The Capture Principle Approach to Sustainable Groundwater Use

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EXECUTIVE SUMMARY

Motivation

The well-documented and scientifically accepted theoretical principles of groundwater flow theory dictate that water withdrawn artificially from an aquifer is derived from a decrease in storage in the aquifer, a reduction in the previous discharge from the aquifer, an increase in the recharge, or a combination of these changes (Theis, 1940, cited in Bennett et al, 1988). This decrease in discharge or increase in recharge has been termed 'capture' of water, and it is the ability of aquifer pumping to capture discharge and enhance recharge that dictates the aquifer's yield. It follows that an assessment of the sustainability of groundwater abstraction would quantify these changes in the flow regime and storage, and determine whether the changes and their associated impacts are considered acceptable; termed here the capture principle approach to sustainable groundwater use.

However, many current tools to support groundwater management broadly apply water balance type calculations for aquifer yield assessments in which groundwater availability is directly related to some portion of pre-abstraction recharge. These assessments are often completed at quaternary catchment scale, and aquifers (or catchments) with high use compared to recharge are generally identified as 'stressed' or 'over-used'. The approach can limit groundwater development based on a perceived stress. Impeding implementation of the capture approach to sustainability is the fact that the approach is intertwined with adaptive management. Management must proceed on less than ideal information, and decisions adjusted as groundwater use continues. This is awkward to regulate.

The ultimate purpose of the project was to promote the capture principle approach to sustainable groundwater use. The project proposed the development of a tool – a 'decision framework' – that could facilitate the translation of theoretical hydrogeological principles for sustainable groundwater use based on the capture principle approach into practice.

Capture Principle Approach to Sustainable Groundwater Use

The maintainable aquifer yield is defined as the pumping rate that can be maintained indefinitely without mining an aquifer continually. Discharge from wells upsets the dynamic equilibrium of an aquifer by producing a loss from aquifer storage; under pumping at the maintainable aquifer yield, over time a new state of dynamic equilibrium is approached at which there is no further loss from storage. The maintainable aquifer yield is therefore a rate that can be maintained by reduced discharge and/or induced recharge in a new dynamic equilibrium of the aquifer, such that the aquifer storage doesn't deplete further. It does not directly depend on the pre-abstraction recharge rates. The length of time required for the new state of dynamic equilibrium to be reached where there is no further loss from storage has been referred to as the aquifer response time.

If sustainable groundwater use is defined as groundwater use that is socially, environmentally and economically acceptable, then long-term abstraction of the maintainable aquifer yield does not necessarily reflect sustainable groundwater use. A critical step from quantification of a maintainable aquifer yield to quantification of sustainable groundwater use, is to determine the contribution from each source under the new dynamic equilibrium (reduced discharge, enhanced recharge), and the equivalent piezometric head distribution (extent of storage depletion), and then take a socio-economic-environmental decision as to whether this is acceptable. If it is acceptable, then the long-term abstraction of the maintainable aquifer yield can be considered sustainable groundwater use, and the maintainable aquifer yield reflects a sustainable aquifer yield.

Groundwater is accessed via pumping from individual boreholes and wellfields, so to implement sustainable groundwater use, wellfield operation must be informed by the maintainable aquifer yield. Aquifer yields and groundwater sustainability therefore cannot be considered in isolation of borehole and wellfield yields and operation.

The sustainable use of groundwater must be implemented through robust operating rules. An idealised approach for the development of wellfield operating rules, which would support or incorporate implementation of the capture principle approach to sustainable groundwater use, is provided in the form of a flowchart.

Impeding implementation of the capture principle approach to sustainability is the fact that the approach is intertwined with adaptive management. The elements taken into account in the development of operating rules each require updating as the wellfield is pumped and the aquifer responds, providing new information. Confidence levels are a useful way to illustrate the necessary adaptive management approach, in which yield estimates are updated, leading to operating rules update. Confidence levels established for numerical modelling have therefore been adapted and combined with levels of information on aquifer yields and infrastructure characteristics that are likely to be available, to provide overall confidence levels (grades) for operating rules.

Mainstreaming the Capture Principle Approach

The study investigated the degree to which the identified idealised approach for the implementation of the capture principle approach to sustainable groundwater use is mainstreamed in the following tools and guidelines:

- Groundwater Resources Assessment Phase II (GRA II) (DWAF, 2006a)
- Groundwater Resources Assessment Phase III (GRA III) (DWA, 2009a)
- Development of a Groundwater Resources Assessment Methodology for South Africa: towards a holistic approach (Allwright et al, 2013)
- Groundwater Resources Directed Measures (GRDM) manual (Dennis et al, 2013)
- A Guideline for the Assessment, Planning and Management of Groundwater Resources in South Africa (DWAF, 2008b)
- Groundwater Management Functions study (Riemann et al, 2011)
- South African Groundwater Decision Tool (Dennis et al, 2002b)
- Groundwater Planning Toolkit for the Karoo (Murray et al, 2012)
- Flow Characteristic (FC)-Method (Van Tonder et al, 2002).

Although several of these existing tools or guidelines do use water balance based methodologies for quantification of groundwater resources, they also contain methodologies appropriate for parts of a capture principle-based groundwater assessment. Lacking in existing tools and guidelines, however, are the following elements of a capture principle-based approach:

- Discussion of upscaling individual borehole yields, or information from pump tests, to a maintainable aquifer yield.
- Specific discussion of the necessity of, and methods for, continued recharge measurement in order to quantify enhanced recharge due to abstraction.
- Guidance on or recognition that the elements of a capture principle-based groundwater assessment require update after an initial estimate, related to adaptive management approaches.
- Guidance on or specific discussion of the process by which the predicted conditions at future dynamic equilibrium (or the impacts of abstraction) are considered acceptable or not (i.e. whether they reflect sustainable groundwater use), and the required hydrogeological input to this decision. This 'sustainability decision' is what the Water Resource Classification System intends to cater for (a trade-off between water resource condition and development: DWAF, 2007; DWA, 2010c), however, the methodologies used are not aligned with a capture approach.
- Furthermore, the mainstreaming assessment highlights that the groundwater use authorisation and licensing process, which takes into account the reserve, is to some degree misaligned with the capture principle (through for example the recommendation of the use of a stress index).

Two supporting measures are developed to address (some of) the gaps identified in the mainstreaming assessment.

Supporting Measure 1: Sustainability Indicators
Implementing the capture principle approach to sustainability requires an initial estimate of the future dynamic equilibrium, and requires a decision to be taken based on the conditions of the future dynamic equilibrium. However, this future dynamic equilibrium cannot be predefined in high confidence. Monitoring designed specifically to provide information on the estimated future conditions is required to support continued update of the estimate of future dynamic equilibrium, and update of operating rule confidence level (and thus effectively manage sustainability).

Sustainability indicators are proposed as a tool to meet this monitoring requirement. The definition of sustainability applied in this study relates to the social, environmental and economic acceptability of the predicted environmental impacts of the resource use, and the indicators relate to updating the groundwater elements of the environmental impact prediction (only). The indicators include:

- Change in natural discharge
    - Detection of change in water table towards discharge point
    - Assessment of discharge
    - Flow in discharge-receiving environment
    - Chemical composition of discharge-receiving environment
- Change in pre-abstraction recharge
    - Detection of change in water table towards recharge zone
    - Direct detection of change in water table in recharge zone
    - Indirect detection of change in water table in recharge zone
    - Assessment of surface water flows in recharge zone
    - Chemical tracer for recharge source
    - Increased recharge
- Change in storage: Detection of change in water table
- Response time/status of aquifer towards new dynamic equilibrium: Age of water
- Associated assessments
    - Projection of timing and magnitude of drawdown cone
    - Recovery assessment.

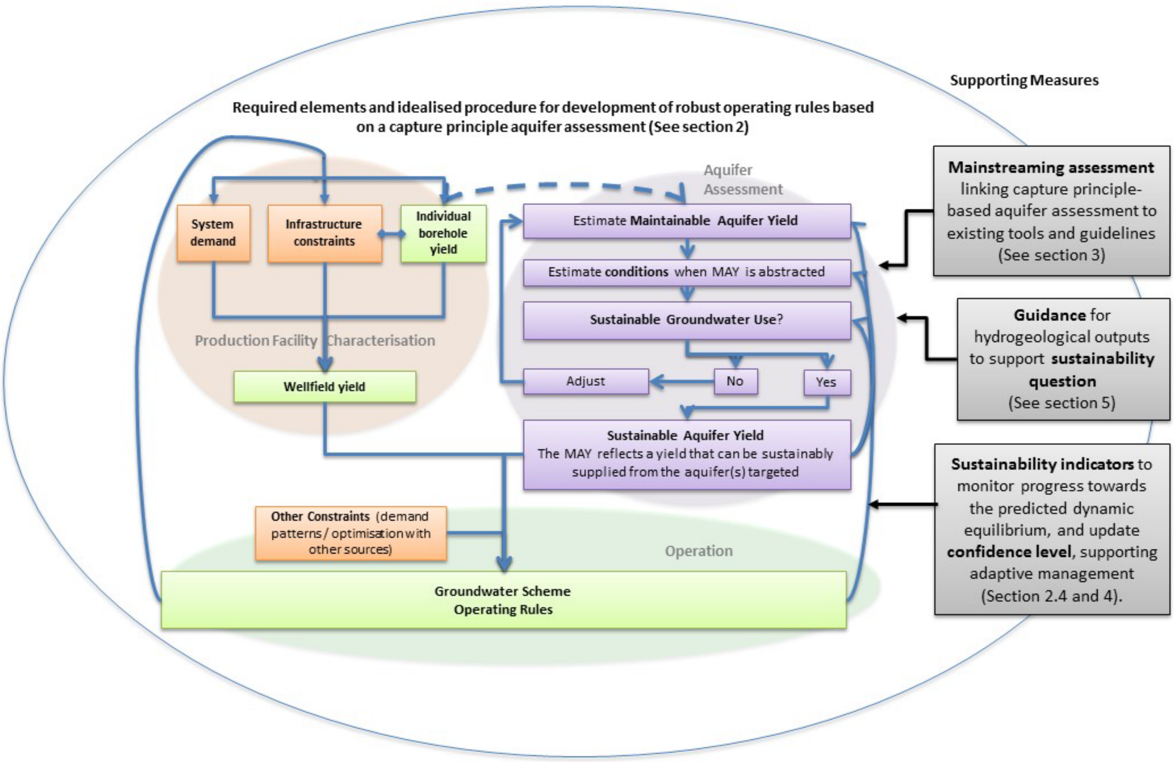
Supporting Measure 2: Hydrogeological Input for Sustainability Assessment
The mainstreaming assessment highlighted a lack of guidance or discussion relating to the decision as to whether the planned use is socially, environmentally and economically acceptable, and so can be considered sustainable. This sustainability decision will have to take different factors into account for each aquifer or situation as the particular social, environmental and economic impacts (or benefits) of abstraction are heavily case specific. The sustainability decision is also one that is to be taken by multiple stakeholders including regulators.

The decision nevertheless requires scientific information: a prediction of the conditions at future dynamic equilibrium. The following two hydrogeological inputs to the sustainability decision are recommended:

1) Quantification of the relationship between increasing abstraction and reducing discharge and other aquifer flows. The shape, gradient and scales of the curves will vary for each system.
2) Estimation of the response time. If this is very long, then the flows at a reasonable water supply planning and environmental timescale should be determined.

Decision Framework

The resulting decision framework tool is an amalgamation of the recommended approach for the development of wellfield operating rules, which would support or incorporate implementation of the capture principle approach to sustainable groundwater use, with supporting measures. The recommended approach has to be applied over the scale of an aquifer, or with consideration of the hydraulic boundaries or capture zone of the abstraction under question. There are hugely variable and case-specific factors to consider when assessing the sustainability (acceptability) of groundwater abstraction. The aquifer assessment steps depicted in the procedure for determining the sustainable groundwater yield are nevertheless applicable in all cases.



Decision Framework for Sustainable Groundwater Use

Case Studies

Various aspects of the capture principle, and of the established decision framework, are illustrated in the following two case studies:

- the West Coast aquifers in the region of Langebaan and Saldanha Bay in the Western Cape, South Africa
- the Maloney's Eye Steenkoppies Compartment, near Magaliesberg in the West Rand District Municipality, Gauteng, South Africa.

For each case study, a numerical groundwater model is developed in order to estimate conditions at the future dynamic equilibrium and hence quantify impacts and changes to the natural aquifer flow regime under abstraction (reduced storage, reduced discharge, enhanced recharge). The first of the recommended hydrogeological outputs to support a multi-stakeholder decision over groundwater sustainability is, therefore, provided for each case study.

The case studies demonstrate the trade-off between groundwater abstraction and reduced discharge, as follows:

- In the West Coast aquifers case study, the maintainable aquifer yield at the West Coast District Municipality (WCDM) wellfield, derived from the capture principle-based assessment, is significantly higher than previous estimates (3.5 to 5.5 million m^3/a, compared to 1.1 million m^3/a) whilst taking the following limitations into account: preventing ingress from the Berg River; preventing dewatering of the confined aquifer.

- In the case of the Maloney's Eye Steenkoppies aquifer, the relationship between abstraction across the aquifer and discharge at Maloney's Eye has been quantified. As there are direct users of the aquifer (i.e. borehole abstractions), and users of the discharge at Maloney's Eye, this relationship can be used to determine allocation to each user group.

Both case studies illustrate the inability of a water balance approach to accurately determine available groundwater yields.

- The recharge to the West Coast aquifers is significantly higher than the combined private and WCDM abstraction, but combined abstraction rates equivalent to only 30% recharge would induce ingress from the Berg River (due to the position of the wellfield) which would likely be considered unacceptable. Application of water balance calculations to estimate yield without taking impacts into account could overestimate available yields and lead to failure of the groundwater supply.
- The Maloney's Eye Steenkoppies aquifer, is heavily utilised (and critically stressed) according to the current abstraction/recharge ratio. Yet the model illustrates that the flow from Maloney's Eye is likely being sustained (albeit at a lower rate) by induced recharge from surface water and irrigation return flows. A water balance based calculation of available yields would have limited abstraction to significantly lower than 23 million m^3/a (current natural recharge estimate) whereas model results show abstraction of 25 million m^3/a can occur with Maloney's Eye discharge remaining at 5–6 million m^3/a.
- Therefore, in one case study, rates significantly less than virgin recharge can be harnessed (at the abstraction location tested), and in the other case study, abstraction rates greater than virgin recharge are feasible supported by induced recharge.

Capture principle-based assessments, which invariably require numerical models, are often assumed to be only appropriate or necessary for moderately (20–65%) to heavily used aquifers (> 65%). Steenkoppies would be classified as heavily used and critically stressed (> 95%). However, the West Coast aquifers would be classified as minimally used, yet application of the capture principle-based approach to groundwater assessment is still deemed necessary as there are limits on yield that need to be considered.

Key Messages

The key message from the study is the decision framework itself; this represents the recommended approach to establishment of operating rules that incorporate the capture principle. The operating rule grades, and associated sustainability indicators, provide a mechanism to implement the adaptive management cycles necessary to update maintainable aquifer yields, and in turn, operating rules.

Although capture principle-based groundwater assessments are recommended as applicable in all cases, they are certainly critical wherever there are sensitive receptors (related to potential risks of abstraction), and wherever the natural discharge is local to the abstraction point.

In many ways, the capture principle approach to sustainable groundwater use is not radically different to existing approaches. Many studies do assess impact on water levels (storage), and make estimates of stream flow reduction (as illustrated by the mainstreaming assessment showing that elements of the capture principle are fully accommodated in some existing tools). But, implicit to the decision framework is a recommendation that a fresh orientation to messaging is required (by hydrogeologists in groundwater assessments), in which the capture principle is put at the centre of planned groundwater use:

- Groundwater levels will decline due to pumping, and this (alone) is not an indication of unsustainability. At the onset of pumping, abstracted water is met by storage, hence water levels will decline when an aquifer is pumped. The (rate of) decline in water levels will reduce as a new dynamic equilibrium is established in which abstraction is met by reduced discharge and/or enhanced storage. At this point, there is no further loss of storage, yet if abstraction continues, the loss of storage is not reversed. This loss of storage/change in water levels must be estimated.
- Natural discharge will reduce *and/or* recharge will increase (at some point), given that the abstracted water must have a source. The reduction in discharge/ increase in recharge, and the response time, must be quantified.

Only two case studies were incorporated in this project to illustrate the capture principle and test the decision framework. It is recommended that these case studies be collated, along with other potentially pre-existing illustrations of the capture principle (ideally including monitored data showing reduction of discharge/enhancement of recharge), into a compendium of capture principle assessments to further demonstrate and promote the capture principle approach to sustainable groundwater use.

Although this project contributes to overcoming the technical challenges associated with the gap between theory and current practices, other significantly wide-ranging issues contribute to this gap. The mainstreaming assessment highlights that the groundwater use authorisation and licensing process, which takes into account the reserve, is to some degree misaligned with the capture principle. This misalignment is also compounded by the prevalent application of water balance approaches in any regional to national scale groundwater assessment in which it is often deemed impractical to assess groundwater resources at the aquifer scale, although this is necessary for a capture principle-based assessment (primarily due to project budget and timescale). The project did not aim to address these gaps, yet without change in these spheres, the gap between theory and implemented methodologies will remain.

It is recommended that South Africa transitions to an approach in which the Department of Water and Sanitation manages numerical models of all the major aquifers for abstraction management and resource protection (i.e. GRDM). This is the only way to change the status quo, fully implement the capture principle approach to sustainable groundwater use, and avoid the prevalence of water balance approaches in regional studies.

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Participation of the project team in a concurrent Department of Water and Sanitation project, (the development of operating rules for water supply and drought management for stand-alone dams/schemes (South)), encouraged incorporation of the link between the capture principle approach to sustainability and wellfield operating rules. Section 2.2 and 2.3 of this report are informed by DWS (2015a).

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List of Acronyms

AAS	Adamboerskraal Aquifer System (West Coast aquifers case study)
CRD	Cumulative rainfall departure
DWA	Department of Water Affairs
DWAF	Department of Water Affairs and Forestry
DWS	Department of Water and Sanitation
EAS	Elandsfontein Aquifer System (West Coast aquifers case study)
EC	Electrical conductivity
GEUS	Geological Survey of Denmark and Greenland
GMU	Groundwater management unit
GRA I	Groundwater Resources Assessment project (first phase)
GRA II	Groundwater Resources Assessment project (second phase)
GRA III	Groundwater Resources Assessment project (third phase)
GRDM	Groundwater resources directed measures
GW	Groundwater
K	Hydraulic conductivity
LAU	Lower aquifer unit (West Coast aquifers case study)
LRAS	Langebaan Road Aquifer System (West Coast aquifers case study)
MAE	Mean annual evaporation
mamsl	Metres above mean sea level
MAP	Mean annual precipitation
MAR	Mean annual runoff
MAY	Maintainable aquifer yield
mbgl	metres below ground level
mS/m	milli siemans per metre
NGA	National groundwater archive
NGDB	National groundwater database
NWA	National Water Act (Act 36 of 1998)
RMSE	root mean square error
PCG	pre-conditioning conjugate gradient
RQO	Resource quality objectives
SAWS	South African Weather Service
SW-GW	Surface water-groundwater
UAU	Upper aquifer unit (West Coast aquifers case study)
WARMS	Water Authorisation Registration Management System
WCA	West Coast aquifers
WCDM	West Coast District Municipality
WMA	Water Mmanagement Area
WRC	Water Research Commission

1. PROJECT BACKGROUND

1.1. MOTIVATION

South Africa will be short of three-billion cubic metres of water by 2030 (NBI, 2012). Meeting this demand is vitally important for the economic security of the country, illustrated by the finding that 71% of South African business respondents indicated that they have already experienced financially material water-related impacts (NBI, 2012). Maximized yet sustainable groundwater use is a priority, and can help meet the vision that before 2030, all South Africans will have affordable access to sufficient safe water, to live healthy and dignified lives (National Planning Commission, 2011).

