Critical Analysis of Environmental Water Quality in South Africa: Historic and Current Trends

Report to the
WATER RESEARCH COMMISSION

by

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Executive summary

BACKGROUND

South Africa is widely recognised as having an admirable water law, and as being a leader in granting a right to water, in terms of quality and quantity, to the environment. However, the water quality of South African water resources is deteriorating (e.g. CSIR 2010, DWA 2011a), although good water quality management structures, strategies, approaches, programmes, instruments, and tools have been developed and implemented nationally. The research reported on here provides a review of changes in water quality management structures, programmes and approaches over the past two decades, and highlights areas where these need updating, completion or revision. As a comparative illustration of changes in water quality with time, changes in 11 water quality parameters in two river systems (the Crocodile River in Mpumalanga, which is moderately impacted, and the Olifants River, in Mpumalanga and Limpopo, which is severely impacted) are presented.

AIMS

In this regard, this report presents:

- A review and critique of the development of policy, management practice and methodologies associated with environmental water quality, within water resource protection;
- Recommendations for research that will support implementation of legal, policy and strategy requirements for environmental water quality, within water resource protection and
- An assessment of the long term water quality trends in two catchments, selected as examples of systems that are moderately (the Crocodile River, Mpumalanga), and seriously (Olifants River, Mpumalanga/Limpopo) impacted by deteriorating water quality.

INTRODUCTION

The South African National Water Acts (No. 36 of 1998) made legal provision for the protection of water resources. Protective measures are termed Resource Directed Measures (RDM) and these include three processes: Classification; Reserve determination; and the setting of Resource Quality Objectives. These three processes are undertaken at differing bio-physical and institutional scales, with larger Integrated Units of Analysis (IUA) being used for classification of the resource, but smaller Resource Units (RU) being used for management at a finer scale and for the setting of Resource Quality Objectives (RQO). Management processes formally integrate these three to achieve overall management of the resource.

Establishment of an understanding of environmental water quality and its underlying information requirements is relatively recent, and has been developed on the basis of an interaction between research and practice. Although much work on environmental water quality has been undertaken, finalization of methods has not occurred and no officially approved methodology has been put in place. In addition, methods for assessing long term trends in water quality are not clear.

REVIEW OF SOUTH AFRICAN ENVIRONMENTAL WATER QUALITY MANAGEMENT

Management of environmental water quality in South Africa is undertaken to ensure water security in terms of quantity and quality to support economic and social development without compromising ecological sustainability. This report will review historic and current management of environmental water quality management with a focus on surface water in rivers, though other resources will receive some attention. For the most part, this report looks at methods and tools dealing with the resource, and other management approaches, such as end-of-pipe assessment and treatment will receive less attention.

Management of water quality at the Department of Water Affairs (DWA) is undertaken via the Chief Directorate: Resource Directed Measures (CD: RDM) and the Sub-Directorate: Water Quality Planning (WQP). The objective is to provide effective management and policy guidance with the context of integrated water resource management. One function of WQP is to undertake national reviews of water quality to determine status and trends in the resource with the aim of supporting strategic management decision making. The management of water quality for the aquatic environment is laid out in a series of documents entitled "Resource Directed Management of water Quality" published in 2006.

The Chief Directorate: Resource Directed Measures was established in 2004 in order to ensure sustainable utilization of the country's water resources to meet ecological, social and economic objectives, and to audit the state of water resources against these objectives. In particular, it is responsible for developing methodologies for RDM, determining and updating targets (as RQOs or the Reserve), assessing and auditing resource quality, and building capacity and providing support for RDM implementation. CD: RDM comprises three Directorates: Water Resource Classification; Reserve Requirements; and Resource Directed Measures Compliance. These interact as follows.

- **Classification**: Weighs up the consequences of different scenarios of managing a catchment to produce a catchment configuration of Ecological Categories and a resultant Management Class.
- **Reserve**: Quantifies the quantity and quality of water to achieve certain Ecological and Basic Human Needs Requirements.
- **Resource quality objectives**: RQOs are numerical and/or narrative descriptive statements of conditions which should be met in the receiving water resource. RQOs will be developed to describe the set MCs, which leads on to monitoring.

Resource Directed Measures interact with Source-directed Controls (SDC) and water use licensing in management of the resource. A decision support system, Assessment of Consideration for Water Use Application (ACWUA), has been developed to inform decisions on licence applications. Decisions are based in multiple criteria including socio-economic factors, race and gender considerations, and alignment to catchment strategy.

Beyond ACWUA, a number of other tools are available or are planned to support water quality management in South Africa. Most of these are available, though uptake and application across the country varies.

Regulatory tools for water quality management have been developed and are available. As management of pollution at source is recognized as an efficient and cost-effective approach, tools addressing this end are important in this regard. Failure to comply with licensing or authorisation conditions may result DWA issuing a directive, the withdrawal of a license, and, usually as a last resort, prosecution.

Economic instruments for water quality management include the Waste Discharge Charge System (WCDS), which has the following three purposes: recovery of management costs; recovery of mitigation costs; and discouragement of water discharge. The WCDS initially only applies to point source discharges, as diffuse discharges are more difficult to quantify. Although WCDS currently operates at a national level, local input and interaction with other government bodies is important.

A number of tools are available for self-regulation of water users' impact on the resource. Some of these, such as ISO 14001 certification, have been widely implemented in certain sectors.

WATER QUALITY ISSUES IN SOUTH AFRICA

South Africa faces a number of challenges with regard to water quality. Some of these are widely recognised, while less is known of the others.

Eutrophication is a major and widely recognised threat to water quality in the country. Eutrophication is a consequence of nutrient enrichment that leads to ecological changes, mostly notably blooms of algae or macrophytes. Eutrophication may impact on ecological systems, and also have aesthetic, recreational, agricultural, and human health impacts. Nutrient input to a resource can be anthropogenic as well as natural. Anthropogenic drivers of eutrophication include increased nutrient from wastewater, either from incorrectly run treatment plants or as a function of an increasing population, nutrient input owing to agricultural practices, and finally input from mines and/or industry. Algal blooms consequent on eutrophication have major ecological consequences, and the cyanobacterial blooms that are more common in the country may also produce toxins, and well as compounds modifying the taste and odour of abstracted water. Macrophyte blooms also have severe ecological consequences, and may lead to blockages of pipes or canals. There are a number of approaches to dealing with eutrophication; none practically address the removal of nutrients from a eutrophied system.

Acid mine drainage (AMD) results largely from the oxidation of sulphide minerals (often pyrites), and is commonly facilitated by acidophilic bacteria. Typical consequences of this process are lowered pH levels, increased salinity (often as sulphates), and mobilisation of a number of metals, many of which are toxic. In South Africa, AMD is particularly associated with gold and coal mining areas. AMD is a major environmental problem in South Africa largely because of the cost in addressing it and because it is persistent and remains a liability after mines have closed. AMD affects both surface and groundwater in the areas that it occurs. The extent of coal and gold mining activities particularly in Gauteng and Mpumalanga have led to major water quality issues in rivers in those areas. Management of increased salinities in these rivers have in the past led to the release of water from upstream impoundments for dilution which in turn stresses water availability upstream of affected areas. Other industries are capable of compromising water quality in a number of ways, and of these salinization is probably the most important.

Contamination of water by poorly managed sewage effluents may lead to high nutrient and salt levels, decreased oxygen levels, and an increase in the number of pathogens present in the water body. The extent of non-compliant sewages discharges across the country led DWA to institute an incentive-based reporting system known as Green Drop. Green Drop seeks to identify and develop key competencies for water management, as well as to establish a baseline of critical risk areas in wastewater management. Since the institution of Green Drop, risk profiles have decreased somewhat, although the majority of wastewater treatment works continue to release effluent that is not treated to a safe and acceptable standard.

Anthropogenic salinization of water in South Africa has a number of causes, including industrial effluent, mining, poor treatment of sewage prior to discharge, irrigation practises, and clear-felling. Diffuse inputs are more difficult than point effluents to address as the extent is not known, monitoring and control are difficult, and effects may only become apparent after prolonged exposure. In addition, groundwater salinization is not easily reversed. Salinization of water impacts on a number of end users, including agriculture (reduced yields, soil salinization), industry (scale and corrosion and potential costs of pre-treatment), and the ecology (changed structure of aquatic communities). Salinity is a measure of all the salts present in water, and not of the individual ions; however, the ionic composition of saline water will determine its impact on all end users. As a result of the differing impacts of ions, salinity *per se* does not always drive regulations around water quality. Insofar as environmental water quality in South Africa is concerned, the relationship between regulatory tools used for routine monitoring, for Reserve determinations and RQOs need to be clarified and refined. A part of this is to expand the toxicological dataset that underlies ecological Reserve determination and monitoring.

The range of toxic organic compounds potentially in water is very high, as toxins could consist of among others, agrichemicals (as herbicides, pesticides, fungicides, etc.), persistent organic pollutants (POP), polyaromatic hydrocarbons (PAH) and endocrine disrupting compounds (EDC). In general, data on all of these is extremely sparse, largely owing to the cost of analysis, the scarcity of appropriate laboratories, the number of potential pollutants, and a general lack of resources. South Africa is a significant user of biocides in agriculture, and the potential for diffuse contamination of water bodies with these compounds is high, though, as noted the extent is not known. The potential for contamination of groundwater by these compounds is of concern, as is their potential to accumulate through the food chain. Sub-acute effects like endocrine disruption are also a potential threat. The National Toxicity Monitoring Programme (NTMP) is tasked with assessment of levels of these compounds in surface and groundwater.

The above comprise the major identified threats to environmental water quality in South Africa. Beyond these, changes in abstraction patterns may exacerbate water quality problems, and these may arise as a result of climate change, unpredictable rainfall, climate variability, alien plant invasion, etc.

MONITORING INITIATIVES

There are a range of monitoring programmes in place in South Africa. Some have been in place for decades, others are more recent. It is important that data collected as part of these programmes be used to guide decision making. A major gap in monitoring is the failure to undertake follow-up monitoring after completion of a Reserve study. Monitoring at DWA involves development of guidelines and procedures for monitoring as well as the management of the various programmes presented below.

The National Chemical Monitoring Programme (NCMP) is well established in South Africa, with ongoing monitoring (despite some recent cutbacks) at hundreds of sites assessing both surface and groundwater. Monitoring data collected as part of this programme may be requested from DWA. Very good data exist for certain aspects of water chemistry of surface waters, such as pH, salinity, major ions, and major nutrients, but data on turbidity, metals, organic pollutants, temperature and oxygen levels are more scarce (and in the case of metals and organic pollutants, often very scarce). Groundwater datasets are less standardized, and although thousands of sample sites are recorded, many have very little data. Improved institutional cooperation between DWA and other organisations assessing groundwater would facilitate standardisation of data and monitoring practices.

The National Eutrophication Monitoring Programme (NEMP) has a brief to undertake monitoring to support assessments of eutrophication in South African reservoirs. Implementation started in 2002 and the programme replaced the previous and smaller Trophic Status Project. NEMP collects water several quality parameters including chlorophyll levels from about 80 reservoirs across the country. NEMP have produced guidelines for monitoring and management of urban impoundments aimed at local authorities and other local water managers.

The National Microbial Monitoring Programme (NMMP) monitors the extent of faecal contamination of surface water in the country. The programme aims to locate and prioritize areas in the country where health risks are highest, and to assess the status, trends and health risks of faecal pollution in these areas. It also assesses the effectiveness of measures to protect surface water resources against these threats. Sampling is undertaken on a two-monthly basis. As much data on faecal pollution is held by local authorities and municipalities, an attempt has been made to merge these data with DWA WMS and in this way rationalize monitoring efforts. Other initiatives for monitoring diseases caused by waterborne microbes have been developed by DWA and the WRC, and these cover a wider range of diseases than those associated with faecal pollution.

The National Toxicant Monitoring Programme aims to measure, assess and report on the status and trends in toxicants in surface and groundwater resources. The programme was initiated as a response to increasing concern about the presence and extent of toxicants in water resources and to respond to the lack of coherent information in this regard. In terms of international treaties signed by South Africa, the country has a responsibility to undertake research on POPs in water resources (to include monitoring of several compounds, their socio-economic impact and release reduction strategies). The programme is relatively recent with pilot testing being undertaken only in 2008-2009.

The National Aquatic Ecosystem Health Monitoring Programme (NAEHMP) provides a means of directly assessing ecological health in water resources. The programme currently consists only of the River Health Programme (RHP), a joint programme in which provincial nature conservation agencies are responsible for much field work and reporting. The programme has produced a number of valuable State of the River reports, which cover various environmental aspects of rivers as well as bioassessment of the resource. As the name suggests, these reports are focussed on river and neither estuaries of wetlands are properly covered. Although monitoring of the ecological Reserve has not commenced at time of writing, this is another potential source of data, and integration of RHP and ecological Reserve monitoring within an adaptive management framework has been proposed. For effective implementation of Ecological Water Resource Monitoring (EWRM) in the light of capacity and resource limitations, a cost effective approach to monitoring is needed. In this regard, rapid methods of assessment have been developed as an initial approach to monitoring, with full monitoring taking place only when necessary.

Beyond the abovementioned programmes, it should be noted that a range of water quality monitoring data are collected by other organisations.

ESTUARIES, WETLANDS AND GROUNDWATER

Notwithstanding their importance, wetlands have been a neglected water resource historically. However, methods for assessment of water quality and the (rapid) ecological Reserve in wetlands have recently been developed. Owing to the recent development of these methods and the extent of differences between wetland and rivers, data on water quality in wetlands over time is rare. Further development of methods for wetlands is of some importance.

Estuaries represent an interface between marine and freshwater systems, and methods for estuaries require a combination of marine and freshwater monitoring approaches with due consideration of the changing distributions of the two water bodies. As such, methods proposed for estuaries are different to those employed upstream in freshwater. DWA have initiated a National Estuaries Monitoring Programme (NESMP) that aims to monitor biotic and abiotic components of estuaries and to ensure collaboration between all those involved in estuarine management. The programme is currently at pilot scale and aims for full implementation in 2015.

The development of methods for groundwater lags behind that of rivers and wetlands owing to the considerable differences between the resources. The vulnerability of groundwater to pollution and the potential of long-term impacts following pollution has ensured that a protective approach has been taken to the management of groundwater in South Africa. However, DWA have implemented strategies for the management of groundwater.

Research programme for environmental water quality to support the National Water Resource Strategy 2

The core of the water resource protection approach outlined in the National Water Resource Strategy 2 (DWA 2013c) lies in six principles to be implemented through ten strategic actions. These are outlined below.

Principles:

- Protection of the resources through classification of the resource with the Reserve as a prior right.
- Water resource protection should be based on a participatory approach, involving users, planners and policymakers at all levels.
- The value of water resources must be recognised from an economic point of view and the social and environmental benefits of the resource.
- Water resource protection must guide setting conditions for water use authorisation.
- Incentive based protection of the water resources.
- Integrated protection of aquatic ecosystems.

Strategic actions:

- Manage for sustainability using resource directed measures.
- Invest in strategic water source areas.
- Strategic investment in the maintenance and rehabilitation of water ecosystem.
- Maintain freshwater ecosystems priority areas in good condition.
- Protect riparian and wetland buffers and critical groundwater recharge areas.
- Rehabilitate strategic water ecosystems to support water quantity and water quality.
- Monitor ecological health to inform management.
- Minimisation of pollution from wastewater treatment works.
- Establishing commitment to sustainable water resource management.
- Target actions with immediate benefits.

In support of implementing the above actions and in accordance with the principles, we propose the following set of research projects. These are coordinated across appropriate KSA thrusts as outlined in the text.

1. Revise and update TEACHA program so that it is accessible and practical to use in ecological Reserve studies and monitoring.
2. Extend salt ecotoxicology research in order to have a better set of data underlying ecological Reserve studies and monitoring. The application of TEACHA is premised on ion and salt toxicity and this research will support revision of TEACHA.
3. Integration of the RDM components (classification, ecological Reserve, and RQOs) in order that their premises and the implications for practice are aligned.
4. Integration of water quality and quantity processes to produce user-friendly quantity/quality models.
5. Assessment and revision of RDM participatory processes.
6. Integration to ensure coherent links between RDM and SDC measures at levels including policy, legislation, governance and practice.
7. Inclusion of a complex social ecological systems view in research projects

The NWRS2 includes a set of key strategic objectives for water resource protection. The benchmark of the EWQ programme success will be the contribution the programme makes to the achievement of the objectives.

Long term trends in Environmental Water Quality

Although changes to South Africa's water law and regulatory frameworks after 1994 were recognised for their focus on environmental sustainability and social outcomes, it is widely recognised that water quality in the country is declining. In order to assess how changes in management may have affected the resource, it is necessary to understand what changes in water quality have taken place.

The analysis presented here looks at trends in several water quality parameters in two example catchments, the Crocodile River in Mpumalanga and the Olifants River in Mpumalanga and Limpopo. The former is moderately stressed and been reported as water-stressed in the past, and the latter is highly impacted and has recently received much attention as regards water quality.

In this analysis, water quality was taken as being chemical water quality as insufficient data on biotic responses and environmental health were available. All data were drawn from DWA's Water Management System (WMS) and no other sources of data were consulted. Trends in the following parameters were assessed: orthophosphate; total inorganic nitrogen (TIN); electrical conductivity; pH; the chemical weathering index (an index reflecting the extent to which chemical weathering contributes to the relative ionic balance); the sulphate contamination index (an index reflecting the extent to which sulphates contribute to the relative ionic balance); the chloride salinization index (an index reflecting the extent to which chlorides contribute to the relative ionic balance and an indicator of salinization); the adjusted sodium adsorption ratio (an index indicating the suitability of the water for irrigation); the corrosion potential ratio (an index indicating the potential of the water to lead to corrosion of metal pipes and fittings); un-ionized ammonia (as an indicator of potential toxicity); *Escherichia coli* counts (as an indicator of faecal pollution); and levels of toxins.

Monitoring points selected for analysis were selected on the basis that they should have enough data for analysis, that they should be representative of environmental water requirement sites, or be spaced relatively regularly along the main river or low on tributaries, and that upstream sites should be included as reference points wherever possible. Sixteen sites on the Crocodile River (including one on the Komati River downstream of the confluence with the Crocodile River), and 28 sites on the Olifants River system were selected.

Although the main aim of the analysis was to assess trends, data were compared against benchmarks for water quality. The majority of these were generic RQOs from DWA (2011a), but benchmarks from Ashton and Dabrowski (2011) and the South African water quality guidelines (DWAF 1996) were used for some parameters.

Trends were analysed using a generalized additive mixed model fitted to DWA data. The model is suited to modelling non-linear trends with seasonal components as were apparent in the data. The model enabled an assessment of the statistical significance of trends after accounting for variation due to seasonal change. Insufficient data were available to assess trends in toxins, and levels of these were assessed by combined data over time and comparing relevant percentiles with identified benchmarks.

Major temporal and spatial trends

As might be supposed, the water quality in both rivers tended to degraded with distance downstream. Overall, water quality from sites in the Crocodile River was better than that from the Olifants River. While localized impacts in the Crocodile River were found, they were of lesser extent than the overall impact of in particular mining in the Olifants River catchment.

Localized trends are addressed in detail in the text. Here we present larger scale trends that may potentially be management related, or may also relate to a stressor acting beyond any one catchment.

Sites in both catchments showed a general tendency to increased orthophosphate levels with time. The increase was not confined to sites low on rivers, but also to relatively undisturbed upstream sites. When compared to other parameters assessed in this study, levels of orthophosphate tended to exceed recommended levels by the greatest degree. As eutrophication has been identified as a major threat to South African water resources, this increase is of considerable concern. However, in a number of sites in both catchments, in particular the Olifants River basin, recorded levels of orthophosphate decreased dramatically after 2009 or so.

Another widespread trend observed at many sites in both catchments (though not where overridden by local impacts) was an increase in pH levels from a stable base that commenced in the late 1980s. Levels increased slowly for about 5 years, then stabilized against at a level that was generally 0.5 to 1.0 pH units greater than before.

The impact of coal mining and associated acid mine drainage is pronounced in the upper Olifants catchment. Despite this, only one site showed the classic signature of low pH levels, high sulphate levels and elevated metal ion concentrations. The remainder of the sites showed increased conductivity and sulphate levels without matching pH decreases. Assessment of data revealed the calcium levels increase together with those of sulphate. This seems likely to be a function of treatment of acid mine drainage, probably as a result of using lime to neutralize acidic waste. The combined effect of this is to elevate the conductivity levels more than would have occurred as a result of increased sulphate alone. The increased calcium levels may have other impacts, such as scale production on pipes. Another impact of increased calcium levels is that, according to the sodium adsorption ratio, the elevated calcium may make the water more suitable for irrigation from a soil sodicity point of view. However, the deleterious effects of elevated salinity cannot be discounted when considering the suitability of the water for irrigation.

Sites in both catchments, though without the elevated sulphate signature characteristic of the upper Olifants basin, showed increased levels of sulphate in the early 1980s. This increase was temporary. No cause is known, however, industrially produced sulphate aerosols may have contributed to this temporary peak in sulphate.

The importance of chemical weathering in contributing to the ionic balance in both rivers generally decreased with time and with distance downstream. This observation is not absolute, as the data records for most sites in the upper Olifants catchment start after mining impacts were established, and these sites for the most part show sulphate mineralization to be a major driver of the ionic balance over the entire data record. Nevertheless, the importance of sulphate contamination and salinization in controlling the ionic balance increases with time and with distance downstream. In general, natural weathering was a greater contributor to the river's ionic complement in the Crocodile as opposed to the Olifants River.

Data on microbial pollution in WMS was relatively sparse and unevenly distributed spatially. As a result, little can be said about spatial trends in microbial pollution. Use of additional records, for example from local authorities, would help in gaining a better idea of microbial pollution levels. No clear temporal trends in microbial levels were found. However, levels of microbial pollution in both catchments were generally unacceptably high.

Very few data on toxins were available, and, at the selected sampling sites, the data consisted only of metal ion levels, with no data on organic toxins. Even at the selected sites, very few data were available at some sites, and the remainder had not data. Overall, the worst impacted sites in the

Crocodile catchment were on the lower Crocodile and Kaap Rivers. In the Olifants River basin, the worst impacted sites were low in the catchment, below the Phalaborwa complex and the confluence of the Ga-Selati River. The scarcity of data (both in terms of sampling events and compounds sampled for) is of concern as pesticide and metal impacts have been found in both rivers.

The upper Olifants River catchment has been mentioned several times above. The catchment is severely impacted with major stressors being coal mining, electricity generation, industry, agriculture, and inefficient wastewater treatment. In many cases these stresses predate the data record used for this analysis. The combined impacts lead to elevated salinity (and sulphate and calcium) levels, often accompanied by elevated nutrient levels. Even sites with better quality show some impact of coal mining, and the cumulative impact is high and shows no sign of abating.

The mining-industrial complex of Phalaborwa is situated adjacent to the Ga-Selati River in the lower reaches of the Olifants River Catchment. Although the impacts in the upper Olifants catchment are severe, the impacts at Phalaborwa are arguably the worst point impact in the catchment. Of the parameters assessed, the major impacts are the elevated conductivity and orthophosphate levels (the latter 10 times upstream values). However, the temporal trend of many of the parameters assessed at this site is to improve with time. Orthophosphate levels, however, are an exception, and these show a continual increasing trend. Sodium adsorption ratio levels increased as well, but these remained acceptable at worst. All other parameters (bar pH, which did not change with time) showed an improving trend. Nevertheless, at the end of monitoring, levels of several parameters remained unacceptably high. These changes seem to underlie, at least partially, improvements in Olifants River water downstream of the confluence with the Ga-Selati River.

The Elands River in the Crocodile River catchment is one of the most impacted rivers in the catchment. Assessment of water quality changes with time along this river shows a clear temporal quality decrease, and a distinct cumulative impact with distance downstream. Unless this temporal trend changes, water quality in discharge from this river will be unacceptable in the relatively near future. Potential impacts identified in this catchment include a ferrochrome smelter, a pulp and paper mill, and increasing human settlement.

The Elands River in the Olifants River catchment is perhaps the clearest example from this study of a river impacted by salinization as indicated by increased sodium and chloride levels. The river supports much agriculture and in particular irrigated agriculture, and passes through an area with relatively low rainfall. Increased levels of sodium and chloride are major drivers of an elevated electrical conductivity. Changes in salinity are correlated with rainfall, and increase in periods of low rainfall. Although the lower Crocodile River has been identified as a region under stress owing to irrigation, the levels of chloride salinization are lower, despite comparative electrical conductivities.

METHODOLOGY AND DATA

The model fitting procedure adopted for this analysis was selected for its ability to fit non-linear trends, to account for seasonality, to utilize irregularly sampled data, and to accommodate temporal autocorrelation. Despite a decision to limit model complexity so as to avoid overfitting, model fits were normally good. High variation in data limited model fits, leading some model to fail to converge, and others to converge on a simple linear fit. Model fit would be improved by tuning model starting parameters and prior assessment and checks on data.

The approach adopted here, of assessing water quality using concentration data, is of considerable value and is widely practised. However, this approach excludes any consideration of flow or discharge, and if these are not known, the results of analyses may be misleading. For example, the severely impacted water leaving the Ga-Selati River might be expected to have a major impact on the

Olifants River, but this impact is mitigated by the much greater discharge of the Olifants River. A consideration of the interaction between discharge, loading and concentration would deepen understanding of the trends observed here.

Issues relating to the lack of availability of data on toxicants are discussed above. The issue of what else could be monitored is also addressed above under discussion on the NCMP. To reiterate, certain parameters are heavily monitored and a great deal of data on these is available. There are a number of others where the data record is sparser, and these include parameters important for management of water quality. Examples include turbidity, chlorophyll *a* levels, microcystin, etc.

In the survey of data availability as part of monitoring point selection, it was noted that a significant number of monitoring points had ceased to be monitored from the late 2000s onward. Another recent trend was a recent decrease, and occasionally irregularity, in frequency of monitoring. This overall decrease in monitoring decreases the number of long-term data sets available for analysis, and the change in frequency reduces the sensitivity of the monitoring programme to short-term events and in some cases, reduces sensitivity to the extent that seasonality could not adequately be sampled. As many parameters showed significant seasonal variation, an understanding of seasonality is important for ongoing monitoring. In addition, consistent and regular records would greatly improve assessment of toxic metals, which were found at significant levels in both catchments.

The detection limits of tests used in monitoring were in several cases too high to return useful data unless the levels of a particular parameter were extremely high. In some or most samples, the detection limits for mercury and lead were more than a factor of ten times the acute effect value selected for that metal. These tests would be unable to detect levels of these metals until profound impacts had occurred. In the case of orthophosphate, the consequences of test and detection limit selection were less dramatic; however, the detection limit in recent samples has the effect that the distinction between ideal and acceptable levels or orthophosphate cannot be resolved.

In the Olifants River catchment, it was noted that no long term data were available for a long stretch of the river central region downstream from Flag Boshielo Dam. This is unfortunate and limits the conclusions that can be drawn about processes in this region of the river.

CONCLUSIONS

Overall, trends with time generally show a movement towards decreasing water quality at sites in the mid to low catchments, although sites high in catchments may be more stable (depending on local impacts). Site that were impacted at the start of the data record generally continue to show that impact over time (though there are some improvements). Changes in water quality are driven by increased orthophosphate levels (though recent records suggest this trend may be changing), increased pH levels, increased salinity, and for sites in the Olifants River catchment, increased sulphate and calcium levels. Some rivers showed elevated chloride levels consistent with salinization, but this was not widespread. Finally, microbial levels were unacceptably high, though no trends were apparent.

Data on potential toxins are inadequate and a reassessment of monitoring in this regard is recommended.

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Acronyms and Abbreviations

ACWUA	Assessment of Consideration for Water Use Applications
AMD	Acid Mine Drainage
BHNR	Basic Human Needs Reserve
BOD	Biological Oxygen Demand
CD: RDM	Chief Directorate: Resource Directed Measures
CMA	Catchment Management Agency
CMS	Catchment Management Strategy
COD	Chemical Oxygen Demand
CPR	Corrosion Potential Ratio
CSIR	Council for Scientific and Industrial Research
DEA	Department of Environmental Affairs
DSS	Decision Support System
DWA	Department of Water Affairs
DWAF	Department of Water Affairs and Forestry
EC	Ecological Category
EC	Electrical Conductivity (as proxy for TDS)
EDC	Endocrine Disrupting Compounds
EMP	Environmental Management Plan
EMPR	Environmental Management Programme
EWQ	Environmental Water Quality
EWR	Ecological Water Requirements
EWRM	Ecological Water Resources Monitoring
GAM	Generalized additive model
GAMM	Generalized additive mixed model
GRDM	Groundwater Resource Directed Measures
ICMA	Inkomati Catchment Management Agency
IUA	Integrated Unit of Analysis
IWRM	Integrated Water Resource Management
MC	Management Class

NAEHMP	National Aquatic Ecosystem Health Monitoring Programme
NCMP	National Chemical Monitoring Programme
NEMP	National Eutrophication Monitoring Programme
NESMP	National Estuaries Monitoring Programme
NMMP	National Microbial Monitoring Programme
NTMP	National Toxicant Monitoring Programme
NWA	National Water Act
NWRS	National Water Resources Strategy
POPs	Persistent Organic Pollutants
RDM	Resource-Directed Measures
R-DRAM	Rapid Diatom Riverine Assessment Method
RHAM	Rapid Habitat Assessment Method
RHP	River Health Programme
RO(s)	Regional Offices(s)
RoD	Record of Decision
RQO	Resource Quality Objectives
RQS	Directorate of Resource Quality Services
RU	Resource Unit
RWQO	Resource Water Quality Objectives
SAEON	South African Environmental Observation Network
SAR	Sodium Adsorption Ratio
SC&A	Scherman Colloty and Associates
SDC	Source-Directed Controls
SPI	Specific Pollution Index
TDS	Total Dissolved Solids
TIN	Total Inorganic Nitrogen
TPC	Threshold of Probable Concern
UCEWQ	Unilever Centre for Environmental Water Quality
WDCS	Waste Discharge Charge System
WfDG	Water for Growth and Development
WMA	Water Management Area

WMS	Water Management System
WQP	Water Quality Planning
WRC	Water Research Commission
WRCS	Water Resource Classification System
WRPS	Water Resource Planning Systems
WUA	Water User Association

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1 GENERAL INTRODUCTION

1.1 Water Resource Protection

The South African National Water Act (No. 36 of 1998) made historic legal provisions for the protection of water resources (RSA 1998). These protective measures are termed Resource Directed Measures (RDM) and are implemented through a three-stage set of processes: i) Resource classification, ii) determination of the ecological and basic human needs Reserve, and iii) setting Resource Quality Objectives (RQOs). Taken together, these processes determine the actions that must be taken to protect the water resource to a desired level (Government Gazette Regulation 810, 33451).

Equity, resource sustainability and resource-use efficiency are the three underpinning principles of the NWA. Resource protection is essential to ensure the long term sustainability of water resources; however, it is clear that not all water resources can be afforded the same level of protection. In order for water to serve as an economic good and fulfil its role as a primary driver of the economy, water resources must be used as well as protected. Therefore decisions have to be made about the level of protection offered in space and time. The classification process defines the level to which specific water resources will be protected and used. Classification happens at the scale of the integrated units of analysis (IUA) (for an example see Figure 1 below).

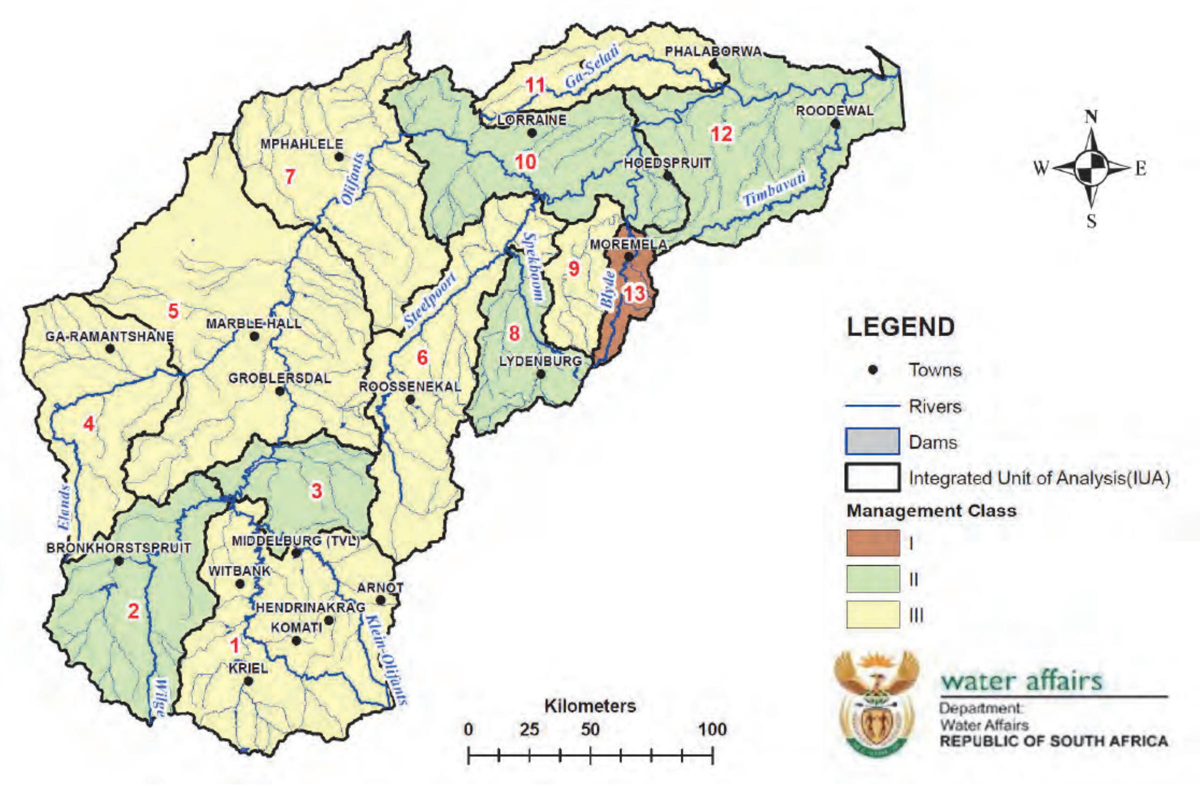

Figure 1 Integrated Units of analysis and Management classes of the Olifants River.

The three RDM processes are undertaken at different bio-physical and institutional scales. The largest scale is that of the Water Management Area, of which there are nine in South Africa, each of which is, or will be, administered by a Catchment Management Agency. Each WMA is divided into smaller areas or sub-catchments identified as IUAs – for which a management class is defined through the classification process.

Since both the water resource and water users in the catchment have particular requirements, the process of deciding on the management class draws on both user and resource requirements, and depends crucially on engaging with the widest possible range of stakeholders. These stakeholders contribute perspectives to government and finally a class is set: Class I (minimally used); Class II (moderately used) or Class III (heavily used). Each class requires a different proportion of the natural water available to be reserved for ecosystem maintenance and thus a different proportion available for other uses. Each class also has a different set of water quality goals, allowing for waste discharge by water users.

However, neither flow nor water quality is uniform across IUAs, and each needs to be assessed and managed at a finer scale. For this reason the management class of an IUA indicates an overall “condition” that is a measure of a mosaic of conditions and uses across the area on the IUA. The smaller unit of the mosaic that makes up the IUA is termed the Resource Unit (for an example see Figure 2) – and the ecological condition of the RU is the ecological category. Figure 1 shows the IUAs and management classes for the Olifants River WMA; Figure 2 shows the Resource Units and ecological categories for the Olifants River WMA; and Figure 3 shows how these relate to each other.

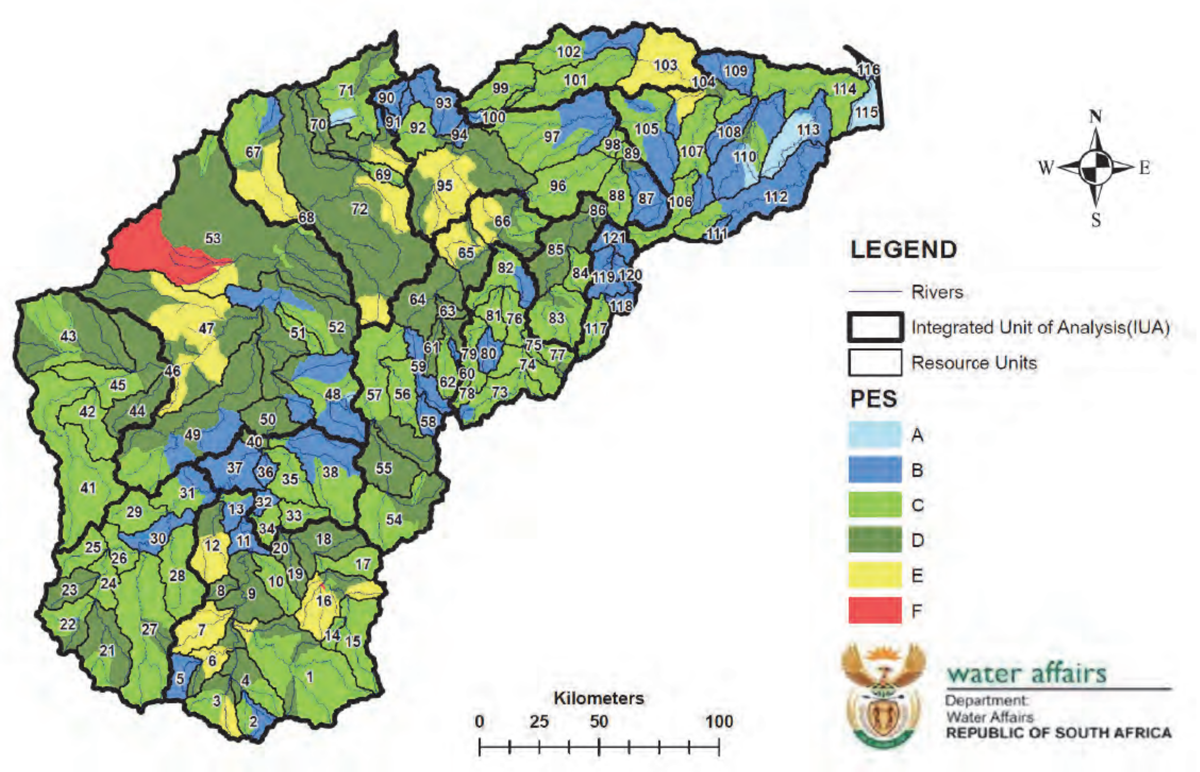

Figure 2 Resource Units (RU) and their Ecological Categories that go to make up the Integrated Units of Analysis (IUA). Each resource unit has a present ecological state (PES) which is one step in an ecological Reserve determination.

Within an ecological Reserve determination, the present ecological state (PES) of the river segments are assessed in terms of multiple components including the habitat requirements of fish, macroinvertebrates and vegetation, as well as geomorphology, hydraulics, flow and water quality. A composite ecological Reserve category is assigned. Then, depending of the Class of the IUA, and the contribution to the IUA that the RU makes – the river in managed to achieve a particular Reserve category. The numerical and descriptive criteria for these categories are termed Resource Quality

Objective (RQOs). Where the present state is degraded, (Reserve categories E and F), management objectives are selected as the D category (Class III).

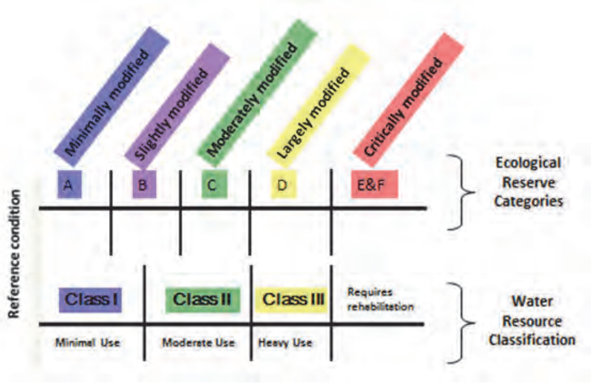

Figure 3 Diagram of resource classification in relation to ecological Reserve categories.

Resource Quality Objectives for water quality are quantitative goals for each ecological category, presented as concentration ranges, loads, magnitudes and durations for a variety of water quality variables; together with explanatory narratives where quantitative data are limited or need to be clarified (DWA 2010a,b, 2013a).

Confidence that achieving RQOs will deliver the estimated level of ecosystem health depends on information monitoring. Fewer data mean lower confidence. Levels of ecological Reserve assessment are: Comprehensive (most data needed, most expensive); Intermediate; Rapid (I, II, III) and Desktop (least data needed, least expensive).

In all three of the RDM processes there are procedures for examining and setting goals for the many variables that comprise water quality (DWAF 2004a, 2008a), but these have not been formally approved and are in need of revision.

1.2 Summarised history of environmental water quality methods in ecological Reserve determinations

The term "Environmental Water Quality" was coined at Rhodes university, with the establishment of the Unilever Centre for Environmental Water Quality in 2002, and was popularized by Palmer *et al*. (2002, 2004a) and scientifically communicated (Palmer *et al*. 2005). The concept focuses on the need to use multiple kinds and sources of data in order to build an understanding of the water quality needs of ecosystems, and threats to them. The emphasis is on the combined use of data derived from water chemistry, ecotoxicology and biomonitoring.

While methods for environmental water (flow) requirements were well developed by the time the NWA was promulgated, this was not the case for water quality. The process to determine ecological Reserve categories and RQOs for water quality variables, was developed on the basis of a close interaction between research and practice (Goetsch [Scherman] and Palmer 1997, Bath *et al*. 1999, Palmer 1999, Louw and Palmer 2000, Palmer and Rossouw 2000, Palmer and Scherman 2000, Palmer *et al*. 2000, Palmer 2001, Palmer *et al*. 2001, Muller and Palmer 2002, Palmer *et al*. 2002, Scherman *et al*. 2003a,b, Palmer *et al*. 2004b,c,d, Slaughter *et al*. 2004, Palmer *et al*. 2005).

Despite this body of work, currently there is no officially approved methodology and most practitioners use DWAF (2004a, 2008a). The procedure reports have not been approved and are in need of revision.

In addition despite an excellent, though now diminished, water quality monitoring network and analysis there is currently no clear method for assessing long term trends in water quality that would act as a warning of trajectories of decline that would affect decisions about Management classes and RQO's.

1.3 Project aims

This report therefore presents:

- A review of development of policy, management practice and methodologies associated with environmental water quality, within water resource protection;
- Recommendations for research that will support implementation of legal, policy and strategy requirements for environmental water quality, within water resource protection; and
- An assessment of the long term water quality trends in two catchments, selected as examples of systems that are moderately (the Crocodile River, Mpumalanga), and seriously (Olifants River, Mpumalanga/Limpopo) impacted by deteriorating water quality.

2 REVIEW OF SOUTH AFRICAN ENVIRONMENTAL WATER QUALITY PROCESSES OF PRACTICE, AND RESEARCH RECOMMENDATIONS

2.1 Background

A document was prepared by Jenny Day and Patsy Scherman in 2009 as part of DWA's Water for Growth and Development Framework (WfGD) project (Appendix E: Environment – Water Quality) (Day and Scherman 2009), which described the status of water quality for the environment and the associated management strategies for water quality. This document is used as the starting point for this review. The drafting of this set of documents was part of developing DWA's framework to guide actions and decisions that will ensure water security in terms of quantity and of quality to support South Africa's requirements for economic growth and social development, without compromising the ecological sustainability of water resources.

The Department also embarked upon rigorous water assessment studies referred to as Reconciliation Strategies in order to reconcile the supply and demand for water, particularly in water scarce areas and areas experiencing relatively high levels of demand. These strategies aim to ensure the supply of water at adequate levels of assurance within the constraints of affordability, appropriate levels of service to users, and the protection of current and possible future water resources. The identification and attempt to manage water quality issues has been a necessary part of developing Reconciliation Strategies.

2.2 Focus of review

The following points clarify what this review entails.

- Note that the project deals broadly with environmental water quality management in the South African context, and focuses on rivers specifically, although the management of estuaries, wetlands and groundwater are mentioned.
- The focus of the review will be environmental water quality management, i.e. methods and tools dealing specifically with the water resource. Although source-directed tools (for example) will be mentioned, the focus is not on listing end-of-pipe treatment methods for sewage or industrial effluents, but rather on the usefulness of such tools in meeting environmental water quality objectives.
- Historical and present water quality tools will be discussed.

- The term environmental will be used broadly and will therefore also focus on water quality users other than ecosystems.
- An introduction will be provided for a number of tools and policies; references will be provided to enable more detailed reading.
- The review will focus on water quality issues of significance in South Africa and possible strategies to alleviate or address these problems.

2.3 Broad introduction to tools for EWQ management

2.3.1 EWQ management at DWA

EWQ management in South Africa relies on a number of tools and policy documents. Water quality management at the Department of Water Affairs is conducted via the Chief Directorate: Resource Directed Measures (CD: RDM) and the Sub-Directorate: Water Quality Planning (WQP). The specific objective of the Water Quality Planning function within DWA is to provide effective management solutions and policy guidance to address the current water quality challenges within the context of integrated water resource management, as shown in Figure 4.

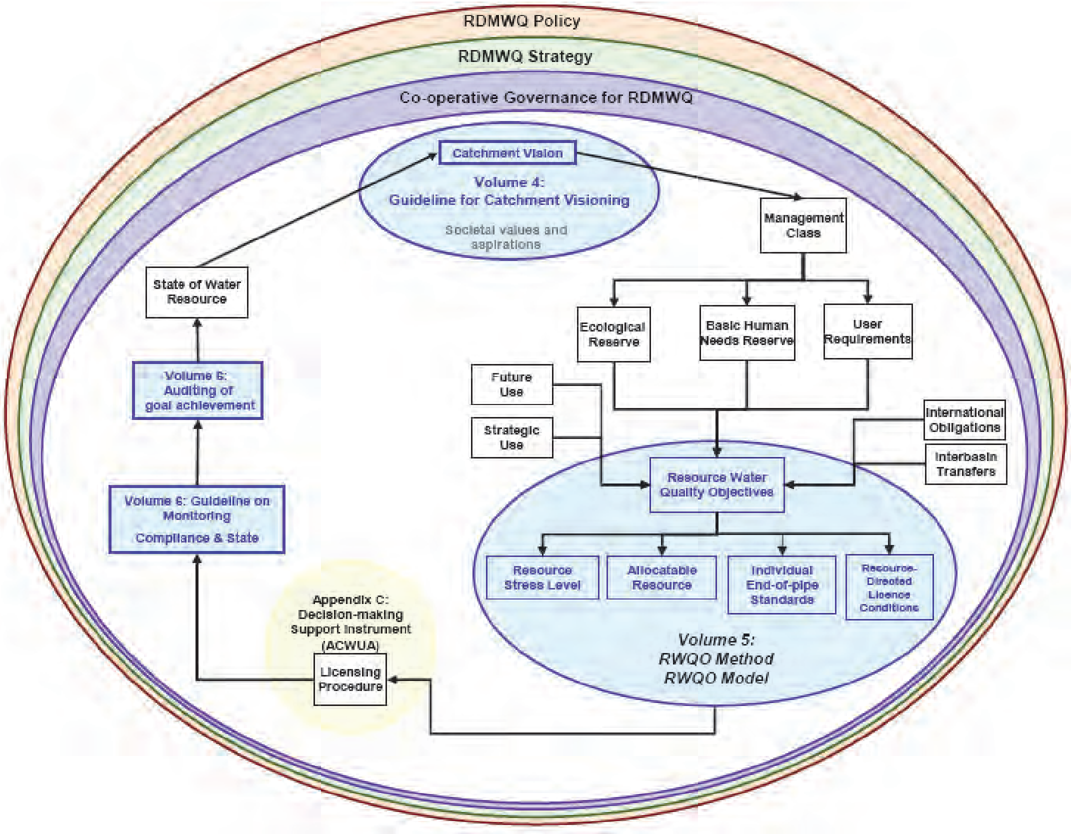

Figure 4 Water quality planning (WQP) within Integrated Water Resources Management (IWRM).

One of the functions of WQP is to conduct national reviews on water quality status and trends that measure, assess and report on the current state and appropriate temporal trends of selected groups of water quality indicators in South African surface water resources. This is aimed at supporting strategic management decisions in the context of sustainable fitness for use of those water resources and for the protection of the integrity of aquatic ecosystems. A document outlining these issues was produced in 2011 (DWA 2011a). Analysis and reporting therein focused on the understanding of water quality status and dominant issues at the WMA scale. The current in-stream water quality was compared to a generic set of Resource Water Quality Objectives (RWQOs) for all users throughout all Water Management Areas (WMAs) and reflected as Ideal, Acceptable, Tolerable and Unacceptable in terms of an indication of ecological Reserve compliance. The boundaries that define these categories were taken from the methods manual produced for determining the Ecological Reserve for rivers (DWAF 2008a).

The management of water quality for the aquatic environment in South Africa is outlined in a series of documents produced by the DWAF Directorate Water Resource Planning Systems: Water Quality Planning. The resource-based tools are outlined in a series entitled Resource Directed Management of Water Quality, i.e. Sub-series no. WQP 1 of the Water Resource Planning Systems Series, published in September 2006. The full series of documents published as part of this series are shown in Table 1 below.

Table 1 List of documents in the Resource Directed Management of water quality series.

Volume no	Report title
	Introduction to the Resource Directed Management of Water Quality
Volume 1.1:	Summary Policy
Volume 1.2:	Policy on the Resource Directed Management of Water Quality
Volume 2.1:	Summary Strategy
Volume 2.2:	Strategy for the Resource Directed Management of Water Quality
Volume 3:	Institutional arrangements for Resource Directed Management of Water Quality
Volume 4.1:	Guidelines on Catchment Visioning for the Resource Directed Management of Water Quality
Volume 4.2	Guideline for determining Resource Water Quality Objectives (RWQOs), water quality stress & allocatable water quality
Volume 4.2.1	RWQOs Model and User Guide
Volume 4.3:	Guideline on Monitoring and for the Resource Directed Management of Water Quality
Appendix A	Project Document: Resource Directed Management of Water Quality: Philosophy of Sustainable Development
Appendix B	Project Document: Conceptual Management of Water Quality: Review of water use licence applications in the context of the Resource Directed Management of Water Quality
Appendix C	Project Document: Guidelines for Setting License Conditions for Resource Directed Management of Water Quality
Appendix D	Project Document: Decision-support instrument for the Assessment of Considerations for Water Use Applications (ACWUA)
Appendix E	Project Document: Glossary of terminology often used in the Resource Directed Management of Water Quality
	RDMWQ Peer Review Report by Dr PJ Ashton

2.3.2 RDM and water quality management

The Chief Directorate: RDM was established at DWA in 2004, with the main objective being to ensure protection of water resources, as described in Chapter 3 of the South African National Water Act (NWA) – 1998 (No. 36 of 1998) and other related water management legislation and policies. The role of CD: RDM is to provide a framework to ensure sustainable utilization of water resources to meet ecological, social and economic objectives and to audit the state of South Africa's water resources against these objectives. Regulation 810 published in Government Gazette No. 33541 dated 17 September 2010, defined the water resource management classes and a procedure (Water Resource Classification System – WRCS) to determine a Class. According to the NWA, once this WRCS has been gazetted all significant water resources must be classified.

More detail regarding CD: RDM can be found in Chapter 3 of King and Pienaar (2011).

In particular, CD: RDM is responsible for:

- Developing policies, strategies, systems, methodologies and guidelines for resource directed measures (RDM), particularly the Reserve determination, water resources classification and specification of associated resource quality objectives (RQO).
- Determining (and updating) the Reserve in significant water resources.
- Prioritisation, piloting and coordination of RDM implementation.
- Capacity building and technical support for RDM implementation.
- Developing indicators and assessing the state of the nation's water resources.
- Auditing implementation of RDM and resource quality against the Resource Quality Objectives and the Reserve.

The Chief Director: RDM reports directly to the Policy & Regulation Branch Manager of the DWA, and comprises the following directorates:

- Directorate: Water Resource Classification;
- Directorate: Reserve Requirements; and
- Directorate: Resource Directed Measures Compliance.

These three directorates can be linked in the following way (taken from a presentation by Nyamande, CD: RDM, prepared for the Mvoti-Mzimkhulu Classification study Stakeholder meeting of March 2013):

Classification

- Weighs up all the consequences of different scenarios of managing a catchment.
- The output of the process is a catchment configuration of Ecological Categories and a resultant Management Class (MC).

Reserve

- Quantifies the quantity and quality of water to achieve certain Ecological and Basic Human Needs Requirements (BHNR).

Resource quality objectives (RQO)

- RQOs are numerical and/or narrative descriptive statements of conditions which should be met in the receiving water resource. RQOs will be developed to describe the set MCs, which leads on to monitoring.

All three processes shown in the text box above are conducted in a step-wise process. The integration of these steps, i.e. the Integrated Steps, is shown clearly in Figure 5 below, which was developed by Louw and Scherman in March 2012 to assist understanding about the linkages between these processes. This diagram was subsequently used in the Inception Report and training workshops for both the Mvoti-Mzimkhulu and Letaba Classification studies initiated in 2012.

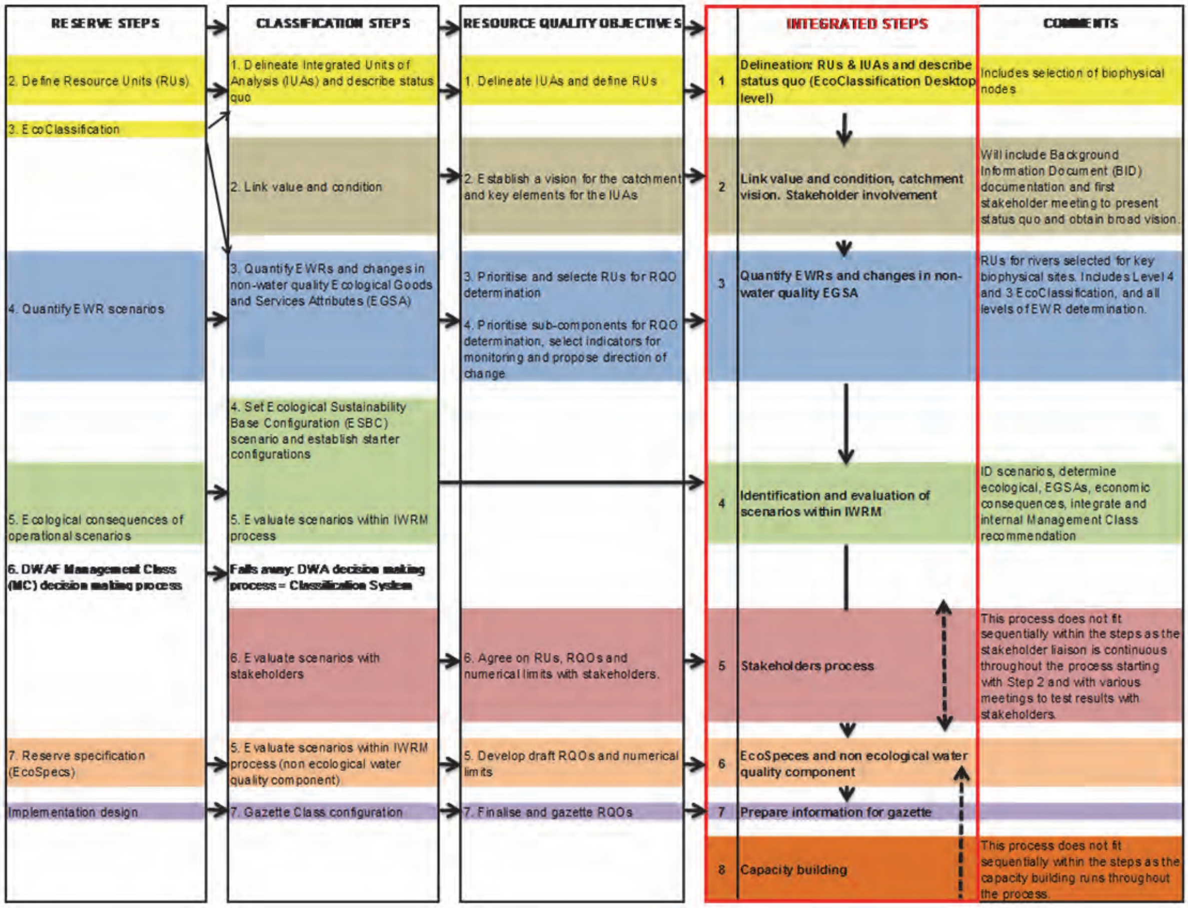

Figure 5 Linkages between the Reserve, classification and RQOs (prepared by Louw and Scherman, 2012).

2.3.3 Licensing and water quality

The link between source-directed controls (SDC), resource-based tools and licensing is shown diagrammatically in the figure below (Figure 6), which outlines the steps to be taken during license preparation to meet water quality objectives. The diagram aims to show which steps may be triggered by a water quality application.

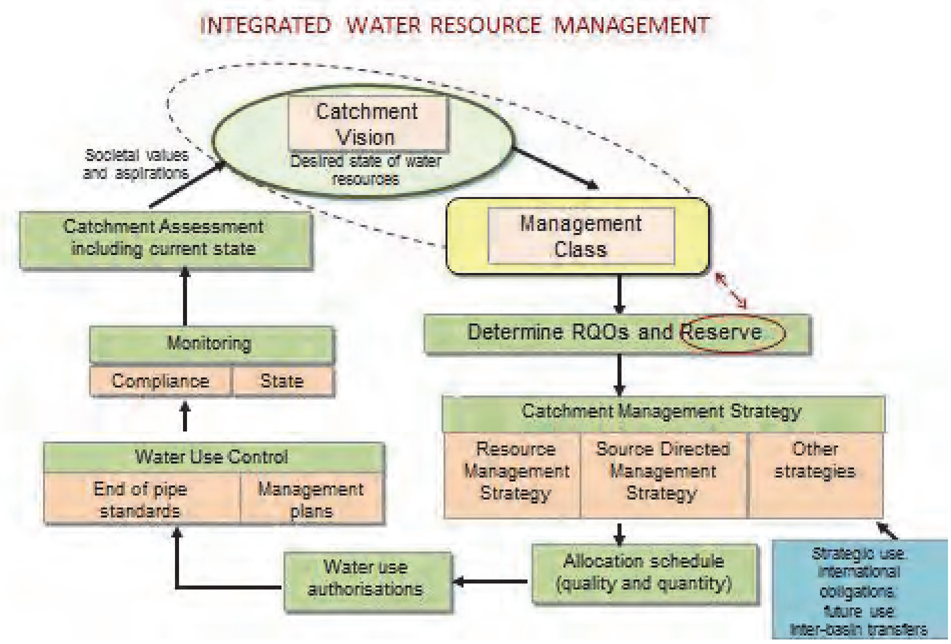

Figure 6 A diagrammatic representation of the links between the functions of CD: RDM

A decision support instrument is referred to in Figure 7, i.e. the Assessment of Consideration for Water Use Applications (ACWUA), which has been developed to allow multiple criteria decision analysis, and utilizes indicators to inform decisions on license allocations. Evidence in terms of indicators is characterized on the basis of impact (extent to which criteria are met) and uncertainty (level of confidence in the available evidence). ACWUA uses Bayesian mathematics to quantify the extent to which one would expect a licence to be granted, given the set of evidence. The user is also prompted to provide narrative support for evidence related to each indicator. The above information is stored and reported as part of the Record of Decision (RoD). ACWUA guides regional authorities by supporting their decision-making despite the limitations caused by incomplete, imprecise, and variable information.

Decisions are based on multiple criteria such as socio-economic factors, race and gender considerations, and alignment with catchment strategy. While it integrates and presents information to inform decision-making, the responsible authority should evaluate the available information and ACWUA results when making a decision. (This paragraph is taken directly from DWAF 2006; Appendix D).

Monitoring against water quality objectives (i.e. Resource Water Quality Objectives or RWQOs) tests compliance with license conditions, i.e. both discharge standards and in-stream objectives. CD: RDM therefore has to develop a strong auditing role to ensure that licensing conditions in terms of the Reserve and the Water Resource Classification System (WRCS) have been met.

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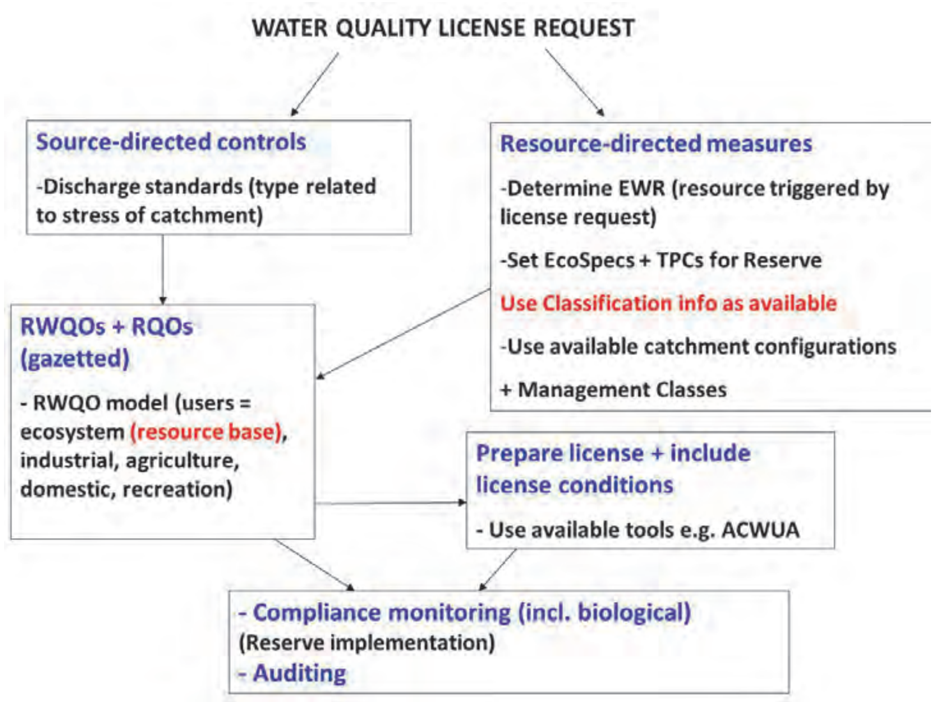

Figure 7 Steps triggered by a water quality license application.

Decisions are based on multiple criteria such as socio-economic factors, race and gender considerations, and alignment with catchment strategy. While it integrates and presents information to inform decision-making, the responsible authority should evaluate the available information and ACWUA results when making a decision. (This paragraph is taken directly from DWAF 2006; Appendix D).

The primary tools currently available to conduct the steps outlined, and their perceived level of availability and implementation, are shown in Table 2 below. Note that this assessment is based on the knowledge of the author during her interactions with the head office and Eastern and Western Cape Regional Offices (RO) of the DWA.

Table 2 Tools available for water quality management (taken from Day and Scherman 2009).

Tool	Availability	Current use by DWA / others
Water Quality Planning Systems documents	Available (DWAF, 2006)	Training workshops conducted in regions, but documents do not appear well known
Regulatory or source-directed control (SDC) tools, e.g. Discharge standards (General + Special Limit Values) + permits	Available	In use by RO of DWA
Market-based management instruments (part of SDC), e.g. remediation, waste discharge charge system, pricing strategy	Available	Limited success??
Self-regulatory management instruments (part of SDC), e.g. ISO 14001 certification	Available	Implemented by a wide range of industries, mines, etc.

Tool	Availability	Current use by DWA / others
Civil management instruments, e.g. civil society, awareness	Available as stakeholder participation in, for example, Water User Forums	Limited applicability at present. Expected to improve with the institution of Catchment Management Agencies (CMAs)
Reserve water quality methods for rivers (all levels of Reserves, i.e. Rapid → Comprehensive).	Available, although the latest river methods manual compiled by Scherman in 2008 has never been finally reviewed and published by DWA (DWAF 2008a Draft).	In use by CD: RDM + Reserve consultants. Training workshops were held with head office, Eastern and Western Cape ROs of DWA in 2007-2009. Training was also conducted as part of an EcoClassification course in May 2011.
Reserve methods for estuaries (Rapid → Comprehensive).	Available as DWAF (2008b). Updated methods are available, but not been finally approved by DWA.	DWAF (2008b) methods in use by Reserve consultants. Elements of latest methods have been tested during recent Reserve studies.
Rapid Reserve water quality methods for wetlands.	Method produced as a WRC project (Malan *et al*. 2013).	Not generally in use by wetland practitioners.
Groundwater water quality Reserve methods.	Water quality is considered as part of the Groundwater RDM (or GRDM) methods (King and Pienaar, 2011).	Methods are applied by groundwater Reserve practitioners.
RWQO model	Version 4.2.0.3. is the latest version (Grobler WQP, pers. comm., March 2013)	Training has previously been conducted in regions but not yet been routinely used.
Licensing tools + license conditions, e.g. ACWUA	Available (but appears to be of limited availability)	Been previously introduced to regions, but not being used by ROs
Catchment-based water quality model, specifically for licensing and Reserve studies	Not available.	-
Monitoring methods + tools	Available as part of DWAF (2006) series, as well as Reserve reports. Updated manual for Ecological Water Resource Monitoring (for the RHP + Reserve monitoring) is available (DWA 2009) – includes a section on water quality	In use as part of license conditions, e.g. biotic and chemical monitoring

All of these tools and systems operate within the National Water Resource Strategy (NWRS) and WRCS.

2.3.4 Regulatory tools (taken from Day and Scherman 2009)

A number of tools are available to the three tiers of government to achieve Integrated Water Resources Management (IWRM) (Burke 2007) and meet water quality objectives (Table 3). The management of pollution at source is a far more efficient and cost effective way of dealing with pollution and achieving resource-based goals. Once an authorization has been issued, certain conditions apply, including the period for which the authorisation is applicable and the conditions that will ensure that the activity does not have a negative impact on the resource. Failure to comply with the conditions may result in:

- the issuing of a directive by DWA or the CMA, which will require the necessary action(s) to be taken to ensure compliance with the conditions;
- withdrawal of the licence; or
- prosecution (usually as a last resort).

Table 3 Water use management tools available to the three tiers of government (Burke 2007; cited in Day and Scherman 2009).

Tool	**National**	**Provincial**	**Local**
Water use authorisations	Section 21 water use authorisation for all users of the water resource – compulsory licensing will come into effect once the CMAs are established.	-	Industrial effluent permits for discharge to sewer
Standards	Resource Water Quality Objectives – quality and quantity	-	Standards or limits set in the by-laws
Environmental authorisations	-	1. Environmental impact assessment process 2. Environmental management programmes for mines	-
Plans/Strategy	1. National Water Resource Strategy and associated strategies 2. Environmental implementation plan / environmental management plan	1. Integrated Development Plan 2. Environmental Implementation Plan	1. IDP 2. Water Services Development Plan 3. Integrated Waste Management Plan 4.IWRM Plan
Water use charges	1. Water Resource Management charge 2. Waste discharge charge system	-	1. Industrial effluent tariffs 2. Potable water use and sewer charges 3. Stormwater management levy
Non legally binding measures to enhance voluntary compliance with regulations	Guidelines, awareness raising and educational programmes	Guidelines, awareness raising and educational programmes	Guidelines, awareness raising and educational programmes

2.3.5 Use of economic instruments (taken from Day and Scherman 2009)

Regulatory economic instruments for water quality management include the Waste Discharge Charge System (WDCS), developed as part of the Pricing Strategy (the original pricing strategy was designed by DWAF in 1998, but was revised in 2004).

The WDCS has three distinct purposes, involving three different water use charges:

- *Management cost recovery*. This water use charge must cover the costs of water resource management activities related to waste discharge.
- *Mitigation cost recovery*. This water use charges must cover the quantifiable costs of infrastructure or other measures for mitigation of existing impacts of waste discharge.
- *Discourage waste discharge.* This water use charge must act as a disincentive to the discharge of waste.

A guarantee will also be retained by the Department in order to finance possible remediation of failed or abandoned activities that have impacted negatively on water resources (DWAF 2006).

Phase 2 of developing the WDCS was completed towards the end of 2005, with the final strategy (Phase 3) originally expected to be completed early in 2006. Phase 3 includes pilot studies being conducted in two areas, the Upper Olifants River in the Witbank area, Mpumalanga, and the Crocodile River West, feeding the polluted Hartbeespoort Dam in the North West Province. It was planned that there would be general implementation of the strategy in South Africa's 19 Water Management Areas by the end of 2006. In the first instance, the WDCS would only apply to direct discharges and disposal of waste, due to the difficulties with quantifying diffuse discharges into ground and surface water (Venter 2004).

The WDCS is primarily a national responsibility and will only be transferred to CMAs once stable, with CMAs being in charge of implementing and regulating the system. However, engagement at the local level is necessary to give effect to the mitigation element, requiring close co-operation with water users, dischargers and impacted communities/enterprises. In developing and reviewing the charge system, co-operation and engagement of Department of Minerals and Energy (mining sector), Department of Trade and Industry, Department of Environmental Affairs (through the Environment Conservation Act) and Department of Agriculture will be required to ensure consistency of the system with other relevant legislation and control and management approaches. As the WDCS flows out of the authorisation process, it will ensure (via control and enforcement, and regulation) and encourage compliance with the authorisation or licensing conditions and ensure achievement of the Catchment Management Strategy (CMS) water quality and water management objectives (DWAF 2006).

2.3.6 Opportunities for self-regulation (taken from Day and Scherman, 2009)

Non-regulatory economic instruments for water quality management include the use of civil society (encouraging general cooperation and awareness) and self-regulatory structures, e.g. ISO 14001 certification (i.e. the international standard relating to good environmental practices). Self-regulation and awareness approaches would be primarily CMA responsibilities and will require the development of co-operative governance at the local level between the CMAs, local government and Water User Associations (WUA), specific users and dischargers (DWAF 2006). Self-regulatory instruments, e.g. ISO 14001, can be included in Environmental Management Plans (EMPs) and Environmental Management Programmes (EMPRs) at the Environmental Impact Assessment stage of a project, and have been widely implemented in the manufacturing and mining industries.

2.4 Water quality issues in South Africa

A number of water quality issues are prevalent in South Africa today – many of which are captured in the list below. Note that many of these issues are exacerbated by changes in flow patterns and modifications of river flow regimes.

2.4.1 Eutrophication

Eutrophication is the process of nutrient enrichment of waters which can result in many changes in the water body, e.g. increased production of algae and aquatic macrophytes (plants), and the deterioration of water quality which is undesirable and interferes with water use. Eutrophication is of concern because it can have numerous negative impacts, which include ecological impacts (such as the deterioration of water quality and loss of biodiversity), aesthetic, recreational and human health impacts. All these impacts also have a significant economic impact. The factors driving eutrophication are high nutrient concentration and stagnation for prolonged periods, with suitable temperature, oxygen concentration and proper light regime. These conditions encourage increased primary growth in the form of algae and macrophytes, culminating in severe blooms and eutrophication or, in extreme cases, a hypertrophic state. Figure 8 is a diagram of eutrophication impact in South Africa as at 2009 (van Ginkel 2011).

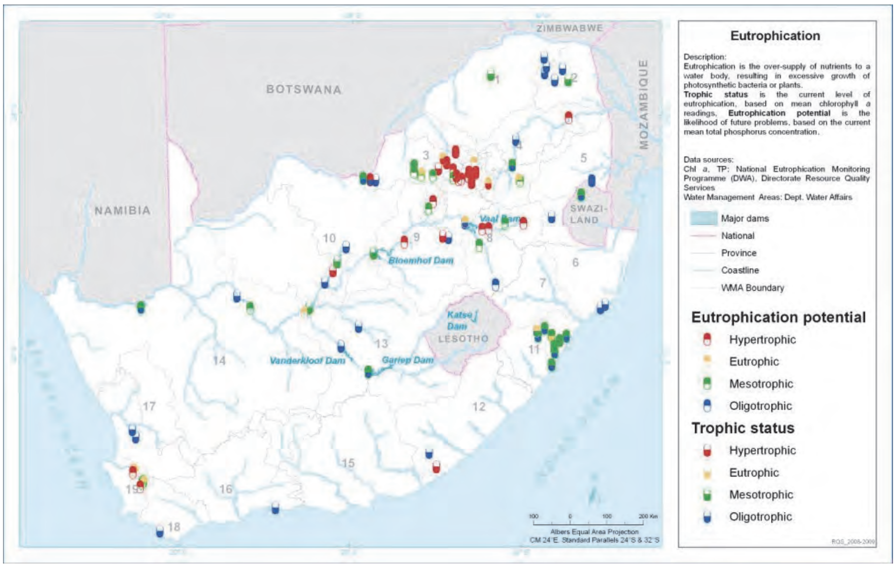

Figure 8 Map showing overall eutrophication impacts in South Africa's 19 catchment management areas (provided by D: RQS, DWA 2009).

High nutrient concentrations are the result of cultural and natural influxes of nutrients. Cultural eutrophication is related to anthropogenic activities – human, social and economic activities. Impacts of cultural eutrophication include:

- Accelerated population growth and associated settlement patterns.
- Watershed or catchment area alterations, such as dams that are built for water storage to supply increasing population needs.
- Increased wastewater treatment works discharges.
- Increased fertiliser applications to increase food production.
- Intensive farming practices that cause increased nutrient-polluted return flows.
- Poor agricultural practices, for example when farmers plough and cultivate the riparian zones of water resources.

Natural eutrophication is caused by the influx of nutrients from natural sources, including the rocks, soil and other natural features within a catchment area. This type of eutrophication is not reversible or controllable, and will therefore continue slowly and inevitably (van Ginkel 2011).

The results of a Downing and van Ginkel (2004) study on cyanobacteria (or blue-green algae) show that cyanobacterial problem events are widespread and typically seasonal, with water resources subject to eutrophication commonly experiencing problems. There exists a geographical variation in the frequency, duration and severity of the problems primarily due to the condition of the catchment, but also the nature of the water source, abstraction points, and regional climatic conditions. *Microcystis* is the dominant problem cyanobacterial genus, with *Anabaena* also being common.

Figure 9 Blue-green algae forming a black crust on Hartbeespoort Dam (source: J Koekemoer).

Both *Microcystis* and *Anabaena* produce hepato- and neurotoxic secondary metabolites, resulting in acute or sub-acute liver toxicity. However, the most common problem associated with cyanobacterial bloom events or increased cyanobacterial biomass, is taste and odour. Data on toxin and geosmin / 2-methyl isoborneol levels are extremely limited due to the limited resources for analysis and the cost of such analyses. Without significant improvement in eutrophication status of South Africa's freshwater resources, increasingly severe problems with longer duration of events can be expected in future.

The presence of high concentrations of nutrients also supports the rapid growth of macrophytes. Excessive macrophyte biomass blocks waterways, impedes access to dams and rivers, clogs drainage systems and contributes to flooding and the destruction of canals. Water hyacinth (*Eichhornia crassipes*) control alone costs South Africa in the order of R12 million per annum. Many aquatic macrophytes are exotic, problem-causing species, including water hyacinth, red water fern (*Azolla* spp.), water lettuce (*Pistia stratiotes*), Kariba weed (*Salvinia molesta*), Hydrilla (*Hydrilla verticillata*) and parrot's feather (*Myriophyllum aquaticum*). Hydrilla is the latest addition to the list of problematic species, although its presence was already recorded in South Africa as early as 1963 (Coetzee *et al*. 2011, cited in van Ginkel 2011).

Management options for reducing nutrient enrichment are as follows (Dallas and Day 2004, van Ginkel 2011):

- Removal or reduction of nutrients from sewage.
- Improvement of agricultural practices, including both land preparation and fertilizer application, such as:
    - Selective and specific application of fertilizers to reduce direct entry into water bodies.
    - Timing of application to avoid heavy rainfall and coincide with peak growing season.

    - Avoidance of fertilizer application to areas susceptible to erosion.
    - The establishment of buffer strips in riparian zones to mitigate the movement of nutrients from steep slopes into water.
    - Manage waste from high-density livestock as a point source of pollution.
- Control of urban non-point source pollution via the use of retention ponds, wetlands, greenways, litter control and street sweeping, reduction of impervious area, and reduction of erosion.
- Reduction of atmospheric deposition of N by more efficient use of fertilizers and improved handling of animal waste.
- Use of low or zero-phosphate detergents (see a WRC study by Quayle and colleagues in 2010 entitled Investigation of the positive and negative consequences associated with the introduction of zero-phosphate detergents into South Africa).
- Physical removal of algal mats – see Figure 10 below for an example.

Figure 10 Physical removal of algae at Hartbeespoort Dam. Note that gas masks are worn (source J. Koekemoer).

- Biomanipulation techniques target food-chain functioning and involve the use or harvesting of non-desirable organisms to eventually control algal growth or other components of the food chain that may cause eutrophication-related problems. The main aim is to control certain key species at critical points in the food web, e.g. fish species that prey on zooplankton to an extent that may alter the normal functioning of the ecosystem. A DWA project currently underway is Harties, Metsi a Me on the Hartbeespoort Dam to combat the severe eutrophication impacts in this system. This project is testing an integrated water-management system, so testing methodologies, monitoring, harvesting fish and improving upstream wetlands. One of the objectives of the project is to test the hypothesis that the gradual increase of filter-feeding cladoceran zooplankton, specifically during summer, is the result of a greater abundance of the edible algal species preferred by zooplankton present in the system, which in turn came about in response to the manipulation of the food chain by fish harvesting. Updates on the project can be found on the following website: www.harties.org.za.
- Epilimnion mixing to combat cyanobacterial blooms (Hart and Hart 2006, cited in van Ginkel 2011), for example, the installation of 16 SolarBee® (patented solar-powered water circulator)

pumping units between 2008 and 2009 near the intake water inflow into Rietvlei Dam. The pumps cause laminar flow that disturbs the water column to such an extent that conditions become unfavourable for cyanobacterial growth. According to Coetzee (2011, cited in van Ginkel 2011) the phytoplankton community has shifted away from the regular annual cyanobacterial blooms since installation of the SolarBee® instruments.

Water quality guidelines used in South Africa for protecting aquatic ecosystems (DWAF 1996e) from nutrient enrichment ensure that the trophic status of the water body does not move in a negative direction, i.e. to a more eutrophic state. This means that it is necessary to both know the natural trophic state of the system and the likely concentrations of P and N that may cause a shift in nutrient status. Table 4 below shows the effects of different levels of P and N in the water (DWAF 1996e).