The well-documented and scientifically accepted principles of groundwater flow theory dictate that water withdrawn artificially from an aquifer is derived from a decrease in storage in the aquifer, a reduction in the previous discharge from the aquifer, an increase in the recharge, or a combination of these changes (Theis, 1940, cited in Bennett et al, 1988). On pumping, water levels will decline, natural discharge may decline, and recharge may increase. The associated decrease in discharge or increase in recharge has been termed 'capture' of water (Lohman, 1972, Bennett et al, 1988, Seward et al, 2006), and it is the ability of aquifer pumping to capture discharge and enhance recharge that dictates the aquifer's yield. It follows that an assessment of the sustainability of groundwater abstraction would quantify these changes in the flow regime, and determine whether the changes and their associated impacts are considered acceptable, termed here the capture principle approach to groundwater sustainability.

However, many current tools to support groundwater management broadly apply water balance type calculations in which groundwater availability is directly linked to some proportion of pre-abstraction recharge. These calculations are often performed at quaternary catchment scale, and aquifers (or catchments) with high use compared to recharge are generally identified as 'stressed' or 'over-used'. The underlying assumptions, (over)-simplifications and limitations of the approach are not always made explicit. This abstraction/recharge approach can be useful for broad-scale resource planning. For under-utilised aquifers it could certainly provide a cheap and easy go ahead. However, the approach can limit groundwater development based on a perceived stress. This is inappropriate given that sustainable groundwater development does not directly depend on recharge. Nevertheless, water balance approaches and misperceptions continue to prevail; individual borehole yields are interpreted and reported as aquifer yields; groundwater level declines are often linked directly to unsustainability of abstraction; and if an aquifer can support abstraction without continued mining this is often assumed to reflect sustainable groundwater use, without an assessment of the impact of this use on the flow regime i.e. defining the source of abstracted water through quantifying reduced discharge and enhanced recharge.

Perhaps the gap between the theoretical and the implementable is the reason that South African policies remain focused on an abstraction/recharge approach. Studies suggest that impeding the implementation of the capture approach to sustainability is the fact that the approach is intertwined with adaptive management. Not all information can be known prior to development, hence the future dynamic equilibrium under a new pumping regime cannot be predefined in high confidence (Sophocleous, 2000, Seward et al, 2006). Management must proceed on less than ideal information, and decisions adjusted as groundwater use continues. This is awkward to regulate. Seward (2010) identified incorporating adaptive management to deal with uncertainty as a key challenge facing environmentally sustainable groundwater use in South Africa. Seward (2010) also points out that selecting appropriate scientific methodology, neither too simplistic nor too involved, is a recurring challenge. This is particularly relevant for regional (water management area) to national scale investigations, for which it is often not practical to assess groundwater resources at the aquifer scale, primarily due to the project budget and timescale that would be necessary, yet such an investigation would be required for an assessment based on the capture principle.

The ultimate purpose of this project was to promote the capture principle approach to sustainable groundwater use. The project proposed a tool, in the form of a 'decision framework', that could facilitate the translation of theoretical hydrogeological principles for sustainable groundwater use based on the capture principle approach into practice. Furthermore, sustainability indicators were proposed as part of this tool. The contents of the tool (the decision framework) were to be informed by research into the necessary interventions to bridge the theory-action gap; hence its details were not explicitly provided at the proposal stage. The project reveals insights into the applicability (in terms of scales and situations) of the capture principle approach to sustainable groundwater use.

This report emanates from an unsolicited Water Research Commission (WRC) project named 'Sustainability Indicators and Decision Framework for Sustainable Groundwater Use', project K5/2311.

1.2. RESEARCH AIMS

The aims of the 3-year research project were:

1. To provide an understandable and accessible description of the capture approach to sustainable groundwater use
2. To promote use of the capture approach for sustainable management of groundwater, especially for moderately and highly used aquifers
3. To develop a decision framework tool to guide a user through the adaptive management cycle of managing groundwater use sustainably, especially for moderately and highly used aquifers
4. To develop sustainability indicators that can be used to qualitatively and quantitatively manage groundwater use during progression through adaptive management cycles, especially for moderately and highly used aquifers
5. To demonstrate the applicability and benefit of the capture approach in (at least) two test cases.

1.3. OUTLINE OF DOCUMENT

The report is presented in two parts:

1. A theoretical discussion of the capture principle. Emanating from this is the decision framework tool that aims to facilitate the translation of theoretical hydrogeological principles for sustainable groundwater use based on a capture approach into practice.
2. Illustration of the capture principle in two case studies. In both case studies a capture principle-based groundwater assessment is implemented, thus (parts of) the decision framework tool are implemented and tested.

Part 1, the theoretical discussion of the capture principle, contains an accessible description of the principles of groundwater flow theory that are part of the capture principle, including a description of aquifer yields, the response time, groundwater sustainability and the significance of recharge (section 2.1). Groundwater is accessed via pumping from individual boreholes and wellfields, so to implement sustainable groundwater use, wellfield operation must be informed by the maintainable aquifer yield. Aquifer yields and groundwater sustainability therefore cannot be considered in isolation of borehole and wellfield yields and operation. Hence section 2.2 and 2.3 provide a summarised description of wellfield infrastructure and yields, and an outline of the idealised approach for the development of wellfield operating rules that would support or incorporate implementation of the capture principle approach to sustainable groundwater use. Section 2.4 provides a discussion of adaptive management and associated confidence levels for operating rules. Section 3 assesses the degree to which the identified idealised approach for the implementation of the capture principle approach to sustainable groundwater use is mainstreamed in selected existing tools and guidelines. The outcome of the mainstreaming assessment is the identification of gaps that impede the implementation of the capture principle approach. The remaining two sections of part 1 aim to meet (a selection of) these gaps: section 4 defines sustainability Indicators that are designed to support adaptive management, and section

5 outlines the recommended hydrogeological outputs required to support the multi-stakeholder decision over groundwater sustainability.

The sustainability indicators (section 4) and the recommended hydrogeological outputs (section 5) both form 'supporting measures' in promoting the established idealised approach (section 2) for the development of wellfield operating rules. These supporting measures are drawn together with a flowchart of the idealised approach, and together make up the decision framework tool.

Part 2 illustrates various aspects of the capture principle, and of the established decision framework, through the following two case studies:

- The West Coast aquifers in the Langebaan and Saldanha Bay regions of the Western Cape, South Africa
- The Maloney's Eye Steenkoppies Compartment, near Magaliesberg, in the West Rand District Municipality, Gauteng, South Africa.

For each case study, a numerical groundwater model is developed in order to quantify impacts and changes to the natural aquifer flow regime under abstraction (reduced storage, reduced discharge, enhanced recharge). The recommended hydrogeological outputs to support a multi-stakeholder decision over groundwater sustainability (one of the supporting measures of the decision framework) are therefore provided for each case study. The reporting of these case studies follows the standard numerical modelling approach (Barnett et al, 2012).

Lastly, sections 9 and 10 provide a summary of the case study outcomes, and the project conclusions and recommendations.

Part 1 Theoretical Discussion of the Capture Principle Approach to Sustainability and the Development of a Decision Framework

2. THE CAPTURE PRINCIPLE APPROACH TO SUSTAINABLE GROUNDWATER USE

2.1. AQUIFER ASSESSMENT

2.1.1. The Water Budget Myth

Many current tools for groundwater management and estimation of groundwater yields broadly apply water budget (or balance) type calculations. The basis for the water balance approach is that an aquifer, as a contained unit, is in a natural balance over the long term or in steady-state; recharge enters the aquifer, and water leaves the aquifer via discharge. Applying thinking consistent with the Law of Conservation of Matter, it is seemingly logical to think then that if an aquifer is pumped more than it is recharged, it will one day run out of water (Delvin & Sophocleous, 2005). This thinking can be expressed as a (quasi steady-state) mass balance:

Recharge (R) = pumping (P) + discharge (D) (Equation 1)

Where R is the recharge rate (L3/T), generally assumed to come from rainfall and be unaffected by pumping, D is the discharge rate of water that is not captured by the pumping (L3/T) (to streams, lakes, evaporation, or evapo-transpiration, etc.), P is the net pumping rate (L3/T) and all three terms are positive numbers (Delvin & Sophocleous, 2005).

Water budget (or balance) type approaches[1] therefore generally compare groundwater use against recharge, and sometimes include the groundwater contribution to baseflow and to the reserve[2] (Dennis et al, 2013). The approach assumes that abstraction should not exceed recharge if it is to be considered sustainable. Aquifers with high use compared to recharge are generally identified as 'stressed' or 'over-utilised'. This is illustrated in the following examples:

1. Groundwater stress, based on a comparison of the ratio of abstraction and (mean annual) recharge, is a suggested criterion as input in the Water Resource Classification System (Dennis et al, 2013, DWA, 2013a). Significant water resources have been classified in the Mokolo catchment, and the resulting groundwater classes were almost wholly based on groundwater stress, with some consideration given to groundwater quality (DWA, 2013b).
2. Groundwater water use licence applications consider, amongst other aspects, the groundwater stress index, calculated based on the ratio of abstraction to (mean annual) recharge in the (surface water derived) quaternary catchment in which the planned abstraction is to take place.
3. Desktop water resources planning assessments routinely compare abstraction from the Water Authorisation Registration Management Systems (WARMS), to mean annual recharge (usually derived from GRA II, DWAF, 2006a), to determine broad-scale groundwater availability. The assessment may be performed over a specified area or aquifer, but is usually at quaternary catchment scale. The Western Cape Government's Integrated Water Resources Management Action Plan applied this approach and identified, for example, 15 quaternary catchments in the (previous) Breede-Overberg Water Management Area (WMA) as 'over-abstracted quaternary catchments'. Based on this assessment, further groundwater development was heavily cautioned against in these catchments (PGWC, 2011).
4. The catchment management strategy for the (previous) Breede-Overberg WMA takes the abstraction/recharge comparison a step further and states that their groundwater management target is to ensure abstraction is less than recharge in all catchments (BOCMA, 2010).

This abstraction/recharge approach to groundwater availability can be useful for broad-scale resource planning. For potentially under-utilised aquifers it could provide a rapid indication of an aquifer with very low use compared to recharge, suggesting further groundwater development may be feasible. However, these examples show how this approach is limiting groundwater development in cases where there is high groundwater use compared to recharge,

[1] In this report, 'water balance approaches' refers to those approaches that directly link sustainable or available groundwater yields to some portion of pre-abstraction recharge, and may include a consideration of groundwater contribution to baseflow.

[2] The reserve is defined in the National Water Act (NWA) (Act 36 of 1998) as the quantity and quality of water required to satisfy basic human needs and to protect aquatic ecosystems.

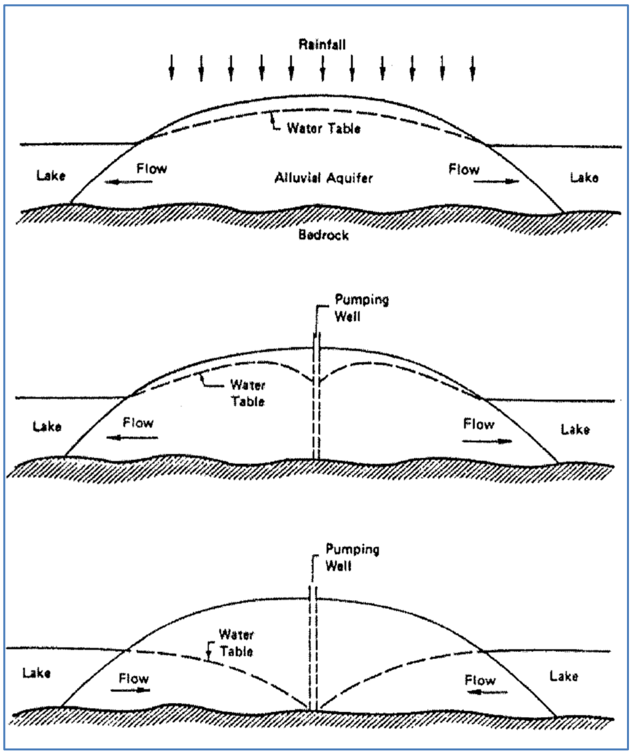
and perhaps incorrectly so, as various authors have shown a number of ways in which water balance type calculations are incorrect, inaccurate and are an inappropriate approach for groundwater management.

Application of the water balance approach implicitly assumes that the recharge rate does not change from the original or natural rate, due to pumping (Delvin & Sophocleous, 2005). This assumption is false as there are a number of mechanisms, each widely accepted and dictated by fundamental groundwater flow theory, by which pumping can affect recharge, as follows:

I. By the radius of influence of pumping capturing groundwater flow which would have otherwise discharged to surface water (including elevated evapotranspiration losses at river banks) or, by reaching a surface water body, generating a 'recharge boundary' in which natural gradient driving discharge from groundwater to surface water is reversed (Kruseman and de Ridder, 1994). Alternatively, if the natural setting is that of a losing stream, then the natural recharge from surface water would be enhanced to an upper limit as groundwater levels are reduced from pumping and the natural gradient driving recharge to groundwater from surface water is increased (Sophocleous, 2002).
II. By increasing recharge if the pumping takes place in areas of rejected recharge (i.e. areas where water tables temporarily rise close to, or to, the ground surface during rains, causing potential recharge waters to be lost as surface runoff (Delvin & Sophocleous, 2005).
III. By increased drainage gradients from the unsaturated zone to the aquifer resulting from lowering of the water table (Delvin & Sophocleous, 2005).
IV. By reduced evapotranspiration losses if the water table falls below the root zone.
V. By increasing drainage gradients through a leaky confining layer to a confined aquifer beneath it whose potential head is reduced through pumping (Theis, 1940, cited in Delvin & Sophocleous, 2005).
VI. By decreasing recharge if the pumping leads to aquifer consolidation and an altered hydraulic conductivity of the recharging units (Delvin & Sophocleous, 2005).

Application of the water balance approach also implicitly assumes that the change in discharge from the original or natural discharge under a pumped regime is equal to the pumped yield (related to equation 1, and Delvin & Sophocleous, 2005). Given that recharge does not remain constant under pumping, the pumped yield cannot only be equated to the change in discharge.

Related to item (I) in the list above, the water balance approach implicitly assumes that an aquifer is a closed system or a fixed directional flow system in which water only enters through prescribed pathways and only leaves through different prescribed pathways. Aquifers may behave as fixed directional systems under some conditions, but they can change when those conditions change. This behaviour is well accepted and illustrated by the example of an island aquifer in Figure 2-1 (Bredehoeft et al, 1982, Bredehoeft, 2002).



The alluvial aquifer sits within the island, surrounded by a lake. Under natural conditions the aquifer receives recharge from rainfall, and discharges into the lake at the boundaries of the island (top sketch in Figure 2-1). When the aquifer is pumped at a low rate (second sketch in Figure 2-1), the radius of influence or 'capture zone' of the well remains within the island boundaries. In this situation equation 1 applies, i.e. the pumped water is sourced from reduced discharge.

If the pumping rate is increased to or beyond the point at which the capture zone reaches the island boundary (bottom sketch in Figure 2-1), the pumping is greater than the pre-pumped discharge rate, the hydraulic gradient is reversed and lake water will enter the system. What was previously a pathway for discharge becomes a pathway for recharge. The pumped yield is made up of the original discharge rate plus enhanced recharge, and equation 1 is invalid.

Figure 2-1 Schematic cross section of an aquifer on a circular island under natural conditions and under pumping (from Bredehoeft et al, 1982)

In summary, the water balance approach fails to consider the dynamics of groundwater flow systems, by the over-simplification of a closed or uni-directional flow system. The water balance approach also considers only the long-term or steady-state of an aquifer, does not consider the dynamic nature of aquifer behaviour, and does not allow for the use or management of water stored in the aquifer (use of stored water is discussed in section 2.1.3). It is essentially equivalent to managing a surface water dam at a constant storage/water level only.

Application of the water balance approach also implicitly requires that one must know the (pre-abstraction) recharge rate to determine maintainable pumping rates. This assumption is also strictly incorrect, as shown in section 2.1.2. This discussion on the inapplicability of basing sustainability of aquifer abstraction directly on recharge echoes that given in DWAF (2010). Yet strategies and methodologies from the national Department of Water and Sanitation (DWS) remain aligned to the water balance approach (DWA, 2010a and 2011a, Dennis et al, 2013).

2.1.2. Maintainable Aquifer Yield and the Capture Principle

Using the water balance approach to calculate yields available for pumping at a particular location, or yields available across an aquifer, ensures that the pumped yield remains less than pre-abstraction recharge, and therefore natural discharge i.e. the situation depicted by the upper sketch in Figure 2-1. Based on the falsehood of the assumptions underlying the water balance approach, limiting groundwater development based on the results of the water balance approach is considered inappropriate, and represents potential under-utilisation of groundwater.

As a more accurate and appropriate alternative to the water balance approach and its application in defining aquifer yields, Delvin & Sophocleous (2005), define the 'sustainable pumping rate' of an aquifer as the pumping rate that can be maintained indefinitely without mining an aquifer *continually*. So as not to confuse a sustainable pumping rate with sustainable groundwater use, the term '**maintainable pumping rate' of an aquifer** is preferred, which is simplified to **maintainable aquifer yield** throughout the text (see glossary, section 12). Under natural conditions, aquifers approach a state of dynamic equilibrium: over the long term (which can be hundreds of years); wet years in which recharge exceeds discharge are offset by dry years when discharge exceeds recharge (Sophocleous, 2000). Discharge from wells upsets the dynamic equilibrium by producing a loss from aquifer storage; and under pumping at the maintainable aquifer yield, over time a new state of dynamic equilibrium is approached when there is no further loss from storage. The maintainable aquifer yield therefore is a rate that can be maintained by reduced discharge and or induced recharge in a new dynamic equilibrium, such that aquifer storage doesn't deplete (and is therefore maintainable in the long term). It does not *directly* depend on the pre-abstraction recharge rates (Bredehoeft et al, 1982, Sophocleous, 2000, Delvin & Sophocleous, 2005).

Maintainable aquifer yields, as defined above, rather depend on the ability of pumping to intercept and decrease natural discharge, or increase recharge (or both). Pumping that decreases discharge and enhances recharge is termed 'capture' (Lohman, 1972, Bennett et al, 1988), or the 'capture principle' (Seward et al, 2006).

> Water withdrawn artificially from an aquifer is derived from a decrease in storage in the aquifer, a reduction in the previous discharge from the aquifer, an increase in the recharge, or a combination of these changes… The decrease in discharge plus the increase in recharge is termed capture. Capture may occur in the form of decreases in the groundwater discharge into streams, lakes, and the ocean, or from decreases in that component of evapotranspiration derived from the saturated zone. After a new artificial withdrawal from the aquifer has begun, the head in the aquifer will continue todecline until the new withdrawal is balanced by capture (Lohman, 1972, p. 3).

Maintainable aquifer yields[3] are determined through pumping and aquifer assessment, and are influenced by land use and a variety of time-varying factors, hence they cannot be considered fixed. The yield must be updated through adaptive management practices (section 2.4). Recharge of course still has relevance to the maintainable aquifer yields and sustainable groundwater use, as summarised in section 2.1.5.

2.1.3. Time to Equilibrium or the Response Time

At the onset of abstraction, prior to reaching any equilibrium, all pumped water is taken from aquifer storage around the borehole. As pumping continues over time, the proportion delivered from storage will reduce, and the proportion delivered from other sources (reduced natural discharge, enhanced recharge) will increase. After a certain amount of time, the aquifer will reach a new state of dynamic equilibrium where no water is contributed from storage. This principle is fundamental because the yield contribution from different sources in the aquifer's new dynamic equilibrium must dictate an assessment of sustainability (section 2.1.4).

The length of time required for the new state of dynamic equilibrium to be reached, where there is no further loss from storage, has been referred to as the aquifer response time. Each of the schematics presented above (Figure 2-1) reflect an aquifer that has reached a new dynamic equilibrium. To illustrate this transition over time to a new dynamic equilibrium, Alley and Leake (2004) established a numerical model of an alluvial aquifer in Paradise Valley, Nevada. Under the original dynamic equilibrium, replicated by the steady-state simulation of a numerical model, the aquifer derives its recharge from leakage from surface waters and groundwater inflow from adjacent aquifers (88%). The aquifer discharges mainly through evapotranspiration (96%), and also through outflow to surface water. The steady-state calibrated model is subjected to 300 years of a pumping regime, followed by 300 years of no pumping (Alley and Leake, 2004). Through analysing the modelled aquifer flow balance over time, it is possible to track the source of pumped water. As would be expected from the capture principle, the results show that initially (year 1.5) the majority of pumped water is derived from aquifer storage (80%). As time progresses (25 years), the contribution from storage has reduced

[3] The term maintainable aquifer yield is compared to borehole yield in section 2.2, and compared to existing South African terminology for aquifer yields in Box 2-1

(44%), and the natural discharge to evapotranspiration has decreased, now contributing 52% of the abstracted water. After 300 years, the contribution from storage has reduced to a minimal amount (3%), and the abstracted water is largely contributed by capturing natural discharge to evapotranspiration (72%), and enhanced recharge from surface water (21%). The response time is somewhat more than 300 years.

A similar exercise is presented in Seyler & Hay (2013), which reveals an aquifer with a much faster response time of ± ten years. The test case is an alluvial aquifer in a large river basin, deriving its inflow from mountain springs which seep into the alluvium at its upstream boundaries ('springs' in Figure 2-2), and discharging to the river at the downstream end of the aquifer ('river' in Figure 2-2). The aquifer is subjected to a hypothetical abstraction, equivalent to 80% of the pre-abstraction recharge rate. The impact over time on the aquifer flow balance is shown in Figure 2-2, with flow into the aquifer shown as positive, and flow out of the aquifer negative. The initial storage imbalance shows that the majority of abstracted water is derived from storage. The largest changes in the natural fluxes occur in the initial five years, and by ten years the system has largely reached a new dynamic equilibrium, illustrated by the loss from storage coming close to zero (Figure 2-2). Although the shape of the curves indicates some initial instability in the model, and there were no constraints applied on the potential supply from mountain springs as recharge, the results showed that the recharge could be increased by almost 100%, driven by the changing hydraulic gradient allowing more inflow. The natural discharge to rivers reduced by almost a half.

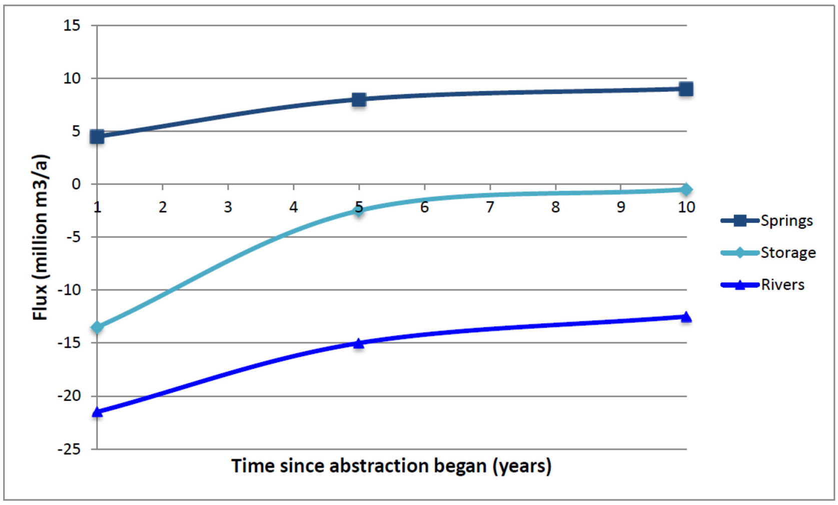


Figure 2-2 Aquifer fluxes over time since abstraction began (adapted from Seyler & Hay, 2013)

Another way to illustrate the change in the aquifer flow regime over time is to collapse what is shown in the y-axis in Figure 2-2, and rather plot the abstracted water source as a percent of storage and a percent of reduced discharge or enhanced recharge, over time. This plot is shown in Figure 2-3, developed for an idealised, two-dimensional, homogeneous and isotropic system (Sophocleous, 2000), with groundwater abstractions near a surface water course, normalised and plotted with dimensionless time on the x-axis. The transition curve illustrates the following fundamental principles (Sophocleous, 2000):

- The general shape of the transition curve is retained in systems with different boundaries and parametric values.
- The rate at which dependence on groundwater storage converts to dependence on surface water depletions is highly variable and is particular to each case. For certain aquifers and pumping situations the x-axis may reflect hours, and the aquifer reaches a new equilibrium in 10 000 hours, just over a year. The x-axis may also reflect 10 000 days, or just over 27 years.
- The extent of groundwater mining depends entirely upon the time frame. Initially, all groundwater developments 'mine' water (i.e. deplete storage) but ultimately they do not.

- The length of time (t) required for the new state of dynamic equilibrium to be reached depends upon (1) the aquifer diffusivity (expressed as the ratio of aquifer transmissivity to storativity, T/S), which is a measure of how fast a transient change in head will be transmitted throughout the aquifer system; and (2) upon the distance from the well to the point of discharge (x). It does not depend on yield abstracted.

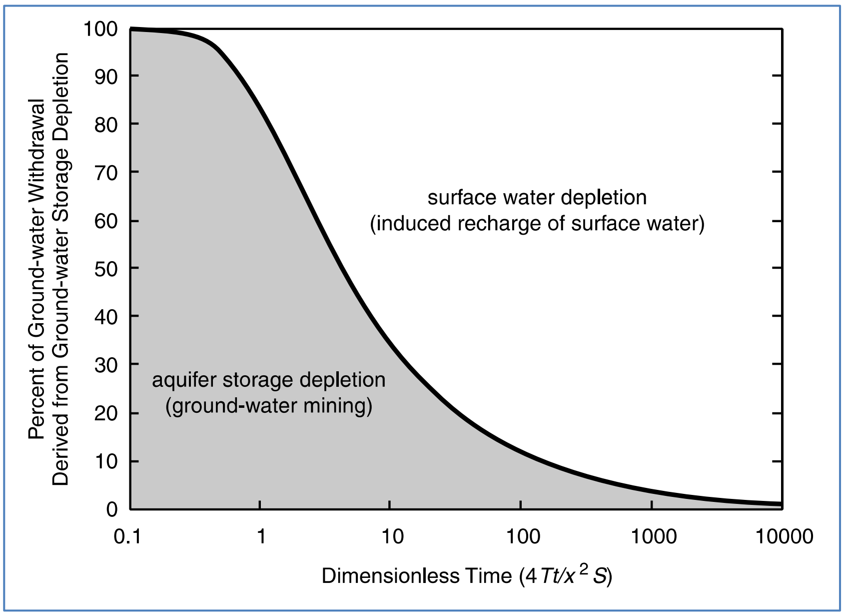


Figure 2-3 Graph showing transition curve from reliance upon groundwater storage at onset of pumping, to induced recharge of surface water. [T is transmissivity, S is storage, x is the distance from pumping well to stream, and t is time]. (From Sophocleous, 2000.)

2.1.4. From Maintainable Aquifer Yields to Sustainable Groundwater Use

Sustainability is generally defined using the so-called 'triple bottom line' which states that economic, social and environmental interests should be balanced. It recognises that everything humans do has an impact on the environment, but that this impact should be balanced against the social and economic benefits. If **sustainable groundwater use** is defined as groundwater use that is socially, environmentally (ecologically) and economically acceptable, then long-term abstraction of the maintainable aquifer yield does not necessarily reflect sustainable groundwater use. As per the definition, the maintainable aquifer yield simply refers to a yield that can be maintained by reduced discharge or enhanced recharge, without continually mining the aquifer or depleting aquifer storage. A critical step from quantification of a maintainable aquifer yield to quantification of sustainable groundwater use is to first determine the yield contribution from each source under the new dynamic equilibrium, and the conditions at this new equilibrium (i.e. the equivalent water level distribution), and then take a socio-economic-environmental decision as to whether this is acceptable. If it is acceptable, then the long-term abstraction of the maintainable aquifer yield can be considered sustainable groundwater use (Sophocleous, 2000, Alley and Leake, 2004), and the maintainable aquifer yield reflects a **sustainable aquifer yield** (see glossary section 12).

A confined aquifer with natural discharge occurring entirely offshore as subterranean seeps, which is recharged in mountainous areas, can illustrate this definition for sustainable aquifer yield. A maintainable aquifer yield that (a) is greater than pre-abstraction recharge, (b)derives the abstracted yield from capturing all the offshore discharge and (c) enhances recharge through capturing some rejected recharge that was previously entering mountain streams, may be considered sustainable if:

- There is no ecological impact, or there is acceptable ecological impact of the cessation of subterranean discharge, and:

- There is no ecological impact, or there is acceptable ecological impact from the stream-flow reduction in the recharge zone, through the capturing of additional recharge from runoff. If there is some impact, then the socio-economic benefit of the groundwater use must outweigh this.

Sustainability considerations must also include groundwater quality; upon abstraction, enhanced recharge may be derived from a neighbouring aquifer unit with unacceptable water quality.

It is clear from the definition provided for a sustainable aquifer yield that its quantification is not purely a scientific exercise undertaken by a hydrogeologist. Significantly, what is considered socially, environmentally and economically acceptable is subjective; different groups will have different perspectives, and regulators must be involved. As the socio-economic-environmental conditions against which the abstraction is being compared will change, due to external factors, so too will the acceptability. Furthermore, the sustainable aquifer yield is dependent on the maintainable aquifer yield, which is also not a fixed number. The sustainable aquifer yield therefore also cannot be considered a fixed number and requires updating through adaptive management practices (see adaptive management discussion section 2.4, and relationship to wellfield operation in section 2.2), as stated by Seward et al (2006):

> A range of 'sustainable yields' is possible for any given situation, dependent on how intervention takes place, and what is deemed acceptable (or at least permissible). It is therefore open to debate whether 'sustainable yield' is the best term to use, since it appears to suggest that there is a single, fixed yield that can be determined. (Seward et al, 2006, p. 481)

A sustainable aquifer yield should always be provided in full, with the predicted impacts of abstraction, why these have been considered acceptable and by whom, and an explanation that as these subjective decisions change and/or knowledge of the aquifer is updated, the sustainable aquifer yield will also change.

In addition to consideration of sustainable aquifer yields, various authors motivate that abstraction policy and management plans be linked to the timescale that the aquifer takes to reach this new equilibrium, or the 'response time' (Sophocleous, 2000, Alley and Leake, 2004, Bredehoeft and Durbin, 2009, Sophocleous, 2012). For example, if the impact on groundwater discharge to surface water under the new equilibrium is ecologically unacceptable, yet the response time extremely slow (i.e. beyond lifetimes), perhaps this groundwater use can be considered sustainable, and can proceed during the time lag for a water supply source to be secured.

The elements making up the sustainable aquifer yield, and their definitions, are connected as a flow chart in Figure 2-4. As is clear from the use of the term 'aquifer' in the definitions, the maintainable aquifer yield, and sustainable aquifer yield are defined across the aquifer or between abstraction points and relevant hydraulic boundaries. These terms and definitions contrast to those widely applied in South Africa. This contrast and the resulting motivation for the derivation of new terms for use in this study is provided in Box 2-1.

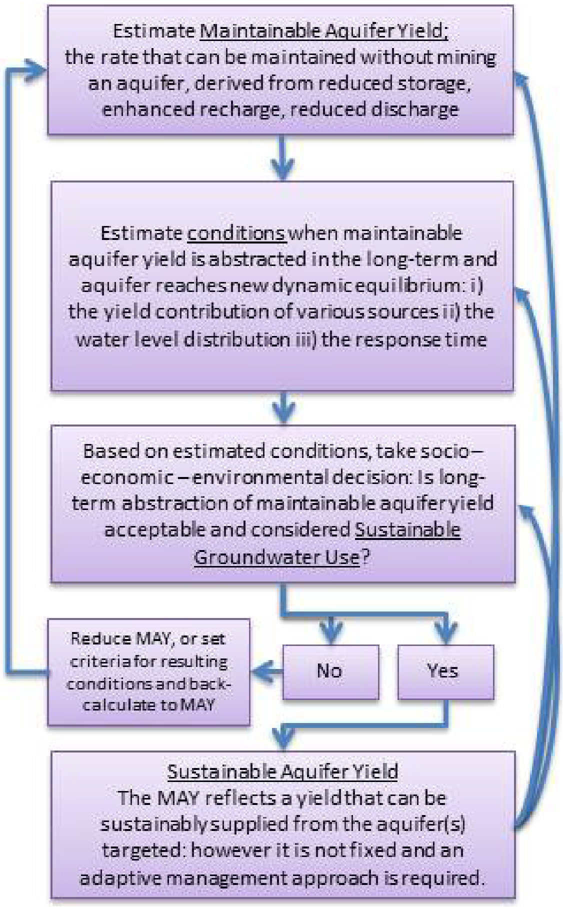


Figure 2-4 Flow chart of elements making up a sustainable aquifer yield

Box 2-1 Comparison to existing aquifer yield terminology

The term **safe yield** of an aquifer is widespread in its use. The definition may be traced back to Theis, (1940) who states that there is a 'perennial safe yield equivalent to the amount of rejected recharge and natural discharge it is feasible to utilize. If this amount is not exceeded, the water levels will finally reach an equilibrium stage' (Theis 1940; cited in Theis, 1957, p. 11). This definition of the term safe yield is clearly aligned with a capture principle-based approach, and it is equivalent to maintainable aquifer yield.

However, safe yield is sometimes used to describe a derived yield based on a proportion of pre-abstraction recharge (Kalf and Woolley, 2005). The terminology documented in the Groundwater Dictionary largely reflects the terminology used in the South African groundwater community (DWA, 2011b). The definition provided for safe yield in the dictionary (considered therein synonymous with aquifer yield) is:

> Definition: Safe yield is defined as the maximum rate of withdrawal that can be sustained by an aquifer without causing an unacceptable decline in the hydraulic head or deterioration in water quality in the aquifer.
> Description: This is a concept or objective embracing the sustainable volume of water that can be abstracted from an aquifer over the long term. At present there is debate in the literature regarding the quantification of this concept, factors influencing safe yield and time frames that should be considered. (DWA, 2011b)

The definition involves a rate of withdrawal that can be sustained, hence can be related in part to the maintainable aquifer yield. However, the definition includes consideration that this yield does not cause unacceptable decline in hydraulic head or water quality, but excludes consideration of other conditions such as reduced discharge and whether this is acceptable or not. This definition is therefore considered imprecise, sitting somewhere between the description for maintainable aquifer yield and sustainable aquifer yield given in section 2, and allowing for uncertainty. This uncertainty is reflected in the description of the term, stating 'there is debate… regarding quantification of [safe yield]'. A definition of yield should be based around its method of quantification. The definition does not cater fully for capture principle-based assessment, which requires an explicit difference between maintainable and sustainable, hence the development of maintainable aquifer yield and sustainable aquifer yield terminology used here.

In an attempt to map and quantify South Africa's groundwater resources, Vegter (1995) published a set of maps titled 'Groundwater Resources of the Republic of South Africa', which included: i) a provisional recharge map based on effective rainfall in the 'absence of other relevant and better-suited information'(Vegter, 2001); and ii) an approach for the classification and depiction of aquifer storage. The information in these maps, considered preliminary by their author, was progressed to provide a **groundwater harvest potential** map of South Africa (DWAF, 1998), where the harvest potential was defined as 'the maximum volume that may be abstracted per square kilometer per annum without depleting the aquifers'. It was derived based solely on the recharge and storage maps through application of a water balance approach. As noted by Vegter (2001): 'Recharge and storage are not the sole factors that determine groundwater availability or harvest potential. The ultimate factor is the extent to which natural groundwater losses may be eliminated and translated into a pumped water supply'. The harvest potential definition is in conflict with a capture approach to yield as it states 'without depleting the aquifer', whereas the theory of the capture principle dictate that all groundwater abstraction must deplete aquifer storage (section 2). The maps and the term are based on a water balance approach and considered inapplicable to a capture principle-based assessment.

2.1.5. The Significance of Recharge

Although the maintainable aquifer yield doesn't directly depend on pre-abstraction recharge rates, recharge rates are still extremely relevant and required in aquifer yield assessments.

Determination of the pre-abstraction dynamic equilibrium of an aquifer, predictions of how this will change under an abstraction regime, and estimating the response time, all generally require numerical modelling. A model cannot be calibrated without a good handle on the pre-abstraction recharge rate. Also, in a pre-abstraction dynamic equilibrium, the storage volume is stable and the recharge and discharge are balanced. So, just as the maintainable aquifer yield and capture concept discuss capture of natural discharge (and enhanced recharge), capture of natural discharge is numerically equivalent to capture of natural recharge. A wellfield that captures all of the natural discharge can equally be argued to have captured all of the natural recharge (Kalf and Woolley, 2005).

If the natural recharge rate can be linked to the capture yield, can the natural recharge rate be linked to the sustainable aquifer yield (based on the above definition that sustainable aquifer yield is the maintainable aquifer yield, if this rate represents sustainable groundwater use)? It is sometimes implicitly suggested (in water balance type approaches), that if a yield to set to be a certain percent less than recharge rates, then it is ensured as a sustainable aquifer yield. All that is actually ensured is that the abstraction yield that could be maintained by reduced discharge. The abstraction may well still induce recharge, and the impact of this, and the impact of the reduced discharge must still be determined to assess whether this abstraction represents sustainable groundwater use. And, on the contrary, the maintainable aquifer yield may well also be higher than natural recharge.

2.1.6. Discussion and Link to Operating Rules

The groundwater flow theory underpinning the description of maintainable aquifer yield, the capture principle and sustainable groundwater use, is well documented and scientifically accepted (note the internationally well-known, peer-reviewed papers referenced, and simply described in Theis, 1957). However, groundwater availability assessments based on a water balance type approach continue to prevail and may unnecessarily limit groundwater development (examples given in section 2.1.1). Misperceptions of groundwater flow hydraulics lead to other prevalent practices, such as: (a) groundwater sustainability being equated with there being 'no impact', and particularly with no impact on general water levels or pressures; (b) where groundwater levels are constant in time, then abstraction is often considered as having 'no impact'; (c) and if actual or projected drawdown doesn't reach a receptor, the abstraction is assumed to have 'no impact' on the said receptor, with an assessment of the associated flow regime or source of abstracted water wholly missing. Given that continued extraction from an aquifer can only be balanced by increased input or reduced output of an aquifer, pressure levels alone are not an indication of impact or sustainability (Kalf and Woolley, 2005). Only considering the groundwater level in an aquifer ignores an assessment of the increased input or reduced output of an aquifer, and the socio-economic-environmental impact of this.

The current discussion is fundamental to the sustainable use of groundwater. Groundwater is accessed via pumping from individual boreholes and wellfields, so to implement sustainable groundwater use, wellfield operation must be informed by the maintainable aquifer yield. Aquifer yields and groundwater sustainability therefore cannot be considered in isolation of borehole and wellfield yields and operation (described in section 2.2 and 2.3). If a sustainable aquifer yield is based directly on the maintainable aquifer yield, without an assessment of the significance of the changes in flow regime, and the operating rules for a scheme are in turn based on this, the scheme may fail. Perhaps the abstracted yield captures the discharge to a distant spring, and the spring dries up with an unacceptable ecological impact. Or perhaps the maintainable aquifer yield is derived from enhanced recharge from surface water, and another surface water user commences abstraction, such that the enhanced recharge is reduced and the groundwater abstraction can no longer be maintained, and the aquifer is continually mined.

2.2. PRODUCTION FACILITY CHARACTERISATION

2.2.1. Borehole Yield

A borehole yield is defined here simply as the yield that can be maintained from a borehole without borehole failure (i.e. inability to continue to pump). This borehole yield is related to the maintainable aquifer yield, but it is not simply equivalent to it:

- If a borehole yield is maintainable in the long term, then it must be less than or equal to the maintainable yield of the aquifer
- If a borehole yield is not maintainable in the long term, then:
    - it may be greater than the maintainable yield of the aquifer, (i.e. a pump test shows runaway dewatering)
    - it may be less than or equal to the maintainable yield of the aquifer, but the borehole is sited poorly within the aquifer and unable to harvest the maintainable aquifer yield (i.e. the borehole is too close to an annealed fault at the boundary of the aquifer)
    - it may be constrained by infrastructure (expanded on below).

The number of boreholes required to harvest the maintainable aquifer yield will vary:

- Few boreholes required: In a highly transmissive aquifer system, the yield of one borehole may exceed the maintainable aquifer yield, as it is able to harvest the maintainable aquifer yield effectively. Or, the maintainable aquifer yield may be so low that one borehole exceeds it.
- Many boreholes required: In a large aquifer system or an aquifer with lower transmissivity, individual borehole yields may be much less than the maintainable aquifer yield and several boreholes may be necessary to harvest the maintainable aquifer yield.