Table 4 Symptoms or effects associated with ranges of inorganic phosphorous and nitrogen concentrations (after DWAF 1996e, cited in Dallas and Day 2004).

Average summer concentration (mg/L)		
Inorganic Phosphorous	**Inorganic Nitrogen**	**Symptoms or Effects**
< 0.005	< 0.5	Oligotrophic conditions
0.005-0.025	0.5-2.5	Mesotrophic conditions
0.025-0.25	2.5-10	Eutrophic conditions
> 0.25	> 10	Hypertrophic conditions

The variables used to define nutrient status for the Ecological Reserve and Classification, are shown below (DWAF 2008a):

- Total Inorganic Nitrogen: TIN-N in mg/L (i.e. the N portion of all inorganic nitrogen sources, viz. $NO_2 + NO_3 + NH_4$). Note: NH_3-N is not included as NH_3 is regarded as a toxicant by the method, and is calculated from the levels of NH_4 and the pH. At pH <10 NH_3 is only a minor fraction of total reduced nitrogen.
- Phosphate: PO_4-P or the P portion of PO_4 in mg/L (also referred to as SRP (Soluble Reactive Phosphorous) or orthophosphate).
- Phytoplankton: chlorophyll-a in µg/L (recommended for Intermediation and Comprehensive Reserve assessments).
- Periphyton: chlorophyll-a in mg/m2 (recommended for Intermediation and Comprehensive Reserve assessments).
- Diatoms: results expressed using the Specific Pollution Index (SPI) (recommended for Intermediation and Comprehensive Reserve assessments).

2.4.2 Acid Mine Drainage and other industrial chemicals

Acid mine drainage (AMD) occurs when water flows over exposed sulphide minerals, which oxidise in the presence of water and oxygen causing the water to become acidic, which then dissolves other toxic metals. Exposure and oxidation of pyrite and other sulphide minerals occur in mine wall rocks, backfill, waste rock piles, low grade ore stockpiles and tailings deposits. In and around South African gold mines, pyrite (FeS_2) present in gold ore dissolves on oxidation and releases iron and sulphuric acids. Apart from iron, the associated decreasing pH is also conducive to the mobilisation of various other metals, such as copper, lead, aluminium, manganese and uranium. Although the AMD-generating reactions also occur in abiotic environments, colonies of microorganisms such as certain acidophiles (bacteria thriving under acidic conditions), greatly accelerate the decomposition of metal ions. AMD, generated through the ingress of water into mine voids, is generally characterised by one or more of the following: low pH, high salt content (mostly made up of sulphates), and high levels of metals – particularly iron (giving it the red-orange colour). In cases where uranium is present, radiological risks may also be present (DWA South Africa website).

Pyrites are also common in coal deposits and AMD is also found in association with coal mining areas in South Africa.

AMD has been described as the largest single environmental problem facing the mining industry, particularly because it is persistent and costly, and tends to be a liability for mines long after they cease to operate. The importance of finding a solution to the rising AMD issue in South Africa and the need for inter-departmental cooperation led to the establishment in 2010 of an Inter-Ministerial Committee (IMC) on AMD, comprising the Ministers of Mineral Resources, Water and Environmental Affairs, and Science and Technology, and the Minister in the Presidency: National Planning Commission.

Due to the potential high impact of AMD discharge into the Vaal River system specifically, a Feasibility Study for a long-term solution to address Acid Mine Drainage (AMD) associated with the East, Central and West Rand underground mining basins in the Gauteng Province was launched in 2011 by the Department of Water Affairs. Phases 1 (study initiation phase) and 2 (pre-feasibility phase) have been completed, with Phase 3 (feasibility phase) currently being concluded. The Feasibility Report should be released by the end of 2013.

Figure 11 AMD contamination at O'Kiep, Northern Cape (source: J Jay).

If AMD, which has not been desalinated, is discharged into the Vaal River System, the high salt load will require large dilution releases to be made from the Vaal Dam to achieve the fitness-for-use objectives set for the Vaal Barrage and further downstream. This would result in unusable surpluses developing in the Lower Vaal River. Moreover, if dilution releases are still required after 2015, the acceptable levels of assurance of water supply from the Vaal Dam would be threatened. This will mean that there would be an increasing risk of water restrictions in the Vaal River water supply area, which will have negative economic and social implications. These negative impacts will be much greater if the catchment of the Vaal River System enters a period of lower-than-average rainfall with drought conditions. Since decant started in the Western Basin in 2002 the continuous flow of untreated AMD, and now the salt load from the continuous flow of the neutralised AMD from the Western Basin, are impacting on the Crocodile (West) River System (DWA 2013b).

The main objective of the feasibility phase is to carry out intensive feasibility level investigations and optimisation of the most feasible layouts for each basin and to select a preferred option to be used as a Reference Project for each basin. The requirements for implementation were also considered and evaluated (DWA 2013b).

The main contributors to acidification in South Africa are therefore the gold mining area of the Witwatersrand and the coal mining industry. The groundwater within the gold mining areas is heavily contaminated and is discharging into streams in the area and contributing up to 20% of total stream discharge, lowering the pH in the stream water while most of the metal load is precipitated. South Africa’s coal mining industry is the second largest mining sector after gold, with sales contributing 16% of export revenue in 2003. Together with its southerly neighbours, the Highveld and Ermelo coal fields, the Witbank coal field represents the largest conterminous area of active coal mining in South Africa. These coal fields produce coal for power generation and support 48% of the country’s total power generating capacity (Tshwete *et al*. 2006, cited in CSIR 2010). The impacts of coal mining on the upper Olifants River catchment as a result of coal mining activities have been noted (Heath *et al.* 2010, Aston and Dabrowski 2011, Dabrowski and De Klerk 2013).

Industrial contributions to pollution depend on the industrial process adopted but can include poisonous and hazardous chemicals, nutrients, elevated salinity and increased sediments (CSIR 2010). The main impacts of industrial chemicals relate to salinization, which may render water unfit for reuse or very costly to treat. Typical pollutants associated with industrial water use include:

- Heavy metals (lead, chromium, cadmium, arsenic, vanadium).
- Dyes.
- Chemicals such as chlorine, phosphate and nitrates.
- High organic compounds in the form of Chemical Oxygen Demand (COD).
- Brine and sewage sludge.
- Organic compounds originating from raw materials, intermediates, products, reagents, solvents and catalysts.

2.4.3 Pollution by sewage effluents

Poorly managed sewage effluents are characterized by high nutrient and salt levels, and often elevated toxics and pathogen levels. Irrigating crops with sewage effluent can lead to crops being heavily contaminated with the four regulated elements, namely cadmium (Cd), copper (Cu), lead (Pb) and zinc (Zn), as was shown in a Zimbabwean study (Muchuweti *et al*. 2006, cited in CSIR 2010). One of the main constituents of domestic sewage is dissolved and particulate organic matter, with sewage being one of the main forms of organic enrichment. The major effects of organic enrichment are a decrease in dissolved oxygen concentrations, an increase in turbidity and the concentration of suspended solids, an increase in nutrient concentrations and possible bacterial contamination of water bodies. The most severe impact is the reduction in oxygen levels, measured as Biological Oxygen Demand (BOD) (Dallas and Day 2004).

Bacterial contamination via sewage discharges are primarily linked to *Escherichia coli*, which is used as an indicator of faecal pollution and may cause diarrhoea and gastroenteritis. Conditions caused by pathogenic organisms present in animal waste which can be transferred to humans via contaminated water include salmonellosis, anthrax, tuberculosis, tetanus and colibacillosis (Dallas and Day 2004). The spread of diseases such as cryptosporidiosis, dysentery, cholera and typhoid is caused by the use of water that is contaminated by faecal matter (Momba *et al*. 2004, cited in CSIR 2010). Surface and drinking water quality, in peri-urban and rural areas, is further compromised by unskilled plant operators, old and inadequate infrastructure and poor maintenance. Interruptions in the water supply and provision of poor quality water are common in these areas.

The extent and impact of non-compliant sewage discharges in South Africa, lead to the DWAF instituting a reporting system in 2008 called Green Drop. Green Drop is an incentive-based regulation adopted by DWA as a means to identify, reward, ensure and encourage excellence in the field of wastewater management. The Green Drop regulation programme was launched by the Minister on 11 September 2008. In parallel, the Department commenced with a full scale assessment of all municipal plants across South Africa and used this baseline to develop the risk-based regulatory approach. This approach is two-pronged and is based on the following:

- The Green Drop Certification incentive-based regulation seeks to identify and develop the core competencies required for the sector that if strengthened, will gradually and sustainably improve the level of wastewater management in South Africa; and
- Risk-based regulation seeks to establish scientific baselines comprising of the critical risk areas within the wastewater services production and to use continuous risk measurement and reporting to ensure that corrective measures are taken to abate these high and critical risk areas.

The 2012 Green Drop Report provides feedback and progress on the following:

- Current status and risk trend of municipal wastewater treatment.
- Current status of public treatment facilities (Department of Public Works).
- Status of four privately owned treatment facilities.
- Progress made using targeted risk-based regulation in specific catchments.

The 2011 Green Drop Report stated that less than half of South Africa’s 821 sewage works were treating the effluent they received each day to safe and acceptable standards.

The Green Drop Risk Profile Progress Report for 2012 is the product of a ‘gap’ year, whereby progress is reported in terms of the improvement or decline in the risk position of the particular wastewater treatment facility, as compare to the previous year’s risks profile. The 2012 Report therefore presents the current risk profile and a 3-year trend analysis of treatment plants on three levels:

- System specific risk data and information pertaining to the performance of each wastewater treatment system per Water Services Institute (WSI).
- Region specific risk figures and information to highlight the strengths, weaknesses and progress for the collective of WSIs within the province or region.
- National overview that collate and elevate the detailed findings on system level to that of a provincial overview, which can then be compared and inculcated as a national view of wastewater treatment performance.

The overall progress on a nation-wide scale for 2012 can be summarised as follows (DWA, 2012a):

- 440 plants showed progress by taking up lower risk positions, whilst 323 plants digressed by taking up increased risk ratios, and 68 plants maintained their status of 2011.
- The majority of plants are in moderate risk (241 plants) and low risk (225 plants), with 212 plants in high risk and 153 plants in critical risk space.

2.4.4 Salinization

Anthropogenic salinization of inland waters emanate from a number of sources, e.g. saline industrial effluents, irrigation, clear-felling and return flows from sewage effluents. Diffuse pollution, resulting from poorly managed urban settlements, waste disposal on land and mine residue deposits, can pose a larger problem than point source pollution because the impact is more widely spread. It is also only detected in the water system after prolonged exposure and is difficult to monitor and control (Oberholster *et al*. 2008, cited in CSIR 2010). The effect of diffuse pollution on groundwater is also often difficult to reverse.

Problems associated with salinization include (CSIR 2010):

- Reduction in the yield and quality of crops and fruit due to soil and water salinization.
- Increased scale formation and corrosion in domestic and industrial water conveyance systems.
- An increased requirement for pre-treatment of selected industrial water uses.
- Changes in the community structure of aquatic biota present in aquatic systems.

The impact of increasing salinity levels on aquatic organisms can be summarized as follows (Dallas and Day 2004):

- It is often the rate of change rather than the final salinity levels that is most critical. Many organisms can physiologically adapt to slow changes or acclimate to higher salinity levels.
- Juvenile stages are generally more sensitive to changing salinity levels.
- In general there appears to be an upper salinity level of around 5000-8000 mg/L above which acclimation is no longer possible.
- Salinity may act antagonistically or synergistically with other toxicants due to its impact on speciation.
- The response of freshwater organisms is also related to their evolutionary origin.
- Toxicity of various salts is related to the toxicity of individual ions.

In South Africa, water quality guidelines for aquatic ecosystems exist for Total Dissolved Solids (TDS) only, and not for any of the individual salt ions (DWAF 1996e). DWAF (2008a), used for Ecological Reserve assessments for rivers, contains values for benchmark categories (i.e. associated with ecological categories A-F) for Electrical Conductivity and the following inorganic salts: sodium chloride, sodium sulphate, magnesium chloride, magnesium sulphate, calcium chloride and calcium sulphate.

Chemical (ionic) data collected during monitoring can be converted to inorganic salts using a tool called TEACHA, i.e. Tool for Ecological Aquatic Chemical Habitat Assessment. This is currently the only tool available in South Africa for conducting inorganic salt assessments and was developed by Dr S Jooste of D: RQS. As a desktop tool, it is also essential for processing water quality licenses for the Ecological Reserve. Future activities for TEACHA use include the further development of TEACHA as a user-friendly tool, with improvements in the graphic user interface and expanded pilot testing.

An issue that currently needs attention in South Africa is the relationship between salinity indicators used for routine monitoring (salt ions), Reserve determinations (inorganic salts, as toxicity testing can only be conducted using this form of salts), and Resource Water Quality Objectives (salt ions for users other than ecosystems). As toxicity testing is the basis of assessing the impact of rising salinity levels of instream organisms, a fully functioning tool is needed to convert monitored ions to salts (i.e. the further development of TEACHA).

Water quality guidelines are based on toxicity data (see Jooste and Rossouw 2003 and Kefford *et al*. 2005), with the drive in South Africa being the use of indigenous riverine organisms for salinity toxicity testing. A number of publications exist on this topic, e.g. Goetsch (Scherman) and Palmer (1997), Palmer *et al*. (1996, 2004b,d, 2005); Palmer and Scherman (2000, 2001), DWAF (2000), Scherman *et al*. (2003a,b), Zokufa *et al*. (2001), and Kefford *et al*. (2004, 2005). The salt tolerances of a range of organisms and life-stages has been investigated, e.g. mayfly adults (such as Tricorythidae, Leptophebiidae and Heptageniidae), limpets (such as *Burnupia stenochorias*), shrimp adults and eggs (*Caradina nilotica*) and *Daphnia*. Some of the work on these organisms culminated in the DWAF (2004a) document used as the basis for the benchmark categories for Electrical Conductivity used in DWAF (2008a). The premise of all this work is to consider the proportion of species at risk from a particular toxicant, based on the species sensitivity distribution (SSD) for that toxicant. This also assumes that the sensitivity data from the species included in the SSD are sufficient to predict the effect on species for which there are no data. For the assumptions of SSDs to be better met, there is therefore a need for tolerance data on more species, from more taxonomic groups and widespread localities, including South Africa (Kefford *et al*. 2005).

Two water quality models are currently used in South Africa for simulations of salinity and sulphates (van Rooyen, WRP Consulting, pers. comm., November 2013):

- The salinity calibration model (WQT) with the purpose of calibrating different modules over an appropriate historical period.
- The Water Resource Planning Model (WRPM), containing the calibrated parameters from WQT. The purpose of this model is to undertake risk analysis of projection scenarios to provide decision support for development and operational planning purposes.

2.4.5 Toxic organic pollutants

Agrichemicals

Biocides, also known as pesticides, are chemicals that kill living organisms and are used in the control of pests. The most commonly used biocides are herbicides, fungicides and insecticides (Dallas and Day, 2004). South Africa is a significant user of pesticides in agriculture, suggesting that the potential for non-point source environmental contamination is high. The type of pesticide and usage pattern is variable across South Africa, depending on climatic conditions and crop types, e.g. fungicides are used widely in the wetter grape, fruit and wheat-producing areas of the Western Cape, but herbicides are more prevalent in the maize-growing areas. Pesticide use also occurs as vector control in public health, commercial pest control, domestic use and in selected industrial or food-processing technologies (e.g. dried fruit and timber-related production activities). However, the infrastructure to monitor and control pesticide use is also poorly developed. Data on pesticides in water sources are sparse, with previous results constrained by high detection limits masking significant levels of pesticides. This is of particular significance as endocrine disrupting effects occur at concentrations far lower than other toxic effects (London *et al*. 2000).

For many biological systems contamination is via polluted waters, although contamination of groundwater is of particular concern due to the duration of impacts and the time taken for recovery. Pollution of surface water normally occurs from runoff, although contamination by volatilization can occur, i.e. the dispersion of pesticides from plant and soil surfaces into the atmosphere. Groundwater contamination is generally through leaching, but can also occur via direct streaming or the movement of pesticides through conduits. The factors determining runoff and leaching include the physio-chemical properties (e.g. water solubility, soil mobility and pesticide persistence) of the pesticide and soil properties (e.g. texture, organic matter content, depth of the soil horizon, erosion potential, pH and microbial content) (London *et al*. 2000). Of primary concern, however, is the persistence and accumulation of pesticides in food chains, and about the role of certain pesticides in causing reproductive failure and endocrine system abnormalities in species not intended as their target. Control of pesticide use is therefore paramount.

Little systematic monitoring of potential environmental impacts of pesticide use takes place in South Africa. The absence of monitoring for organic contaminants is largely linked to expensive analytical method, the scarcity of suitable laboratories for organics analyses, and lack of resources on the part of local authorities who have no option but to focus firstly on the provision of services.

Table 5 shows standards in place to protect aquatic ecosystems for a selection of pesticides (Dallas and Day 1993). More detail can be seen in Appendix A of London *et al*. (2000).

Table 5 Water standards to protect aquatic ecosystems (µg/L).

Substance	South Africa	United Kingdom	US EPA	Australia	Canada
Aldrin*	0.01	0.01		0.01	0.004
Chlordane*	0.025			0.004	6.0
DDT*	0.0015	0.025		0.0005	0.001
Dieldrin*	0.005	0.01		0.002	
Chlorpyrifos				0.001	
Endosulfan	0.003			0.001	
Endrin*	0.002		0.05	0.003	0.0023
Azinphos-methyl				0.01	
Heptochlor*	0.005			0.0003	
Lindane	0.015			0.003	
Malathion	0.1			0.07	
Methoxychlor	0.02			0.04	
Mirex*	0.001			0.001	
Parathion	0.008			0.004	
Toxaphene*				0.00	
2,4 Dichlorophenol	4.0				

* original chemicals listed by the Stockholm Convention on POPs

The use of pesticides in South Africa is controlled by the Pesticide Management Policy (Government Notice 1120, Government Gazette of 24 December 2010), which was promulgated by the Department of Agriculture, Forestry and Fisheries (DAFF) under the Fertilizers, Farm feeds, Agricultural Remedies and Stock Act, 1947 (Act No. 36 of 1947). The Policy was written to provide information and guidelines to support legislation and regulations. It sets out a framework to ensure improvements of pesticide production, and use and disposal throughout the full life-cycle of pesticide use, so as not to pose significant adverse effects and health and the environment. The Notice refers to the National Toxicity Monitoring Programme (see Section 5.4) initiated by DWA, which monitors the level of pesticides in groundwater and surface water. There will also be requirements and restrictions in place regarding the minimization of pesticide use, buffer zones and restrictions on aerial spraying. Meeting international obligations also need to be ensured, e.g. compliance with the Rotterdam Convention on Prior Informed Consent (PIC) which requires that an exporter of chemical obtains consent of the receiving country before delivery and the Stockholm Convention of Persistent Organic Pollutants (POPs) (Government Notice 1120).

POPs

The Stockholm Convention on POPs originally focussed on 12 chemicals or chemical classes. This list consisted of eight pesticides (see Table 5), the industrial chemicals polychlorinated biphenyls (PCBs) and hexachlrobenzene (HCV), and dioxins and furans which are by-products of industrial processes. Nine additional POPs were listed in 2009, and in 2011 endosulfan joined the list. The US EPA has classified 16 polyaromatic hydrocarbons (PAHs) as priority pollutants based on toxicity, potential for human exposure, frequency of occurrence at hazardous waste sites, and the extent of information available. PAHs are composed of carbon and hydrogen atoms in fused benzene rings. Their widespread occurrence is due to their formation and release during the incomplete combustion of organic materials in furnaces, fireplaces, gas and oil burners, and in the production of coke and carbon, in petroleum processing and aluminium sintering, in coal power plants and in cars and trains. Natural sources such as volcanic activity contribute slightly to environmental levels, although the role of vegetation fires is currently unknown (ORASECOM 2013).

Research in South Africa has shown that very little is known about levels of POPs and PAHs or distribution patterns, whereas more information is known about heavy metals. For example, South Africa has been listed as the country with the second highest mercury emissions in the world (Pacyna *et al*. 2006, cited in ORASECOM 2013), based on estimates of total mercury emissions from gold mining and coal combustion.

2.4.6 Other factors, e.g. climate change

Other factors contributing to changes in water quality in South Africa include factors such as climate change, unpredictable rainfall patterns, highly variable climate, invasions by alien plants and excessive water abstraction which exacerbate water quality issues. Given the predictions associated with global and climate change and the down-scaled forecasts of increased temperatures across South Africa, a rise in air temperature of 2 degrees celsius will likely have far-reaching effects on the quality of water in river systems and water storage reservoirs. In particular, higher water temperatures will alter water-gas equilibria and increase the rates of microbial processes, which will in turn accelerate nitrification, denitrification, respiration and methanogenesis (the generation of methane by anaerobic bacteria). Higher water temperatures will also lead to increased rates of evaporation, thereby reducing the volumes of water needed for a growing population (CSIR 2010).

The development of a climate change response for the water sector in South Africa is a requirement of the White Paper on the National Climate Change Response (the National Water Resource Strategy 2 (DWA 2012b)). The Vision for the Climate Change Strategy is as follows:

The vulnerability of people, the ecosystems and the economy for climate change is considered and integrated into both short-and medium-term water sector planning approaches.

Due to the significant water quality changes which can be brought about by the impact of climate change, water quality state will have to be addressed in the Climate Change Response. Another issue of significance is groundwater responses to climate change, and specifically technical issues such as groundwater recharge. As South Africa's surface water resources are largely being used to full capacity, and groundwater is more resilient to drought conditions, groundwater will form an important part of any climate change adaptation strategy (DWA 2010c).

2.5 Monitoring initiatives in South Africa

Although a wide range of monitoring programmes exist in South Africa, care should be taken that the analysis and use of data generated by these programmes takes place. Monitoring should not be for monitoring's sake, but to enable effective decision-making. The biggest gap in monitoring activities in South Africa at present is follow-up monitoring due to take place upon completion of a Reserve study. Resources are being spent in conducting Reserve or EWR studies, during which baseline monitoring is conducted and ecological specifications (EcoSpecs) and Thresholds of Probable Concern (TPCs) set up to enable management against measurable objectives, but monitoring is generally then not initiated. If five years have elapsed since a Reserve determination study has been completed, results would need review and a new baseline may need to be set up, thereby requiring the commitment of more resources. These costs could be avoided by inclusion of the sites into a proper monitoring programme. This concept is explored further in the section below on Ecological Water Resources Monitoring (EWRM), which has a water quality component.

Resource Quality Monitoring at DWA therefore includes a number of reporting activities, including the development of guidelines and procedures for the monitoring and assessment of water resource quality. It also involves management of national monitoring programmes, namely the National Microbial Monitoring Programme (NMMP), the National Aquatic Ecosystem Health Monitoring Programme (including the River Health Programme or RHP), the National Eutrophication Monitoring Programme (NEMP), the National Toxicity Monitoring Programme (NTMP) and the National Chemical Monitoring Programme (NCMP). Other areas of activity include the application of ecotoxicology (ecological risk assessment), and support for the RDM office.

2.5.1 Chemical monitoring

Chemical monitoring is conducted at a number of river gauging weirs in South Africa. Day and Scherman (2009) state that National Chemical Monitoring Programme (NCMP) monitors about 40 core sites and a total of more than 700 sites, roughly 330 of which are sampled at an average frequency of about two weeks. The raw data are curated on DWA's Water Management System (WMS) database. Data can be requested via DWA and is accessible on the website of the Resource Quality Services Directorate (http://www.dwaf.gov.za/iwqs/water_quality/NCMP.htm) as carefully synthesised long-term data sets. The Directorate: Water Resources Information Management has also begun compiling internal annual reports, with the latest available being for 2007-2008 (DWA 2011b).

DWA's Resource Quality Services (RQS) section has a very good record of water (resource) quality monitoring going back some forty years and covering hundreds of sites, mostly rivers, throughout the country. There is very good knowledge of some aspects of the water chemistry of rivers, particularly levels of Electrical Conductivity or salts (i.e. Total Dissolved Solids, TDS), major ions and nutrients. Gaps in WMA include metals data, organic pollutants, turbidity, temperature and oxygen – all which are required for determining water quality state of rivers.

Groundwater is also monitored by DWA as part of long-term monitoring. Groundwater monitoring stations around the country are shown in Figure 5 below (DWA 2010c). Data is stored in a web-based database called the National Groundwater Archive (NGA) that was released internally to the Department users in October 2008. The NGA contains data with in excess of 249 000 geosites, of which 242 000 are boreholes. However, a number of issues exist (DWA, 2010c):

- There is evidence that groundwater data holdings are declining in parts of the country (e.g. the Tshwane dolomites near Pretoria).
- Inaccessibility of data and fragmentation of databases is also a serious problem.

- Earlier records have estimated locations based on the cadastral farm name on which the borehole is found – this can lead to inaccuracies in position of several kilometres.
- Records date from the early part of the last century to the present day; some records are many decades long, others consist of only a single point and date.
- Much groundwater data is held by the private sector, where it is difficult to access, and private sector drillers do not submit borehole data to the Department.
- A lack of standard data capturing formats also hampers the integration of databases.
- Improved institutional arrangements for data collection are required.

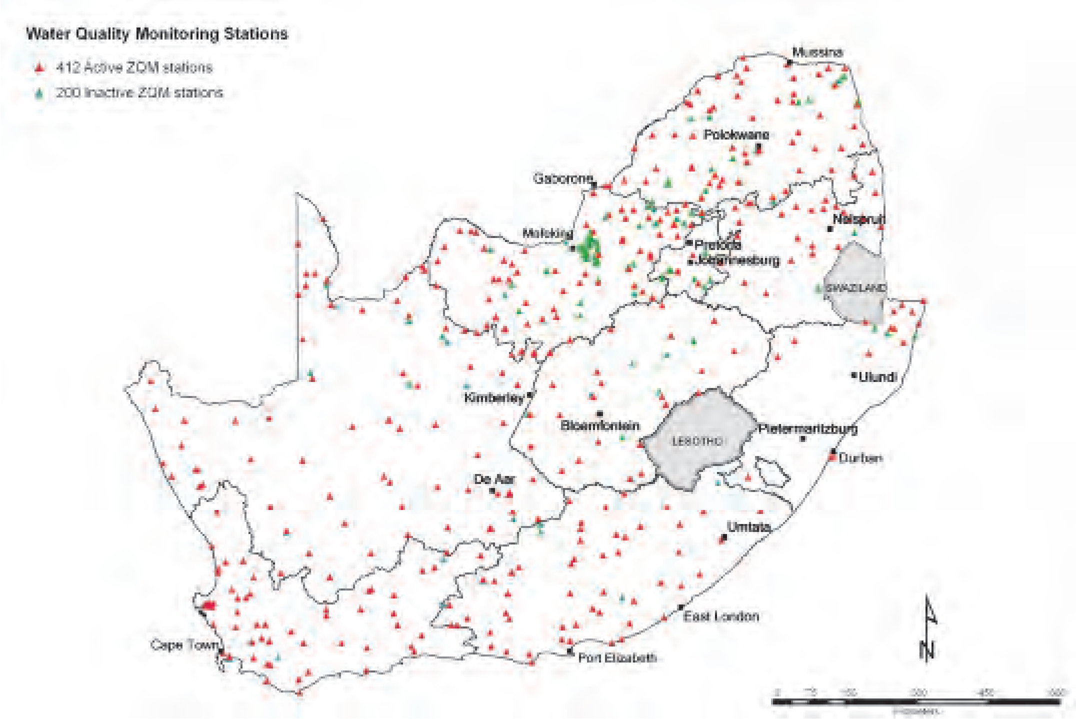

Figure 12 Groundwater monitoring stations around South Africa (DWA 2010c).

See Section 2.6.3 for more information on water quality issues in groundwater.

Estuary monitoring is covered in the section on Water quality and Estuaries, i.e. Section 2.6.2.

2.5.2 Eutrophication monitoring

The National Eutrophication Monitoring Programme (NEMP) monitors variables such as chlorophyll, dissolved oxygen, total nitrogen and total phosphorus in addition to those measured by the NCMP and is focused on dams. Information exists for about 80 reservoirs (dams). Monitoring eutrophication assists in tracking nutrient pollution. A short report for 2003 is available on the NEMP website (http://www.dwaf.gov.za/iwqs/eutrophication/NEMP/default.htm). A WRC report is also available which looked at cyanobacterial blooms in 1999-2000 (Downing and van Ginkel 2004). Additional documents include a guideline manual for management urban impoundments in South Africa (Freeman *et al*. 2000). The purpose of the Guideline Manual is to assist those responsible for, or with an interest in, the management of the water quality of urban impoundments. It is thus aimed predominantly at those persons in local authorities who must manage the water bodies in their areas. A framework for managing urban impoundments is presented, which examines the planning and design of the impoundment and how management techniques can be deployed to address water-quality problems within the categories of catchment, pre-impoundment and in-lake management. A range of management techniques are described in the Manual which can be used to combat commonly occurring water quality problems.

2.5.3 National Microbial Monitoring

The National Microbial Monitoring Programme (NMMP) monitors the extent of faecal contamination in surface waters. The specific objectives of the NMMP are as follows and are taken from the website (http://www.dwaf.gov.za/iwqs/microbio/nmmp.asp, March 2013):

- To locate, assess and prioritise those areas in the country where potential health risks related to faecal pollution of water resources are highest.
- To provide information on the status and trends in extent of faecal pollution, in terms of the microbial quality of surface water in the potential high risk areas.
- To provide information to help assess the potential health risk to humans associated with the possible use of faecally polluted water resources.
- To help assess the effectiveness of measures to protect water resources against faecal pollution in terms of trends in the microbial water quality.

The term "potential high health risk area" used in the NMMP is to describe an area that has (at the time that the assessment was done) been exposed to factors that might have contributed to a potential health risk, due to faecal pollution of surface water resources, in that area.

Sample analysis occurs at RQS and at a variety of contracted laboratories across South Africa. The results are reported to the regions every two months and country-wide results are mapped. The Water Services Directorate has a separate system for municipal data and recently municipal statistics have been released for the whole country. Extensive municipal databases are available for some cities and are used to track problematic sites. A challenge for DWA is the accessibility of municipal data in the same format as WMS, thereby enabling the easy use of these data for relevant studies (Day and Scherman 2009). An example of a NMMP in action is that of the Berg River WMA (Rossouw 2004). A presentation of results is available on the DWA website.

Other initiatives for monitoring diseases caused by water-related microbes have been developed by the WRC and DWA, e.g. the following set of documents which formed part of the series of Guides on the Management of Water-related Microbial Diseases:

Vol 1: What is the problem? – Disease Characteristics.
Vol 2: What causes the problem? – A What to Do for Water Suppliers following Diarrhoea Incidents.
Vol 3: How great is the problem? – Health Impact Assessment.
Vol 4: How dangerous is the problem? – Communicating the Risk.
Vol 5: What we and our children need to know – Health & Hygiene Awareness.

The following water-related infectious diseases were covered in the series: Amoebic dysentery, Bilharzia (schistosomiasis), Campylobacteriosis, Cholera, Cryptosporidiosis (“Crypto”), Gastroenteritis, Giardiasis, Hepatitis A, Leptospirosis (Weil’s disease), Malaria, Poliomyelitis (“polio” or “Infantile paralysis”), Shigellosis (shigella dysentery), Swimmer’s itch (non-human bilharzia), Trachoma and Typhoid fever.

Monitoring of the microbiological quality of a groundwater source gives important information about the groundwater system, and early warning of problems.

2.5.4 National Toxicant Monitoring Programme

Toxicants are chemical pollutants capable of exhibiting a toxic effect. The objectives of the National Toxicant Monitoring Programme (NTMP) is to measure, assess and regularly report on the status and trends of the nature and extent of the following in a manner that will support strategic management decisions in the context of fitness for use of those water resources, and will be mindful of financial and capacity constraints, yet, be soundly scientific:

- Potentially toxic substances in South African water resources (watercourses, groundwaters and estuaries).
- The potential for toxic effects to selected organisms.

The NTMP was designed in response to increasing local and international concerns about the detrimental effects of toxicants that are being released into the environment and to address the current lack of a coherent source of information on the occurrence of toxic substances in South African water resources. The project was initiated in 2002, the same year that South Africa signed the Stockholm Convention on Persistent Organic Pollutants (POPs), which came into force on 17 May 2004. Obligations for South Africa in terms of the Convention include the development and maintenance of appropriate information dissemination programmes (Article 10) as well as the undertaking of research on all matters relating to POPs. This includes monitoring, socio-economic impacts and release reduction (Article11). In order to meet the information requirements of the Stockholm Convention regarding the presence of POP's in fresh surface water resources, the monitoring of POPs (excluding dioxins, fire retardants, furans and hormones) were included as variables of concern in the conceptual design of the NTMP. The NTMP project comprises of the following four distinct phases (Jooste *et al*. 2008):

Phase 1: Needs assessment (completed in March 2003);
Phase 2: Development of implementation plan (completed in March 2006);
Phase 3: Testing and refinement of NTMP design and implementation plan;
Phase 4: Implementation and evaluation of the monitoring programme.

The pilot testing and implementation of the design of the NTMP was undertaken in 2008-2009, with the relevant reports being available on the DWA website at http://www.dwaf.gov.za/iwqs/water_quality. The focus of the original design of the NTMP was on toxicity and selected trace organic toxicants. The list of toxicants selected for testing in the implementation phase was the following: Aldrin, Chlordane, DDT and selected breakdown products (DDD and DDE), Dieldrin, Endosulfan, (α-endosulfan, β-endosulfan and endosulfan-sulphate), Endrin, Heptachlor, Hexachlorobenzene, Lindane and selected breakdown products (α-BHC, γ-BHC and δ-BHC), Mirex, Monochrotophos, four PCB congeners (2',5' dichloro-4-hydroxybiphenyl; 2',5' dichloro-3-hydroxybiphenyl; 2',4',6' trichloro-4-hydroxybiphenyl and 2',3',4',5' tetrachloro-4-hydroxybiphenyl), Toxaphene; and three triazines (atrazine, simazine and terbutylazine) (Jooste *et al*. 2008).

The suite of toxicity tests selected for inclusion was as follows (Jooste *et al*. 2008):

- The *Vibrio fischeri* bacterial bioluminescence inhibition test.
- An algal 24-well microplate growth inhibition (AGI) test.
- A *Daphnia pulex* reproduction test (lethality and sub-lethality).
- 96 hour acute *Poecilia reticulata* (Guppy) test.
- Semi-static *Brachydanio rerio* [Zebra fish] development test.
- The recombinant yeast (hER) method.

2.5.5 Ecological Water Resources Monitoring

From an environmental point of view the most useful reporting comes from the National Aquatic Ecosystem Health Monitoring Programme (NAEHMP), specifically the River Health Programme (RHP) and EWR / Ecological Reserve studies. The NAEHMP was instituted as a requirement of the NWA and currently consists only of the River Health Programme (RHP), a joint venture in which provincial nature conservation agencies are responsible for much of the field work and reporting. Coverage is uneven, in that some provincial agencies are under-capacitated and less able to do the work necessary for detailed reporting, while others are producing series of valuable volumes entitled State of the River reports. These reports cover various aspects of the environment, including bio-assessment of water quality. As yet neither wetlands nor estuaries are properly covered, but an estuarine monitoring programme has recently been initiated under the auspices of D: RQS (contact: Gerhard Cilliers). The coastal Elwandle node of the South African Environmental Observation Network (SAEON) has also developed an estuarine database. Phase 1 was the production of a PDF database, i.e. the curation of existing data available in published form, while Phase 2 was the curation of all biotic and abiotic data, including water quality (Day and Scherman 2009).

Integrated Ecological Water Resource Monitoring was borne from a project to develop monitoring guidelines and a Decision Support System (DSS) for Ecological Water Requirements as required for the Ecological Reserve determination. The aim of the project was to provide guidelines to operationalize the Reserve, and was led by Water for Africa (now Rivers 4 Africa) in collaboration with D: RQS. RHP monitoring as such was not included. However, the implications of simultaneously operating two separate ecological monitoring programmes (ERM and RHP) have serious resource implications. To mitigate this and still maintain an operational ecological monitoring programme that provides useful management information, integration of the ERM and RHP within an adaptive management approach was proposed (Kleynhans *et al*. 2009). This forms the basis of the integrated Ecological Water Resource Monitoring (EWRM) approach (DWA 2009).

Due to capacity and resource limitations, all EWRM sites cannot always be monitored at the same time and highest intensity that may be desirable. These restrictions necessitate different levels of monitoring and require the setting of Threshold of Probable Concerns (TPCs) for the different levels of monitoring. It follows that a relatively low intensity monitoring survey will provide TPCs with a larger margin of error than surveys done at a more intensive level. However, the results of TPC assessment from a lower to a higher level of monitoring must be linked. These concepts form the basis of a structured decision-making approach (DWA 2009).

It was recognized that to ensure the effective implementation of EWRM, all efforts must be made to design a programme and methods that are as cost-effective as possible. Developing easy methods to monitor in stream habitat led to the development of the Rapid Habitat Assessment Method (RHAM). This method aims to provide a rapid approach to assess instream habitat conditions in wadeable and to a more limited degree, non-wadeable streams. RHAM data is used to assess habitat suitability for indicator instream biota (fish and macro-invertebrates) and is fundamental in the setting of TPCs that indicate the suitability of the habitat to sustain biota that is considered to be suitable indicators of the EC of the stream. The premise of the RHAM is that suitable habitat conditions will indicate the likely presence, abundance and frequency of occurrence of particular biota. Baseline conditions are used to indicate the change in habitat conditions and the derived impact on the indicator biota. Available data and expert knowledge is used to associate particular habitat conditions with different Ecological Categories (DWA 2009).

The issue of Water Quality monitoring has also frequently been raised. In terms of EWRM, the monitoring of physico-chemical variables is specifically designed to determine whether the ecological objectives, defined by the Ecological Category, EcoSpecs and TPCs, will be met. It must be emphasized that the aim of monitoring physico-chemical variables are to aid in determining causes of biological responses in terms of the Ecological Reserve and not to indicate general water quality problems which are assessed through separate monitoring programs. The physico-chemical monitoring forms part of the EcoClassification system and cannot be seen as an independent monitoring action. The monitoring of physico-chemical variables can also be included in the EWRM at different levels of intensity (see Figure 13).

The monitoring of diatoms has been included to provide a more complete picture of ecological water quality. A Rapid Diatom Riverine Assessment Method (R-DRAM) was developed by Koekemoer (2009) as part of the EWRM as a Level 3 assessment approach within the DSS framework. More detailed diatom assessments are routinely used in Reserve studies. A manual is available for R-DRAM and some training has been provided, but widespread testing of the method is still required.

2.5.6 Other monitoring initiatives

ESKOM is responsible for monitoring acid rain; indications are that it is not yet significant in South Africa (Day and Scherman 2009). Numerous other organizations conduct their own monitoring, specific to their needs, e.g. Umgeni Water, Rand Water, municipalities and water boards, mining companies, Sappi and Mondi.

2.5.7 Conclusion

In summary, geographical coverage of some aspects of water (resource) quality in South Africa is excellent and of others is poor, while reporting is limited except from the National Microbial Monitoring Programme and in the State of the Rivers reports, which cover only some parts of the country. Data are also available from the EWR / Reserve studies, but access to these reports and data are poor (Day and Scherman 2009).

2.6 Environmental water quality in wetlands, estuaries and groundwater

This section of the report provides a summarized overview of the state of environmental water quality methods for wetlands, estuaries and groundwater.

2.6.1 Water quality and Wetlands

Wetland water quality and the Ecological Reserve is summarized in Malan and Day (2005), and then more recently Rountree *et al*. (2013), which covers the development of a method for the Rapid Reserve determination for wetlands, including water quality (in Appendix 7 of the document; Malan *et al*. 2013).

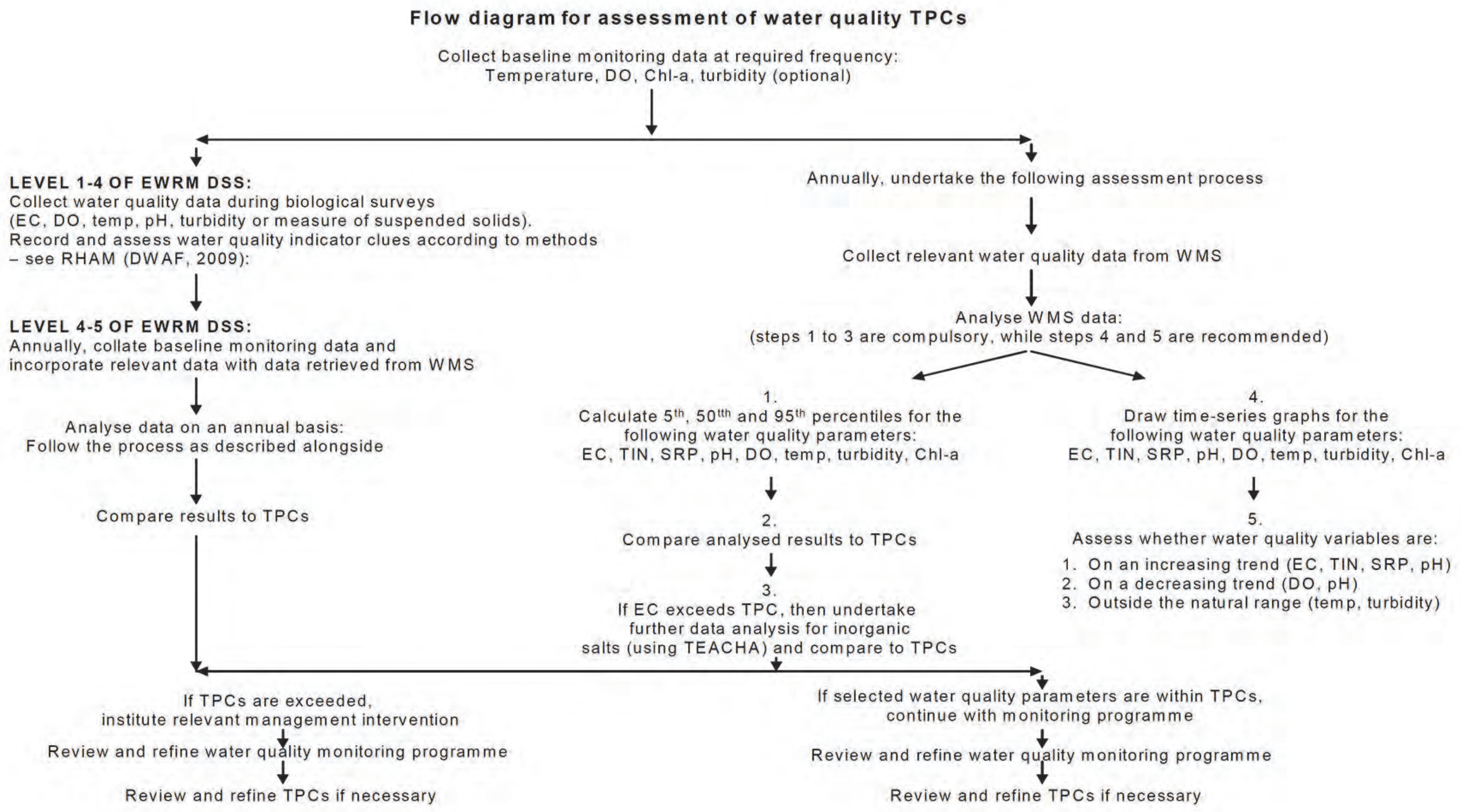

Figure 13 Proposed DSS to be followed for the water quality monitoring programme (Scherman and Muller 2009, DWA 2009).

Wetlands have been a neglected component of aquatic systems, despite the importance of these systems. There has been a large amount of work conducted over the past few years to develop methods for the Reserve and wetlands, with the most recent being the development of Rapid methods, particularly for water quality. Difficulties were experienced due to the nature of wetlands, i.e. they are naturally variable with regard to water quality (both spatially and temporally), particularly in terms of water chemistry, geology and inundation time; and methods need to be linked to the different levels of utilization of the resource. For non-marine influenced wetlands there are four basic wetland water quality types, based on drainage pattern and water source. These are riverine, runoff-fed, groundwater-fed and freshwater lakes. Further sub-divisions cover seeps, springs (or “eyes”) and endorheic wetlands or pans (Malan and Day 2005).

One of the constraints that exist for method development is the paucity of water quality monitoring data for wetlands, in conjunction with limited research that has been carried out to understand the ecological functioning of these systems. Methods also need to be developed specifically for wetlands, e.g. nutrient status of wetlands is very different to that of rivers, meaning that river-specific methods as outlined in DWAF (2008a) cannot be used for wetland assessments. There are also little data on links between quantity and quality requirements. Databases that have been developed to assist in this regard are the Wetlands Water Quality Database and the National Wetlands Inventory. Cognizance needs to be taken of the form of the database housing the National Wetlands Inventory to ensure that the two databases can eventually be merged.

2.6.2 Water quality and Estuaries

Estuarine methods for the Ecological Reserve, including those for water quality, are outlined in the method manuals of DWAF (2004b and 2008b). The purpose of these programmes are to collect data and information to characterise and understand the ecosystem functioning of a specific system so as to be able to determine Resource Directed Measures such as the reference condition, Present Ecological State, Ecological Importance, Ecological Reserve Category and Resource Quality Objectives (RQOs). The following should be noted for the water quality method (Taljaard and van Niekerk 2007):

- The analytical techniques used in the processing of marine and estuarine water quality samples vary greatly from those used in the analysis of freshwater samples. It is therefore crucial that the analyses of water quality samples be conducted by an accredited marine analytical laboratory.
- Estuaries receive water from two sources, i.e. the river and sea, each with distinctively different water quality characteristics, particularly in terms of system variables and nutrients. In turn, the water quality characteristics along the length of an estuary depend on the extent of the influences of each of these sources (governed by hydrodynamic processes), as well as biochemical processes (e.g. organic degradation, eutrophication) taking place at that point within the estuary. The influence of biochemical processes is particularly evident in parts of an estuary where residence time of water becomes longer, often observed along the middle reaches of an estuary during the low flow season. It is therefore also crucial that water in sampled in the two sources, i.e. river and sea.
- River water quality requires longer-term data sets and it is therefore necessary to start such baseline monitoring programmes well in advance (at least five years). For

example, monitoring points at the head of estuaries could be included in the water quality monitoring programme of DWAF.

- At present (i.e. as at 2007) water quality of near-shore waters is not measured on a routine basis along the South African coast, as is the case for some rivers. Because the seawater quality may show strong seasonal variability, particularly along the West Coast, a short-term monitoring survey may not necessarily be representative. In the short term, data on near-shore seawater quality therefore needs to be derived from available data sources, including the South African Water Quality Guidelines for Coastal Marine Waters. Volume 1: Natural Environment (DWAF, 1995; cited in Taljaard and van Niekerk, 2007), until such time as routine water quality monitoring programmes are implemented along the South African coast.
- For toxic substances (e.g. trace metals and hydrocarbons) it is considered more appropriate to sample environmental components that tend to integrate or accumulate change over time, such as sediments. However, these surveys need not be done in all estuaries, only in systems where river water quality or human activities along the banks of the estuary suggest possible contamination (e.g. industrial effluents or storm water run-off from large urban developments).
- For long-term monitoring programmes, water and sediment quality data are particularly important for interpretation of specific biological responses and must therefore be collected by the relevant biotic components as indicated during their sampling surveys.
- Malfunctioning septic tanks, situated in close proximity to the banks of estuaries, may have an influence on water quality in the estuary. However, unlike point-source discharges, e.g. effluents from wastewater treatment works, it is often difficult to quantify the inputs from such diffuse sources. Even so, where septic tanks are known to be a problem or potential problem in a particular estuary, inputs need to be taken into account in the water quality assessments.

Environmental water quality objectives for the estuary component of the Reserve are shown as a series of tables with objectives relating to concentrations to be achieved in the estuary for the following variables (taken from DWAF 2009, the Intermediate Reserve for the Knysna Estuary):

- System variables: temperature, pH, dissolved oxygen and suspended solids/turbidity (transparency).
- Dissolved inorganic nutrients: dissolved inorganic nitrogen, dissolved reactive phosphate, total phosphorus and dissolved reactive silicate.
- Toxic substances.

Sampling takes place at different times for different variables (DWAF 2009):

- Salinity and temperature: longitudinal profiles.
- System variables (turbidity/suspended solids, dissolved oxygen and pH): surface and bottom at high tide).
- Dissolved inorganic nutrients (nitrite, nitrate, ammonia, phosphate and silicate): surface and bottom at high tide).

In addition to Reserve methodologies (which outline monitoring requirements as an output of a Reserve study), the DWA has initiated the National Estuaries Monitoring Programme (NESMP), as required by the National Water Act. The aim is to monitor biotic and abiotic components collaboratively between all role players involved in the management of estuaries.

This will ensure cooperative governance, the prevention of duplication of monitoring effort and the efficient use of public funds. Central to the successful implementation of the programme are the estuary management forums that are currently being established according to the requirements of the Integrated Coastal Management Act. These forums will facilitate implementation and problem-solving, based on monitoring data generated through the programme. The programme is currently being pilot-tested on 18 different estuaries throughout South Africa. Full-scale implementation will proceed in 2015. As at January 2012, monitoring task teams and estuary specific monitoring programmes had been established on the Kosi, Umfolozi, Umlalazi, Breede, Bot, Klein and Berg estuaries (Cillers, D: RQS on http://www.solutionsforwater.org).

Details regarding estuary monitoring can be found in Taljaard and van Niekerk (2007), which define the setting of ecological objectives, the selection of indicators, the significance of scale (i.e. spatial and temporal scales), sampling methods and analytical techniques, and data curation and reporting.

2.6.3 Water quality and Groundwater

Methodology development for groundwater lags behind that of rivers and wetlands, largely because groundwater characteristics are so very different from surface water and that it requires indirect methods of measurement and quantification (DWAF 1999). In the case of the water quality component of the Reserve for groundwater, it is stated that the quality of the aquifer “should not deteriorate and should remain in its ambient state”. The RQOs would therefore be set to prevent deterioration of groundwater quality. In addition, groundwater resources must be protected from point-source and diffuse sources of pollution, saline intrusion and induction of poorer quality from adjoining aquifers. A protective approach has to be taken for groundwater due to their vulnerability to pollution and the long-term impacts should pollution occur (Malan and Day 2005).

Recent developments in terms of policy and methods include the Policy and Strategy for Groundwater Quality Management in South Africa, published by DWA in 2000 and the development of the DWA’s Groundwater Strategy 2010 (DWA 2010c). Some of the action items pertaining to water quality within the 2010 document are the following:

- Review and continue to implement the policy on Groundwater Quality Management in South Africa.
- Finalize the policy and strategy on the rehabilitation of abandoned mines and mine closures, ensuring that all groundwater aspects are addressed.
- Licensing of water use and enforcement of licensing conditions are essential. Water quality aspects should be included in both procedures, with prosecution as required.
- Incidences of groundwater pollution must be resolved as soon as they are detected. The cost of groundwater contamination has been estimated
- Groundwater contributions to the Ecological Reserve need to be determined where possible in all of the water management areas.
- Implement RQOs where Water Resource Classification has been determined for groundwater.
- Protect groundwater quality by establishing areas or “protection zones” around groundwater abstraction points (and sometimes well fields and whole aquifers). Regulations and control over what happens within these protection zones are also critical.

- The single regulatory tool that will adequately protect groundwater, through protection zoning, is through the Water Resource Classification System.

2.7 The development of an environmental water quality research programme to support implementation of the National Water Resource Strategy 2 (2013)

2.7.1 The NWRS 2 and an EWQ research programme

The National Water Resource Strategy 2 was published in 2013. The core of the water resource protection approach lies in six principles to be implemented through ten strategic actions. These are outlined in Table 6 and Table 7 below.

Table 6 Principles from the NWRS2 chapter on Water Resource Protection (http://www.dwaf.gov.za/nwrs/NWRS2013.aspx)

Principle	Principle Title	Description
1	Protection of the resources through classification of the resource with the Reserve as a prior right	The most critical resource protection imperative over the next five years is the use of the gazetted classification process to classify all the major rivers, wetlands and aquifers. This should involve stakeholder engagement to create ownership of water resources. The amount of water available to allocate will be determined after accounting for the Reserve, international obligations and the water requirements for power generation, which is considered a strategic sector.
2	Water resource protection should be based on a participatory approach, involving users, planners and policymakers at all levels	The participatory approach to water resource protection should involve raising awareness of the importance and value of water among policymakers and the general public. It means that decisions are taken at the lowest appropriate level, with full public consultation and involvement of users in the management of our water resources.
3	The value of water resources must be recognised from an economic point of view and the social and environmental benefits of the resource	It is important that society recognises determining the economic value of water, accounting for the use of water (for example, household water supply and irrigation for agriculture) and the ecosystem services provided or supported by water resources (for example, nutrient cycling, habitat provision, and recreation).

Principle	Principle Title	Description
4	Water resource protection must guide setting conditions for water use authorisation	Water resource protection is effected through Resource Directed Measures (RDM), which set the goals for optimising the allocative efficiency of the water resource among its competing demands, and Source Directed Controls (SDC), which set abstraction and discharge licence conditions, financial and economic measures, and other regulatory processes for controlling water use. The potential impacts on the quality of the resource (this includes the quality of all aspects of the water resource, including water quality, the integrity of riparian and instream habitats and aquatic organisms), will be considered when granting a licence in order to ensure that water resources are protected.
5	Incentive based protection of the water resources	To manage the quality of the water resource and protect the ecosystems, the waste discharge charge system must be used as an instrument to improve the quality of the degraded rivers wetlands and aquifers.
6	Integrated protection of aquatic ecosystems	The complex and interconnected nature of catchments as social-ecological systems must be recognised and the aquatic ecosystem (water quantity and quality, habitat and biota) are to be managed in an integrated manner.

Table 7 Strategic Actions from the NWRS2 Chapter on Water Resource Protection (http://www.dwaf.gov.za/nwrs/NWRS2013.aspx)

Strategic Action	Title	Description
1	Manage for sustainability using resource directed measures	A management class, and associated Reserve and resource quality objectives (RQO) have been set and approved for every significant water resources in the country. Resource quality objectives are regularly monitored for compliance, which informs enforcement and a strategic adaptive management cycle. Water ecosystems are maintained in the desired state. ‘The Reserve includes the water quantity and quality needed to maintain aquatic ecosystems in a particular state, as well as the water required to meet basic human needs’.
2	Invest in strategic water source areas	National Strategic Water Source Areas are endorsed and acknowledged as strategic national assets at the highest level in all sectors. They all enjoy legal protection that allows land to be managed in a way that does not significantly undermine their role as key water sources. The costs of catchment management of these areas are factored into the water price, and revenues are reinvested in the management of these areas for their water resources.

Strategic Action	**Title**	**Description**
3	Strategic investment in the maintenance and rehabilitation of water ecosystem	Sufficient financial investment, through the Water Pricing Strategy and the waste discharge charge system must be allocated towards the maintenance and rehabilitation of key identified water ecosystems.
4	Maintain Freshwater Ecosystems Priority areas in good condition	All National Freshwater Ecosystem Priority Areas, which identify priorities for conserving water ecosystems and supporting the sustainable use of water resources, are considered in the determination of Resource Directed Measures.
5	Protect riparian and wetland buffers and critical groundwater recharge areas	Buffers and critical groundwater recharge areas are recognised as critical ecological infrastructure supporting water security and are kept intact, maintained and restored to support water quantity and quality.
6	Rehabilitate strategic water ecosystems to support water quantity and water quality	The priority rehabilitation needs of water ecosystems are identified, and an appropriate level of investment in the rehabilitation of degraded ecosystems is in place to improve the sustainability and performance of key water-related ecological infrastructure.
7	Monitor ecological health to inform management	Sound monitoring (indicators, sites and frequency) is conducted by experienced inter-departmental / agency teams at the scale of WMA's. The resulting information on the state of, and trends in, ecosystem health is packaged and communicated to inform relevant water resource management, decision making and policy processes (see Chapter 13). Dynamic feedbacks between monitoring and research ensure that emerging concerns are investigated and that the monitoring programme(s) remains relevant. Existing monitoring programmes, in varying phases of maturity, serve as a basis for refining, expanding and strengthening the monitoring of the health of water ecosystems. These programmes include the national River Health Programme, Wetland Health Programme and Estuary Health Programme. There is an urgent need to initiate an Aquifer Health Programme to monitor the health and extent of pollution of significant aquifers. Where necessary, programmes are revitalised, expanded and revised so that the location of monitoring sites considers NFEPAs.

Strategic Action	Title	Description
8	Minimisation of pollution from wastewater treatment works	Ensuring that efficient water use is sustainably implemented to reduce the amount of municipal sewage produced (see Chapter 7) should include the development and implementation of wastewater recycling systems to minimize the discharge of sewage into the water resources. Sustainable implementation of Wastewater Risk Abatement Plans, such as the Green Drop certification across all municipal and private wastewater treatment works, should be assessed.
9	Establishing commitment to sustainable water resource management	Water valuation must be integrated into water resource decision-making by ensuring that water resource protection gains acceptance among all stakeholders involved. Current and appropriate valuation methods and processes need to be further developed and refined where the decision context can play a role in addressing water resource protection and enabling communities, business and decision makers to have ownership in the decisions made on water resource protection. This includes stakeholder engagement in creating an enabling environment for appraising decision where trade-offs between use(s) of water and/or services supported by it are evident. The pricing of water (See Chapter 12) needs to better reflect its value.
10	Target actions with immediate benefits	The sector, led by DWA, will identify priority actions that will result in immediate benefits for the country. This includes availing additional water for irrigation use by emerging farmers, on land available for irrigation agriculture, estimated at between 100 000 and 200 00 hectares.

In support of implementing the actions, in line with the principles we recommend a research programme, comprising a set of projects, co-ordinated across the appropriate KSA thrusts.

2.7.2 Proposed EWQ Research Programme

In order to revise, update, and secure sign-off for the water quality methods and implementation approaches for Resource Directed Measures (ecological Reserve determination, setting RQOs and classification) and relevant Source Directed Controls, the following co-ordinated set of projects are proposed:

1. TEACHA update:
   Revise the TEACHA program so that it is usable on a generally accessible platform with DWA WQ data. At present it requires MATLAB which is expensive, technically demanding and therefore not generally accessible.
   This is a necessary first step to update the water quality methods within an ecological Reserve determination. This will require collaboration between DWA RQS and EWQ researchers.

2. Salt toxicity update:
   TEACHA is premised on ion and salt toxicity. There is a need to update the salt toxicity data used in the programme nationally and internationally, and to assess the need for additional eco-toxicity experiments. This would enable an understanding of the eco-toxicity of particular ions that are currently thought to be under- or over-estimated by TEACHA (for example magnesium, potassium and sulphate). This project would then feed into the update of TEACHA (Project 1).
   Project 1 and 2 will result in a validated use of TEACHA.

3. Integration of RDM components
   The application of the methods and procedures for RDM components (ecological Reserve, RQO and Classification) evolved at different times, and attention has not been paid to integrating their premises, or the implications for the resulting practice. Issues include, for example, the up-scaling from resource units (which have RQOs) to the integrated units of analysis (which have management classes and an unclear way of amalgamating, or prioritising the RQO's within an IUA). There are no guidelines for such prioritisation which would need to take into account a consideration of the role of refugia, and the possibility decisions not to rehabilitate.
   This project should include a set of national workshops to canvass practice-based experience, and must include the active participation of the DWQ RDM Chief Directorate.

4. Integrating Water quality and quantity
   There has been a long standing call for the integration of water quality and flow in RDM (and SDC) processes. Currently there is research on the development of user-friendly water quality/quantity models. These need to be fast-tracked into RDM processes and into robust, transparent, meaningful stakeholder participation in during the RDM processes.

5. RDM participatory processes
   Participatory processes are important and challenging. There needs to be research-based evaluation of current participatory process. This can draw on the WRC initiative for social science in water research, and will result in more theoretically supported participatory processes that are integrated, transparent and robust. Current understandings of complex social-ecological systems and ecosystem services will be included in this. The concept of ecosystem services and benefits can act to mediate social and ecological values.

6. Integrating RDM and SDC
   To date there is little connection in ensuring coherent links between EWQ RDM and SDC measures. (UCEWQ has an MSc student who has recently completed the first empirical study to link waste water treatment works with licence requirements and green drop performance with instream river health – a useful starting point). This task needs to be tackled at a range of levels through policy, legislation, governance, and practice – supported by research. The project will include all work to date on the Waste Discharge Charge System.

7. Integration into the Towards a New Paradigm for IWRM process
   If these projects are undertaken with and understanding of complex social ecological systems, in an integrated way, taking a systems approach – that will support and feed into and support the TNP project (proposal 1003122).

The manner in which these proposed research thrusts might support implementation of NWRS2 is outlined in Table 8 and Table 9 below.

Table 8 NWRS2 (DWA 2013c) Principles tabulated against the proposed EWQ projects (listed above), with an explanation of their contribution to implementation. Projects that offer necessary support are shown in parentheses.

Principle Number	EWQ project Number	Research contribution to implementation
1	(1,2) and 3	Resource classification draws on knowledge about ecological and societal water needs, quantified and described through Reserve determinations and RQOs. This project develops a sound basis for all the RDM components to work together integratively. (Water quality components of ecological Reserve determinations and RQOS depend on Projects 1 and 2 – which support Project 3.) These three projects underpin the clear, justifiable EWQ RQO's necessary for transparent stakeholder participation (Principle 2).
2	(1-4), 5 and 6	The concept of "full public participation" needs to be clarified for stakeholders to have reasonable expectations and to engage in processes that are informed, robust and transparent (Project 5). This depends on reliable defensible RQOs (Projects 1-3), efficient SDCs, and the ability to understand water quality and quantity (Project 4) in an integrated way. Substantive public participation needs to include an understanding of the link RDM and SDC (Project 6)
3	5 and 6	Ecosystem services and benefits, and the waste discharge charge system can both be used to bridge social and ecological values. Projects 5 and 6 develop a practical link between bio-physical RQOS and users' needs.
4	6	This project will provide clear explicit steps to link RDM and SDC processes. Ideally a guide to practice will emerge from engaged case-study research.
5	6	The waste discharge charge system, as a SDC will be clearly and practically linked to RDM and other SDC processes
6	6 and 7	Integration is the key challenge. Each of these projects fills a current gap in the EWQ management system and Projects 6 and 7 work together bringing social-ecological components together systemically.

Table 9 NWRS2 (DWA 2013c) strategic actions tabulated against the proposed EWQ projects (listed above), with an explanation of their contribution to implementation. Projects that offer necessary support are shown in parentheses.

Strategic Action Number	EWQ project Number	Research contribution to implementation
1	(1-3), 4-7	Projects 1-3 are necessary support steps enabling Projects 4 -7 focus clearly on how to practically USE the RDM steps to achieve effective resource protection.
2	(1-5), 6 and 7	Water users invest in water resources through the level of responsibility of their use. These users rely on the clarity of information shared and the opportunities to share their knowledge – robust stakeholder processes as well as specific tools such as ecosystem services and benefits, and the waste discharge charge system underpin water resource investment.

Strategic Action Number	EWQ project Number	Research contribution to implementation
3	1-7	Strategic investment requires an understanding of value and risk. The costs of rehabilitation need to be linked to benefits of functional systems. The criteria defining functionality are the RQOs. Therefore the whole RDM-SDC system should encourage strategic investment.
4 & 5	(1-5), 6 and 7	Priority areas, riparian strips and groundwater systems are embedded in connected landscapes. We can only protect them when we engage in the appropriate protection of whole systems. This needs an understanding key dependencies and linkages. The basics of determining the Reserve with specified RQOs, and the selection of management classes, backed up by controls and enforcement, are the mechanisms for protection
6	1-7	Strategic Action 3 and 6 are closely linked and the same argument applies.
7		Monitoring is intrinsic to all of the data required for RDM and SDC procedures. RQOs in particular are directly derived from monitored data. Monitoring enables or constrains the accuracy of both setting goals (RDM) or measuring compliance with SDC. None of the project deals explicitly with monitoring, but all depend on it.
8	1-7	WWTW waste minimisation will be supported by stronger EWQ management systems for RDM and SDC. The programme explicitly builds EWQ management strength.
9	1-7	Commitment comes with recognition of value. This EWQ research programme is based on the concept of engaged, transformative research which acts to develop the values of equity and sustainability, the premise of the NWA and the NWRS2.
10	3-7	An outcome of each project must be specific actions that each contribute to social and ecological wellbeing.

The NWRS2 includes a set of key strategic objectives for water resource protection:

- Ensure sustainable management of the water resources through resource directed measure and source directed controls.
- Protect and maintain existing freshwater ecosystem priority areas in good condition and well-functioning water resource ecosystems by managing riparian and wetland buffers and critical groundwater recharge areas.
- Carry out rehabilitation of strategic water ecosystems.
- Ensure prevention of water resources from point source and non-point source pollution by managing at source.
- Create awareness among communities, business and decision makers about the value of water and ensure commitment to sustainable water use practices.
- Create an enabling environment for water resource protection through incentive based approach to water resource management.
- Monitor the ecological health of our resources through an integrated information management system.

The benchmark of the EWQ programme success will be the contribution the programme makes to the achievement of the objectives.

3 LONG TERM TRENDS IN ENVIRONMENTAL WATER QUALITY IN OLIFANTS RIVER AND CROCODILE RIVER CATCHMENTS

3.1 Introduction

3.1.1 Water quality in South Africa

South Africa's revision of water law after 1994 was recognised for changes that geared legal and regulatory frameworks towards sustainable environmental and social outcomes (Godden 2005). Despite this focus on sustainable management, the quality of water in the country has been recognised as decreasing (CSIR 2010, DWA 2011a) and the challenges involved in management of water quality in the country recognised (Bohensky 2008, Quinn 2012). A delay often appears between proposal or acceptance of a management approach or structure and its implementation. Delays have been attributed to difficulties in slow institutional reform, as well as difficulties in meeting goals of management practices and in institutional and stakeholder cooperation and coordination, and a disjunction between scientific and management communities (Dickens 2007, Cobbing 2008, Pollard and du Toit 2010, Quinn 2012).

While this paints a generally bleak view of the state of water quality management in the country, it is important that management processes are reviewed in the light of water quality changes to assess what has worked, what has no, and, ideally, why this might be so.

The analysis presented here is an assessment of changes in water quality to support an assessment of water quality management practices While the overall focus of this project is national, the research process will make use of case study or focus areas in order that detailed investigations of the links between management practices and tools and environmental water quality can be undertaken. Case study areas and justifications for these follow:

The Crocodile River in the Inkomati catchment: This river is selected as it has been identified as water-stressed in the past, but has recently received management intervention from a CMA recognized as engaged and enthusiastic (Pollard and du Toit 2010). Water in the Crocodile River contributes to the flow of the Inkomati River, and South Africa has transboundary management agreements with Mozambique regarding the quantity and quality of water passing the border.

The Olifants River in the Limpopo catchment: The water quality in this river has been identified as impacted (Ashton and Dabrowski 2011). A functional CMA has not been established. The Olifants River is also part of a transboundary agreement with Mozambique regarding shared water resource management.

3.1.2 Study areas

Brief outlines of the study catchments follow. More details on site specific impacts are presented in the results and discussion. For more details on either catchment, please consult the references cited.

The Crocodile River catchment

This catchment is located entirely within Mpumalanga province in South Africa. The river flows in a generally easterly direction until it passes over the border to Mozambique and joins the Komati River. Much of the catchment, particularly the upper catchment, is dominated by afforestation and commercial and emerging agriculture (DWA 2011a, ICMA 2011). The catchment has a subtropical climate, with predominantly summer rainfall. As the river passes along the southern boundary of the Kruger National Park, it has considerable ecological importance (ICMA 2011). Relatively few dams are present in the catchment.

Noted impacts in the catchment include the conurbation around Nelspruit, a pulp and paper mill and a ferrochrome smelter along the Elands River, a sugar mill along the lower Crocodile River, mining in the Kaap and Queens Rivers, and diffuse return flows from intensive irrigated agriculture (DWAF 2004c, ICMA 2011, DWA 2011a). Agriculture accounts for 59% of water requirements (ICMA 2011). The impact of alien invasive plants on the catchment is also of concern (DWAF 2004c). Water quality in the catchment was “notably good” in the past (DWAF 2004c), but may be degrading (DWA 2011a), with areas in the elands River, the lower Kaap River, and the lower Crocodile River being of concern (ICMA 2011).

South Africa has a signed agreement with Mozambique regarding the quantity and quality of waters passing the border (TPTC 2002). Standards for water quality related to this have been promulgated.

The Olifants River catchment

The Olifants River catchment lies within the provinces of Gauteng in the western reaches, Mpumalanga in the southern reaches, and Limpopo in the north. The river starts flowing northwards, but on reaching Limpopo curves to the east, from where it flows through the Kruger National Park and thence to the Mozambique border. Once in Mozambique it joins the Limpopo River before flowing to the Indian Ocean.

Major land uses in the catchment include mining (coal, platinum, vanadium, chrome, copper, phosphate), coal-fired power generation, industry, irrigated, emerging and subsistence agriculture, livestock farming and conservation (DWAF 2004d, Ashton and Dabrowski 2011, DWA 2011a). The climate ranges from cool highveld to subtropical, with predominantly summer rainfall and strong seasonality in flows (Ashton and Dabrowski 2011).

Like the Crocodile River, the Olifants River has ecological importance as it passes through Kruger National Park, and the catchment includes many other game reserves or conservation or tourism areas (DWA 2011a). The catchment is heavily dammed (Heath *et al*. 2010).

Water quality in the catchment is known to be impacted. In brief, major impacts include salinization, sulphate contamination, pH changes with high levels at many sites but occasional acidified stretches, increasing nutrient levels leading to mesotrophic or eutrophic conditions, and elevated levels of heavy metals and potentially of organic toxins (DWAF 2004d, DWA 2011a). Particular impacts are a largely a function of mining and associated industries (especially coal mining in the upper catchment, and phosphate mining at Phalaborwa), irrigated agriculture, and human settlement.

South Africa has an agreement with Mozambique regarding management of water quality and quantity passing the border, although standards are not yet established (LIMCOM 2003).

3.1.3 Rationale

Although the Olifants River catchment is relatively well studied (e.g. Hohls *et al*. 2002, DWAF 2004d, Heath *et al*. 2010, Ashton and Dabrowski 2011, DWA 2011a, van Veelen and Dhemba 2011) and information on trends in water quality in this catchment is available, there is a relative dearth of data, and particularly trends in data, from the Crocodile River catchment (but see Hohls *et al*. 2002, DWAF 2004c, DWA 2011a). As such, it proved more practical for the purposes of this analysis to undertake a full trend analysis for long term data for both catchments. This ensured that maximal use was made of available data and that the same parameters were assessed in both catchments.

The aim is to identify trends in environmental water quality that might be linked to management practises at a national or local scale in both catchments. Although consideration will be given to, for example, the state of the resource, and consideration as to possible drivers of water quality change, the primary focus of the module is to identify and assess patterns of change in water quality.

3.2 Methodology

3.2.1 Approach

The issue of what may loosely be termed water quality has in management guidelines commonly been phrased in terms of utility to users. In this sense, water quality is generally reflected in a number of relations between what are often fairly easily measured water quality parameters, for example water hardness, and their effects on particular user groups, for example, production of scale on pipes, or effects on soil chemistry. Some of these relations are well defined and understood. When the user is the environment, the complexity of the relation and of interactions between of driving variables on biotic response is less clear, and it is desirable to include direct measures of biotic response, and, ideally, to develop a large dataset of these to better understand the relations and how these might be mediated by other parameters. This philosophy underlies the use of biotic and physicochemical together for environmental water quality management.

In assessing the data available for analysis in this report, it becomes apparent that the mismatch between data available on drivers (and in particular major salts, major plant nutrients, pH and temperature) and biotic response is vast. Physicochemical records from monitoring points in the two catchments date from the 1960’s, and are better established since the 1970’s, while biotic data are relatively recent and sparse. For the purpose of undertaking a fairly long term trend analysis, explicit inclusion of biotic data was therefore abandoned.

3.2.2 Data availability

Data for the all analyses presented here were sourced from the Department of Water Affair’s Water Management System in April 2013. Full data sets of all available physicochemical parameters over as long a period as were available in the Olifants and Crocodile River catchments were received from the Department of Water Affairs (DWA) Directorate of Resource Quality Services (RQS). These data were transferred to a database for processing prior to analysis.

3.2.3 Data pre-processing

Records were combined into a single record per day per monitoring point, largely to combine data collected or measured using different methods and stored separately in the source data. Multiple records of any one parameter were rare, and when encountered these were combined by taking a median of available data. Only data collected from rivers was assessed, and records from reservoirs, canals, and waste and potable water treatment works were discarded.

Left-censored data (data less than the detection limit) were replaced with a value of half the detection limit following DWAF (2008a). This approach to dealing with left censored data has been criticized (Helsel, 2006), but it underlies the proposed methodology for the water quality component of the ecological Reserve and is in common application elsewhere, and has been found to perform capably in the derivation of summary statistics when censoring levels are not high, as in the data assessed here (Antweiler and Taylor, 2008).

The prevalence of right-censored data was also assessed. Right-censored data were extremely rare (e.g. 0.5% of all data in the Crocodile catchment), only occurred in microbial counts and then were almost completely restricted to total coliform counts (which were not assessed in the analyses presented here). As a result, no correction for right-censored data was undertaken.

3.2.4 Parameters and indices assessed

The indices selected for assessment are for the most part widely used to assess impacts of eutrophication, salinization (industrial, agricultural and otherwise) and microbial pollution. Index selection was also guided so as to include indices of importance to a range of water users including industry, agriculture and the environment. Index and water quality parameter selection was constrained by data availability and only those measures that had sufficient data in the underlying data set were selected.

- Orthophosphate and total inorganic nitrogen levels: Orthophosphate is simple the quantity of dissolved phosphate in solution, and total inorganic nitrogen is the combined quantity of dissolved nitrate, nitrate and ammonium in solution. These were selected as indices of eutrophication or the potential for eutrophication. Insufficient data on chlorophyll levels negated their application as a more direct indicator of algal growth in this regard.

- Electrical conductivity: This represents the capacity of ions in solution to conduct electricity and as such is a proxy for ionic levels and thereby for total dissolved solids. The data supplied included a field for TDS data, but this is currently a calculated field, though older data was measured (Sebastian Jooste and Esna Portwig, pers. comm.), and changes in results, if any, as a result of methodological changes were not known. Use of electrical conductivity rather than TDS data follows van Niekerk *et al*. (2009).

- pH: This is a major and commonly assessed water quality parameter. Changes in pH are associated with the impact of acid mine drainage.

- Chemical weathering index: This index was proposed by Huizenga *et al*. (2013) and reflects the extent to which chemical weathering contributes to relative ionic levels in rivers. The index indicates the relative extent to which combined carbonate and bicarbonate levels (as derived from total alkalinity levels in data) contribute to dissolved ions levels in water.

- Sulphate contamination index: This is another index proposed by Huizenga *et al*. (2013) that reflects the relative extent to which sulphate ion enrichment contributes to relative ion levels in rivers. The index can be used to assess increases in sulphate levels owing to anthropogenic activity, in particular acid mine drainage, air pollution leading to elevated atmospheric sulphate and the use of gypsum-containing fertilisers (Huizenga, 2011).

- Chloride salinization index: This is the third water characterization index proposed by Huizenga *et al*. (2013). It reflects the relative extent to which chloride ion enrichment contributes to relative ion levels in rivers. Elevated chloride levels are commonly associated with secondary salinization owing to irrigation, the removal of natural vegetation, and marine impacts on freshwater systems (Huizenga, 2011).

- Adjusted sodium adsorption ratio: This is a widely used index that evaluates the sodium hazard associated with a use of water in irrigation. The broad definition of the ratio is the square root of the ratio of sodium concentration over the square root of the combined concentration of calcium and magnesium. Derivation of this index followed the method of Lesch and Suarez (2009).

- Corrosion potential ratio: This index was used by Ashton and Dabrowski (2011) in a previous survey of water quality in the Olifants River. It acts as an indicator of the likelihood of corrosion of metal pipes and fittings, and is calculated by assessing the combined ionic activity of chloride and sulphate over alkalinity.

- Un-ionized ammonia levels: We aimed to include a measure of water toxicity in this survey, but neither catchment had sufficient toxin data for trend analysis. In order that some indication of toxicity was included, we followed DWA (2011a) and included un-ionized ammonia as a toxicity indicator. As raw data on un-ionized ammonia were insufficient, un-ionized ammonia levels were derived from ammonium levels and pH following the method recommended for the water quality component of the ecological Reserve (DWAF, 2008a).

- *Escherichia coli* counts: These were included as an indicator of microbial pollution in riverine water. Several indicators of microbial pollution were present in the data; *E. coli* was selected as it had the most records and had little right-censored data. It should be noted that the time frame of this data differs from other data assessed here (the earliest records from the Crocodile River catchment were from 2004, and in the Olifants catchment from 2008). The monitoring points assessed also differ.

3.2.5 Monitoring point selection

Monitoring points for assessment of water quality were selected using number of criteria as follows:

- Sites should have sufficient data for trend analysis. Generally 500 points or more were deemed sufficient. Sites should also have at least 100 points that passed data quality checks (see 3.2.7 below). Exceptions were made where sites were particularly important for analysis on the basis of other selection criteria.
- All designated environmental water requirement sites should be represented.

- Sites should be spaced along the main stem of the river and should be placed low on each major tributary to assess water inflows to the main river stem.
- A suitable upstream reference site(s) should be included.
- Sites other than the above but with sufficient data and clearly spaced from other sites were included where it seemed appropriate.

The location of selected physicochemical water quality monitoring points is shown in Figure 14 and Figure 15.