In most cases, the borehole yield will be less than the maintainable aquifer yield, constrained by borehole infrastructure and construction. Borehole yield is influenced by time-varying factors as infrastructure can impact yield over time (e.g. by clogging), and aquifer conditions can change, impacting borehole yield. It therefore cannot be considered a fixed number.

Given that the primary tool to investigate a maintainable aquifer yield is pump testing at individual boreholes, the maintainable aquifer yield and the individual borehole yield are often confused. In the case where the individual borehole yield has been constrained by infrastructure, or if it is less than a maintainable aquifer yield, then assuming the borehole yield is directly related to the maintainable aquifer yield can again limit groundwater development.

Box 2-2 Comparison to existing borehole yield terminology

The term **borehole yield** is used widely, often with different meanings, and having been derived in different ways.

An individual borehole yield is defined here simply as the yield that can be maintained from a borehole without borehole failure (i.e. inability to continue to pump). The maintainable aquifer yield is defined here as a rate that can be maintained by reduced discharge and or induced recharge in a new dynamic equilibrium of the aquifer, such that the aquifer storage doesn't deplete. Both the borehole yield and maintainable aquifer yield are influenced by time-varying factors and therefore cannot be considered fixed numbers; rather they require updating through adaptive management.

A widely used tool in South Africa for pump test analysis to determine borehole yields is the Flow Characteristic (FC)-method (Van Tonder et al, 2002). This is a spreadsheet tool in which a minimum water level is defined by the user (often level of pump inlet), and the FC-Method provides a borehole yield that maintains water levels above that level for the duration specified, (using various analytical equations). The borehole yield calculated has been termed the 'sustainable borehole yield', which is defined as 'the safe amount of water that can be abstracted from a borehole for a long time (usually one or two years), without the water level reaching the position of the pump or the main water strike' (Van Tonder 2001, cited in Van Tonder et al, 2002). This was developed to enable a simple, easily applicable management rule to prevent borehole failure. The resulting yield depends on the position of the pump, which is generally limited by several infrastructure and budget factors rather than a level which represents some sustainable use of the aquifer. Basing the yield on a user-defined water level, and interpreting the outcome as a 'sustainable yield' implicitly assumes that any impacts associated with the abstraction of this yield are considered acceptable. They may not be. A borehole yield derived by the FC-method is equivalent to the **borehole yield** as defined here, and should not necessarily be termed a sustainable borehole yield.

2.2.2. Infrastructure Constraints

Borehole and wellfield infrastructure can limit borehole yield (i.e. beyond that which is hydraulically available as controlled by transmissivity). As an example of these constraints, Table 2-1 provides a list of primary wellfield infrastructure and the potential impact it may have on borehole yield. Each of these impacts alone may be minor (much less than 1 ℓ/s) yet together sum to be more significant (several ℓ/s).

Table 2-1 List of wellfield infrastructure and its potential impact on individual borehole yield

Wellfield infrastructure	Examples of potential impact on borehole yield, wellfield yield and borehole performance
The borehole itself: casing, screening, cement and gravel pack, head-works	• Poorly designed or installed gravel pack can reduce borehole yields if their transmissivity is less than that of the aquifer formation • Poorly installed casing can seal off productive parts of the aquifer
Pump infrastructure: the pump, its capacity, the pump cable and associated electrics	• A borehole that has been drilled too narrow may impact on the (submersible) pump motor size. This is possible if the borehole yield was underestimated prior to drilling and a narrower hole selected for cost reasons • The installed pump may have been selected incorrectly and be unable to deliver the yield against the hydraulic head • The electricity supply to the borehole must be sufficient to power the pump • Pumps have differing requirements for the water level to be maintained above the pump inlet, and for water flow past the pump for cooling. These may impact yield attainable in boreholes with a short water column
Pipeline infrastructure: rising main, delivery pipeline	• The pipeline diameter may be insufficient for the flow that the borehole can produce, hence constrain the yield • With more than one borehole pumping into the same pipeline, back pressure or friction in the pipeline may impact on the yield
Water treatment facilities	• Whatever the treatment technology is, it may only be able to function under a certain bracket of flow rates, thus potentially constraining yields
Ancillary infrastructure includes monitoring equipment such as flow meters and water level equipment	• Impacts on yield from ancillary equipment should always be avoided, for example in the case of an impellor flow meter slowing flow, and a coiled water level meter pipe crowding the borehole reducing the borehole's transmissivity
Maintenance of all of the above infrastructure	• The lack of routine maintenance of all of the above equipment leads to a reduction in borehole efficiency (i.e. same drawdown produces lower yield, or drawdowns increase for the same yield). Preventative operation and routine maintenance is required to manage chemical and biological incrustation, iron biofouling, and physical plugging of screen and surrounding formation. • Wellfield downtime for routine maintenance and breakdown should to taken into account when estimating the annual/long-term yield available from the wellfield.
Other boreholes within the wellfield	• Individual borehole yields cannot be summed to provide a wellfield yield, given that each borehole may sit within the radius of influence or cone of depression of another borehole. The combined yield achievable with these drawdown impacts must be determined. • Hydraulic interference from neighbouring wells increases the drawdown at the pumped hole beyond that generated by pumping within the pumped hole, and in a pump with fixed rate or power, this slightly reduces the yield due to the higher pumping head.

2.2.3. Production Facility or Wellfield Yield

Natural controls on borehole yield (transmissivity and aquifer setting), interference from other boreholes in the wellfield, and constraints from other wellfield infrastructure, together dictate the total yield of the complete production facility or wellfield infrastructure. The term production facility is preferred to wellfield, as groundwater abstraction may be from spring capture and diversion, or elements other than boreholes. However, as wellfield is a more widely used term, it is applied here. Consumptive demand on the scheme is also a factor that influences wellfield yield, given that infrastructure is initially established with an eventual use in mind. These aspects influencing wellfield yield are summarised in Figure 2-5. Wellfield yield, as defined here, does not incorporate 'sustainable', as the infrastructure may be oversized and able to pump at yields higher than the sustainable aquifer yield. Given that borehole yield is influenced by time-varying factors and cannot be considered a fixed number, neither then can the wellfield yield be considered a fixed number.

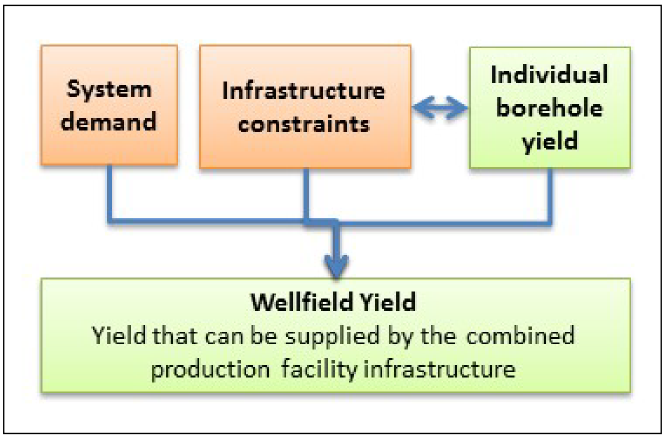


Figure 2-5 **Flow chart connecting aspects that influence wellfield yield**[4]

2.3. GROUNDWATER SCHEME OPERATING RULES

Natural constraints on the groundwater system are accommodated in the sustainable aquifer yield (i.e. impact on other users, the environment, protection against saline intrusion), and infrastructure constraints are accommodated in the wellfield yield. Prior to translation of these two yields into operating rules, various other constraints must be considered. These are specific to each system and an example of what these may include are:

- constraints introduced by seasonal demand patterns;
- interaction with other schemes or sources to meet demand, and their constraints, and an optimisation between sources;
- potential water quality factors that motivate to operate schemes in a particular way, i.e. for dilution or mixing of water.

Taking the wellfield, sustainable aquifer yield and other constraints into account, operating rules can be developed. In combining the elements considered to make up robust operating rules, Figure 2-6 presents an idealised approach to the development of operating rules (definitions are repeated in Box 12-1 for reference). The rules should also consider, where possible, maximising cost effectiveness, i.e. minimisation of electricity consumption and energy footprint (and by association, complete life cycle water footprint). The aquifer assessment elements shown in Figure 2-6 are defined at aquifer scale, or up to the hydraulic boundaries or capture zone of abstraction under consideration.

In summary, a scheme is considered to have **robust operating rules** if:

The scheme operator operates the scheme based on a living document, updated through adaptive management cycles, containing a set of rules describing the required operation of the wellfield and any associated infrastructure that makes up the complete scheme. These operating rules are based on, or take into account:

- The sustainable aquifer yield, incorporating natural constraints
- The wellfield yield and infrastructure constraints
- The demand on the system and any constraints this may introduce, and optimisation approaches.

[4] Orange represents manmade influences; green represents items influenced by a combination of natural (aquifer) aspects and manmade aspects.

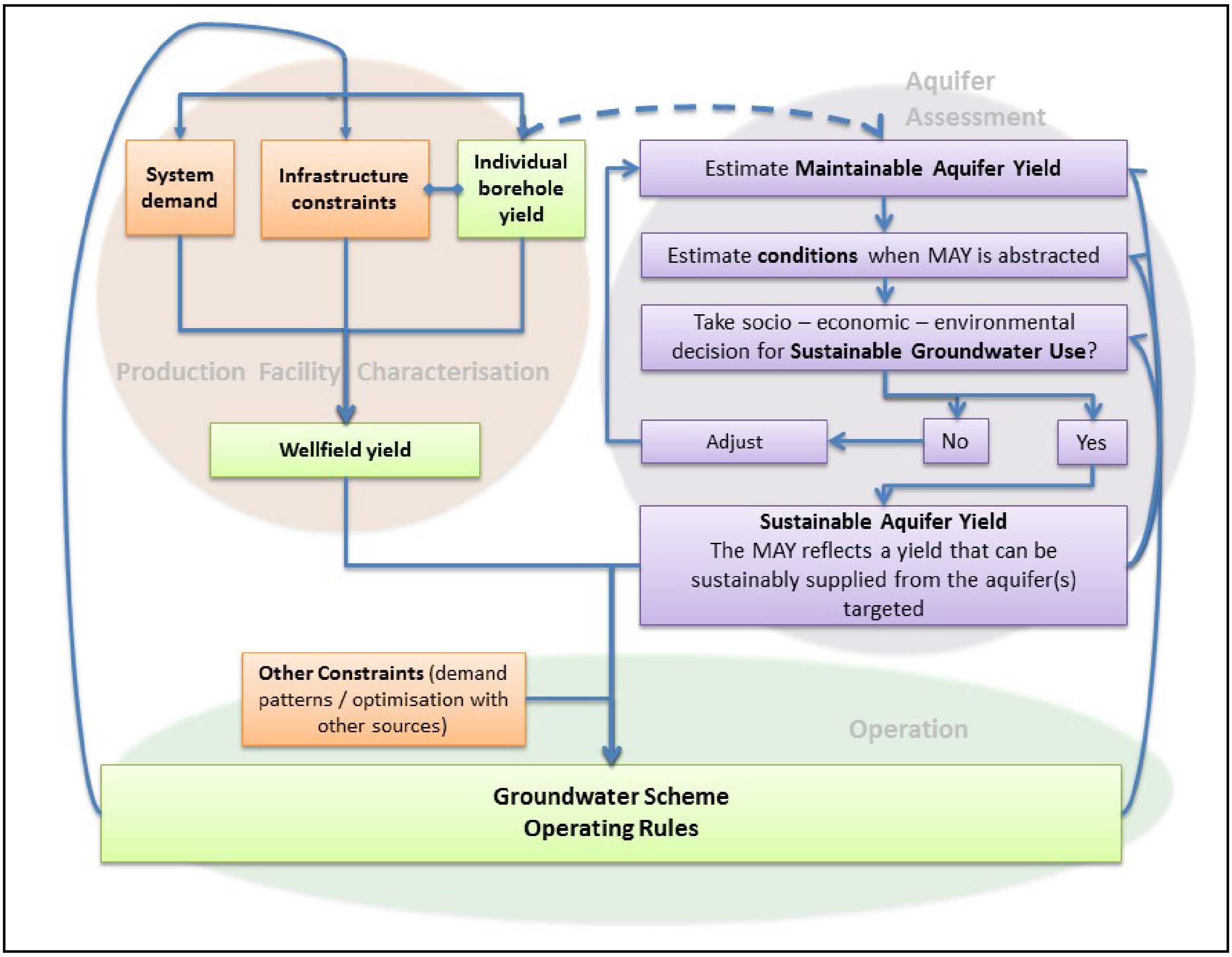


Figure 2-6 Idealised procedure for development of robust operating rules incorporating a capture principle-based aquifer assessment[5]

2.4. ADAPTIVE MANAGEMENT AND CONFIDENCE LEVELS

The procedure shown in Figure 2-6 is idealistic, yet it does depict the various elements required for robust operating rules that incorporate a capture principle approach.

The capture principle approach to sustainable groundwater use is motivated in Seward et al (2006), but perhaps the gap between the theoretical and the implementable is the reason that South African policies remain focused on a water balance or abstraction/recharge approach. Impeding the implementation of the capture principle approach to sustainability, is the fact that the approach is intertwined with adaptive management. Not all information can be known upfront, hence the future dynamic equilibrium cannot be predefined in high confidence (Sophocleous, 2000, Seward et al, 2006). Management must proceed on less than ideal information, and decisions must be adjusted on a regular basis, illustrated by the upward arrows in Figure 2-6. This is awkward to regulate. Seward (2010) identified incorporating adaptive management to deal with uncertainty as a key challenge facing sustainable groundwater use in South Africa. The required adaptive management approach needs to be entrenched into the operation and management of the production facility or wellfield through the routine update of operating rules.

[5] Purple represents terms relevant to an aquifer (greater detail in Figure 2-4), orange represents manmade influences, green represents items influenced by a combination of natural (aquifer or borehole) aspects and manmade aspects.

The procedure is hence shown as a linear one with aquifer assessment and infrastructure characterisation (or development) occurring in parallel, and with cyclic updates (upward arrows in Figure 2-6), yet this in itself is idealistic. In practice, schemes are often developed on a borehole by borehole basis and may be pumping with little to no assessment of maintainable aquifer yield (nor the sustainable aquifer yield as defined here). Aquifer assessment often lags far behind infrastructure development, and the elements required under aquifer assessment are often partly developed at some later stage based only on operational results. Nevertheless, the procedure holds relevance in these situations, and that scheme would need an aquifer assessment (i.e. development of the right hand side of the flow chart) to improve the operating rules to those considered here as robust.

Although this infrastructure-focused approach may yield a groundwater supply perhaps quicker than more detailed upfront assessment, not having estimated the maintainable aquifer yield (and estimated the source of water and predicted the aquifer conditions when the maintainable aquifer yield is abstracted), means allowing an undefined future dynamic equilibrium to establish. Sequencing aquifer characterisation after infrastructure development limits the user to acting in hindsight; if monitoring results show unacceptable (unsustainable) impacts corrective measures must be put in place. Implementing a capture principle approach requires an upfront estimation of the conditions at the future dynamic equilibrium. This generally requires a numerical model. There is a common perception that data to support an aquifer assessment/numerical modelling does not exist prior to extensive exploration. However, enough regional data does generally exist for a low-confidence model to be developed in early exploration stages, and, if completed early, it can provide indicative results, provide assistance in defining a monitoring protocol, and the detail and confidence of output results can grow as more data becomes available, keeping pace with exploration. New information gained from monitoring is used to update predictions (Lugar & Hay, 2002), a process coined as 'monitor-model-manage' in Hay et al (2006, p. 3). A model initiated early in the exploration stage starts as a useful demonstrative tool but with large uncertainties; however, over time the model becomes an accurate quantitative prediction tool for wellfield operation (Seyler et al, 2013). Potential problems can be flagged early from model results, acting pre-emptively rather than acting in hindsight.

Regardless of the approach used in aquifer assessment (numerical modelling or other), little aquifer information can be gleaned until there is some pumping, and the maintainable aquifer yield can only be estimated through pumping. So, more fundamental than applying adaptive management in order to update an early estimation of the conditions at a future dynamic equilibrium, adaptive management is essential for abstraction management given that some pumping must start to inform borehole yields and maintainable aquifer yields, with only preliminary operating rules.

Referring back to the procedure in Figure 2-6, a useful way to illustrate the necessary adaptive management cycles is with confidence levels. An ideal approach would use available data during the establishment of a groundwater supply and the development of infrastructure, to develop a numerical model for an initial estimate of maintainable aquifer yield. The likely data availability would render this estimate 'low confidence' (Barnett et al, 2012). This low-confidence estimate in turn renders the operating rules low confidence. Confidence in operating rules would increase as the procedure in Figure 2-6 is iterated. Likewise, in early stages of a groundwater supply, the infrastructure will not yet have been used under a range of situations, and not all infrastructure constraints would be known in detail.

Barnett et al (2012) provide confidence levels for numerical modelling results. As numerical modelling is the most appropriate tool for undertaking the assessment required for estimating the maintainable aquifer yield and, in turn, operating rules, these confidence levels have been adapted and combined with likely levels of information on aquifer yields and infrastructure characteristics, to provide overall confidence levels (grades) for operating rules (Table 3-2). Those schemes that have operating rules based only on infrastructure characterisation are considered less than low confidence (Grade 0 Preliminary, Table 7-1). Schemes with no operating rules, and with Grade 0 operating rules are considered unacceptable. Grade 1 low-confidence rules may be acceptable if the risk of abstraction is very low (i.e. no sensitive receptors, natural discharge is significantly distant).

Whether low-confidence operating rules are considered acceptable for a scheme, and little effort is required to improve the confidence, or whether a high-confidence operating rule is necessary, depends on the aquifer setting in question and the potential risks of poor definition of sustainable aquifer yield, as stated by Lugar & Hay:

> The nature of the abstraction programme and the complexity of the monitoring programme should be based on the potential risks, sensitivity and value of the potentially affected environment, the social and economic need as well as the long-term risks associated with limited understanding. (Lugar & Hay, 2002, p5)

Table 2-2 Operating rule grades (adapted from Barnett et al, 2012)

Operating rule grade	Status of operating rule	Status of supporting infrastructure characterisation	Status of supporting aquifer assessment				
			Summary	Status of model used to support aquifer characterisation			
				Data	Calibration	Prediction	Key Indicator
None	None exist	Boreholes exist but little information is available.	None or some assessment of the aquifer setting (geology, hydrostratigraphy, hydraulic properties, natural water balance) is available	n/a			
Grade 0: Preliminary	Preliminary operating rules based mostly on infrastructure characterisation elements	Boreholes have been tested, and individual borehole yields or combined wellfield yields dictate operating rules.	Some assessment of the aquifer setting (geology, hydrostratigraphy, hydraulic properties, natural water balance) is available.	n/a			
Grade 1: Low confidence	Operating rules are in place, based on an assessment of (at least most of) the required elements. Reflects one iteration of the required process.	Individual borehole yields estimated through pump tests, and infrastructure constraints assessed to estimate wellfield yield	An assessment of the aquifer setting (geology, hydrostratigraphy, hydraulic properties, natural recharge and discharge) is available, and used as input to an assessment of the maintainable aquifer yield. This assessment is low confidence based on status of model used. A sustainability decision has been taken by relevant stakeholders and/or regulatory authority, understanding the limitations	• Few or poorly distributed data on groundwater and geology available • Data unavailable or sparsely distributed in areas of greatest interest • Little or no data on land use, soils, river flows and stage elevations	• Calibration is based on an inadequate distribution of data. • Calibration only to datasets other than that required for prediction.	• Predictive model time frame far exceeds that of calibration. • Temporal discretisation is different to that of calibration	• Model predictive time frame > 10 times longer than transient calibration period. • Stresses in predictions > 5 times higher than those in calibration. • Stress period or calculation interval is different from that used in calibration.