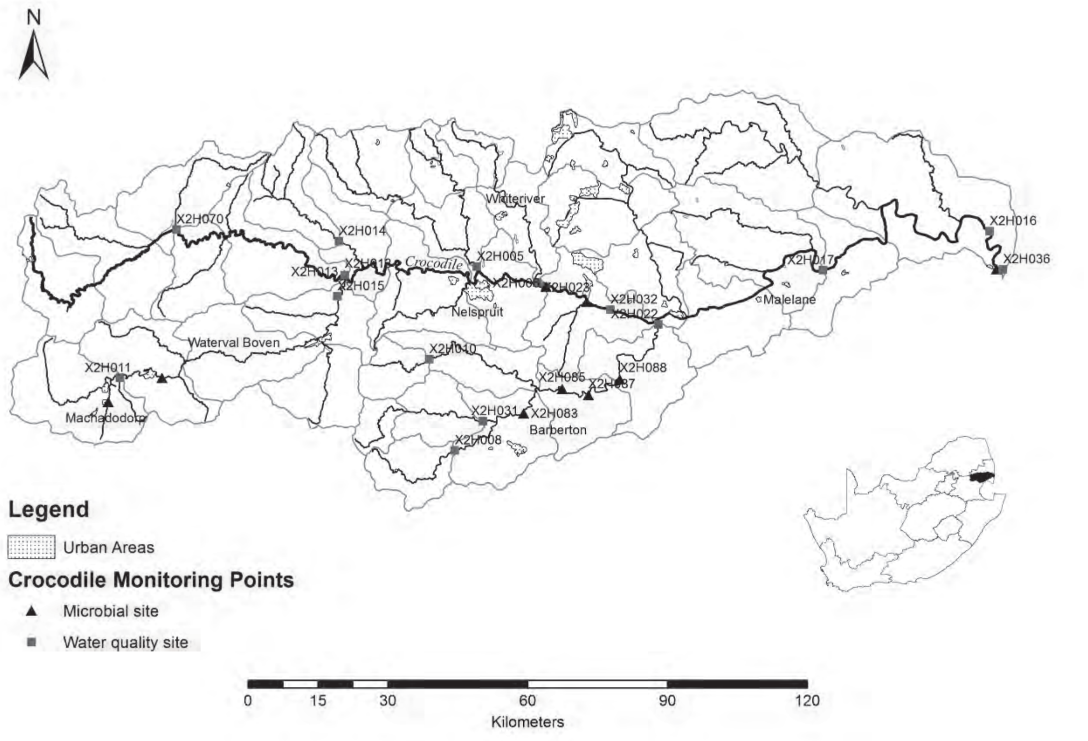

Figure 14 Sites selected for assessment of physicochemical and microbial water quality in the Crocodile River catchment.

Sites for assessment of microbial pollution were heavily data-limited, and after assessment of the amount of data available per site across both catchments, site selection was finally driven by data availability. This decision was undertaken with the hope that trends with time might be detected. The spatial distribution of sites is as a consequence uneven and does not comply to site selection criteria listed above.

In many cases sites for microbial monitoring were found to be located immediately downstream of wastewater treatment works discharge, and as a result the microbial load at these sites cannot be taken as representative of overall river conditions. Despite this, such sites were selected for analysis in some cases owing to their superior data record.

The location of sites selected for microbial trend analysis are presented in Figure 14 and Figure 15.

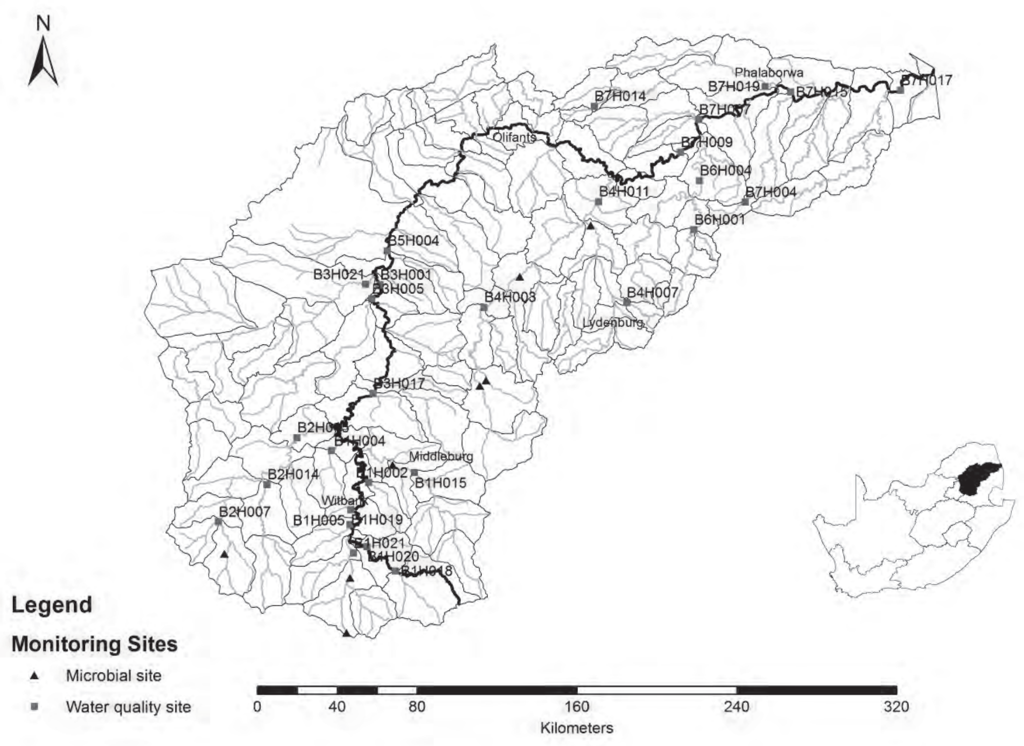

Figure 15 Sites selected for assessment of physicochemical and microbial water quality in the Olifants River catchment.

3.2.6 Guidelines/RQOs adopted

The guidelines adopted for this analysis are largely drawn from DWA generic RQOs applied by DWA in their recent review of national water quality (DWA, 2011a). Guidelines are presented below in Table 10. Where these differ from DWA generic RQOs, the sources are indicated in Table 10, and the rationale is presented below.

Aston and Dabrowski (2011), in selecting guidelines for their survey of surface water quality in the Olifants River catchment, used the target water quality range and chronic effect values ranges of various compounds in deriving their guidelines, and these were used for the current analysis. The guidelines for total inorganic nitrogen were taken from the ecological target water quality range and chronic effect value range, and the upper bound of tolerable was taken from the upper bound of the chronic effect value range for domestic use for nitrate. The values adopted for the corrosion potential ratio boundaries derive from the upper bound of the target water quality range and the upper bound of the chronic effect value range for domestic use.

Electrical conductivity boundaries were derived from DWA (2011a) total dissolved solids boundaries, converted to mS/m using the commonly applied factor of 6.4.

Ideal and acceptable bounds for *E. coli* counts were derived from the upper bounds for slight risk and significant risk posed by faecal coliforms in water for domestic use (DWAF 1996a). As *E coli* can make up 97% of faecal coliforms in human faeces (DWAF 1996a), it was deemed appropriate to use boundaries defined for faecal coliforms in deriving guidelines for *E. coli* in this analysis. The upper bound of tolerable water quality is derived from the upper bound for faecal coliforms of the significant risk class for watering of young livestock (DWAF 1996d).

Table 10 Resource quality objectives or guidelines adopted for this analysis. Where they are not taken directly from referenced source, they are marked with asterisks and the rationale behind the measures chosen explained in the text.

Water quality parameter	Units	Bound	Ideal	Acceptable	Tolerable	Source
Orthophosphate	mg P/ℓ	Upper	0.005	0.015	0.025	DWA (2011a)
Total inorganic nitrogen	mg N/ℓ	Upper	0.5	2.5	10	* Ashton & Dabrowski (2011)
Un-ionized ammonia	mg N/ℓ	Upper	0.015	0.044	0.073	DWA (2011a)
Electrical conductivity	mS/m	Upper	31	55	125	* DWA (2011a) TDS RQOs
pH		Upper	≤8	<8.4		DWA (2011a)
		Lower	≥6.5	>8		DWA (2011a)
Sodium adsorption ratio		Upper	2	8	15	DWA (2011a)
Corrosion potential ratio		Upper	0.4	1.0		* Ashton & Dabrowski (2011)
E. coli counts	cfu/ 100mℓ	Upper	20	200	1000	* DWAF (1996a & d)

3.2.7 Data quality

Major salt data were assessed for quality using the approach of Huizenga *et al*. (2013). This approach uses tests applied to water quality data from the Olifants River by Ashton and Dabrowski (2011) along with other quality control measures in an attempt to identify records containing unreliable data. The calculation template supplied in support of Huizenga *et al*. (2013) needed some modification before it could be applied. The modified inorganic water quality check uses information on record completeness, stoichiometric charge balance (following Appelo and Postma 2005), and a comparison of measured and calculated electrical conductivity (following McCleskey *et al*. 2012).

Comparisons based on measured versus calculated total dissolved solids were abandoned owing to the impracticality of DWAs calculated TDS with another calculated using a (presumably) different approach (see 3.2.4 above). Application of the stoichiometric charge balance in quality control has been critiqued by Murray and Wade (1999) (though a different carbonate/bicarbonate derivation technique underlay their critique). Another difficulty inherent in using quality control measures based on major salt concentrations is that acceptance or rejection of a record based on these criteria may be appropriate for salt data, but not necessarily for other data, most notably microbial data where sampling and analysis are quite different.

Application of data filtering based on these quality control systems considerably reduced the amount of data available for trend analysis, and thereby decreased the power of the trend analysis models. Because of this, and the concerns and difficulties applying the methods, data filtering was not applied prior to analysis, and the assumption was made that possible flaws in the data were randomly distributed and therefore did not introduce systematic bias to analyses. The results of the filtering process are nevertheless presented as an indication of salt data quality.

3.2.8 Multivariate data trend assessment

A basic principal components analysis (e.g. see Mardia *et al.*, 1979) was undertaken on datasets containing all water quality parameters underlying the indices assessed here at all selected monitoring points. Owing to data restrictions, no microbial data were used in this analysis. All PCA analysis using data that were zero-centered, scaled and rotated.

This analysis aimed to briefly determine the level of covariance between a range of water quality parameters with the potential outcome of identifying water quality "syndromes" characterised by a number of co-occurring water quality parameters. This approach would also determine the potential for dimensionality reduction in the selected water quality parameters which itself would suggest particular patterns of water quality change in a catchment.

3.2.9 Trend analysis

The approach taken in trend analysis was informed by the aims of this component of the project, namely to identify temporal trends in water quality in the two model catchments. The key part of this was not to identify where water quality had decreased, but to determine whether significant changes had taken place with time at the identified monitoring points and what the nature of these trends might be.

The initial approach had been to undertake a time-series analysis of water quality parameters at identified monitoring sites. However, assessment of data revealed that the spread of data across with time was never regular, that monitoring frequencies varied, and that seasonality in the various water quality parameters was common and often pronounced. Given that most time series analysis relies on regular, repeated measurements and is compromised in their absence, an alternate approach was identified that could identify complex trends without overfitting of data, that could separate seasonal variability from longer term trends, that would account for temporal autocorrelation, and that could use data that were not smoothly distributed temporally.

The approach adopted used a general additive mixed model (GAMM) to model water quality parameter change over time (Wood 2011). GAMMs have been found appropriate for modelling nonlinear temporal trends with seasonal components (Polansky and Robbins 2013). Temporal trends were modelled as a combination of temporal change (as year) using a penalized thin plate regression spline (Wahba 1990, Wood 2003), with seasonal changes modelled as a separate term using a cubic spline. Lag effects were accounted using an AR1 autoregressive model on elapsed time. Tests on a number of datasets found that a normal distribution of residuals was appropriate and this was applied throughout. Given the number of analyses undertaken, this general model was applied throughout all trend analysis (except where fewer than 100 data points were available when no trend analysis was undertaken) and no model refinement was undertaken. Although model refinement would be necessary to accurately model responses, application of the general model was deemed sufficient for the purpose of trend identification.

Trend models are presented throughout the text as model predictions overplotted in scatterplots showing the raw data. The significance of the long term trends, but not the seasonal trends, is presented together with the graphs.

3.2.10 Toxin analysis

Insufficient data were available to undertake a trend analysis on the vast majority of potential toxins, with the exception of un-ionized ammonia. Examples include levels of arsenic, cadmium and zinc noted as potentially problematic in the Crocodile catchment (DWA 2010d, Palmer *et al*. 2013), and levels of pesticides and heavy metals in the Olifants catchment (Heath and Claassen 1999, Ansara-Ross *et al*. 2012).

In order to determine whether the potential for toxic effects might exist in the catchment, 5th, 50th (median) and 95th percentiles for toxins were derived for available data for comparison with standards. As this project deals specifically with environmental water quality, standards were taken from the South African water quality guidelines for aquatic ecosystems (1996e) by preference. If no guideline for aquatic ecosystems was available, standards for domestic use, livestock watering or irrigation use were selected (1996a, c & d). Generally, the most sensitive guideline from these was selected.

3.2.11 Statistics and software

All statistical analyses undertaken used R 3.0.1 (R Core Team, 2013). Trend analysis used packages mgcv (Wood, 2011) and nlme (Pinheiro *et al.*, 2013). Graphics were produced with the ggplot2 (Wickham, 2009) and gridExtra (Auguie, 2012) packages.

3.2.12 Disclaimer

It needs to be emphasized that the analyses presented in this report are based only on data sourced from DWA, with no attempts at ground-truthing of data or trends, or cross-referencing with other data sets. It is also important to note that monitoring points were selected to provide a spatial spread, with some distance (and many potential impacts) between data points. As such, the results presented here are presented as trends and no categorical and absolute identification of particular impacts is possible. Where likely impacts are identified in the text, this may indicate that some effect is likely. It does not indicate a proven link between the proposed impact and the trends identified in this report.

3.3 Results

3.3.1 Crocodile River

Data quality

Although selected monitoring points in the Crocodile data had large datasets associated with them, data quality checks reveal issues with the vast majority of these (Table 11). If strict filtering was applied, the amount of data available from this catchment is limited. The best of the selected sites was X2H036 at Komatipoort, on the Komati just after the Crocodile River has joined the Komati, where 70% of records pass all checks. However, there are a large number of sites where only 20%, or one record in five, passes, and the worst of the selected sites is one the Crocodile River at Montrose (X2H013) where only 16% of records pass the quality control procedures.

Table 11 The number of records available for analysis at monitoring points in the Crocodile catchment (following initial data processing), and the number and proportion of these that remained after data quality control checks had been applied.

Monitoring Point Name	All records	Filtered records	
X2H005 Nels River at Boschrand	751	163	22%
X2H006 at Karino on Crocodile	663	189	29%
X2H008 Queens River at Sassenheim	534	173	32%
X2H010 North Kaap River at Bellevue	464	116	25%
X2H011 at Geluk on Elandsrivier	656	267	41%
X2H013 Crocodile River at Montrose	1149	185	16%
X2H014 Houtbosloop Spruit at Sudwalaskraal	573	122	21%
X2H015 at Lindenau on Elandsrivier	1289	490	38%
X2H016 at Ten Bosch/Kruger National Park on Crocodile River	1417	980	69%
X2H017 Crocodile River at Thankerton/Kruger National Park	1054	657	62%
X2H022 Kaap River at Dolton	995	454	46%
X2H023 Wit River at Goede Hoop	375	110	29%
X2H031 South Kaap River at Bornmans Drift	528	130	25%
X2H032 Crocodile River at Weltevrede	1220	322	26%
X2H036 at Komatipoort/Kruger National Park on Komati River	746	521	70%
X2H070 Kwena Dam on Crocodile River: downstream weir	286	90	31%

Overview of major multivariate trends

The PCA biplot of water quality parameters from selected monitoring points in the Crocodile River catchment is presented in Figure 16 below. It can be seen from this that changes in major salts generally occur together, and that increases in salinity are accompanied by changes in all major salts, with the possible exception of potassium. Changes in salinity are also accompanied by changes in pH and in alkalinity.

Changes in major nutrient parameters are largely independent of changes in salts. This indicates that salinity, pH and alkalinity changes across the catchment are largely independent of nutrient changes.

The first principal component explains 56% of the variation in the data, which suggests that the majority of variation in water quality in this catchment is due to changes in major salts and pH.

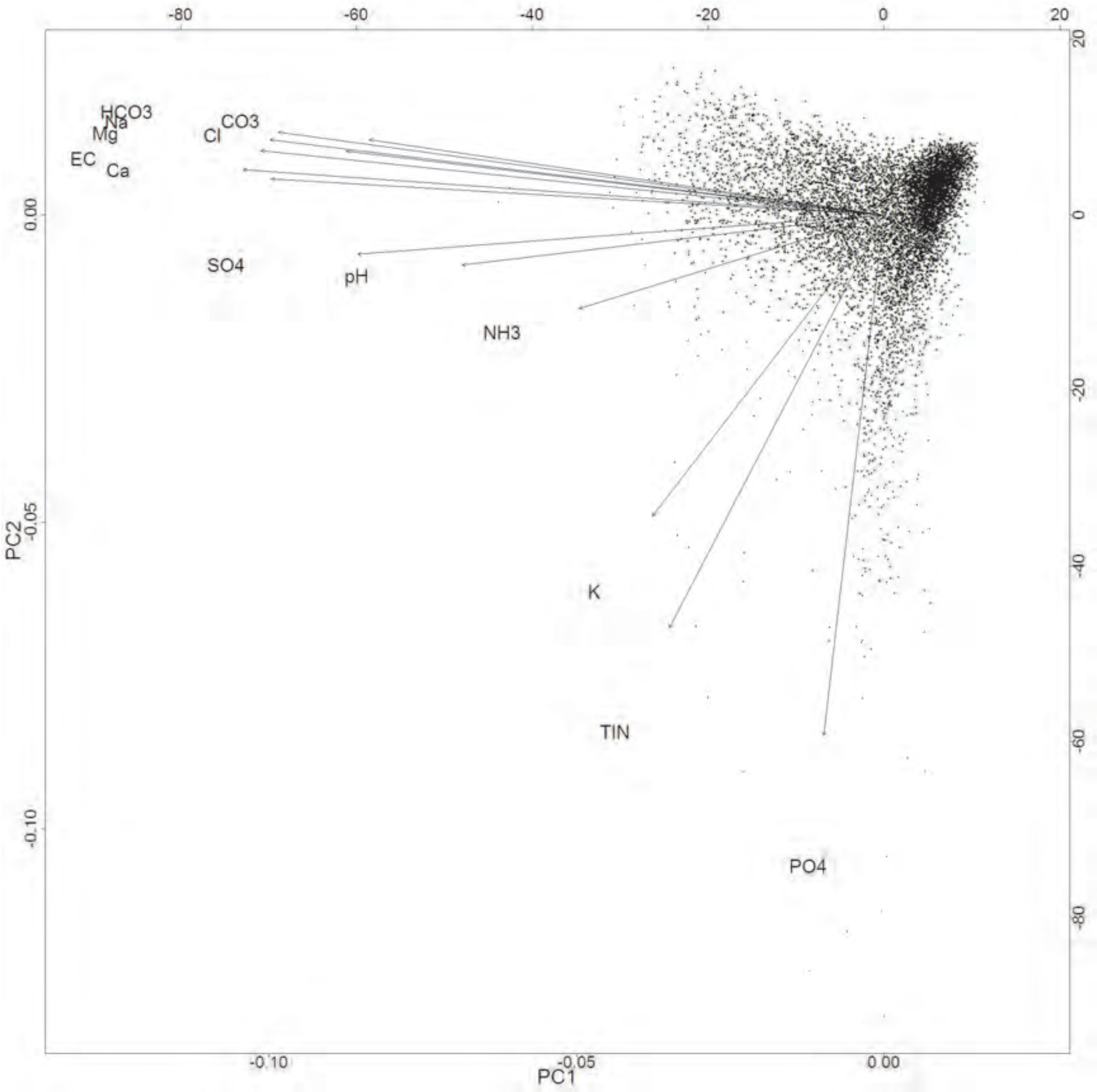

Figure 16 PCA biplot of the first two principal components from the PCA analysis of water quality parameters in the Crocodile River catchment. The first two principal components together explain 67% of the variation in the data.

Spatial overview of water quality

This section assesses major spatial trends throughout the catchment without explicit consideration of a temporal aspect. Although temporal trends will be mentioned at times, they do not enter into major consideration of spatial patterns revealed in the figures presented here. As a result, the information presented here may help identify spatial patterns of water quality change, but the absence of temporal variation mean that temporal analyses at various monitoring points will need to be consulted before one can conclude that the changes indicated by figures presented here represent the current state of the catchment.

Microbial spatial trends will not be analyzed here owing to the uneven distribution of microbial monitoring sites.

Orthophosphate

Spatial trends on orthophosphate levels in the Crocodile River catchment are presented in Figure 17. Although no sites have ideal orthophosphate levels, it can be seen that the majority of sites have median orthophosphate level at a tolerable or better state. The two exceptions to this are the sites on the Crocodile at Karino (X2H006) and at Weltevrede (X2H032). Looking at the temporal trends at these sites, both show a steady increase in orthophosphate levels with the highest levels recorded recently. The two sites are situated just below Nelspruit and KaNyamazane, and, as these orthophosphate levels are associated with high level of *E. coli*, may be caused by wastewater treatment works effluent.

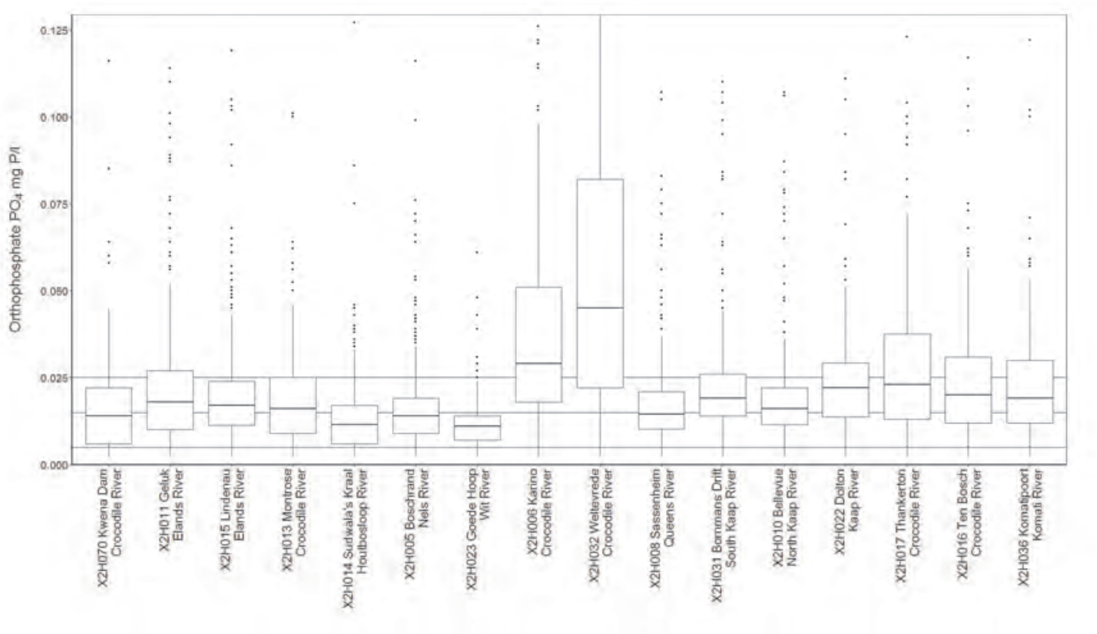

Figure 17 Box and whisker plots of orthophosphate levels at sites in the Crocodile River catchment. Boxes show the interquartile range, with medians indicated. Whiskers show levels within 1.5 times the interquartile range, and dots show outliers. The upper 2% of data are not plotted. Sites are roughly ordered along the axis by distance upstream, with those higher in the catchment to the left. Horizontal lines show guidelines adopted for this analysis.

Although, as noted above, the majority of sites have median orthophosphate levels that are tolerable or better, no site has a median that is ideal. This includes the selected instream reference site below Kwena dam (X2H070), whose median level is just below the transition point from acceptable to tolerable. The best sites are at Goede Hoop in the Wit River (X2H023) and at Sudwala's Kraal in the Houtbosloop River. There is a general trend, particularly in terms of higher 25th and 75th percentiles, of a greater likelihood of higher levels of orthophosphate at sites lower on the river, particularly below the confluence with the Kaap River.

Total inorganic nitrogen

Total inorganic nitrogen levels across the Crocodile River catchment are shown in Figure 18. The majority of sites are well-placed in the ideal class, having only occasional outliers moving beyond this class. These sites are all located in upstream reaches of the Crocodile River or its tributaries.

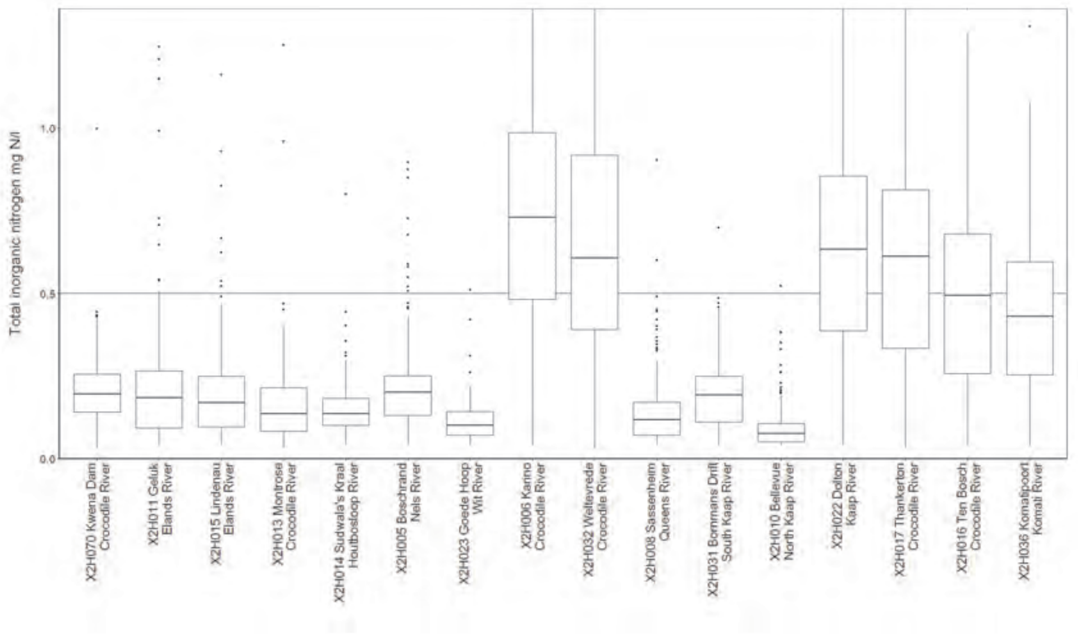

Figure 18 Box and whisker plots of total inorganic nitrogen levels at sites in the Crocodile River catchment. Boxes show the interquartile range, with medians indicated. Whiskers show levels within 1.5 times the interquartile range, and dots show outliers. The upper 2% of data are not plotted. Sites are roughly ordered along the axis by distance upstream, with those higher in the catchment to the left. Horizontal lines show guidelines adopted for this analysis.

The remaining sites have distinctly higher nitrogen levels, although none pass beyond the acceptable class. As is the case with orthophosphate, levels of total inorganic nitrogen are dramatically elevated at Karino (X2H006) and at Weltevrede (X2H032) in the Crocodile River downstream from Nelspruit and KaNyamazane, and in the same way, it appears likely that this may be owing to the impacts of wastewater treatment plants serving these large populated areas.

All sites in Crocodile River downstream of its confluence with the Kaap River, and the furthest downstream site in the Kaap River (X2H022 Dolton) also have elevated total inorganic nitrogen levels, and this appears due to the input of waters from the Kaap River. There are three sites further upstream in the Kaap River catchment (X2H008, X2H010 and X2H031) that do not show this impact, and this locates the source of nitrogen in the lower Kaap River or lower North or South Kaap Rivers. The input from the Kaap River ameliorates slowly downstream. It is not known from this survey whether the nitrogen levels in the Komati River just below the confluence of the Crocodile and Komati Rivers (X2H036) are largely due to input from the Crocodile/Kaap Rivers.

Un-ionized ammonia

Levels of un-ionized ammonia in the Crocodile River catchment are all with the ideal range (Figure 19). The only notable trend is an increase in sites in the Lower Kaap River and downstream of this in the Crocodile River. This would appear most likely to be associated with the increased levels of total inorganic nitrogen in this reach. Levels also increase slightly at Karino (X2H006) below Nelspruit, but this increase is not maintained thereafter. Levels in the Elands River are also slightly elevated.

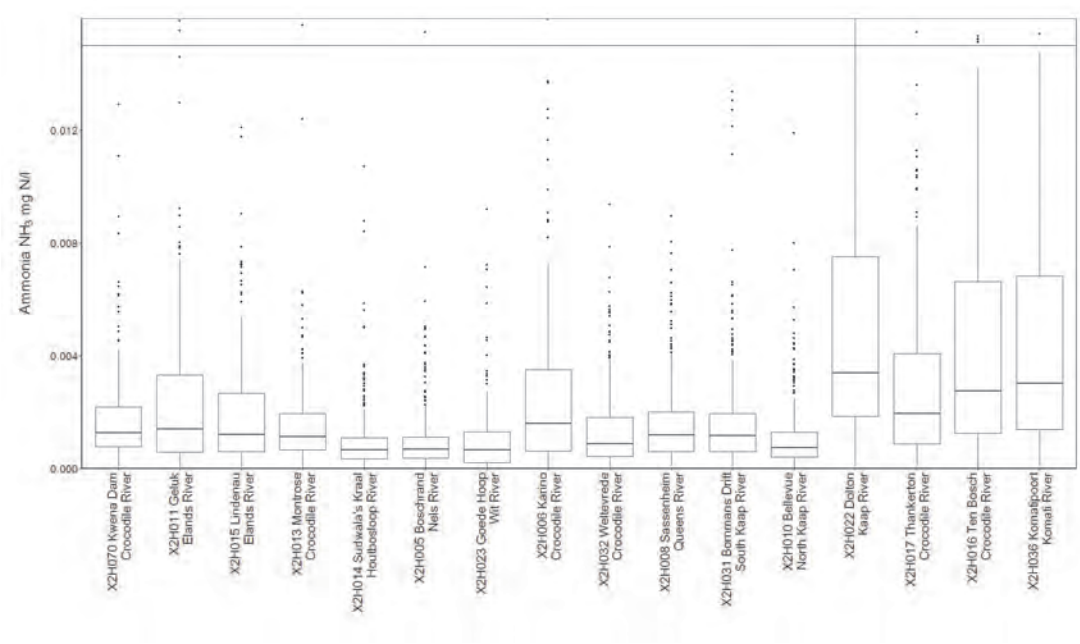

Figure 19 Box and whisker plots of un-ionized ammonia levels at sites in the Crocodile River catchment. Boxes show the interquartile range, with medians indicated. Whiskers show levels within 1.5 times the interquartile range, and dots show outliers. The upper 2% of data are not plotted. Sites are roughly ordered along the axis by distance upstream, with those higher in the catchment to the left. Horizontal lines show guidelines adopted for this analysis.

Electrical conductivity

The majority of sites in the Crocodile River catchment have electrical conductivity levels that are classed as ideal (Figure 20). Only the lowest site on the Kaap River and sites thereafter along the Crocodile and Komati Rivers can be considerable as acceptable (though the 75^{th} percentile of data from Dolton (X2H022) in the lower Kaap is in the tolerable class, and the median is close to the class boundary).

For the most part, spatial trends in conductivity follow those described for un-ionized ammonia levels. One notable difference is elevated conductivity at Goede Hoop in the Wit River (X2H023). This can be contrasted with the adjacent Nels River (X2H005). Conductivity remains high in the mainstream Crocodile River, and impacts owing to Wit River input cannot be separated from the likely effect of the conurbations of Nelspruit and KaNyamazane. Conductivity at sites in the Elands River is elevated in comparison to others in the upper catchment, and an assessment of temporal change at these site reveal levels increasing with time. Finally, dramatically increased conductivity at Dolton in the Lower Kaap appears to lead to the lower Crocodile having higher, yet still acceptable levels of conductivity.

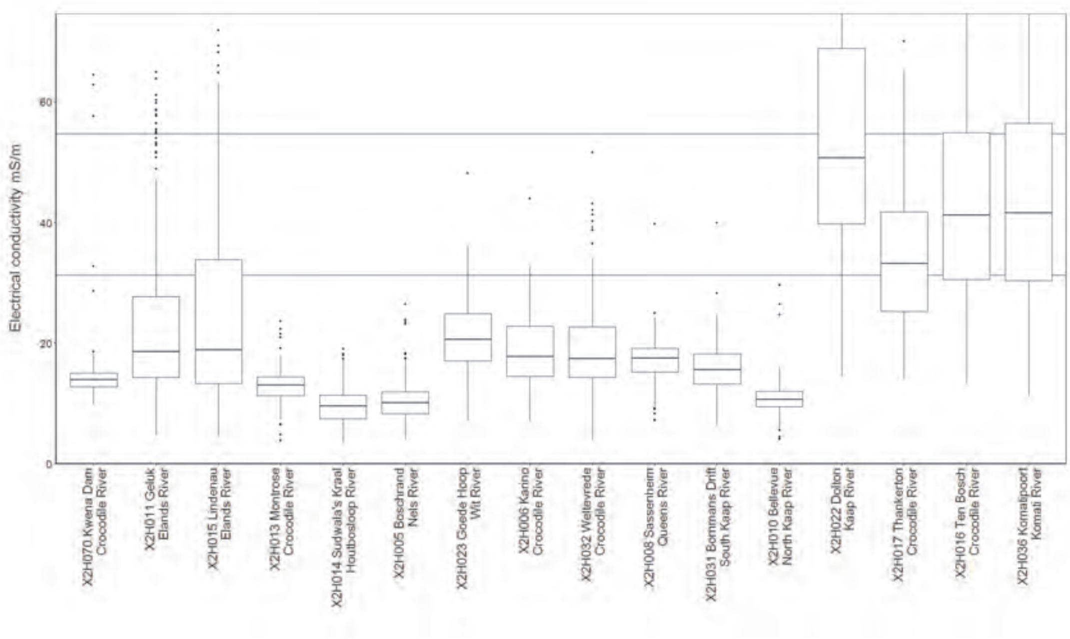

Figure 20 Box and whisker plots of electrical conductivity levels at sites in the Crocodile River catchment. Boxes show the interquartile range, with medians indicated. Whiskers show levels within 1.5 times the interquartile range, and dots show outliers. The upper 2% of data are not plotted. Sites are roughly ordered along the axis by distance upstream, with those higher in the catchment to the left. Horizontal lines show guidelines adopted for this analysis.

pH

Spatial trends in pH in the Crocodile River catchment show most of the monitored sites to be classed as ideal (Figure 21). Temporal trends will be addressed below, but at this point it is worth noting that increases in pH across sites in this catchment is a general trend over the previous 20-30 years, and the current situation is not necessarily reflected in Figure 21.

pH levels in the Elands River are slightly higher overall than others in the upper catchment. A similar slight increase is found in the South Kaap River and the Queens river, compared to the North Kaap River and indeed the Crocodile mainstream below Nelspruit/KaNyamazane.

The site with the highest conductivity levels, and the only one whose 25th percentile is located in the acceptable class, is Dolton (X2H022) in the lower Kaap River. Compared to other sites, the data from this site also show low variability. As with many other parameters assessed here, input from the Kaap River seems to elevated conductivity in the Crocodile River downstream of the confluence of the two, although increases in pH between Thankerton (X2H017) and Ten Bosch (X2H016) along the Crocodile River suggest that other impacts are present.

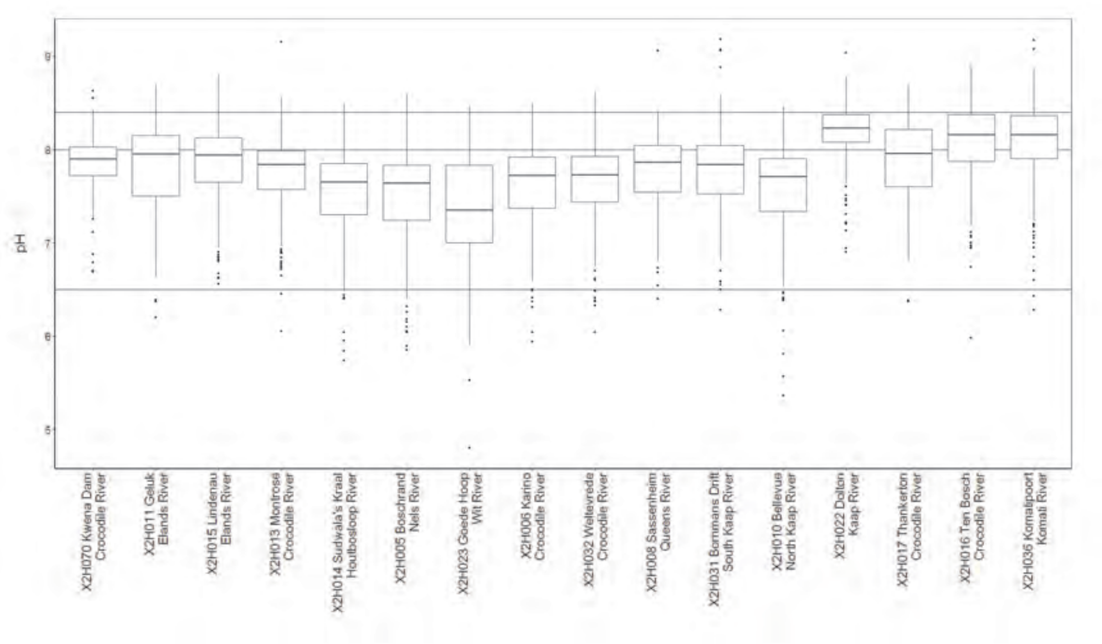

Figure 21 Box and whisker plots of pH levels at sites in the Crocodile River catchment. Boxes show the interquartile range, with medians indicated. Whiskers show levels within 1.5 times the interquartile range, and dots show outliers. Sites are roughly ordered along the axis by distance upstream, with those higher in the catchment to the left. Horizontal lines show guidelines adapted for this analysis.

Chemical weathering index

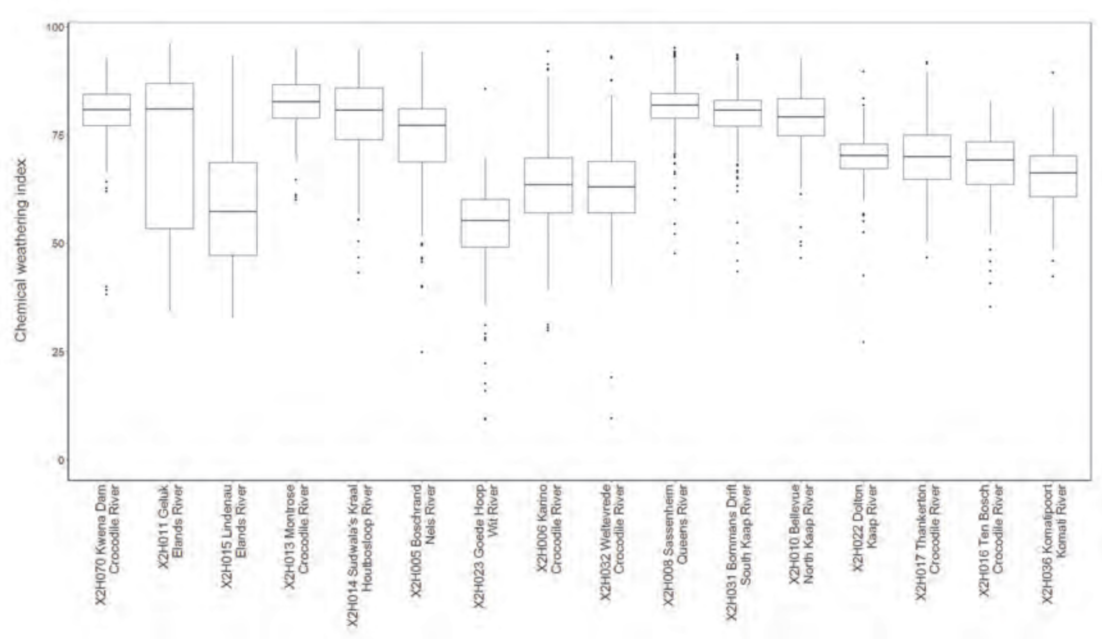

Figure 22 Box and whisker plots of the chemical weathering index proposed by Huizenga *et al.* (2013) at sites in the Crocodile River catchment. Boxes show the interquartile range, with medians indicated. Whiskers show levels within 1.5 times the interquartile range, and dots show outliers. Sites are roughly ordered along the axis by distance upstream, with those higher in the catchment to the left.

The chemical weathering index, unlike the majority of parameters assessed here, is indicative of the extent to which chemical weathering of rocks, as opposed to largely anthropogenic impact, contributed to dissolved ion characterization of the water. As this index assesses chemical weathering as a function of the ratio of carbonates and bicarbonates to other ions, it will be influenced by the geology of the region. Figure 22 shows that at all sites in the Crocodile River catchment, a relatively large component of the ionic makeup of the water derives from chemical weathering, and that no site has an ionic balance dominated by ions associated with salinization, sulphate contamination, or eutrophication.

The trends in Figure 22 largely confirm trends found in several other parameters. The chemical weathering index is depressed in the Elands River, particularly at the lower site (Lindenau XaH015). The Wit River also has a relatively low median chemical weathering index, and this is matched by sites in the Crocodile River downstream of the confluence with the Wit River (Karino X2H006 and Weltevrede X2H032). As these sites are likely to be affected by impacts at Nelspruit and KaNyamazane, changes at these sites cannot simply be attributed to input from the Wit River. Finally, the chemical weathering index is depressed at Dalton (X2H022) on the lower Kaap River, in contrast to sites higher in the Kaap River catchment. Sites in the Crocodile River downstream of the confluence with the Kaap River (Thankerton X2H017 and Ten Bosch X2H016) also have depressed chemical weathering indices.

Sulphate contamination index

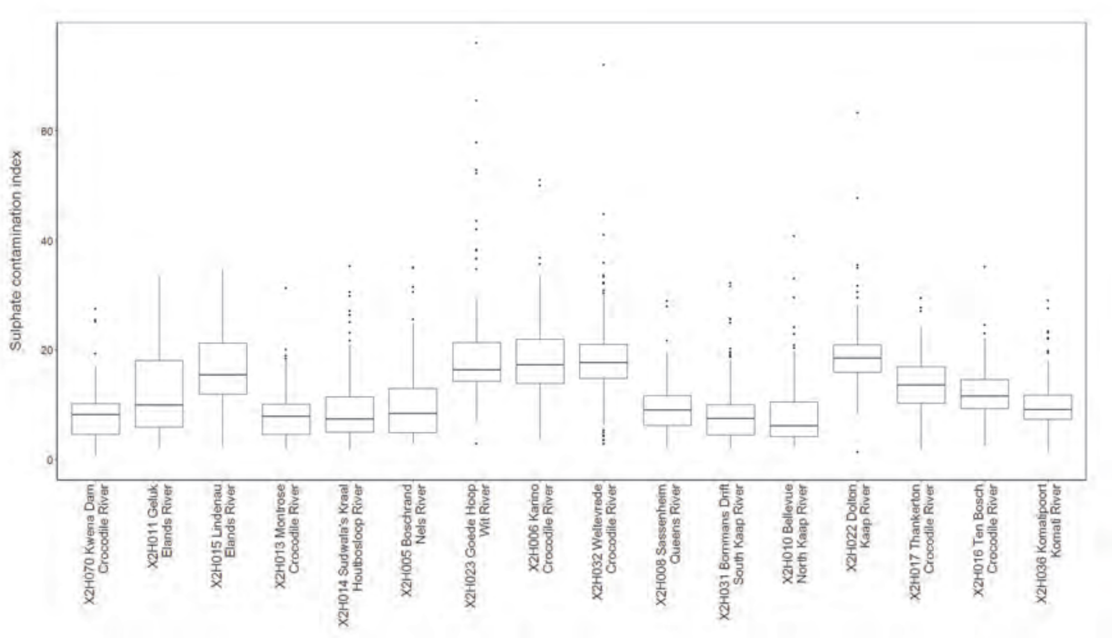

Figure 23 Box and whisker plots of the sulphate contamination index proposed by Huizenga *et al.* (2013) at sites in the Crocodile River catchment. Boxes show the interquartile range, with medians indicated. Whiskers show levels within 1.5 times the interquartile range, and dots show outliers. Sites are roughly ordered along the axis by distance upstream, with those higher in the catchment to the left.

The spatial trends in the sulphate contamination index, presented in Figure 23, together with those of the chloride salinization index (Figure 24, below) go some way towards identifying the drivers of changes in other parameters, in particular electrical conductivity and the chemical weathering index.

Before proceeding, it is important to recall that this index is a relative measure, and that the scores from this catchment are all very good in comparison to those in the Olifants River catchment. In addition, median levels of sulphate at all sites in this catchment fall into the ideal class as defined by DWA (2011a) generic RQOs, and even at the site with the worst median sulphate levels (Dolton X2H022), 87% of data are classed as ideal and only 13% are in the acceptable class. Relative levels of the sulphate contamination index therefore do not necessarily indicate high absolute levels of sulphate.

The index increases along the Elands River suggesting sulphate input in this stretch. Wit River levels are relatively high, as are sites in the mainstream Crocodile River below the Wit River confluence (and Nelspruit and KaNyamazane). Relatively high index scores are found at Dolton on the lower Kaap River, though sites higher in this catchment have low scores. Finally, the Crocodile River has elevated index scores below the confluence with the Kaap River, and this effect decreases as one move down the Crocodile towards the confluence with the Komati River.

Chloride salinization index

The results from the chloride salinization in index, presented in Figure 24 below, present a complementary picture to the chemical weathering and sulphate contamination indices. A major difference between patterns observed in Figure 24 and those seen in several other parameters is that index scores from Dolton (X2H022) on the lower Kaap River are low, and it appears that the impacts on this stretch of river do not lead to the introduction of chloride to the system.

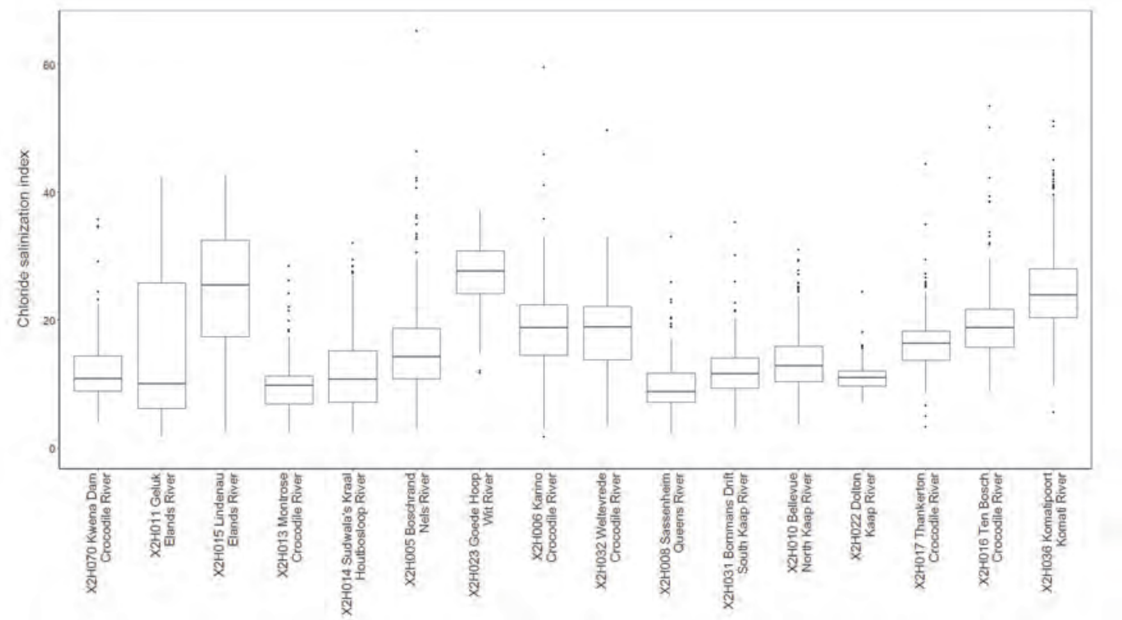

Figure 24 Box and whisker plots of the chloride salinization index proposed by Huizenga *et al.* (2013) at sites in the Crocodile River catchment. Boxes show the interquartile range, with medians indicated. Whiskers show levels within 1.5 times the interquartile range, and dots show outliers. Sites are roughly ordered along the axis by distance upstream, with those higher in the catchment to the left.

As with the sulphate contamination index, one must recall that the index indicates the relative amount of chloride in solution. When absolute levels are assessed, the median chloride levels at all sites can be classed as ideal. The worst site of those surveyed is at Komatipoort (X2H036) on the Komati River, followed by Ten Bosch (X2H016), low on the Crocodile River. Most of the records (75%) for Ten Bosch fall into the ideal class, and, with a small handful of exceptions, the rest are classed as acceptable.

Relative levels of chloride increase along the Elands River suggesting an input in this reach. Samples from Goede Hoop (X2H023) lower Wit River also have high index scores. Sites in the Crocodile just below the Wit River confluence have relatively elevated scores, but overall less that samples from the Wit River. Index scores from sites in the Kaap River catchment are relatively low overall. However, the lower reaches of the Crocodile show increasing index scores and this suggests that salinization may be taking place in this region.

Sodium adsorption ratio

The spatial patterns in the sodium adsorption ratio, an index of the suitability of water for irrigation, are shown in Figure 25. Despite variation between sites, all sites overall can be classed as ideal, with the worst site in the Crocodile catchment being Ten Bosch (X2H016), where 76% of samples would be classed as ideal.

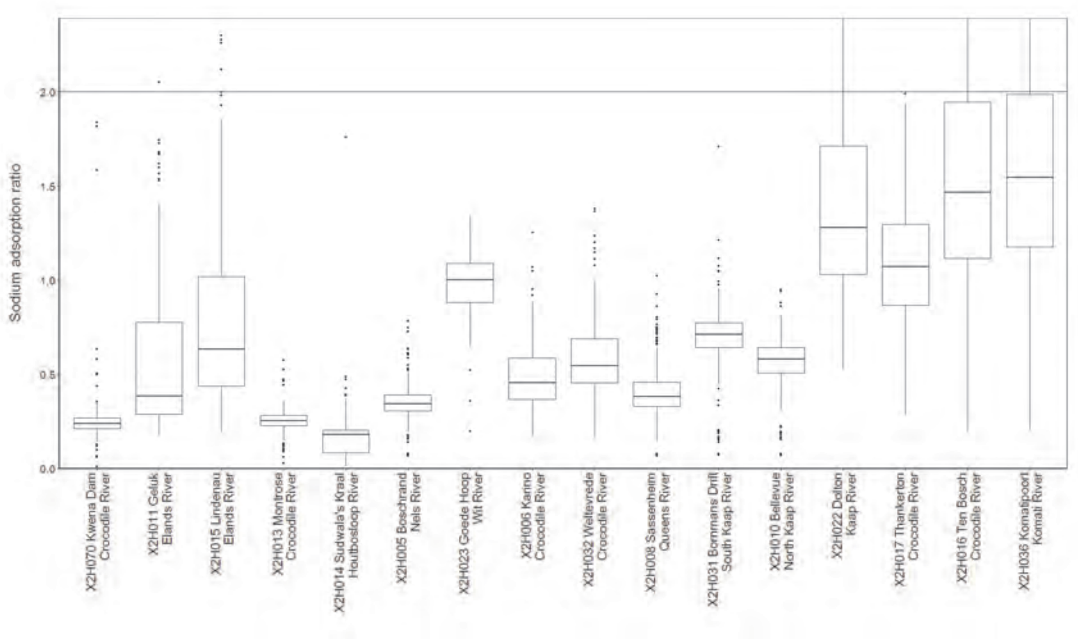

Figure 25 Box and whisker plots of sodium adsorption ratios at sites in the Crocodile River catchment. Boxes show the interquartile range, with medians indicated. Whiskers show levels within 1.5 times the interquartile range, and dots show outliers. The upper 2% of data are not plotted. Sites are roughly ordered along the axis by distance upstream, with those higher in the catchment to the left. Horizontal lines show guidelines adopted for this analysis.

Many of the patterns observed for other parameters are present in Figure 25. Scores increase along the Elands River, and are elevated in the Wit River compared to other sites in the area. Scores are high at Dalton (X2H022) in the lower Kaap River in comparison to sites higher in the catchment. Finally, scores along the lower Crocodile are relatively high, and samples from this region have a higher chance of being classed as acceptable rather than ideal.

Corrosion potential ratio

The median score of half of the sites assessed here would be classed ideal, with the remained being classed as acceptable (Figure 26). Sites in the Elands River show wide variation in scores compared to other sites; this is driven by increases over in scores over time, so recent scores from the Elands River would be classed as acceptable to tolerable in the upstream site at Geluk (X2H011) and as tolerable in the downstream site at Lindenau (X2H015).

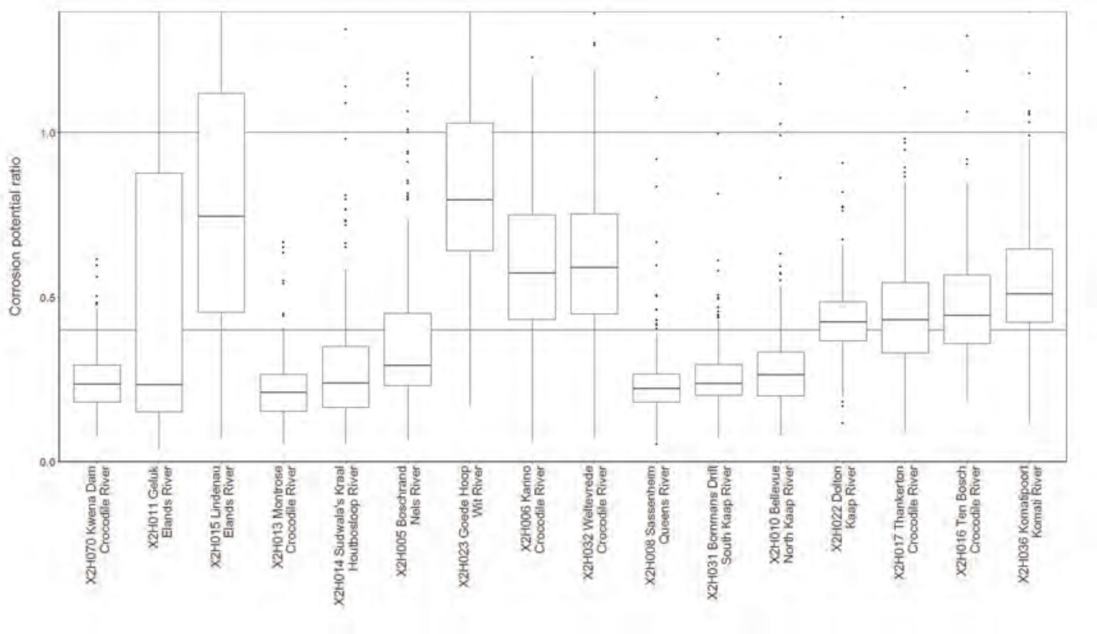

Figure 26 Box and whisker plots of the corrosion potential ratio described by Ashton and Dabrowski (2011) at sites in the Crocodile River catchment. Boxes show the interquartile range, with medians indicated. Whiskers show levels within 1.5 times the interquartile range, and dots show outliers. The upper 2% of data are not plotted. Sites are roughly ordered along the axis by distance upstream, with those higher in the catchment to the left. Horizontal lines show guidelines adopted for this analysis.

Water from the Wit River is more likely to cause corrosion than water from the adjacent Nels River. Corrosion potential is also elevated in Crocodile River sites below the confluence with the Wit River, and the settlements of Nelspruit and KaNyamazane. Samples from sites in the upper Kaap River catchment have a low potential for corrosion, but this increases in the lower Kaap River. Sites in the Crocodile downstream of the Kaap River confluence have a similar corrosion potential to that of the lower Kaap River.

Temporal trends in water quality

This section of the report assesses changes with time in selected water quality parameters at monitoring points across the Crocodile River catchment. The monitoring points selected have datasets ranging from 286 samples at Kwena Dam (X2H070) at the top of the catchment to 1417 samples at Ten Bosch (X2H016), low in the catchment just before the confluence with Komati River. The results from this section complement the basic spatial analysis of trends in the catchment, and enables assessment and quantification of temporal change, which is the primary aim of this module.

Temporal changes can be caused by a range of things, including anthropogenic impact and management choices, but also including natural causes. Under natural causes, flow and runoff changes owing to changing rainfall changes can be expected to modify some or all of the parameters assessed here. Changes in rainfall owing to seasonal changes have been included in the statistical models used in tests, but the results of statistical tests cited here do not reflect seasonal change. The statistical results only apply to temporal change that excludes seasonal change.

As rainfall patterns will vary outside of predicable seasonal change, reaching extremes in droughts and floods, these are likely to modify water quality parameter and their effects are covered by the statistical tests applied.

Microbial data could not be sourced for the monitoring points selected to provide spatial cover of the catchment. Microbial data is also of more recent provenance than the other water quality parameters assessed here. For this reason, temporal trends in microbial data will be presented separately from the other water quality parameters.

Kwena Dam (X2H070) on the Crocodile River

This site is the uppermost site on the Crocodile River and is close to the headwaters of the river. Although the data record from this site is sparse (in comparison to other selected sites) there are sufficient data for analysis and this site will serve as the best reference site of the various alternatives that were assessed.

Data from this site dates from the mid 1980's, and thus it covers most of the timespan of importance in this analysis. However, there are several gaps in the record where sampling was evidently abandoned (Figure 27).

For the most part, parameters from this site are in an ideal state, and evidence for major change is low. Exceptions to this generality follow.

Orthophosphate levels show relatively linear increase with time, from mostly acceptable levels in the mid-1980's to levels that are often only tolerable or worse. Particularly high values are recorded over the period 2004-2008. Data collection since that time is sparse and it is not clear whether this trend is continuing. A clear and significant seasonality is seen in orthophosphate levels ($p=0.032$). Total inorganic nitrogen levels are also seasonal ($p<0.001$), though there is no significant change in nitrogen levels over time.

pH levels have changed over time, increasing from the mid-1980s to a peak in approximately 1998, and decreasing somewhat since then. In early and more recent samples, the pH would be classed as ideal; however, when levels were high they would mostly be classed as acceptable.

The sulphate contamination index shows some change over time in the relative abundance of sulphate in the water, and, since 2000, shows a decrease in sulphate importance. Levels of sulphate throughout are low. The chloride salinization index has decreased, with fluctuations over time. As might be expected from this, the chemical weathering index is high and has increased over time, and bodes were for water quality at this site.

The sodium adsorption ratio had increased slowly over time, but the levels remain low and would be classed as ideal.

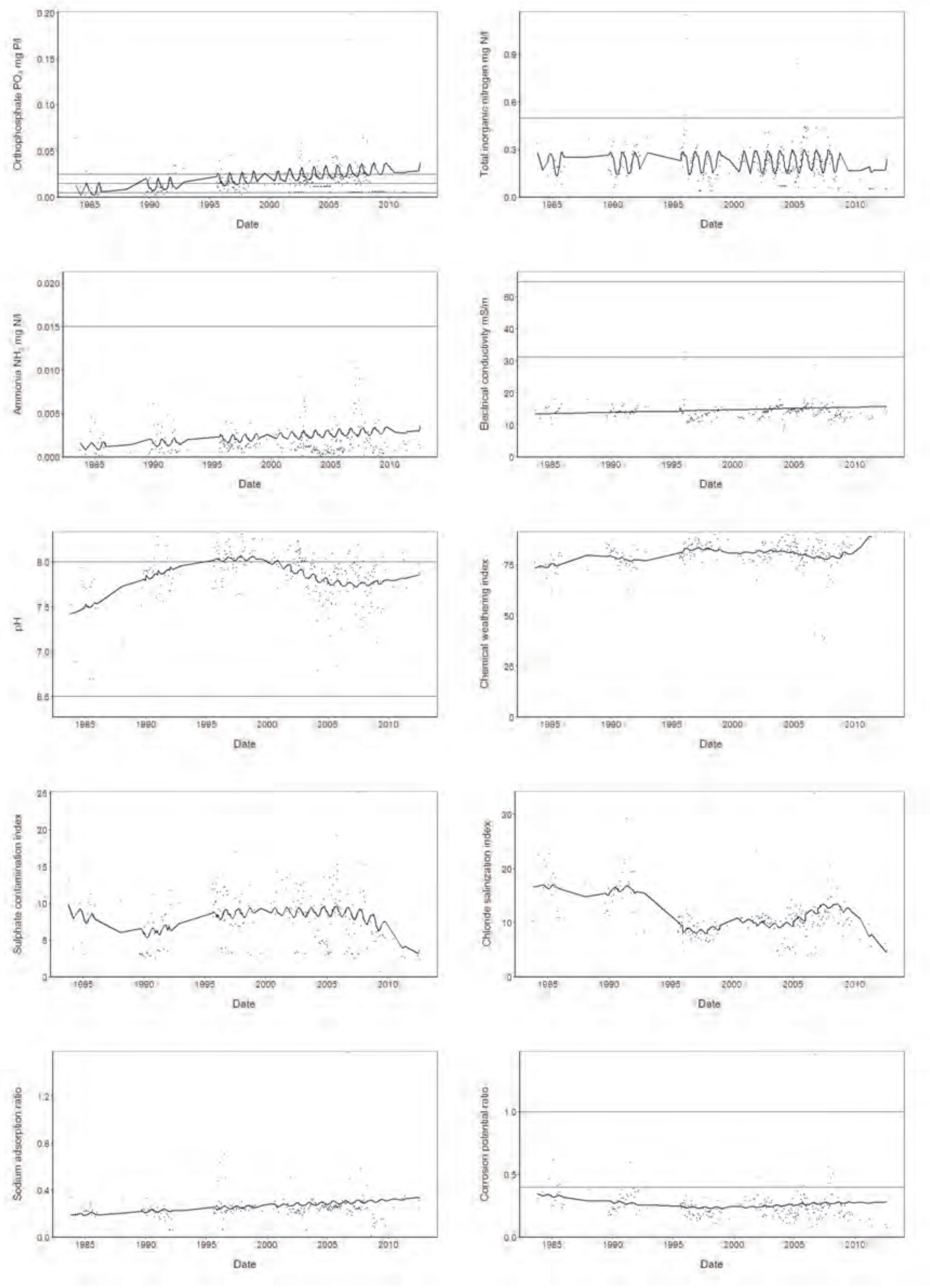

Figure 27 Temporal trends in several water quality parameters at monitoring point X2H070 downstream of the Kwena Dam on the Crocodile River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate p=0.011; TIN p=0.670; ammonia p=0.216; EC p=0.126; pH p<0.001; chemical weathering index p<0.001; sulphate contamination index p=0.001; chloride salinization index p<0.001; SAR p=0.013; and CPR p=0.143.

Overall water quality at this site is good. The only threat at this point in the river is the increasing levels of orthophosphate. Recent data is rare, though, and the on-going nature of this threat is not clear.

Geluk (X2H011) on the Elands River

Data at this site are available in the late 1970's, although very little data are available from the period 1986-1994. Earlier data show a river in a largely ideal state, with the exception of orthophosphate levels that were high and occasionally strayed beyond the tolerable class. In more recent times though, and in particular since 1998-2000, dramatic changes for the worse in all parameters assessed can be seen Figure 28. In the cases of orthophosphate, electrical conductivity, pH, and the corrosion potential ratio, these changes lead to a change classification, usually from ideal to acceptable. In this regard, the changes have not led to a severely modified state, but their dramatic onset and trajectory will bear monitoring. This is particularly the case as this site is located fairly high in the catchment (although it is downstream of Machadodorp), and would be expected to be fairly unimpacted compared to downstream sites.

Orthophosphate levels were in general high, but since approximately 2000 have shown an overall increase, with an increase overall to a tolerable or worse class, peaks reaching approximately 0.25 mg P/ℓ, and common exceedance of 0.05 mg P/ℓ. The increases in recent years show no sign of slowing (although short-term variation is very high), and these are a serious threat to the integrity of the upstream reaches of the Elands River.

Total inorganic nitrogen and un-ionized ammonia levels also show recent increases and increased short-term variation, but levels are not high enough to pose a threat.

Recent increases in salinity are accompanied by increases in the sulphate contamination index and the chloride salinization index (and concomitant decreases in the chemical weathering index). Together, these suggest that increased levels of sulphate and chloride might contribute significantly to the increased dissolved solid load. Sulphate loads remain in the ideal class (DWA 2011a) but chloride levels have a large number of samples in the acceptable class and a few in the tolerable class (DWA 2011a). Likewise, increases in sulphate and chloride levels cause the corrosion potential ratio to increase and move from ideal to acceptable and worse.

pH levels also increase to reach a peak around 2000 (although data from the 1990's are sparse), and move from a largely ideal class to an acceptable class in the process. It appears that these may decline somewhat in recent years, though they remain above levels from the 1980's.

Irrigation potential as measured by sodium adsorption ratios also increases considerably but remains in the ideal class.

All indices and parameters assessed, with the exception of orthophosphate, were found to show detectable and significant seasonal changes.

The rapid onset of the changes in water quality in this catchment suggest that the cause of these changes relates in all probability to a single impact.

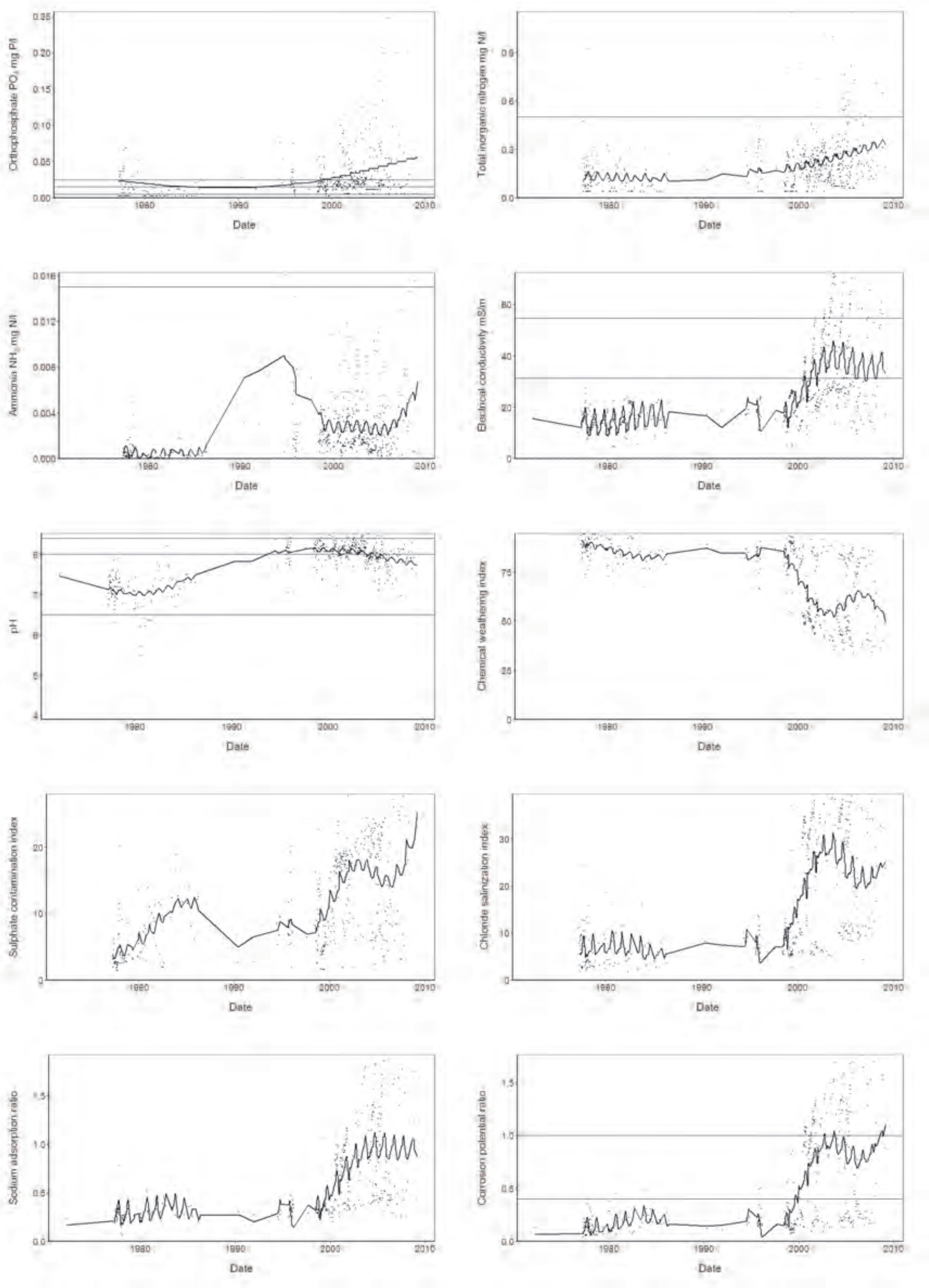

Figure 28 Temporal trends in several water quality parameters at monitoring point X2H011 at Geluk on the Elands River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate p=0.001; TIN p<0.001; ammonia p<0.001; EC p<0.001; pH p<0.001; chemical weathering index p<0.001; sulphate contamination index p<0.001; chloride salinization index p<0.001; SAR p<0.001; and CPR p<0.001.

Lindenau (X2H015) on the Elands River

The patterns of temporal change at Lindenau, low on the Elands River, in general ways mirror the change at Geluk (X2H011) upstream, with the exception that the sudden onset of change observed at Geluk is not observed at Lindenau, as the changes at the latter site have been more gradual and have had an earlier onset (Figure 29). Like Geluk, data at Lindenau start in the mid-1970s, but unlike the upstream site, no obvious gaps in the data record are present.

Orthophosphate levels show a steady, largely linear, increase over time with no significant seasonal variability. Although at the start of the data record, the site could not be classified as ideal, levels were largely within the acceptable range, although higher values were present. Thereafter values increased and at approximately 2000, passed beyond the tolerable class. 23% of the data from this site show a site with less than tolerable levels of orthophosphate. Total inorganic nitrogen shows an increase, but this plateaus in the mid-1990's and never exceeds ideal levels. Un-ionized ammonia likewise shows an increase, but never approaches the ideal-acceptable class boundary.

Electrical conductivity remains in the ideal class until the mid-1990s, when it increases to yield an acceptable level overall. Indications that the increase may have slowed since 2010 are difficult to support owing to sparse data. The increase in conductivity is accompanied by relative increases in the proportion of sulphate and chloride ions. At the end of the period of monitoring, chloride ion levels have passed from ideal to acceptable, and sulphate levels are approaching the class boundary. (DWA 2011a) These increases are reflected in an increase in the corrosion potential ratio, which, driven by increasing sulphate and chloride levels, passed from ideal to beyond acceptable.

The increase in chloride and sulphate is accompanied by a decrease in the chemical weathering index, which falls in a generally linear fashion from a start in the region of 0.85 to end around 0.35-0.40. This indicates a considerable shift in the ionic makeup of river water over this period.

pH levels show a similar change to that observed at other sites, increasing from an ideal level at the start of the data record, increasing in the mid-1980's to peak at an acceptable or worse level around 2000, and level off and decrease slightly thereafter.

Sodium levels also appear to increase as reflected in the sodium adsorption ratio, which climbs after 1990, but never passes out of the ideal range.

Beyond orthophosphate and total inorganic nitrogen, and to a less extent the sulphate contamination index, all changes in the water quality parameters assessed here have a seasonal component.

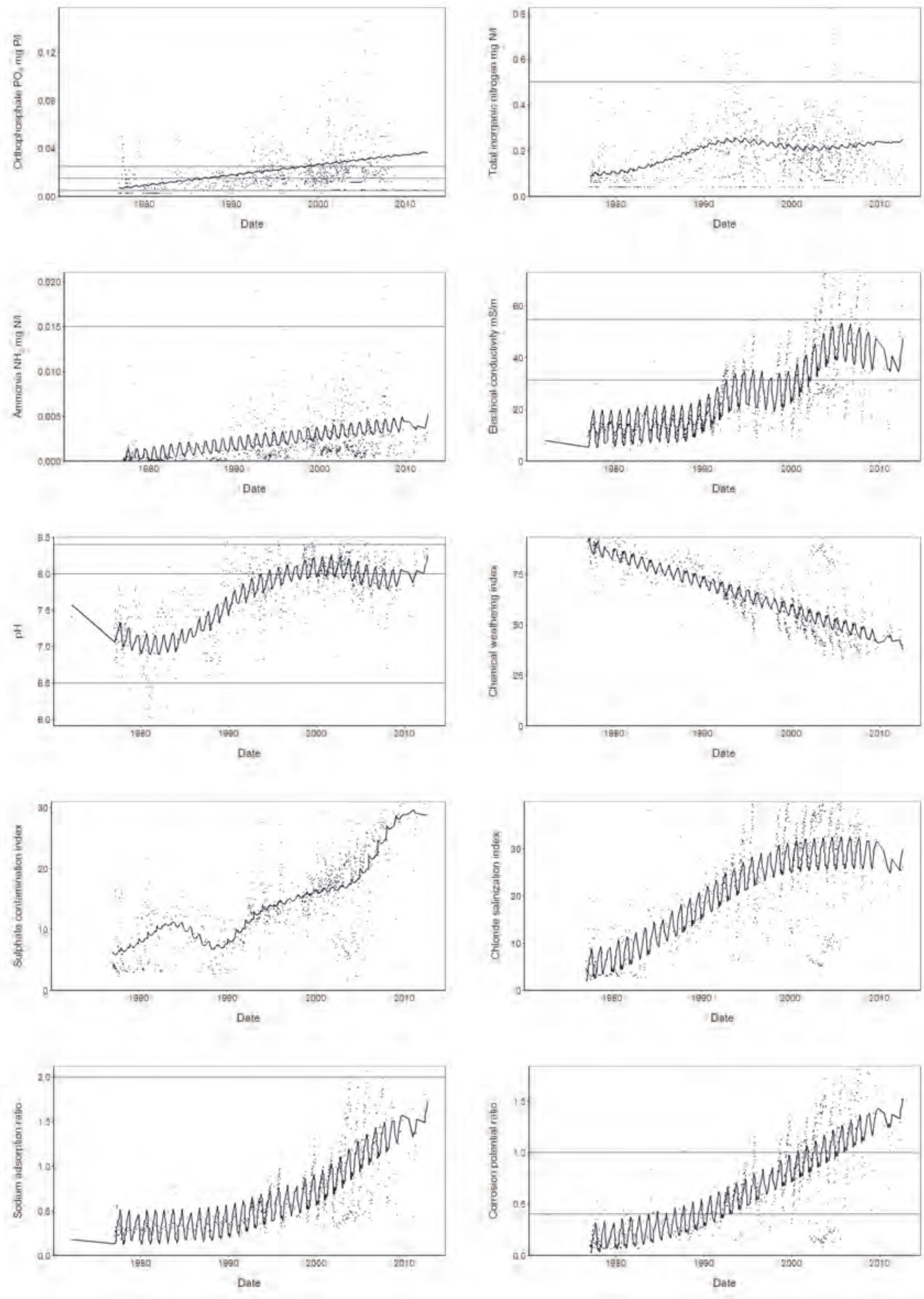

Figure 29 Temporal trends in several water quality parameters at monitoring point X2H015 at Lindenau on the Elands River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate $p<0.001$; TIN $p<0.001$; ammonia $p<0.001$; EC $p<0.001$; pH $p<0.001$; chemical weathering index $p<0.001$; sulphate contamination index $p<0.001$; chloride salinization index $p<0.001$; SAR $p<0.001$; and CPR $p<0.001$.

Montrose (X2H013) on the Crocodile River

Inspection of the trends in selected water quality parameters at Montrose on the Crocodile River reveals a river in good condition according to most parameters, bar nutrients (Figure 30). The data record is good from 1978 to the current day, with some parameters having a few records from the late 1960's.

Orthophosphate levels were acceptable until approximately 1990, and tolerable until around 2003. Thereafter levels have risen steeply to unacceptable levels. In samples collected after 2008, orthophosphate test results seem to show greatly reduced resolution, and nearly all results are 0.005, 0.05, 0.1 or 0.2 mg P/ℓ. This is in contrast to results from previous years. The cause of this is unknown and, unless this trend is purely coincidental, such lowered resolution will hamper assessment and consequent management of orthophosphate at this site.

Total inorganic nitrogen appears to show an increase with time, as does un-ionised ammonia, but both remain in the ideal range.

pH shows a trend common throughout this catchment, where levels rise from an ideal state to plateau around pH 8.0 in approximately the year 2000, where the upper levels pass from ideal to acceptable. Thereafter a slight decrease is observed, with the fitted curve suggesting that the decrease has slowed and pH levels appear to be reaching a new stable state, in the ideal range, but above that observed at the start of monitoring.

Electrical conductivity changes over time, with minor peaks around 1993 and 2004, but shows no ongoing trend over the monitored period, and remains classed as ideal throughout. Seasonal change is pronounced.

There is some change in the chloride salinization index and sulphate contamination index over the monitored period, but levels of both remain low, which is reflected in the high scores shown by the chemical weathering index. Sulphate and chloride levels are ideal over the monitored period.

The water from this site appears suitable for a range of uses as the sodium adsorption ratio and the corrosion potential ratio are both ideal and, despite temporal variation, neither show a trend towards increasing.

As is common in upper catchment sites, all parameters beyond nutrients displayed detectable and significant seasonality.