Operating rule grade	Status of operating rule	Status of supporting infrastructure characterisation	Status of supporting aquifer assessment				
			Summary	Status of model used to support aquifer characterisation			
				Data	Calibration	Prediction	Key Indicator
			of the information on which it is based.				
Grade 2: Medium Confidence	Operating rules are updated, based on a second iteration of (at least most of) the required elements.	Low-confidence characterisation updated by > 1 year of operational data	Low-confidence assessment of the maintainable aquifer yield updated to medium confidence, based on status of model used. The sustainability of the medium-confidence maintainable aquifer yield has been assessed by relevant stakeholders and/or regulatory authority, understanding the limitations of the information on which it is based.	• Piezometric head data, metered extraction data and borehole log data available but may not provide adequate coverage • Stream flow data and baseflow estimates available at a few points	• Validation not undertaken • Long-term trends not replicated in all parts of model domain • Transient calibration to historic data but not extending to present • Seasonal fluctuations not adequately replicated in all parts of model domain	• Transient calibration over a short timeframe compared to prediction • Temporal discretisation used in predictive model different from that used in transient calibration • Level and type of stresses included in predictive model outside the range of those used in the transient calibration	• Model predictive time frame is three to ten times the duration of transient calibration • Stresses are two to five times greater than those included in calibration • The model has been reviewed and deemed fit for purpose by an independent hydrogeologist

Operating rule grade	Status of operating rule	Status of supporting infrastructure characterisation	Status of supporting aquifer assessment				
			Summary	Status of model used to support aquifer characterisation			
				Data	Calibration	Prediction	Key Indicator
Grade 3: High Confidence	Operating rules are updated, based on a third iteration of (at least most of) the required elements.	Medium confidence characterisation updated by > 5 years' operational data	Medium-confidence assessment of the maintainable aquifer yield updated to high confidence, based on status of model used. The sustainability of the high-confidence maintainable aquifer yield has been assessed by relevant stakeholders and/or regulatory authority, understanding the limitations of the information on which it is based.	• Spatial and temporal distribution of head observations adequately define groundwater behaviour, especially in areas of greatest interest • Clearly defined aquifer geometry • Reliable metered groundwater extraction data available • Reliable baseflow estimates at a number of points	• Adequate validation • Long-term trends adequately replicated where important • Seasonal fluctuations adequately replicated where important • Transient calibration uses recent data • Model is calibrated to heads and fluxes • Observations of the key modelling outcomes (i.e. dewatering volumes, contamination plume) dataset used in calibration	• Length of predictive model not excessive compared to calibration period • Temporal discretisation used in predictive model consistent with transient calibration. • Level and type of stresses in the predictive model within the range of transient calibration • Model validation suggests calibration appropriate for locations and/or times outside the calibration model	• Model predictive time frame < 3 times the duration of transient calibration. • Stresses are < 2 times greater than those included in calibration. • Temporal discretisation in predictive model is the same as in calibration. • Model has been reviewed and deemed fit for purpose by experienced, independent hydrogeologist with modelling experience

3. MAINSTREAMING THE CAPTURE PRINCIPLE APPROACH

3.1. Overview

Research into the perspectives of South African water sector experts revealed that one of the highest ranking tools or measures available for implementing sustainable and adaptive groundwater management in South Africa was to 'implement existing groundwater legislation and regulations', with 'formulation of new groundwater legislation and regulations' being ranked of minor importance (Knüppe, 2011, p. 73). The theme of the Geological Society of South Africa's groundwater division conference in 2015 was 'From Theory to Action', and carries Ralph Waldo Emerson's quote that 'an ounce of action is worth a ton of theory'. This conference theme and the perspectives reflected by Knüppe (2011) highlight the lethargy within at least the groundwater sector. South Africa has even developed countless strategies, yet real change remains slow. However, the current focus in the groundwater sector and wider water sector has shifted to action.

Although cognisant of this lethargy and the need for action over theory, the premise of this study was the need to promote implementation of the capture approach, given that application of water balance type assessments can lead to under-use of groundwater (limiting abstraction yields) or overuse (ill-defined yields and impacts, ill-defined concept of sustainability). The elements required to follow a capture principle approach to groundwater use (Figure 2-6) are diverse. They relate to groundwater management aspects including monitoring, the approach of an exploration programme, what needs to be included in a groundwater assessment, socio-economic-environmental considerations, and are also related to authorisation. A stand-alone guideline, or even set of tools, to implement the capture principle therefore is not possible, nor necessary, because the capture principle relates to almost all elements of groundwater management. Rather than develop a new isolated guideline an analysis of how the capture approach, and the elements that must be quantified to follow a capture approach, is mainstreamed or already accommodated within existing guidelines and tools was completed. Based on this analysis, requirements to support the implementation of the capture approach were determined, and (some) are addressed in subsequent sections.

The following questions shape an analysis of how the capture principle is, or can be, mainstreamed into existing policies, guidelines and tools:

- Where is the capture principle accommodated (implicitly or explicitly) in existing processes?
- Where is the capture principle misaligned with or excluded from existing processes, thus suggesting an update of that process would be required in order to implement the capture principle?
- What guidelines and tools are available to support the required assessments for a capture principle-based approach to groundwater assessment (those in Figure 2-6)?
- What are the challenges to implementing a capture principle-based approach to groundwater assessment (as per Figure 2-6), and which of these challenges are not (yet) adequately supported by existing processes?

3.2. Selected existing guidelines

The intention of the second phase **Groundwater Resources Assessment (GRA II)** was to build on the first phase (GRA I), develop a set of generic methodologies or approaches for quantifying groundwater resources in South Africa and establish the datasets required for applying the methodologies at national and aquifer scale (DWAF, 2006a). In assessing available groundwater resources the GRA II study's intention was to develop a robust algorithm that can be readily applied to both regional and national groundwater resources assessments. The GRA II project provides approaches and datasets (ranging from quaternary catchment scale, with some disaggregated to per km^2) for:

- groundwater storage (Task 1)
- recharge and the proportion of recharge that is available for abstraction (Task 2)
- groundwater use (Task 5)
- estimation of what should remain in the aquifer for a specific management criteria (i.e. to meet the reserve or prevent land subsidence) (Task 4).

Using the above listed numbers the 'surplus groundwater that can further be allocated' is quantified through direct application of the water balance approach for quantification of available groundwater resources. The following terms are defined within the methodology for quantification of groundwater resources, each of which are quantified across South Africa (in $m^2/km^2/a$):

- Average Groundwater Resource Potential: based on recharge plus storage, divided by some factor for drought during which no recharge occurs hence abstraction comes from storage minus baseflow.
- Current Groundwater Resource Potential: equivalent to the average groundwater resource potential, minus current use. The calculation for average groundwater resource potential uses current aquifer storage (not mean).
- Groundwater Exploitation Potential: equivalent to the groundwater resource potential multiplied by an exploitation factor based on borehole yields.
- Potable Groundwater Exploitation Potential furthermore takes into account likely groundwater qualities and excludes areas with poor groundwater quality.
- Utilisable Groundwater Exploitation Potential furthermore takes into account the requirements of the reserve.

The GRA II methodology for quantification of groundwater resources (groundwater potential) conflicts directly with a capture principle-based assessment. However, the GRA II study also provided detailed literature reviews and recommended methodologies for surface-groundwater interaction, including groundwater contribution to baseflow, and approaches for assessing the impact of abstraction on streamflow (DWAF, 2006b). A consideration of the impact on surface water must, to some degree, acknowledge the capture principle for the derivation of pumped water, and this methodology therefore contradicts the one provided for groundwater potential (and supports a capture principle-based assessment). The internal contradiction is acknowledged within the GRA II in a discussion about surface-groundwater (SW-GW) interaction:

> The methodology [that provided for quantification of SW-GW interactions] also presents quantitative results of the impacts of abstraction on baseflow. This would force geohydrologists to consider the impacts of recommended pumping rates on streamflow, and impacts on the ecological reserve. This may have implications on the Harvest Potential concept, since maximum pumping rates may no longer be controlled by recharge, storage and permeability, but also by allowable impacts on baseflow. The methodology could therefore be used to consider pumping allocations in terms of the Reserve.
>
> The methodology also considers the nonlinearity between groundwater abstraction and baseflow. Abstraction of groundwater cannot simply be removed from baseflow and considered equivalent to surface water abstraction, since groundwater abstraction may increase recharge and transmission losses during peak flows, reduce evapotranspiration losses and groundwater outflow. This should cause a revision in currently held views of interactions; that groundwater abstraction is considered based on recharge, ignoring impacts on surface water (view held by or applied in practice by many geohydrologists), or the opposite view that pumping is equivalent to baseflow depletion hence can be considered equivalent to a surface water use (a few held by many hydrologists). The distance dependency of groundwater abstraction and baseflow depletion suggests that groundwater allocations (to meet baseflow requirements for the ecological reserve) cannot simply be based on a quantitative value dependent on recharge calculations. The impacts on baseflow, hence on the Reserve, would be dependent on the location of the abstraction relative to the groundwater baseflow discharge area. A small abstraction next to the river may have a bigger impact than much larger abstraction away from a river. This has an implication that allocations should be based on location as well as recharge. A quantitative groundwater reserve may therefore be a fallacy, and allocations for abstraction should be based on allowable impacts rather than fixed recharge based values. (DWAF, 2006b, p. 11)

Given the national scale of the GRA II datasets, and the intention to develop a robust algorithm that can be readily applied to both regional and national groundwater resources assessments, the application of water balance approaches is understandable. In discussing persistence of water balance approaches and the challenges facing sustainable groundwater use in South Africa, Seward (2010) also points out that selecting appropriate scientific methodology, neither too simplistic nor too involved, is a recurring challenge. This is particularly relevant for regional- (WMA) to national scale investigations, where water balance approaches dominate, and in which it is often not practical to assess groundwater resources at the aquifer scale necessary for a capture principle-based assessment (primarily due to project budget and timescale). In summary, the GRA II results have become a key dataset used for desktop, preliminary (low-confidence), and regional groundwater resources assessments, where other local or site-specific data is not available.

Although the GRA II generated datasets incorporate methods based on water balance approaches (the groundwater resource potential methods) that conflict with a capture principle-based assessment, several aspects of the GRA II methodologies (the description of methodologies to determine recharge, and recommended model approaches for calculating surface-groundwater interaction) can be applied for certain aspects of a capture principle-based groundwater assessment. However, the groundwater resource potential (yield) approaches and datasets would need a significant update for the capture principle to be fully incorporated and supported by GRA II. The National Groundwater Strategy includes estimates of groundwater availability all taken directly from the data generated under GRA II (DWA, 2010a and DWA, 2011a).

The **Groundwater Resources Assessment Phase III (GRA III)** process was initiated in 2008 to update the methodologies and datasets and aimed to provide 'a broader methodology ultimately aimed at raising the profile of groundwater in South Africa and ensuring wider and more sustainable groundwater use' (DWA, 2009a, p. vi). As part of GRA III the previous GRA I and GRA II were reviewed, along with international groundwater assessment methodologies (DWA, 2009b). Several aspects of GRA II methodologies were discredited and it was motivated that these be discontinued. The GRA III study largely moved away from an attempt to standardise a groundwater assessment approach and generate national datasets, and therefore also moved away from regional water balance approaches. Some of the key outcomes of GRA III include a discussion of how to incorporate assurance of supply in groundwater; a series of recommendations for hydrogeological data gathering and storage which focus on centralisation of existing data largely held by private consultants; and recommendations and a generic approach for an aquifer-level assessment of groundwater (which utilises numerical modelling). The move towards numerical quantification and depiction of groundwater resources at a local scale, and calls for groundwater assessment methodologies to be tailored to the specific needs of the assessment, are in line with the international approaches reviewed (DWA, 2009b). The GRA III methodologies are generally aligned with a capture principle approach and provide applicable tools when carrying out a capture principle-based groundwater assessment (i.e. recharge methodologies are further critiqued beyond that presented in GRA II). However to explicitly support the capture principle some aspects of GRA III would require an update.

The Water Research Commission (WRC) supported the update of the GRA II methodologies (i.e. the development of GRA III). The study, titled '**Development of a Groundwater Resources Assessment Methodology for South Africa: towards a holistic approach**' developed a new methodology for quantification of SW-GW interaction based on water quality (Allwright et al, 2013). The study includes a detailed literature review of appropriate methods and tools for quantification of SW-GW interaction that can be a useful support tool when carrying out a capture principle-based groundwater assessment.

The **Groundwater Resources Directed Measures (GRDM) manual** (Dennis et al, 2013) outlines steps required to carry out a GRDM assessment, with the outcome being determination of the reserve, classification of water resources, and determination of resource quality objectives (RQOs), all processes legislated in the National Water Act (NWA) (Act 36 of 1998). The GRDM manual applies a water balance approach to define aquifer stress, and provides aquifer stress as a key parameter to determine the present class of a groundwater resource. The present class, in turn, is a key consideration in the future management class of the resource within the documented classification system for a groundwater resource. Although quantification of impacts of abstraction on surface water are discussed and appropriate quantification methods supplied, methods applied in the GRDM manual implicitly support a water balance approach and do not easily accommodate a capture principle approach in its entirety. Certain descriptions, approaches and methodologies would need to be updated for this. Nevertheless, the GRDM manual contains tools that can be applied for certain aspects of a capture principle-based assessment, such as a description of methodologies to determine recharge, and recommended model approaches for estimating surface-groundwater interaction.

The GRA II and, more specifically, the GRDM tool are related to implementation of the reserve, which in itself is a concept misaligned with the capture principle. As stated by Vegter, (2001): 'The question is not how much groundwater should be reserved for ecological purposes, but rather, how much groundwater may be captured from the natural system for man's use without causing unacceptable damage to the environment'. Any guideline or tool designed to support determination of the reserve will, to some extent, be misaligned with the capture principle. More significantly, the groundwater use authorisation and licensing process, which takes into account the reserve, is to some degree misaligned.

The **Guideline for the Assessment, Planning and Management of Groundwater Resources in South Africa** was released by the (then) Department of Water Affairs and Forestry (DWAF) in 2008 (DWAF, 2008b). The guideline describes the assessment, planning and management functions as they relate to groundwater use, outlining the appropriate approach to each function. This comprises listing the steps required in each function; for example, groundwater assessment includes: undertake desk study; undertake hydrocensus; carry out drilling/testing; develop a conceptual model; with a procedure provided for each step in the form of a checklist. Where the recommended step is 'reporting', a standard reporting template is provided. The guideline doesn't incorporate any methodologies that specifically promote a capture principle-based groundwater assessment, and in order to do so the following updates would be necessary:

- The reason for carrying out a groundwater assessment, in many cases, is to determine available yields for abstraction. Hence checklists for determination of sustainable aquifer yield, as a key step in groundwater assessment, should perhaps be incorporated.
- There is no explicit mention of determining the source of the abstracted water or, in other words, where the abstraction will cause impact (e.g. on surface water bodies, in discharge or recharge zone). The procedure for surface-groundwater interactions focusses on identifying springs that require protection and maintaining baseflow. To fully support a capture approach this procedure would need to be amended to rather determine the source of water/impact of abstraction, and link to procedures to determine whether this is acceptable.

However, neither does the guideline incorporate any methodologies that misalign with a capture principle-based groundwater assessment (i.e. water balance approaches), hence parts of the guideline can certainly be applied. Some elements of the guideline are superseded by, or further detailed within, the **Groundwater Management Functions** study (Riemann et al, 2011), which assessed the responsibility of various parties in local government towards the management of groundwater use. It mapped those elements necessary for comprehensive groundwater management (pollution prevention, licensing, wellfield operations and maintenance and groundwater assessment, amongst others), and integrated these with the mandated and legislative local municipal structures. The Groundwater Management Framework includes aquifer utilisation as one part of groundwater management, which is subdivided into groundwater assessment and wellfield operation and maintenance (Figure 3-1). These two sub-categories are not further defined as this was not the aim of the report. This work expands on the groundwater assessment box (as aquifer assessment, Figure 2-6), and the wellfield operation and maintenance box (as infrastructure characterisation and operating rules, Figure 2-6), and explores how these are linked and how they can be updated in adaptive management cycles to ensure sustainability. Although the resulting **Groundwater Management Framework** does not explicitly promote implementation of the capture approach (which was not its interest), a neat alignment with this framework is therefore possible.

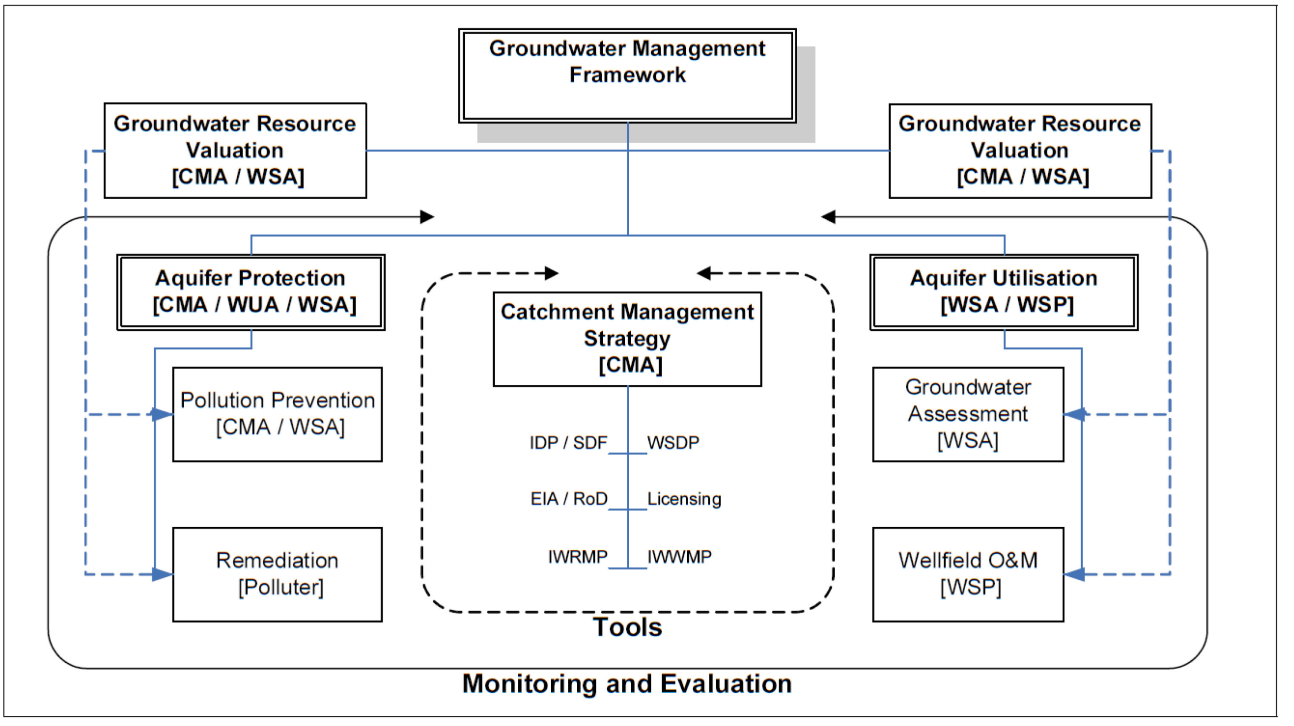


Figure 3-1 Groundwater management framework (from Riemann et al, 2011).

3.3. SELECTED EXISTING TOOLS

The **South African Groundwater Decision Tool** (Dennis et al, 2002b) aimed to guide groundwater resource managers in utilisation of groundwater. It includes information (tools, approaches), on determining aquifer parameters and borehole yields through pump tests, and on determination of transport parameters, and a fuzzy-logic based framework for groundwater risk assessment which 'takes into account the sustainability of a groundwater resource, the potential contamination of groundwater, human health risks and the impacts of changes in groundwater (quantity and quality) on aquatic ecosystems'.