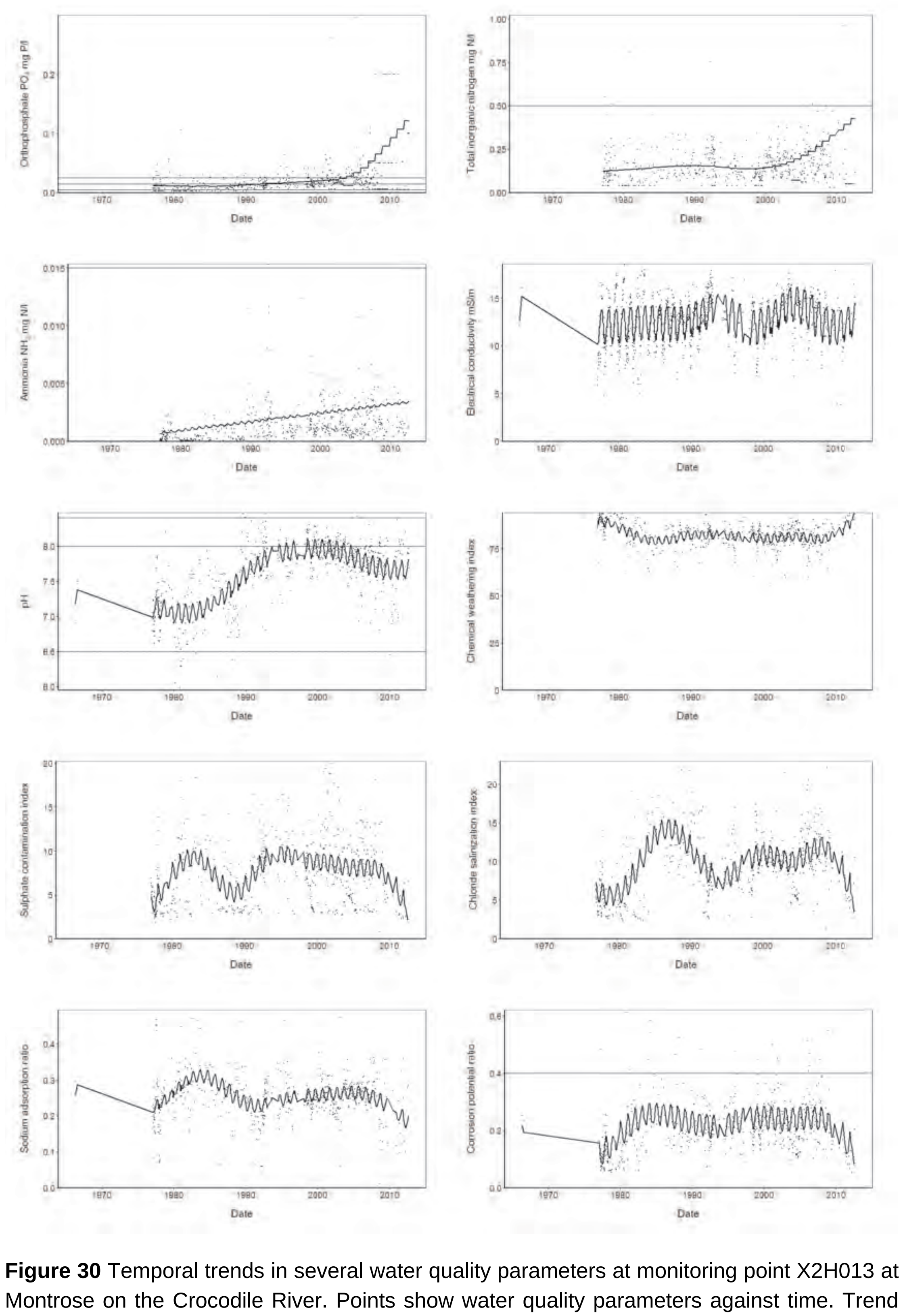

Figure 30 Temporal trends in several water quality parameters at monitoring point X2H013 at Montrose on the Crocodile River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate $p<0.001$; TIN $p<0.001$; ammonia $p=0.032$; EC $p<0.001$; pH $p<0.001$; chemical weathering index $p<0.001$; sulphate contamination index $p<0.001$; chloride salinization index $p<0.001$; SAR $p<0.001$; and CPR $p<0.001$.

Sudwala’s Kraal (X2H014) on the Houtbosloop River

In common with a number of sites in this catchment, this site has good data coverage from 1977 until the current time, with one sample from 1966 and another from 1972. The overall picture presented in Figure 31 is that of a site with good water quality, though with increasing levels of orthophosphate that may pose a problem in the future. The patterns seen in Figure 31 have much in common with those seen at Montrose (X2H013) on the Crocodile River.

Orthophosphate levels show an increase with time that seems largely linear from the late 1970’s. Over this period, the site shifts from a largely acceptable to a tolerable state. In contrast to earlier samples, when a number of sample were found to classed as ideal, later samples fall only into acceptable, tolerable or worse states. This trend of increasing orthophosphate over time is common, and often more pronounced, in upper catchment samples.

Total inorganic nitrogen and un-ionized ammonia show similar trends with an increase until 1992-1994 and somewhat of a decrease thereafter. Samples are all classed as ideal throughout.

Electrical conductivity changes over time but shows no overall increasing or decreasing trend over the monitored period, and remains in the ideal class throughout. There is a notable peak around 1993 that matches one noted at nearby Montrose in the Crocodile River. Samples from Montrose also showed a peak in approximately 2004 that is not as clear in the Sudwala’s Kraal samples, although the fitted model suggests that a lesser peak is present.

Despite temporal variation, no overall trends in sulphate contamination or chloride salinization indices were found, and evidence for contamination by either of these ions is lacking. This is corroborated by a high chemical weather index. The latter appears to show a slight drop after monitoring commenced, and sulphate and chloride influence shows a matching increase. Nevertheless, the three indices paint a picture of a river with an ionic complement dominated by natural processes.

As noted before for other sites, pH changes over time with an increase from approximately 1983 until around 1994, thereafter levels show a slight decrease. The majority of samples remain in the ideal state throughout.

In general, the water from this site is suitable for a range of uses as nearly all samples have ideal sodium adsorption and corrosion potential ratios. The latter approaches the ideal-acceptable class boundary though. Both indices show temporal variation with no overall increasing or decreasing trend.

All parameters bar nutrients, un-ionized ammonia and the sodium adsorption ratio showed a detectable and significant seasonal variation. Inspection of Figure 31 suggests that reparameterization of the GAMM for sodium adsorption ratio would have revealed a seasonal pattern that is currently concealed by variation in the data.

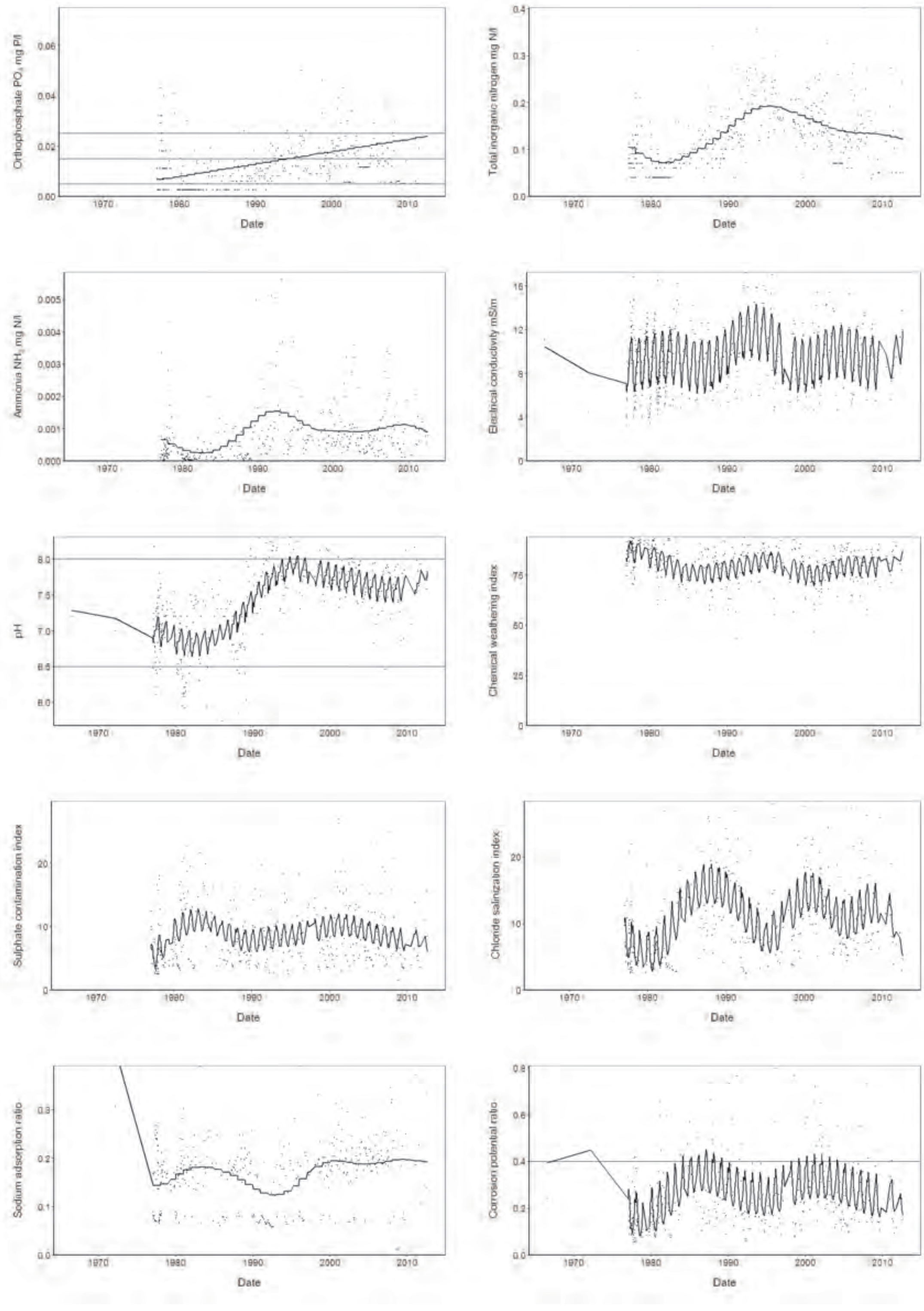

Figure 31 Temporal trends in several water quality parameters at monitoring point X2H014 at Sudwala's Kraal on the Houtbosloop River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate $p<0.001$; TIN $p<0.001$; ammonia $p<0.001$; EC $p<0.001$; pH $p<0.001$; chemical weathering index $p<0.001$; sulphate contamination index $p<0.001$; chloride salinization index $p<0.001$; SAR $p<0.001$; and CPR $p<0.001$.

Boschrand (X2H005) on the Nels River

As with most of the upper catchment sites assessed here, data from this site show a site where water quality is generally good, with the exception of high and increased orthophosphate levels (Figure 32). The data record for this site is good from 1977 until 2008. A few samples are available since 1962, though not for most parameters. Since 2008, sampling has been irregular and infrequent.

Short term variation in orthophosphate levels is high but nevertheless an increasing trend can be seen, and over the monitored period overall levels climb from acceptable to the upper bound of tolerable. Throughout, and in particular recently, a proportion of samples have orthophosphate levels that are beyond tolerable (15% of the full dataset).

Total inorganic nitrogen levels also increase over time but do not leave the ideal class. As with orthophosphate, variation between samples is high.

Un-ionized ammonia levels increase overall from approximately 1980 to 1990, and remain stable thereafter. Levels are ideal and no toxicity hazard is posed.

Electrical conductivity varies with time and shows a slight increase over the monitored period, but remains classed as ideal throughout. The peaks in conductivity in approximately 1993 and 2004 observed at Montrose and Sudwala's Kraal are present here as well.

pH followed the same pattern of temporal change found across this catchment, and increased from around 7 to nearly 8 from about 1980 to 1993. Levels thereafter showed a slight and slow decrease. Unlike the results from some other sites, pH levels at Boschrand remained classed as ideal throughout.

Sulphate influence as indicated by the sulphate contamination index is low and varies with time. Samples from recent years suggest that the sulphate contribution to ionic makeup is decreasing. The chloride fraction is a little larger but still fairly low for most of the monitored period. However, in the mid-to-late 1980's, the chloride salinization index increased considerably, a change reflected in a dip in the chemical weathering index. It is important to note that the chloride salinization is a relative index, and given low conductivity throughout, the peak in the index reflects chloride concentrations that remain ideal (DWA 2011a). Even with this change, the chemical weathering index shows a river whose ions are dominated by those resulting from natural processes.

The peak in chloride levels referred to above is also reflected in a peak in the corrosion potential ratio that moves samples in this period from ideal to acceptable corrosion potential. Thereafter this index declined until the end of the monitored period. The sodium adsorption ratio reveals that water from this site would be suitable for irrigation throughout the monitoring period.

Seasonality in all parameters bar nutrients and un-ionized ammonia was significant.

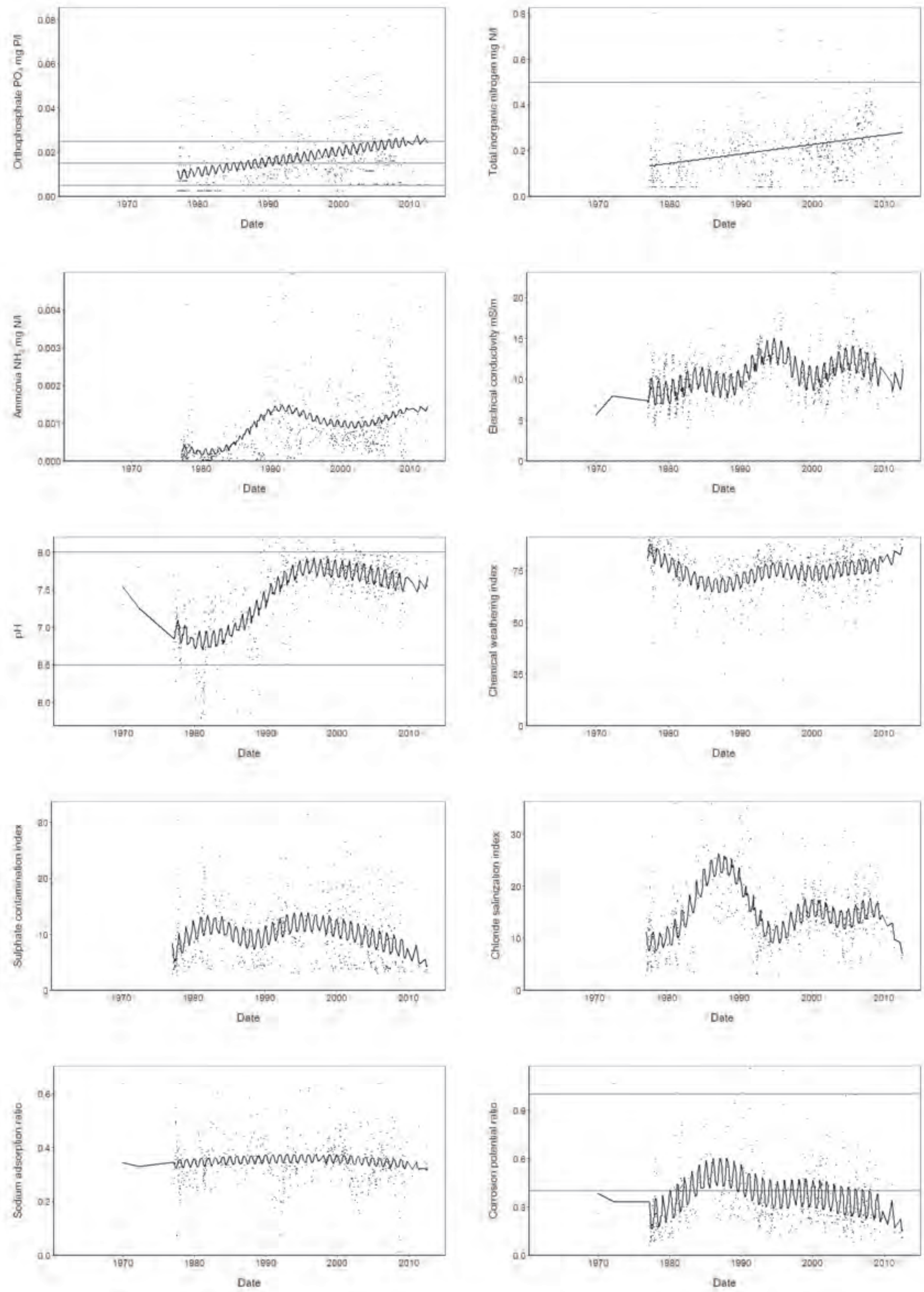

Figure 32 Temporal trends in several water quality parameters at monitoring point X2H005 at Boschrand on the Nels River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate p<0.001; TIN p<0.001; ammonia p<0.001; EC p<0.001; pH p<0.001; chemical weathering index p<0.001; sulphate contamination index p<0.001; chloride salinization index p<0.001; SAR p=0.253; and CPR p<0.001.

Goede Hoop (X2H023) on the Wit River

Inspection of the plots in Figure 33 show trends that differ from other monitoring points assessed in the mid-to-upper catchment. It is important when comparing this site with others to note that the data record presented here does not cover the timeframe covered at most monitoring points. In common with many other points, records begin in earnest with 1977, with a few dispersed prior records going back to 1968. However, records stop in 1992, when monitoring appears to have terminated. Data from this site will not be useful in assessing recent change, but it does provide a contrast in some respects to other sites in the mid-upper catchment.

In contrast to most sites in the upper Crocodile catchment, orthophosphate levels do not show an increase, and no trend in these was detected. Levels from most samples fall into the ideal-to-acceptable range. Total inorganic nitrogen levels are also low, and decline with time.

Un-ionized ammonia levels show a significant increase with time (despite decreased nitrogen) but do not pass out of the ideal range.

Conductivity levels increase initially, but the increase slows and stops in the late 1980's. Samples remain largely in the ideal range throughout. The pronounced seasonality common in samples from other sites is not visible in Figure 33.

pH levels begin to increase in the early 1980's and by end of the sampling period were still increasing such that samples from this period were starting to shirt from ideal to acceptable. The timing of this increase matches the generic trend to increased pH levels this catchment.

Inspection of the various indices show a river whose ionic makeup shows the effect of high levels of sulphate and higher levels of chloride. The chemical weathering index is far lower than at other comparable sites in this survey, suggesting some anthropogenic impact over the period surveyed. Nevertheless, neither sulphate nor chloride reached levels that were out of the ideal class according to DWA (2011a).

Although the water was suitable for irrigation according to the sodium adsorption ratio, the increased corrosion potential ratio, driven by higher ratios of chloride and sulphate, has samples moving from ideal to acceptable or worse, and the use of the water may have led to piping, etc. corrosion.

The seasonal patterns common in other datasets are not found here. GAMM analysis was unable to detect a seasonal pattern in any parameter except the sodium adsorption ratio, and inspection of data in Figure 33 does not reveal obvious seasonality.

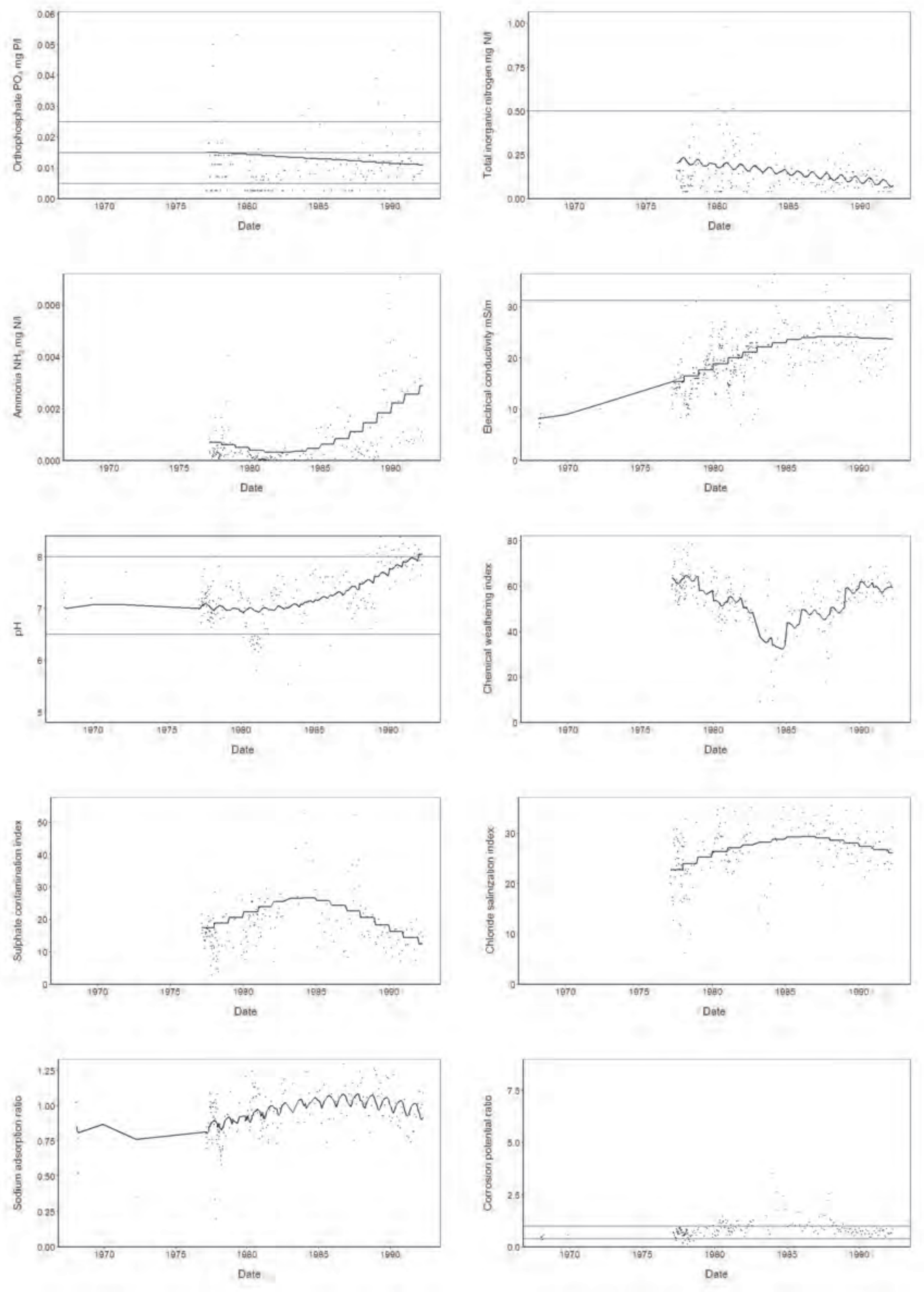

Figure 33 Temporal trends in several water quality parameters at monitoring point X2H023 at Goede Hoop on the Wit River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate $p<0.484$; TIN $p=0.025$; ammonia $p<0.001$; EC $p<0.001$; pH $p<0.001$; chemical weathering index $p<0.001$; sulphate contamination index $p=0.072$; chloride salinization index $p<0.001$; SAR $p<0.001$; and CPR model non-convergence.

Karino (X2H006) on the Crocodile River

This site is located on the Crocodile River after the confluence with the Wit River, and after the Crocodile River has passed Nelspruit and been exposed to the impacts associated with a large settlement. The overall picture seen in Figure 34 shows a river experiencing several impacts, the extents of many of which are increasing, even if current levels are not at yet serious in themselves. The data record spans 1977 to 2008, with some prior records extending back to 1962, and infrequent and irregular sampling since 2008.

Orthophosphate levels show a steady, linear increase with time. At the start of the data record, most samples were ideal-to-tolerable, but by the early 1990’s, the majority of samples were beyond tolerable, and by the end of the record very few samples were tolerable or better.

Total inorganic nitrogen levels showed a sharp and dramatic increase in the late 1970’s, moving from ideal to acceptable in the process. Since then there has been a general improvement, with temporal fluctuations, until 2000, when samples had ideal-to-acceptable nitrogen concentrations.

Un-ionized ammonia concentrations show increases that temporally match those of nitrogen, but levels remain well below the ideal-to-acceptable boundary.

pH levels show the changes found throughout the catchment, increasing from around 1980 to peak just before 2000, and decline slightly thereafter.

Levels of electrical conductivity show a largely linear increase across the data record, and by the end of the monitored period are approaching the boundary between ideal and acceptable. This trend is in sharp contrast to the next site upstream at Montrose, where no overall change in conductivity was noted. This increase in conductivity, and therefore charged ion concentrations, is accompanied by increases in the chloride salinization and sulphate contamination indices, indicating the greater relative dominance of these ions. By the end of the data record, scores for both indices were around 20, in comparison to scores of 5 to 10 for sites upstream on the Crocodile River for the same period and scores from the early data record at this site. Despite these increases, neither chloride nor sulphate levels pass out of the ideal class according DWA (2011a). The increases in these indices at this site is accompanied by an decrease in the chemical weathering index, indicating the reduced relative importance of natural weathering processes in defining ionic composition of the water.

Changes in ionic makeup are accompanied by changes in the suitability of the water for use. The sodium adsorption ratio shows an ongoing increasing trend though levels are still well within the ideal range. The increases in the relative importance of chloride and sulphate causes the corrosion potential index to move from ideal to acceptable in the mid 1980’s and never to recover thereafter.

Significant seasonal changes were found for most of the parameters assessed, with the exception of the chloride salinization and sulphate contamination indices.

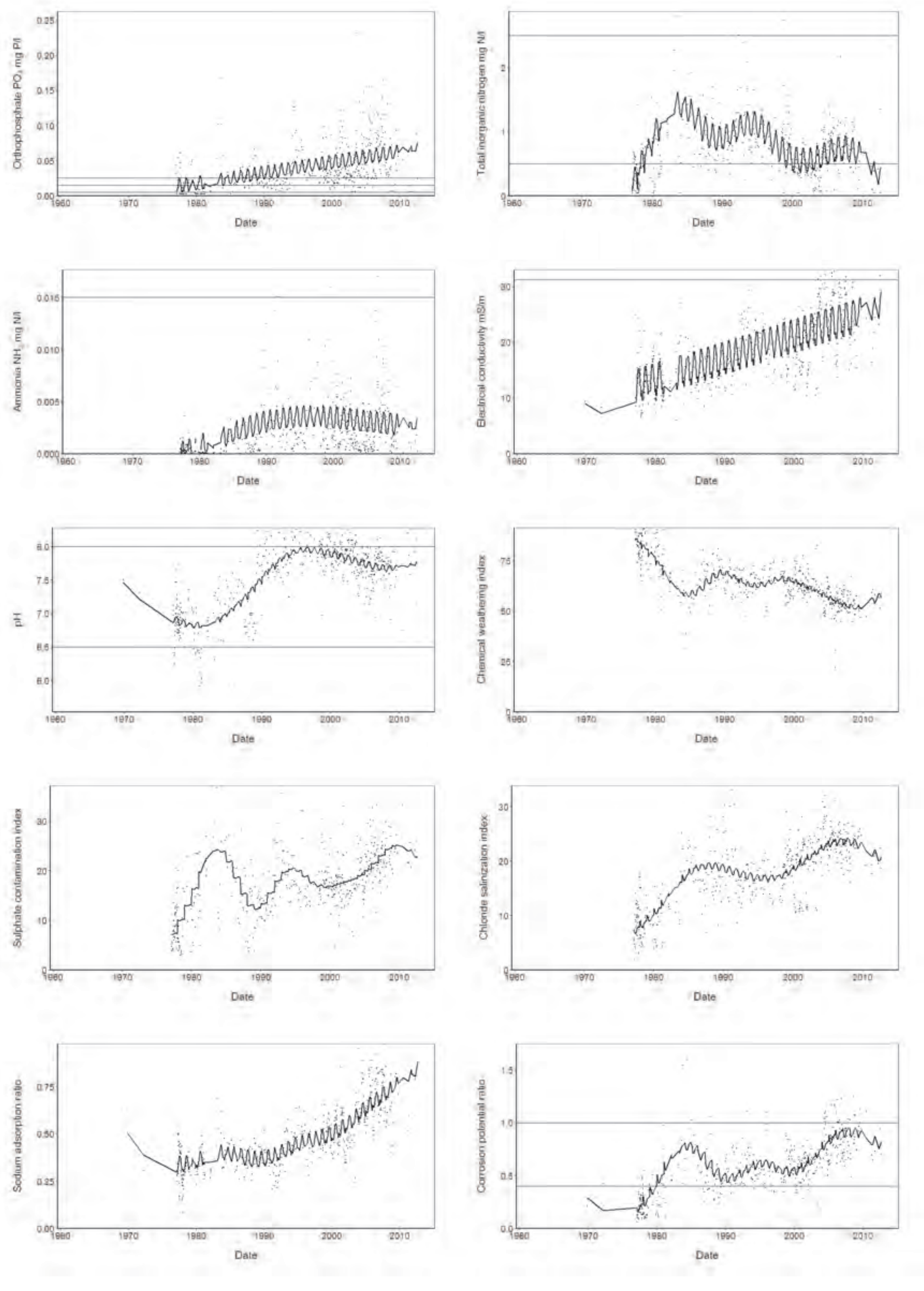

Figure 34 Temporal trends in several water quality parameters at monitoring point X2H006 at Karino on the Crocodile River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate $p<0.001$; TIN $p<0.001$; ammonia $p<0.001$; EC $p<0.001$; pH $p<0.001$; chemical weathering index $p<0.001$; sulphate contamination index $p<0.001$; chloride salinization index $p<0.001$; SAR $p<0.001$; and CPR $p<0.001$.

Weltevrede (X2H032) on the Crocodile River

Weltevrede is on the Crocodile downstream of Karino, and it can be expected that the changes in water quality noted at Karino may be repeated here. Inspection of the plots presented in Figure 35 reveals this largely to be the case. The data record for this site, like several others, is well populated from 1977 to 2008. There is one record from 1972, and data after 2008 is available but is irregular and infrequent.

Orthophospahe levels show a generally linear increase with time, much as at Karino. However, levels at Weltevrede are generally higher than those at Karino at the start of the data record, and remain higher thereafter. The outcome of this is that samples from Weltevrede would overall be classed as beyond tolerable, and, while a proportional of samples at the start of sampling fall into better classes, even ideal levels, by the end of the data record very few samples can be classed as ideal, acceptable or tolerable.

Total inorganic nitrogen trends at Weltevrede parallel those at upstream Karino, and the levels of nitrogen change little, indicating little recovery in this stretch of the river, but no added discernable impacts. A trend towards some improvement with time is noted here and at Karino.

Un-ionized ammonia levels increase with time but never pass out of the ideal class.

Electrical conductivity shows an increase with time from the start of sampling until, at the end of the sampled period, samples reach and occasionally exceed the ideal-to-acceptable class boundary. While the model fitting procedure fitted a curve showing temporal fluctuations at this site, and a largely linear model at Karino, assessment of the data reveals little difference between the two sites.

The changes in pH with time show the common catchment theme of an increase from a stable level beginning in the 1980's and ending around 2000 with a pH near the ideal-to-acceptable class boundary. Some temporal variation is noted at this site which has not been seen at other sites, but this does not modify the overall pattern of change.

Chloride salinization and sulphate contamination index change at this site reflects upstream changes closely. Both indices increase overall with time, with some, sometimes large, temporal fluctuations (for example, peaks in both in the mid-1980's). Despite the index changes, absolute concentrations of these ions do not pass beyond the ideal range as defined in DWA (2011a).

The chemical weather index shows a similar overall decreasing trend at this site as it does at Karino upstream, but levels are slightly better at this site.

The sodium adsorption ratio increases across the time of monitoring, but never exceeds the ideal class. The curve fitted to the corrosion protection ratio in Figure 35 shows a linear increase, and remains in the acceptable range. Comparison of the fitted curve and data show that a better fit would be obtained after model reparameterization. Nevertheless, it is clear that an increase in time occurred, and that data are for the most part classed as acceptable, after being largely ideal at the start of the data record.

Significant seasonal change in data from this site is limited in comparison to results from upstream.

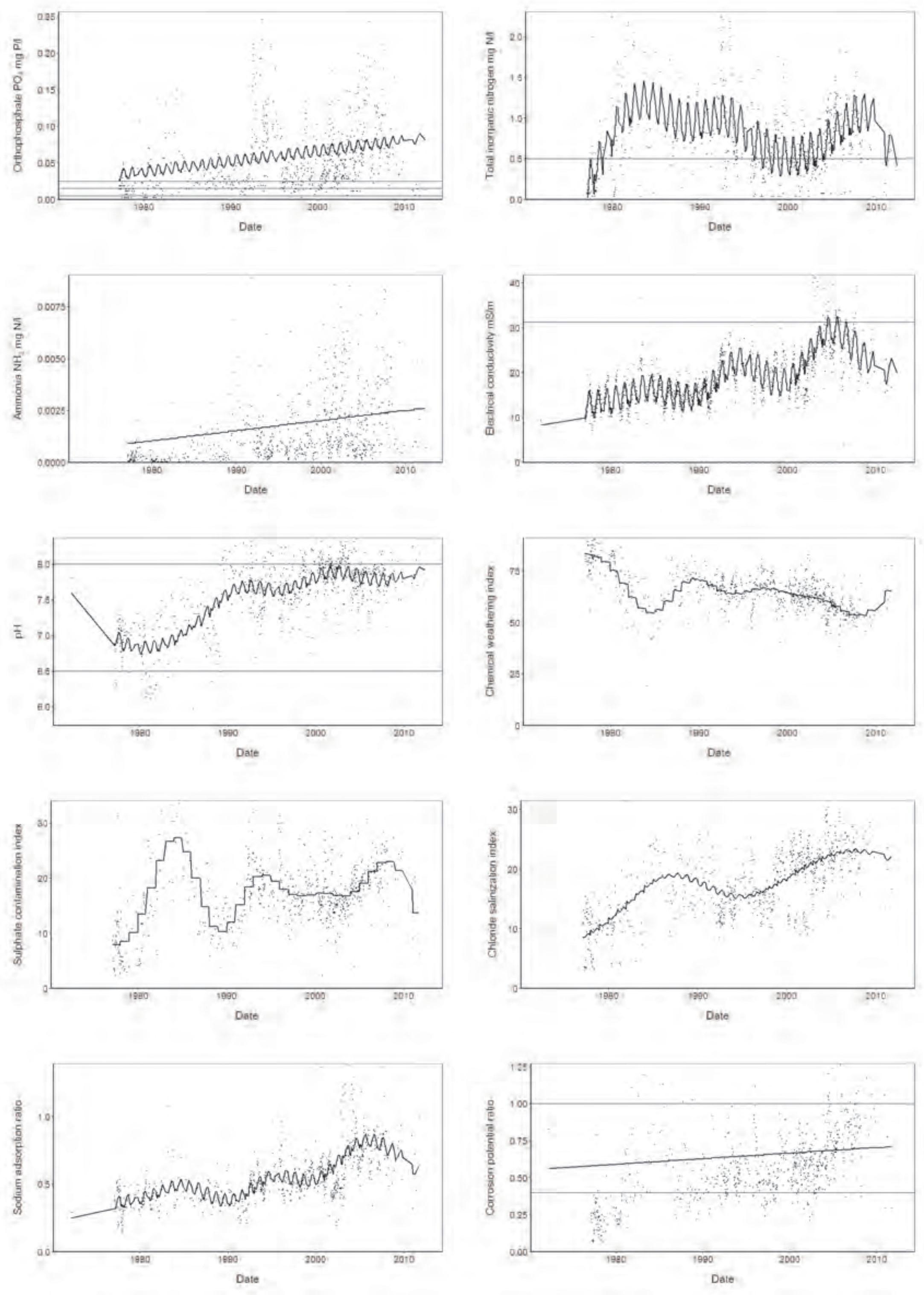

Figure 35 Temporal trends in several water quality parameters at monitoring point X2H032 at Weltevrede on the Crocodile River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate p=0.011; TIN p<0.001; ammonia p=0.043; EC p<0.001; pH p<0.001; chemical weathering index p<0.001; sulphate contamination index p<0.001; chloride salinization index p<0.001; SAR p<0.001; and CPR p=0.593.

Sassenheim (X2H008) on the Queens River

In contrast to the previous sites on the Crocodile, this site is located fairly high in the Kaap River catchment, where impacts of upstream activities might be expected to be low. Inspection of the data reveals this to be the case. Most parameters assessed reflect ideal conditions, with the exception of orthophosphate, levels of which are at least tolerable, and pH, which in common across this catchment, reaches ideal-to-acceptable levels in later samples (Figure 36). As is common in this catchment, the data record is well populated from 1977 to 2008. Two records are available prior to this period, and sampling thereafter is sporadic and irregular.

The GAMM model fitted for changes of orthophosphate levels indicates a linear decrease in orthophosphate levels with time. Statistical support for this is low, and an inspection of the data in Figure 36 suggest that the curve fitting algorithms were misled by a significant number of high values at the start of the data record, along with the high variation found in the data record. Reparameterization and rerunning of the model would likely improve the fit. Inspection of the data reveals a slow increase in orthophosphate from acceptable to tolerable levels over the course of the data record. Increases in orthophosphate with time are common in this catchment; at this site though, the increase is small in comparison with others.

Total inorganic nitrogen levels fluctuate over time, with a notable peak in the mid-1980's, but show no overall trend over the monitoring period and remain low throughout. Un-ionised ammonia increase from 1980 to 1990, but are also very low and well within the ideal class.

Electrical conductivity levels fluctuate with time but show no overall trend and remain in the ideal range. The ionic composition of the water underlying this is largely derived from natural processes, as the chemical weathering index scores are high, despite a small drop over the first 10 or so years of the data record. The sulphate contamination and chloride salinization indices are correspondingly low and show no real trend with time beyond an increase in the early data record. There is an increase at the start of the data record of the sulphate contamination index in the early 1980's, followed by a decrease and then a stabilization, which can be observed at many sites in the catchment.

The pH shows an increase characteristic of this catchment, increasing from about 7.3 to 8.0, and a slight decreased and stabilization afterwards, so the majority of recent samples would be classified as ideal.

Finally, the water from this site is eminently suitable for irrigation and industrial use, according to the indices employed here, and, beyond an increase in the corrosion potential index at the start of the data record, no obvious trends are noted.

Significant seasonality was only detected in nitrogen levels, conductivity, the chemical weathering index, the sodium adsorption ratio and the corrosion potential ratio.

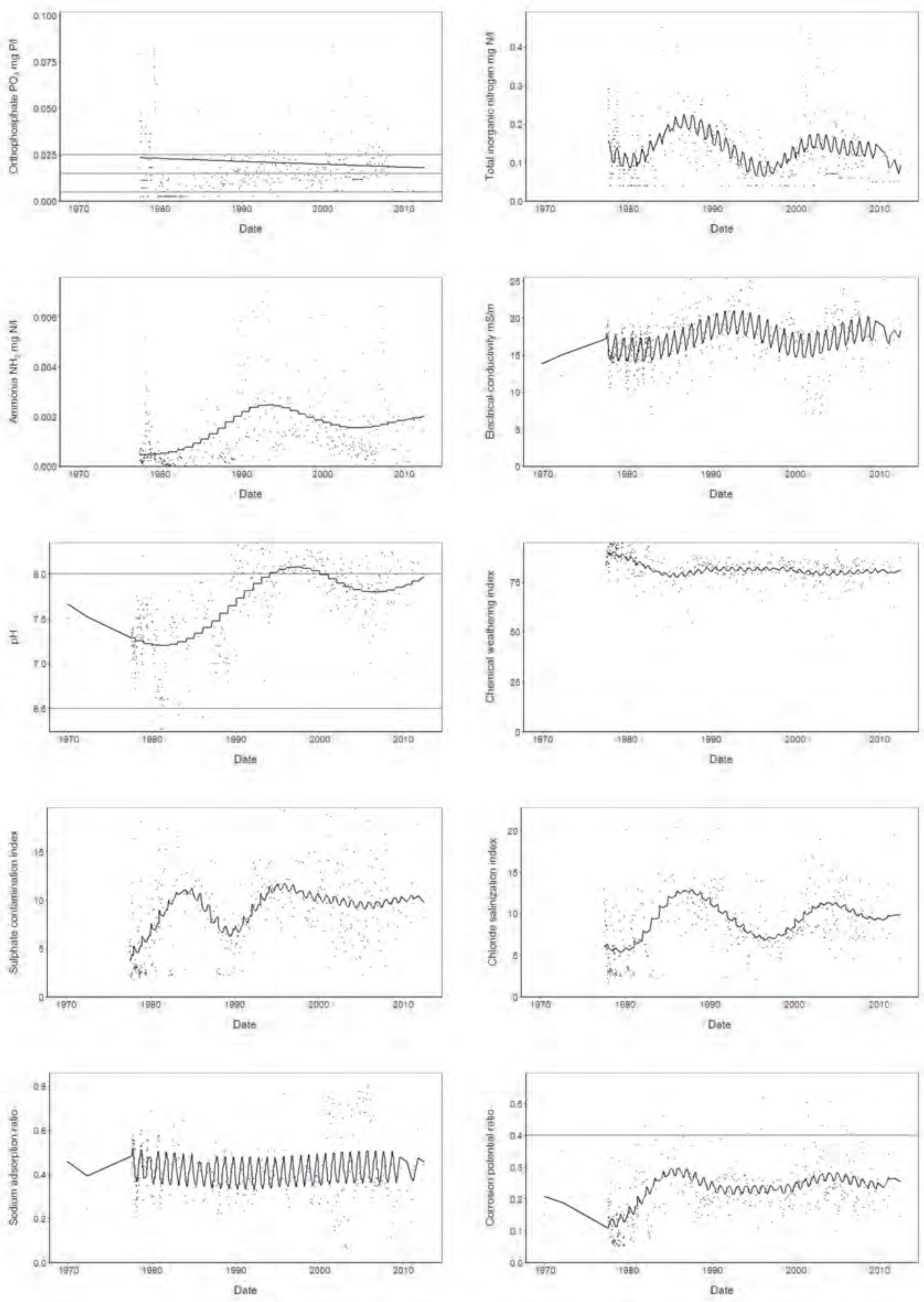

Figure 36 Temporal trends in several water quality parameters at monitoring point X2H008 at Sassenheim on the Queens River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate p=0.614; TIN p<0.001; ammonia p=0.002; EC p<0.001; pH p<0.001; chemical weathering index p<0.001; sulphate contamination index p<0.001; chloride salinization index p<0.001; SAR p=0.104; and CPR p<0.001.

Bornman's Drift (X2H031) on the South Kaap River

This site is close to the Sassenheim and is situated on the South Kaap River adjacent to the Queens River, close to the confluence of the two. The trends in data at this site may be expected to be similar to results from the Queens River site, and for the greater part, they are (Figure 37). Differences will be indicated below. The reliable data record extends from 1977 to 2008, with two samples before, and sporadic, infrequent sampling afterwards.

The GAMM model fitting algorithms was not able to converge on a suitable model for orthophosphate at this site. Inspection of the data (Figure 37) reveals the majority of samples to be in the acceptable-to-tolerable range, with no clear temporal trend over the period sampled.

Total inorganic nitrogen levels fluctuated over time, with much variation and without any clear temporal shit, and were within the ideal range. Beyond an increase since the start of sampling, un-ionized ammonia levels remained low with little temporal fluctuation.

Likewise, electrical conductivity shows a small overall increase over the first years of the data record, and thereafter some temporal variation but no overall temporal trend. The chemical weathering index indicates the ionic makeup is heavily dominated by natural processes. The latter shows a small decrease since the start of data collection, suggesting a small shift in ionic composition probably owing to anthropogenic impact.

The sulphate contamination and chloride salinization indices have similar results to those from the Queens River, though relative sulphate levels at this site are lower than those at Sassenheim.

The water from this site was suitable throughout the monitored period for use according to the results from the sodium adsorption and corrosion potential ratios. The former is low and shows a decreasing trend with time; the latter a little higher with regard to the class boundary values, but showing no sign of increasing in recent years and still largely classed as ideal.

All parameters except for orthophosphate and un-ionized ammonia showed a detectable and significant seasonal variation.

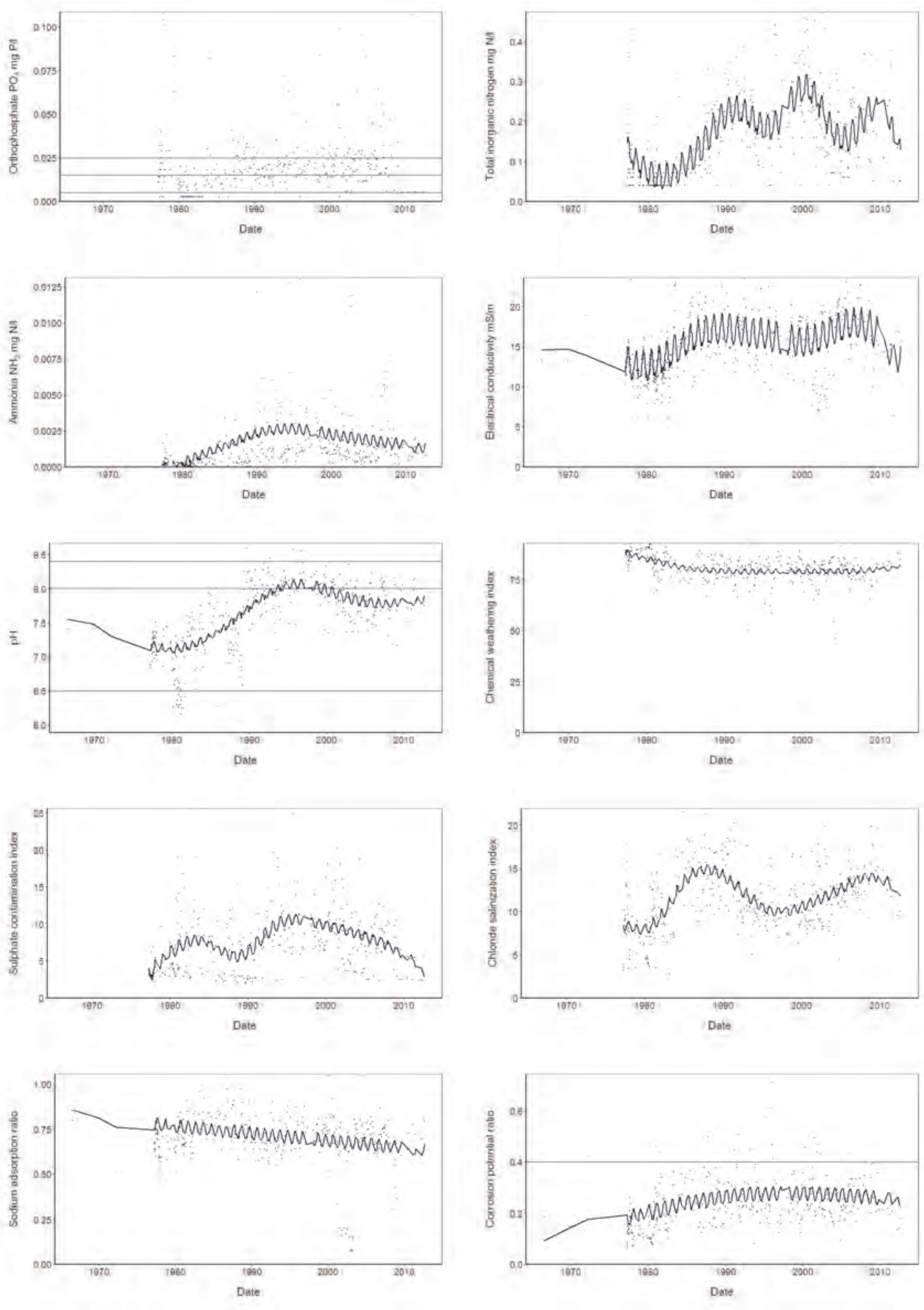

Figure 37 Temporal trends in several water quality parameters at monitoring point X2H031 at Bornmans Drift on the South Kaap River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate model non-convergence; TIN p<0.001; ammonia p<0.001; EC p<0.001; pH p<0.001; chemical weathering index p<0.001; sulphate contamination index p<0.001; chloride salinization index p<0.001; SAR p=0.002; and CPR p<0.001.

Bellevue (X2H010) on the North Kaap River

This site is located relatively high in the North Kaap River, which joins the South Kaap River to form the Kaap River, which then flows into the mid to lower Crocodile River. The water quality parameters at this site may be expected to show the same trends as those in the South Kaap and Queens River sites, and for the most part they do (Figure 38). Differences will be noted below. The data record for this site is well populated from 1978 to 2008. A few sporadic samples precede this timeframe, and samples thereafter are irregular and infrequent.

While the GAMM model fitting procedure often had difficulty with orthophosphate data, the fit for this site seems fairly good. In common with samples from the Queens and South Kaap rivers, early samples have a number of high orthophosphate readings. The scale of this peak, its rapid disappearance, and its occurrence in three high-catchment sites either indicates a sudden change in land use or management approaches, or potentially also a systematic error in the data. At this site, levels of orthophosphate after the early peak increase until approximately 2000, and decline somewhat afterwards. The majority of samples over the monitored period are classified as being acceptable or tolerable.

Total inorganic nitrogen shows a small but steady increase with time, and all samples are classified as ideal. Un-ionized ammonia levels increase until the early 1990’s and decline slightly thereafter, and are low throughout.

Electrical conductivity increases steadily with time, but levels remain very low, more so than in samples from the Queens or South Kaap Rivers. The contribution of natural weathering to the ions underlying conductivity levels is high, despite a slight decrease following the start of monitoring. Correspondingly, the contribution of chloride and sulphate to overall ionic makeup is low. Samples from this site show a slight relative increase in sulphate levels in the early 1980’s that is present in the data records from a number of other sites in this catchment. They also show a late-1980’s peak in relative chloride dominance also observed at several sites. Trends in these ions are relatively stable given temporal changes, and the relative sulphate levels seem to be decreasing in later records.

pH record show the catchment-wide trend of increasing from the early 1980’s to peak in the late 1980’s and stabilize or decrease slightly thereafter. At the end of the monitoring period, most records would be classed as ideal.

Finally, with a low (and decreasing) sodium adsorption potential, this water should be highly suitable for irrigation. The corrosion potential ratio is higher relative to the class boundary, but is relatively stable over time and the corrosion potential of most samples is low.

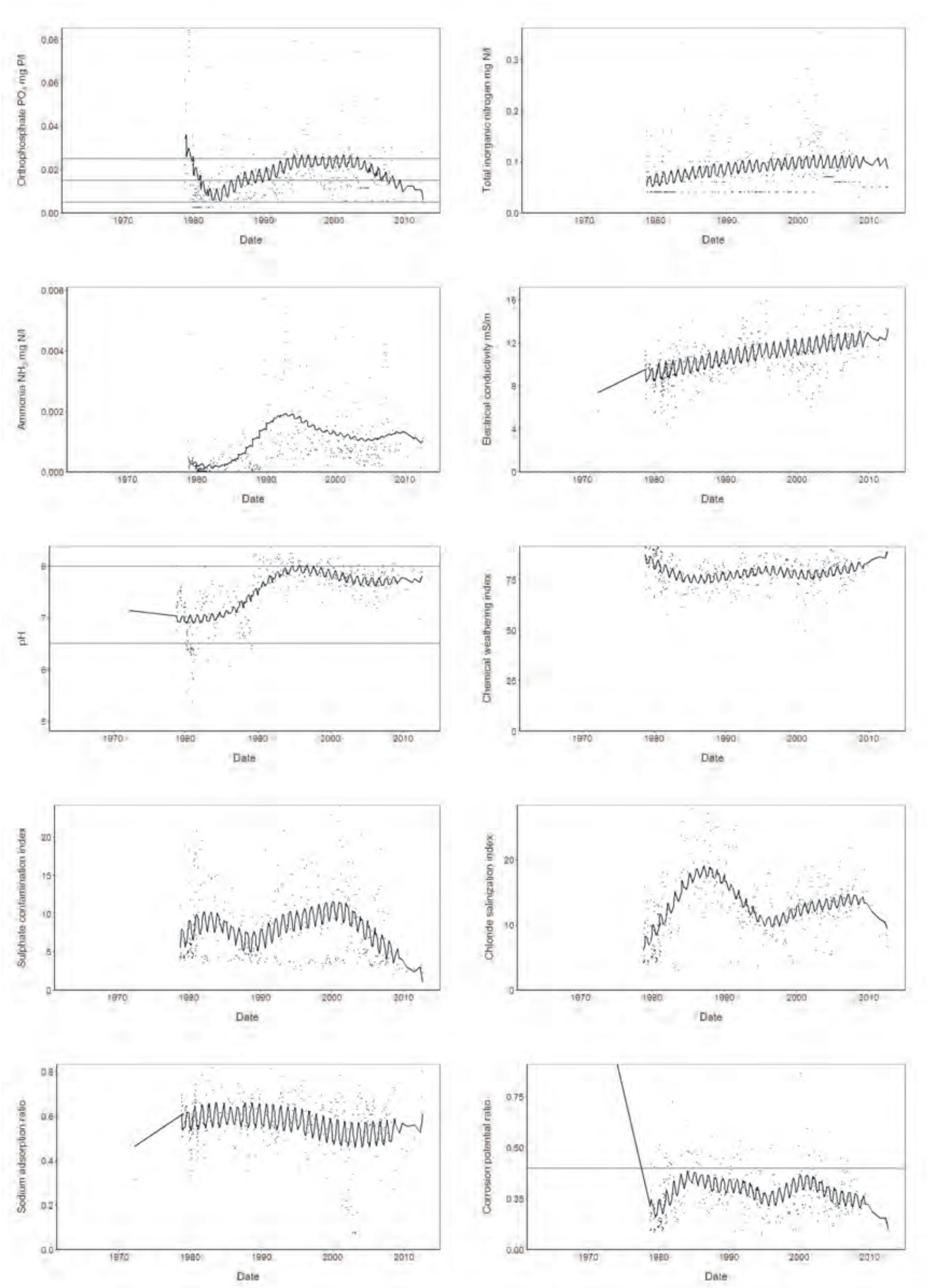

Figure 38 Temporal trends in several water quality parameters at monitoring point X2H010 at Bellevue on the North Kaap River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate p<0.001; TIN p<0.001; ammonia p<0.001; EC p<0.001; pH p<0.001; chemical weathering index p<0.001; sulphate contamination index p<0.001; chloride salinization index p<0.001; SAR p=0.006; and CPR p<0.001.

Dolton (X2H022) on the Kaap River

This site is located low in the Kaap River catchment and lies just before the confluence of the Kaap and Crocodile rivers. In contrast to sites higher in the Kaap River catchment, water quality at this site has move to the acceptable or worse class according to many of the water quality parameters assessed here (Figure 39). The data record for the site extends from 1977 to 2008, with some scattered samples prior to this stretching back to 1962, and irregular and infrequent sampling since.

At the start of sampling, orthophosphate levels are largely acceptable, but dissolved orthophosphate levels increased over the monitored period until by the end of sampling the majority of sampling have levels greater than tolerable. The curious “spike” in orthophosphate levels noted at the start of sampling at other sites in the Kaap River catchment is present here too.

Total inorganic nitrogen levels here are much higher than at sites higher in the Kaap River catchment, and show an ongoing increasing trend. Samples were split between ideal and acceptable levels at the start of sampling, but by the end the majority of samples are acceptable. This parameter shows strongly seasonal change. Increasing nitrogen is accompanied by increasing un-ionized ammonia, but levels of the latter remain classified as ideal.

Electrical conductivity levels fluctuate over the monitored period between acceptable and tolerable, and show strong seasonal variation such that a shift between classes happens on an annual basis. The variations in conductivity are not accompanied by obvious shifts in any of the indices or ratios based on relative ion concentrations, suggesting that changes in conductivity probably reflect changes in all major ions.

The pH increases from around 1980 until approximately 2000, when it stabilizes at a higher value. This trend is common at sites in the Crocodile River catchment. However, samples from the majority of sites stabilize at pH 8.0 or slightly lower, while samples from this catchment stabilize just below 8.5, with annual variation pushing upper pH limits past 8.5 and therefore beyond the range classified as acceptable.

The chemical weathering index decreases with time throughout the monitored period; however, values remain fairly high throughout and natural weathering remains the dominant source of dissolved ions in the river. The sulphate contamination index increases steadily over the monitored period such that the values encountered at the end of the monitored period are high for this catchment, and absolute sulphate concentrations reach and exceed the ideal-to-acceptable boundary according to DWA (2011a). Sulphate mobilisation and contamination is therefore a potential threat to water quality at this point. The chloride salinization index varies slightly with time but shows no increasing trend and values remain low.

The sodium adsorption ratio shows a strong pattern of seasonal change and decreases overall through the monitored period. While values were close to the ideal-to-acceptable boundary around 1980, present day values have declined and the water should be suitable for irrigation. Seasonal changes affect the water’s suitability considerably.

The corrosion potential ratio increases throughout the monitored period, largely as a function of increasing sulphate levels.

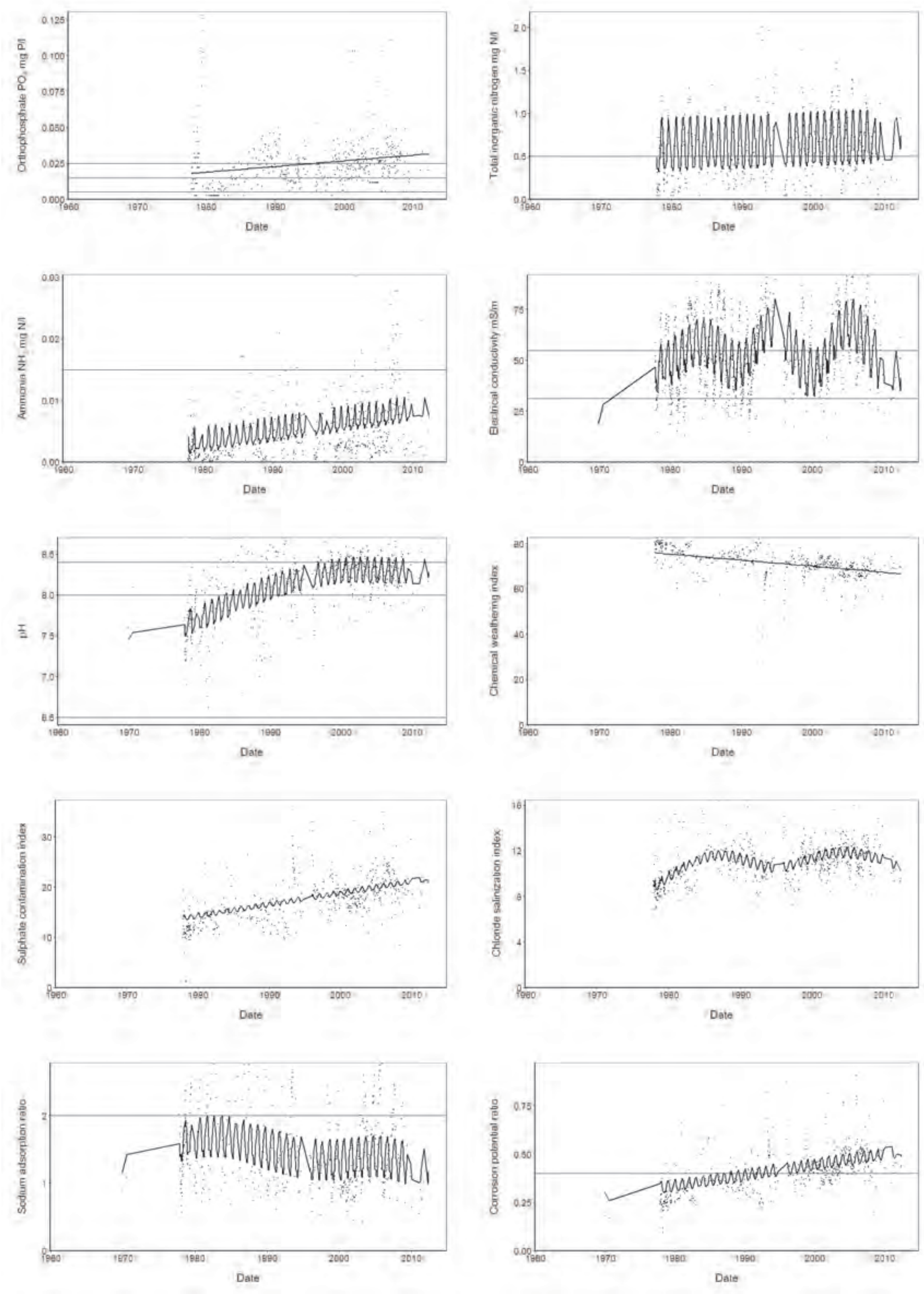

Figure 39 Temporal trends in several water quality parameters at monitoring point X2H022 at Dolton on the Kaap River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate p=0.002; TIN p=0.360; ammonia p<0.001; EC p<0.001; pH p<0.001; chemical weathering index p<0.001; sulphate contamination index p<0.001; chloride salinization index p<0.001; SAR p=0.057; and CPR p<0.001.

All parameters except orthophosphate and the sulphate contamination index showed significant seasonal change.

Thankerton (X2H017) on the Crocodile River

This site is on the Crocodile River some distance down from the confluence with the Kaap River and fairly low in the catchment. At this point, the river is flanked by the Kruger National Park on the northern bank. Despite the presence of a nature reserve, a number of the water quality parameters assessed here show disturbing trends (Figure 40). The data record stretches from 1977 to 2008, with gaps in the 1990's, and with a few scattered samples on either side of this period.

Orthophosphate levels show an increase over the period samples, from being acceptable or tolerable when monitoring started, to recent samples where very few samples are even tolerable. The GAMM curve plotted on the figure (Figure 40) suggests that the increase may be exponential rather than linear; this may be an artefact caused by the several very high values at the start of the data record. If not, this is a disturbing trend. Regardless, at the end of the sampling period, orthophosphate levels are unacceptably high.

Total inorganic nitrogen levels increase sharply from the start of monitoring from ideal to acceptable. The GAMM curve fitting procedure suggests a dip during the 1990's, but owing to a paucity of data from the 1990's, this may be an artefact. After the initial increase, then, levels remain acceptable, with much seasonal and short-term variation.

Un-ionized ammonia levels increase though the monitored period, but remain ideal.

Electrical conductivity levels fluctuate around the ideal-to-acceptable boundary, considerable seasonal and longer-term variation. Of the ions underlying conductivity levels, the relative proportion of sulphate and chloride increase over time, and the relative dominance of ions from natural weathering decreases, though these remain the dominant ions through the monitored period. Both relative chloride and sulphate levels are high for this catchment. Absolute concentrations of both remain ideal for the most part according to DWA (2011a).

The pH shows the same trend observed across the catchment, with an increase from around 1980 to around 2000, followed by stabilization, with a suggestion that overall levels may be decreasing in recent times. Overall, levels peaked between 8.0 and 8.5, which is higher than most sites in the catchment.

The water from this site is suitable for irrigation according to the sodium adsorption ratio, and no clear trends with time were noted for this index. The corrosion protection ratio, on the other hand, shows a steady increase with time with some localised fluctuation, and change the class of water from ideal to acceptable according to this index.

Seasonality is apparent in most of the parameters assessed from this site, with only orthophosphate and un-ionized ammonia not having statistically significant seasonal variation.

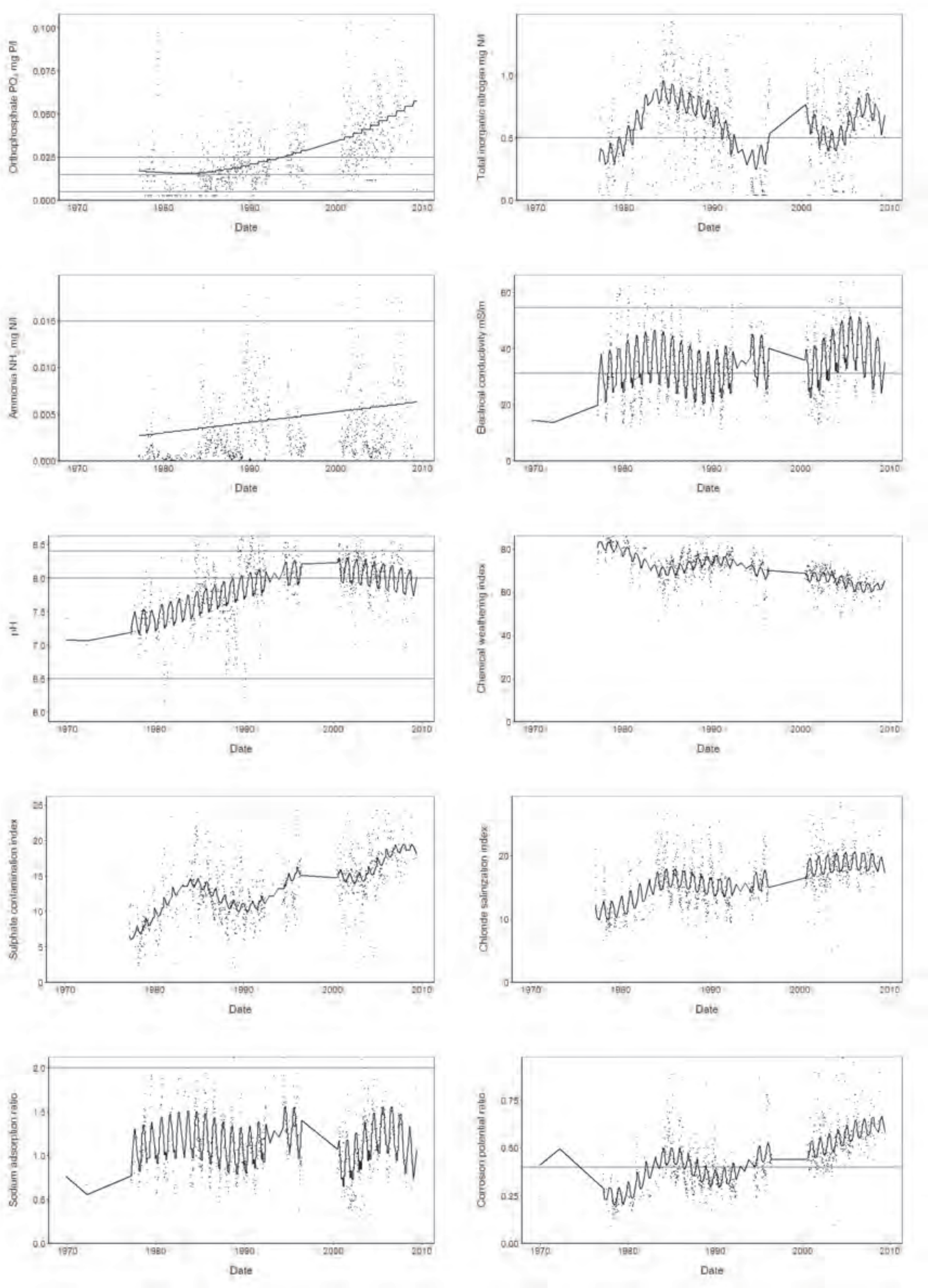

Figure 40 Temporal trends in several water quality parameters at monitoring point X2H017 at Thankerton on the Crocodile River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate $p<0.001$; TIN $p<0.001$; ammonia $p=0.361$; EC $p<0.001$; pH $p<0.001$; chemical weathering index $p<0.001$; sulphate contamination index $p<0.001$; chloride salinization index $p<0.001$; SAR $p<0.001$; and CPR $p<0.001$.

Ten Bosch (X2H016) on the Crocodile River

This is the lowest of the selected monitoring points along the Crocodile River, and the quality of water at this point represents the quality of water that will flow into the Komati River and thence to Mozambique. Like Thankerton upstream, the river at this point has the Kruger National Park on the northern bank. To a large extent, the patterns in water quality at this site reflect those upstream at Thankerton (Figure 41). The reliable data record at this point stretches from 1977 to 2008, with one sample from 1970 and infrequent, scattered samples thereafter.

In common with sites throughout this catchment (with the exception of the upper Kaap River sites), orthophosphate levels increase over the time sampled. At the start of monitoring, samples were for the most part acceptable, but by the end most samples were beyond tolerable. It is worth noting perhaps that orthophosphate levels here are improved in comparison to those from Thankerton, and some recovery would seem to have taken place between the sites.

While variation is high, total inorganic nitrogen shows a decrease with time, and at the end of sampling, the majority of samples at the end of the monitoring would be classified as ideal. Un-ionized ammonia levels vary with time and season, but with the exception of some outliers, samples remain classed as ideal.

Conductivity levels show high seasonal and short-term variation, and, a slow increasing trend across the monitored period is apparent. On average, samples remain classed as acceptable through this period, but short-term variation leads to rapid changes in class from ideal to tolerable.

The patterns in the indices reflecting relative ionic balance are similar to those at Thankerton. With, time, the influence of natural weathering in determining the ionic composition of the water decreases, but this remains the major source of ions throughout. The relative importance of chloride and sulphate increases over time with some temporal fluctuation. The absolute concentrations of sulphate remain ideal throughout the sampled period, but increases in chloride cause samples since the mid 1990’s to more frequently be classed as acceptable.

The pH shows the standard catchment-wide pattern of increasing from about 1980, then stabilizing at a higher value, here during the mid-1990’s. The final levels are fairly high though and with upper levels passing 8.4 are no longer acceptable. Then data from this site show less of a tendency to (some) recovery in comparison to other sites in this catchment.

For the first time in this catchment, sodium loads have increased to the extent that the sodium adsorption ratio passes at times beyond the bounds of ideal to become acceptable. While the water can still be used for irrigation, the occasional shift in classification highlights the ongoing degradation of water quality in this catchment. However, despite high seasonal variation medium-term fluctuation, no obvious trend to change in this parameter can be seen.

The corrosion potential index, on the other hand, shows the steady increase with time common in sites in the mid-to-lower catchment. This change is underwritten by increasing dominance of chloride and sulphate in the water. Water remains acceptable though at the end of monitoring.

Significant seasonality was detected for all water quality parameters at this site.

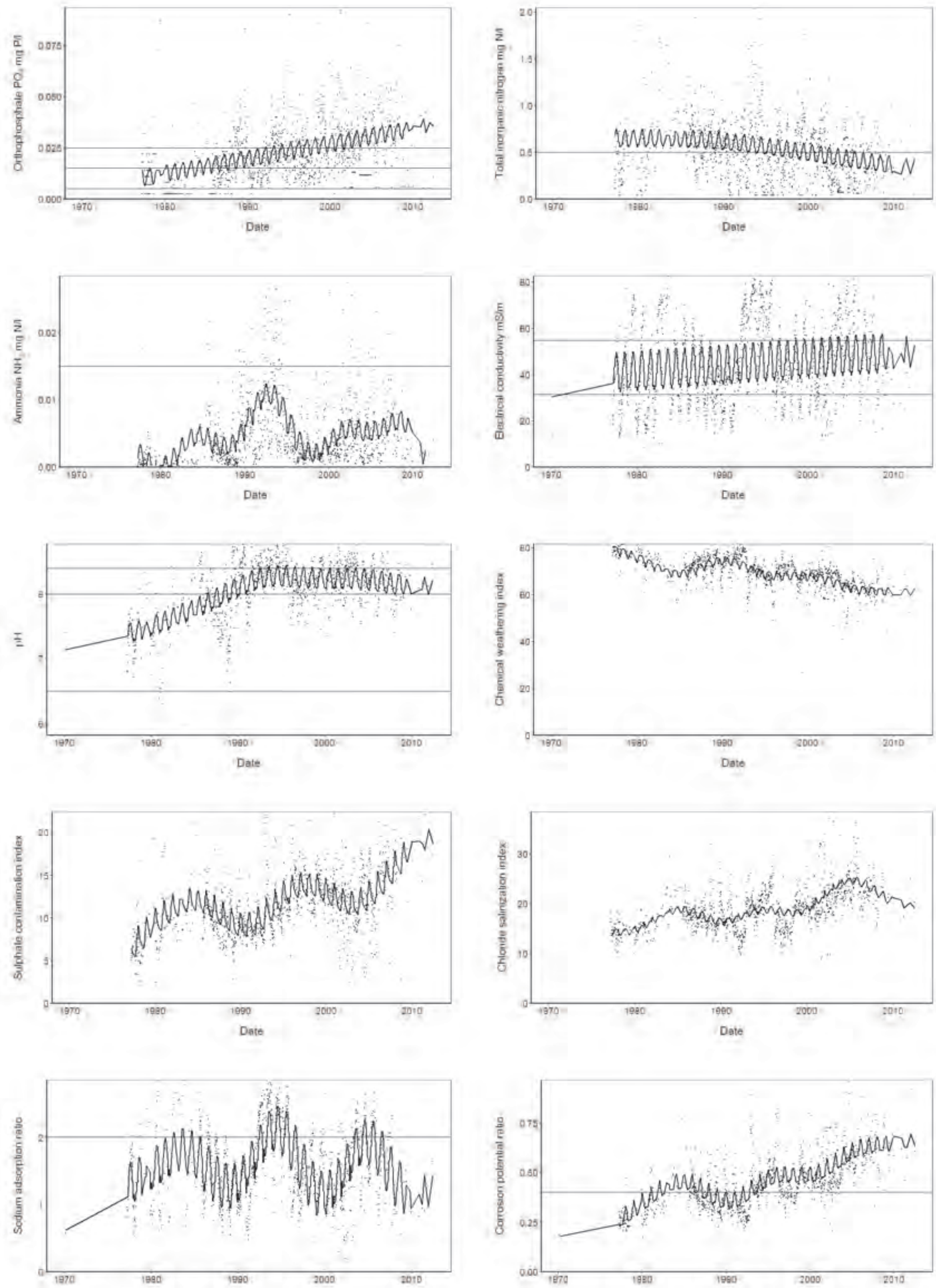

Figure 41 Temporal trends in several water quality parameters at monitoring point X2H016 at Ten Bosch on the Crocodile River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate $p<0.001$; TIN $p<0.001$; ammonia $p<0.001$; EC $p=0.005$; pH $p<0.001$; chemical weathering index $p<0.001$; sulphate contamination index $p<0.001$; chloride salinization index $p<0.001$; SAR $p<0.001$; and CPR $p<0.001$.

Komatipoort (X2H036) on the Komati River

This site was included to determine whether trends in the Crocodile catchment continue after the river has joined the Komati River. Another reason for the site's inclusion is that this is the water that passes from South Africa to Mozambique, and whose quantity and quality is governed by the Incomaputo Accord (TPTC 2002).

The trends in water quality apparent at this site have much in common with those from sites in the lower Crocodile River (Figure 42). Where mixing of the Komati and Crocodile River waters led to different temporal trends, this will be indicated below. The data record starts in 1982, and extends to the current day. As in sites in the Crocodile catchment, records after 2008 are sparse and sampling is infrequent.

As in results from the lower Crocodile, orthophosphate levels show a steady increase with time, and while most were acceptable to tolerable at the start of monitoring, by the end the majority are beyond tolerable. Simultaneously, there is a decrease in total inorganic nitrogen concentrations across the monitoring period, and by the of the data record, most samples would be classed as ideal.

Levels of un-ionized ammonia increase with time, but levels overall are ideal with few samples classed as acceptable or tolerable.

Electrical conductivity varies seasonally and over the medium term but shows no clear increasing or decreasing trend. For the most part samples would be classified as acceptable, but during peaks around 1995 and 2005 the majority of samples would be classed as tolerable; likewise, around 1988 and 2000, significant numbers of samples would be classed as ideal.

The indices indicating relative ion dominance indicate similar trends to those observed in the lower Crocodile, but change across the monitored period is lower. The chemical weathering index is lower at the start of monitoring that at lower Crocodile sites, and the change with time less, making these and the Crocodile values comparable at the end of the data record. While there is an increasing trend exhibited by the sulphate contamination index, the change is slow and values here less than those from the later part of the Crocodile data record. On the other hand, although a discernable temporal trend is difficult to identify in the face of medium-term variation, levels of the chloride salinization index from this site are greater than those from the Crocodile throughout the data record. While absolute concentrations of sulphate at this site remain ideal throughout, after 1995, half or more of samples have only acceptable chloride concentrations and a small handful are only tolerable (following DWA 2011a).

Variation and a shorter data record make it difficult to come to a firm conclusion as to whether the pH rose from 1980 to 2000 as in Crocodile samples. Certainly the pH rose overall between 1990 and 1995 to stabilize between pH 8.0 and 8.5, and in terms of the more recent data record, trends in pH in the Komati and Crocodile rivers are similar. As noted for the Crocodile, upper pH values of more than 8.4 which are common in recent years mean the condition of samples in the Komati at the border is not acceptable.

There is little to distinguish patterns in the Komati and lower Crocodile Rivers in the use indicators sodium adsorption and corrosion potential ratios. The former fluctuates considerably to match changes in electrical conductivity, but most samples would be classified as ideal. The latter shows a steady increase while remaining for the most part acceptable, though with some outliers beyond acceptable.

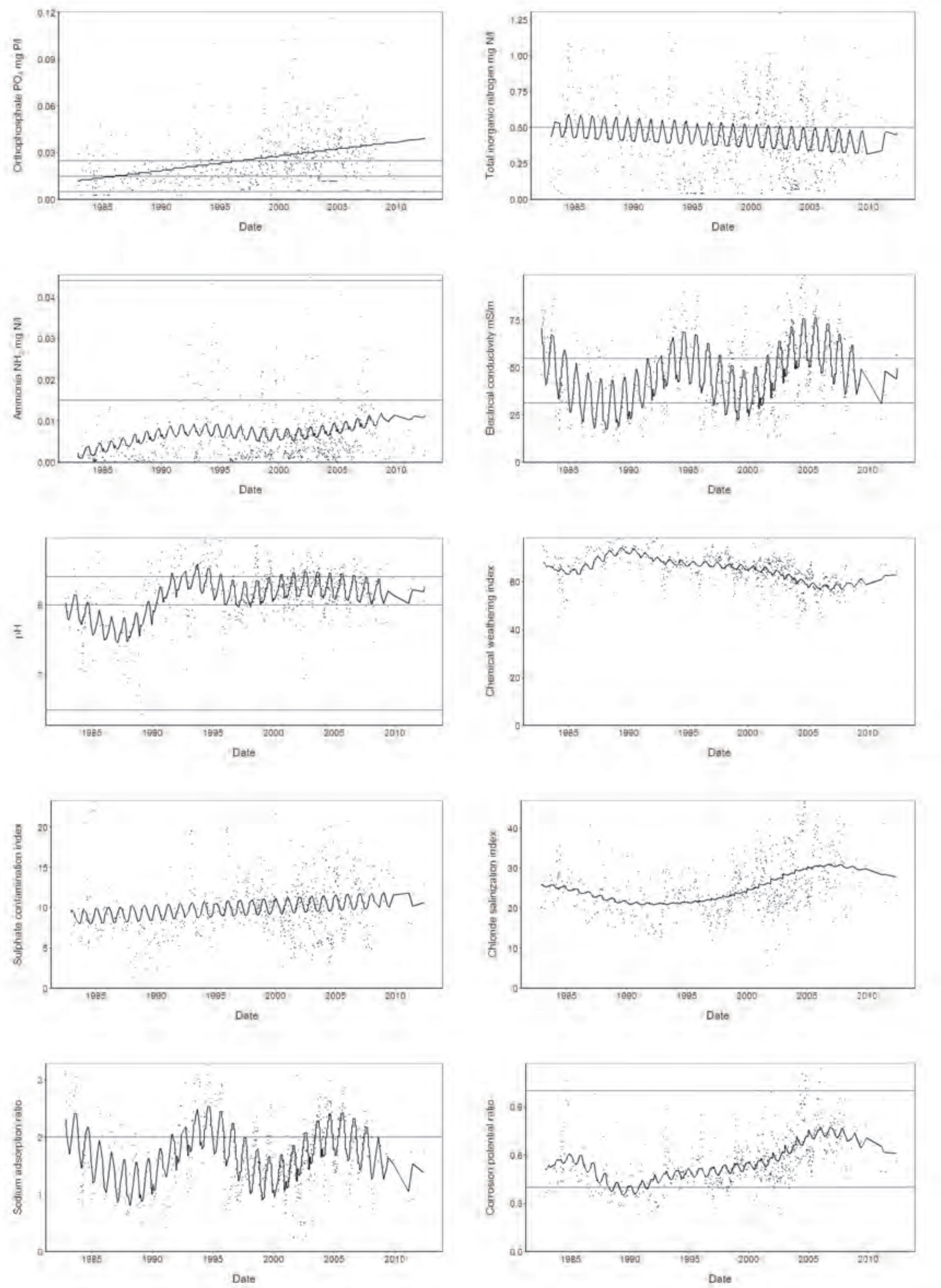

Figure 42 Temporal trends in several water quality parameters at monitoring point X2H036 at Komatipoort on the Komati River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate $p<0.001$; TIN $p=0.032$; ammonia $p=0.026$; EC $p<0.001$; pH $p<0.001$; chemical weathering index $p<0.001$; sulphate contamination index $p<0.001$; chloride salinization index $p<0.001$; SAR $p<0.001$; and CPR $p<0.001$.