Acknowledging that the capture principle dictates that abstracted water is sourced from some combination of depleted storage, or captured discharge, enhanced recharge, means that to follow a capture principle-based groundwater assessment, these elements must be estimated as to when, where and by how much they occur. These impacts are considered as *risks of abstraction* within the decision tool, with the stance being to determine *whether,* when and by how much they will occur, and to minimise their effects. The risk assessment incorporates:

- A 'sustainable risk assessment' which determines the risk of failure, or of unsustainable groundwater use. This is a set of user-friendly front-end screens that link to calculations for determining whether a borehole can maintain a particular pump rate, through application of analytical solutions such as the Cooper-Jacob equation. If appropriate for the aquifer and situation in question, this can be considered a tool for determining the maintainable borehole yield and maintainable aquifer yield.
- An ecological risk assessment for which the analysis method includes determining surface-groundwater interactions, determining the dependence of vegetation on groundwater, the depths of rooting systems compared to groundwater levels, and the amount of water required to sustain vegetation – dependent on baseflow estimates. The risk of ecological impact is linked to abstraction proportions compared to baseflow (i.e. it is unfavourable for abstraction to be greater than 0.3 x baseflow in rapid assessment resulting in a high risk proportion – with this level based on expert opinion). The ecological risk assessment does not cater for assessment of the source of water to be abstracted, and therefore quantify the effect on surface waters.

In summary, the language and overall approach of the decision tool doesn't fully align with a capture approach, and certain methodologies incorporate water balance approaches. Where elements of the capture principle-based assessment are covered by the tool (i.e. pump test interpretation to inform borehole yield and maintainable aquifer yield), it can be applied.

The 'aquifer firm yield' concept has been developed as part of the **Groundwater Planning Toolkit for the Karoo** by Murray et al (2012). The concept was developed because 'it was deemed necessary for groundwater resource assessments to be brought on par with that applied by surface water planners'. As an equivalent to the level of the dead storage volume in a dam, a maximum allowable drawdown is defined which is a user-specified management level, below which it is considered undesirable to draw down the water level. This approach has some connection to a capture principle-based assessment (impact on storage being one of the conditions of a future dynamic equilibrium to consider in a sustainability assessment), however it does not directly incorporate consideration of the impact of capture of discharge or enhanced recharge. Sustainable groundwater use has been linked solely to a groundwater level which is considered to cause negative impacts. The location of this groundwater level is (a minimum established for the entire aquifer, or near sensitive receptors only), and how it is to be determined (based on an assessment that does take into account the other aquifer conditions at this water level) is not clear. The specified level could relate to a capture of discharge to surface water which is considered unacceptable. The language and overall approach of the aquifer firm yield does not fully align with a capture approach.

The **Groundwater Planning Toolkit for the Karoo** incorporates the Cooper-Jacob Wellfield Model, a spreadsheet and GIS tool. Spatial data is read from GIS layers and linked to aquifer parameters entered in spreadsheet format, in order to calculate hydraulic interference, resulting drawdown and resulting wellfield yield for a specified maximum drawdown, through application of the Cooper-Jacob equation (Baker & Dennis, 2012). This model can be applied in a capture principle-based groundwater assessment for determination of wellfield yields, where the conditions of the aquifer sufficiently meet the assumptions of the Cooper-Jacob equation (Kruseman & de Ridder, 1991)

The primary investigation technique for estimation of the maintainable aquifer yield is pumping tests. Methods and guidelines for pump test interpretation are widely documented internationally (for example, Kruseman & de Ridder, 1991) and have also been tailored for South African contexts (for example Van Tonder et al, 2002 and Chiang and Riemann, 2001). The widely used Flow Characteristic **(FC)-Method** (published within Van Tonder et al, 2002), is an excel spreadsheet tool for the interpretation of pump test data and development of a borehole yield (also described in Box 2-2). A minimum water level is defined by the user (often level of pump inlet), and the FC-Method provides a borehole yield that maintains water levels above that level for the duration specified, (using various analytical equations). The borehole yield calculated has been termed the 'sustainable borehole yield', which is defined as the safe amount of water that can be abstracted from a borehole for a long time (usually one or two years), without the water level reaching the position of the pump or the main water strike (Van Tonder 2001, cited in Van Tonder et al, 2002). This was developed to enable a simple, easily applicable management rule to prevent borehole failure. The resulting yield depends on the position of the pump, which is generally limited by several infrastructure and budget factors rather than a level which represents some sustainable use of the aquifer. Basing the yield on a user-defined water level, and interpreting the outcome as a sustainable yield, assumes that any impacts associated with the abstraction of this yield are considered acceptable. They may not be. The FC-Method may be useful for definition of a (maintainable) borehole yield, but is not an appropriate tool to determine maintainable or sustainable aquifer yield. Nevertheless, several of the methods and guidelines in Van Tonder et al (2002) can be used for pump test interpretation to build information towards defining the maintainable aquifer yield.

Estimation of the conditions at the future dynamic equilibrium, as input to a sustainability decision, and update of this estimation based on operational results, generally requires analytical or numerical modelling. Several studies, e.g. DWA, (2010a), document the appropriate model tool for a particular aquifer setting or a particular problem at hand (i.e. high-level planning versus wellfield management). Guidelines are particularly well developed for discussion of models for quantification of surface-groundwater interactions, and a review of model methodologies for surface-groundwater interactions is incorporated into the GRA II and GRA III reports (DWAF, 2006a and DWA, 2009b respectively), and also features in the GRDM manual (Dennis et al, 2013).

3.4. SUMMARY OF GAPS FOR MAINSTREAMING THE CAPTURE PRINCIPLE APPROACH

Although several existing tools or guidelines use water balance methodologies for quantification of groundwater resources, these existing tools or guidelines also contain methodologies appropriate for parts of a capture principle-based groundwater assessment, including:

- Guidelines on groundwater exploration including necessary steps for a desktop study, and drilling/testing procedures.
- Several guidelines on pump test interpretation as input to the maintainable aquifer yield.
- Several guidelines on recharge quantification.
- Several guidelines on surface-groundwater interactions (quantification of natural aquifer discharge), the appropriate models or techniques for quantification and the impact of abstraction on discharge to surface water (stream flow reduction).
- Guidelines on the appropriate models or techniques for different stages of groundwater development.

Lacking in existing tools and guidelines are the following elements of a capture principle approach:

- How individual borehole yields, or information from pump tests, are related to or up-scaled to a maintainable aquifer yield.
- Specific discussion of continued recharge measurement in order to quantify enhanced recharge due to abstraction.
- Acknowledgement that each of the elements required for a capture principle-based groundwater assessment are to be updated after an initial estimate, i.e. guidance on implementing adaptive management approaches.
- Guidance on or specific discussion of the process by which the estimated conditions at future dynamic equilibrium and the impacts of abstraction are considered acceptable or not (hence reflect sustainable groundwater use), and the required hydrogeological input to this decision. This 'sustainability decision' is what the water resources classification system intends to cater for; however, their methodologies are not aligned with a capture approach.

Cognisant of these gaps, the following were selected for further exploration in this project and are discussed in section 4 and section 5 respectively:

1) Adaptive management remains a challenge. Other researchers have recognised the need for adaptive management guidelines (Knüppe, 2011, Seward et al, 2006, and Riemann et al, 2011). Table 2-2 suggests operating rule confidence levels and related grades can be used to illustrate adaptive management cycles and the number of iterations completed of the operating rule procedure (Figure 2-6). However, the specific analyses required to progress through the confidence levels and support adaptive management require definition.
2) Guidance on the required hydrogeological input to determine whether the predicted conditions at future dynamic equilibrium (or the impacts of abstraction) are acceptable or not, and therefore can be considered sustainable groundwater use.

Furthermore, the mainstreaming assessment highlights that the groundwater use authorisation and licensing process, which takes into account the reserve, is to some degree misaligned with the capture principle (through, for example, recommending the use of a stress index). As reserve determination and classification studies are generally carried out at regional scale (in addition to aquifer scale for particular licence applications), this misalignment is also compounded by the prevalent application of water balance approaches at regional to national scale. Within a project of at national or regional scale, it is often deemed impractical to assess groundwater resources at the aquifer scale, necessary for a capture principle-based assessment, (primarily due to project budget and timescale). Regional to national scale assessments and legislation are not directly addressed in this report, but are discussed further in section 10.

4. SUSTAINABILITY INDICATORS TO SUPPORT ADAPTIVE MANAGEMENT

4.1. PURPOSE OF PROPOSED SUSTAINABILITY INDICATORS

Implementing the capture principle approach to sustainability requires an initial estimate of the future dynamic equilibrium, and requires a decision to be taken based on the conditions of the future dynamic equilibrium. However, this future dynamic equilibrium cannot be predefined with high confidence, hence implementing the capture principle approach to sustainability requires an adaptive process in which management decisions are adjusted as necessary. Monitoring designed specifically to provide information on the estimated future conditions on which the sustainability decision was taken is required to support the adaptive process, and enable updating of the estimate of future dynamic equilibrium and hence updating of the operating rule confidence level.

Sustainability indicators are therefore proposed as a tool to meet this monitoring requirement. The indicators must be capable of:

1. Monitoring or calculating the changes in the aquifer water balance or dynamic equilibrium, and comparing these to initial estimations:
    a. changes in the pre-abstraction aquifer discharge
    b. changes in the pre-abstraction aquifer recharge
    c. changes in the pre-abstraction aquifer storage
2. Monitoring or calculating any other observed changes in aquifer conditions, and comparing these to initial estimations
3. Monitoring or calculating progress towards a new dynamic equilibrium and updating the initial estimation of response time (i.e. position on x-axis of a transition curve, Figure 2-3).

The definition of sustainability applied in this study relates to the social, environmental and economic acceptability of the predicted environmental impacts of the resource use, and the indicators relate to updating the environmental impact prediction. Human capacity, institutional arrangements and instruments to support human capacity in carrying out the necessary activities towards sustainable groundwater use, will also impact the sustainability of groundwater use. Indicators for groundwater sustainability that consider human capacity, institutional arrangements and knowledge bases have been developed elsewhere (Pandey et al, 2011). These capacity and institutional aspects are not considered here.

The indicators also do not consider the affordability of the groundwater scheme, i.e. pumping, maintenance and infrastructure costs are not considered, nor do the indicators consider social aspects such as public education in groundwater or allocation decisions and restrictions. Steinmann et al (2007) developed groundwater indicators to support basin-wide groundwater management that include these aspects.

Selected existing indicators are assessed in section 4.2 to determine whether existing indicators can meet the above-listed requirements, followed by proposed indicators in section 4.3.

4.2. SELECTED EXISTING GROUNDWATER INDICATORS

4.2.1. UNESCO-IHP Groundwater Resource Sustainability Indicators

A groundwater indicators working group was established that included the International Hydrological Programme (IHP) of UNESCO, the International Atomic Energy Agency (IAEA) the International Association of Hydrogeologists (IAH) and a group of select professionals. This working group developed a set of groundwater indicators, described as 'Groundwater Resources Sustainability Indicators', (Vrba et al, 2007, referred to within this section as 'the report').

Several UN meetings had been held on indicator development prior to and as part of this UNESCO effort, the outcome of which was an agreement that a 'balanced scientific and policy-based approach should be employed in deriving groundwater indicators'. Useful insights are that a 'limited set of quality indicators with excellent data backup' is more meaningful than ‘a large number of lesser quality indicators’ and as such data availability must play a role in selecting appropriate indicators. The report describes the main functions of indicators (in general) to be:

- To simplify, quantify and communicate information in a way that allows for comparison between different countries and regions and different aspects.
- To provide information in an understandable way and therefore act as an important communication tool for policy-makers, managers and the public.
- To evaluate the effect of policy actions and plans and help to develop new actions
- To help to translate information needs into data that has to be collected, and to translate collected data into policy relevant information.
- A description of the state of the resource, and, if measured over time, may provide information on the functioning of the system or its response to management.
- For predicting the future; ‘When models are linked to indicators, a time series can be extended into an estimated future. Alternative scenarios can be assessed in terms of how well each one moves towards a desired state’ (Vrba et al, 2007, p. 1).

These generic functions of indicators are then related to the required function of groundwater indicators, stating that groundwater indicators ‘support sustainable management of groundwater resources, provide summary information about the present state and trends in groundwater systems, help to analyse the extent of natural processes and human impacts on groundwater systems in space and time and facilitate communication and public participation in resource planning and policy and indicators generation’ (Vrba et al, 2007, p. 6).

Although the goals of the UNESCO groundwater resources indicators are not explicitly stated, it is inferred that they were developed with the intention of meeting each of these functions, and specific reference is made to quantifying and communicating groundwater resources information in a way that allows for comparison between different countries and regions and different aspects. These indicators have been applied at country scale in the case studies within the report, although the text states that groundwater indicators have been proposed for application at global, national or aquifer levels.

The intended purpose of these proposed indicators, as compared to the goals of the UNESCO indicators, is a more targeted one, which is to track specific information rather than paint a picture of the state of a resource. Nevertheless, the UNESCO indicators may also be able to meet some requirements of these proposed indicators. The ten UNESCO indicators are summarised in Table 4-1, with a comment on their relevance, i.e. their ability to meet the requirements of the proposed indicators.

Table 4-1 Summary of UNESCO –IHP groundwater resources sustainability indicators

Description	Relevance
UNESCO-IHP Indicator 2.3.1: **Renewable groundwater resources per capita**	
The indicator expresses the total annual amount of renewable groundwater resources (m^3 per year) per capita at national or regional level, suggested as an important factor for the social and economic development of a country. The renewable groundwater resources are defined and calculated as: ➢ natural recharge within the geographic boundary of the country, ➢ plus the total rate of groundwater inflow from the neighbouring countries minus that leaving to the neighbouring countries, ➢ plus seepage from surface water bodies (streams, lakes) to groundwater, minus discharge to surface water through springs and base flow, ➢ plus artificial recharge where relevant. Suggestions are provided on how each of the above can be quantified or collated from existing datasets.	If applied at aquifer scale, this indicator is useful in describing the ability of the aquifer to meet the water demand of the area indicating its significance for water supply. However, even if the groundwater resource per capita is relatively low, groundwater may still be an essential part of a water supply system by providing peak season demand or drought resilience. This indicator is not directly relevant for capture-based sustainability indicators.
UNESCO-IHP Indicator 2.3.2: **Total groundwater abstraction/groundwater recharge**	
The indicator expresses: $\dfrac{\textit{total groundwater abstraction}}{\textit{groundwater recharge}} \times 100$ Definitions are provided for total groundwater abstraction and groundwater recharge, along with suggestions of how each of the above can be quantified or collated from existing datasets. Acknowledging the challenges of data scarcity, three scenarios are proposed for interpreting and giving significance to the estimated indicator values: 1. Scenario 1: abstraction ≤ recharge, i.e. < 90% 2. Scenario 2: abstraction = recharge, i.e. = 100% 3. Scenario 3: abstraction > recharge, i.e. > 100% It is implicitly suggested that (local variability aside) abstraction in a region with this indicator falling within scenario 1 and 2 is sustainable, and scenario 3 represents over-abstraction	This indicator describes use compared to recharge. The indicator is similar to the aquifer stress indices used in South Africa. The indicator can indicate relative use of an aquifer (low or high compared to recharge), but relating the outcome of this indicator is to sustainability in any way implicitly applies the water balance approach, and is contrary to the capture approach. This indicator cannot contribute to the requirements of capture-based sustainability indicators.
UNESCO-IHP Indicator 2.3.3: **Total groundwater abstraction/exploitable groundwater resources**	
The term 'exploitable groundwater resources' is defined as the amount of water that can be abstracted annually from a given aquifer under prevailing economic, technological and institutional constraints and environmental conditions. It is noted that the exact meaning of exploitable groundwater resources (and whether and how it includes considerations such as ecological reserves) may vary from one country to another. Three scenarios are again proposed for the estimated indicator values: 1. Scenario 1: abstraction < exploitable amount, i.e. < 90% 2. Scenario 2: abstraction ≈ exploitable amount, i.e. ≈ 100% 3. Scenario 3: abstraction > exploitable amount, i.e. > 100% Scenario 1 describes a country with underdeveloped groundwater resources, probably with potential for further development. Scenario 2 is likely to be a country with groundwater resources developed, and probably an understanding and appreciation of sustainability aspects in water resource management. Scenario 3 depicts the situation in a country with overexploited groundwater resources and the resulting stress needs to be addressed in managing water resources.	The potential applicability of this indicator revolves around the definition of 'exploitable groundwater resources'. If the definition is based on water balance approaches as is suggested, then as per indicator 2.3.2, it is not meaningful to relate the outcome of this indicator to sustainability in any way and this indicator cannot contribute to the requirements of capture-based sustainability indicators.

It is suggested that 'Indicators 2.3.2 and 2.3.3 should be used jointly to reflect the status of resource development from a given groundwater system from a ***water balance*** perspective' (Vrba et al, 2007, p. 12).	
UNESCO-IHP Indicator 2.3.4: **Groundwater as a percentage of total use of drinking water at national level**	
The indicator is intended to illustrate the dependency on groundwater as a drinking water source. It is suggested that the indicator be expressed as a percentage of groundwater contribution to the total drinking water supply.	Potentially useful to indicate the importance of the aquifer in question, and subsequent resource quality objective or management class. However the indictor is unrelated to monitoring aspects of the capture principle sustainability concept (i.e. information on the estimated future conditions), so cannot contribute to the requirements of capture-based sustainability indicators.
UNESCO-IHP Indicator 2.3.5: **Groundwater depletion**	
It is acknowledged that any groundwater exploitation leads to water level declines and affects groundwater storage. The stated issue for consideration is how much water can be withdrawn from an aquifer without producing non-reversible impacts on groundwater quantity and quality, ecosystems or surface geotechnical stability, hence the indicator is intended to express 'excessive groundwater withdrawal'. It is recognised that groundwater storage depletion 'may also be associated with a long transient evolution from one steady-state to another, and does not necessarily represent a problem of unsustainable exploitation of an aquifer' and that 'the most difficult problem in aquifers that are subjected to exploitation is to distinguish permanent and regional depletions from only temporal and local interferences caused by the proximity of production wells' (Vrba et al, 2007, p. 14). Suggestions are provided for how to detect a groundwater depletion problem related to excessive withdrawal, including (among others): • changes in baseflow to surface water bodies. • detection of land subsidence which can be used as an indirect indicator of unsustainable groundwater exploitation. • Given that drastic changes in groundwater quality are not expected under abstraction, changes in age and origin of groundwater at specific locations in the aquifer can be an indication of groundwater depletion. Using these methods to detect groundwater depletion problems, the indicator is quantified as: $\dfrac{\sum \text{areas with a groundwater depletion problem}}{\text{Total area studied}} \times 100$	Some elements in the description of this indicator are aligned with the capture principle, such as the recognition that all groundwater developments lead to an impact on storage, and the acknowledgement that this doesn't necessarily represent unsustainable exploitation as an aquifer may be evolving to a new steady-state. That an impact on storage may be non-reversible is not considered problematic in the capture principle approach to sustainability. Abstraction may cease and recharge may not be capable of overcoming the storage deficit. If this change is considered socially, environmentally and economically acceptable then the groundwater use is considered sustainable. 'Permanent depletions' are therefore not a direct indication of unsustainability. However, the predicted change in storage should be part of a sustainability assessment and as such should be monitored through the use of indicators, and distinguished from seasonal and cyclical (drought) depletion. In suggesting monitoring changes in baseflow to detect groundwater depletion, there is an implicit assumption that a detected impact on baseflow indicates 'excessive withdrawal'. It does not. As per the capture principle definition, all abstraction will eventually derive its water from reduced discharge and/or enhanced recharge. Baseflow should however be monitored to assess and update the predicted impact. Where abstraction has caused subsidence due to the depressurisation of clay aquitard layers, or the dissolution of dolomites, this is unlikely to be considered socially, environmentally and economically acceptable, and so is considered a useful indicator to consider.