All parameters excluding orthophosphate, un-ionized ammonia and the chloride salinization index showed statistically detectable and significant seasonal variation.

Temporal trends in microbial levels

Microbial data were not available for the same monitoring points as were used for other water quality parameters, and as a result these are presented separately (Figure 43). Before considering the data presented here, one needs to bear in mind that the time frame of data available is far less for microbial than physicochemical data, as available data only date back to 2004 at best in this catchment. Additionally, data are sparse making statistical analysis less practical. Here, a statistical approach was attempted where more than 100 data points were available. While curves were fitted to the data, none of these is well supported statistically, and no conclusions regarding changes with time can be supported. Significant seasonal changes were only detected at KaNyamazane.

It is also important to bear in mind the spatial location and distribution of monitoring points. Point selection criteria included the requirement that sufficient data be available for trend analysis; while, strictly speaking, no point met this criterion, points with large datasets were selected on this basis. The resulting spatial distribution of points is as a result uneven, and points are focussed around Machadodorp on the Elands River, along the midstream reaches of the Crocodile River, along the Kaap and South Kaap Rivers, and at the end of the Crocodile River at Komatipoort.

Finally, monitoring points are often located downstream of a wastewater treatment works. This makes a great deal of sense for monitoring the effects of these plants; however, the results from samples collected at these sites cannot be considered to be representative of the river as a whole.

While conclusions about temporal trends cannot be drawn from Figure 43, there are important conclusions to be drawn from the data. The major point is that levels of *E. coli* in riverine waters in this catchment are intolerably high. The guidelines used in this analysis are based on risk factors of using water for domestic use or livestock watering. The ideal class upper boundary marks where the risks of using water domestically are slight. Very few samples fall into this class. The upper bound of the acceptable class marks where significant risks are taken using the water domestically. A significant number of records fall into this class, where risks to domestic users are significant. However, a significant number of records exceed this boundary, and at the worse sites, the majority of records exceed this boundary, effectively making domestic use of water a high-risk endeavour for users. At the worse sites, significant numbers of records exceed the upper bound of the tolerable class, where a significant risk is posed to young livestock. Use of unprocessed water at the worse sites is high risk, whether it is used domestically or for watering livestock.

Spatially, sites along the South Kaap and Kaap Rivers have the best *E. coli* levels found in the catchment. Nevertheless, use of water along this stretch poses a significant risk to domestic users. Results from Montrose, in the mid-to-upper Crocodile River, are also fairly good in comparison to other results presented here. In contrast are samples from around Machadodorp (where quality seems to degrade slightly on passing the town), samples from Karino and KaNyamazane in the mid-reaches of the Crocodile River, and samples from Komatipoort at the end of the river.

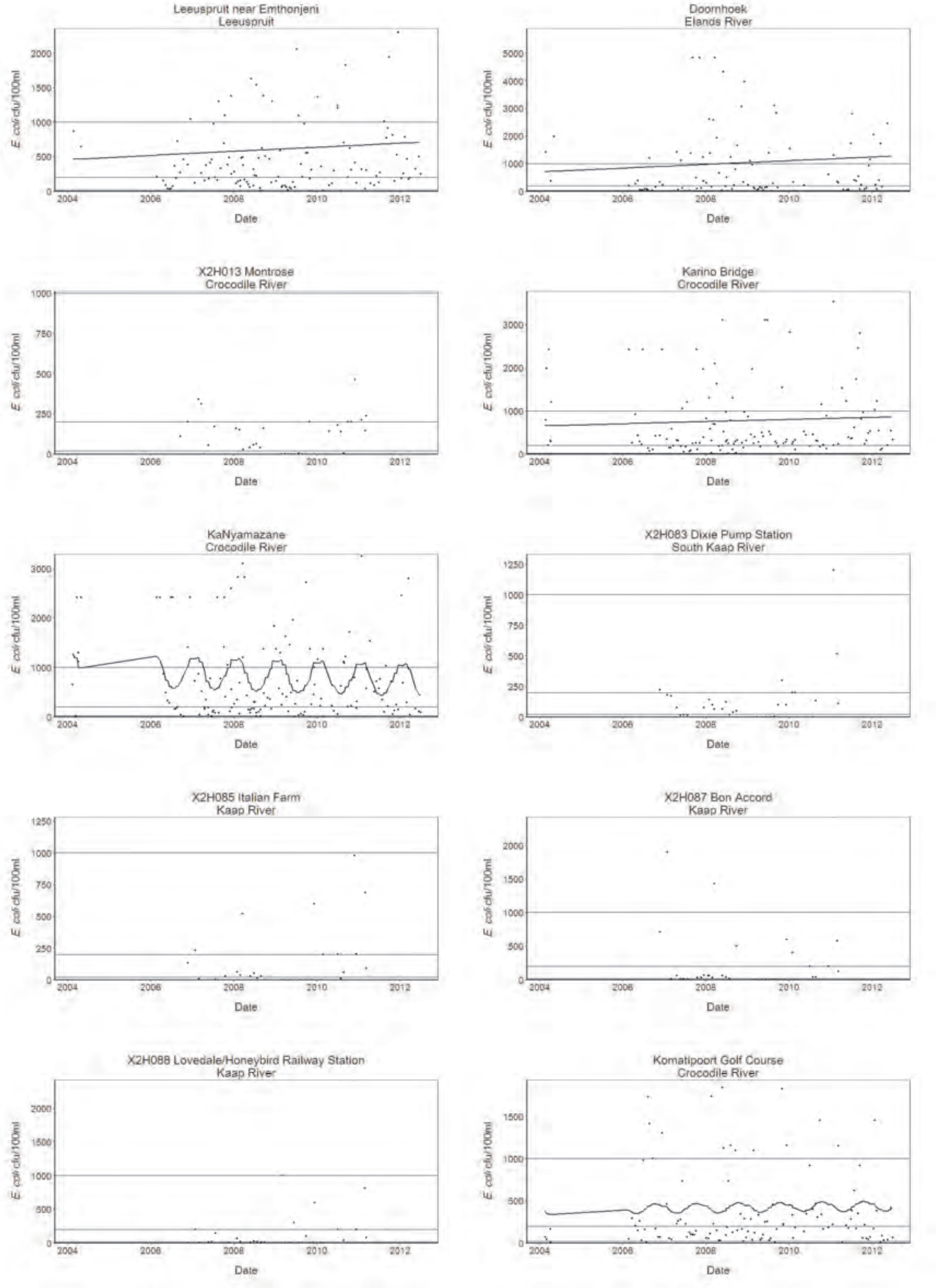

Figure 43 Temporal trends in *E. coli* colony forming units at several monitoring points in the Crocodile River catchment. Points show colony forming units against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends at sites with sufficient data were: Leeuspruit near Emthonjeni p=0.690; Doornhoek p=0.419; Karino Bridge p=0.583; KaNyamazane p=0.494; and Komatipoort Golf Course p=0.853.

Potential toxins

Summary data on a range of toxins collected from monitoring points in the Crocodile River catchment are presented in Table 12 below. Only sites with data (bar fluoride data which are widely available) are shown; similarly, only parameters with some data are presented. As is apparent from the table, there are a great number of toxins that are not monitored and the importance of these in the Crocodile River system is not known.

Prior to commenting on the data, a few points need to be made. The first is that few data are available, to the extent that only 5 of the 16 monitoring points selected for analysis had any data from the monitored period, and the datasets at these points were generally very small. A second point is that standards for comparison were taken from the South African water quality guidelines (DWAF, 1996), ideally the aquatic ecosystem guidelines. When standards were not available from this source, other guidelines were used. As the guidelines for, for example, irrigation, are drawn up with a very different end use in mind, the guideline source must be considered when assessing the results presented here.

In assessing the results presented in Table 12, the fairly conservative approach was taken that the 95th percentile of the potential toxin in question should equal or exceed the acute effect value from the guideline that was applied. This isolates data where an impact is highly probable and does not deal, for the purpose of this analysis, with chronic effects of potential toxins.

Overall, the compounds that were most likely to be associated with impacts in the catchment are aluminium (50% of sites with data), cadmium (50% of sites with data), copper (75% of sites with data), molybdenum (50% of sites with data) and zinc (50% of sites with data). The results from Table 12 suggest that lead and mercury seem also to be a problem; however, inspection of the underlying data reveals that the detection limits of the methods used for analysis as recorded in WMS data are too high to detect significant levels of these compounds. In case of mercury, all the 55 records available displayed results below the detection limit, and thus no conclusions can be drawn about mercury levels in the catchment (except that levels are not higher than 0.02 mg/ℓ, the detection limit across the dataset). In the case of lead, detection limits varied from 0.004-0.126 mg/ℓ, and detectable lead levels that exceeded these limits were occasionally found, and one can conclude that lead levels are a potential hazard at a number of sites.

Of the sites with some data, only Lindenau on the Elands River was found to have no potentially hazardous levels of the compounds that were tested for. However, the frequency of testing, and the range of compounds tested for, at this site was very low. In addition, the compounds tested for were not ones found to be problematical in this catchment and one was assessed against irrigation guidelines as not data for aquatic ecosystem health, domestic water use, etc. were available. Little can therefore be concluded as to the lack of impact at this site.

Of the remaining sites, copper was found at high levels in all Crocodile River sites. Beyond this, Montrose had high levels of aluminium and zinc, and Ten Bosch and Thankerton had high levels of cadmium. Dolton, low on the Kaap River, had elevated levels of aluminium and zinc.

Table 12 Percentiles (5^{th}-50^{th} (median)-95^{th}) of data from a range of dissolved toxins collected at a range of monitoring points in the Crocodile River catchment. Data on all known and recognised toxins in the dataset are presented here. Only monitoring points and toxins for which data are available are presented (widely collected data on fluoride levels excepted). Target water quality range, chronic effect value and acute effect value for the most sensitive criterion from the South African water quality guidelines (DWAF, 1996) are also presented.

	X2H013 Montrose Crocodile River	**X2H015 Lindenau Elands River**	**X2H016 Ten Bosch Crocodile River**	**X2H017 Thankerton Crocodile River**	**X2H022 Dolton Kaap River**	**Upper bounds (DWA, 1996) TWQR-CEV-AEV**
Aluminium (mg/ℓ)	0.07-0.5-10		0.01-0.04-0.14	0.03-0.04-0.10	0.01-0.50-0.50	0.01-0.02-0.15 [1]
Arsenic (mg/ℓ)	0.03-0.05-0.05		0.002-0.005-0.01		0.03-0.05-0.05	0.01-0.02-0.13 [1]
Barium (mg/ℓ)	0.01-0.03-0.04		0.001-0.02-0.04	0.001-0.01-0.04	0.01-0.03-0.13	
Beryllium (mg/ℓ)	0.001-0.001-0.002				0.001-0.001-0.001	0.1-0.5 [2]
Boron (mg/ℓ)	0.001-0.001-0.05		0.001-0.07-0.13	0.006-0.03-0.07	0.001-0.03-0.06	0.5-4.0-15.0 [2]
Cadmium (mg/ℓ)	0.001-0.003-0.003		0.001-0.01-0.01	0.004-0.005-0.01	0.001-0.003-0.003	0.00025-0.0005-0.006 [1]
Chrome (mg/ℓ)	0.002-0.003-0.10		0.002-0.003-0.01	0.003-0.003-0.01	0.002-0.003-0.02	0.012-0.024-0.34 (for CrIII) [1]
Chrome VI (mg/ℓ)	0.02-0.02-0.02	0.02-0.02-0.02				0.007-0.014-0.2 [1]
Cobalt (mg/ℓ)	0.01-0.01-0.01				0.01-0.01-0.01	0.05-5.0 [2]
Copper (mg/ℓ)	0.001-0.003-0.01		0.004-0.01-0.02	0.01-0.01-0.02	0.001-0.003-003	0.0008-0.0015-0.0046 [1]
Fluoride (mg/ℓ)	0.05-0.13-0.27	0.05-0.13-0.24	0.14-0.28-0.53	0.12-0.24-0.45	0.17-0.33-0.73	0.75-1.5-2.54 [1]
Iron (mg/ℓ)	0.01-0.46-68	0.03-0.18-0.37	0.003-0.01-0.06	0.003-0.01-0.06	0.002-0.10-0.35	0.1-30-3000 [3]
Lead (mg/ℓ)	0.01-0.03-0.20		0.002-0.05-0.06	0.02-0.05-0.06	0.01-0.03-0.66	0.0005-0.001-0.007 [1]
Manganese (mg/ℓ)	0.004-0.02-8.06	0.01-0.01-0.01	0.001-0.001-0.01	0.001-0.001-0.01	0.001-0.05-0.13	0.02-10 [2]
Mercury (mg/ℓ)	0.01-0.01-0.01				0.01-0.01-0.01	0.00004-0.00008-0.0017 [1]
Molybdenum (mg/ℓ)	0.003-0.003-0.01		0.003-0.02-0.03	0.01-0.02-0.02	0.003-0.003-0.01	0.01-0.02 [4]
Nickel (mg/ℓ)	0.003-0.01-0.05		0.004-0.004-0.03	0.004-0.01-0.04	0.003-0.01-0.01	0.2-2.0 [2]
Vanadium (mg/ℓ)	0.001-0.001-0.004		0.001-0.003-0.03	0.003-0.004-0.03	0.001-0.002-0.01	0.1-1.0 [2,3]
Zinc (mg/ℓ)	0.002-0.002-0.08		0.002-0.004-0.01	0.002-0.004-0.02	0.002-0.002-0.06	0.002-0.0036-0.036 [1]

South African water quality guidelines: [1] aquatic ecosystems (DWAF, 1996e); [2] irrigation (DWAF, 1996c); [3] domestic use (DWAF, 1996a); [4] livestock watering (DWAF, 1996d)

3.3.2 Olifants River

Data quality

Initial monitoring point selection for sites in the Olifants River catchment was based on, amongst other things, availability of data for trend analysis. The number of data points (post processing) per selected monitoring, and the results of data quality control checks on these datasets, is shown in Table 13.

Implementation of quality control as outlined in 3.2.7 above would reduce the size of datasets to between 15 and 86% of the original size. In the extreme, the dataset from the Klaserie River at Fleur de Lys (B7H004) would be reduced to 84 samples for further analysis. Such a decrease, together with whatever alteration of sample frequency as might occur, would have the effect that trend analysis, as undertaken in this survey, would not have the power to resolve all but highly pronounced trends in the data.

Table 13 The number of records available for analysis at monitoring points in the Olifants River catchment (following initial data processing), and the number and proportion of these that remained after data quality control checks had been applied.

Monitoring Point Name	All records	Filtered records	
B1H002 at Elandspruit on Spookspruit	1442	600	42%
B1H004 Klip Spruit at Zaaihoek	1416	591	42%
B1H005 Olifants River at Wolvekrans	989	744	75%
B1H015 Middelburg Dam on Little Olifants River: downstream	1285	1104	86%
B1H018 Olifants River at Middelkraal	661	482	73%
B1H019 Naauwpoort 335 Js on Noupoortspruit	834	684	82%
B1H020 at Vaalkranz U/S Vandyksdrift on Koringspruit	811	561	69%
B1H021 Steenkool Spruit at Middeldrift	770	594	77%
B2H007 at Waaikraal on Koffiespruit	916	577	63%
B2H014 at Onverwacht on Wilgerivier	616	359	58%
B2H015 at Zusterstroom on Wilgerivier	503	339	67%
B3H001 Olifants River at Loskop North	724	547	76%
B3H005 Moses River at Mosesriviermond	282	130	46%
B3H017 Loskop Dam on Olifants River: downstream weir	498	405	81%
B3H021 Elands River at Scherp Arabie	398	301	76%
B4H003 Steelpoort River at Buffelskloof	1274	707	55%
B4H007 Little Spekboom River at Potloodspruit	1183	209	18%
B4H011 Steelpoort River at Alverton	605	423	70%
B5H004 Flag Boshielo (Arabie) Dam on Olifants River: downstream weir	547	424	78%
B6H001 Blyde River at Willemsoord	734	264	36%
B6H004 Blyde River at Chester	1015	403	40%

Monitoring Point Name	All records	Filtered records	
B7H004 Klaserie River at Fleur De Lys	550	84	15%
B7H007 at Oxford on Olifants River	973	561	58%
B7H009 at Finale Liverpool on Olifants River	541	382	71%
B7H014 Selati River at Calais	339	137	40%
B7H015 Olifants River at Mamba/Kruger National Park	718	573	80%
B7H017 Olifants River at Balule Rest Camp/Kruger National Park	443	359	81%
B7H019 Ga-Selati River at Loole/Foskor	488	343	70%

Overview of major multivariate trends

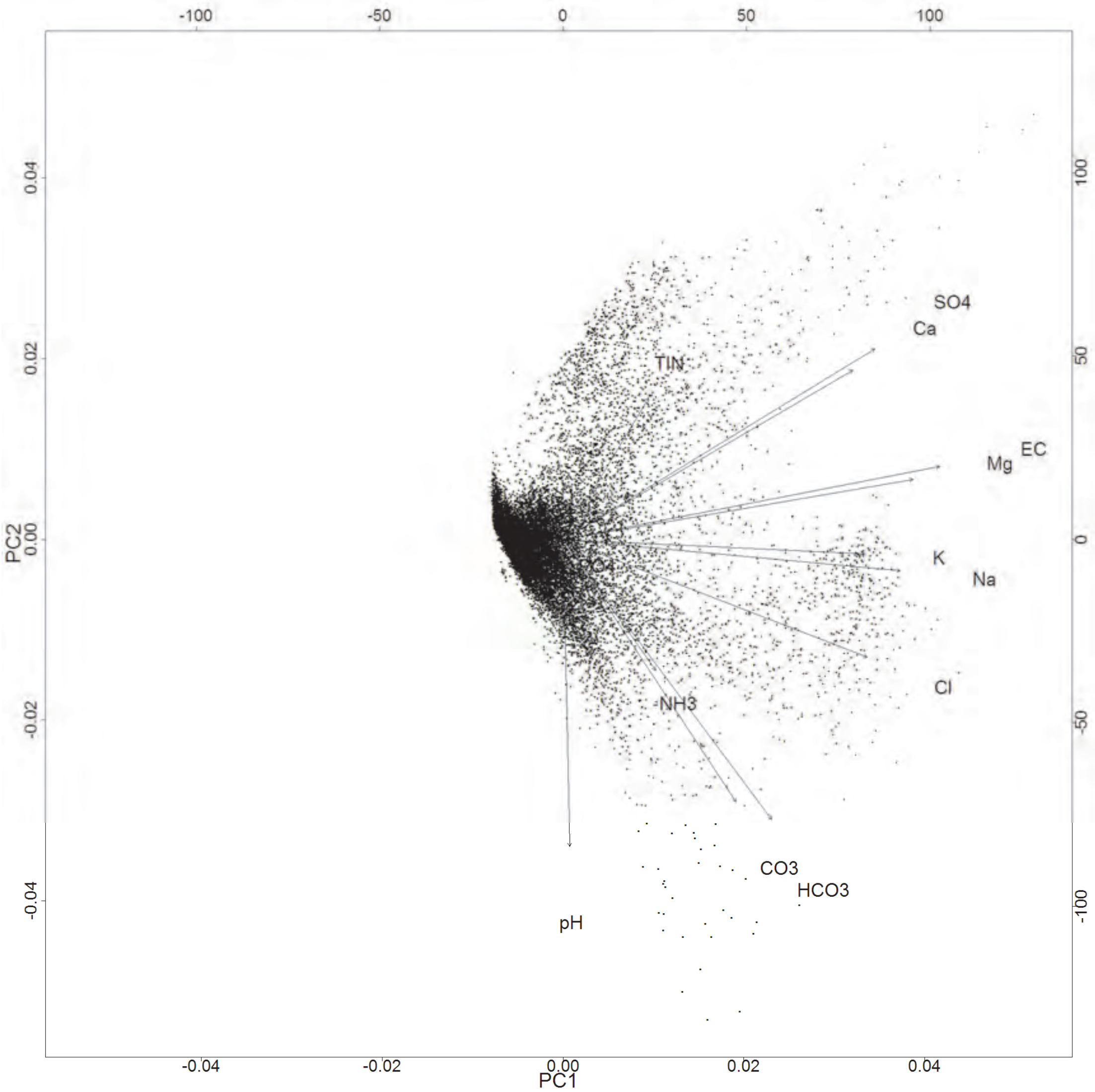

Figure 44 PCA biplot of the first two principal components from the PCA analysis of water quality parameters in the Olifants River catchment. The first two principal components together explain 61% of the variation in the data.

The PCA biplot summarizing trends in major water quality drivers is shown in Figure 44. The first principal component explains 42% of the overall variation, and most of the variation in the major cations. The second principal component contains most of the variation in pH and hardness.

Although variation in total inorganic nitrogen levels is apparent in Figure 44, much variation in this parameter is along higher level principal components not shown in the biplot. Variation in orthophosphate is barely visible in the plot, and is mostly in higher level principal components.

Overall, variations in major salts appear to underlie the majority of changes in water quality. However, changes in measured ions do not always concur and there seems to be a range of ionic compositions in water from across the Olifants catchment. Changes in major ions are largely independent of changes in pH and hardness. Finally changes in nutrients as orthophosphate and total inorganic nitrogen are independent of salt levels, pH or hardness, and also of each other. Together, this indicates a complex pattern of shifting water quality across the catchment, with much temporal and/or spatial variation.

Spatial overview of water quality

This section addresses spatial patterns over the Olifants River catchment in the various water quality parameters assessed. Temporal changes at sites are addressed below.

Orthophosphate

Figure 45 shows a spatial overview of trends in orthophosphate across the Olifants River catchment. Perhaps the most striking patterns recorded there relates to the extremely high levels of orthophosphate at two sites, Middeldrift (B1H021) on the Steenkoolspruit, and Loole/Foskor (B7H019) on the lower Ga-Selati River, with the latter having the highest levels. Both of these sites are associated with mining impacts. The Steenkoolspruit is a tributary of the Olifants River in the far upper catchment, and, as its name suggests, drains an area containing a number of coal mines. It also has Kriel and Duvha coal-fired power stations, with a combined installed capacity of 6600 MW. Further north and downstream, the Ga-Selati is a tributary of the Olifants River that joins the Olifants shortly before the river enters the Kruger National Park. The Ga-Selati River passes the copper and phosphate mining complex at Phalaborwa just before Loole/Foskor.

No other sites in the upper catchment have orthophosphate levels that approach those at Middledrift, although some have relatively high orthophosphate levels compared to others in the region. For example, Middelkraal (B1H018), the uppermost site on the Olifants River, and Vaalkrantz (B1H020) on the Koringspruit, both have median phosphate levels that are only tolerable, and 75th percentiles that are not tolerable. Orthophosphate levels at Wolvekrans (B1H005) on the Olifants River downstream of the confluence with the Steenkoolspruit are much lower than those in the Steenkoolspruit itself, though levels (and peaks especially) are moderately high for the area. However, sites on the Olifants River in the lower catchment downstream of the confluence with the Ga-Selati do have visibly elevated orthophosphate levels. Other sites with elevated orthophosphate levels include Scherp Arabie (B3H021) low on the Elands River, below several settlements and irrigated agriculture, and Fleur de Lys (B7H004) on the Klaserie River, also downstream of settlements and some irrigated agriculture.

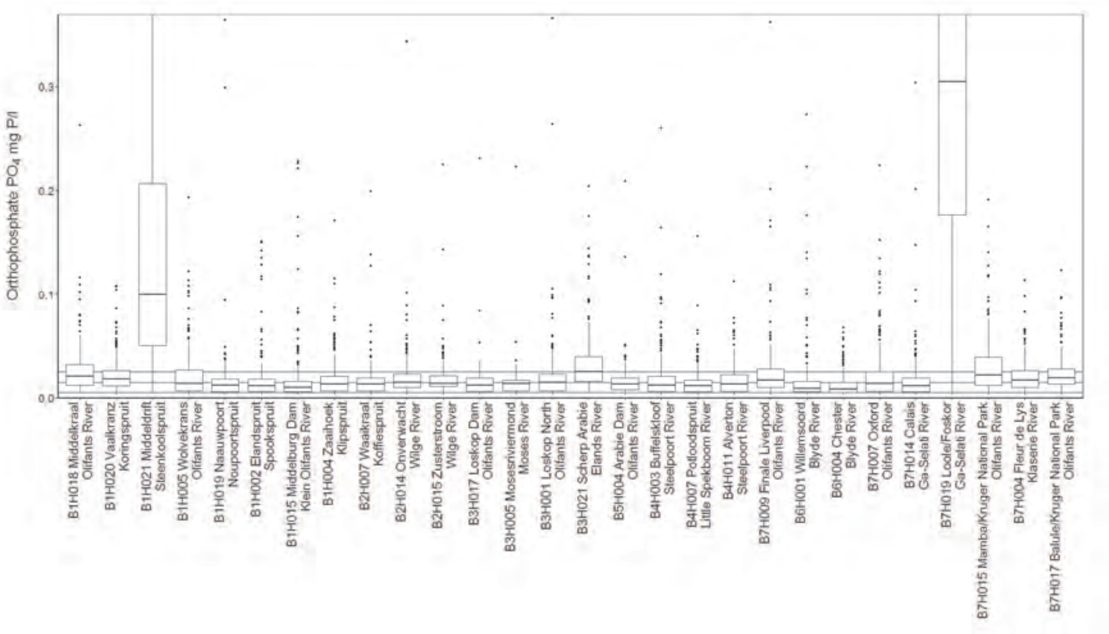

Figure 45 Box and whisker plots of orthophosphate levels at sites in the Olifants River catchment. Boxes show the interquartile range, with medians indicated. Whiskers show levels within 1.5 times the interquartile range, and dots show outliers. The upper 2% of data are not plotted. Sites are roughly ordered along the axis by distance upstream, with those higher in the catchment to the left. Horizontal lines show guidelines adopted for this analysis.

All other sites had median orthophosphate levels in the acceptable-to-tolerable range, and, with a few exceptions, 75th percentiles also fell into this range. No site in the catchment had a median orthophosphate level that fell into the ideal range.

Total inorganic nitrogen

Box and whisker plots of temporally aggregated levels of total inorganic nitrogen in the Olifants River catchment are shown in Figure 46. The majority of sites have median nitrogen levels well within the ideal range, and in most cases, even the 75th percentile is ideal.

The site with the highest median inorganic nitrogen levels is Zaaihoek (B1H004) on the Klipspruit. This site also has much variation over time in inorganic nitrogen levels. Another site with less high overall but variable inorganic nitrogen is Mosesriviermond (B3H005) on the Moses River, which is downstream of settlements and irrigated agriculture. Other sites with median inorganic nitrogen levels over the ideal range include Waaikraal (B2H007) on the Koffiespruit (agriculture and settlements), Alverton (B4H011) on the Steelpoort River (agriculture, mining and settlements), and Loole/Foskor (B7H019) on the Ga-Selati River (mining and settlements).

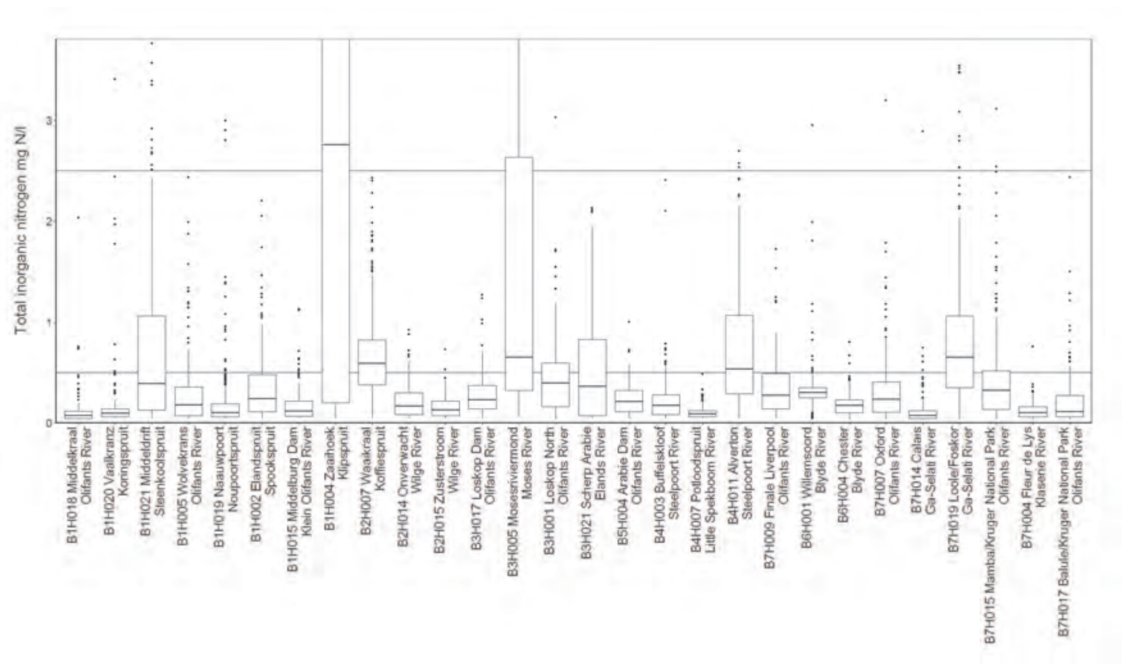

Figure 46 Box and whisker plots of total inorganic nitrogen levels at sites in the Olifants River catchment. Boxes show the interquartile range, with medians indicated. Whiskers show levels within 1.5 times the interquartile range, and dots show outliers. The upper 2% of data are not plotted. Sites are roughly ordered along the axis by distance upstream, with those higher in the catchment to the left. Horizontal lines show guidelines adopted for this analysis.

Un-ionized ammonia

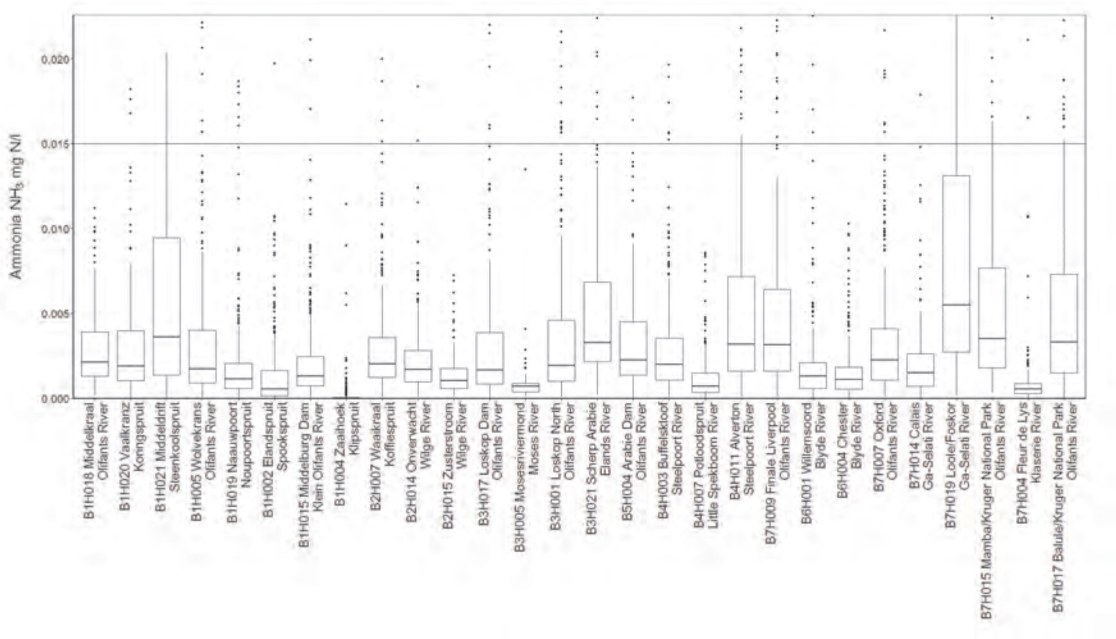

Figure 47 Box and whisker plots of un-ionized ammonia levels at sites in the Olifants River catchment. Boxes show the interquartile range, with medians indicated. Whiskers show levels within 1.5 times the interquartile range, and dots show outliers. The upper 2% of data are not plotted. Sites are roughly ordered along the axis by distance upstream, with those higher in the catchment to the left. Horizontal lines show guidelines adopted for this analysis.

Levels of un-ionized ammonia at sites throughout the Olifants River catchment are presented in Figure 47. Median and 75th percentile un-ionized ammonia levels at all sites fell into the ideal range, though nearly all sites had outliers above the ideal level. Although levels of ammonia were for the most part low, several sites had somewhat elevated levels. The highest levels were found at Middeldrift (B1H021) on the Steenkoolspruit and at Loole/Foskor (B7H019) on the Ga-Selati River. Both of these sites had greatly elevated levels of orthophosphate, and, to a lesser extent, of total inorganic nitrogen. Alverton (B4H011) on the Steelpoort River and the first site downstream of the Steelpoort confluence, Finale Liverpool (B7H009) on the Olifants River, both had elevated levels of ammonia, along with a number of sites from the far upper catchment, sites on the Olifants Elands Rivers north of Loskop Dam, and Olifants River sites downstream of the confluence with the Ga-Selati River.

Electrical conductivity

Spatial variation in electrical conductivity levels in rivers in the Olifants River catchment are presented in Figure 48. Relatively few sites had median conductivities in the ideal range, and of these, none were on the Olifants River itself. Sites with ideal conductivities were on the Koffiespruit, the Wilge, Blyde and Klaserie Rivers, and upstream in the Steelpoort and Ga-Selati River catchments.

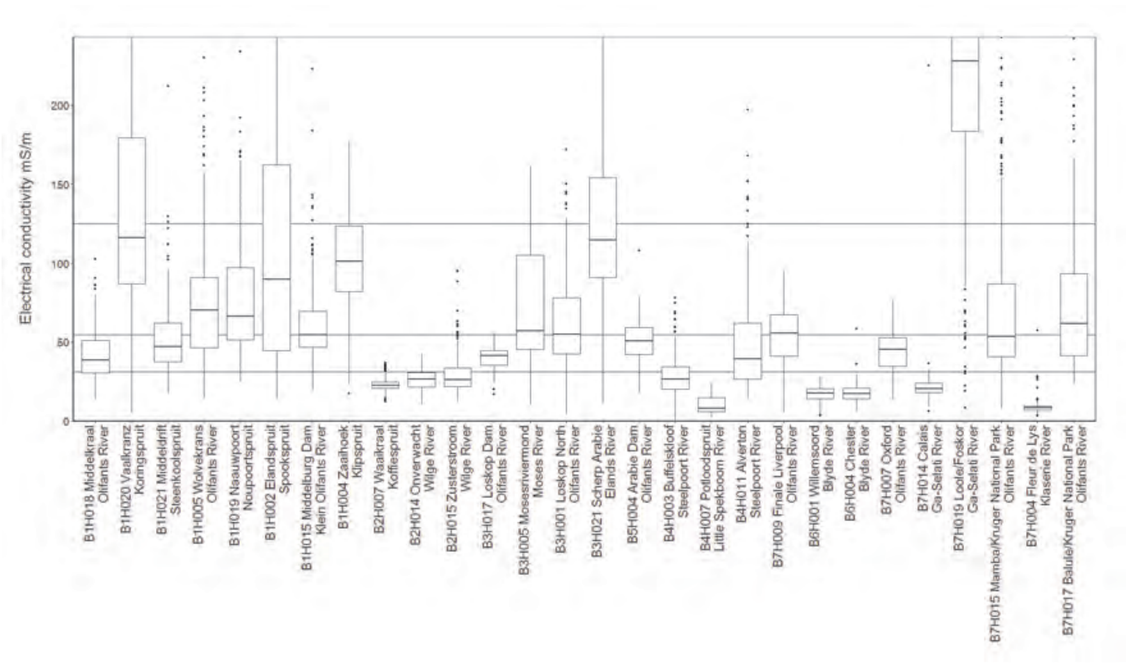

Figure 48 Box and whisker plots of electrical conductivities of water at sites in the Olifants River catchment. Boxes show the interquartile range, with medians indicated. Whiskers show levels within 1.5 times the interquartile range, and dots show outliers. The upper 2% of data are not plotted. Sites are roughly ordered along the axis by distance upstream, with those higher in the catchment to the left. Horizontal lines show guidelines adopted for this analysis.

Only one site had median conductivity that was worse than tolerable, and this site was downstream on the Ga-Selati River at Loole/Foskor (B7H019). This correlates with the findings of several other parameters assessed in this analysis, and highlights the impact of the mining complex and settlements around Phalaborwa. Sites on the Olifants River downstream of the confluence with the Ga-Selati River have electrical conductivities in the acceptable-to-tolerable range and it seems likely that inflow from the Ga-Selati maintains salinity levels in this region.

Sites in the upper catchment are located in an area with much mining, and in particular coal mining, activity, and all sites in that region have slightly to seriously elevated conductivities.

The conductivity of the Olifants River increases after Loskop Dam until the river leaves South Africa. Along this stretch, all sites are on the acceptable-to-tolerable boundary. Inputs from tributaries along this length vary considerable in salinity.

The central region of the catchment has fairly high conductivity, and inputs from the Moses and Elands Rivers are among the most saline of tributaries in this catchment.

pH

Box and whisker plots of temporally aggregated pH data from a range of sites in the Olifants River catchment are presented in Figure 49. Sites for the most part are largely within the ideal range for most of the time, with most problems coming from pHs that rise from ideal to acceptable. The one notable exception, where serious acidification is present, is found at Zaaihoek (B1H004) on the Klipspruit. This small river drains Ferrobank to the west of Emalahleni and, particularly along the Blesbokspruit, shows signs of acidification very early on. That this is the only site of those assessed in the upper catchment that shows signs of severe acidification, despite the extent of coal mining in this region, is positive.

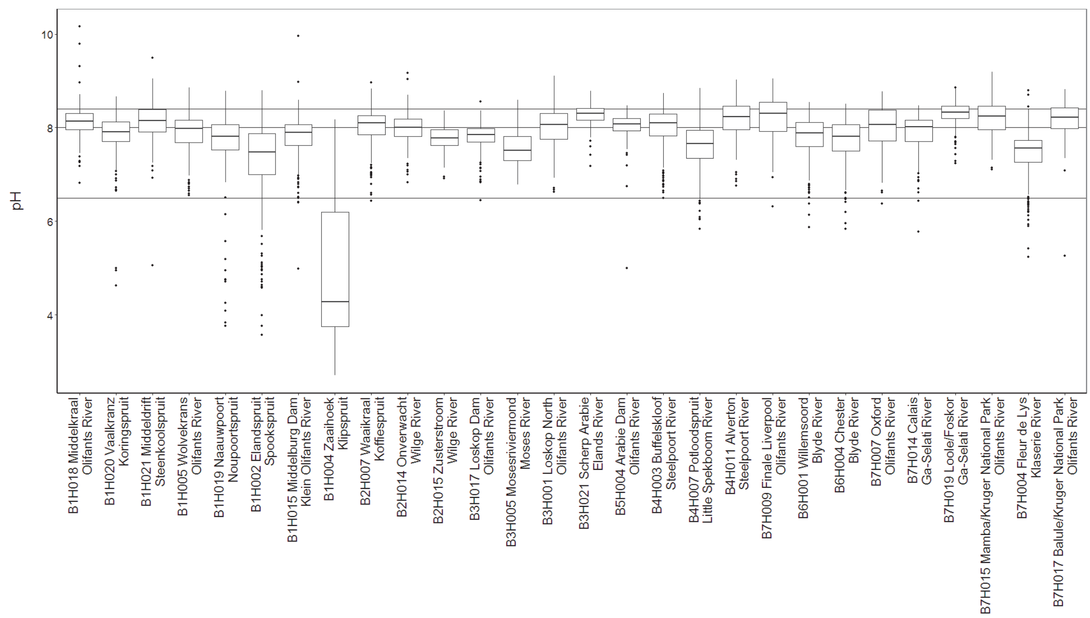

Figure 49 Box and whisker plots of pH levels at sites in the Olifants River catchment. Boxes show the interquartile range, with medians indicated. Whiskers show levels within 1.5 times the interquartile range, and dots show outliers. Sites are roughly ordered along the axis by distance upstream, with those higher in the catchment to the left. Horizontal lines show guidelines adapted for this analysis.

The other trend in pH apparent in Figure 49 is the alkalinity of upper and occasionally lower pH values at a number of sites. Notable examples include Alverton (B4H011) in the lower Steelpoort River and Finale Liverpool (B7H009), the first site downstream of the confluence of the Steelpoort and Olifants Rivers, as well as Loole/Foskor (B7H019) in the lower Ga-Selati River, and Mamba/Kruger National Park (B7H015), the first site downstream of the confluence of the Ga-Selati and Olifants Rivers. The spatial arrangement of these sites suggests that input from tributaries is having an effect on water quality in the Olifants River (although this effect is not clear in the case of low pH water from the Klipspruit).

Chemical weathering index

Temporally aggregated spatial patterns in results of the chemical weather index across sites in the Olifants River catchment are presented in Figure 50. This index indicates what proportional of certain major ions derive from natural chemical weathering of rock, and a higher score indicates an ionic complement largely due to natural processes. Examination of Figure 50 reveals a high degree of spatial variation in index scores, with median scores ranging from 1 to 86. The former occurs at Zaaihoek (B1H004) on the Klipspruit, the same site where notable acidification was found above. The latter is from Calais (B7H014), the upstream site on the Ga-Selati River. This can be contrasted with a median score of 20 from Loole/Foskor (B7H019), about 66 km further downstream on the same river.

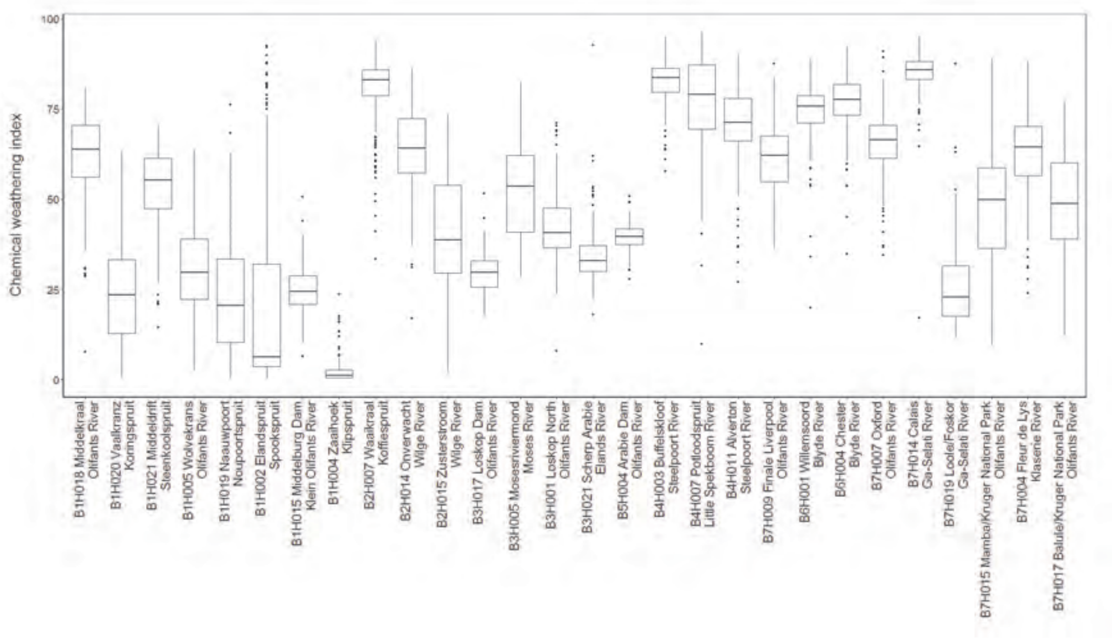

Figure 50 Box and whisker plots of the chemical weathering index proposed by Huizenga *et al.* (2013) at sites in the Olifants River catchment. Boxes show the interquartile range, with medians indicated. Whiskers show levels within 1.5 times the interquartile range, and dots show outliers. Sites are roughly ordered along the axis by distance upstream, with those higher in the catchment to the left.

As noted above, there is a great deal of variation in results of the chemical weather index from this dataset. Some general trends are that sites from the upper catchment generally have low index scores, and with two exceptions (Middelkraal (B1H018) in the Olifants River and Middeldrift (B1H021) in the Steenkoolspruit), median scores are in the early 30's or below. Scores along the Wilge River start fairly high in the upper catchment but degrade with distance downstream. Scores along the upper and mid Olifants River are fairly low, but increase after the Loskop Dam to peak around Oxford (B7H007) in the mid-to-lower catchment. Inputs from low-to-mid catchment tributaries have low scores and likely do not contribute to recovery in the mid-Olifants River. Scores from the Steelpoort River catchment are fairly high, as are those from sites in the Blyde River catchment. Finally, scores from the lower Ga-Selati River are low, and the input of this tributary has a distinct impact on scores in the lower Olifants River.

Sulphate contamination index

Temporally aggregated spatial changes in the sulphate contamination index are presented in Figure 51 below. Results cover a wide range of possible values and show sites with natural or near natural sulphate levels as well as sites where sulphate ions heavily dominate the ion complement of samples. Median sulphate contamination index scores range from 2 to a rather extreme 90. The former is from Calais (B7H007) on the upper Ga-Selati River, and the latter is from Elandspruit (B1H002) on the Spookspruit. These scores translate to sulphate levels ranging from ideal to beyond tolerable with a median of 765 mg/ℓ following the generic RQO’s of DWA (2011a).

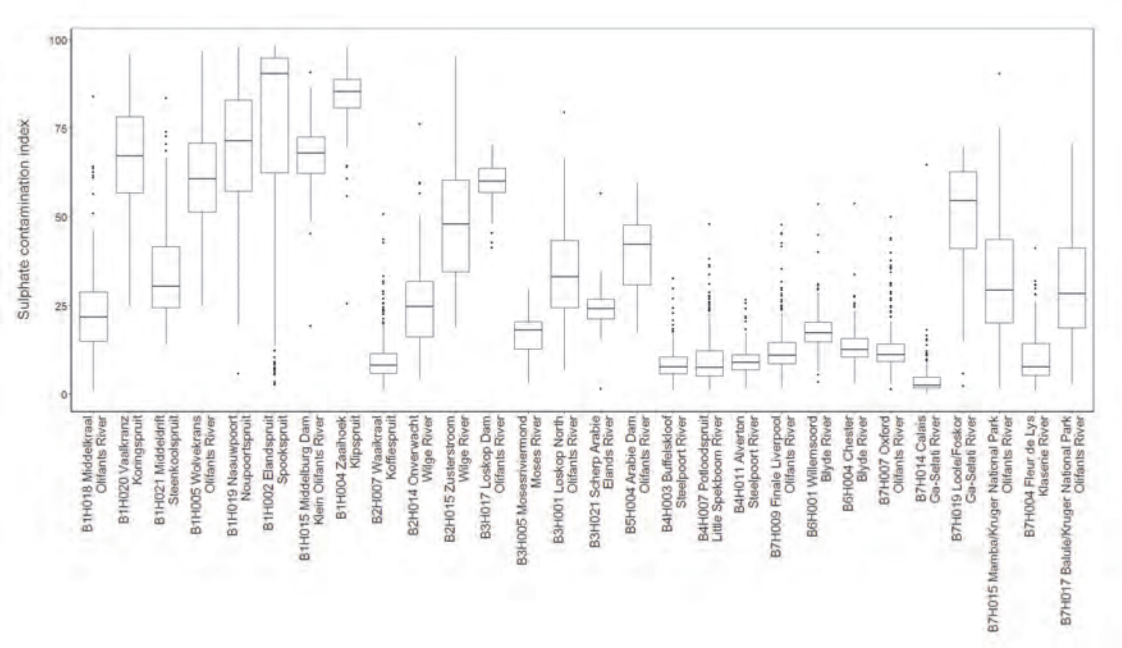

Figure 51 Box and whisker plots of the sulphate contamination index proposed by Huizenga *et al.* (2013) at sites in the Olifants River catchment. Boxes show the interquartile range, with medians indicated. Whiskers show levels within 1.5 times the interquartile range, and dots show outliers. Sites are roughly ordered along the axis by distance upstream, with those higher in the catchment to the left.

The upper catchment has the highest levels of sulphate contamination overall. This is not surprising given the extent of coal mining in this area. Two sites in the upper catchment, Middelkraal (B1H018) in the Olifants River and Middeldrift (B1H021) in the Steenkoolspruit, have relatively low scores for this area. These are the same two sites that had better chemical weathering index scores. Scores along the Wilge River start fairly low in the upstream reaches, but increase with distance downstream. Despite these increased scores, downstream scores in the Wilge River do not approach the levels found in most upper catchment sites. Scores along the Olifants River from Loskop Dam (B3H017) to Oxford (B7H007) in the mid-to-lower catchment show a decreasing trend. Along this stretch of the Olifants, inputs from tributaries indicate moderate to low sulphate contamination, with the highest median score being 24. Inputs from the Steelpoort and Blyde Rivers in particular have low scores. However, inputs from the Ga-Selati River, with a median score of 60 and median sulphate concentrations of 765 mg/ℓ (the upper bound of tolerable being 250 mg/ℓ) lead to increase index scores in the lower Olifants River.

Chloride salinization index

Spatial patterns in temporally aggregated data on chloride salinization index scores from sites across the Olifants River catchment are presented in Figure 52. The range of scores encountered is far less than that found for either the chemical weather index or the sulphate contamination index. The scores encountered here are matched by absolute chloride levels, assessment of which indicates that most sites have ideal chloride levels, and that only two sites have levels have chloride levels that are beyond acceptable (following DWA 2011a). The upper bound of tolerable chloride is 175 mg/ℓ (DWA 2011a). Scherp Arabie (B3H021) on the Elands River has median chloride levels of 177 mg/ℓ, while Loole/Foskor (B7H019) on the Ga-Selati River has median levels of 197 mg/ℓ.

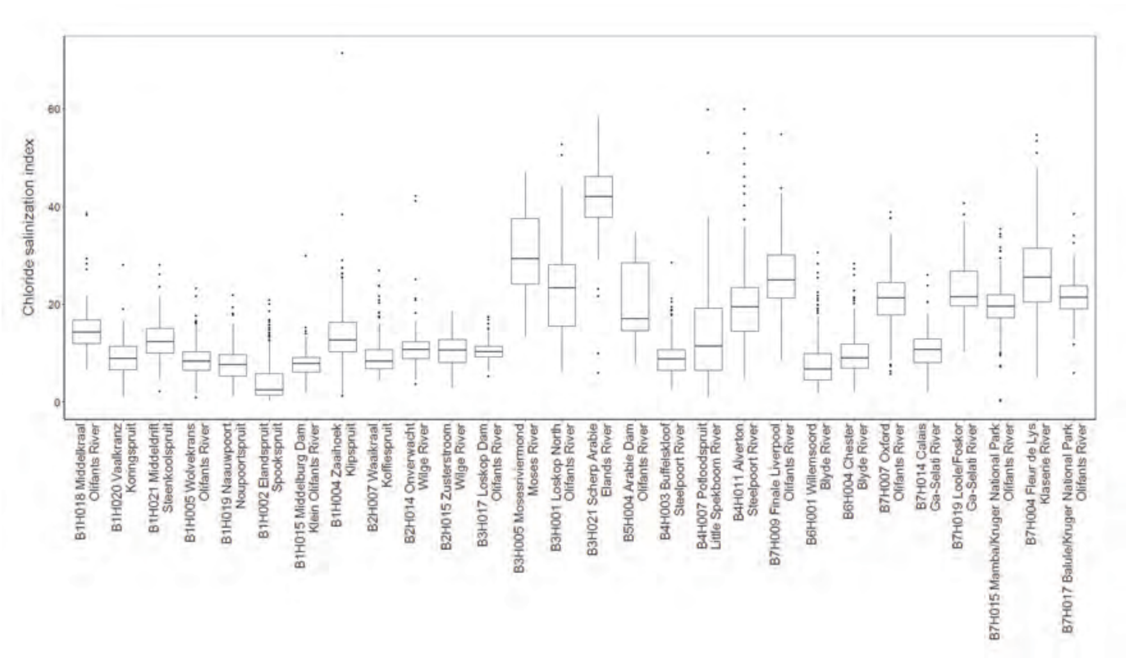

Figure 52 Box and whisker plots of the chloride salinization index proposed by Huizenga *et al.* (2013) at sites in the Olifants River catchment. Boxes show the interquartile range, with medians indicated. Whiskers show levels within 1.5 times the interquartile range, and dots show outliers. Sites are roughly ordered along the axis by distance upstream, with those higher in the catchment to the left.

General trends across the catchment include the observation that scores at upper catchment sites are relatively low. This is in contrast to results from the sulphate contamination index above. Although sulphate levels are used in the calculation of the chloride salinization index, and inflated levels of sulphate might depress this index, inspection of chloride levels in the upper catchment reveal them to be ideal (following DWA 2011a). Chloride salinization scores are also low in the Wilge River, and in the Olifants River until Loskop Dam (B3H017).

Beyond this, there is a tendency for higher index scores in the Olifants and, in particular, the mid catchment tributaries. Both the Moses River and the Elands River have high salinization scores; in the former this translates to acceptable chloride levels, in the latter, median chloride levels are beyond tolerable. The difference in the scaling between the relative index and the absolute ionic levels is a function of the index being a relative measure, and the differing salinity (measured as electrical conductivity) between the two sites.

Tributaries below the once just mentioned and above the Ga-Selati River end to have lower index scores than the Olifants river mainstream. The Blyde River has particularly low scores and these change only slightly along the length of the river; the Steelpoort River shows increased scores along its length.

As with all indices considered, there is a profound change along the length of the Ga-Selati River. The Klaserie River also has high scores for this index. None of these inputs from tributaries have a pronounced effect on the Olifants River scores, which remain fairly stable after Loskop North (B3H001).

Sodium adsorption ratio

Spatial changes in temporally aggregated data on the sodium adsorption ratio at a range of sites across the Olifants River catchment are shown in Figure 53. All sites fall in the ideal to acceptable range for this parameter. This is despite electrical conductivities for a number of sites falling into the tolerable or worse class. This appears to be a function of conductivity changes not being associated with high relative sodium loads.

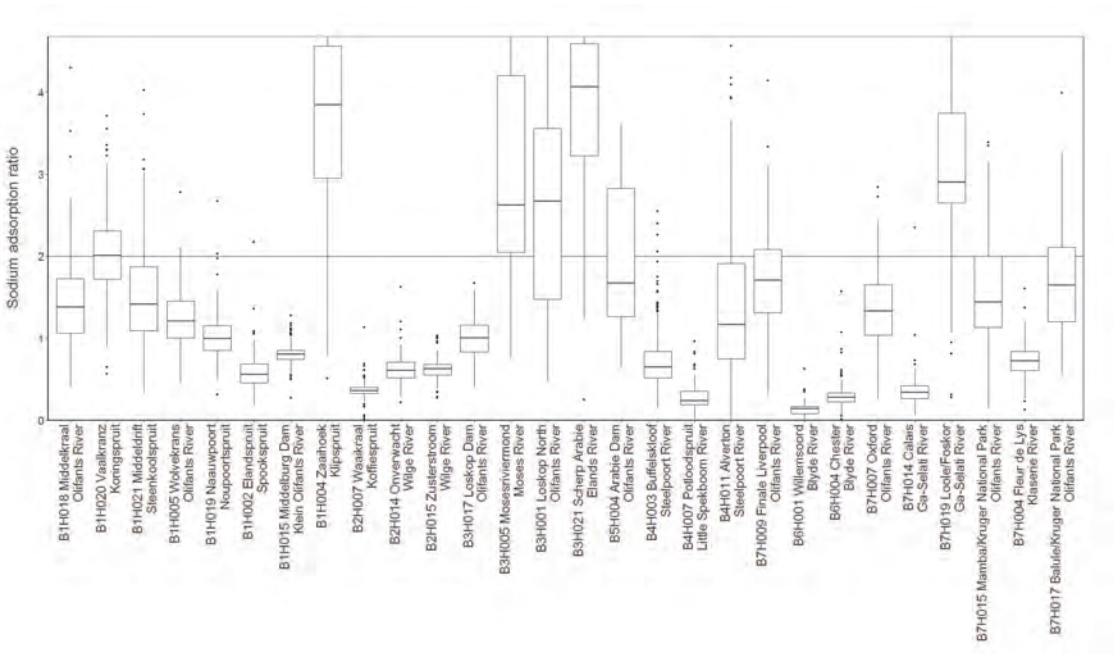

Figure 53 Box and whisker plots of sodium adsorption ratios at sites in the Olifants River catchment. Boxes show the interquartile range, with medians indicated. Whiskers show levels within 1.5 times the interquartile range, and dots show outliers. The upper 2% of data are not plotted. Sites are roughly ordered along the axis by distance upstream, with those higher in the catchment to the left. Horizontal lines show guidelines adopted for this analysis.

Sites in the upper catchment have for the most part ideal levels of this index. An exception to this generality is found at Zaaihoek (B1H004) where levels are acceptable. This site was also found to be acidified, with high levels of sulphate and nitrogen, and a high corrosion potential. While many of the upper catchment sites are broadly similar, some of the impacts and drivers for this river seem likely to be unique.

Sites in the Wilge River have low sodium adsorption ratio scores and this, together with the low conductivities found at these sites, recommends the water for irrigation.

Beyond this point, as with results from the chloride salinization index, scores tend to increase. Inflows from the Moses and Elands Rivers have sharply elevated (though still acceptable) index scores, and these are accompanied by increased index scores on the Olifants River at Loskop North (B3H001). This stretch of the river with its tributaries has evidence of salinization and reduced suitability for irrigation; from inspection of satellite images, however, the number of visible centre-pivot irrigation systems suggests that irrigated agriculture is widespread and well-established.

Beyond this point, sodium adsorption ratios along the Olifants River change little, and, with one exception, tributaries joining the Olifants have low index scores. The exception is the Ga-Selati River, which despite having ideal water in the uppermost reaches, degrades sharply after passing Phalaborwa. This profound negative change along the Ga-Selati River is in common with the results from all water quality parameters assessed in this study.

Corrosion potential ratio

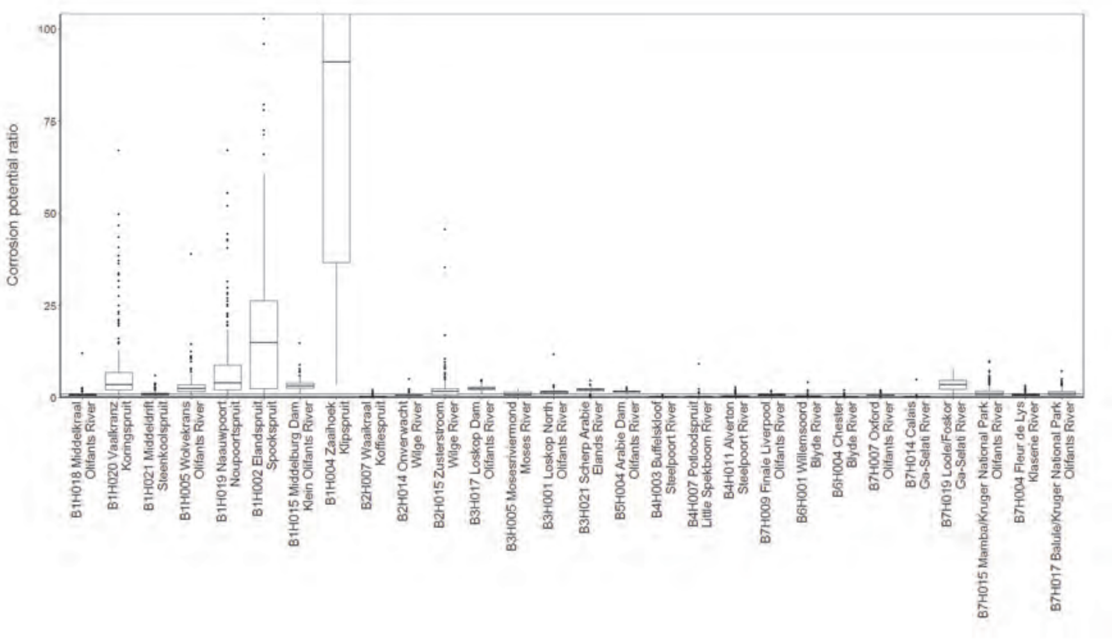

Figure 54 Box and whisker plots of the corrosion potential ratio described by Ashton and Dabrowski (2011) at sites in the Olifants River catchment. Boxes show the interquartile range, with medians indicated. Whiskers show levels within 1.5 times the interquartile range, and dots show outliers. The upper 2% of data are not plotted. Sites are roughly ordered along the axis by distance upstream, with those higher in the catchment to the left. Horizontal lines show guidelines adopted for this analysis.

The temporally aggregated corrosion potential ratio scores at a range of sites across the Olifants River catchment are shown in Figure 54. It can be seen that changes along the catchment are dwarfed by the massive scores found at some sites in the upper catchment. All but two sites in the upper catchment have corrosion potential scores that are beyond acceptable. Scores at Middeldrift (B1H021) on the Steenkoolspruit and Middelkraal (B1H018) on the Olifants River are acceptable. Several other indices have found these sites to have above-average water quality. These scores from the upper catchment, where mining, industry and agriculture are important water users, indicate the difficulties in use of untreated water for these end-users.

Although levels at sites downstream of the upper catchment are lower than those in upstream sites, high index scores may still occur. Water quality along the Wilge River degrades from ideal to beyond acceptable along the course of the river. Input from the Moses River is acceptable, while that from the Elands River is beyond acceptable. In this region of the Olifants River, scores have improved in comparison with the upper catchment, but are still not acceptable. Scores along both the Steelpoort and Byde Rivers are ideal. The condition of the Olifants River in this region has improved and scores are acceptable. As is the case with many other indices, the corrosion potential along the Ga-Selati River increases from ideal to beyond acceptable, and the condition of the Olifants River after the confluence of the Ga-Selati River is not acceptable, despite acceptable input from the Klaserie River.

Temporal trends in water quality

This section of the report assesses changes with time in selected water quality parameters at monitoring points across the Olifants River catchment. The monitoring points selected have datasets ranging from 142 samples at Mosesriviermond (B3H005) on the Moses River to 1258 samples at Middelburg Dam (B1H015) on the Little Olifants River. The results from this section complement the basic spatial analysis of trends in the catchment, and enables assessment and quantification of temporal change, which is the primary aim of this module.

Temporal changes can be caused by a range of things, including anthropogenic impact and management choices, but also including natural causes. Under natural causes, flow and runoff changes owing to changing rainfall changes can be expected to modify some or all of the parameters assessed here. Changes in rainfall owing to seasonal changes have been included in the statistical models used in tests, but the results of statistical tests cited here do not reflect seasonal change. The statistical results only apply to temporal change that excludes seasonal change.

As rainfall patterns will vary outside of predicable seasonal change, reaching extremes in droughts and floods, these are likely to modify water quality parameter and their effects are covered by the statistical tests applied.

Microbial data could not be sourced for the monitoring points selected to provide spatial cover of the catchment. Microbial data is also of more recent provenance than the other water quality parameters assessed here. For this reason, temporal trends in microbial data will be presented separately from the other water quality parameters.

Middelkraal (B1H018) on the Olifants River

This site is the highest on the Olifants River and was selected to act as something of a reference site for this catchment. An inspection of data presented in Figure 55 makes it clear that the site is not free of impacts however. This site had 612 records available from 1991 until the current day.

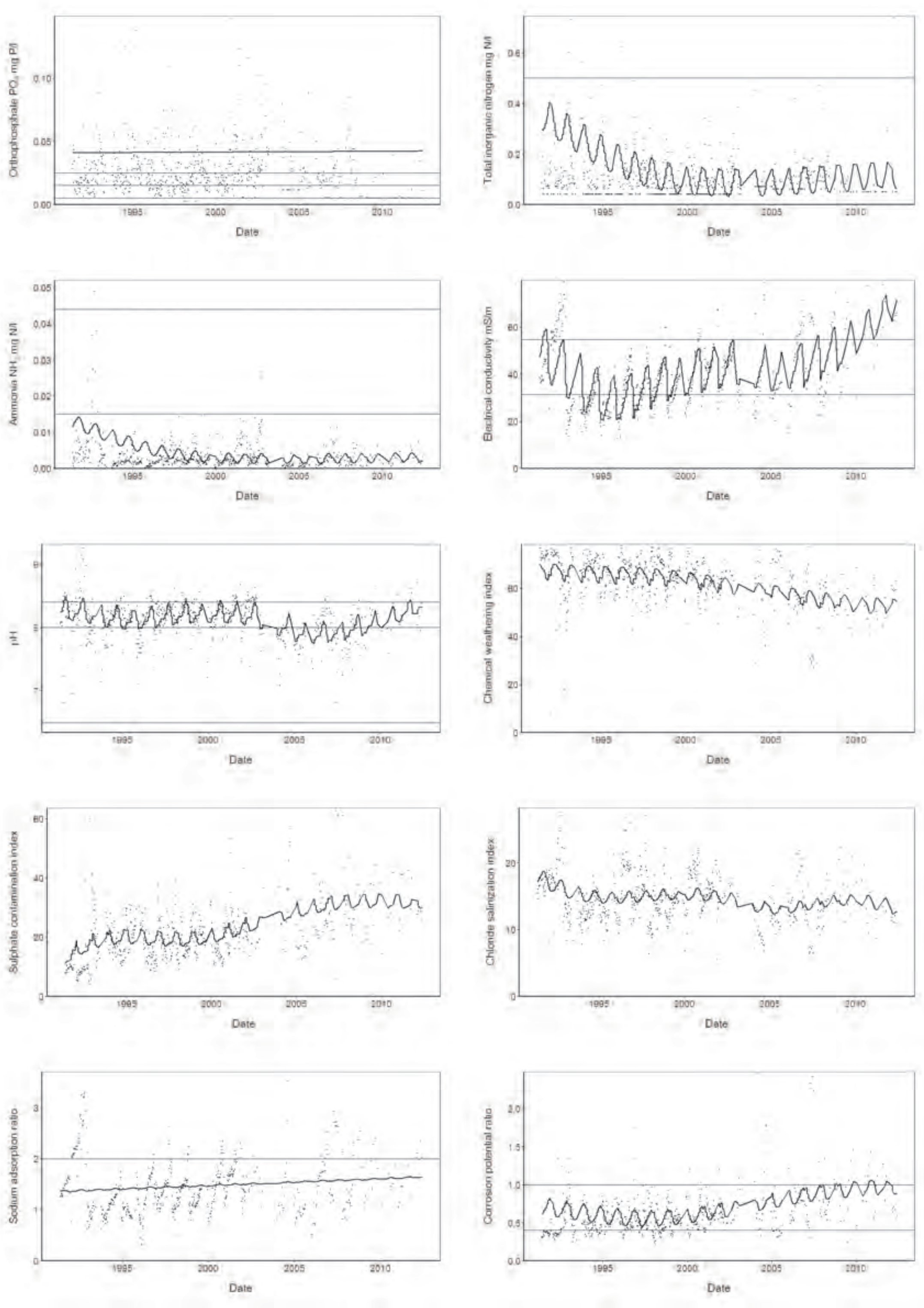

Figure 55 Temporal trends in several water quality parameters at monitoring point B1H018 at Middelkraal on the Olifants River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate p=0.987; TIN p=0.010; ammonia p=0.014; EC p<0.001; pH p<0.001; chemical weathering index p<0.001; sulphate contamination index p<0.001; chloride salinization index p=0.004; SAR p=0.201; and CPR p<0.001.

The model-fitting procedure was not able to fit a model for change with time in orthophosphate levels, probably owing to the considerable variation in data apparent in the plot. Inspection of the data reveals no apparent overall trend with time or season. However, few samples were ideal, and, while a significant number were acceptable or tolerable, and large number had levels higher than tolerable. Levels of total inorganic nitrogen, on the other hand, were nearly always ideal and showed a decreasing trend from 1991 until roughly 2000, and remained at that level thereafter.

Changes in un-ionized ammonia levels follow a similar trend to changes in nitrogen, decreasing from 1991 until 2000, and remaining low thereafter. The great majority of samples had ideal levels of this toxicant. Electrical conductivity levels have varied over time, decreasing with time until approximately 1995, when a large number of samples would be classified as ideal, and then increasing to a tolerable level in recent years. Seasonality in this parameter was pronounced ($p<0.001$).

For most of the data record, pH levels fluctuate seasonally between 8 and 8.4, making them acceptable. However, for a period between roughly 2004 and 2008, pH levels decreased towards a more ideal level. Increases thereafter returned pH to its original level. The cause of this change is not known.

The chemical weathering index shows a decreasing trend over time. To certain extent, this is cause by increasing sulphate levels, as the sulphate contamination index shows a reciprocal increase. The chloride salinization index also shows a drop over the monitored period. This is not a function of decreasing levels of chloride, but rather its relative contribution to the ion complement of the water in the face of increasing levels of sulphate. While chloride levels at this site are largely ideal throughout, sulphate levels over the monitored period increase from ideal to largely acceptable (after DWA 2011a).

The model-fitting procedure failed to derive an adequate model for sodium adsorption ratio data at this site. Inspection of the data suggests a slightly increasing and highly seasonal trend in this parameter. Despite this apparent increase, levels of this parameter are mostly ideal and by this parameter, water from this site is suitable for irrigation. The corrosion potential ratio shows a steady increase over the monitored period, and index scores move from the ideal-to-acceptable boundary to the upper acceptable bound. Changes in this parameter are largely a function of increasing sulphate levels. The suitability of the water for use according to this index is low by the end of monitoring.

Vaalkranz (B1H020) on the Koringspruit

This site is located low on the Koringspruit just before the confluence with the Olifants River. This river drains a small catchment where land use is dominated by agriculture, coal mining and power generation. The data record at this site extends from 1990 until the current day, and 881 records are available for analysis.

Over the period that data is available, few orthophosphate records could be classified as ideal, with the majority being acceptable or tolerable or worse. The statistical model fitted to the data indicate some change in overall orthophosphate levels with time, with a decrease in levels between approximately 1994 and 2000, returning to overall intolerable levels in about 2008, and a small hint at an increasing trend thereafter. Total inorganic nitrogen levels were nearly all ideal, and levels of this parameter decreased over time to stabilize after 2005.

Un-ionized ammonia levels have also decreased over time, and nearly all records show levels of this toxicant to be ideal.

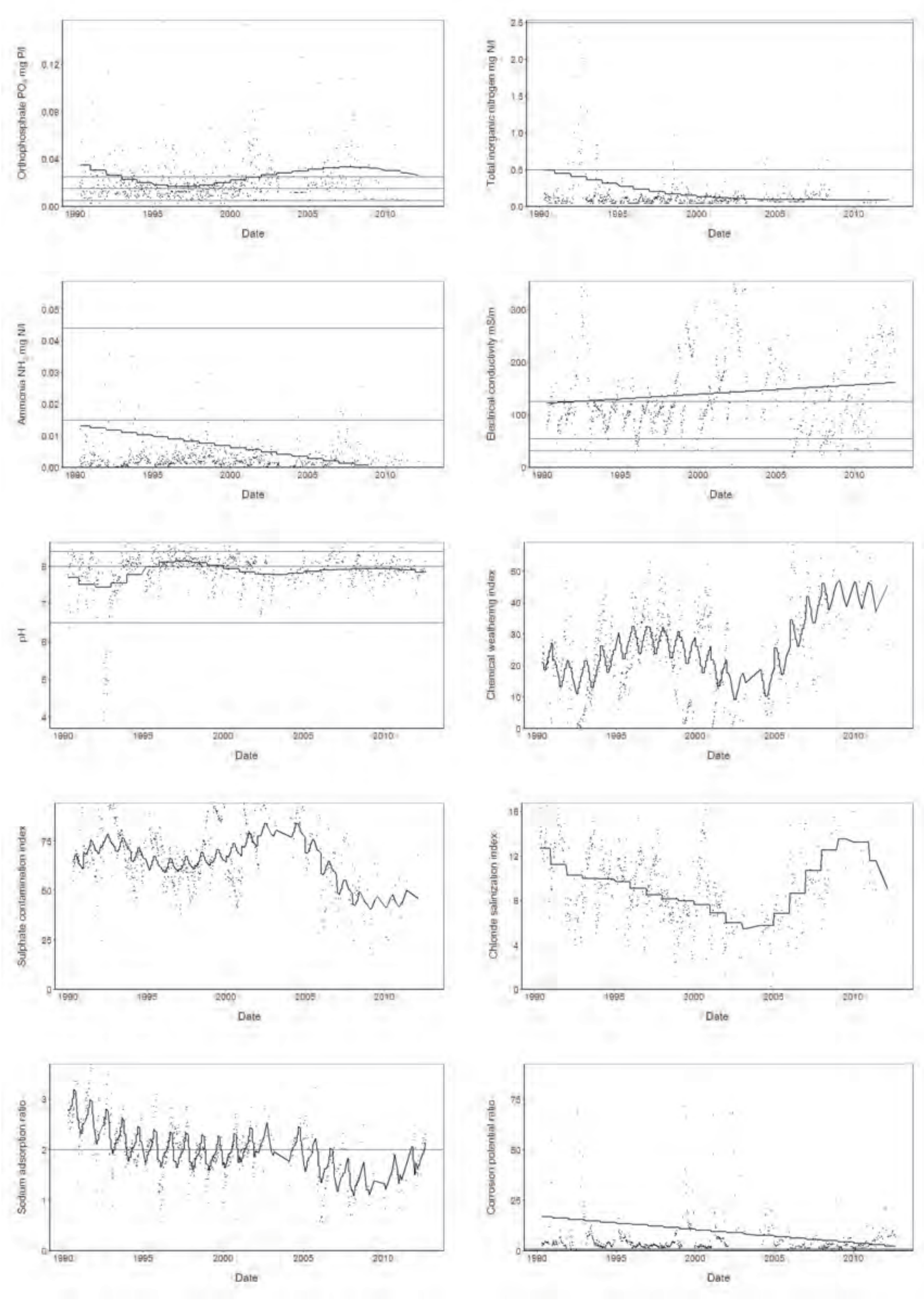

Figure 56 Temporal trends in several water quality parameters at monitoring point B1H020 at Vaalkranz on the Koringspruit. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate p=0.002; TIN p=0.034; ammonia p=0.078; EC p=0.103; pH p<0.001; chemical weathering index p<0.001; sulphate contamination index p<0.001; chloride salinization index p<0.001; SAR p<0.001; and CPR p=0.048.

The statistical model fitting process was unable to define a model that closely followed the data, likely as a function of seasonal variability and stochasticity apparent in Figure 56. Inspection of the plot suggests that a slight increase in overall levels may have occurred over the monitoring period, but also that variation, either random, seasonal or other, is high. The majority of data are in a tolerable or worse state.

The pH at this site showed significant temporal variation but no overall trend for the monitoring period, and pH around levels of 8. As such, pH levels at this site can be classed as ideal-to-acceptable for the greater part.

The chemical weather index varied considerably with time and with season, and with a tendency to better scores since 2005. This improvement in scores of this index (from a very low base) seems to be driven largely by decreasing relative contribution of sulphate, although absolute sulphate levels remain to a large extent only tolerable or worse (following DWA 2011a). The chloride salinization index is low throughout, and decreases between 1990 and approximately 2004, whereafter it increases sharply to 2010. Absolute chloride levels are with very few exceptions, ideal or acceptable throughout the monitored period.

The results of the sodium adsorption ratio show in increasing suitability of water for irrigation, moving from acceptable to ideal overall over the monitored period. The corrosion potential ratio, on the other hand, is high throughout with a decreasing overall trend driven by fewer peaks towards the end of the monitored period, and the risk of corrosion owing to use of water from this site is beyond acceptable throughout.

Middeldrift (B1H021) on the Steenkoolspruit

This site is fairly low on the Steenkoolspruit shortly before its confluence with the Olifants River and drains a catchment with major land-uses including mining, power generation and agriculture. The data record from this site consists of 770 records from 1990 until the recent day. Water quality parameters for this site are presented in Figure 57.

Levels of orthophosphate are unacceptably high at this site and showed a steady increase over time, with the highest levels (bar the occasional peak) being found towards the end of the data record. With relatively few exceptions, and those from the early 1990s only, samples from this site are classified as having worse than tolerable levels of orthophosphate. The levels of orthophosphate encountered at this site are far greater than at surrounding sites in the upper catchment.

Total inorganic nitrogen levels also show a steady increase with time, with the majority of records moving from the ideal to the acceptable class.

Un-ionized ammonia levels have changed with time, showing a trend of increase from the start of the data record until approximately 2004, whereafter levels decreased until the end of the data record. The great majority of records over this period was either classed as ideal or acceptable.

Electrical conductivities of samples from this site have for the most part fluctuated between acceptable and tolerable over the data record, with relatively few records classed as ideal and fewer as beyond tolerable. Peaks in conductivity have occurred in approximately 1998 and 2004.

pH levels in samples from this site have been fairly high for much of the data record, with somewhat of a decrease in samples collected after 2002. Overall, the pH values from this site are greater than surrounding upper catchment sites.

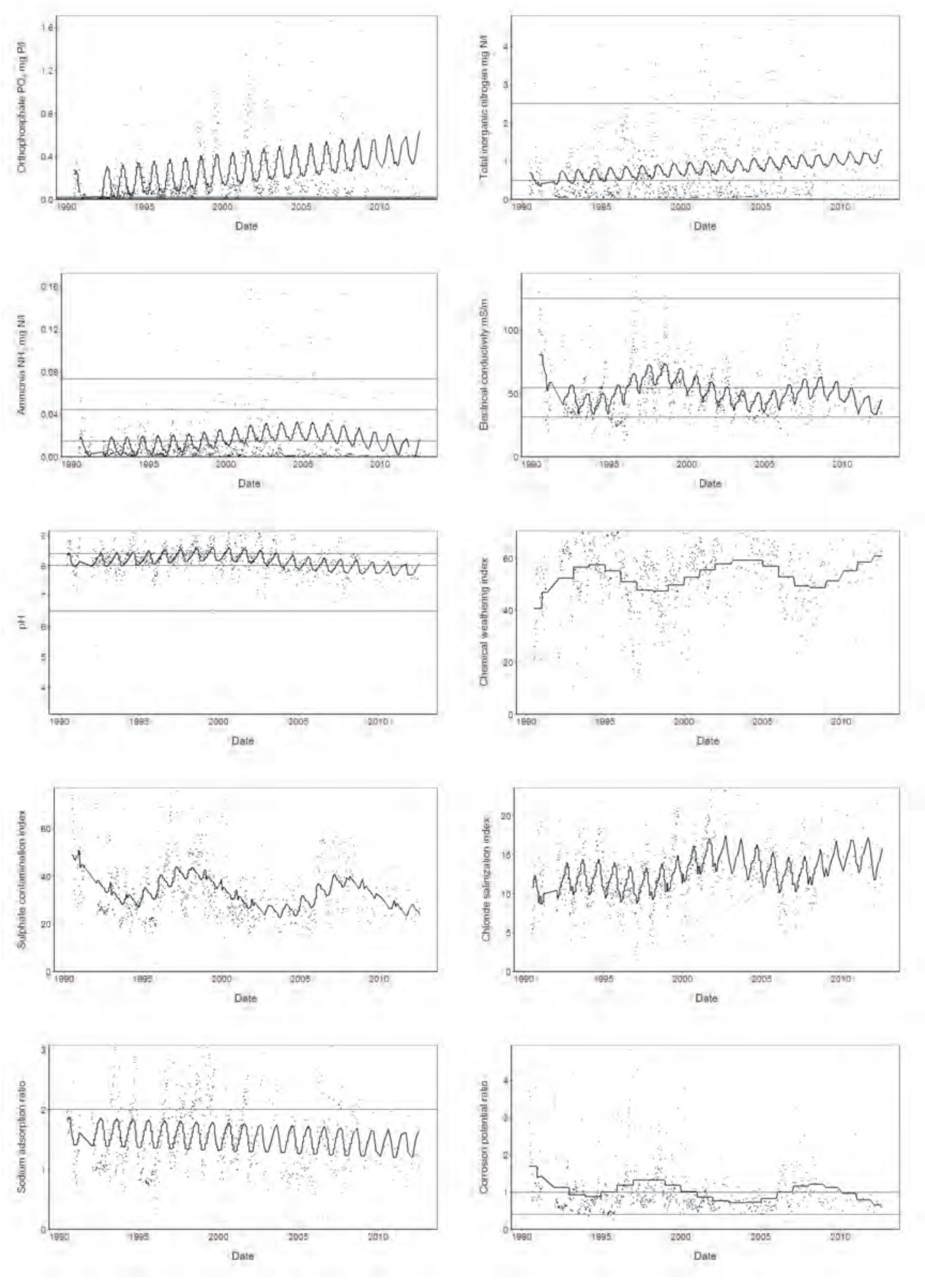

Figure 57 Temporal trends in several water quality parameters at monitoring point B1H021 at Middeldrift on the Steenkoolspruit. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate p=0.062; TIN p=0.009; ammonia p<0.001; EC p<0.001; pH p<0.001; chemical weathering index p<0.001; sulphate contamination index p<0.001; chloride salinization index p<0.001; SAR p=0.119; and CPR p<0.001.

The chemical weathering index results from this site show some regular fluctuation with time that might be a function of changes linked to changing salinities as reflected in electrical conductivity levels from this site. The chemical weathering index scores from this site are somewhat higher than most sites in the upper catchment.

Relative sulphate levels at this site are low in comparison to sites in the surrounding catchments. However, they cannot be considered low in a general way, and they still reflect the level of sulphate contamination of the Olifants River and the upper catchment in particular. Levels fluctuate with time in a manner that suggests that relative changes in sulphate may underpin changes in the chemical weathering index. The chloride salinization index increases slightly over time and with some temporal fluctuation, but levels remain low throughout. Absolute levels of chloride are largely ideal, and those of sulphate mostly ideal or acceptable, throughout the monitored period (following DWA 2011a).

The water from this site appears ideal for irrigation according to the sodium adsorption ratio results. However, the corrosion potential of the water is mostly acceptable and sometimes worse. Changes in the corrosion potential ratio are to a fair extent a function of changes in sulphate levels at the site.

Wolvekrans (B1H005) on the Olifants River

This site lies on the Olifants River just upstream of the Witbank Dam in an area heavily dominated by coal mining. The data record from this site starts in 1979, but sampling is sparse and irregular until 1986, thereafter the data record is good until the current day. The data set consists of 989 records over the monitored period. Data on selected water quality parameters from this site are presented in Figure 58.

Orthophosphate levels range between acceptable and intolerable for most of the data record. There is evidence of a rapid increase in orthophosphate between 1990 and 1995, as well as an apparent decrease at the end of the monitored period. Total inorganic nitrogen shows an overall decrease with time and remains largely ideal throughout the monitored period.

Ammonia levels increase slightly over the monitored period but remain classed as ideal throughout.

Although levels of electrical conductivity increase slightly with time, conductivity records are for the most part acceptable or tolerable throughout the monitored period. pH levels increased from around 7 in 1985 to range around 8 from approximately 1995 until the end of the data record.

The chemical weathering showed no detectable trend with time, though seasonal variation was statistically significant ($p=0.002$). In common with sites in the upper catchment, the values for this index were low, indicating that natural weathering processes were fairly insignificant in determining the ionic complement of water at this site. In contrast with this observation and in common with upper catchment sites, the sulphate contamination index scores from the site were high. No overall trend with time in this index was found. The absolute levels of sulphate at the site fell between acceptable and intolerable according to DWA (2011a). The median sulphate concentration was tolerable, and 37% or samples had concentrations of sulphate higher than tolerable.

In contrast to the sulphate scores, the chloride salinization index scores were low and decreased with time. The great majority of samples from this site had absolute chloride levels that are classed as ideal in DWA (2011a).

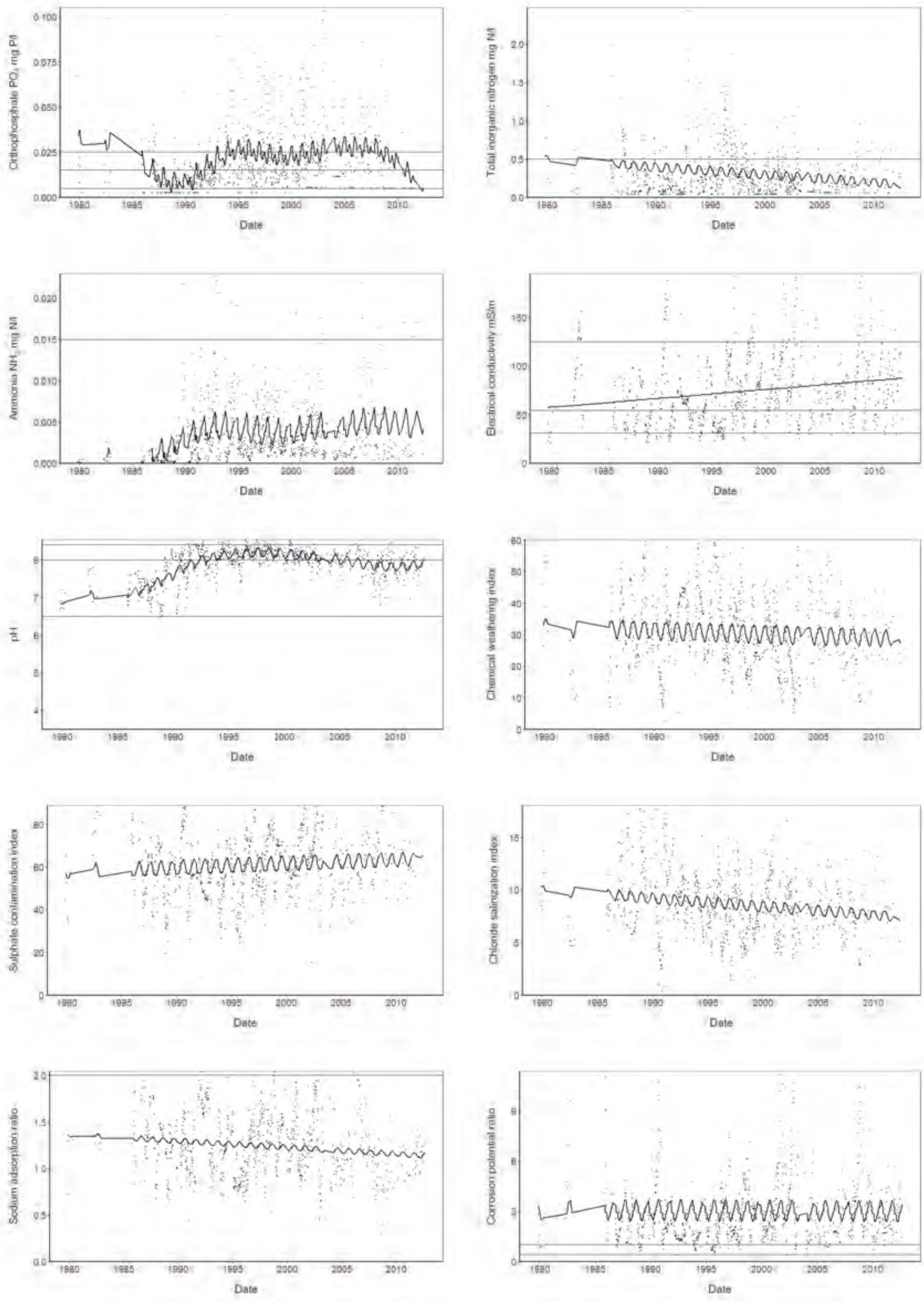

Figure 58 Temporal trends in several water quality parameters at monitoring point B1H005 at Wolvekrans on the Olifants River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate $p<0.001$; TIN $p=0.016$; ammonia $p<0.001$; EC $p=0.012$; pH $p<0.001$; chemical weathering index $p=0.313$; sulphate contamination index $p=0.145$; chloride salinization index $p=0.006$; SAR $p=0.028$; and CPR $p=0.893$.

The results of the sodium adsorption ratio scores at this site indicate that, from the viewpoint of sodicity, the water from the site is appropriate for irrigation of crops. However, corrosion potential ratio scores suggest a high likelihood of corrosion should the water be used. As chloride levels are low, the corrosion potential is driven by the high levels of sulphate encountered at this site.

Several of the datasets portrayed in Figure 58 showed considerable short-term variation, and, from the scatterplots, this variation was combined with pronounced but variable seasonality. Under these conditions, the model-fitting algorithm was unable to reliably detect variations over time. This was particularly the case when temporal variation was not high and/or changed frequently.

Naauwpoort (B1H019) on the Noupoortspruit

This site is located on the Noupoortspruit, which flows from the west into the Witbank dam. The small catchment is dominated by agriculture, coal mining and expanding residential use. Suburb of eMalahleni that surround the river on both sides have seen substantial construction and expansion in recent years. The data record for the site stretches from 1990 until the current day, and contains 834 samples. Plots of selected water quality parameters at this site are presented in Figure 59.

Over most of the monitored period, orthophosphate levels increased from ideal to tolerable to for the most part intolerable. The increase over that period is for the most part linear with seasonal variation. However, samples collected since 2009 have with few exceptions been ideal (a change that the model fitting procedure did not adequately capture). Contrariwise, levels of inorganic nitrogen showed little change from the start of monitoring until approximately 2004, but since then they have increased considerably, such that recent samples are acceptable rather than ideal. The cause of recent changes in both nutrient measures is not known.

Un-ionized ammonia levels show a recent increase in the same way that inorganic nitrogen does, but all levels are comfortably within the ideal class.

The electrical conductivity at this site shows several temporal (and seasonal $p=0.007$) changes, but no clear trend to increase or decrease. Levels are for the most part classed as tolerable, though, when mean levels increase, as at about 1998, a significant number of samples have higher conductivities than are considered tolerable.

The pH at this site varies around 8 for most of the data record, with ranges that varied from ideal to acceptable for the most part. The scatterplots indicates that occasional sharp decreases in pH throughout the data record, most notably in approximately 1996.

The chemical weathering index showed considerable change over time. Despite several large changes in index scores no clear overall trend with time is apparent, bar somewhat elevated values in around 2005 when scores of this index were good in comparison to other sites in the upper catchment. Scores over this period range from near zero to 50 and above, and this represents a great deal of short term change in the index. This in turn suggests temporally varying impacts on rivers, possibly as contamination events or changes in effluent quantity or composition.

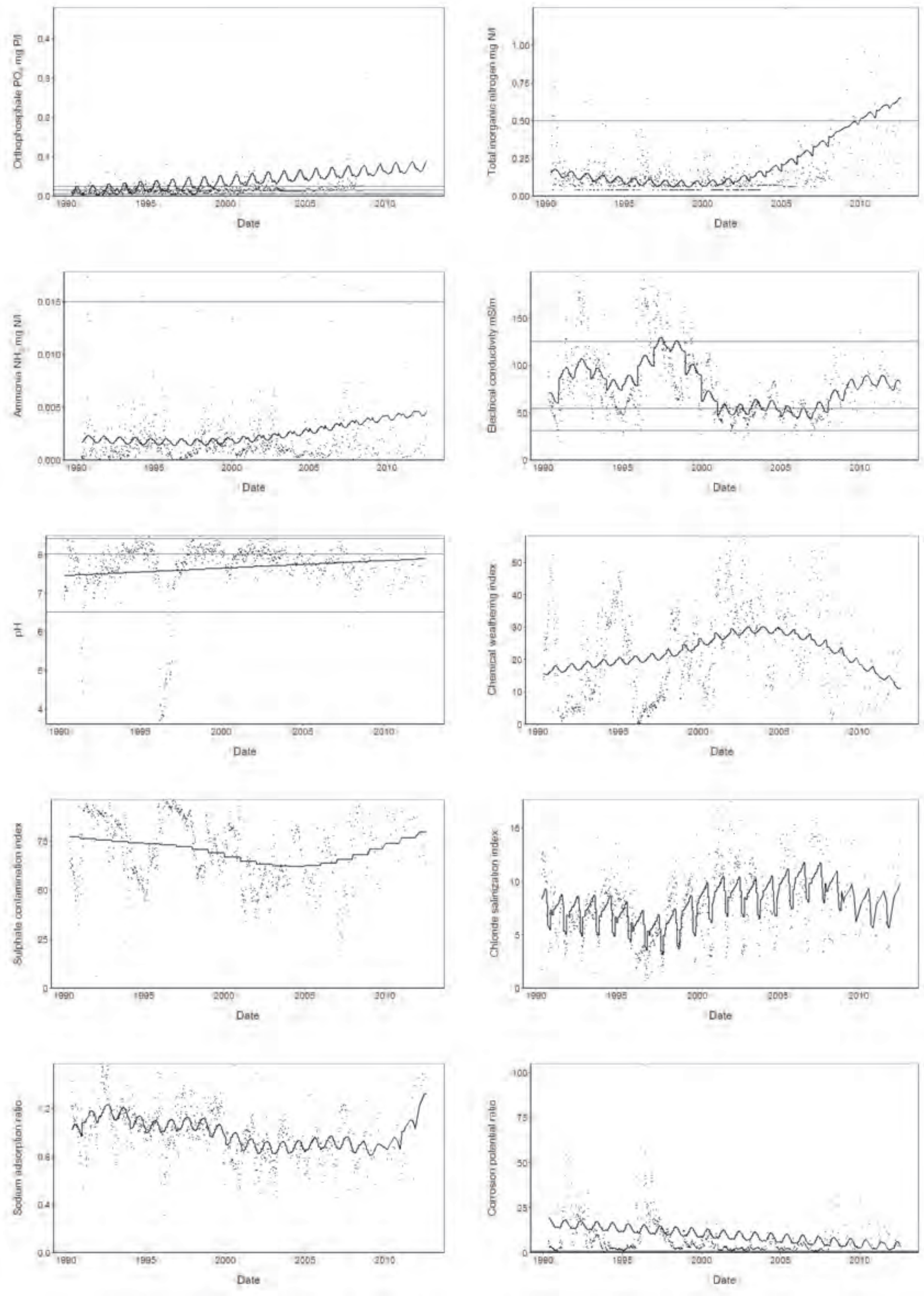

Figure 59 Temporal trends in several water quality parameters at monitoring point B1H019 at Naauwpoort on the Noupoortspruit. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate p=0.017; TIN p<0.001; ammonia p<0.001; EC p<0.001; pH p=0.178; chemical weathering index p=0.003; sulphate contamination index p=0.006; chloride salinization index p<0.001; SAR p<0.001; and CPR p=0.095.

The sulphate contamination index scores from Naauwpoort were high overall, with some temporal variation. The temporal patterns shown by the sulphate score are the inverse of those observed in the chemical weathering index, suggesting that changes in the latter are largely driven by changes in sulphate contamination of the water at this site. Assessment of absolute sulphate levels reveals peaks in around 1993 and 1998, and an increase from around 2008 to the end of the monitored period. During the two peaks, sample concentrations were beyond tolerable, and in the later increase, samples levels are tolerable or worse. In comparison, over the period from around 2003 to 2006, most sulphate levels were acceptable. These patterns confirm the importance of sulphate levels in changing both scores.

It is of interest to note that the pH decrease of around 1996, the increase in salinity after 1996, a 1996 decrease in ammonia, and the changes in sulphate levels were all contemporaneous and likely to have the same cause. A likely possibility would be the introduction of unneutralized acid mine drainage to the stream.

The chloride salinization index shows some change over time, but largely follows the pattern of the chemical weathering index, though with a pronounced seasonality ($p<0.001$). This supports the contention that the main impact on this stream's chemistry is the one mentioned above. Levels of the index are low, and there is consequently little evidence of chloride build-up owing to salinization.

The sodium adsorption ratio shows minor temporal change and values remain within the ideal range throughout the monitored period. However, as is common in upper catchment sites, the results of the corrosion potential index reveal water likely to cause corrosion if used domestically, industrially or agriculturally. The scores obtained during the 1993 and 1996 sulphate peaks are particularly high.

Elandspruit (B1H002) on the Spookspruit

This site is located low on the Spookspruit just before it joins the Olifants River slightly downstream from Witbank Dam. The stream drains a long catchment to the east and south of the Olifants River. Land use in the catchment is the mix of agriculture and coal mining typical of sites in the upper Olifants River catchment. The data record extends from 1979 to the current day, with an isolated sample from 1970. 1442 samples underlie the results presented in Figure 60. This is one of the longest datasets in the upper catchment. The results indicate an increasing impact over time that is likely associated with mining in this catchment.

The model fitting algorithm was unable to fit a trend model to orthophosphate data from this site. Inspection of the raw data in Figure 60 reveals that the majority of orthophosphate records have tolerable or better levels of orthophosphate. There is also an apparent trend to increasing levels of orthophosphate in samples from around 1998 onwards.

Levels of total inorganic nitrogen in samples from this site are for the most ideal or at worst acceptable. Levels increase overall from the start of the data record until around 1998, and thereafter they decrease until the end of the data record. Seasonal variation in this parameter is pronounced ($p<0.001$).

Un-ionized ammonia levels vary with time at this site, and the great majority of samples have ideal levels of ammonia. The greatest peak in ammonia levels occurs in the early 1990s.

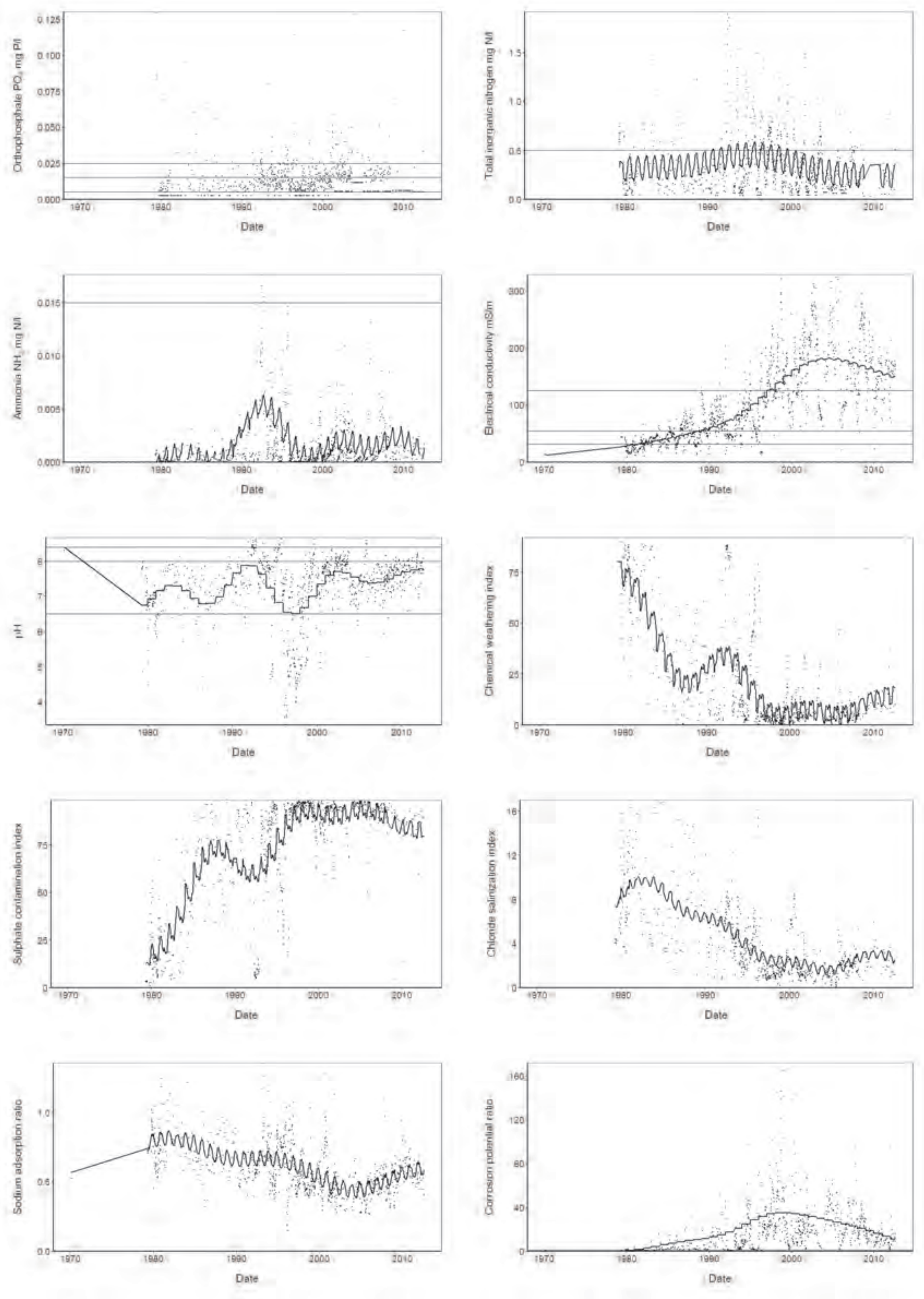

Figure 60 Temporal trends in several water quality parameters at monitoring point B1H002 at Elandspruit on the Spookspruit. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate model non-convergent; TIN $p=0.005$; ammonia $p<0.001$; EC $p<0.001$; pH $p<0.001$; chemical weathering index $p<0.001$; sulphate contamination index $p<0.001$; chloride salinization index $p<0.001$; SAR $p<0.001$; and CPR $p<0.001$.

Electrical conductivity levels at this site show a strong temporal trend to change that is greater than that found at other sites in the upper Olifants River catchment. Levels of conductivity increase with time from the start of the data record; however, early samples have either ideal or acceptable levels of conductivity. The rate of change increases with time, and in the early to mid-1990s accelerates until levels peak in the early to mid-2000s. At this point, the majority of samples have levels that are worse than tolerable. Thereafter, overall levels decrease but most remain worse than tolerable. This leads to a situation where median conductivity in 1982 was acceptable at 35 mS/m, and 1996 was the last year median conductivities were tolerable at 101 mS/m. Beyond this, levels increased to a median of 252 mS/m in 2002 and decline thereafter. Median conductivities increased by a factor of 7 over a 20 year period.

pH levels at this site are low for a site in the upper Olifants River catchment, and fluctuate over time without any clear trend to change. Levels are for the most part ideal throughout. The dip in pH levels in approximately 1998, when levels as low as 4 were recorded, suggests that this was impact driven. Together with results from other parameters shown in Figure 60, this change seems likely to be a result of an introduction of insufficiently neutralized acid mine effluent to the river.

Results from the chemical weathering index show changes that indicate the river ionic complement changed considerably over the monitored period. At the start of the data record, chemical weathering index scores from this site were indicative of a river whose ionic complement was to a great extent a function of chemical weathering (median score for 1979 was 92). In the early to mid-2000s, the median annual index score at this site had declined to 2, indicating that the ions in solution at this site were derived almost exclusively as a result of impacts. A small and brief recovery was found between the late 1980s and the mid-1990s, and there is some indication of a small improving trend at the end of the data record (although the median score in the last year of the data record is still small at 9).

Inspection of the plot of the sulphate contamination index in Figure 60 shows a curve that is largely a mirror image of the chemical weathering index plot, indicating that the changes in the latter are driven by elevating levels of sulphate in the water. Over the monitored period, 68% of samples had absolute sulphate levels above the upper bound of tolerable (following DWA 2011a). The trend in increasing sulphate levels means that very few samples taken after 1997 have even tolerable levels of sulphate.

Relative levels of chloride are low, and changes in the chloride salinization index are a function of changes in sulphate levels. Absolute levels of chloride show no change with time and nearly all samples are ideal after DWA (2011a).

Levels of the sodium adsorption ratio show a slight decrease with time. This decrease may be a function of index scores responding to elevated levels of calcium that occur over the monitoring period. Calcium levels rise steady, and overall, 54% of them are beyond tolerable (after DWA 2011a). This is likely to be a response to the use of lime in neutralizing acid mine drainage/effluent.

Elevated sulphate levels lead to an increase in the corrosion potential ratio scores until around 2000. The levels found at this site are high for the upper catchment and corrosion risks associated with the use of water from this site correspondingly high.

Downstream of the Middelburg Dam (B1H015) on the Klein Olifants River

This site is located at a small weir about 500 m downstream of the wall of the Middelburg Dam. The fairly large catchment upstream of this site contains a mix of agriculture, coal mining, power generation and some residential land use. The data set from this site extends from 1983 until the current day. The data set assessed here contains 1285 records. Plots of selected water quality parameters are presented in Figure 61.

Orthophosphate levels at this site show a fairly smooth increase with time across the monitored period. There is a suggestion that levels have improved in recent years, but such a change would be dramatic and requires further data for confirmation. Levels are for the most part ideal at the start of the data range, and rise to be largely tolerable in the mid to late 2000s. Scores from the final few years are in general ideal again.

Total inorganic nitrogen levels decrease somewhat from the start of sampling, and then fluctuate around 0.1 mg/ℓ. Levels of this parameter are ideal throughout.

Un-ionized ammonia levels show a tendency to increase through the sampled period but remain low and all but a few samples are classified as ideal.

Electrical conductivity at this site increases unevenly over time, with a lesser peak in about 1994, and a higher peak in 2010. Over this period levels moved from an acceptable class to a tolerable one, with very few samples exceeding the upper bound of tolerable.

The pH at this site increased slightly from the start of the data record to stabilize at around pH 8. After the pH had reached 8 it remained very stable in ongoing samples.

The chemical weathering index at the start of the data record was about 30, which indicates that natural weathering was no longer the dominant driver of water samples' ionic complement. Over the monitored period the chemical weather index decreased until around 1993; thereafter, values increased again until approximately 2002, when they dropped sharply until around 2005, when they recovered somewhat. An overall trend is a slight reduction over the monitored period in chemical weathering scores.

Inspection of the sulphate contamination index reveals that, as in most sites in the upper catchment, the cause of changes in several indices is changing sulphate levels. The trends revealed on the plot of the sulphate contamination index mirror those of the chemical weathering index, the chloride salinization index, and the corrosion potential index. The plot also reveals that sulphate is a dominant ion at this site throughout the monitored period. Trends in the absolute sulphate levels reveal that levels fluctuated but seldom exceeded the bounds of the tolerable class until 2005, and thereafter the majority samples are worse than tolerable (after DWA 2011a).

Absolute chloride levels are ideal throughout, and changes in the chloride salinization index are caused largely by changes in the amount of sulphate present.

The sodium adsorption index fluctuates throughout the monitored period and shows no overall change with time. Absolute levels of calcium, used in the denominator of the formula used to calculate this index, increase over time showing a pattern very similar to that of sulphate, and at the end of monitoring, calcium levels are tolerable or worse (following DWA 2011a).

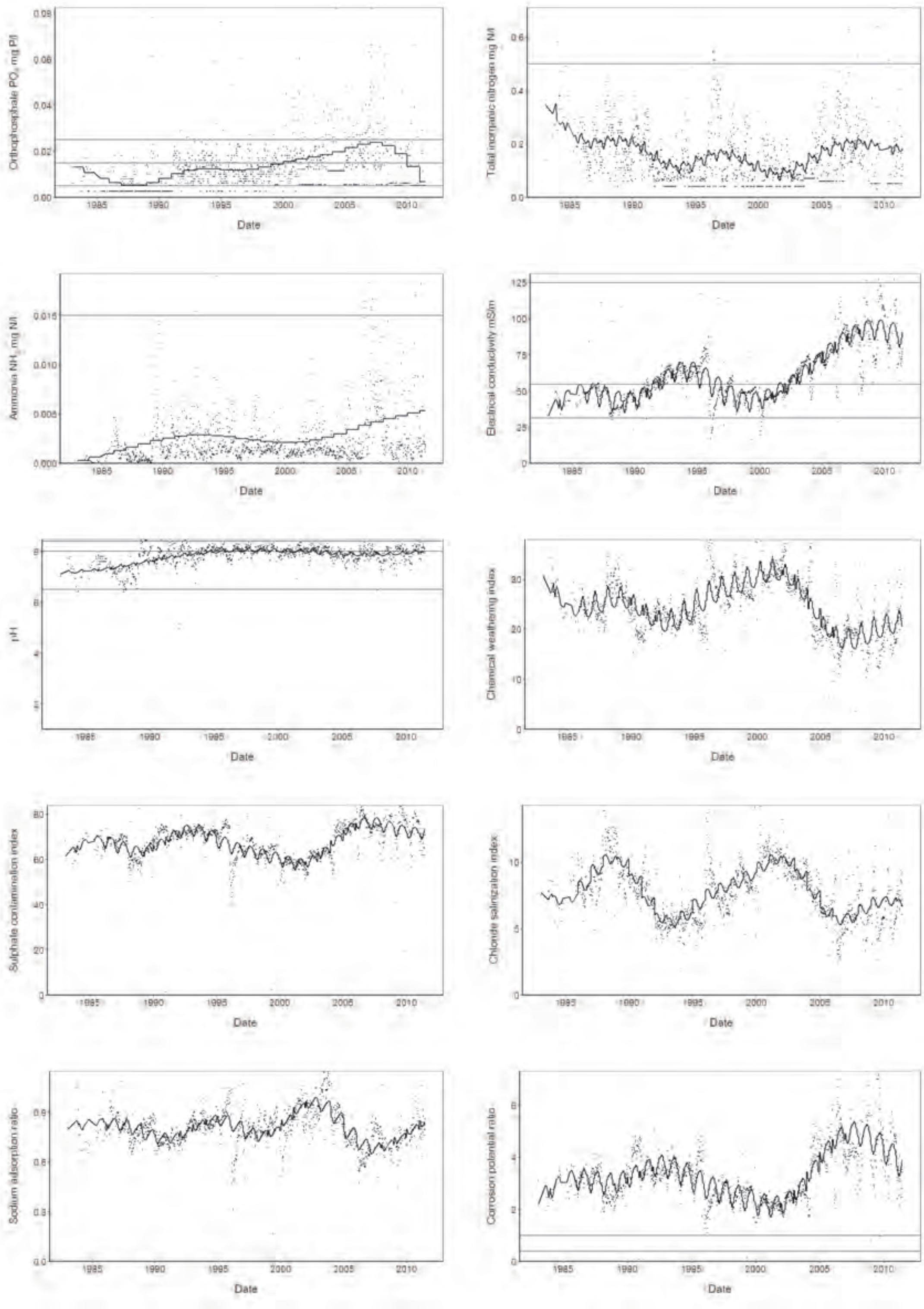

Figure 61 Temporal trends in several water quality parameters at monitoring point B1H015 downstream of Middelburg Dam on the Klein Olifants River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate $p<0.001$; TIN $p<0.001$; ammonia $p<0.001$; EC $p<0.001$; pH $p<0.001$; chemical weathering index $p<0.001$; sulphate contamination index $p<0.001$; chloride salinization index $p<0.001$; SAR $p<0.001$; and CPR $p<0.001$.

As noted above, changes in sulphate levels drive changes in the corrosion potential ratio, and the pattern of changes over time matches that of sulphate and the sulphate contamination ratio. The levels of this index are not acceptable throughout the monitored period.

Zaaihoek (B1H004) on the Klipspruit

This site is midway along the lower Klipspruit before it joins the Olifants River. The catchment contains the normal mix of land uses for the upper catchment, namely coal mining and agriculture, with some residential and industrial use. The fraction the catchment used for coal mining is larger here that in some others in the upper catchment. The catchment contains the old Transkei and Delagoa Bay and other abandoned collieries. The data record stretches from 1976 until the current day, with some scattered samples going back to 1966, and contains 1416 records. Of the catchments impacted by coal mining in the upper Olifants, the others covered in this report control the acidity of released effluent, but in the Zaaihoek the effects of acidification owing to mine drainage are apparent. Figure 62 presents trends in selected water quality parameters.

Orthophosphate levels in this catchment show considerable short-term variation that does not appear to seasonal. At most points in the data record, samples that are ideal can be found near those that have intolerable orthophosphate levels. The model fitting algorithm fitted a significant model to the data that showed average orthophosphate levels to be tolerable or acceptable throughout most of the monitored period, with peaks in orthophosphate levels occurring around 1989 and 2003. Data from the last few years of the data record suggest a decreasing trend in recent times.

Total inorganic nitrogen levels in this catchment are overall higher than at most upper catchment sites, and they show considerable change over time. With the exception of relatively few samples, levels do not pass out of the acceptable class during the monitored period. Peaks in nitrogen correspond broadly with those shown by orthophosphate, as peaks in inorganic nitrogen occur around 1989 and 2000. It is also noteworthy that when levels of nitrogen are low, they are very low and firmly within the ideal class. Unlike orthophosphate, levels of inorganic nitrogen increase towards the end of the monitored period.

Levels of un-ionized ammonia are low throughout with the exception of two periods in the mid-1990s and mid-2000s, which corresponds with times when pH levels increased. As pH levels affect the balance between un-ionized ammonia and ammonium, this would seem to be a direct function of pH changes. The model fitting process was not able to construct an adequate model for this parameter.

Electrical conductivity levels at this site are variable and no satisfactory model for this parameter was found. The majority of samples from this site are in the tolerable class, with a significant number being worse than tolerable. Overall trends with time are not easily discerned through inspection of data in Figure 62; however, a brief period of lower than normal values occurs in the early to mid-1990s, which corresponds with increases in pH. When pH levels increased in the mid-2000s, on the other hand, conductivity levels were high.

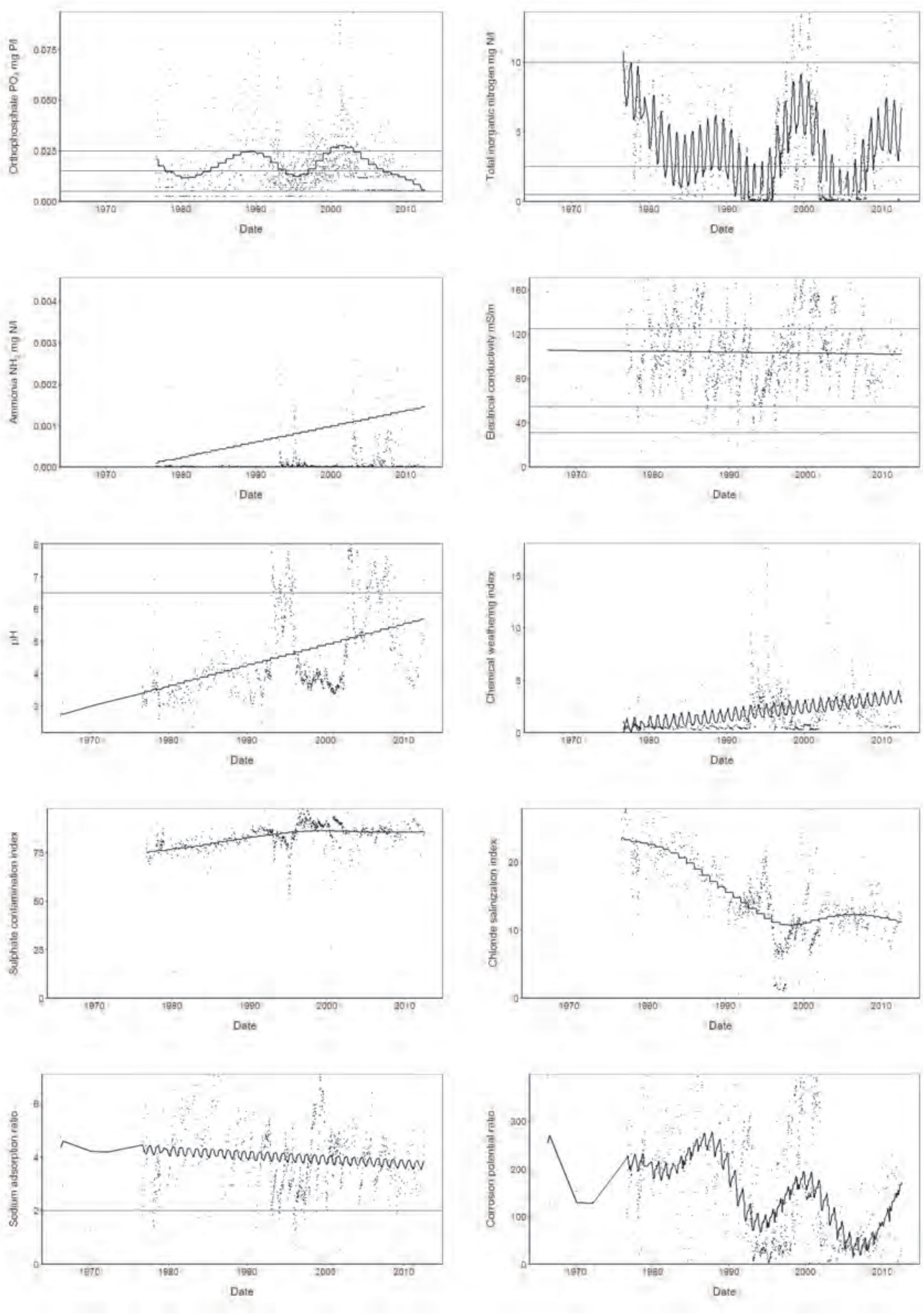

Figure 62 Temporal trends in several water quality parameters at monitoring point B1H004 at Zaaihoek on the Klipspruit. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate $p<0.001$; TIN $p<0.001$; ammonia $p=0.549$; EC $p=0.686$; pH $p<0.001$; chemical weathering index $p<0.001$; sulphate contamination index $p<0.001$; chloride salinization index $p<0.001$; SAR $p=0.139$; and CPR $p<0.001$.

Although the model fitted to the changes in pH with time explains enough of the variation in this parameter to be rated as a significant fit, inspection of the model fit reveals it to be lacking. Base pH levels are low for most of the data record, causing this parameter to be rated as beyond acceptable. However, for two periods, the first in the mid-1990s, and the second in the mid-2000s, pH levels rose, not to level that was acceptable, but significantly higher than before. As noted above, only the first of these is matched by a drop in conductivity levels. Inspection of levels of calcium (which can originate from lime used in treating acid water) suggest that the first pH increase was a function of decreased acidic effluent (calcium and sulphate levels drop), while in the second effluent was treated with lime to increase the pH (calcium levels rise, sulphate is unchanged). This hypothesis would explain the changes in conductivity encountered.

The scores of the chemical weathering index from this site are extremely low and only show a slight increase when the pH levels increase. This is probably a function of the acid conditions keeping levels of carbonate and bicarbonate (which make up the numerator of the index) low, while less acidic conditions were accompanied by increased carbonate and bicarbonate levels. Nevertheless, the high levels of sulphate in samples from this site would ensure that this index remained low in the absence of pH effects.

As might be expected, the sulphate contamination index scores are high throughout, with the exception of a dip in the mid-1990s, which is considered above. There is a slight increase in this index over time. Absolute sulphate levels are largely worse than tolerable throughout the monitored period (with the exception of the mid 1990s decrease) (after DWA 2011a).

The chloride salinization index is higher earlier in the data record than at many other sites in the upper catchment, but it decreases with time closer to the values encountered at other sites. As noted above, changes in pH and sulphate affect this index and have some influence in this trend. Absolute chloride levels in samples from the catchment decrease with time from largely acceptable to the upper bound of ideal according to DWA (2011a).

The sodium adsorption ratio scores from this site fall into the acceptable class. As with several of the other water quality parameters assessed at this site, the model fitting process as not able to fit a satisfactory model to the data from this parameter. Inspection of the data shows a slight decrease in this parameter over time. This parameter is high at this site in comparison to others in the upper Olifants catchment. The corrosion potential ratio scores vary with time, showing two decreases that match the two pH increases. Corrosion potential ratio scores from this site are very high and the use of water from this site in metal piping, etc. is high risk.

Waaikraal (B2H007) on the Koffiespruit

This site is in the upper Wilge River catchment, low on the Koffiespruit and just before its confluence with the Bronkhorstspruit. The catchment above this point is dominated by agriculture (at times intensive) with some residential impact in the far upper catchment. The data record from this site stretches from 1985 to the present day, and contains 916 records. Selected water quality parameters from this site are presented in Figure 63. They show a site with good water quality according to most parameters and with little trend to change over time.

Levels of orthophosphate at this site are mostly either acceptable or tolerable, with a few samples classed as ideal, and several as worse than tolerable. The model fitting process was not able to fit a model to the data. Inspection of the plot in Figure 63 reveals a considerable amount of variability with no obvious seasonal or temporal trend.

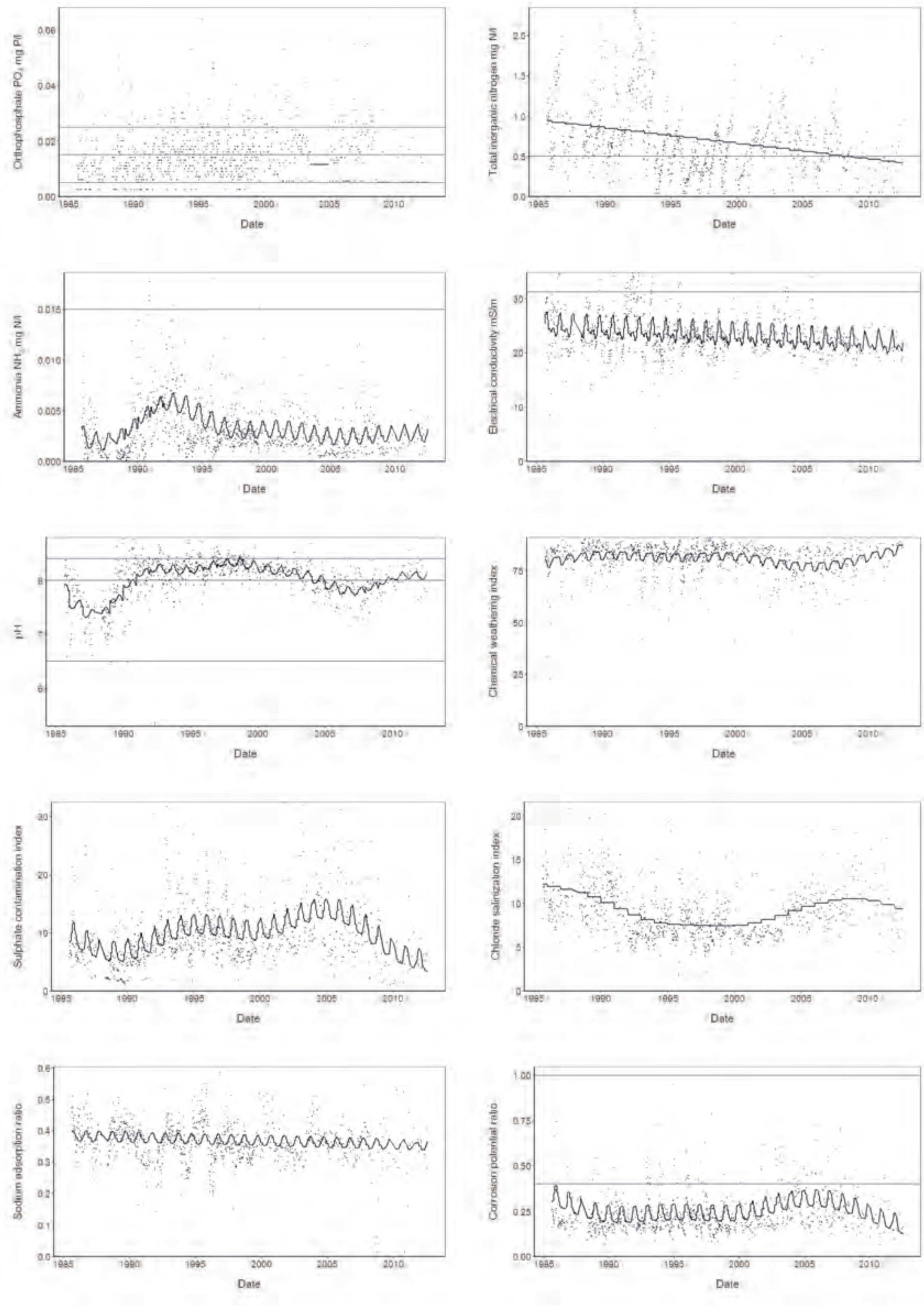

Figure 63 Temporal trends in several water quality parameters at monitoring point B2H007 at Waaikraal on the Koffiespruit. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate model non-convergent; TIN $p<0.001$; ammonia $p<0.001$; EC $p<0.001$; pH $p<0.001$; chemical weathering index $p<0.001$; sulphate contamination index $p<0.001$; chloride salinization index $p<0.001$; SAR $p=0.182$; and CPR $p<0.001$.

Levels of inorganic nitrogen are also variable, but show a tendency to decrease over time. Samples from the start of the monitored period were largely classed as acceptable, while those from the end of the data record are for the most part ideal.

Un-ionized ammonia levels show a seasonal trend (p=0.003), and, with the exception of a peak in approximately 1992, show little trend to temporal change. Levels of this parameter are ideal throughout.

Electrical conductivity levels are also seasonal (p<0.001) and beyond this, show a steady but slight decrease with time. With few exceptions, all samples from this site are classed as ideal.

pH levels change more than one during the data record from this site. Immediately after the start of the data record, pH levels shift from a median of 8.1 in 1985 to 7.2 in 1987. Thereafter the pH level increases again until 1991, when it stabilizes around 8.2. After 2000, pH levels start to drop again, until in 2008 median levels are 7.9. pH levels at this site would be classed as mostly acceptable for the greater part of the data record, and would be ideal in the late 1980s and mid to late 2000s.

The scores of the chemical weathering index, and those of the sulphate contamination and chloride salinization indices, from this site are a sharp contract to those from the upper Olifants River catchment sites presented above. The chemical weathering index scores show a site where natural weathering contributes heavily to the ionic complement of samples from the site. Both the sulphate contamination and chloride salinization index scores change with time, but scores are low and no clear increasing trend can be observed.

Both the sodium adsorption ratio and the corrosion potential ratio are predominantly ideal throughout the data record and, though some changes over time are encountered, particularly in the corrosion potential ratio, the results from these indices indicate that the water from this site is suitable for multiple uses.

Onverwacht (B2H014) on the Wilge River

This site is located on the Wilge River just upstream of its confluence with the Bronkhorstspruit. The catchment above this point is dominated by agriculture, and some coal mining is found, particularly in the mid to upper catchment. Kendal Power Station is located in the catchment and Kusile Power Station is under construction upstream of this monitoring point and scheduled to begin operating in December 2014. The data record for this site stretches from 1991 to the current day and contains 616 records. The overall picture is of a site with reasonable water quality but with evidence of certain impacts modifying the water quality.

The model fitted for orthophosphate changes at this site is not satisfactory and the model fitting process seems to have been overly affected by outliers. Inspection of the data suggests an increase in orthophosphate levels with time until 2009, when there is a distinct increase in records with lower concentrations in samples. Over the period 2005 to 2008, median orthophosphate levels are just below the upper bound of tolerable, indicating that a substantial number of samples have levels that are not tolerable. Variability in the orthophosphate data set is high.

Levels of inorganic nitrogen vary with time but the great majority of samples are classed as ideal. Overall increases over time are centred around the late 1990s and around 2010.

Un-ionized ammonia levels are variable but all would be classed as ideal. No obvious trend over time in this parameter can be seen.

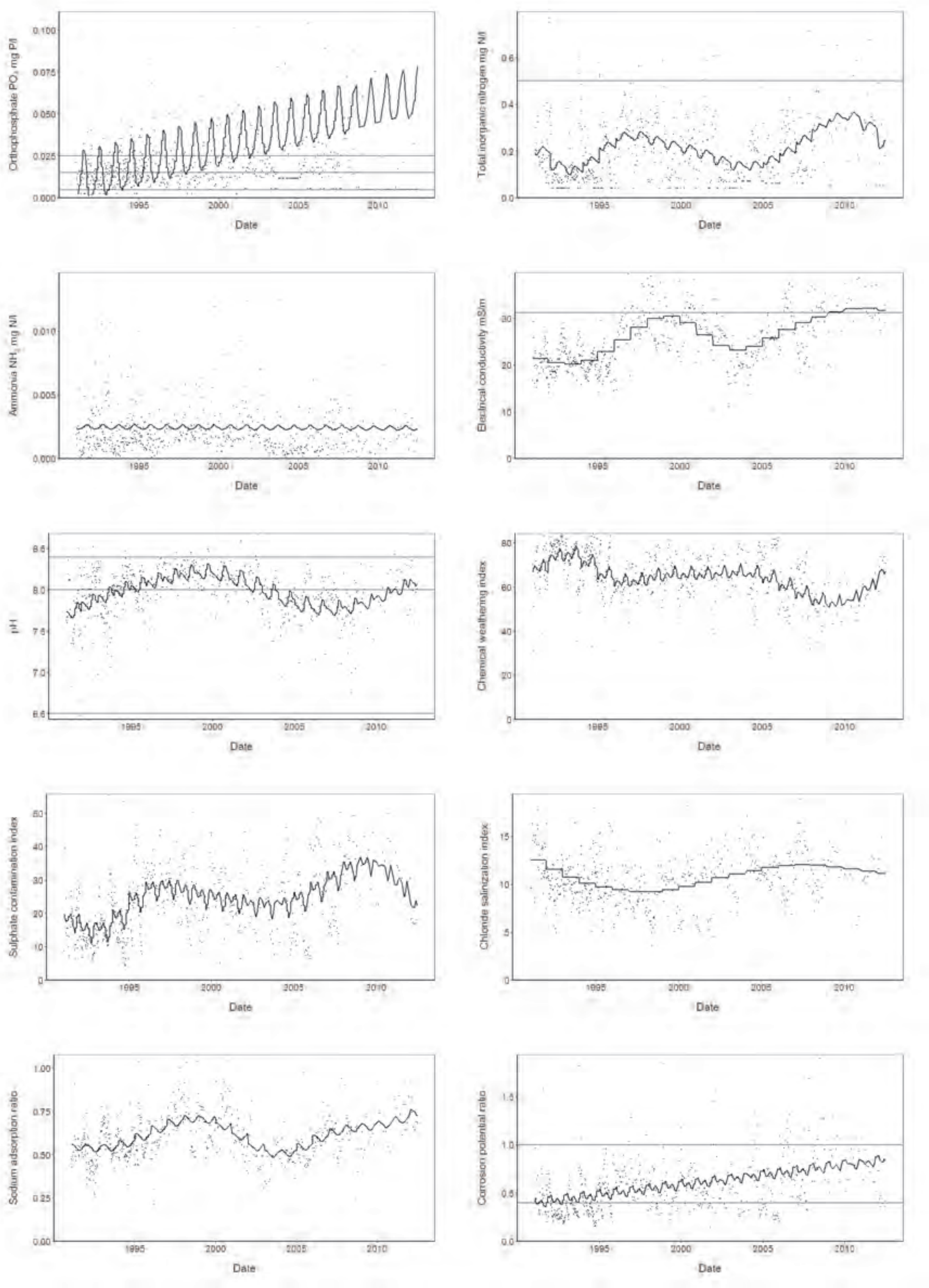

Figure 64 Temporal trends in several water quality parameters at monitoring point B2H014 at Onverwacht on the Wilge River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate p=0.354; TIN p<0.001; ammonia p=0.880; EC p<0.001; pH p<0.001; chemical weathering index p<0.001; sulphate contamination index p<0.001; chloride salinization index p<0.001; SAR p<0.001; and CPR p<0.001.

A similar pattern to that observed for inorganic nitrogen was found in electrical conductivity changes over time, as temporal fluctuations in this parameter lead to peaks in the late 1990s and around 2010. For the most part, conductivity levels are ideal, but in the last few years of the data record, levels shift until a large proportion of samples would be classed as acceptable. Over the monitored period, there is an overall increase in conductivity levels.

pH levels in samples in this dataset vary with time, shifting from an annual median of 7.7 to 8.3. Samples from the lower periods would be classed as ideal, while those from period with higher pH levels would be classed as acceptable. Elevated pH levels were found over the period 1997 to 2001. While there are significant changes with time, no overall increasing or decreasing trend is apparent in the dataset.

Chemical weathering index scores are high at this site compared to sites in the upper Olifants River catchment, but are substantially reduced compared to Waaikraal (B2H007) in the upper Wilge River catchment. Scores of this index show an overall decrease with time, suggesting that anthropogenic disturbances are mounting above this site. Median scores for years in the early 1990s lie between 70 and 80, while those from 2008 to 2010 are around 51.

The sulphate contamination index scores show an overall trend of increasing over time indicating that the relative dominance of sulphate in the ionic complement of water at this site is increasing. Two peaks can be seen on the plot that correspond with those noted for electrical conductivity and inorganic nitrogen. These are associated with increases in the absolute sulphate levels, which are nevertheless largely ideal throughout the data record using the classification of DWA (2011a). Peaks of calcium are found that match those referred to above, which suggest the use of lime in treating effluent. Calcium levels are acceptable throughout but also show an increasing trend with time.

The levels of the chloride salinization index are low throughout, and, while there is temporal change, do not show and increasing or decreasing trend. Absolute chloride levels are ideal throughout (after DWA 2011a).

The sodium adsorption ratio does not show a clear changing trend, but has the sample temporal fluctuation noted for several other parameters where peaks in the late 1990s and around 2010 are observed. As this index is based to a large extent on relative amounts of sodium, magnesium and calcium in samples, this suggests that increases in electrical conductivity, inorganic nitrogen, etc. are accompanied by changes in sodium, and inspections of the data confirm this (though sodium levels remain ideal after DWA (2011a)). As chloride levels do not show the same pattern, it would appear that simple dilution or lack thereof does not account for these trends.

Finally, the corrosion potential ratio shows a steady increase with time across the dataset and, by the end of the sampled period, is nearing the upper bound of acceptable. As such, risks of using water from this site are increasing and show no sign of slowing.

Zusterstroom (B2H015) on the Wilge River

This site is on the lower Wilge River before it joins the Olifants River. Land uses in the catchment include those mentioned for Waaikraal (B2H007) and Onverwacht (B2H014) as well as further agricultural, mining, residential and nature conservation uses. The data record for this site extends from 1994 until 2009 and contains 503 records. Data from this site are presented in Figure 65.

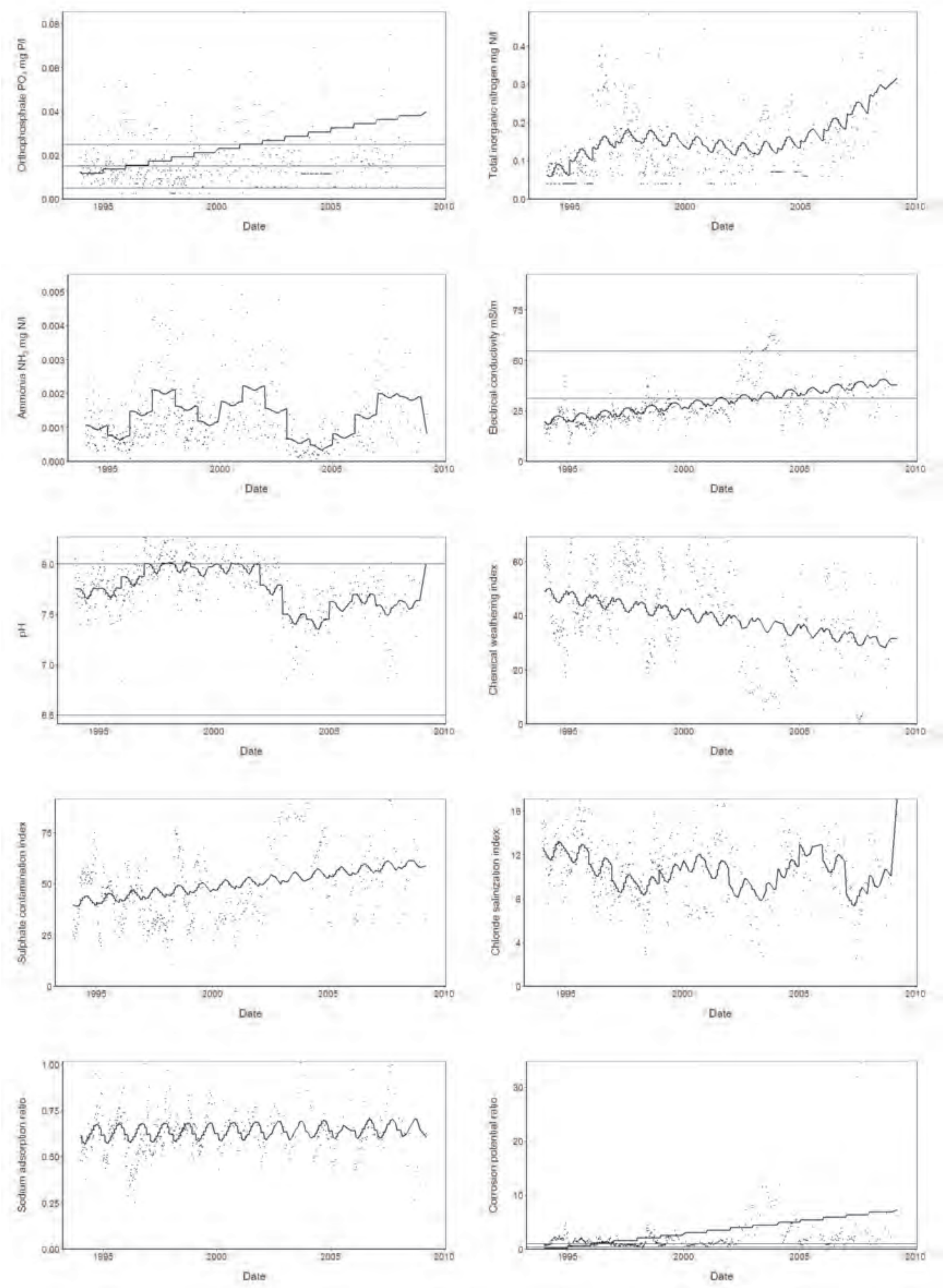

Figure 65 Temporal trends in several water quality parameters at monitoring point B2H015 at Zusterstroom on the Wilge River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate p=0.174; TIN p<0.001; ammonia p<0.001; EC p=0.016; pH p<0.001; chemical weathering index p=0.014; sulphate contamination index p=0.018; chloride salinization index p<0.001; SAR p=0.453; and CPR p=0.065.

Orthophosphate levels are very variable and the model fitting process was not able to satisfactorily fit a model to the orthophosphate data from this site. Inspection of the plot in Figure 65 suggests that an increase in levels of this parameter across the data set may have occurred. Annual median levels are acceptable or tolerable throughout the data record.

Levels of total inorganic nitrogen vary with time, but show a distinct increasing trend towards the end of the data record. Levels of this parameter are ideal throughout the data record. Un-ionized ammonia levels likewise show temporal variation, but no overall trend is apparent and levels are ideal throughout the data record.

Electrical conductivity levels show a largely steady and consistent increase throughout the data record. Overall levels change from ideal in the early to mid data record to acceptable towards the end. Around 2003 there is a peak in conductivity levels that the model fitting process did not detect. This peak is matched by changes in the chemical weathering index, the sulphate contamination index, the chloride salinization index and the corrosion potential index (and is commonly not detected by the model fitting process).

pH levels show some fluctuation with time but no clear overall increasing or decreasing trend. For the greater part of the data record levels would be classed as ideal; however, for a period from around 1997 to 2002, slightly elevated overall levels lead to have values greater than 8, and therefore to an acceptable classification.

Levels of the chemical weathering index are highly variable, but show a decrease over the (fairly short) data record at this site. This suggests that anthropogenic contamination of water at this site is becoming more and more important in determining the ionic complement of water here. The levels toward the end of the monitored period are approaching levels typical of impacted sites in the upper Olifants River catchment. As is also common in those sites, changes in the chemical weathering index seem to primarily be a function of changes in the sulphate contamination index and consequently absolute sulphate levels. Absolute sulphate levels do increase with time, and also show a sharp and major increase around 2003 that matches changes in several water quality parameters. Absolute sulphate levels are largely ideal until 2003, but thereafter levels that are acceptable, tolerable or worse are encountered (following DWA 2011a). Trends in absolute levels of calcium in water at this site largely follow those shown by sulphate, and it likely that these derive from liming of sulphate-rich effluent. Most calcium records would be classed as acceptable after DWA (2011a).

The chloride salinization index showed considerable variation with time but no overt trend, and the scores of this index are low suggesting salinization is low. Absolute chloride levels are all ideal following DWA (2011a).

While seasonality in the sodium adsorption ratio is apparent ($p<0.001$) there is no detectable temporal trend in this index. Levels throughout are ideal. However, levels of the corrosion potential index are high and show an increasing trend. This seems largely to be driven by changes in dissolved sulphate levels.

Downstream of Loskop Dam (B3H017) on the Olifants River

This site is located immediately downstream of Loskop Dam on the Olifants River. Land uses upstream of this include those mentioned for all upstream sites, viz. mining, power generation, agriculture, residential, and nature conservation. The data record stretches from 1993 until the present day and contains 498 records. Figure 66 contains trends in selected water quality parameters from this site.

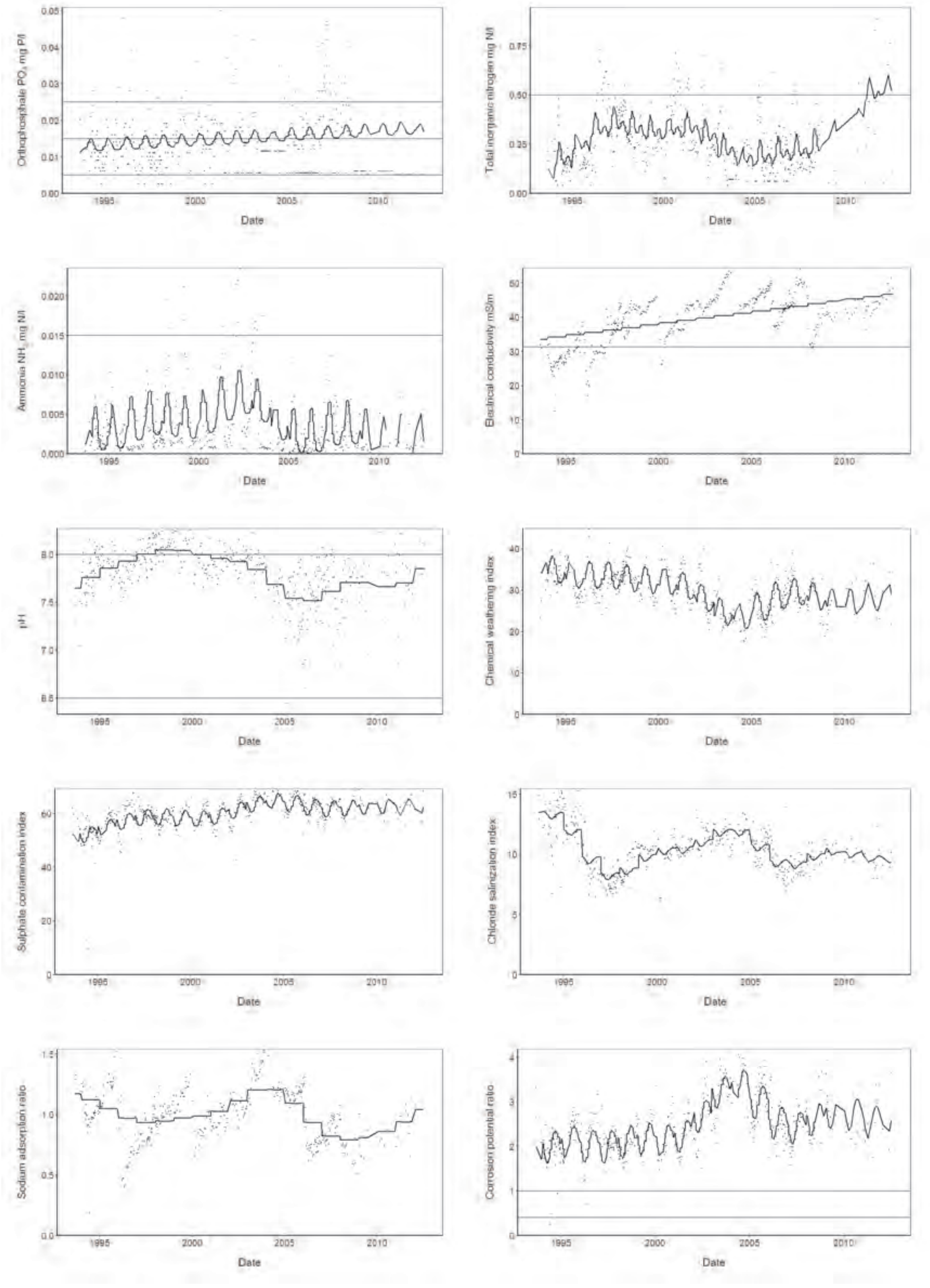

Figure 66 Temporal trends in several water quality parameters at monitoring point B3H017 downstream of Loskop Dam on the Olifants River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate p=0.042; TIN p<0.001; ammonia p<0.001; EC p<0.001; pH p<0.001; chemical weathering index p<0.001; sulphate contamination index p<0.001; chloride salinization index p<0.001; SAR p<0.001; and CPR p<0.001.

Orthophosphate levels at this site are variable, as is common in sites covered by this report, and show a steady increase with time. The fitted model did not reflect a shift to low values for the last 4 years of the data record that is apparent in the plot of the data. Levels of orthophosphate were mostly acceptable to tolerable at the start of the data record, shift to tolerable or worse with time, and after 2009 levels drop back to ideal to acceptable.

Total inorganic nitrogen levels vary with time in this dataset, increasing from 1993 until around 1996, where levels remained until around 2002, whereafter levels decreased again until about 2005. Nearly all samples until this point are classed as ideal. Thereafter levels increase until the end of the dataset. This increase is based on infrequent samples with very variable nitrogen levels, however, and should be accepted with care.

Un-ionized ammonia levels vary with time across the dataset, are strongly seasonal ($p<0.001$), but show no increasing trend overall and are classed as ideal throughout.

The model fitting algorithm was not able to derive a model that closely fitted the data on electrical conductivity at this site; however the simple linear increasing trend line fits the data well enough to be significant. Conductivity increased from the start of the monitored period, when the majority of samples were classed as ideal, to an acceptable level later in the dataset. Inspection of the data in Figure 66 suggests little overall increase since about 2005.

Trends in pH at this site seem to be influenced by trends at Zusterstroom (B2H015) in the lower Wilge River. The pH fluctuates with time, with a period of increased pH levels from the late 1990s to the early 2000s, and lower levels thereafter. With the exception of the period of elevated pH levels, pH is for the most part ideal.

Trends in the chemical weathering index reveal that water quality, which did not reflect a strong influence of natural weathering processes at the start of the monitored period, decreased with seasonal variation ($p<0.001$) across the monitored period. There is a distinct dip in the index centred around 2004. It has been noted at other sites that changes in the chemical weathering index are commonly driven by changes in sulphate levels, and in this case the same applies. With regard to the 2004 dip in chemical weathering index scores, this is matched by changes in sulphate levels (and, as is common elsewhere, calcium levels) and also chloride levels, and is therefore driven by increases in two ions commonly associated with water quality changes. Levels of sodium are not used in calculation of these indices, and these increase also. This change seems therefore likely to be associated with decreased dilution or increased upstream evapotranspiration resulting in greater concentrations of a the less labile ions in solutions.

As noted above, sulphate contamination index scores increased across the monitored period, to end in values that indicate considerable dominance of sulphate in water from this site. Absolute sulphate levels increased from ideal to acceptable across the monitored period (following DWA 2011a).

Chloride salinization index scores are low throughout the monitored period, decreasing sharply from the start of monitoring, and showing a peak around 2004 as noted above. Absolute chloride levels are ideal throughout the monitored period (following DWA 2011a).

The sodium adsorption ratio in this catchment fluctuates with time, and shows a slight increase around 2004 as noted above, but remains ideal and, from the perspective of sodicity, the water from this site is suitable for irrigation. The corrosion potential ratio, on the other hand, is beyond acceptable throughout. It shows the same peak in 2004 as noted in several parameters assessed here.

Mosesriviermond (B3H005) on the Moses River

This site is located low on the Moses River just before it confluence with the Olifants River. Land use in the catchment includes agriculture (often intensively irrigated) mostly in the mid to lower catchment, and residential use with some coal mining in the upper catchment. The dataset for this site stretches from 1976 to 1987, with a few sparse samples thereafter. This data is not ideal to assess trends to the current day; however, this reflects on the paucity of data from the mid Olifants River (particularly further downstream from this site). Trend analysis can only correctly be undertaken until 1987. Any data thereafter may act as indicators of later conditions, but in the absence of further data, cannot be used to draw conclusions as to trends after 1987 at this site. The overall picture of this site is of one where water quality decreased considerably from 1976 to 1987 with increasing salinization as a major driver, with the few more recent samples indicating that this trend may have been reversed. Data are presented in Figure 67.

Orthophosphate levels at this site, unusually, decreased from the start of monitoring until 1987, and the few samples thereafter are tolerable or better. Inorganic nitrogen, on the other hand, showed a sharp increase until 1987, indicating a change from ideal to acceptable levels, to acceptable to tolerable level in 1987. Samples thereafter were all ideal, and this trend may have ended, though more data is required for confirmation of this.

Un-ionized ammonia levels showed an increase that paralleled that of inorganic nitrogen, but levels remained ideal throughout.

Electrical conductivity increased sharply from the beginning of monitoring, with levels passing from acceptable to tolerable, and, in some cases, beyond. Variability, whether seasonal ($p=0.009$) or otherwise, was high. The few more recent records are all acceptable.

pH levels show a steady increasing trend from well within an ideal class to mostly ideal to acceptable, with considerable variation in the dataset. Unlike most water quality parameters considered for this site, the trend in pH does not change when the few more recent samples are assessed, as these suggest a continued, slowing increase in pH levels.

The chemical weathering index data from site reveal a site where chemical weathering was more important than salinization and sulphate accumulation in shaping the ionic complement of the water at the start of the data record. At this time, scores of this index were fairly good in comparison to other upper and mid catchment sites. However, over the brief monitored period, the scores of this index fell considerably. More recent data suggest this trend may have been reversed. Changes in the chemical weathering index were driven by changes in sulphate and chloride levels, as evidenced by changes in the sulphate contamination and chloride salinization indices respectively. Absolute levels of sulphate changed from ideal to acceptable or tolerable over the monitored period, while those of chloride changed from ideal or acceptable to tolerable or worse (classifications following DWA 2011a).

The suitability of the from this site for irrigation worsens according to the sodium adsorption ratio scores, but samples from this site are for the most part classified as acceptable throughout. The appropriateness of the water for use from a corrosion viewpoint also worsens and changes from an acceptable to largely a tolerable level. Data from after 1987 suggest that trends in both indices improve considerably.

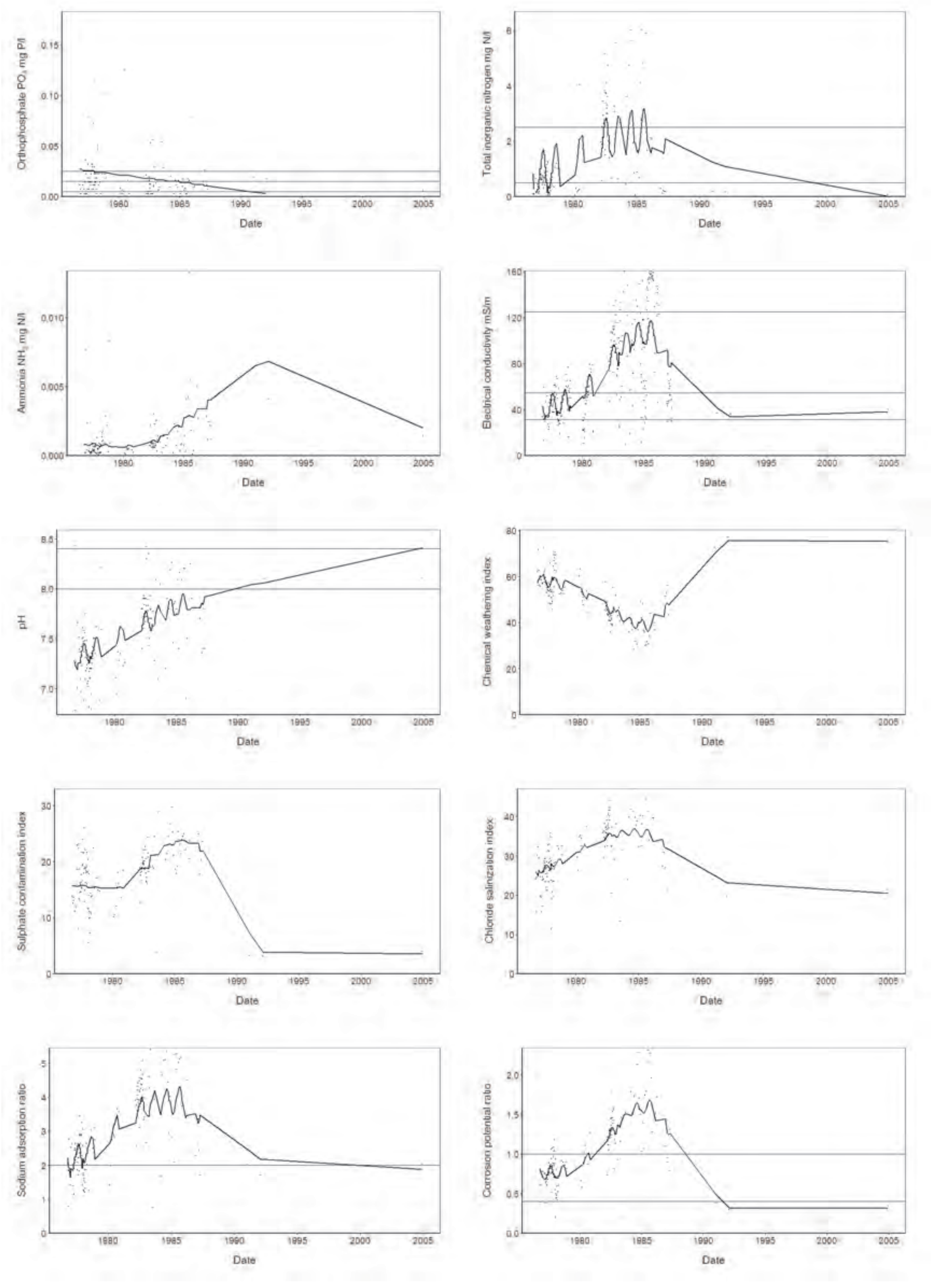

Figure 67 Temporal trends in several water quality parameters at monitoring point B3H005 at Mosesriviermond on the Moses River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate $p=0.036$; TIN $p=0.093$; ammonia $p<0.001$; EC $p<0.001$; pH $p<0.001$; chemical weathering index $p<0.001$; sulphate contamination index $p<0.001$; chloride salinization index $p<0.001$; SAR $p<0.001$; and CPR $p<0.001$.