	As the origin of water is expected to change as abstraction continues (from stored water around the well to discharge or enhanced recharge), the water chemistry (or specific parameters) may change (unless the water has sufficient time to equilibrate with the aquifer material). Water chemistry characteristics such as isotopic signature and age of water can assist in detecting this source change, and are relevant for consideration.
UNESCO-IHP Indicator 2.3.6: **Total exploitable non-renewable groundwater resources/annual abstraction of non-renewable groundwater resources**	
The indicator expresses: $\frac{\textit{total exploitable non-renewable groundwater resource } (m^3)}{\textit{annual abstraction of non-renewable groundwater resource } (m^3/a)}$ The indicator is essentially an adaptation of indicator 2.3.3, total exploitable groundwater resources/annual abstraction of groundwater resources, for non-renewable groundwater resources. It is recommended that this indicator be plotted as a trend-line over time, and can also therefore be used to estimate the total lifetime of a non-renewable aquifer. Two main criteria are listed to define non-renewable groundwater resources: (a) mean annual recharge should be less than 0.1% of the stored volume; (b) exploitation of the groundwater concerned should not have a significant impact on neighbouring renewable systems or recharged groundwater bodies.	The general assumption with non-renewable groundwater resources is that the aquifer will be mined, i.e. water will always be sourced from storage given that pre-abstraction recharge (and therefore discharge) is negligible. Continued groundwater mining may still be considered socially, environmentally and economically acceptable, and hence represent sustainable groundwater use. The capture principle (and the associated assessments such as determining source of abstracted water) is still applicable in a non-renewable groundwater resource hence this situation is unlikely to need specific sustainability indicators.
UNESCO-IHP Indicator 2.3.7: **Groundwater vulnerability**	
The formulation of groundwater vulnerability indicators is intended to support groundwater protection policies by giving guidance on sound land-use planning and sustainable managerial purposes. It is also intended to create public awareness about groundwater protection. The indicator is quantified as: $\frac{\sum \textit{areas with different classes of groundwater vulnerability}}{\textit{Total area studied}} \times 100$ Groundwater vulnerability is defined for this indicator solely as a function of hydrogeological factors – the characteristics of an aquifer and the overlying soil and unsaturated geological material. The indicator uses the following parameters to define aquifers with low, moderate and high vulnerability aquifers: soil properties, lithology of the unsaturated zone, and thickness of the unsaturated zone.	Although potentially useful to indicate the vulnerability of an aquifer to pollution, the indictor is unrelated to monitoring aspects of the capture principle sustainability concept (i.e. information on the estimated future conditions), so cannot contribute in this form to the requirements of the proposed indicators.
UNESCO-IHP Indicator 2.3.8: **Groundwater quality**	
The indicator is intended to provide information about the present status and trends in groundwater quality and help to analyse and visualise groundwater quality problems in space and time. The indicator for naturally occurring quality problems is defined by the relationship:	The indicator is targeted at quantifying or mapping areas of varying groundwater quality (either natural or contamination related), to indicate the water quality status. The indicator is unrelated to monitoring aspects of the capture principle sustainability concept (i.e. information

∑ areas with natural groundwater quality problem x 100 *Total area studied* It can be implemented in relation to drinking water standards, food processing, irrigation requirements, industrial use and others. The area of aquifer that presents natural groundwater quality problems is defined as the sum of those parts of the aquifer in which the concentration of the indicator parameter exceeds the maximum level specified in the standards being used. The indicator for groundwater under human stress is established based on the relationship: *∑ areas with increment of concentration for specific variable x 100* *Total area studied* The total area where an increment of concentration for a specific variable was detected is defined by the sum of all areas where an increase in concentration of the variable was detected during the observation period.	on the estimated future conditions), so cannot contribute to the requirements of the proposed indicators.
UNESCO-IHP Indicator 2.3.9: **Groundwater treatment requirements**	
This indicator describes whether groundwater can feasibly be made potable (drinking water), or usable for other purposes (e.g. agricultural water, industrial water, cooling water) with treatment.	The indictor is unrelated to monitoring aspects of the capture principle sustainability concept (i.e. information on the estimated future conditions), so cannot contribute to the requirements of the proposed indicators.
UNESCO-IHP Indicator 2.3.10: **Dependence of agricultural population on groundwater**	
The indicator is defined by the relationship: *No. of farmers dependent on groundwater for agriculture activities x 100* *Total population of the country* It is recognised that rapid groundwater development has made a great contribution to the objectives of food security and improved rural livelihoods. The indicator is therefore designed to signify the importance of groundwater in rural livelihoods and household incomes. It provides the percentage of a country's population that depends on groundwater for supporting livelihoods and household income.	Although valuable as an indicator of the importance of groundwater in rural livelihoods and household incomes (as well as for commercial farmers), this indicator is unrelated to monitoring of sustainability aspects as defined here, so cannot contribute to the proposed indicators.

As part of the UNESCO-IHP publication of groundwater resource sustainability indicators, several country-wide case studies were carried out to test implementation of the indicators. South Africa was one of these case studies, contributed by the then DWAF. The first three indicators in Table 4-1 were assessed through application of GRA II data.

4.2.2. Parsons and Wentzel Indicators to Classify Groundwater in South Africa

Parsons and Wentzel (2006) proposed indicators to classify groundwater in South Africa. Their aim was that the indicators be understandable, clear, simple and unambiguous, conceptually well founded, limited in number and dependent on data that was readily available, or available at a reasonable cost. The authors suggest that groundwater sustainability indicators include sinkhole formation, saline intrusion, decrease in river or spring flow and vegetation die-off. They suggest that should any of these conditions be observed, a management intervention modify the abstraction to 'within sustainable limits'. Due to the (for all practical purposes) irreversibility of saline intrusion, and the risks associated with sinkhole formation, it is supported that observation of these impacts should relate directly to unsustainability and amending abstraction. However, observation alone of a reduction in river or spring flow, and the associated vegetation die-off, cannot be related directly to extraction being unsustainable, given that the capture principle dictates that all abstraction will have an impact on pre-abstraction natural discharge and recharge.

4.2.3. GEUS Exploitable Groundwater Indicators

Henriksen et al (2007) of the Geological Survey of Denmark and Greenland (GEUS), used what they refer to as 'ensemble indicators' to assess the (sustainably) exploitable groundwater resources of Denmark. The paper is aligned with much of the sustainable pump rate and capture principle literature as this study, and numerical modelling is motivated as a necessary tool for estimating elements of the future dynamic equilibrium. Based on large scale national modelling and other analyses, the authors then propose indicators, each with thresholds expressed as percent (therein termed criteria), for allowable changes from current aquifer flow balance to future dynamic equilibrium. For example, indicator 4 is reduced baseflow which has a threshold of up to 50% of pre-abstraction baseflow, based on the ecological requirements of the river. The indicators have several aims, referred to throughout the paper, including:

- To guide, enable, quantify, and provide normality in model outputs, for the decision over sustainability.
- To manage indirect water quality risks and adverse impacts on aquatic ecosystems coming from intensive abstraction, with numerical flow models as the tool.
- To overcome a common misperception which has its basis in the water balance approach, which states that the development of a groundwater system is 'safe' if the average annual rate of groundwater withdrawn does not exceed the average annual rate of natural recharge.
- European legislation makes provision for environmental flows to be met. The paper intends to provide a tool to quantify and assess the influence of abstraction on environmental flows (i.e. through numerical modelling), but also then to categorise this into acceptable ranges, in a tool that is based on robust science, with a transparent approach useful for communication with stakeholders.

Indicator 1 is decreased water level and indicator 2 is increased recharge. The thresholds for indicators 1 and 2 have been developed based on an analysis of water quality data, abstraction rates compared to pre-abstraction recharge, and abstraction rates compared to current (i.e. induced) recharge (a modelled result) in several basins. The modelling software and approach used (national models separated into regional domains, 1 km x 1 km grids, using MIKE-SHE software), enabled the model to quantify impacts such as enhanced recharge, but does not allow for estimation of water level impacts around abstraction points. For indicator 1, an abstraction/pre-abstraction recharge ratio is used as a proxy for decreased water level, yet the theoretical motivation for this is not clear. The analysis showed that with ratios above 35% for abstraction/pre-abstraction recharge, and above 30% for abstraction/current (i.e. induced) recharge (a modelled result), each of these basins had some evidence of water quality impacts. The observed water quality impacts include sulphate and chloride, pesticides and organic micro-pollutants. The source of the water quality impacts is not described explicitly, but a general reference is made to the fact that lowering the groundwater table and associated transformation from anaerobic to aerobic conditions causes increased release of toxic solutes from aquifer sediments. It is likely that the groundwater abstractions sustained farming, which in turn adds these pollutants to the system. However, the lowering of the water table also enables increased retardation in the unsaturated zone, which isn't discussed. In a confined aquifer setting the authors suggest that abstraction-induced increased recharge will allow pollutants such as nitrate and pesticides located in the upper soil layers to move faster towards the deeper aquifers. The authors therefore propose that these levels (35% and 30% for indicator 1 and 2 respectively) are set as maximum thresholds for sustainable abstraction given that below these levels no water quality impacts that could be related to abstraction were observed.

The findings of the assessment implicitly suggest that these levels of 35% and 30% reflect a natural constraint of the system, above which negative impacts are noted. One would have expected environmental impact criteria and thresholds to be very case or aquifer specific, and this result suggests the basins must all be of similar terrain and hydrogeology. The authors do note that the thresholds are based on data from one area in Denmark, and that it is not known as to whether these findings (the 35% and 30%) can also apply to hydrogeological conditions in other parts of the country.

The threshold set for indicator 4 (depletion of low flows) adopt the levels established for environmental flows in legislation. In Danish and European legislation rivers are classified according to ecological status (Class A to Class F, similarly to South Africa), and in addition, the reduction in low flow corresponding to each class is established. It is therefore fairly straightforward to establish thresholds for the model output for reduced flow. In South Africa, the corresponding reduction in low flow for each class is not standardised, and is to be determined by an ecologist on a river by river basis. The authors do note that 'there is a strong need for new and better estimates of threshold values for critical streamflow depletion for both average flow and low flow conditions linked to ecological parameters, e.g. using habitat models' (Henriksen et al, 2007, p. 235).

Indicator 3 is reduction of runoff (by enhanced recharge), although it should rather be reduction of interflow as groundwater abstractions are unlikely to significantly influence infiltration capacity. The threshold for the indicator is set through 'the translation of the abstraction-runoff balancing principle' which showed that a 10% reduction of the average accumulated river runoff from the entire catchment was acceptable (Henriksen et al, 2007, p. 234).

The national model is used to derive an abstraction rate that meets the threshold for each indicator (reduction or increase in elements of the aquifer flow regime), and these abstraction rates are translated into the exploitable groundwater resources. The entire research work, establishment of indicators and setting of thresholds, was a participatory process with review by a wide range of experts, and liaison in an open forum of invited water managers, researchers, stakeholders and the project group. There is discussion over the future implementation of the indicators at different scales, and an acknowledgement that a weakness in the approach is that it is designed for a specific scale (300–2000 km^2). It is suggested that use of the same approach on smaller scales would require re-assessment of the thresholds for the four indicators. In addition, at aquifer management scale (for management of licences and optimisation of abstractions in an area), the use of more specific and physically based indicators, and the use of a model that can define variations in water level, is suggested.

There are strong similarities in theoretical foundation between the GEUS work and the proposed indicator work. However, the orientation of the GEUS indicators is towards defining thresholds for sustainable groundwater use, rather than monitoring predicted aquifer impacts and evolution towards the future dynamic equilibrium, as per the proposed indicators. Referring to the flow chart of elements making up sustainable groundwater use (Figure 6-1), the GEUS methodology can be a tool to support the socio-economic-environmental decision, but taking into account environmental costs only. This is discussed further in section 5.

4.2.4. Overstrand Municipality Sustainability Indicators

Indicators have been used to monitor and assess the sustainability of groundwater abstraction for domestic water supply in the Overstrand Municipality, in the Western Cape of South Africa (Umvoto Africa, 2011). The need for monitoring and an adaptive management approach for groundwater use is well entrenched in the operation of this wellfield, with a 6-monthly compilation of abstraction monitoring results to assess aquifer performance and sustainability, and update operating rules if necessary. The report highlights that the ideal approach would be to use numerical modelling to predict the future response of the aquifer to abstraction, monitor responses, and update the model based on results, and update management practices based on actual and predicted results. In the absence of available numerical modelling results, the authors use various analytical assessments (termed indicators) to 'broadly address the question of the available yield and the sustainability of the abstraction' and 'to determine whether yields that have been applied to date are sustainable' (Umvoto Africa, 2011, p. 44-45). There is no formal definition of sustainability provided, and it appears to refer both to pumping yields that are maintainable, and to the acceptability of the predicted or monitored impacts on aquifer mass balance elements.

The wellfield targets a deep, confined, fractured-rock aquifer with offshore subterranean discharge being the only known discharge, and no other known groundwater users drawing from the same aquifer. The aquifer is recharged where it outcrops in an ecologically sensitive mountain catchment, with some perennial and presumed groundwater-fed springs. The indicators and assessments carried out are listed in Table 4-2, with their potential relevance.

Table 4-2 Summary of Overstrand Municipality sustainability indicators

Description	Relevance
1. Hydraulic property estimations	
The hydraulic properties derived from an initial constant-rate pump test on one wellfield borehole (several days in duration) were compared to hydraulic properties derived from an operational pumping session on the same individual borehole of three months in duration. The derived transmissivity reduces between these tests from an initial 200 m^2/day to 80 m^2/day. The reduction is attributed to a larger radius of influence generated in the longer test (as similar yields were pumped) and a larger proportion of lower transmissivity aquifer material encountered, hence lowering the overall transmissivity.	The detection of varying hydraulic properties in itself isn't a direct indicator of any of the capture principle-based elements of sustainability. However, if there is an absence of direct measurement of drawdown across the aquifer (i.e. a lack of monitoring boreholes), repeating this analysis until hydraulic properties stabilise could be used to indicate that the radius of influence has stabilised in growth. This occurs when the cone of depression intercepts enough flow in the aquifer to meet the pump rate through induced recharge, or natural infiltration to the aquifer across the volume influenced. Such a measurement could also serve as an indirect assessment of response time – and whether the new dynamic equilibrium has been reached (stabilised hydraulic properties) or not (hydraulic properties continuing to change). The challenge is that in an operational wellfield, data that easily fits this analytical analysis is scarce. In addition, changing hydraulic properties can be clouded by reduced well efficiency through clogging or siltation.
2. Drawdown character	
The drawdown character of successive pump sessions over time is assessed to detect any changes in aquifer behaviour. The curves continue to show confined aquifer behaviour and do not, for example, encounter a barrier boundary at any point which would indicate abstraction in excess of the maintainable aquifer yield. Under specific conditions (combinations of pumped holes and at particular pump rates), recharge boundaries are noted in two of the four wellfield boreholes.	The indicator can provide useful information on capture principle-based elements of sustainability. For example, if a recharge boundary is reached and maintained, this can indicate that the new dynamic equilibrium has been reached in which water is derived from enhanced recharge.
3. Recovery assessment	
Whether the water levels recovered or not (or can be projected to recover using analytical analyses) during times when the wellfield has not been used, has been taken as an indication that the yield pumped prior to the rest period is sustainable [reflects a maintainable aquifer yield].	If an aquifer reaches new dynamic equilibrium, with associated reduction in storage, and pumping then ceases, the aquifer will go through another transition curve to equilibrate with the lack of pumping, and over this time, storage will be replenished if all other factors return to previous. Depletion of aquifer storage is certain, even with sustainable groundwater use. Hence lack of recovery is not an indication of abstraction being unsustainable groundwater use, using the capture principle-based elements of sustainability.
4. Assessment of impact	
The available hydrogeochemical monitoring results are used to build an understanding of the impact and source of abstracted water, and the aquifer's progress towards a new equilibrium:	The applied approaches can provide useful information on capture principle-based elements of sustainability and are relevant for consideration:

• The lack of water level impact at two far-field upstream monitoring boreholes is used as an indication that the radius of influence has not yet reached these boreholes and the recharge zone beyond. An analytical projection of the speed of propagation of the radius of influence suggests that it is likely to reach the recharge zone in another 5–8 years of continual pumping. • The water abstracted at the wellfield is dated with Carbon-14 isotopes to be 9000 years old, suggesting its origin in stored water rather than fresh recharge water. • These two observations are interpreted as an indication that the source of abstracted water is stored water, and the aquifer has not reached a new dynamic equilibrium. • A drawdown/distance relationship is also used to project the radius of influence, suggesting it should have already reached the recharge zone. Departure from ideal conditions in analytical analyses is suggested as a reason for the different results (i.e. aquifer heterogeneity, recharge and regional groundwater flow characteristics). • The position of the downstream end of the radius of influence is also discussed, and indicates that natural discharge to the ocean has been reduced, although this has not been quantified.	• Given the challenges in applying analytical approaches (often limited by their simplistic assumptions) to real-world situations, a combination of analytical calculations and numerical model assessments would be recommended to develop projections of impact and timing of impact. • If carried out routinely, carbon-14 dating could detect a reduction in the age of the pumped water, as the source transitions from stored water to enhanced recharge. If the age stabilises, this may be a direct indication that the aquifer has reached dynamic equilibrium. • Aquifer water levels should be interpreted to determine the position of the radius of influence, and compared to that which was projected. If sufficient monitoring boreholes close to the discharge point are available, the reduction of aquifer discharge could be calculated.
5. Saline intrusion	
Subterranean oceanic discharge is the only (known) discharge of the coastal aquifer targeted, and therefore saline intrusion is a risk. Water level cut off points are in place to prevent saline intrusion through upconing. Their robustness is tested using density dependent numerical modelling (and results summarised in Umvoto Africa, 2011). The cone of depression in the downstream direction towards the offshore discharge is assessed to ensure a hydraulic gradient towards the ocean remains.	Where saline intrusion is a risk, the reduction in discharge to the ocean will have to be carefully quantified and the impact of this reduction taken into account in determining the acceptability of abstraction. The same is true for an aquifer discharging to ecologically sensitive surface water. In both cases, monitoring is required to determine if the reduction in discharge is in line with predictions, and update assessments. Specific indicators for saline intrusion therefore do not appear necessary.
6. Isotopic signature of pumped water	
Oxygen and hydrogen isotope composition of groundwater can be used to indicate providence of a sample (i.e. recharge characteristics). Oxygen and hydrogen isotopes were analysed routinely for several years, in order to determine the source of water in certain boreholes for which no geological information was available.	The long-term analysis of isotopes has direct relevance to the proposed indicators, as it could illustrate (seasonal variation aside) a change in providence of pumped water from stored water (with one recharge source) to enhanced recharge (with perhaps a different recharge source).
7. Recharge zone monitoring	
Monitoring is targeted to detect potential impact in the ecologically sensitive recharge zone. The monitoring is three-tiered in order to detect ecological impact in stages, before loss of habitat may occur (Umvoto Africa, 2011). The approaches include: • Tier 1: Water levels are monitored in the confined aquifer upstream of the wellfield, and in the unconfined aquifer within the recharge zone, as groundwater levels would reduce before causing surface impacts. • Tier 2: The flow rate in the groundwater-fed springs is monitored, as, if the groundwater levels reduce in the recharge zone, discharge to springs would decrease. This is enhanced with chemical monitoring in the springs to determine groundwater contribution to flow. In this case, dissolved organic carbon is used as an indicator of surface water contribution, and silica content as an indicator of groundwater. Data from local weather stations and regional remote sensing analysis would be used to distinguish between climate-induced and abstraction-induced impacts.	Each of these approaches is directly relevant to indicators for monitoring the predicted impact on recharge zone, or change of recharge.

• Tier 3: Botanical monitoring is conducted in the ecosystems around the groundwater-fed springs. As no impact is yet noted at tier 1 or 2, this monitoring currently serves as a baseline.	

4.3. PROPOSED SUSTAINABILITY INDICATORS

Through conceptualising means to monitor the elements considered in a capture principle-based sustainability assessment, and using insights from section 4.2, proposed indicators are provided in Table 4-3. Indicators must be able to monitor impacts on storage, recharge and discharge, and monitor aquifer conditions at or towards the new dynamic equilibrium. Furthermore, the indicators must enable updating of the original predictions of each of these.

The indicators are therefore predicated on the assumption that sustainable groundwater use has been determined using the recommended process (Figure 2-6); i.e. an initial estimate of the conditions at the future dynamic equilibrium has been established, a social-environmental-economic decision has been taken based on the conditions of the future dynamic equilibrium, and abstraction has proceeded based on this. In most cases numerical modelling is required to estimate the conditions at the future dynamic equilibrium. As the indicators will monitor these sustainability aspects to see if the aquifer is progressing as predicted (given that a management decision was taken based on a prediction), the results of indicators can therefore provide model verification data, enabling updating of the prediction.

The indicators are necessarily generic and would need to be tailored to suit each aquifer and environmental setting and each sustainability decision, selecting those most appropriate and adapting the principles where necessary. Some of these necessary adaptations or applications are discussed in Table 4-3 ('discussion' column). A robust monitoring programme is pre-requisite to implementing these indicators, including a well-distributed network of groundwater level and hydrochemistry monitoring points, weather stations for atmospheric conditions, monitoring of surface water flow where applicable, ecological monitoring where applicable, and monitoring of flow and water level at the wellfield. Although generic, the indicator list may be a useful framework or approach to design, inform and update the required monitoring network towards elements of a sustainability assessment.