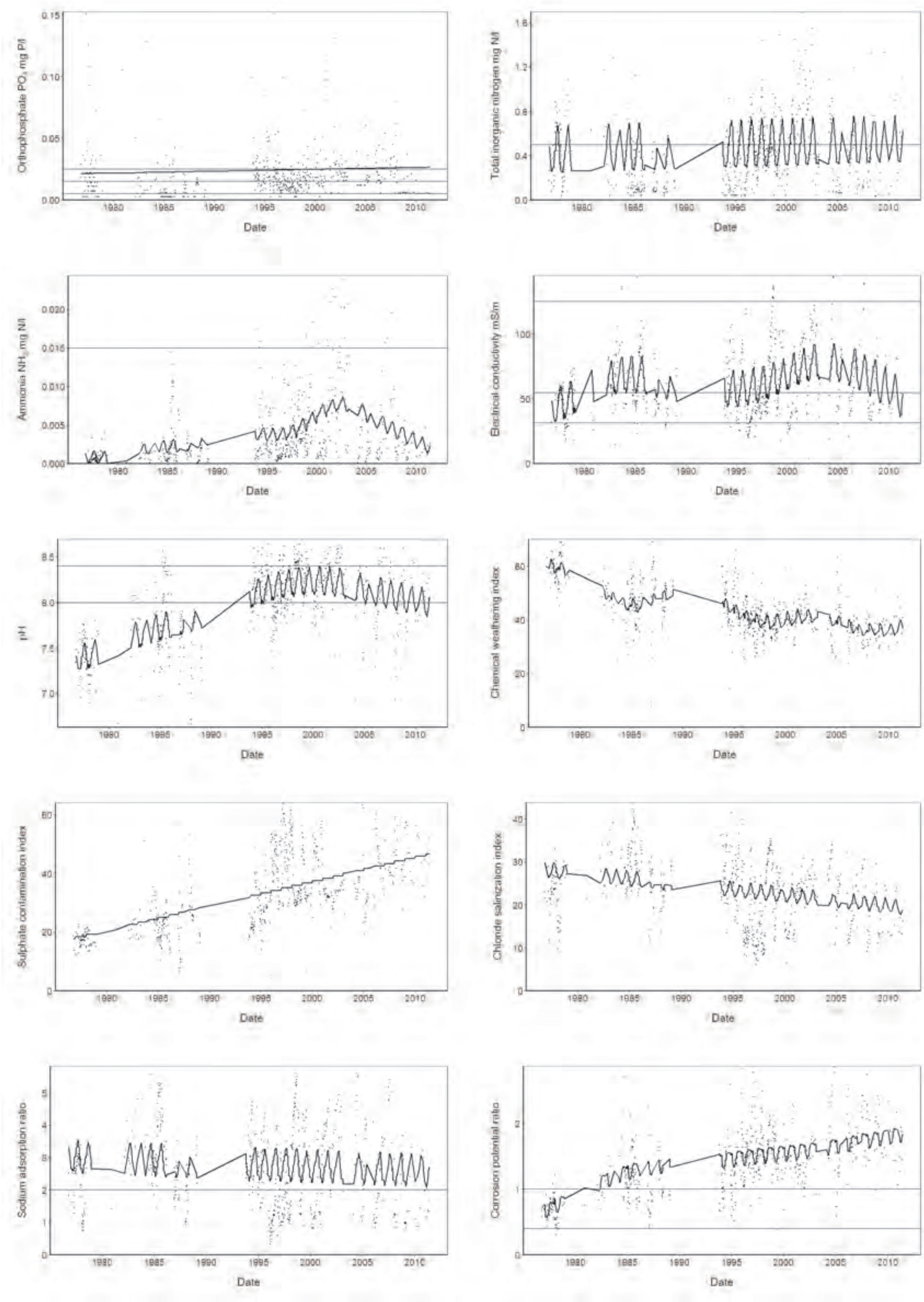

Figure 68 Temporal trends in several water quality parameters at monitoring point B3H001 at Loskop North on the Olifants River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate p=0.695; TIN p=0.417; ammonia p<0.001; EC p<0.001; pH p<0.001; chemical weathering index p<0.001; sulphate contamination index p<0.001; chloride salinization index p<0.001; SAR p=0.237; and CPR p<0.001.

Loskop North (B3H001) on the Olifants River

This site is located on the Olifants River approximately 7km downstream of the confluence of the Olifants and the Moses Rivers. Land uses of the upstream catchment are those listed for upstream sites, together with agriculture (often intensively irrigated) and residential use along the Olifants River. The data record from this site stretches from 1976 until the current day, though data was not collected consistently throughout this period and clear gaps in the record are apparent. There are 724 records from this site. Data on selected water quality parameters are presented in Figure 68. Water quality at this site showed a generally decreasing trend with time, with pronounced seasonal change in most parameters.

Orthophosphate levels at this site were very variable the model fitting algorithm was not able derive a satisfactory model that adequately summarized trends in the data. Inspection of the data indicates that most samples had levels of the nutrient that were classed as acceptable or tolerable for the most part, with a significant number of records classed as worse than tolerable. Between roughly 2002 and 2007 few acceptable records were found, and most records were tolerable or worse. From around 2008 onwards, a significant number of records that were ideal to acceptable are present.

Total inorganic nitrogen levels showed no significant temporal trend but values of this parameter were distinctly seasonal ($p<0.001$). Values of this parameter ranged between ideal and acceptable throughout.

Un-ionized ammonia levels increased with time until approximately 2002, then decreased thereafter. The great majority of data were classed as ideal, with a few classed as acceptable.

Electrical conductivities at this site showed some temporal fluctuation, and pronounced seasonal variation ($p<0.001$), but no clear increasing or decreasing trend over the monitored period was apparent. The majority of data from this site were either classed as acceptable or tolerable, with the majority of samples classed as tolerable in periods when conductivity levels were higher.

pH levels at this site increased from between about 7.0 and 7.5 at the start of monitoring to peak around 8.3 in around 1997, and in the process samples moved from ideal to acceptable and worse classes. Levels remained at this point until about 2005, and thereafter decreased somewhat so that at the end of sampling pH samples were as largely acceptable with some tolerable. As in most parameters at this site, seasonality in this parameter was pronounced ($p<0.001$).

The chemical weathering index declined over the monitored period at this site in a roughly linear fashion, a change that was driven by changes in sulphate at this site, as the other factors used in calculation of this index showed little change over the monitored period. The results of the sulphate contamination index show sulphate contamination increasing fairly linearly over the monitored period. Absolute sulphate levels change from ideal at the start of monitoring to acceptable, tolerable, or worse at the end (following DWA 2011a). As in sulphate-impacted upstream sites, absolute levels of calcium increased together with sulphate, though these remain acceptable to tolerable throughout.

Salinization, as measured by the chloride salinization index, decreased across the monitoring period. This change is driven by increased sulphate, as this affects the index. Absolute chloride levels in fact show an apparent minor increase across the monitored period, but levels remain tolerable or better (following DWA 2011a).

The sodium adsorption ratio does not show any change with time, despite considerable seasonal variation ($p<0.001$). Levels are acceptable throughout. The corrosion potential ratio, on the other hand, shows a steady increase, driven by increases in sulphate in river water. Levels of this index are acceptable at the start of the data record, but from the mid-1980s onwards, scores are not acceptable.

Scherp Arabie (B3H021) on the Elands River

This site is located low on the Eland River, just above the confluence of the Elands River and the Olifants River in the Flag Boshielo Dam. Land uses in the catchment are largely agricultural, with significant levels of irrigated agriculture, and also residential use. The data record for this site extends from 1994 until the present day, and contains 398 records. Data from this site are plotted in Figure 69. In contrast to many of the sites assessed above, this site shows a greater impact of salinization than of sulphate contamination. All water quality variables assessed here show significant seasonal change.

Orthophosphate levels at this site are very variable, and, with the exception of more recent samples, range from acceptable to far beyond tolerable. The fitted model indicates an increase in orthophosphate levels from the start of monitoring in 1994, but with improved orthophosphate levels in recent years, leading to a situation where the median orthophosphate level in the final two years of the data record being 0.005 mg P/ℓ.

Total inorganic nitrogen levels varied over time and fluctuated between ideal and acceptable levels. Since around 2005, inorganic nitrogen levels have showed an increasing trend, so that by the end of monitoring, most samples are classed as acceptable.

Un-ionized ammonia levels were variable but largely ideal throughout the data record. No evidence of a significant change with time (over and above seasonal change) was found.

Electrical conductivity levels show a sigmoid increase from the start of monitoring until the mid to late 2000s, and a decrease thereafter. The is considerable seasonal variation in this parameter ($p<0.001$) and, with the exception of the mid to late 2000s when levels were largely beyond tolerable, electrical conductivities at this site fluctuate between tolerable and intolerable on a seasonal basis.

pH levels at Scherp Arabie show a largely linear, small decrease with time, and in the process decrease from a mostly unacceptable state to an acceptable class.

The chemical weathering index shows no significant trend across the monitored period. However, levels of this index at this site are fairly low. The reason for this is that sulphate and in particular chloride levels in water from this site are elevated. The sulphate contamination index and the chloride salinization index show complementary changes with time, with levels of the chloride salinization index considerably exceeding those of the sulphate contamination index. Absolute levels of sulphate are, with the exception of data from the mid to late 2000s, between ideal and tolerable, with an acceptable median of 141 mg/ℓ (after DWA 2011a). Absolute levels of chloride are acceptable, tolerable or worse, with an overall intolerable median of 177 mg/ℓ (after DWA 2011a).

Results of the sodium adsorption ratio show temporal fluctuation with levels remaining acceptable for irrigation throughout the data record. Absolute levels of sodium range from ideal to intolerable, with an intolerable median of 125 mg/ℓ (after DWA 2011a). Levels of the corrosion potential ratio are not acceptable throughout the data record. These results are a function of elevated chloride, and to a lesser extent sulphate, levels.

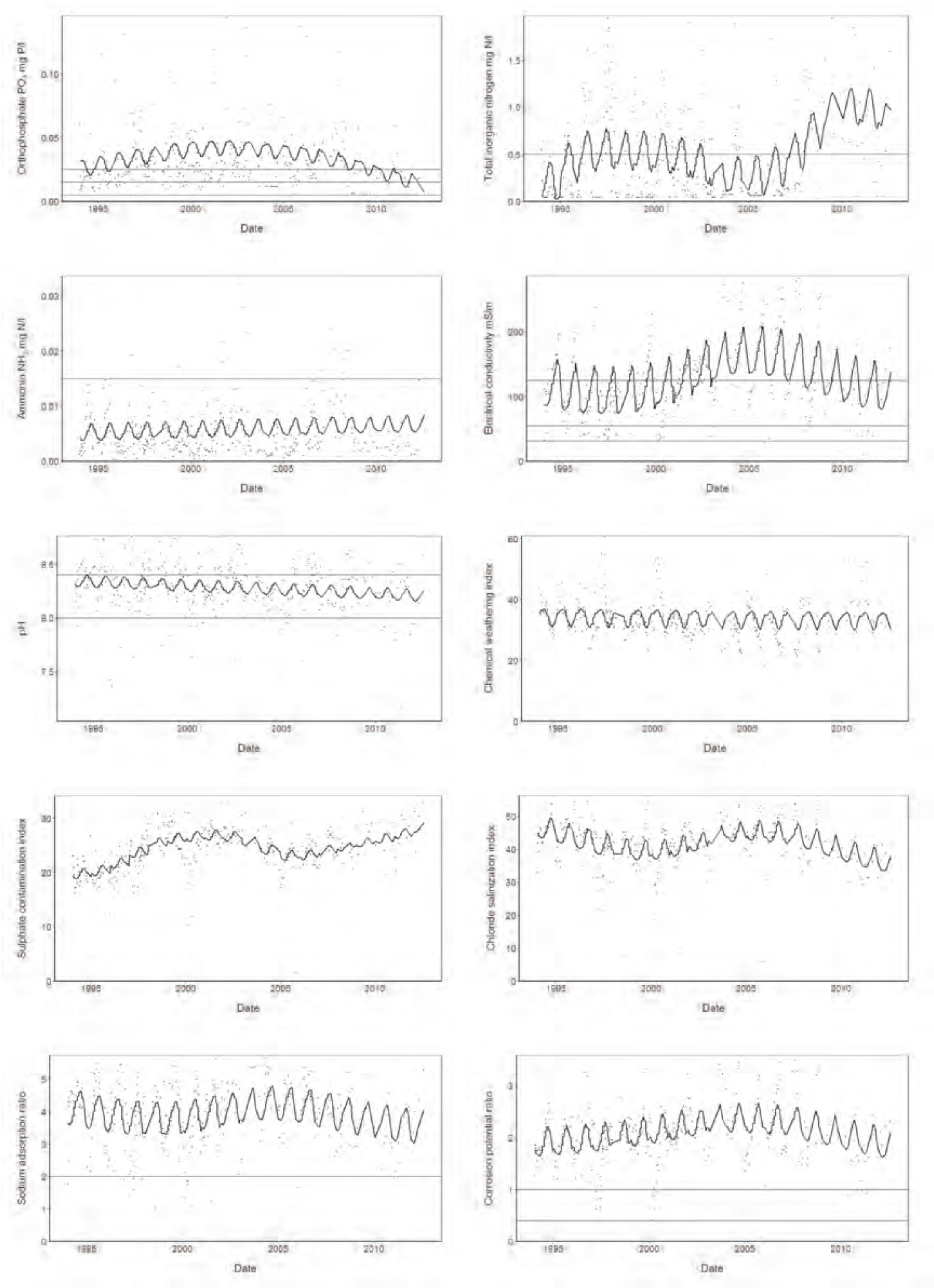

Figure 69 Temporal trends in several water quality parameters at monitoring point B3H021 at Scherp Arabie on the Elands River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate $p=0.002$; TIN $p<0.001$; ammonia $p=0.377$; EC $p<0.001$; pH $p=0.009$; chemical weathering index $p=0.590$; sulphate contamination index $p<0.001$; chloride salinization index $p<0.001$; SAR $p=0.015$; and CPR $p=0.003$.

Downstream of the Flag Boshielo (Arabie) Dam (B5H004) on the Olifants River

This site is located below the Flag Boshielo Dam on the Olifants River, approximately 20 km downstream of the confluence with the Elands River. This the last monitoring point with long term data on the Olifants River for a considerable distance. Land uses along the river include agriculture (some irrigated agriculture but largely subsistence agriculture) and nature conservation. The data record from this site starts in 1993 and extends to the current day, with 547 records available for analysis here. Many of the water quality parameters assessed show considerable fluctuation with time (Figure 70). A trend apparent at this site, and often not captured by the model, is medium term increases in several parameters followed by sharp decreases.

Orthophosphate levels were mostly acceptable to tolerable through most of the data record, with samples since 2010 having median orthophosphate levels on the boundary between ideal and tolerable (recorded detection limits on orthophosphate analyses do not allow lower values). The overall median phosphate levels were 0.015 mg P/ℓ, on the boundary between acceptable and tolerable. There appears to have been a slight increase in this parameter from the start of the data record until around 2008; thereafter, orthophosphate levels from this site decrease sharply.

Total inorganic nitrogen levels vary over time, with a peak in the late 1990s, and lower values at sampling start and in the mid to late 2000s. Values throughout the data record are largely ideal.

Variation in un-ionized ammonia levels is high, though no overt long term change was detected and levels are ideal throughout the data record.

Electrical conductivity levels show a trend of medium-term increases followed by sharp drops, a pattern suggesting that level of this parameter are affected by weather and flow pattern changes. Inspection of DWA (Hydrology) data confirms that sharp drops correlate with high flows at Loskop North (B3H001) just upstream of the dam. Levels of this parameter were acceptable or tolerable over the monitored period.

pH levels were very consistent, with some seasonal variation ($p<0.001$) from the start of monitoring until 2005, when values of this parameter would mostly be classed as acceptable. After 2005, pH levels at this site drop slightly, and samples are classed as ideal or acceptable. This change is correlated with a decrease in pH at Loskop North upstream.

The chemical weathering index at this site remains fairly stable with time and indicates that chemical weathering is not the major contributor to the ionic complement of the water at this site. Although there is no overall change in the chemical weathering index, this masks changes that occur in the indicator ions chloride and sulphate. Changes in the sulphate contamination index and the chloride salinization index are to a large extent complementary. The former fluctuates between 20 and 50, indicating a significant to high contribution of sulphate contamination to the ionic complement of water at this site. The range of changes in the chloride contamination index is lower at roughly 10 to 30, which still indicates an at times significant contribution of salinization to ions in water at this site. Absolute levels of sulphate at this point increase with time from ideal to acceptable to only a certain extent follow the same trends displayed by conductivity levels at this site (classification after DWA 2011a). Calcium ions are acceptable throughout and show no increase with time and also follow conductivity trends to a certain extent. Absolute chloride levels, however, vary from ideal to acceptable and strongly follow the trends of conductivity, and trends in sodium levels are similar.

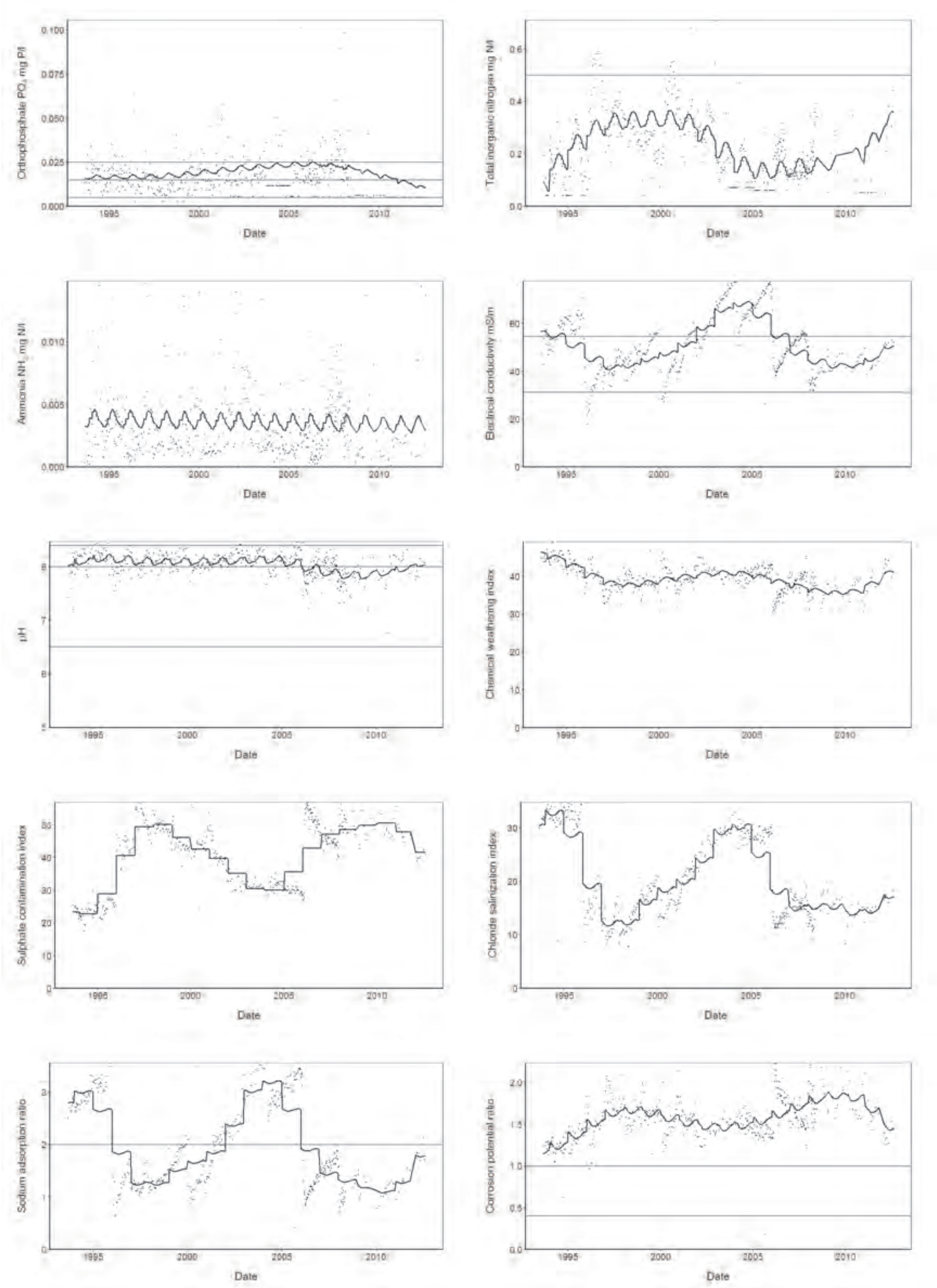

Figure 70 Temporal trends in several water quality parameters at monitoring point B5H004 downstream of the Flag Boshielo (Arabie) Dam on the Olifants River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate $p=0.044$; TIN $p<0.001$; ammonia $p=0.474$; EC $p<0.001$; pH $p<0.001$; chemical weathering index $p<0.001$; sulphate contamination index $p<0.001$; chloride salinization index $p<0.001$; SAR $p<0.001$; and CPR $p<0.001$.

As a result of the ionic changes discussed above, the sodium adsorption ratio exhibits a trend that broadly follows that of conductivity at this site, moving from ideal to acceptable when flows were low, and decreasing after high flow periods. The corrosion potential ratio is more stable and, driven by significant levels of chloride and or sulphate, is worse than acceptable throughout the data record.

Buffelskloof (B4H003) on the Steelpoort River

This site is in the upper catchment of the Steelpoort River, approximately 15km upstream of the De Hoop Dam. Land use in the catchment is largely agricultural, with some mining and residential use. The data record at this site extends from 1977 to the current time and contains 1274 records. Plots with the data and fitted models are presented in Figure 71. The data presented show this site to be one with generally good, if very variable, water quality. All water quality parameters assessed, bar orthophosphate, showed significant seasonal trends.

Orthophosphate levels at this site are very variable, a common observation with regard to this water quality parameter. At the start of sampling, the majority of samples are ideal or acceptable. Median levels increase steadily with time however, until from 2002 until 2008, median levels of this parameter approach or reach levels of 0.02 mg P/ℓ, in the centre of the tolerable class. At this time, a significant number of samples exceed the upper boundary of the tolerable class. From 2009 onwards, median levels of this parameter decrease considerably, so that median orthophosphate levels at this site are at the boundary between ideal and acceptable.

Total inorganic nitrogen levels are likewise very variable, but the great majority of samples from this site are classed as ideal. The fitted model indicates a slight increase with time in this parameter.

Un-ionized ammonia levels are low throughout the sampling record and very few samples have levels that are not ideal. There does not seem to be any overt increasing or decreasing trend with time.

Electrical conductivity levels changed slightly over the monitored period, with the greater part of changes in this parameter showing a seasonal pattern ($p<0.001$). Levels of this parameter shifted slightly with time, but no overall increasing or decreasing trend is observed. Samples were classed as ideal or acceptable throughout the data record.

pH levels at this site did not change over the first few years of the data record, and median levels of this parameter were around 7.6. From roughly 1990 to 1995, pH levels increased to a median of around 8.3. From the early to mid-2000s, a slight decrease in pH was observed. This trend of increasing pH levels in the early 1990s, followed by a maintained or slightly decreasing pH thereafter, is common at sites in the Crocodile catchment. In this light, it is worth noting that this sample site is only about 60 km from the uppermost sample site in the Crocodile River.

Scores of the chemical weathering index are good throughout and little change with time is noted. Temporal changes in the sulphate contamination and chloride weathering indices are apparent in Figure 71; however, scores of both these indices are low and absolute levels of sulphate and chloride are classed as ideal throughout the data record (following DWA 2011a).

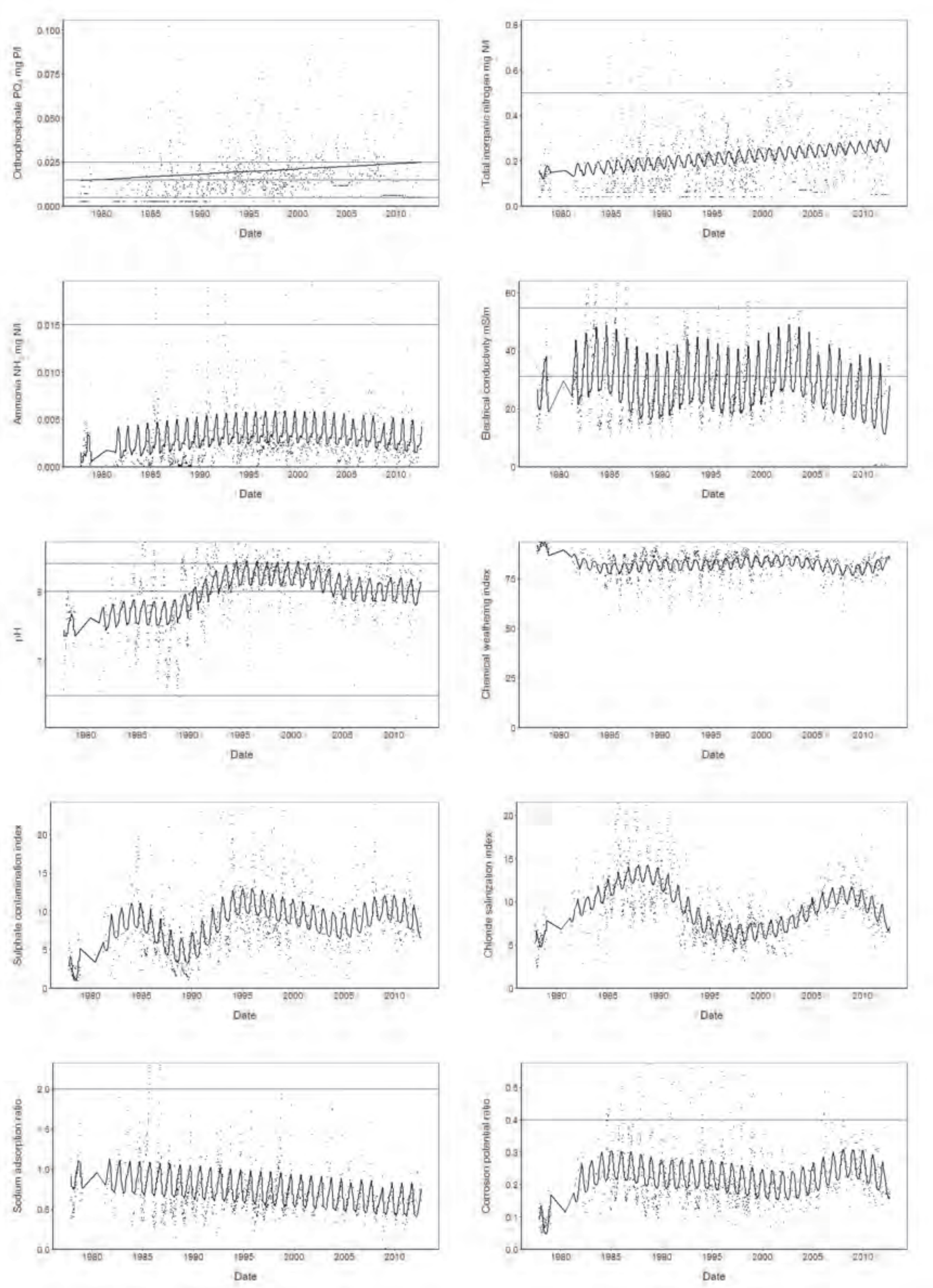

Figure 71 Temporal trends in several water quality parameters at monitoring point B4H003 at Buffelskloof on the Steelpoort River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate p=0.221; TIN p=0.001; ammonia p=0.001; EC p<0.001; pH p<0.001; chemical weathering index p<0.001; sulphate contamination index p<0.001; chloride salinization index p<0.001; SAR p<0.001; and CPR p<0.001.

Sodium adsorption ratio scores are low and classed as ideal throughout, and show a decreasing trend with time. Corrosion potential ratio scores show an increase at the start of the data record that is soon arrested, and thereafter score change little and are largely ideal throughout the rest of the data record. As a consequence, the water at this site appears suitable for use by a range of potential end users.

Potloodspruit (B4H007) on the Little Spekboom River

This site is upstream on the Little Spekboom River in an area with few identified impacts. This river flows into the Spekboom River, which in turn joins the Steelpoort River in the Burgersfort area. Land use in this catchment is limited, with limited agriculture and nature conservation as major land users. The data record at this site starts in 1977 and extends until the current day, and contains 1183 records. The data plotted in Figure 72 below show a site with generally good water quality that changes little with time but, with the exception of orthophosphate, displays significant seasonal variation.

Orthophosphate levels increase across the dataset from this site until 2008, whereafter median levels drop suddenly. In the process, orthophosphate levels in samples from this site change from largely ideal to acceptable, to acceptable to tolerable, and at the end of sampling end on the border between ideal and acceptable. Throughout the dataset, the majority of samples are tolerable or better.

No changes over time in the levels of inorganic nitrogen are found. Samples throughout are classed as ideal.

Un-ionized ammonia levels show a slight increasing trend from the start of monitoring; however levels encountered are far below the upper boundary of ideal.

Electrical conductivity at this site changed little with time although seasonal variation was profound. All samples were classed as ideal.

The pH at this site showed the same pattern of change noted at Buffelskloof (B4H003) and a number of sites in the Crocodile River catchment. After a period of relative stability at the start of the data record, pH levels increase from the late 1980s to around 1995, and change little thereafter except to decrease slightly later in the data record. For much of the record the pH levels are ideal, but from approximately 1995 to 2000, levels overall drop to being acceptable.

The chemical weathering index scores from this site are high indicating that natural weathering processes are the major contributors to the ionic complement of samples from this site. There is relatively little change with time, though seasonal variation is pronounced ($p<0.001$). Changes in the chemical weathering index can be explained by changes in the relative amounts of sulphate and chloride in samples. Although changes do occur over time, the magnitude of such changes is small, and the absolute amount of sulphate and chloride in samples from this site is ideal according to DWA (2011a).

The sodium adsorption ratio score for this site changes little with time, and, according to this index, water from this site is suitable for irrigation. The corrosion potential ratio varies with season ($p<0.001$) to the extent that the index changes from ideal to acceptable on an annual basis. Beyond this, no significant change in the index with time was detected.

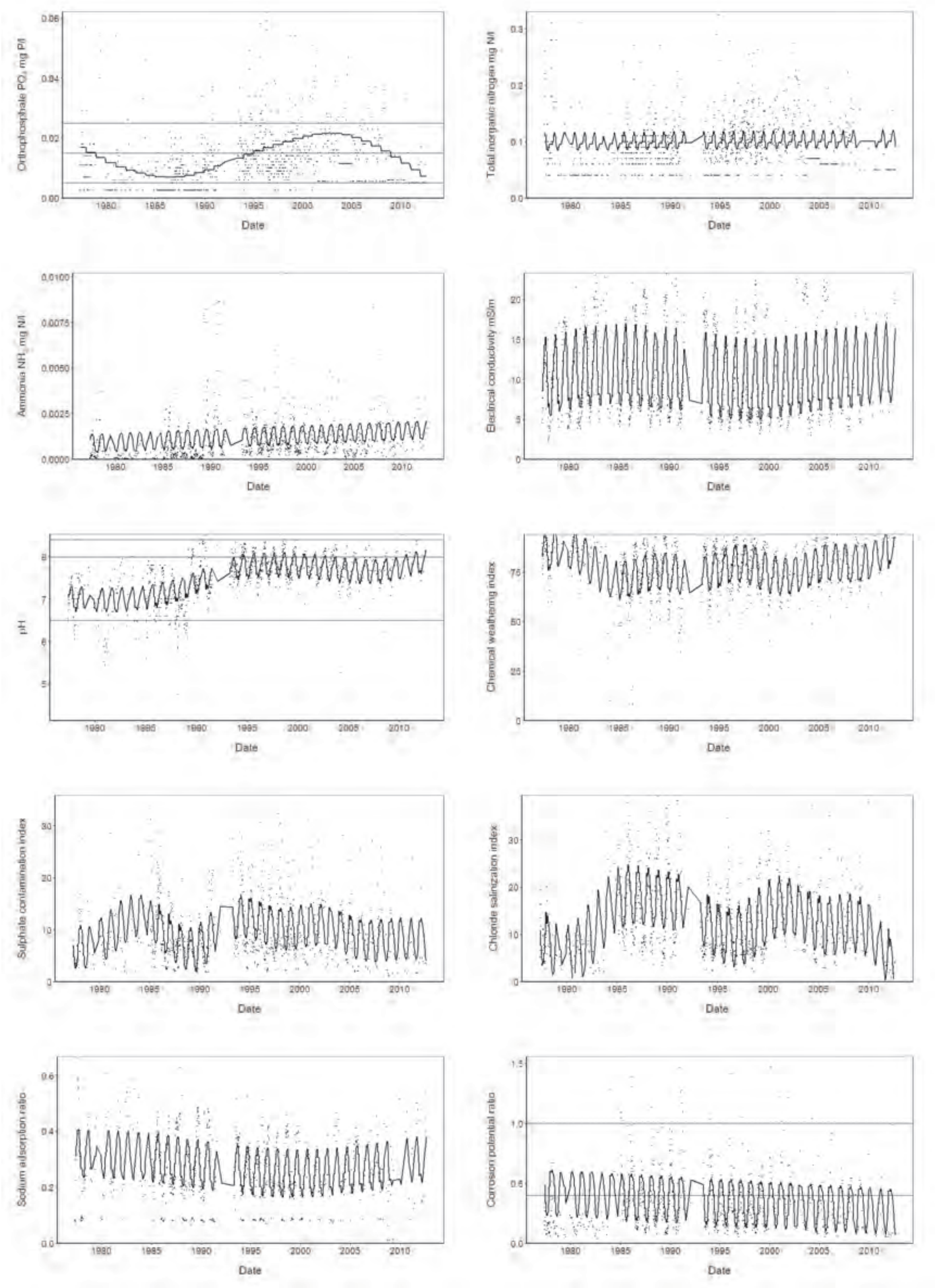

Figure 72 Temporal trends in several water quality parameters at monitoring point B4H007 at Potloodspruit on the Little Spekboom River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate $p<0.001$; TIN $p=0.634$; ammonia $p=0.085$; EC $p=0.015$; pH $p<0.001$; chemical weathering index $p<0.001$; sulphate contamination index $p<0.001$; chloride salinization index $p<0.001$; SAR $p=0.052$; and CPR $p=0.175$.

Alverton (B4H011) on the Steelpoort River

This site is located relatively low on the Steelpoort River, about 23 km before the confluence with the Olifants River. At this point land uses include those mentioned for sites in the upper catchment, together with further agriculture and considerable residential and mining activity. The data record for this site stretches from 1984 until the current day; however, two gaps are present in the data. The first stretches from the early to late 1990s, and the second is centred around the year 2000. The data record contains 605 records. The data from this site are plotted in Figure 73. A worsening of water quality in comparison with upstream sites is apparent; however, the change is fairly minor and many of the parameters assessed here show improvements in recent times. As in upstream sites in the Steelpoort catchment, seasonal variability is pronounced and all parameters in Figure 73 show statistically significant seasonal variation. Seasonality in several parameters decreases with time after the start of monitoring.

At the start of the data record, orthophosphate levels at this site fluctuated for the greater part between acceptable and tolerable (though with several ideal and fewer intolerable records). Levels thereafter increase slowly until about 2003, when the rate of increase accelerates, leading to a peak in 2007 when the median orthophosphate concentration was a worse than tolerable 0.033 mg P/ℓ. Levels thereafter decrease rapidly to the boundary between acceptable and ideal.

Inorganic nitrogen levels at the start of sampling were acceptable, and samples showed strong seasonal variation ($p<0.001$). No real change in this parameter was detected until levels decreased in the early 2000s to a point where levels switched seasonally between ideal and acceptable.

Un-ionized ammonia levels were ideal throughout the data record, and decreased with time.

Electrical conductivity levels at the start of the data record were fairly high though still mostly tolerable and highly variable seasonally ($p<0.001$). Over time, conductivity levels decrease slowly until at the end of the data record, most records are either ideal of acceptable. The marked seasonality in this parameter also decreased with time.

pH levels at the start of sampling were fairly high compared to upstream sites, and highly variable seasonally ($p<0.001$). Levels increased with time until the mid to late 1990s, after which they decreased somewhat until at the end of sampling levels were much the same as when sampling started. The increase in the 1990s, with a slight decrease thereafter, mirrors changes at upstream sites, though the magnitude of change at Alverton is far smaller.

The chemical weathering index was fairly highly at this site and no change with time that could not be attributed to seasonal patterns was detected. There was an increase of levels of the sulphate contamination index over time; however, levels of this parameter remain low and sulphate cannot be considered a significant contaminant at this site. Absolute sulphate levels are nearly all classed as ideal after DWA (2011a). Relative chloride levels, as reflected in the chloride salinization index, are higher than sulphate, and do not vary significantly with time. Absolute levels of chloride were acceptable or ideal over most of the data record.

The sodium adsorption ratio decreases over time until, at the end of the data record, most samples are ideal. This change is driven by decreasing levels of sodium in the water. The corrosion potential ratio did not change with time and remains at the border between ideal and acceptable throughout.

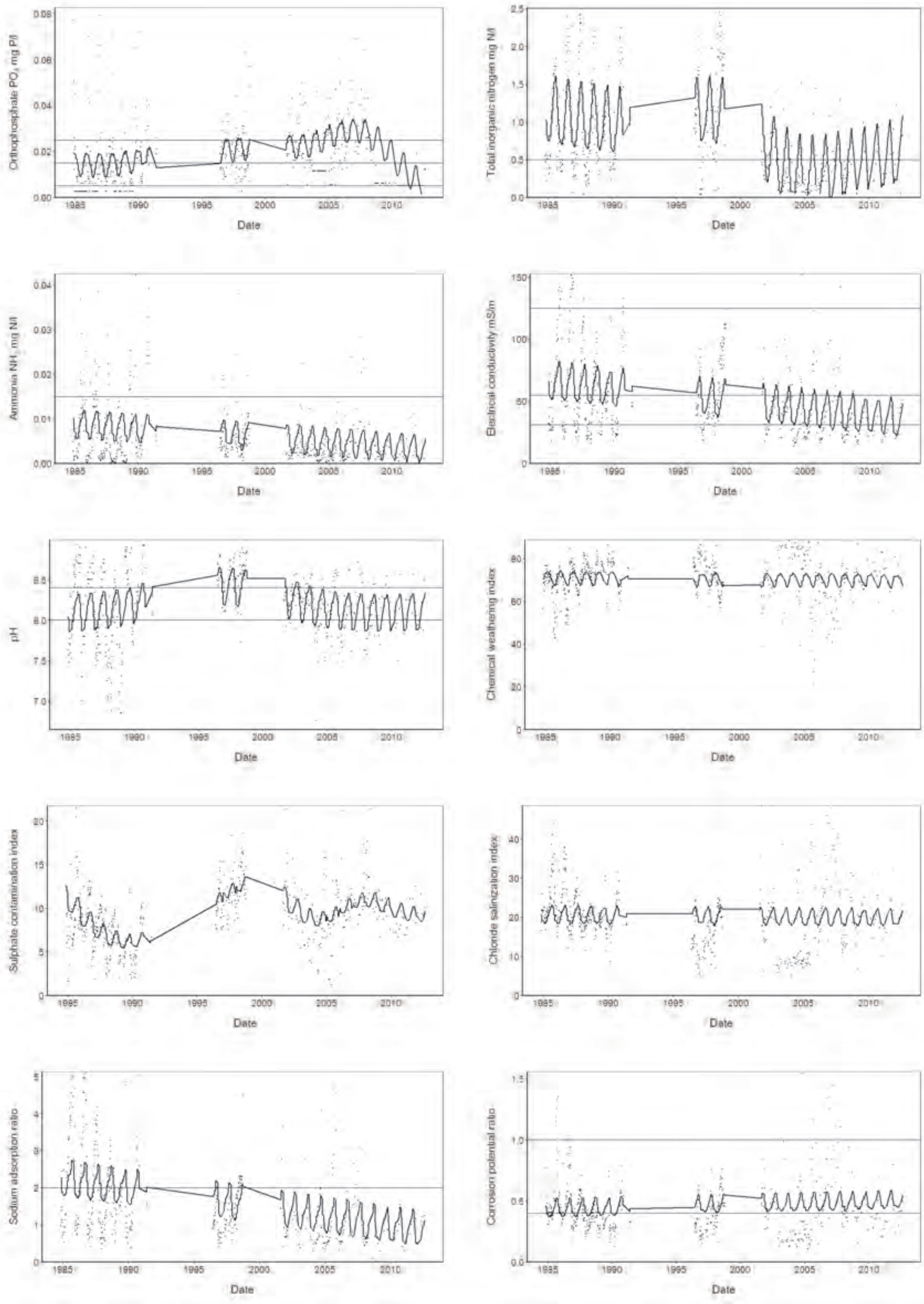

Figure 73 Temporal trends in several water quality parameters at monitoring point B4H011 at Alverton on the Steelpoort River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate $p<0.001$; TIN $p<0.001$; ammonia $p<0.001$; EC $p<0.001$; pH $p=0.004$; chemical weathering index $p=0.706$; sulphate contamination index $p<0.001$; chloride salinization index $p=0.789$; SAR $p<0.001$; and CPR $p=0.508$.

Finale/Liverpool (B7H009) on the Olifants River

This site is the next on the Olifants River approximately 230 km downstream after the Flag Boshielo Dam, after the confluence with the Steelpoort River. It is the first site along this reach with a record set suitable for analysis, and the first site after the river passes through the escarpment. No long term data was available for a long stretch between the sites along the mid Olifants River. Land uses above this point include all those listed for the upper catchment and the Steelpoort River, along with mixed residential, agricultural (commercial and subsistence), mining, nature conservation and tourism, etc. The data record for this site starts in 1962, but only contains a few samples in that and several subsequent years. Sampling in earnest began in 1979, and ended in 2007. The data record here contains 541 samples. Plots of selected water quality parameters for this site are presented in Figure 74. A comparison with upstream data reveals that the most notable change along the river is a decrease in sulphate with consequent improvements in the chemical weathering index and corrosion potential ratio.

Orthophosphate levels at this site are variable and show a steady increase with time, causing median levels of this parameter to increase from ideal to acceptable to the upper bound of tolerable over the monitored period. The common decrease in orthophosphate levels noted in recent samples from other sites was not observed at Finale/Liverpool; however, as this dataset ended in 2007 rather than the current day, one cannot draw any conclusions about this trend at this site.

Inorganic nitrogen levels were largely ideal throughout and showed no significant trend with time. Exactly the same can be said for ammonia levels at this site.

Electrical conductivity at this site did not show any significant trend with time, though seasonal change was pronounced ($p<0.001$). Seasonal changes underlie regular changes in classes of this parameter, with summer levels being on the whole acceptable and winter conductivities only tolerable.

pH levels at this site increased from 1979 until around 1995, then stabilized, and recent samples show an apparent decreasing trend. Although data on the increase is not available from the next upstream site in the river, the following site upstream at Loskop North (B3H001) shows a similar pattern of change. The change shifted pH levels at this site from largely ideal to acceptable or worse.

The results of the chemical weathering index at Finale/Liverpool show regular temporal fluctuation with no overt increasing or decreasing trend. Levels of this index at this site are significantly better than the next site upstream below Flag Boshielo Dam (B5H004). This change between sites is to a large extent a function of decreased relative levels of sulphate at Finale/Liverpool, and absolute levels are largely ideal at this site (classification after DWA 2011a). On the other hand, chloride salinization index levels at Finale/Liverpool are slightly higher than upstream, and absolute chloride levels range between ideal and acceptable (again after DWA 2011a).

The sodium adsorption ratio scores from this site illustrates that, from a viewpoint of sodicity, most samples would be classed as ideal for irrigation. However, pronounced seasonal variation of this index ($p<0.001$) mean that winter months are more likely to be classed as acceptable. Corrosion potential ratio scores at Finale/Liverpool are acceptable throughout and are a clear improvement on scores from upstream.

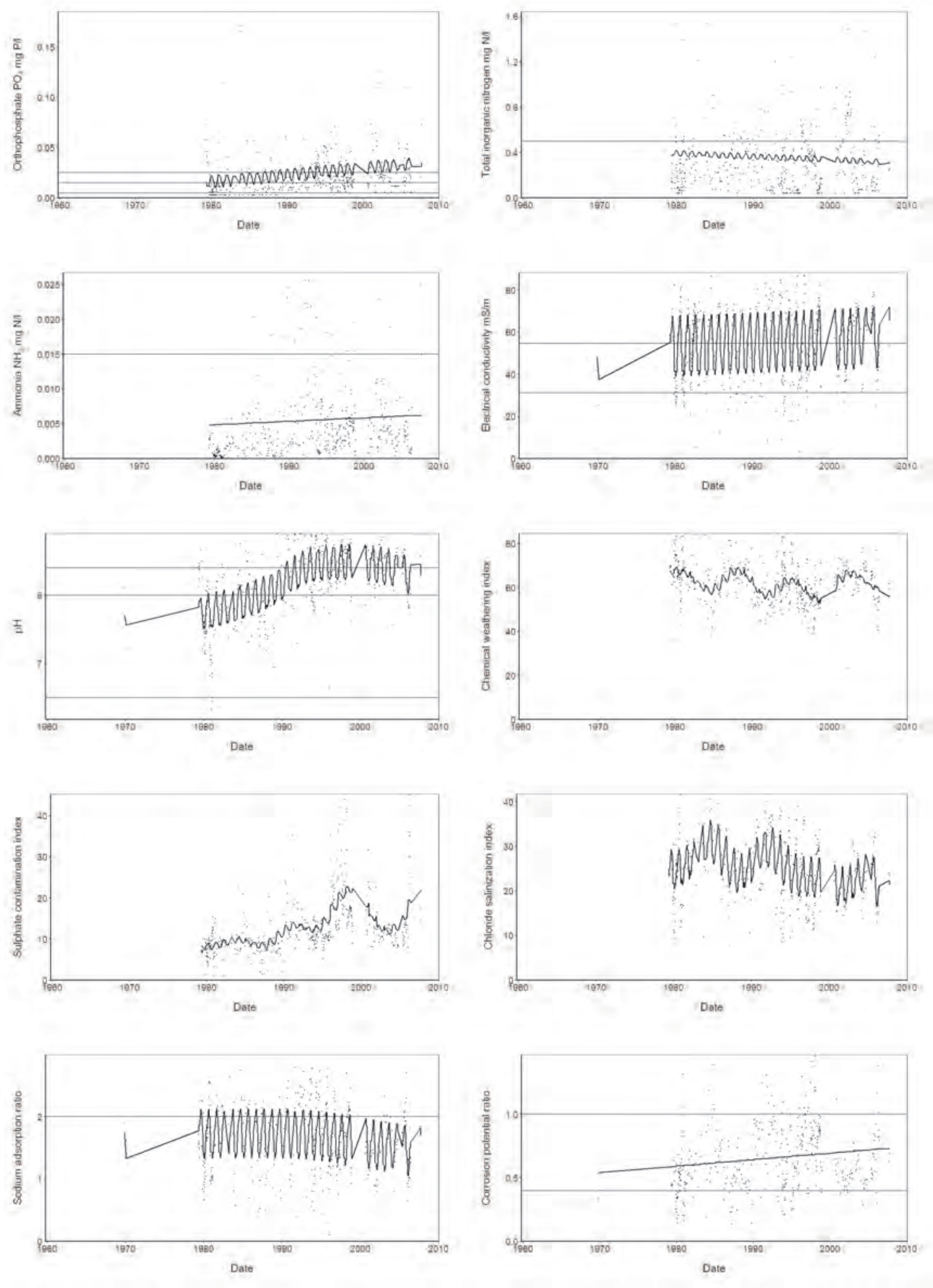

Figure 74 Temporal trends in several water quality parameters at monitoring point B7H009 at Finale/Liverpool on the Olifants River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate p=0.005; TIN p=0.390; ammonia p=0.591; EC p=0.111; pH p<0.001; chemical weathering index p<0.001; sulphate contamination index p<0.001; chloride salinization index p<0.001; SAR p=0.002; and CPR p=0.036.

Willemsoord (B6H001) on the Blyde River

This is the uppermost site on the Blyde River. Land use upstream of this point is dominated by agriculture, forestry and nature conservation/tourism. There is one sample from this site from 1966, and continuous sampling started in 1976 and extends until the current day, leading to 734 samples from this site. The data presented in Figure 75 reveal this site to be one with generally good water quality, although, in common with most sites, orthophosphate levels are a concern. Many parameters have a significant pattern of seasonal variation.

Orthophosphate levels at this site showed a trend to increasing with time, with samples mostly ideal to acceptable at the start of the data record, and acceptable to tolerable in the early 2000s. Recent samples have mostly low scores, and are classed as ideal to acceptable, a common observation in this catchment.

Inorganic nitrogen levels were largely ideal over the monitored period, and no statistically significant temporal trend could be fitted to the data. The same can be said of levels of un-ionized ammonia at this site.

Electrical conductivity levels are low and show some temporal fluctuation, but without any clear increasing or decreasing trend over the monitored period. Seasonal variability is relatively high ($p<0.001$). Levels of this parameter are ideal throughout.

pH levels at this site show the same shift from a lower stable state at the start of monitoring, to increase through the 1990s until stabilizing again at a higher pH as was noted in at sites in the Steelpoort and Crocodile River catchments. pH levels from early in the data record would be classed as ideal, while those from later on would be considered as acceptable.

The chemical weathering index scores from Willemsoord vary little with time and show some seasonal variation ($p<0.001$). Scores of this index are generally high and it appears that the ionic complement of water from this site is largely determined by natural weathering processes. Sulphate contamination index scores were surprisingly high and decreased with time, indicating that sulphate is a significant contributor to the ionic makeup. However, as conductivity levels at this site were low, an indication that sulphate is relatively important does not translate to concerning absolute levels of sulphate, which were ideal throughout the data record (classification after DWA 2011a). Scores of the chloride contamination index were low and variable throughout the data record, and absolute chloride levels are ideal (also after DWA 2011a).

Scores of the sodium adsorption ratio showed an increasing trend with time across the data set, but levels of this parameter remained ideal. The corrosion potential ratio scores varied between ideal and acceptable, with higher scores generally being found in winter and lower scores in summer.

Chester (B6H004) on the Blyde River

This site is the lower of the two sites that were selected on the Blyde River, and lies downstream of the confluence of the Blyde and Ohrigstad Rivers at the Ohrigstad Dam. Major land uses in the catchment are agriculture, forestry, and nature conservation/tourism, with some residential and mining impact. The data record from Chester runs from 1978 until the current data and contains 1015 records. The data presented in Figure 76 show many of the same trends as were observed at the upstream site. All selected water quality parameters had statistically significant seasonal variation

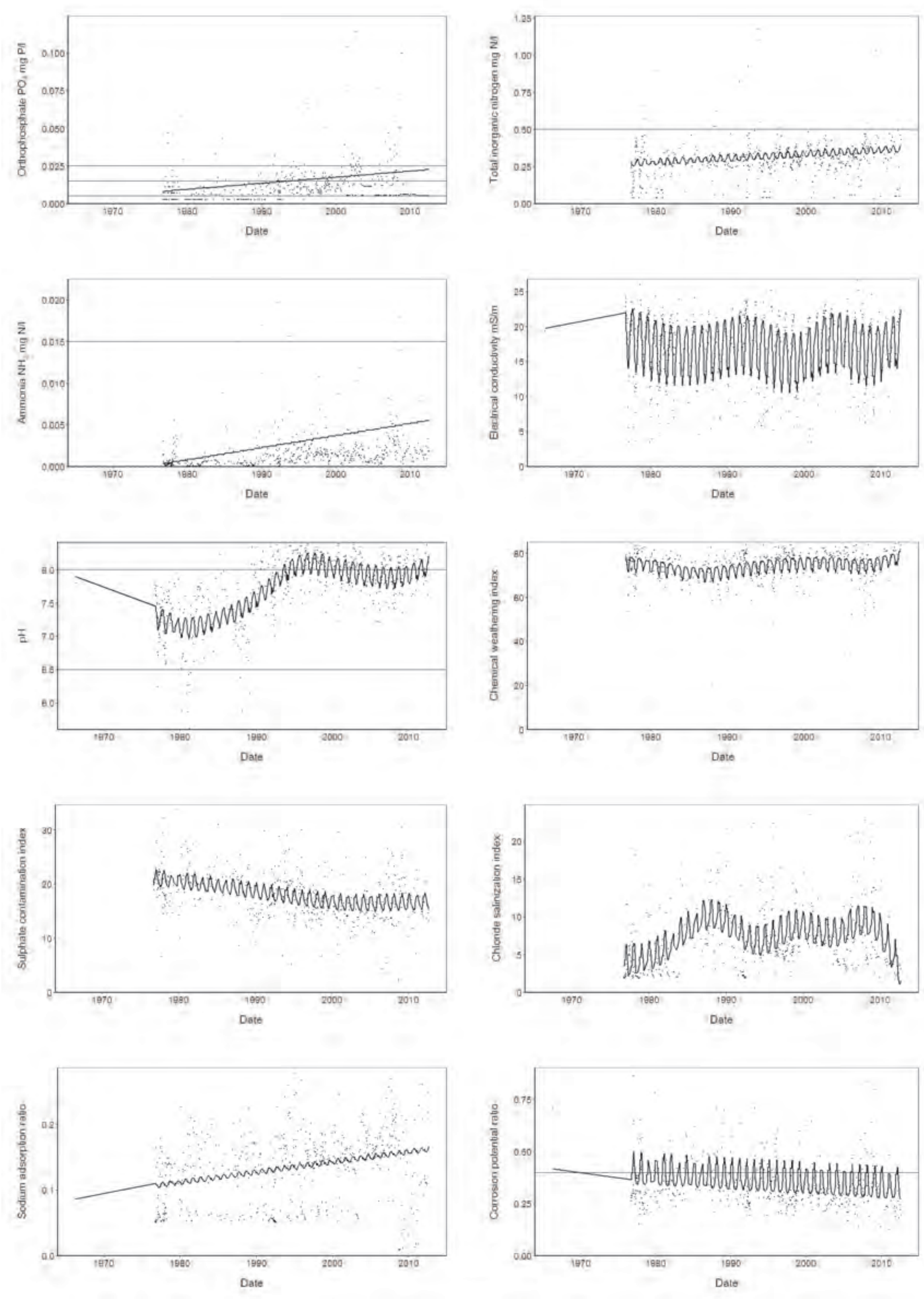

Figure 75 Temporal trends in several water quality parameters at monitoring point B6H001 at Willemsoord on the Blyde River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate p=0.001; TIN p=0.108; ammonia p=0.124; EC p<0.001; pH p<0.001; chemical weathering index p<0.001; sulphate contamination index p<0.001; chloride salinization index p<0.001; SAR p<0.001; and CPR p=0.130.

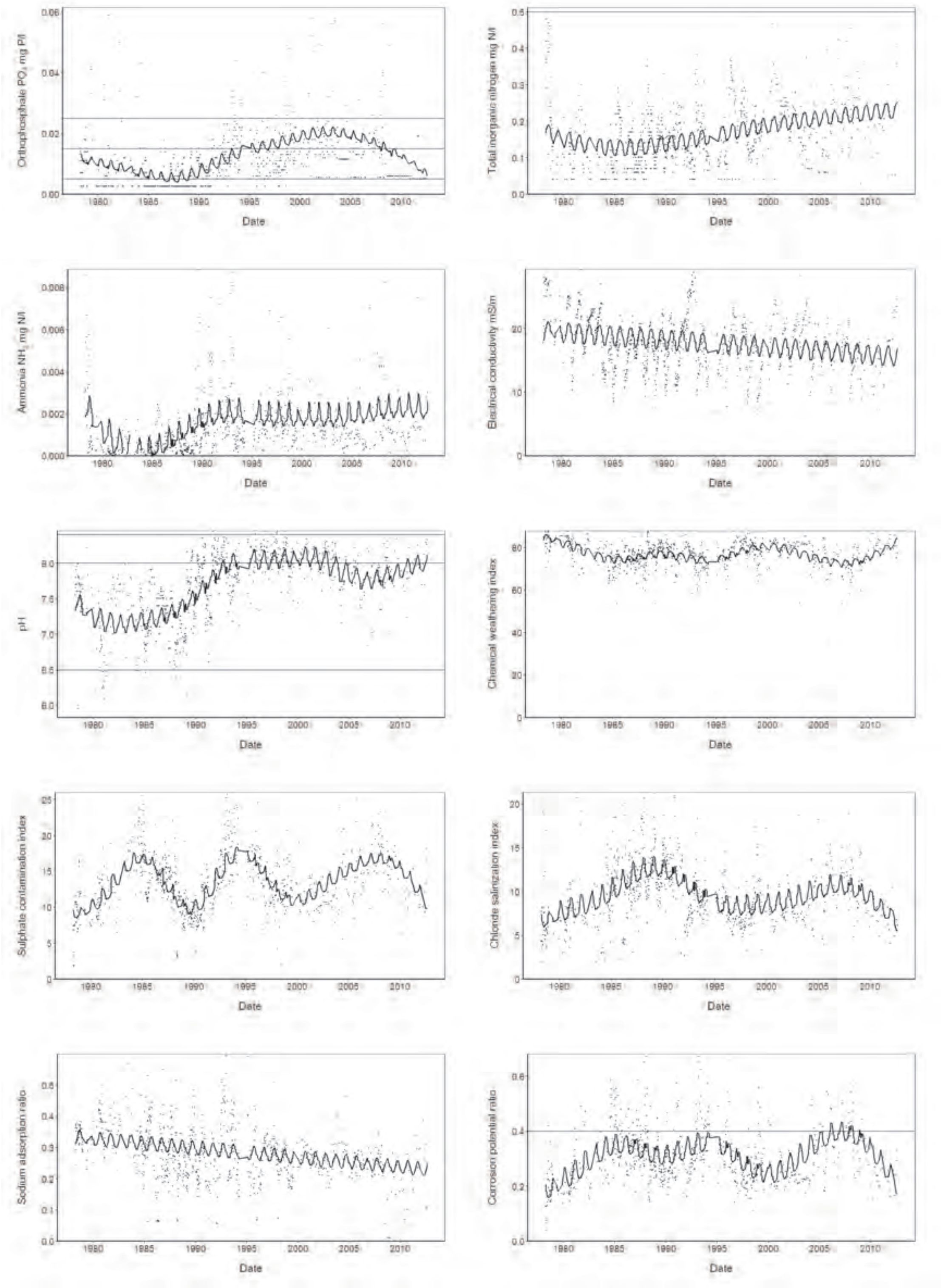

Figure 76 Temporal trends in several water quality parameters at monitoring point B6H004 at Chester on the Blyde River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate $p<0.001$; TIN $p<0.001$; ammonia $p<0.001$; EC $p<0.001$; pH $p<0.001$; chemical weathering index $p<0.001$; sulphate contamination index $p<0.001$; chloride salinization index $p<0.001$; SAR $p<0.001$; and CPR $p<0.001$.

Orthophosphate levels at Chester showed an increase for most of the data record, taking samples from ideal or acceptable to acceptable of tolerable in the early to mid-2000s. Thereafter, as is common at a number of sites, levels decline sharply for the last years of the data record such that most records are below the detection limit of the test employed, and most samples are at the boundary of ideal and acceptable.

Inorganic nitrogen levels show an increase with time, but levels remain in the ideal range. Un-ionized ammonia also increases, and then stabilizes, and remains at ideal levels throughout the data record.

Electrical conductivity levels show a slight decrease across the data record, and are ideal throughout. The seasonality apparent in this parameter means that higher levels are encountered in winter.

The pH shows a trend observed in samples from the upstream Blyde River, the Steelpoort River catchment and the Crocodile River catchment. Lower and stable pH values at the start of monitoring start to increase in the late 1980s until about 1995, when levels stabilize at a higher pH. There is also a slight decrease towards the end of the data record. pH levels from the start of the data record can comfortably be classed as ideal, while higher levels shift to an acceptable class.

Levels of the chemical weathering index fluctuate slightly with time and remain high for the entire data record. Both the sulphate contamination index and the chloride salinization index vary with time, but have fairly low scores with the former being slightly higher. Despite temporal change, neither index shows an overt increasing or decreasing trend. Absolute levels of chloride and sulphate are ideal throughout the data record (following DWA 2011a).

The sodium adsorption ratio and the corrosion potential ratio indicate that the water is suitable for throughout the data record. Levels of the sodium adsorption ratio are all ideal and show a slight decline with time, while levels of the corrosion potential ratio are mostly in the upper reaches of the ideal class, with some samples passing into the acceptable class.

Oxford (B7H007) on the Olifants River

This site lies about 10 km downstream of the confluence with the Blyde River in an area dominated by dryland agriculture and nature conservation/tourism. This site has one record from 1969, and regular monitoring commenced in 1975 and continues until the current day. The dataset for this site contains 973 records. Data on selected water quality parameters for this site are presented in Figure 77. The trends at this site are very similar to those encountered at Finale/Liverpool (B7H009), the next site upstream on the Olifants River.

Levels of orthophosphate show a general increase with time until the early to mid-2000s, with a slight and temporary decrease during the 1980s. Throughout this period, occasional high levels occur regularly. In the process median orthophosphate levels shift from an acceptable 0.01 mg P/ℓ in 1976 to a worse than tolerable 0.03 mg P/ℓ in 2008. Levels after this decline sharply to give an ideal to acceptable 0.005 mg P/ℓ in 2012 at the end of the data record.

Inorganic nitrogen levels are mostly ideal throughout the data record, with a significant number falling into the acceptable class. The fitted model indicates an increase in this parameter towards the end of the data record; however, inspection of the data suggests that this increase was caused by a small number of sample with particularly high levels, and further monitoring would be required to confirm this apparent trend.

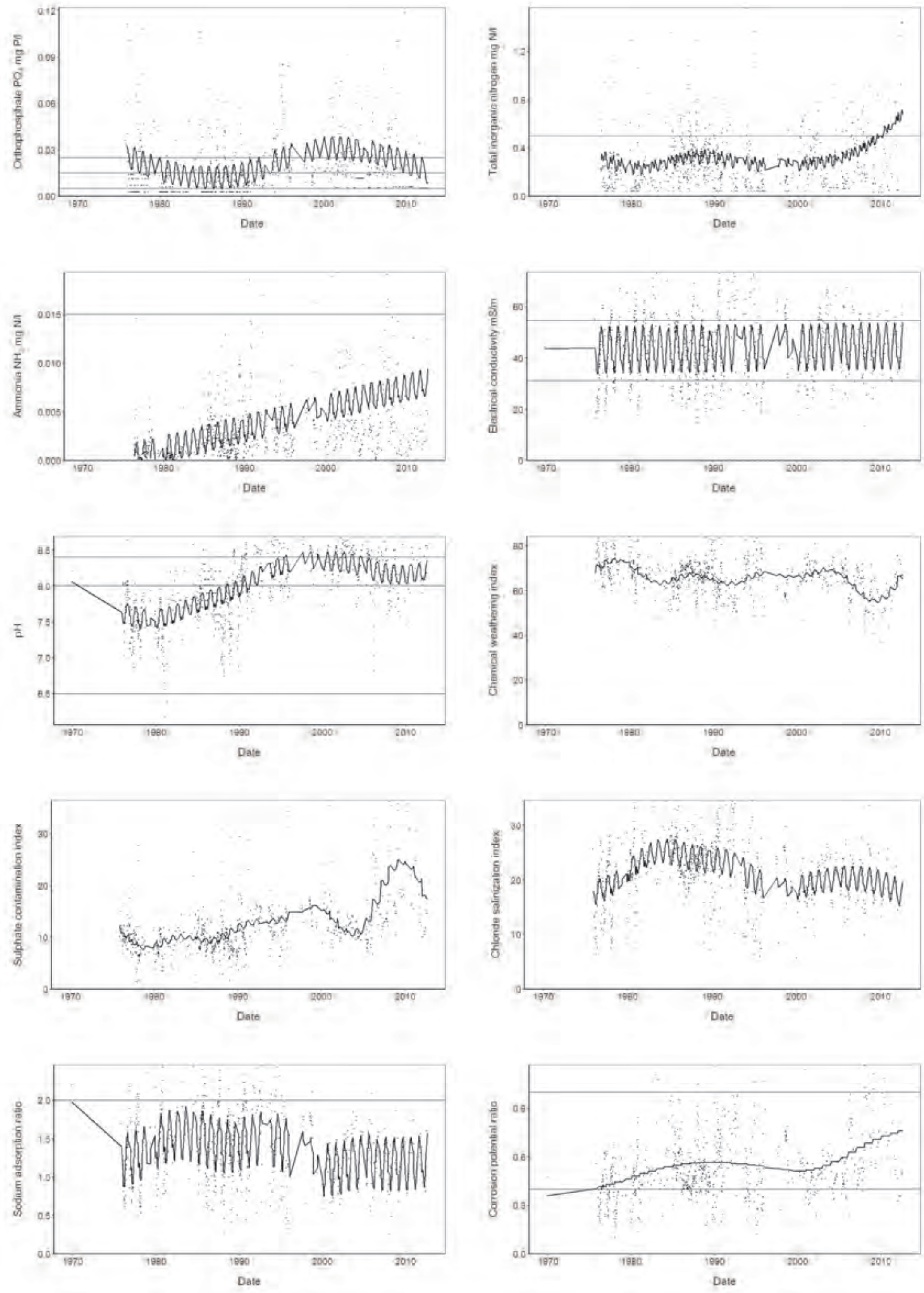

Figure 77 Temporal trends in several water quality parameters at monitoring point B7H007 at Oxford on the Olifants River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate $p<0.001$; TIN $p<0.001$; ammonia $p<0.001$; EC $p=0.413$; pH $p<0.001$; chemical weathering index $p<0.001$; sulphate contamination index $p<0.001$; chloride salinization index $p<0.001$; SAR $p<0.001$; and CPR $p<0.001$.

Levels of un-ionized ammonia at this site show a distinct increase with time. Nevertheless levels in the monitored period are for the greater part ideal and unless the increasing trend is maintained, levels of this compound should not be an issue at this site.

Electrical conductivity levels at this site show no evidence of change with time, although seasonal variation in this measure is pronounced ($p<0.001$). Levels are greater in the winter to autumn and improve again in the summer. As inspection of Figure 77 reveals, the seasonal variation means that the classification of water at this site can shift from ideal to tolerable on an annual cycle.

The pH at this site increases from a pre-1980 stable base until around 2000, when the levels stabilize at a higher level. This shift in pH is common in sites in this area. Recent years show a slight decrease, another trend observed at a number of sites. In the process levels shift from ideal at the start of monitoring to acceptable, and for a period, to a level worse than acceptable before shift back to being for the greater part acceptable.

In common with data from Finale/Liverpool upstream, the scores of the chemical weathering index at this Oxford are fairly good and a great improvement on sites further upstream in the catchment. Levels of this index fluctuate with time and show no overt directional trend. Sulphate contamination index scores increase from a low base to somewhat elevated scores in recent years, with fluctuations indicating the importance of short-term changes. Further monitoring will be necessary to determine whether the increased scores in recent samples represent a short-term change or whether sulphate levels at this site are increasing. Absolute levels of sulphate do increase with time, but levels remain classed as ideal after DWA (2011a). Relative levels of chloride as indicated by the chloride salinization index are higher, and, although changes with time are apparent, this index does not increase overall throughout the monitored period. Absolute chloride levels are ideal or acceptable at this site (after DWA 2011a).

The sodium adsorption ratio changes somewhat with time, while showing greater seasonal variation ($p<0.001$). Most water from this site would be classified as ideal according to this index. There is a slight increase in the corrosion potential index, probably driven by increasing sulphate levels in water at this site. Levels of this parameter shift from ideal to acceptable during the period covered by the data set.

Calais (B7H014) on the Ga-Selati River

This site is far upstream on the Ga-Selati River, slightly below the small settlement of Calais and close to the start of the river against the escarpment in the Lekgalameetse Nature Reserve. Land use in this area includes conservation and agriculture, with limited residential use. The data record for this site starts in 1977, but regular sampling stopped in 1982, and, bar a few sporadic samples from the interim, only commenced again around 2000. The data record for this site comprises 339 samples. This site was selected as something of a reference site for the Ga-Selati River, and has one of the best overall water quality levels of all sites assessed in the Olifants River Catchment (Figure 78).

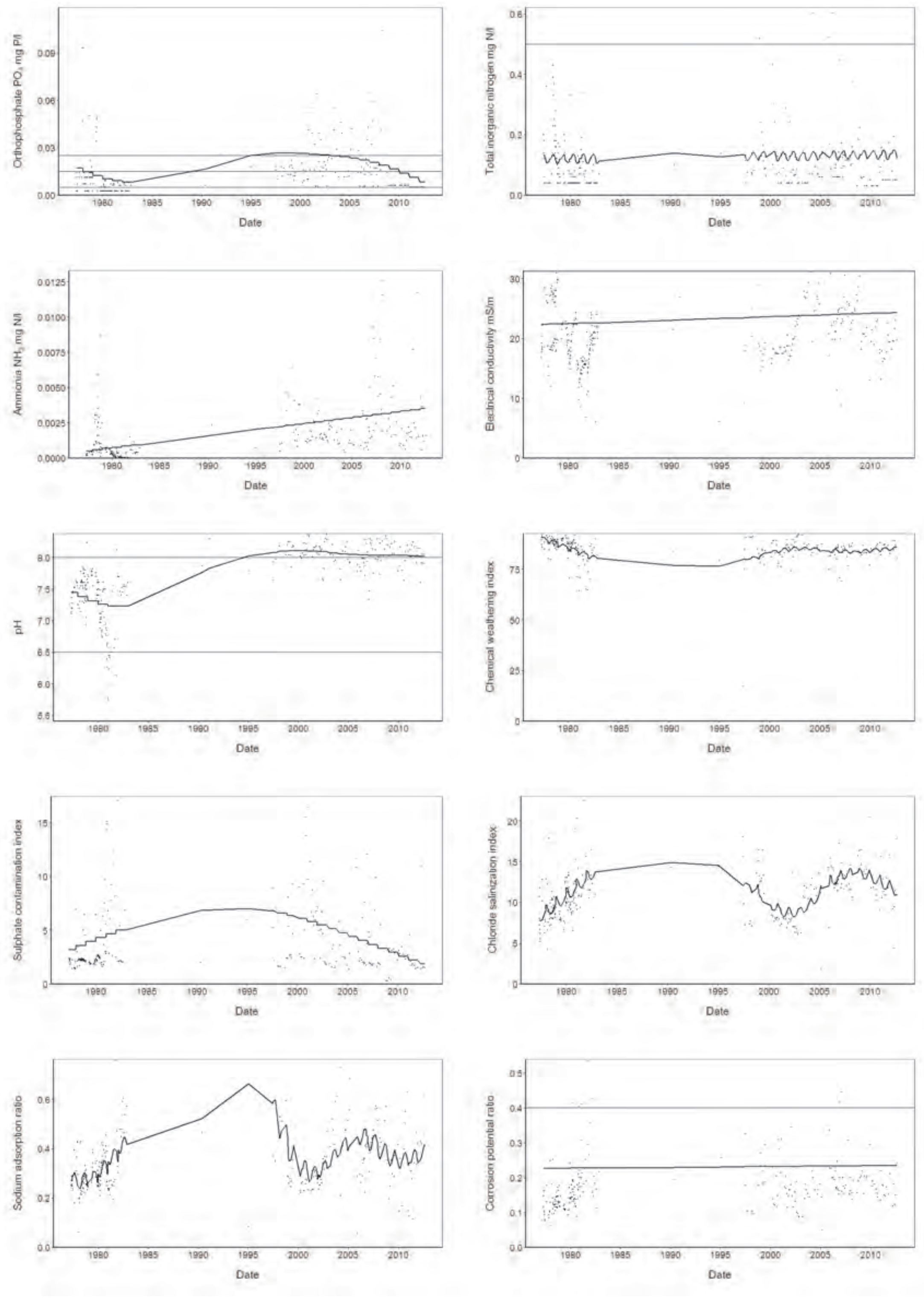

Figure 78 Temporal trends in several water quality parameters at monitoring point B7H014 at Calais on the Ga-Selati River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate p=0.030; TIN p=0.826; ammonia p<0.001; EC p=0.810; pH p<0.001; chemical weathering index p<0.001; sulphate contamination index p=0.037; chloride salinization index p<0.001; SAR p<0.001; and CPR p=0.957.

Orthophosphate levels at this site, like many others in the Olifants River catchment, increase over time from a base where levels are classed as ideal or acceptable to a point where median levels of orthophosphate are classed as worse than tolerable, and then improve thereafter so that recent samples are again classed as ideal to acceptable. As the majority of the river upstream is contained inside a nature reserve, it seems that orthophosphate levels in this river are likely to result from farming activity just above the sampling point and/or the impact of the Calais settlement.

Inorganic nitrogen levels are ideal throughout the data record and show no overt trend with time. Un-ionized ammonia levels on the other hand do show an increasing trend, but levels of this parameter remain well within the ideal class.

Electrical conductivity levels are low, and, while some temporal change is apparent in the plotted data, the model fitting algorithm was not able to fit a satisfactory trend to the data. Visual assessment of the data shows relatively short-term and minor temporal changes with no evidence of an overall increasing or decreasing trend and no clear seasonal pattern for this parameter.

pH levels show the same trend observed in a number of sites in the western or mid to lower Olifants catchment. pH levels increase from the start of monitoring to stabilize at higher levels in the early to mid-2000s, and decrease slightly thereafter. Missing data mean that the timing and rate of the increase is not known; however, that levels are higher from 2000 and decrease slightly thereafter is clear from available data. The cause of this change in an upstream site like Calais is not known.

Chemical weathering index scores are high throughout and indicative of a river with a largely natural salt complement. Scores of the sulphate contamination index were low and variable, while those of the chloride salinization index were slightly higher and showed some temporal variation but without either a detectable pattern of seasonality or an overt increasing of decreasing trend. Absolute levels of chloride and sulphate were low through the data record, and samples would be classed as ideal following DWA (2011a).

The results of the sodium adsorption ratio and the corrosion potential ratio show the water from this site to be suitable for use by a number of user groups. While some changes with time were found, neither index shows a clear trend to increase or decrease.

Loole/Foskor (B1H019) on the Ga-Selati River

This site is located low on the Ga-Selati River just prior to its confluence with the Olifants River. While much of the mid and upper Ga-Selati River passes through land dominated by agricultural use and nature conservation/tourism with a minor mining presence, the lower part of the river passes the town of Phalaborwa and the associated copper and phosphate mining complex located on the south-east of the town. The data record from this site starts in 1989 and runs to the current day, albeit with sparse sampling for several short periods. The dataset contains 488 records, and this data is presented in Figure 79. Inspection of this figure reveals that water quality at this site is severely degraded in comparison with Calais (B7H014) upstream. It is also apparent from the plots that, for most but not all of the parameters assessed, the water quality at this site has improved over the monitored period. Most parameters at this site showed significant seasonal variation.

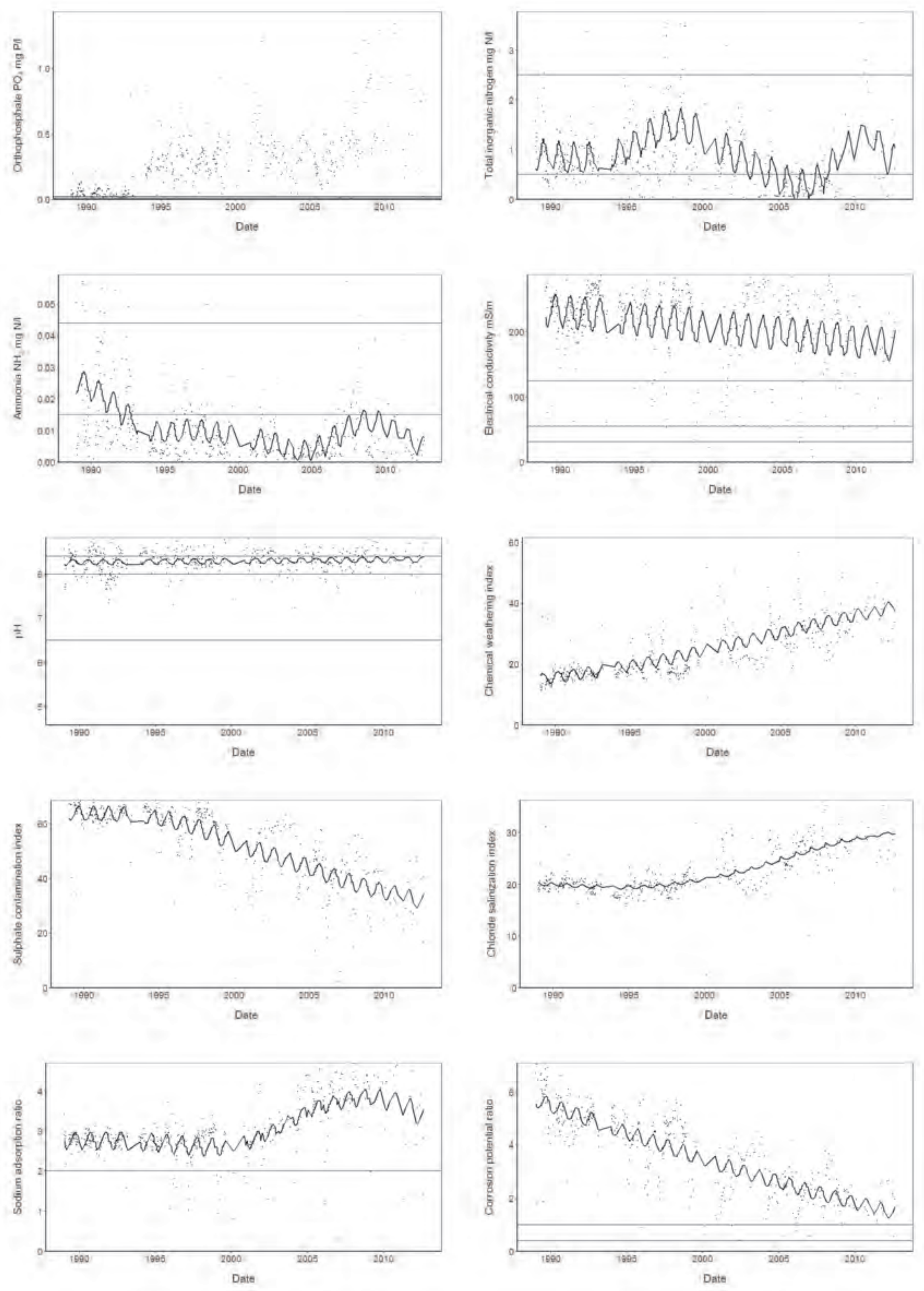

Figure 79 Temporal trends in several water quality parameters at monitoring point B7H019 at Loole/Foskor on the Ga-Selati River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate not convergent; TIN p<0.001; ammonia p<0.001; EC p<0.001; pH p=0.127; chemical weathering index p<0.001; sulphate contamination index p<0.001; chloride salinization index p<0.001; SAR p<0.001; and CPR p<0.001.

Orthophosphate levels at this point are the worst in the entire Olifants River catchment (see Figure 45), and levels of this parameter are increasing at the monitored point. High variability in the dataset made fitting a trend-line problematical, however, inspection of the data show this increase. Another point that can be made here is that, after a sparse collection of records in the early 1990s, measures of central tendency and variation in this parameter increased. The cause of the elevated and increasing orthophosphate levels at this site is not known; however, it seems likely that mining activities in an area where phosphate rock is actively mined underlie the unusual orthophosphate levels at this site.

Inorganic nitrogen levels at this site varied between acceptable for the main part, and ideal for a brief period just after 2005. Seasonal change in this parameter is high ($p<0.001$) and switching of classes on a seasonal basis is likely. Levels of inorganic nitrogen varied over time though a clear increasing or decreasing trend is not obvious.

Levels of un-ionized ammonia at this site have decreased with time and in recent samples levels are likely to be ideal. Levels of ammonia at this site are higher than most in the Olifants River catchment (Figure 47). There is considerable variability shown by this parameter and seasonal changes are also apparent ($p<0.001$).

Levels of electrical conductivity at Loole/Foskor are very high: at the start of the monitored period, conductivities at this point were roughly 10 times their upstream values; and, overall, this site has the highest conductivity levels in the entire Olifants River catchment (Figure 48). With the exception of a few samples, levels of this parameter are worse than tolerable throughout the data record. On the other hand, conductivity levels decreased significantly over the monitored period, and therefore, though levels are still intolerable, improvement in this regard is apparent in the dataset. Given the current rate of decrease, levels should reach tolerable in around 20 years from the present.

pH levels at this site are fairly high and seasonally variable ($p<0.001$) but stable over time. This is in contrast with a number of sites in this part of the catchment, including Calais upstream on the Ga-Selati River. Levels of this parameter at this site would be classed as acceptable or worse throughout the data record.

Chemical weathering index scores at this site were very low at the start of monitoring but showed a consistent improvement with time. This improvement is underlain by a decrease in the sulphate contamination index with time, and an accompanying decrease in absolute sulphate levels (though absolute sulphate levels remain for the greater part far worse than tolerable after DWA 2011a). In contrast, the chloride salinization index shows an increase; however, this is not accompanied by an increase in absolute chloride concentrations (although these are also largely beyond tolerable). The change in this index is driven rather by an increase in sample alkalinity, which is also beyond tolerable according to DWA (2011a). The latter change also partially drives the improving chemical weathering score.

The sodium adsorption ratio is acceptable throughout the data record, but shows an increase towards the end of the data record. This increase appears to be function of changes in several ions used in derivation of this index (magnesium and calcium decrease while sodium increases; absolute levels of all are largely beyond tolerable according to DWA 2011a). A similar reason lies behind the change in corrosion potential ratio scores, where decreasing sulphate levels interact with increasing alkalinity to bring the corrosion potential ratio scores almost to acceptable levels over the monitored period.

Mamba/Kruger National Park (B7h015) on the Olifants River

This site is about 4 km inside the Kruger National Part and just upstream of the confluence with the Klaserie River. At this point, the Olifants River has passed the Phalaborwa complex and merged with the Selati River. The water at this site should be appropriate to use in nature conservation, but the influences that modified the water quality are all exterior to the reserve. The data record extends from 1983 to the current day, with few records from around 2003. There are 718 samples available for analysis. An inspection of the data in Figure 80 reveals a site that shows a mixed signature of mainstream Olifants River water and some impact from Ga-Selati River inputs. As was observed at Loole/Foskor (B7H019) in the Ga-Selati River, the water quality at this site, according to a number of parameters, is improving with time. All parameters bar the major plant nutrients showed significant seasonal change at this point.

Trends in orthophosphate levels at this site reflect a mix of those at Oxford (B7H007) upstream on the Olifants River and at Loole/Foskor upstream on the Ga-Selati River. The trends at Mamba/Kruger National Park show the same changes in orthophosphate over time as at Oxford, but with slightly elevated orthophosphate levels, which is probably the result of Ga-Selati water entering the Olifants River. Although the concentrations of orthophosphate in Ga-Selati water inflow are very high, the volume of flow from the Ga-Selati River is low in comparison to the Olifants River flow and concentrations are diluted by Olifants River water to yield the increases seen in Figure 80. The end result is a pattern of orthophosphate concentrations that increase from a tolerable median concentration at the start of monitoring to a worse than tolerable median. Towards the end of the data set, median orthophosphate concentrations are ideal to acceptable, a common pattern in this catchment, and a surprising one at this point given the concurrent increasing orthophosphate concentrations at Loole/Foskor.

Inorganic nitrogen levels decrease over the data record and by 1995, most samples would be classed as ideal. Un-ionized ammonia levels are likewise mostly ideal, with a decrease over the data record.

Electrical conductivities of samples from this site start the data record with most samples classed as tolerable, and improve over time to stabilize from roughly 2000. There is a temporary increase in the early 1990s not matched at either upstream site. This parameter is highly seasonal ($p<0.001$) and seasonal fluctuations are such that classification of water from this site can change at the end of the data record from ideal to tolerable over one year.

The trends in pH levels show changes common in this catchment, with an increase from around 1990 to stabilize around 1995, and a slight further decrease after 2000. These changes closely reflect the situation upstream at Oxford on the Olifants River. pH levels at Loole/Foskor seem not have much changed the trends at Mamba/Kruger National Park, probably as levels are similar (from 1995 onwards anyway) and the greater volume of Olifants River water dilutes the effects owing to Ga-Selati River water.

The chemical weathering index scores at this site improved considerably over the monitored period, and short-term variation in this parameter (mostly due to seasonal variation $p<0.001$) decreased. The slight decrease around 2010 reflects changes upstream at or above Oxford on the Olifants River. Changes in this index are driven by decreasing relative levels of sulphate and chloride in the water (absolute levels of these ions drop over the monitored period to ideal and ideal to acceptable respectively; classification after DWA 2011a). At the same time there is an increase in alkalinity, leading to absolute levels that are tolerable at the end of the monitored period. All of these act to improve the chemical weathering index.

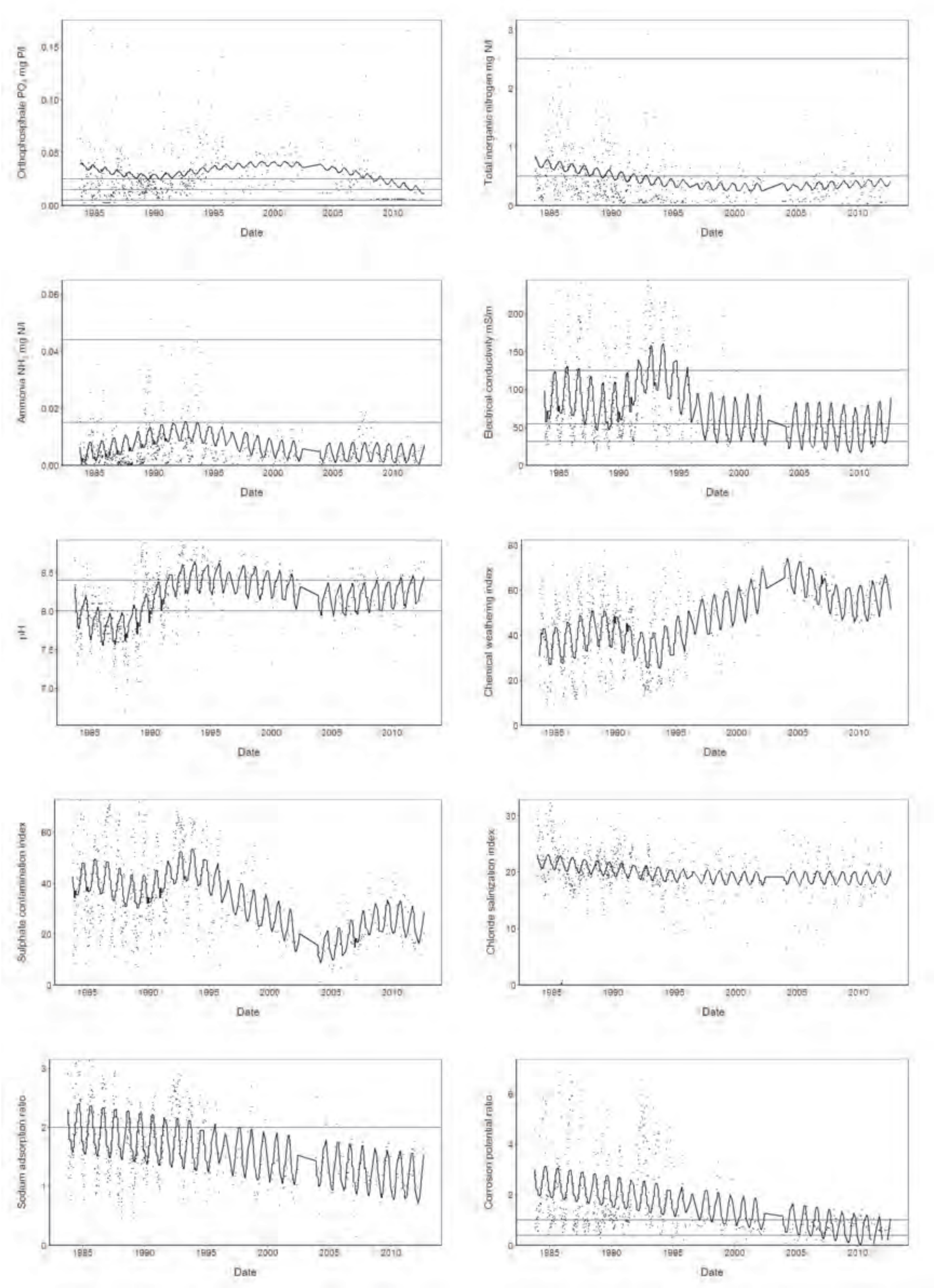

Figure 80 Temporal trends in several water quality parameters at monitoring point B7H015 at Mamba/Kruger National Park on the Olifants River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate p=0.031; TIN p<0.001; ammonia p=0.002; EC p<0.001; pH p<0.001; chemical weathering index p<0.001; sulphate contamination index p<0.001; chloride salinization index p<0.001; SAR p<0.001; and CPR p<0.001.

The sodium adsorption ratio at this site decreased steadily over the monitored period and sample from this site shift from being ideal to acceptable to largely ideal over the monitored period. Absolute sodium concentrations over the period shifted from ideal to worse than tolerable (and highly variable) at the start of the monitored period to ideal at the end. Finally, The corrosion potential ratio move from unacceptable to largely acceptable over the monitored period.

Fleur de Lys (B7H004) on the Klaserie River

This river was selected to include a river from the lower end of the Olifants River catchment. The site is fairly near the top of the Klaserie River about 3 km above the Klaserie (Jan Wassenaar) Dam. The river rises in the escarpment in the Blyde River Nature Reserve and flows past the settlements of Violet Bank, Moloro, Brooklyn and Ga-Maotele, east of Acornhoek in Mpumalanga, and immediately after the sampling point enters a region dominated by nature conservation and tourism before finally entering the Kruger National Park. The data record for this site begins in earnest in 1977, though there are a few prior samples stretching back to 1969, and runs to the current day. The data record from this site contains 550 records. Data from this site are presented in Figure 81. Data trends in this site show a site with generally good water quality that shares trends with several other sites in this region. All parameters bar major plant nutrients and un-ionized ammonia show statistically significant seasonality.

Levels of orthophosphate at this site show a trend common to sites in this region, where levels increase from the start of monitoring until the early 2000s, then decrease sharply thereafter. In common with several other sites too is the relatively small number of high samples from the late 1970s. Median levels from the start of the data record are ideal or acceptable, but in the early 2000s median phosphate levels are beyond tolerable. More recent samples are once again ideal to acceptable.

Inorganic nitrogen levels show an increase with time, but remain classed as ideal throughout the monitored period. The same trend is observed in un-ionized ammonia levels.

Electrical conductivity also increases with time and shows significant seasonality ($p<0.001$). However, levels are low and all samples are classed as ideal.

pH levels at this site show a trend common in this region, with an increase from a lower, stable level starting in the late 1980s and ending around 1995, when levels stabilize at a higher pH than before. There is a suggestion of a slight decrease thereafter. In contrast with a number of sites showing this trend, the later stable levels at this site are at a slightly lower pH than observed elsewhere, and samples from this site would, with few exceptions, be classified as ideal.

The chemical weathering index fluctuates with time and remains fairly high throughout. Changes in this index are a function of varying levels of chloride and sulphate in the water, with scores of the chloride salinization being slightly higher overall than the sulphate contamination index. Absolute levels of both these ions are ideal throughout by the classification of DWA (2011a).

The sulphate adsorption ratio shows seasonal variation ($p<0.001$) but no change with time and levels are ideal throughout. Corrosion potential ratio scores are largely acceptable throughout, with changes in this index being driven by changes in the levels of chloride in the water.

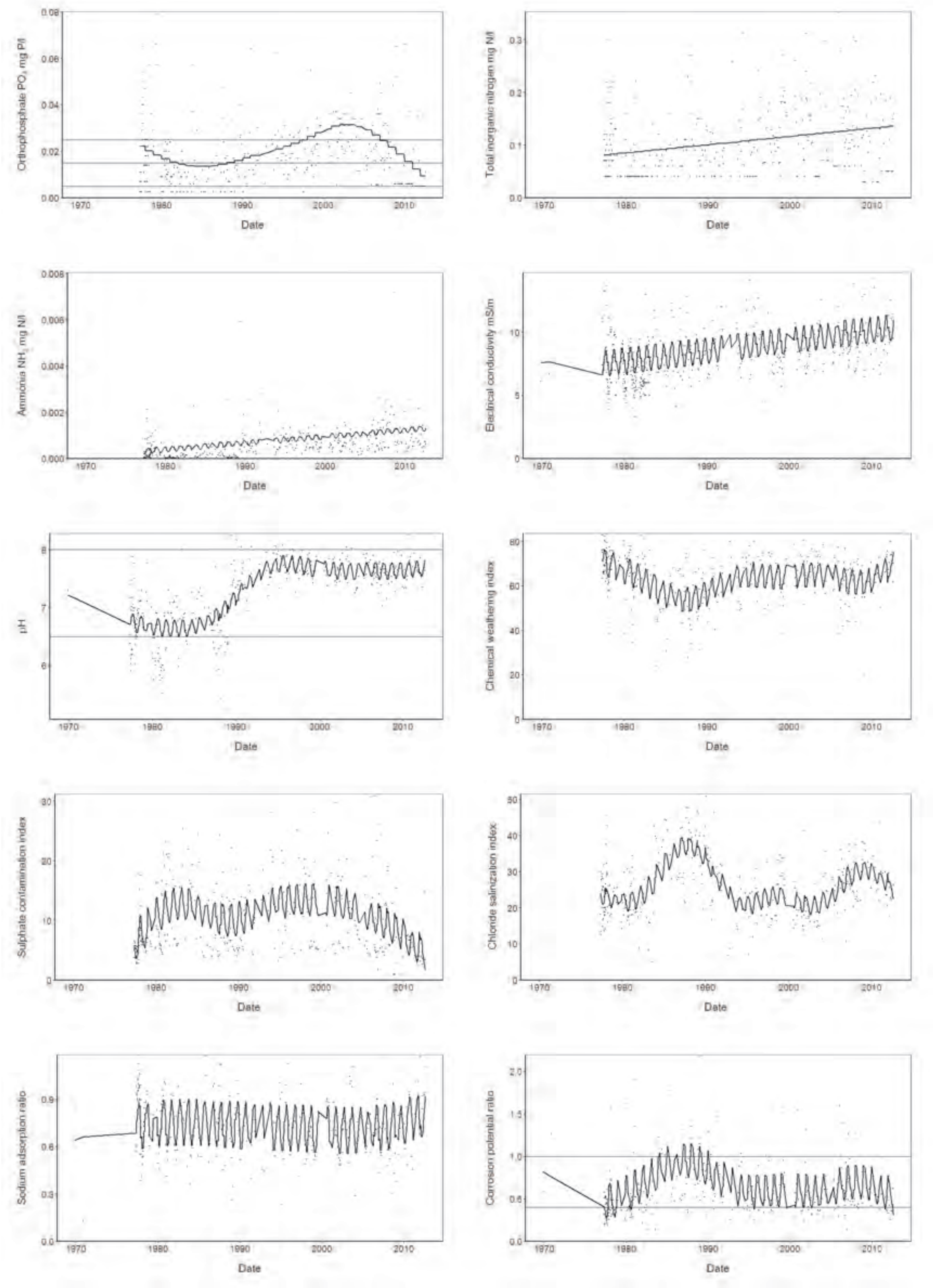

Figure 81 Temporal trends in several water quality parameters at monitoring point B7H004 at Fleur de Lys on the Klaserie River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate $p<0.001$; TIN $p<0.001$; ammonia $p=0.007$; EC $p<0.001$; pH $p<0.001$; chemical weathering index $p<0.001$; sulphate contamination index $p<0.001$; chloride salinization index $p<0.001$; SAR $p=0.158$; and CPR $p<0.001$.

Balule/Kruger National Park (B7H0017) on the Olifants River

This site is the furthest downstream in South Africa on the Olifants River, and is located in the Kruger National Park 17 km upstream of the confluence with the Letaba River and 25 km upstream of the South African-Mozambican border. After passing the border, the river enters the Massingir Dam. Land uses upstream of this point and downstream of upstream sites Mamba/Kruger National Park (B7H015) and Fleur de Lys (B7H004) are nearly exclusively nature conservation and tourism, as the majority of land is within the Kruger National Park. The data record at this site extends from 1983 to the current day and contains 443 records. Trends at this site are much the same as those at Mamba/Kruger National Park, approximately 70 km upstream, and apparently recovery of river condition between the sites is low. A trend noted at this site, and also apparent upstream, is the profound decrease in scattered high or very high values of various ions, the effects of which can be seen in several ion-based indices in Figure 82. All water quality parameters presented in Figure 82 show statistically significant seasonality. Sparse and infrequent sampling in the more recent part of the data record complicated the process of fitting trend models to the data.

Orthophosphate levels at Balule/Kruger National Park are variable and show an increase with time. The fitted model did not detect a drop in orthophosphate at the end of the data record as was observed at upstream sites. Inspection of the data suggests that this has happened but that sampling was sparse and infrequent towards the end of the data record and this will have limited the capacity of the analysis to detect such a decrease. A slight recovery in this index is apparent between the two sites as annual median orthophosphate levels at this site are nearly always lower than upstream at Mamba/Kruger National Park.

Inorganic nitrogen levels at this site are consistently ideal and show no consistent trend with time. Comparison of annual median levels of inorganic nitrogen at this site with levels from upstream at Mamba/Kruger National Park reveals that levels of available nitrogen have decreased with distance downstream.

Levels of un-ionized ammonia at this site also show no obvious trend with time and are generally ideal.

Electrical conductivity at this site shows a slight decrease with time consistent with that observed upstream at Mamba/Kruger National Park. As at the upstream site, seasonal change in this parameter can change site classification from ideal to tolerable on a regular basis.

Changes in pH are very similar to those upstream at Mamba/Kruger National Park and follow a trend observed at many other sites in the Olifants and Crocodile River catchments.

Although model fits vary owing to sampling effort differences between the sites, trends in the chemical weathering, sulphate contamination and chloride salinization indices at this site are very like those upstream at Mamba/Kruger National Park. Assessment of ions underlying the indices confirms this observation.

Finally, as at Mamba/Kruger National Park, the sodium adsorption ratio and the corrosion potential ratio scores increase over time.

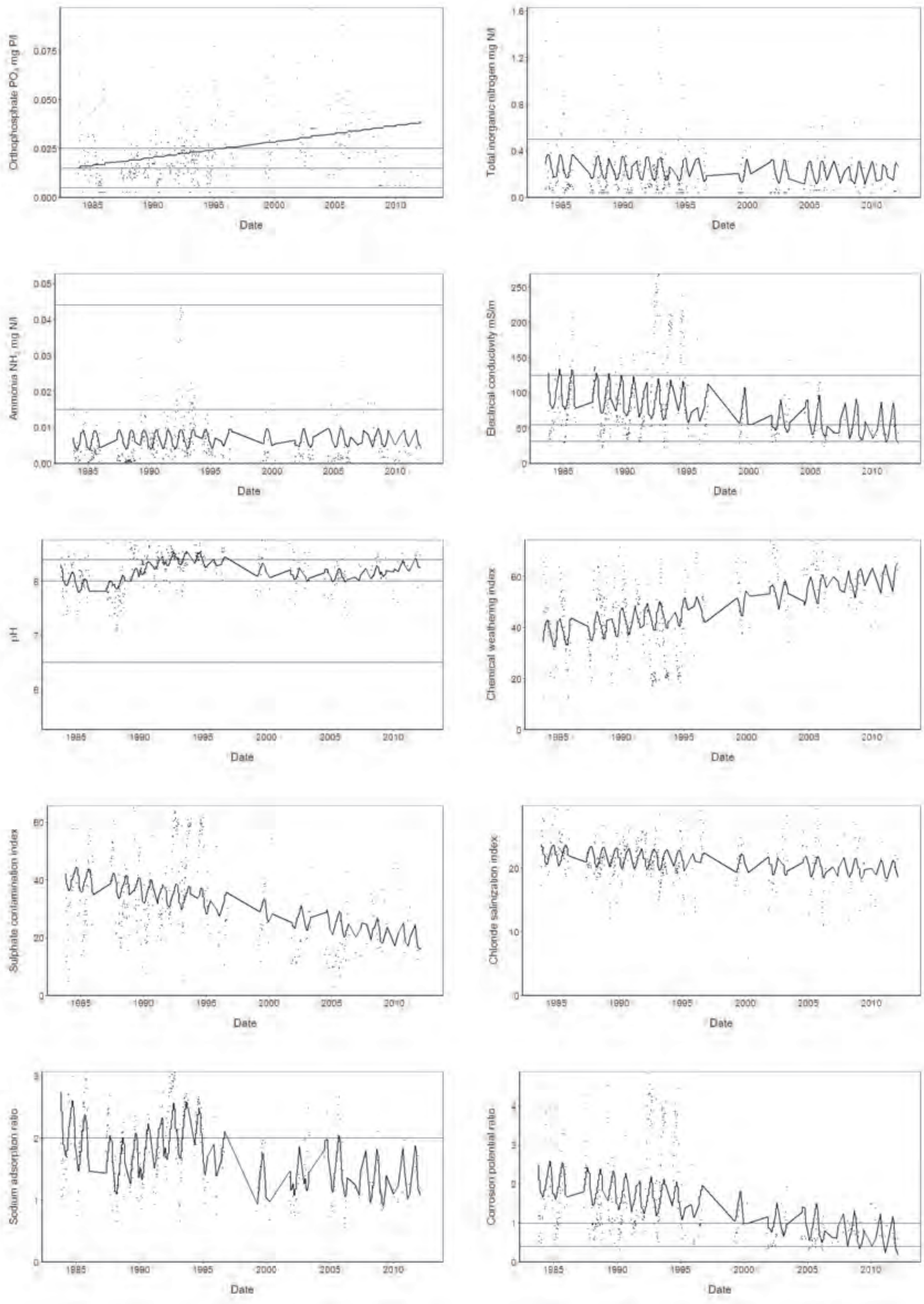

Figure 82 Temporal trends in several water quality parameters at monitoring point B7H0017 at Balule/Kruger National Park on the Olifants River. Points show water quality parameters against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends were: orthophosphate p=0.001; TIN p=0.286; ammonia p=0.697; EC p=0.003; pH p<0.001; chemical weathering index p<0.001; sulphate contamination index p<0.001; chloride salinization index p=0.037; SAR p<0.001; and CPR p<0.001.

Temporal trends in microbial levels

Changes in microbial pollution with time and between monitoring points in the Olifants River catchment are presented in Figure 83. Many of the caveats or problems associated with these are in common with the results from the Crocodile River catchment. Issues include the fact that the data record is relatively short, dating from late 2005 in sites with enough data to attempt trend analysis. The number of records since the start of sampling is also low, even when sites in the catchment with relatively good data sets for this parameter. Finally, the geographic distribution of sample sites with better data sets is extremely uneven, and, for this analysis, only sites from the upper Olifants River catchment and the Steelpoort River catchment had data sufficient for analysis. This precludes any spatial analysis of microbial pollution in this catchment.

The size of the available datasets severely limited the capacity of the statistical model fitting process to derive accurate models for the data (assuming that there are any temporal or seasonal trends in this parameter at the various sites. Of the temporal trends that were tested, only the increasing trend at site L74 Burgersfort Sewage Works was close to the widely accepted limit for statistical significance, and it is likely that with more data, an overall increase in *E. coli* levels from this site would have proved statistically significant. However, of interest is that, of the four sites with enough data to attempt a model fit, two sites, at the L44 Bridge on the R555 Bridge over the Witpoort River and the R555 Bridge at Steelpoortdrift over the Steelpoort River, had showed significant seasonal variation in E. coli levels ($p<0.001$ in both cases). In addition, data from the site at the R555 bridge at Blinkwater on the Steelpoort River showed a close to significant seasonal pattern ($p=0.065$). This indicates the probability of seasonal variation in levels of this parameter. Little can be said as to whether this trend is specific to the Steelpoort River catchment, as the tests were only conducted on data from this catchment, as data records here were best of all in the entire Olifants River catchment.

Attention should be paid to the levels of *E. coli* found, regardless of whether trends, either seasonal or temporal, were found. The upper bound of the tolerable class selected for this analysis was the upper bound of the significant risk class for watering of young livestock. The upper bound of the acceptable class was the upper bound of the significant risk class for faecal coliforms in domestic water. An inspection of the data plotted in Figure 83 reveals that water from all sites considered would pose significant risk for domestic use, and in many cases would have more than significant risk for livestock watering. Water from sites at S8 Delmas and Middelburg Town have few samples within the tolerable or better class, and water from this sites poses a high risk for use. The level of microbial potential at the site at Kriel is slightly better but would still pose significant to high risk for use. There is no site where even the majority of samples fell into the ideal class, and it appears that use of water in the regions assessed here carries at best a significant risk for domestic risk, and frequently is worse.

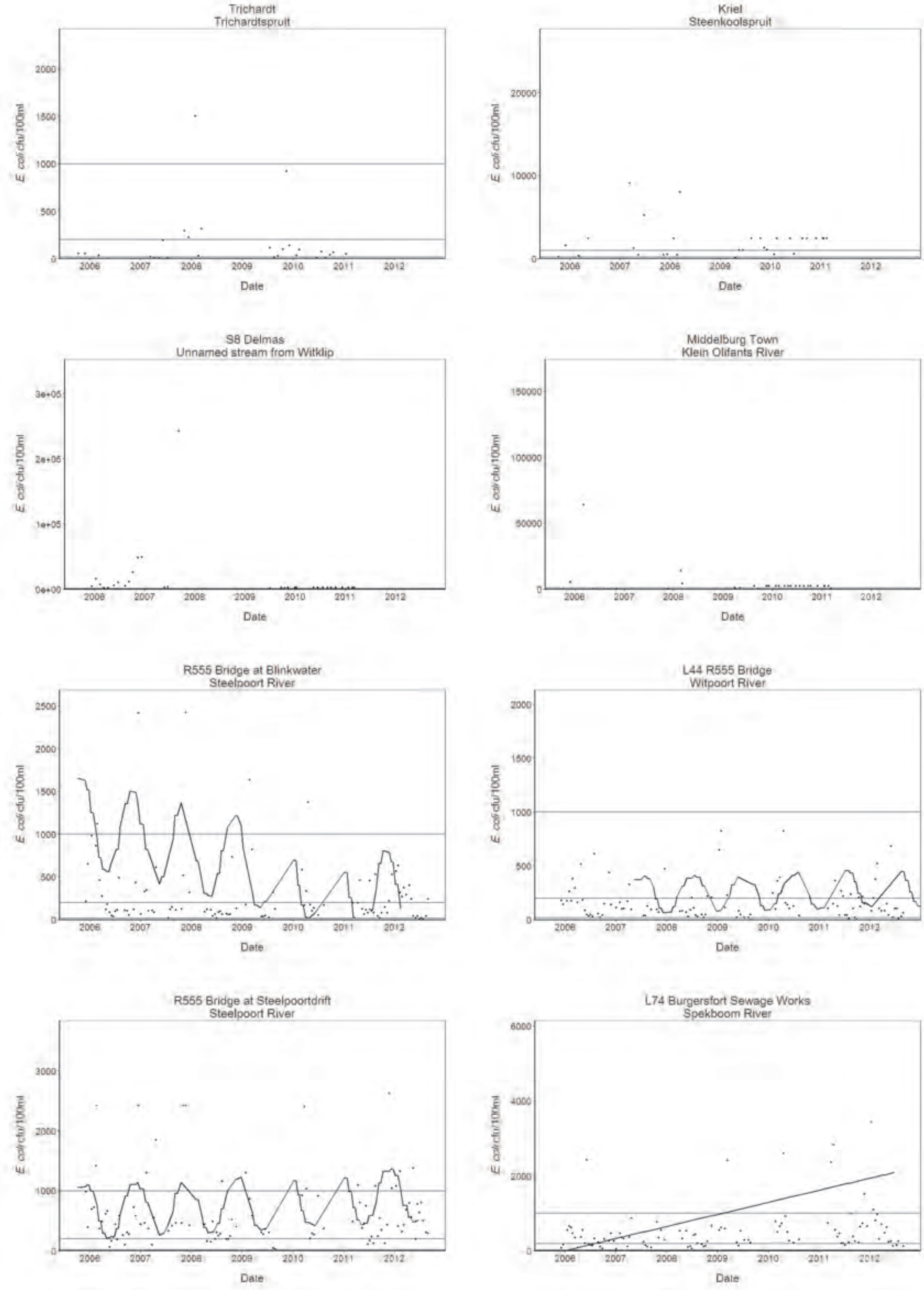

Figure 83 Temporal trends in *E. coli* colony forming units at several monitoring points in the Olifants River catchment. Points show colony forming units against time. Trend lines indicate GAMM model fits to data. Horizontal lines indicate RQO's adopted for this analysis. GAMM model fits for non-seasonal temporal trends at sites with sufficient data were: R555 Bridge at Blinkwater p=0.189; L44 R555 Bridge p=0.673; R555 Bridge at Steelpoortdrift p=0.238; L74 and Burgersfort Sewage Works p=0.063.

Potential toxins

Summary data on a range of toxins collected from monitoring points in the Olifants River catchment are presented in Table 14 below. Only sites with data (bar fluoride data which are widely available) are shown; similarly, only parameters with some data are presented. As is apparent from the table, there are a great number of toxins that are not monitored and the importance of these in the Olifants River system is not known.

Prior to commenting on the data, a few points need to be made. The first is that little data is available, to the extent that only 15 of the 28 monitoring points selected for analysis had any data from the monitored period, and the datasets at these points were generally very small. A second point is that standards for comparison were taken from the South African water quality guidelines (DWAF, 1996), ideally the aquatic ecosystem guidelines. When standards were not available from this source, other guidelines were used. As the guidelines for, for example, irrigation, are drawn up with a very different end use in mind, the guideline source must be considered when assessing the results presented here.

In assessing the results presented in Table 14, the fairly conservative approach was taken that the 95^{th} percentile of the potential toxin in question should equal or exceed the acute effect value from the guideline that was applied. This isolates data where an impact is highly probable and does not deal, for the purpose of this analysis, with chronic effects of potential toxins.

As was observed in the Crocodile River catchment analysis, data in Table 14 indicate that lead and mercury are potentially hazardous in most or all of the sites with data. Inspection of raw underlying data reveal the same underlying cause: detection limits for tests used to assess these parameters preclude identification of hazards following the guidelines applied here. In the case of mercury, none of the samples were tested with appropriate resolution, and in the case of lead, 18% of samples were tested with appropriate resolution. One can conclude from this that mercury levels did not exceed the detection limit of 0.02 mg/ℓ (though only 8 samples underlie this tentative conclusion). In the case of lead, 12% of samples had lead levels of 0.013 mg/ℓ, measured above the detection limit in all cases, and far above the acute effect guidelines adopted here. All these samples were collected in 2009 and 2010 at Mamba or Balule in the Kruger National Park. For the remainder of the samples for which data on lead were available, levels did not exceed the detection limit, which varied from 0.004-0.126 mg/ℓ.

Several sites where a number of potential toxins were found have no likely hazard according to the results. These sites were Wolvekrans, Middelkraal and Oxford on the Olifants River, and Naauwpoort on the Noupoortspruit. The results for these sites in Table 14 were, for many of these compounds, based on one to a few samples, and further sampling would be useful to confirm this.

Beyond this, several sites had data on a few parameters only. These sites include Zaaihoek on the Klipspruit, Onverwacht on the Wilge River, and Loskop Dam on the Olifants River. Of these sites, two (Zaaihoek and Onverwacht) had disturbingly high levels of aluminium. Given this observation, monitoring of more compounds at these sites is strongly recommended. This is especially so given the low pH levels encountered at Zaaihoek. At the remaining site, beyond fluoride, sampling of which is common when common salts are assessed, only aluminium was sampled for, and then rarely (4 samples). As such, little can be said about levels of potentially toxic compounds at this site.

Table 14 Percentiles (5^{th}-50^{th} (median)-95^{th}) of data from a range of dissolved toxins collected at a range of monitoring points in the Olifants River catchment. Data on all known and recognised toxins in the dataset are presented here. Only monitoring points and toxins for which data are available are presented (widely collected data on fluoride levels excepted). Target water quality range, chronic effect value and acute effect value for the most sensitive criterion from the South African water quality guidelines (DWAF, 1996) are also presented.

	B1H002 Elandspruit Spookspruit	**B1H004 Zaaihoek Klipspruit**	**B1H005 Wolvekrans Olifants River**	**B1H015 Middelburg Dam Klein Olifants R.**	**B1H018 Middelkraal Olifants River**	**Upper bounds (DWAF, 1996) TWQR-CEV-AEV**
Aluminium (mg/ℓ)	0.01-0.05-0.05	9.25-10.70-10.87	0.05-0.05-0.05	0.01-0.05-0.45	0.01-0.04-0.12	0.01-0.02-0.15 [1]
Arsenic (mg/ℓ)	0.05-0.08-0.10			0.003-0.03-0.03		0.01-0.02-0.13 [1]
Barium (mg/ℓ)	0.002-0.02-0.05		0.06-0.08-0.09	0.07-0.08-0.10	0.11-0.11-0.11	
Beryllium (mg/ℓ)	0.001-0.001-0.002					0.1-0.5 [2]
Boron (mg/ℓ)	0.002-0.03-0.05		0.002-0.008-0.03	0.03-0.05-0.20	0.002-0.002-0.002	0.5-4.0-15.0 [2]
Cadmium (mg/ℓ)	0.01-0.01-0.01		0.001-0.001-0.001	0.001-0.001-0.001	0.001-0.001-0.001	0.00025-0.0005-0.006 [1]
Chrome (mg/ℓ)	0.003-0.003-0.03		0.002-0.002-0.002	0.002-0.002-0.002	0.002-0.002-0.002	0.012-0.024-0.34 (as CrIII) [1]
Chrome VI (mg/ℓ)	0.02-0.02-0.02					0.007-0.014-0.2 [1]
Cobalt (mg/ℓ)	0.01-0.01-0.05					0.05-5.0 [2]
Copper (mg/ℓ)	0.003-0.004-0.01		0.001-0.001-0.001	0.001-0.001-0.001	0.001-0.001-0.001	0.0008-0.0015-0.0046 [1]
Fluoride (mg/ℓ)	0.20-0.45-0.82	0.05-0.42-1.05	0.33-0.47-0.71	0.27-0.37-0.54	0.27-0.40-0.80	0.75-1.5-2.54 [1]
Iron (mg/ℓ)	0.05-0.05-0.11	0.30-0.68-0.95	0.01-0.05-0.32	0.002-0.06-0.39	0.08-0.14-0.40	0.1-30-3000 [3]
Lead (mg/ℓ)	0.03-0.04-0.05		0.01-0.01-0.01	0.01-0.01-0.01	0.01-0.01-0.01	0.0005-0.001-0.007 [1]
Manganese (mg/ℓ)	0.05-0.05-1.51	2.32-3.71-3.99	0.05-0.05-0.30	0.001-0.001-0.003	0.003-0.02-0.03	0.02-10 [2]
Mercury (mg/ℓ)				0.01-0.01-0.01		0.00004-0.00008-0.0017 [1]
Molybdenum (mg/ℓ)	0.003-0.01-0.02		0.003-0.003-0.003	0.003-0.003-0.02	0.003-0.003-0.003	0.01-0.02 [4]
Nickel (mg/ℓ)	0.01-0.05-0.15		0.003-0.01-0.02	0.003-0.003-0.01	0.003-0.003-0.003	0.2-2.0 [2]
Selenium ((mg/ℓ)				0.001-0.001-0.001		0.002-0.005-0.03 [1]
Vanadium (mg/ℓ)	0.001-0.01-0.02		0.001-0.001-0.002	0.001-0.001-0.001	0.001-0.001-0.001	0.1-1.0 [2,3]
Zinc (mg/ℓ)	0.002-0.05-0.26		0.002-0.002-0.01	0.002-0.002-0.01	0.002-0.002-0.002	0.002-0.0036-0.036 [1]

South African water quality guidelines: [1] aquatic ecosystems (DWAF, 1996e); [2] irrigation (DWAF, 1996c); [3] domestic use (DWAF, 1996a); [4] livestock watering (DWAF, 1996d)

	B1H019 Naauwpoort Noupoortspruit	**B1H020 Vaalkranz Koringspruit**	**B1H021 Middeldrift Steenkoolspruit**	**B2H014 Onverwacht Wilge River**	**B3H001 Loskop North Olifants River**	**Upper bounds (DWAF, 1996) TWQR-CEV-AEV**
Aluminium (mg/ℓ)	0.01-0.01-0.04	0.04-0.05-0.05	0.01-0.01-0.46	0.50-0.50-1.53	0.50-0.50-0.50	0.01-0.02-0.15 [1]
Arsenic (mg/ℓ)		0.10-0.10-0.10			0.05-0.05-0.05	0.01-0.02-0.13 [1]
Barium (mg/ℓ)	0.04-0.07-0.10	0.001-0.03-0.08	0.07-0.09-0.12			
Beryllium (mg/ℓ)					0.001-0.001-0.001	0.1-0.5 [2]
Boron (mg/ℓ)	0.002-0.01-0.02	0.002-0.007-0.03	0.002-0.002-0.02		0.02-0.04-0.29	0.5-4.0-15.0 [2]
Cadmium (mg/ℓ)	0.001-0.001-0.001	0.001-0.003-0.005	0.001-0.001-0.001		0.003-0.003-0.003	0.00025-0.0005-0.006 [1]
Chrome (mg/ℓ)	0.002-0.002-0.002	0.002-0.002-0.003	0.002-0.002-0.002		0.003-0.003-0.003	0.012-0.024-0.34 (as CrIII) [1]
Chrome VI (mg/ℓ)						0.007-0.014-0.2 [1]
Cobalt (mg/ℓ)					0.01-0.01-0.01	0.05-5.0 [2]
Copper (mg/ℓ)	0.001-0.001-0.001	0.001-0.01-0.02	0.001-0.001-0.001		0.003-0.003-0.003	0.0008-0.0015-0.0046 [1]
Fluoride (mg/ℓ)	0.27-0.41-0.64	0.31-0.60-0.95	0.26-0.39-0.58	0.20-0.30-0.50	0.32-0.88-1.35	0.75-1.5-2.54 [1]
Iron (mg/ℓ)	0.003-0.08-1.02	0.003-0.05-0.10	0.002-0.01-0.66	0.10-0.33-1.13	0.10-0.10-0.10	0.1-30-3000 [3]
Lead (mg/ℓ)	0.01-0.01-0.01	0.01-0.03-0.05	0.01-0.01-0.01		0.03-0.03-0.03	0.0005-0.001-0.007 [1]
Manganese (mg/ℓ)	0.01-0.06-0.16	0.05-0.31-3.21	0.001-0.01-0.02	0.001-0.001-0.04	0.001-0.001-0.001	0.02-10 [2]
Mercury (mg/ℓ)					0.001-0.001-0.001	0.00004-0.00008-0.0017 [1]
Molybdenum (mg/ℓ)	0.003-0.003-0.003	0.003-0.01-0.02	0.003-0.003-0.003		0.003-0.003-0.003	0.01-0.02 [4]
Nickel (mg/ℓ)	0.003-0.01-0.01	0.003-0.004-0.02	0.003-0.003-0.003		0.01-0.01-0.01	0.2-2.0 [2]
Selenium ((mg/ℓ)						0.002-0.005-0.03 [1]
Vanadium (mg/ℓ)	0.001-0.001-0.002	0.001-0.002-0.02	0.001-0.001-0.001		0.001-0.001-0.001	0.1-1.0 [2,3]
Zinc (mg/ℓ)	0.002-0.002-0.002	0.002-0.002-0.003	0.002-0.002-0.002		0.002-0.002-0.002	0.002-0.0036-0.036 [1]

South African water quality guidelines: [1] aquatic ecosystems (DWAF, 1996e); [2] irrigation (DWAF, 1996c); [3] domestic use (DWAF, 1996a); [4] livestock watering (DWAF, 1996d)

	B3H017 Loskop Dam Olifants River	**B7H007 Oxford Olifants River**	**B7H009 Finale Liverpool Olifants River**	**B7H015 Mamba/KNP Olifants River**	**B7H017 Balule/KNP Olifants River**	**Upper bounds (DWAF, 1996) TWQR-CEV-AEV**
Aluminium (mg/ℓ)	0.01-0.01-0.01	0.04-0.04-0.04	0.01-0.02-0.02	0.003-0.04-0.09	0.01-0.03-0.04	0.01-0.02-0.15 [1]
Arsenic (mg/ℓ)		0.03-0.03-0.03	0.03-0.03-0.03	0.01-0.01-0.02	0.01-0.01-0.02	0.01-0.02-0.13 [1]
Barium (mg/ℓ)		0.08-0.08-0.08	0.001-0.003-0.03	0.01-0.04-0.06	0.01-0.04-0.05	
Beryllium (mg/ℓ)			0.001-0.001-0.001			0.1-0.5 [2]
Boron (mg/ℓ)		0.15-0.15-0.15	0.04-0.05-0.07	0.01-0.04-0.05	0.010.04-0.05	0.5-4.0-15.0 [2]
Cadmium (mg/ℓ)		0.001-0.001-0.001	0.001-0.001-0.001	0.001-0.004-0.01	0.001-0.004-0.01	0.00025-0.0005-0.006 [1]
Chrome (mg/ℓ)		0.002-0.002-0.002	0.002-0.002-0.002	0.001-0.004-0.01	0.001-0.003-0.01	0.012-0.024-0.34 (as CrIII) [1]
Chrome VI (mg/ℓ)						0.007-0.014-0.2 [1]
Cobalt (mg/ℓ)		0.03-0.03-0.03	0.01-0.01-0.01	0.01-0.01-0.01	0.01-0.01-0.01	0.05-5.0 [2]
Copper (mg/ℓ)		0.001-0.001-0.001	0.001-0.001-0.01	0.002-0.01-0.02	0.001-0.01-0.01	0.0008-0.0015-0.0046 [1]
Fluoride (mg/ℓ)	0.25-0.32-0.44	0.13-0.28-0.50	0.15-0.32-0.59	0.22-0.53-2.85	0.24-0.60-2.91	0.75-1.5-2.54 [1]
Iron (mg/ℓ)		0.002-0.002-0.002	0.002-0.002-0.004	0.001-0.01-0.31	0.001-0.02-0.07	0.1-30-3000 [3]
Lead (mg/ℓ)		0.01-0.01-0.01	0.01-0.01-0.01	0.002-0.01-0.06	0.002-0.01-0.05	0.0005-0.001-0.007 [1]
Manganese (mg/ℓ)		0.001-0.001-0.001	0.001-0.001-0.001	0.001-0.003-0.07	0.001-0.002-0.01	0.02-10 [2]
Mercury (mg/ℓ)			0.01-0.01-0.01			0.00004-0.00008-0.0017 [1]
Molybdenum (mg/ℓ)		0.003-0.003-0.003	0.003-0.003-0.003	0.001-0.02-0.03	0.001-0.02-0.03	0.01-0.02 [4]
Nickel (mg/ℓ)		0.003-0.003-0.003	0.003-0.003-0.003	0.001-0.01-0.02	0.001-0.004-0.01	0.2-2.0 [2]
Selenium ((mg/ℓ)						0.002-0.005-0.03 [1]
Vanadium (mg/ℓ)		0.01-0.01-0.01	0.003-0.00-0.02	0.002-0.01-0.02	0.001-0.01-0.02	0.1-1.0 [2,3]
Zinc (mg/ℓ)		0.002-0.002-0.002	0.002-0.002-0.01	0.008-0.01-0.14	0.001-0.003-0.25	0.002-0.0036-0.036 [1]

South African water quality guidelines: [1] aquatic ecosystems (DWAF, 1996e); [2] irrigation (DWAF, 1996c); [3] domestic use (DWAF, 1996a); [4] livestock watering (DWAF, 1996d)

In the site where a wider range of potential toxicants were assessed, the compounds, excluding mercury and lead for reasons given above, with elevated levels at the most sites were as follows: aluminium (33% of sites assessed), cadmium (25% of sites assessed), copper (42% of sites assessed), fluoride (13% of sites assessed), molybdenum (33% of sites assessed), and zinc (17% of sites assessed).

Of the sites in the upper catchment, the site with the widest range of potential toxicants was Elandspruit in the Spookspruit, where elevated levels of cadmium and molybdenum were detected. Other common upper catchment toxicants include aluminium and copper. As noted above, further sampling in, for example, the acidified Klipspruit, would likely yield more.

The lower part of the Olifants River, where it flows through the Kruger National Park towards Mozambique, contains a unique cocktail of potentially toxic compounds. Toxicants found in this stretch include cadmium, copper, fluoride, molybdenum and zinc, and, in addition, lead levels far over the detection limit were also observed. It may be speculated that these enter the river in the Ga-Selati River; however, in the absence of any data from the monitoring point at the end of the Ga-Selati River, this must remain speculation. Monitoring of this point for potential toxicants is highly recommended given the water quality impacts that are observed at this point together with the elevated metal levels in the lower Olifants River.

Another river from which no data were found is the Steelpoort River system. As mining is well established in this area, and as elevated levels of copper were found in the Olifants River at Finale/Liverpool after the confluence with the Steelpoort River, monitoring of these compounds is recommended.

3.4 Discussion

The aim of this analysis was to identify trends in water quality in the two study catchments, and these are discussed below. The initial discussion will focus on trends that occur across multiple monitoring points in one or both catchments. While there are a number of sites with localized trends, which often can be attributed to land use or relatively focussed or point impacts, these are presented in the results above. Localized trends that can simply be ascribed to a simple, local impact receive little attention here as these can largely be addressed by local intervention and not by addressing management patterns in a broader sense.

Speculation on the causes of identified trends in water quality in the catchments will be limited at this point as clear evidence of causality is often lacking, and also as many have been addressed on a site by site basis.

As noted in 3.2.12 above, the results that are discussed here are based only on reported values from DWA, and no cross-referencing with other data sets has been undertaken.

3.4.1 Orthophosphate trends

General orthophosphate increase

A common trend in both catchments is for a general increase in orthophosphate levels over most or all of the monitored period. In comparison with other water quality parameters assessed in this survey, the levels of orthophosphate are generally the worst when compared to the RQOs that were adopted. Given that eutrophication has been identified as one of the major threats to water quality in South Africa (CSIR 2010, DWA 2011a, van Ginkel 2011), this general trend of orthophosphate levels to exceed tolerable levels is of considerable concern (but see below for recent changes).

When considering this overall trend, it is instructive to note that the trend extends in many cases to monitoring points relatively high in their respective catchments, located in many cases below small drainage areas with relatively few identified upstream impacts. Though levels at upstream sites are often lower than at downstream ones, the same trend to increasing orthophosphate levels is common and widespread.

Sudden orthophosphate decrease

A more recent trend in orthophosphate levels, particularly in sites in the Olifants River catchment, but also at some sites in the Crocodile River catchment, is a notable decrease in reported orthophosphate levels in recent samples (usually from 2009 onwards). In the Olifants River catchment, recent reported levels of orthophosphate are generally below detection limits, and therefore all that is known of these parameters is that they would be classed as ideal or acceptable. In many cases, inspection of the plots shows the extent of this decrease, with levels falling suddenly from intolerable or better levels to below the detection limit. This equates to a five-fold or greater decrease over a very short period. Despite this decrease, some sites still report occasional high values.

Given the variability of orthophosphate data, this trend was relatively infrequently detected by the model fitting algorithm.

Early orthophosphate peak

Another curious trend in orthophosphate levels across both major catchments is a brief peak in orthophosphate levels at sites in the late 1970s, generally followed by a period of lower levels. Many of the monitoring points selected for this analysis do not have data from this period and no conclusions can be drawn about them; however, sites with data often reflect this trend.

As in 0.0.0 above, this trend was often not detected by the model fitting algorithm, most likely as a result of high variability in orthophosphate levels coupled with a relatively small number of samples with such high values.

3.4.2 pH shifts over time

Another widespread trend that can be observed at many sites in both catchments where local impacts permit is an increase in pH from a stable base level starting in the late 1980s or early 1990s. Levels increased for about 5 years, then stabilized again at a higher pH than the initial base level. In many cases, this shift to a higher pH is followed by a slight drop in more recent samples. As was noted for the trend in increasing orthophosphate levels, this trend to a shifted pH is widespread and is found in a range of rivers, and at sites in upper and lower catchments. The overall character of this trend is that local conditions and changes at each site might modify various characteristics of the change (extent of pH shifted, for example, or degree of recent recovery), however this trend is very common in both catchments. In most cases, the shift in pH over the monitored period, discounting seasonal changes that might occur, was between 0.5 and 1.0 pH units.

The cause of this pH change is not known; however, the spatial distribution of the trend suggests that it is a consequence not of spatially variable drivers such as geology, but rather of something more generalized. Potential general causes might include management, aerosol and rainfall quality changes, or other widespread changes.

3.4.3 Acid mine drainage, treatment and water quality

The upper Olifants River catchment has a great deal of coal mining and associated electricity generation, and impacts in this area have been commented on by other authors (Hohls *et al*. 2002, DWAF 2004d, Heath *et al*. 2010, Ashton and Dabrowski 2011, amongst many others). In the survey of water quality presented here, only one of the selected sites displayed the classic signature of acid mine drainage with abnormally low pH values, high relative and absolute levels of sulphate, and, of the few metals that were checked, very high levels of aluminium. This site was Zaaihoek (B1H004) on the Klipspruit, a stream whose condition has been remarked on before (e.g. Hill 1997, Hohls *et al*. 2002, McCarthy 2011, Dabrowski and De Klerk 2013).

However, Zaaihoek is not the only site that is associated with or downstream of coal mines (one of the streams assessed is even named after coal) and yet such severe acidification is not noted elsewhere. However, high, and often increasing levels of salinity, are common in sites in this region, as are high absolute and relative levels of sulphate, which are also commonly show an increasing trend with time. Calcium levels at non-acidified sites increased together with sulphate levels until they were beyond tolerable (after DWA 2011a), probably as a result of using liming to treat acid drainage. Together with sulphate, the elevated calcium levels contribute to increased salinity levels. While calcium itself is not particularly toxic and may even mitigate against the impacts of certain metals (DWAF 1996a, Wood *et al*. 2006, Grosnell and Brix 2009), the effects of calcium include the production of scale on pipes and interference in a number of processes (DWAF 1996a, b) and as such are undesirable for a number of users. Increased levels of calcium cause the sodium adsorption ratio, used here as an indicator of the suitability of water for crop irrigation, to decrease. This is because increasing the ratio of calcium to sodium in irrigation water decreases clay soil dispersion and improves soil structure. The improvement in this ratio must, from an irrigator's perspective, be balanced against the elevated salt load and the ionic balance in the water and their potential impact on soils and crops (see Annandale *et al*. 2002 for an example). Other ions that were not directly considered in this trend analysis might also need assessment in this case. Candidates would include iron (from pyrite oxidation) and other co-solubilized metals (McCarthy 2011, Dabrowski and De Klerk 2013).

In the absence of pH changes, the changed salinity and ionic balance, in surface water, together with concomitant nutrient loading, will severely impact on ecosystem health (Ashton and Dabrowski 2011, Dabrowski *et al*. 2013).

There is a great deal more that can be said about acid mine drainage and its impact on the upper Olifants River catchment (e.g. see Hobbs *et al*. 2008, Ashton and Dabrowski 2011, Dabrowski and De Klerk 2013, Dabrowski *et al*. 2013). The aim of this report is to identify trends in water quality, however, and cannot deal with all issues around acid mine drainage. It is also important to note that although only one acidified stream was found in this survey, the impact of acid mine drainage on ground and surface water is extensive (e.g. see Expert Team of the Inter-Ministerial Committee under the Coordination of the Council for Geoscience 2010, McCarthy 2011).

3.4.4 Temporary sulphate increases in the early 1980s

Nearly all sites in the Crocodile catchment that were monitored through the 1980s reveal an increase in the relative abundance of sulphate in the early 1980s. Absolute levels of sulphate in this catchment remain for the most part ideal, but inspection of sulphate levels reveal that this increase was driven by absolute sulphate level increases across the catchment (rather than by changes in other ions used in index calculations). Sites in the Olifants River catchment have for the most part high sulphate levels determined by the impact of coal mining. A few do not, though, and several of these sites (with data for the period) show the same trend. These are all in tributaries of the Olifants River in the mid to lower catchment, and include sites in the Steelpoort River, Blyde River and Klaserie River.

Sulphate levels decrease again after this peak at all sites showing this trend. Thereafter, there is a general trend for sulphate levels to increase again to varying levels.

Industrially produced sulphur aerosols are major components of haze over South Africa before being transported to the Indian Ocean (Piketh *et al*. 1999), and sulphate aerosols over the Kruger National Park have been found to originate largely from fossil fuel burning and industrial activity over the Highveld (Maenhaut *et al*. 1996, Zunckel *et al*. 2000). Against the relatively low sulphate backgrounds at the sites that exhibit this trend, wet and dry deposition of aerosol sulphate is likely to contribute to some extent to the observed trends.

3.4.5 Changes in the importance of natural weathering on the ionic balance

The chemical weathering index used in this analysis focuses largely on anions, and compares those present owing to natural weathering with those that derive from sulphate contamination, salinization and eutrophication. The scores returned by this index range from very high in some upstream sites to very low, particularly at sites where mining impact is pronounced. Although the data record at most sites extends mostly from the 1970s or later to the current day, impacts at many sites were well established prior to the start of monitoring. This is particularly apparent at sites in the upper Olifants River catchment, where all sites bar one show evidence of significant sulphate contamination at the start of monitoring. The exception to this generality was the Spookspruit, which had good scores around 1980 when monitoring started, but these changed soon after as sulphate contamination increased dramatically. As some mines around the Spookspruit have already closed and contribute much to Olifants River pollution (Dabrowski and De Klerk 2013) it is something of a surprise to find this stream looking relatively unimpacted around 1980.

As noted above, many sites in the upper Olifants River catchment were impacted before monitoring started. In general, scores of this index at these sites may have fluctuated with time, but no site shows overt signs of recovery. However, upstream sites in the Wilge, the Steelpoort, the Blyde, and the Ga-Selati River catchments all have high chemical weathering index scores at the start of monitoring, and maintain these scores throughout. Sites in the upper to mid Olifants River catchment, and the mid to lower Wilge river showed a decreasing trend in index scores with time, largely as a result of sulphate contamination and salinity increases. Site in both the Steelpoort and Blyde River catchment have high index scores at upstream sites, with no temporal trend to change. In the Steelpoort River, sites lower in the catchment have lower index scores, but in not the Blyde River. Finally the impact of the mining industrial complex of Phalaborwa leads to exceedingly low scores at the start of monitoring which improve thereafter. The patterns seen at the end of the Ga-Selati River are reflected in scores in the lower Olifants River, which have mid to low index scores that improve with time.

In contrast, sites from the Crocodile River catchment generally had a high score on this index at the start of monitoring. As a general rule, sites highest along any river usually maintained a high score throughout, while sites lower on rivers showed the increasing importance of contamination in driving river ionic balances. Exceptions where a greater decrease in water quality was noted were around Nelspruit and along the Elands River. Overall, scores from the Crocodile River catchment are far better than those from the Olifants River catchment.

3.4.6 Microbial pollution

This analysis relied only on DWA data on microbial levels, and did not attempt to access other sources. As such it must be acknowledged that considerably more data is likely to be available in other records, for example local government records, individual wastewater treatment works' records, or Green Drop reports. As noted in the results section, the selection of points for assessment was driven by data availability and as a result the points selected do not have an even spatial distribution, making comparisons between regions untenable.

One clear conclusion that can be drawn from the data presented here is that levels of microbial pollution across both catchments are sometimes acceptable, but often only tolerable or worse. As such, the risk from the use of water from the assessed sites is significant at best for the domestic user, and often worse. At certain sites, *E. coli* levels exceeded the upper bound of tolerable by several orders of magnitude. The best sites overall were in the Kaap River catchment off the Crocodile River.

The observation that microbial levels, when checked, are found to be high and that the risk of using river water as drinking water is significant, is in accord with material published on South African rivers by other authors (e.g. Paulse *et al*. 2009, Greenfield *et al*. 2010, Lin *et al*. 2012).

Scarcity of data and variation meant that fitting significant trend lines to data was rarely possible and in general no conclusions on trends can be drawn from the data presented here.

3.4.7 Toxins

In the assessment of toxins presented, only data available from DWA were used. Too few data were available to allow a trend analysis and no conclusions can be drawn about changes in these parameters with time. In addition, many of the selected monitoring points had no data on the toxins assessed and nothing can be concluded from these analyses regarding these locations.

Taking the fairly conservative approach where sites were flagged only if the 95^{th} percentile of the toxin in question exceeded the respective guideline's acute effect value, several metals were found to pose a threat in the two catchments. Despite the differences in the two catchments, the set of metals identified as problematic was largely the same in both. Of the compounds for which data was available and for which test detection limits were suitable, aluminium, cadmium, copper, lead, molybdenum and zinc were found to have high values in both catchments. High levels of copper were found across most sites in both catchments.

Only five of the 16 assessed sampling points in the Crocodile River catchment had any data on the compounds assessed, and at one of these sites very few compounds had any data. The remaining four sites had high levels of most of the set of problematic metals at most or all sites. Three of these sites were on the Crocodile River, with two in the lower reaches, and the remaining site was at Dolton in the Kaap River below Barberton.

In comparison, fewer of the sites for which data were available in the Olifants River catchment had high levels of assessed metals. Of the 28 sites that were assessed, 15 had data on the one or more of the compounds for which data were available. Of these, three sites in particular had data on very few compounds. Four of the sites where a wider range of compounds were sampled were found not to have high levels of any of the metals identified as problematic. Of the sites remaining sites, the worst impacted sites were found low in the catchment, after the confluence with the Ga-Selati River, and within the Kruger National Park.

No data was available in the dataset that was analysed for any herbicide or pesticide, or for endocrine disruptors, or for any other more complex organic toxins. Pesticide impact and bioaccumulation has been recorded in both the Crocodile (van Dyk 1978, Heath 1999) and Olifants (Pick *et al*. 1981, van Dyk and Greeff 1977, Grobler 1994, Visser 2008) River systems (Heath and Claassen 1999, Ansara-Ross *et al*. 2012). As noted above, few data are available for a range of metals, and this scarcity does not indicate any planning for a regular, structured sampling plan. Metal impacts and bioaccumulation have also been recorded in both these rivers (e.g. Roux *et al*. 1994, Wepener 1997, Heath 1999, Oberholster *et al*. 2012, Dabrowski *et al*. 2013, Dabrowski and De Klerk 2013). Given that impacts have been identified, it would be highly advantageous that regular sampling for a range of toxic compounds be instituted as part of routine chemical monitoring. The issue of missing data is discussed further under 3.4.11 below.

3.4.8 Seasonality

The trend analysis method selected for this analysis was applied with the aim of detecting mid to long term changes in the various water quality parameters assessed, after accounting for the variation caused by seasonal changes. As such, assessment of seasonality was not a part of the original aims of this analysis. However, from the viewpoint of management of water quality, it is important to note that many of the parameters assessed had statistically significant seasonal changes, and that in several cases and at particular locations, the extent of seasonal changes have implications for the class that a water body would be assigned to following seasonally invariant Resource Quality Objectives.

Following the Resource Quality Objectives selected for this analysis, sites in a number of rivers across both catchments (and particularly towards the lower reaches of the respective rivers) would change over one class boundary within the course of a year as a result of seasonal change. At an extreme, seasonal change could cause a site to shift from ideal to worse than tolerable state.

Of the various parameters assessed, orthophosphate levels were the least likely to show a detectable seasonal pattern.

3.4.9 Particular impacts

A few notable impacts are reviewed below. This is not an exhaustive list of impacts across the two catchments, but rather major examples of the types of impacts encountered.

General impact: Upper Olifants River catchment

Much of the general change in the upper Olifants River catchment is discussed above. This section will draw attention to the extent to which water quality in this region is driven by multiple and complex local impacts. The major stressors across this region are coal mining, coal-fired electricity generation, industry, agriculture and often inefficient wastewater treatment (Dabrowski and De Klerk 2013). Many of the small catchments contain at least two or three of these. While the impacts of, for example, coal mining, can be found across the region, the differences between sampling points often appear to depend to a large extent on the management of impacts at that site.

As an example, it is instructive to compare Middeldrift on the Steelkoolspruit with Zaaihoek on the Klipsruit. The former has ideal or acceptable levels of many parameters, with, in several cases, trends to improvement in a number of these. The exceptions are nutrient levels, which in the case of orthophosphate, are worse than tolerable and increasing, and in the case of inorganic nitrogen, acceptable and increasing. Potentially hazardous metals include copper and lead, out of 14 for which data are available. The latter has exceptionally low pH levels, with accompanying worse than tolerable sulphate levels. Potential for use of this water is low. However, levels of orthophosphate are improving toward acceptable or ideal, and levels of inorganic nitrogen varied and were tolerable at the end of monitoring. Few toxins have any data from this site; of the few that do, aluminium levels are exceedingly high.

Even the sites in a better condition show the signs of stresses associated with coal mining, and the cumulative effect of this across the upper catchment is severe and is shows every indication of worsening.

Point impact: Phalaborwa area on the Ga-Selati River

Although the cumulative impact of coal mining and associated activities in the upper Olifants River catchment are severe and show no signs of improving, the impacts associated with the Phalaborwa area mining-industrial complex are arguably the worst point impact in the Olifants River catchment. The difference between water quality upstream in the Ga-Selati River and downstream before the confluence with the Olifants River was noted when presenting results from the sites in question. The impact of water from the Ga-Selati River on the water quality of the lower Olifants River, after the quality of Olifants River water has largely improved along the central reaches of the river, has been noted before (DWAF 2004d, Ashton and Dabrowski 2011).

While the scale of impacts at this point are indubitably high, over the period for which data was available, an improving trend in many of the water quality parameters assessed here was noted. Of the parameters assessed here, the most notable exception is the very high and increasing level of orthophosphate. Inspection of absolute ionic levels underlying the indices used in this analysis reveals that while alkalinity increases, sodium and chloride levels change little, and sulphate, magnesium and to a lesser extent calcium levels decrease with time. These ionic changes drive a decrease in conductivity over time. Despite the changes, the overall absolute level of the ions in question tends to remain worse than tolerable, though often by a small margin (in contrast with changes in orthophosphate).

Inspection of water quality parameters in the Olifants River immediately upstream and downstream of the Ga-Selati River confluence shows the impact of Ga-Selati River water on the downstream Olifants to be a function of the various parameters' concentrations in both upstream sites, as well as the discharge levels in the two rivers (as well as input from the Mohlabetsi and Sedumoni Rivers). So, for example, the effect of the very high orthophosphate levels in Ga-Selati River water is diluted by the greater volume of water from the Olifants River upstream. The end result is generally improving water quality at the first site downstream of the Ga-Selati River confluence, with many water quality parameters falling into the ideal or acceptable classes, and the timing and levels of some of the changes suggesting that improved water quality from the Ga-Selati River at least partially underlies the improvement in the Olifants River downstream. This optimistic view is offset though against the levels of various toxins in Olifants River water at this point, particularly as several records with levels above the acute effect value for that toxin were collected in the last few years covered by the data record.

Cumulative impact: Elands River, Crocodile River catchment

The Inkomati Catchment Management Agency (ICMA 2011) identifies the Elands sub-catchment as having worse water quality than other areas in the upper Crocodile River catchment. Identified impacts along the river include expanding human settlements, trout farming and a paper mill (ICMA 2011). Other impacts include agriculture and forestry and a ferrochrome smelter (Assmang Chrome

2013). This region is discussed here as both monitoring points along the river shift from good water quality at the start of monitoring and degrade thereafter, and thereby exhibit a clear directional trend.

At the upstream site (Geluk, below Machadodorp), most water quality parameters assessed were classed as ideal in the late 1970s when monitoring started, though orthophosphate levels were usually acceptable or better. Regular sampling at this site stopped during the 1980s and recommenced in the late 1990s. Until the mid-1980s, water quality was good overall, and samples showed little trend with time (with the exception of a slight increase in sulphate levels). After monitoring recommenced, increases in nutrients, un-ionized ammonia, electrical conductivity, sulphate, chloride, calcium, magnesium and sodium were recorded. Despite the rapid and dramatic change, these parameters would be classed as ideal or acceptable at the end of monitoring, with the exception of orthophosphate levels which were worse than tolerable.

At the downstream site, water quality at the start of monitoring is good. In the early to mid-1990s, levels of many water quality parameters assessed started to increase dramatically. These ions or measures include orthophosphate, sulphate, chloride, alkalinity, magnesium, calcium and sodium. For the most part, this involved a shift from an ideal to an acceptable state (orthophosphate levels increased to tolerable or worse, then improved dramatically as discussed in 0.0.0 above). As at the upper site, the increasing impacts cause a decline in chemical weathering index scores.

Several of the water quality parameters showed a slowing rate of increase in recent samples that suggests that the worsening of water quality may be coming to an end. However, if continued degradation does not slow, water quality could shift to tolerable or worse in the relatively near future. Even if degradation is slowed, this river has experienced a sharp change over a relatively short period.

Processes in the catchment that may account for some of the changes in water quality over the monitored period include expansion of the ferrochrome smelter which reached completion in 2005 (Assmang Chrome 2013), and expansion of a pulp and paper mill that reached completion in 1985, followed by commissioning of ozone rather than chlorine bleaching in 1995 (Sappi 2013). However, changes that might stem from these processes are not easily disentangled from the impacts of, for example, increased human settlement in the area.

Irrigation impact: Elands River, Olifants River catchment

This river flows through much agricultural land, with significant levels of irrigated agriculture. The area has a relatively low rainfall, and a warm climate (at least in summer). Water quality impacts at this site are unusual for the upper to mid Olifants catchment, in that the impacts of salinization, particularly as indicated by elevated sodium and chloride levels, are apparent, while those of coal mining are absent. Elevated relative chloride levels in South Africa are associated with salinization (Huizenga 2011), and salinization in turn may be a result of improper irrigation practices (Rabie and Day 2000).

The levels of sodium and chloride at this site contribute to tolerable or worse conductivity levels indicating high salinities in instream water, as well as an elevated, though still acceptable sodium adsorption ratio, indicating that sodium levels may have some impact on soil structure if river water is used for irrigation.

Inspection of monthly discharge data for this site reveals that, over the period assessed, the late 1990s and the period around 2010 were wetter, and the period around 2005 was somewhat drier. Discharge patterns correlate well with several other water quality parameters. Levels of inorganic nitrogen are high when discharge, and presumably rainfall, are higher, which may indicate that increased runoff or return flows are removing nitrogen from soil and carrying it to the river. Not surprisingly, salinity levels are low when discharge is high, and vice versa, likely as a result of simple dilution.

Changes in major ions follow salinity trends overall, with levels increasing when discharge is low (sodium, chloride, and alkalinity climbed to levels beyond tolerable). Changes in the sulphate contamination index, chloride salinization index and sodium adsorption ratio indicate that changes in the relative amounts of various ions follow discharge trends. There is an indication that sodium and chloride become more dominant when discharge is low.

The lower Crocodile River has been identified as a region under stress from irrigation (DWA 2011a), with levels of irrigation leading to low water availability (ICMA 2011). However, the trends in that region do not show the impact of salinization as clearly as in the Elands River, although salinity levels are comparable.

3.4.10 Trend analysis and methodology

Trend analysis

As outlined the rationale for the selected methodology, the analytic approach taken aimed to tease apart seasonal and longer term trends, and to be able deal with non-linear trends across time. Another factor in selection of a method was that it should be capable of using as much data as was available, regardless of temporal irregularity of sample and the potential for autocorrelation. As such, dropping of excess data to improve temporal spacing, or generation of data by interpolation was to be avoided. Despite the attractiveness of a non-parametric approach, the seasonal Kendall test was not appropriate as it tests for linear trends (Helsel and Hirsch 2002). Although some methods do exist for irregular time series analysis (e.g. Zeileis and Grothendieck 2005), the approach of Polansky and Robbins (2013) showed promise and was selected for this analysis.

The approach selected yielded good results, for the most part, although good fits for some parameters were not always produced. This is the result of limited model complexity, lack of model parameter tuning, and variation and outliers in the data.

A deliberate decision was taken to limit model complexity so as to avoid overfitting of the data, which would result in a model that was accurate for one compound at one site over the given time frame, but not generalizable. However, limiting complexity, particularly in the presence of high variation and more complex temporal change, resulted in fairly simple, near-linear model fits being generated for some datasets.

Given the number of models run in the full analysis, model parameter tuning was limited to selection of generic model parameters that performed well on a subset of the data sets assessed. This generic model was then applied to all datasets. For a good model fit, more time needs to be taken to set good parameters for each dataset separately. However, given the approach selected, the technique performed more than adequately for a broad survey of trends across a large set of data.

Prior to data analysis, it is good practice to undertake a process of data exploration and potential consequent dataset editing (e.g. Zuur *et al*. 2010). A part of this is detection of and dealing with outliers. This was not adequately undertaken prior to this analysis, and outliers could sometimes be seen to have undue effects on model fits. It is recommended that the checks proposed by Zuur *et al*. (2010) be undertaken, where applicable, prior to model fitting.

Finally, variation in datasets affected the efficacy of model fits. This is to be expected. Failure of the model fitting algorithms to come up with an appropriate model was particularly apparent in orthophosphate datasets, where variation was high and not clear seasonal trends were apparent.

Measures of water quality

The use of concentration data as practised in this analysis gives a valuable idea of the amount of a water quality parameter available to biota. However, in the absence of some understanding of flow or discharge, these data can be somewhat misleading. In the current analysis, an example, comparison of the concentrations of various parameters at Loole/Foskor in the lower Ga-Selati River and at Mamba/Kruger National Park downstream in the Olifants River and determination of the changes owing to Ga-Selati River input can only be done by assessing the relative flows of the respective rivers. Concomitant assessment of flow was not undertaken in this analysis, as the goals here relate to environmental water quality trends. However, a deeper understanding of the interaction between flow rates and concentration would deepen understanding of the trends observed here. Owing to the trend of reservoirs to retain nutrients entering them, assessments of loading are needed to assess the likely change owing to inflows (Harding 2008, Dabrowski *et al*. 2013).

This analysis selected three indices proposed by Huizenga *et al*. (2013) that have not seen widespread application. The indices were selected as they seemed a useful way of assessing the relative extent of various impacts, even when salinity and absolute concentration levels were fairly low. In this regard, they were useful, and in some cases highlighted impacts that assessment of absolute concentrations would not easily have highlighted. However, given that the effects of particular compounds are generally linked to available concentrations, it was often useful to apply the indices in conjunction with assessments of absolute concentrations.

3.4.11 Data record

Data availability

Issues related to data availability for potential toxins are discussed 3.4.7 above. Beyond major salts, WMS data for the two catchment contained fields for other parameters that would be useful in analysis. For example, levels of chlorophyll *a* might act as a more direct indicator of eutrophication and algal growth than orthophosphate, but little data were available. Likewise, data on total microcystin was available for two sites in the Crocodile River catchment, and this would be appropriate as a direct measure of algal toxins produced following eutrophication.

Consistent records

The value of a long-term data record with consistent data collection was highlighted by van Niekerk *et al*. (2009), in a survey of trends in salinity in South Africa, and sampling programmes have been devised to derive rational and useful data from all the monitoring programmes in place (DWAF 2004e, Nomquphu *et al*. 2007).

The approach taken in the analysis here was to attempt to identify long-term records from a range of sites. In many cases sites were rejected owing to scarcity of data, often owing to termination of sampling at sites, or to large gaps in the data record. Certainly, a common observation was that monitoring at a number of sites was terminated in the late 2000s, even when long-term data sets at these sites had already been compiled.

Another general observation was that frequency of monitoring varied both spatially and temporally. With regard to the latter case, a common trend observed across many sites was a decrease in sampling frequency in recent years. In many cases, the decrease in sampling frequency was accompanied by highly irregular sampling. The outcome of this is that, at certain sites, recent samples are not able to adequately sample seasonal variation. As seasonal variation at sites can be high, adequate and evenly spread sampling of these trends is necessary in order that seasonal variation can be included in trend analysis.

Another area where consistent records would be valuable is monitoring of what are traditionally less frequently monitored compounds. A useful example of this would be potentially toxic metals. The issue of detection limits with respect to some of these is discussed below. Here attention should be given to consistency, or lack thereof, in monitoring. For example, mercury levels at selected sites in the Olifants River catchment are, with one exception, only monitored from 1993 to 1994, and only at Middelburg Dam and Finale/Liverpool. Lead is monitored more widely, but timeframes of monitoring at different sites vary. Monitoring was maintained to the current day at two sites low on the Olifants River, but most other sites have records for lead spanning a few years during the 1990s. Aluminium is a recognised problem in the Olifant's River catchment (Ashton and Dabrowski 2011, Oberholster *et al*. 2012, Dabrowski and De Klerk 2013), and the extent of monitoring at certain sites reflects this. However, samples from the only site in the acidified Klipspruit with long term records, where our analysis found very high levels, derive only from 1991 and 1992. The rationale, costs and benefits of monitoring these toxins would benefit from a re-assessment, particularly given the known and proven impacts of these compounds.

Detection limits

Detection limits in the datasets used for this analysis were occasionally set somewhat or definitely too high. I will mention three examples in particular. Two relate to toxic metals, and one to a routinely monitored nutrient.

The detection limit for mercury in samples from selected monitoring points in this analysis was, with a single 1988 example, set at 0.02 mg/ℓ. The acute effect value of this compound for aquatic ecosystem health was 0.0017 mg/ℓ (DWAF 1996e). From an aquatic ecosystems viewpoint, mercury would be generally not be detected until levels had exceeded the acute effect value by a factor of nearly 12 times.

The detection limits for lead in samples from monitoring points selected for this analysis varied more far more than mercury. In samples from the both catchments, the detection limits of tests ranged from 0.004-0.126 mg/ℓ. In water of medium hardness, the acute effect value for this toxin was 0.007 mg/ℓ (DWAF 1996e). In order that lead levels approaching the acute effect value be detected, a detection limit of 0.004 mg/ℓ is entirely appropriate (although this is not sensitive enough to detect that the chronic effect value had been reached). Most of the remaining detection limits for tests will not detect lead levels until they have passed the acute effect value, and, in the worst case, until they are 18 times higher than the acute effect value.

Detection limits for tests for orthophosphate in more recent samples from both catchments are generally around 0.01 mg P/ℓ (older samples generally had more sensitive detection limits). This means that orthophosphate levels below acceptable cannot be identified by these tests. As noted above, orthophosphate levels are often high and this might not be an issue. One can also argue that knowing that levels are acceptable or better is sufficient for management of this nutrient. However, given the widespread decrease in orthophosphate levels in recent years referred to in 0.0.0 above, recent data from many sites are simply below the detection limit and nothing is known of orthophosphate levels or trends, beyond that levels are acceptable or better. It is somewhat contradictory that a class boundary for a compound is established, but in practice that boundary cannot be measured.

Monitoring point placement

No monitoring points with long term data were found in the central region of the river. It is recommended that some level of monitoring in this region be maintained such that long term data may exist in the future.

3.5 Conclusions

The analysis presented here was undertaken with the explicit aim of detecting trends in water quality in the two focus catchments. The data available allow one to explore a great number of facets of water quality and the drivers of water quality in the catchments. Nevertheless, throughout we have maintained a focus on the primary aims of the research, while necessarily not shifting focus to other issues thrown up by the data, regardless of how intriguing they may seem.

Overall, temporal trends often show decreasing water quality at sites in the mid to lower catchments, while sites in the high catchment may change relatively little. Sites that were impacted at the start of the data records generally continue to show that impact, with some showing an improvement over time. Overall, impacts are driven by increased orthophosphate (although recent samples suggest this trend has ended at many sites), increased pH levels, increased salinity, and, for sites in the Olifants River, increased or elevated sulphate and calcium levels. Some sites showed elevated chloride levels consistent with salinization, but this was less widespread than sulphate impacts. In addition, microbial levels were high, though no trend was apparent.

This analysis used all data available from DWA WMS, and inspection of this data reveals that while data from general chemical monitoring is fairly extensive, data from potential toxins is inadequate, particularly given the stresses on various parts of the study catchment. Reconsideration of sampling programme structure is recommended in this regard.

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