There are several indicators for the same purpose, because, for example, several methods are available to detect a change in natural discharge. Given that there is generally a hydraulic progression of impact, and thus a progression in the detection of impact, these multiple indicators fall naturally into various tiers related to timing of detection (tier column in Table 4-3). For the detection of change in natural discharge, the first tier indicator would be to detect a change in water levels (and therefore gradients) towards the discharge point, as the impact propagates away from the abstraction point. The second tier indicator would be to use this information to calculate the change in discharge. The third tier indicator would be to directly monitor the receiving environment to detect this reduction in discharge. A fourth tier is available, which is an indirect monitoring of the groundwater contribution to the receiving environment.

The tiers can also correspond to confidence levels. An aquifer monitored only by first tier indicators (as is common) provides lower confidence information on the capture principle elements of sustainability than if monitoring by all tiers is employed. As the sustainability indicators are envisaged as a means to enable the update the estimate of future dynamic equilibrium and hence update the operating rules, so too does the operating rule confidence level increase as the various tiers of sustainability indicators are implemented. Each indicator (or, at least, tier 1 in each purpose category) should be assessed routinely, perhaps 6-monthly in high risk environments, or annually once aquifer impacts are well understood.

There is deliberately no acceptable level or threshold set for each of the indicators. The value or level is case specific, and is determined through assessment of the acceptability of the future dynamic equilibrium. For example, in an ecologically sensitive surface environment, an ecologist may have determined that aquifer discharge to surface can only reduce by 30%, hence this would be the criteria against which the indicators for change in natural discharge are assessed for this particular case (discussed further in section 5).

Table 4-3 Indicators for monitoring capture principle-based sustainability and response time

Purpose	Tier	Indicator	Description and measurement	Discussion
Change in natural discharge	1	Detection of change in groundwater level towards discharge point	Use water levels in boreholes downstream of the wellfield to determine propagation of radius of influence and impact on water levels towards discharge point.	In the case where abstraction is intended to directly capture all natural discharge (i.e. capturing a groundwater-fed spring), the abstraction is at the natural discharge point, and this indicator would require adaptation.
	2	Assessment of discharge	Use water level monitoring points to calculate aquifer discharge using analytical techniques.	Depending on the hydrogeological setting, analytical techniques may not be applicable even for a low-confidence estimate.
	3	Flow in discharge-receiving environment	Use surface water flow gauging station to calculate groundwater contribution to baseflow.	Impact in discharge environment may only occur after several years depending on the distance to the river, by which time changes in the catchment hydrology (e.g. urbanisation, forestry, new dams etc) may override measureable impacts Not economically feasible for aquifers with offshore subterranean discharge (unless budgets are very high and research-based). If aquifer discharges solely through evapotranspiration, groundwater discharge may require indirect monitoring through vegetation monitoring and remote sensing.
	4	Chemical composition of discharge-receiving environment	Use chemical composition of discharge-receiving environment to establish contribution of groundwater	Applicable where groundwater has a geochemical identity different to surface water. May be challenging to use quantitatively.
Change in pre-abstraction recharge	1	Detection of change in water table towards recharge zone	Use water levels in boreholes upstream of the wellfield to determine propagation of radius of influence and impact on water levels towards recharge zone.	Only relevant in a confined aquifer. In an unconfined aquifer abstraction is from within the recharge zone.
	2	Direct detection of change in water table in recharge zone	Use groundwater levels in the recharge zone to determine potential for enhanced recharge to be established	Only relevant in a confined aquifer. In an unconfined aquifer abstraction is from within the recharge zone, hence this indicator overlaps with/is identical to the change in storage indicator.
	2	Indirect detection of change in water table in recharge zone	Groundwater may discharge in recharge zone as springs (rejected recharge in a mountainous recharge zone), or as baseflow into rivers. In these cases measurement of: i) surface water flow gauging and/or ii) chemical composition of surface water, can provide indirect measurement of groundwater contribution and potential reduction of contribution.	Especially relevant if impact on surface water in recharge zone is a key consideration for acceptability and hence sustainability.

Purpose	Tier	Indicator	Description and measurement	Discussion
	3	Assessment of surface water flows in recharge zone	Use surface water flow-gauging station to determine whether surface water is being lost to groundwater.	In unconfined setting this indicator can be applicable to an upstream surface water course. Reduction in surface flows can be an indication that surface water is being lost to groundwater. May be challenging to quantify as other processes potentially impacting surface water flow (land use, climate) will need to be assessed.
	3	Chemical tracer for recharge source	Depending on the chemistry of groundwater and surface water in recharge zone, chemical tracers may be usable to detect recharge, and changing recharge characteristics.	Applicable where groundwater has a geochemical identity different to surface water. May be challenging to use quantitatively.
	4	Increased recharge	Apply various and most appropriate method(s) to estimate or directly measure recharge to the aquifer.	Several indirect and direct recharge estimation methods are available, but each with advantages and disadvantages (Kinzelbach et al, 2002). Uncertainties in recharge estimation are high, and may mask the detection of any change in recharge, unless for the simplest systems (i.e. abstraction close to a river inducing recharge). Nevertheless errors may stay constant, and so changes may be detected if the same method is repeated over time.
Change in storage	1	Detection of change in water table	Monitor water levels across the aquifer to understand changes in storage. Requires monitoring over long term to distinguish storage changes from response to climatic variability. Stabilisation of drawdown can indicate new dynamic equilibrium has been reached.	Indicator may include monitoring other abstractions to determine cause of storage loss.
Response time/status of aquifer towards new dynamic equilibrium	n/a	Age of water	Establish age of abstracted water and monitor change in age, to detect potential transition from older (stored) water to younger (enhanced recharge) water. Aquifer system reaches new dynamic equilibrium once age stabilises.	Similarly to recharge estimation techniques, estimation of the age of water also involves several uncertainties
Associated assessments	n/a	Projection of timing and magnitude of drawdown cone.	Use observed changes in water table to project timing and magnitude of water level impact at discharge point and recharge zone (supporting the Tier 1 indicator for change in natural discharge and supporting Tier 1 indicator for change in natural recharge)	It may be feasible to provide estimations based on analytical techniques if good monitoring data exists and depending on the hydrogeological setting, but commonly numerical models will be required

Purpose	Tier	Indicator	Description and measurement	Discussion
	n/a	Recovery assessment	Use water level monitoring (change in storage tier 1) to determine whether aquifer recovers fully on cessation of pumping (i.e. in seasonally used wellfields). Replenishment of storage may indicate that the pre-abstraction aquifer balance is restored on cessation of pumping.	

5. HYDROGEOLOGICAL INPUT FOR SUSTAINABILITY DECISION

5.1. CASE-SPECIFIC SUSTAINABILITY CONSIDERATIONS

The mainstreaming assessment highlights a lack of specific guidance (or discussion) on the socio-economic and environmental decision as to whether the planned use is socially, environmentally and economically acceptable, and so can be considered sustainable. For each aquifer case this sustainability decision will have to take different factors into account. The following examples illustrate the wide range in these factors:

- A confined aquifer with natural discharge occurring entirely offshore as subterranean seeps, is recharged where the aquifer is unconfined in mountainous areas. The maintainable aquifer yield is greater than pre-abstraction recharge, and the abstracted yield is derived from capturing all the offshore discharge and enhancing recharge through capturing some rejected recharge that was previously entering streams. This may be considered sustainable if there is no ecological impact, or there is acceptable ecological impact from the cessation of subterranean discharge, and if there is acceptable ecological impact from the stream flow reduction in the recharge zone. The socio-economic benefit of the groundwater use outweighs this ecological impact.
- Groundwater abstraction in the San Joaquin valley, USA, has caused land subsidence through the depressurisation of clay aquitard layers. The subsidence has caused ground surface effects that are clearly undesirable, and measures to curtail abstraction and manage the subsidence are in place (Galloway and Riley, 1999). Nevertheless, the economic benefit of large scale abstraction to the region has been significant, enabling it to become one of the world's most productive agricultural regions. Economically this certainly outweighs the cost to repair structures impacted by land subsidence. The challenge when considering whether on balance the abstraction has been socially, environmentally and economically acceptable, is that those who benefit (large agricultural businesses, shareholders) are often a different group from those who are negatively impacted (agricultural workers and landowners).
- The high plains aquifer, USA, is also heavily utilised as a source of irrigation water to one of the major agricultural regions of the world, and also as a domestic water supply source to just over 80% of the population living within the aquifer boundaries (Dennehy et al, 2002). Data shows that more water is abstracted than enters the aquifer, and that the pre-abstraction aquifer storage has reduced by around 7%, equivalent to a 30–40 m water table decline (Dennehy et al, 2002). The limit to sustainability in this case may be that water levels cannot decline beyond that which farmers can economically abstract. As stated by Hiscock (2005), 'there is no fundamental reason why the temporary over-exploitation of aquifer storage for a given benefit should not be allowed as part of a logical water resources management strategy as long as the groundwater system is sufficiently well understood to evaluate the impacts'.
- Non-renewable groundwater resources are mined in the Great Man-made River Project in Libya. The project involves abstraction of fossil groundwater beneath the Libyan Desert, recharged during the last Ice Age, and transfer of this water to coastal cities and towns. Estimations show that the groundwater resource can supply most of the population of Libya for several decades. Although the water source is not renewable, hence with lasting environmental impact, the benefit to the economy is deemed to outweigh this impact (Hiscock, 2005).

The aquifer assessment steps depicted in the procedure for operating rules and establishing sustainable groundwater use (Figure 2-6) are applicable in each of the above cases i.e. the groundwater system must be sufficiently well understood to evaluate the impacts of abstraction. However, the particular social, environmental and economic impacts (or benefits) of abstraction are heavily case specific.

The GEUS indicators (Henriksen et al, 2007, discussed in section 4.2.3) provide a tool to support the environmental aspects of the sustainability decision (impact on discharge, enhanced recharge, etc) without social and economic considerations. The authors use a combination of assumptions and empirical data to establish thresholds for changes in the aquifer flow balance that depict unacceptable environmental impacts of abstraction. A similar assessment for South Africa could be extremely beneficial and provide a framework from which hydrogeologists, surface water engineers and ecologists can discuss the impact of groundwater abstraction on surface water and associated ecology. Assessments such as water resources classification would be dramatically simplified if it was possible to establish blanket rules such as that a decrease in groundwater contribution to baseflow greater than 80% (for example) relates to a category D river ecosystem and is unacceptable. However, the GEUS authors had the benefit of water quality, abstraction, and recharge data from basins across an area of Denmark, detailed regional modelling of the entire country and several researchers working on the project over several years (the modelling effort, at least). More fundamentally, their findings of a natural constraint do suggest that the basins studied have similar terrain and hydrogeology, and the authors note that the applicability of the findings across Denmark is unknown. The establishment of natural constraints applicable across the country is unlikely to be possible in South Africa with greatly varied geology and aquifer settings. The feasibility, therefore, of generic guidelines for a user to weigh the impacts and benefits of abstraction and make a sustainability decision, is questionable.

5.2. MULTI-STAKEHOLDER AND REGULATORY DECISION

In addition to the particular social, environmental and economic impacts (or benefits) of abstraction being heavily case specific, the sustainability decision is to be taken by multiple stakeholders, including regulators. These three dimensions of sustainability (economic growth, social equity and ecological integrity) are highlighted in DWAF (2008b), which states they are 'seldom in balance as there are always trade-offs based on local needs and circumstances, and the point of sustainability differs from project to project. Public participation assists decision-makers to establish the point of sustainability for each project. It contributes essential local knowledge and wisdom to project assessment, planning and design, and management, and clarifies the degree to which stakeholders are willing to accept, or live with, the trade-offs' (DWAF, 2008b, p. 1-20).

This 'sustainability decision' is what the Water Resource Classification System intends to cater for (DWAF, 2007a), however, their methodologies are not aligned with a capture approach. Integration of a capture approach with authorisation processes (the reserve, RQOs, licensing) may be required to optimally support the implementation of the capture approach, including actions such as updating the guide to licensing, tying license application forms to a capture approach (see recommendations, section 10.3).

5.3. HYDROGEOLOGICAL CONTRIBUTION TO THE SUSTAINABILITY DECISION

Although the decision is to be made by multiple parties, and is case specific, the sustainability decision requires scientific information (a prediction of the conditions at future dynamic equilibrium), and the role of hydrogeologists is to provide this information, as stated by Seward et al (2006, p. 481); 'the role of scientists should be to identify a range of sustainability options – each with a probable consequence – while it would be the managers' and stakeholders' role to select a preferred option. Scientists would then monitor the outcomes of that option and revise the sustainability scenarios as needs be'.

Cognisant of the case-specific nature of impacts and the sustainability decision, and the fact that the sustainability decision is entrenched in authorisation processes requiring multi-stakeholder input, a guideline with thresholds to weigh the impact of abstraction, as input to the sustainability decision, is not provided. Nevertheless, the following two recommendations for necessary hydrogeological contribution to the sustainability decision can, at minimum, be part of the decision framework.

1) As depicted in the procedure for operating rules based on capture assessment, the future conditions when abstracting a maintainable aquifer yield are predicted and, based on this, a sustainability decision taken. If these conditions are not considered sustainable, the maintainable aquifer yield must be adapted (Figure 2-6). A powerful hydrogeological result that can provide significant support to this process is shown in Figure 5-1 (Henriksen et al, 2007). Increasing abstraction rates are shown on the x-axis, against aquifer flow on the y-axis. Separate series are shown for the various elements of the aquifer flow regime that are impacted i.e. reduced discharge to baseflow and enhanced recharge (their 'indicators' 4 and 2). The shape, gradient and scales of the curves will vary for each system but the principle remains the same: for example, in the case of reduced discharge, an ecologist can determine the maximum reduction the ecology can sustain, stakeholders can

consider the acceptability, and the graphed result can provide the maximum corresponding abstraction yield. It is recommended that this relationship between abstraction and aquifer flow regime be quantified.

2) The results shown in Figure 5-1 are for once the system has reached a new dynamic equilibrium. Estimation of the response time remains an additional requirement. If this is very long, then the flow at some reasonable water supply planning and environmental timescale (100 years?) should be plotted, rather than steady-state.

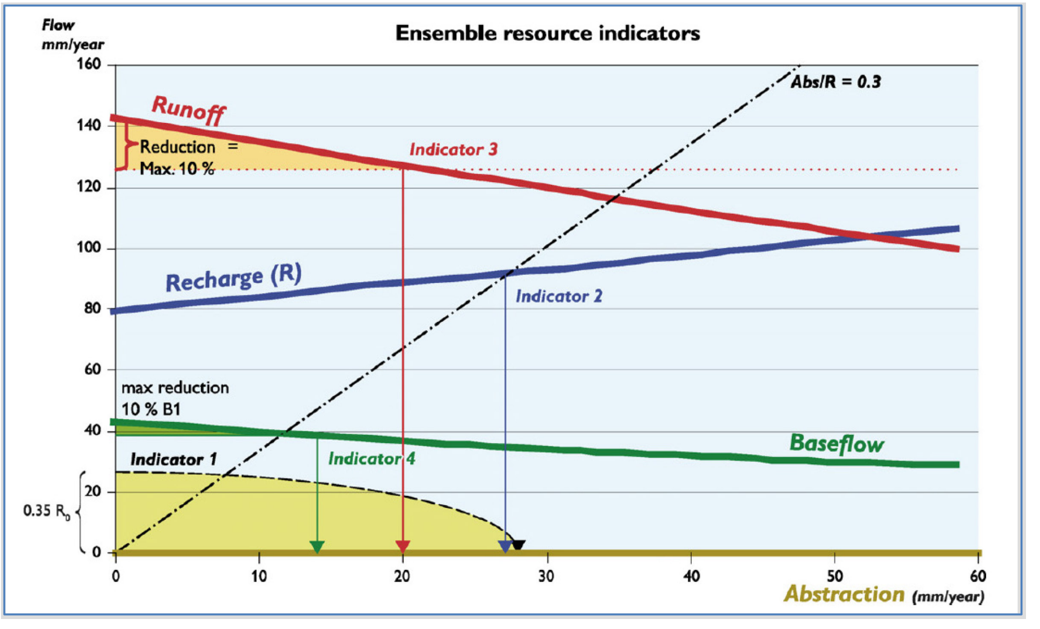


Figure 5-1 Example of relationship between aquifer flow regime and abstraction yield, required to support a sustainability decision (Henriksen et al, 2007)

6. DECISION FRAMEWORK

The aim of the decision framework (section 1) is to facilitate the translation of theoretical hydrogeological principles for sustainable groundwater use, based on the capture approach, into practice. The flowchart presented in Figure 2-6 summarises the elements to be considered and steps to be taken to derive operating rules based on the capture principle. In this format, the flow chart achieves the goal of providing a user with an accessible description of the capture principle (research aim 1). To further support the implementation of the capture principle approach to groundwater use as shown in Figure 2-6, various supporting measures have been explored and developed. These include:

- Links to appropriate (existing and new) tools and suggested processes to undertake the required assessments of a capture principle approach (section 3)
- A set of sustainability indicators which can monitor progress towards the estimated future dynamic equilibrium and therefore assist the user in updating the confidence level of the sustainability assessment, in various cycles of adaptive management (section 2.4 and 4).
- Recommendations for hydrogeological input to the sustainability decision (section 5).

A decision framework that links the elements to be considered and steps to be taken to derive operating rules based on the capture principle to these supporting measures is presented in Figure 6-1. In essence, the decision framework represents an amalgamation of recommended approaches. The recommended approach has to be applied aquifer scale, or with consideration of the hydraulic boundaries or capture zone of the abstraction in question. Water balance approaches persist particularly within projects carried out at regional or national scale (section 3.2), in which it is often not deemed practical to assess groundwater resources at the aquifer scale (primarily due to project budget and timescale). Potential ways forward to overcome this are discussed further in section 10.

There are hugely variable and case-specific factors to consider when assessing the sustainability (acceptability) of groundwater abstraction (examples given in section 5.1). The aquifer assessment steps depicted in the procedure for operating rules and establishing sustainable groundwater use (central part in Figure 2-6) are nevertheless applicable in all cases. The groundwater system must be sufficiently well understood to evaluate the impacts of abstraction. The response time and propagation of radius of influence (and impact) is not related to pump rate, only to hydraulic diffusivity, duration pumped and distance to discharge point (Figure 2-3). Low abstraction in close proximity to a river, wetland or spring will (at some point) reduce groundwater discharge, which, depending on the ecological status and sensitivity of the surface water environment, may not be acceptable. Hence, in this situation, application of the decision framework, or the capture principle-based recommended approach, is still applicable.

In several areas of South Africa, aquifers are large, hydraulic boundaries potentially very distant, and there is limited surface water (for example, aquifers of the Karoo Supergroup in the dry interior). In these situations the distance between abstraction and aquifer discharge may be high. The speed with which the source of abstracted water transitions from aquifer storage to reduced discharge/enhanced recharge is proportional to the square of the distance to discharge point (Figure 2-3, Sophocleous, 2000). For very long distances (large aquifer systems) with low diffusivity (high storativity and low hydraulic conductivity or transmissivity), the speed may be excessively slow, i.e. beyond the planning horizon, and the abstracted water is essentially derived from aquifer storage. In these cases, once a very long response time has been established, the quantification of reduced discharge/enhanced recharge may not be relevant, and the key indicator for assessment of acceptability will be storage/water level impact. Nevertheless, even in this case, the decision framework, or the capture principle-based recommended approach, is still applicable, in order to come to this conclusion and motivate aquifer management based on storage/water level change.

Where the risks of abstraction are low, (i.e. no sensitive receptors, natural discharge is significantly distant from abstraction point), and the potential risk of poor definition of sustainable aquifer yield is low, a preliminary assessment of each element in Figure 2-6 may be sufficient, linked to low-confidence operating rules (section 2.4). As risks of abstraction increase, iteration of the procedure in Figure 2-6 becomes more critical and leads to increased operating rule grades, through the implementation of sustainability indicators.

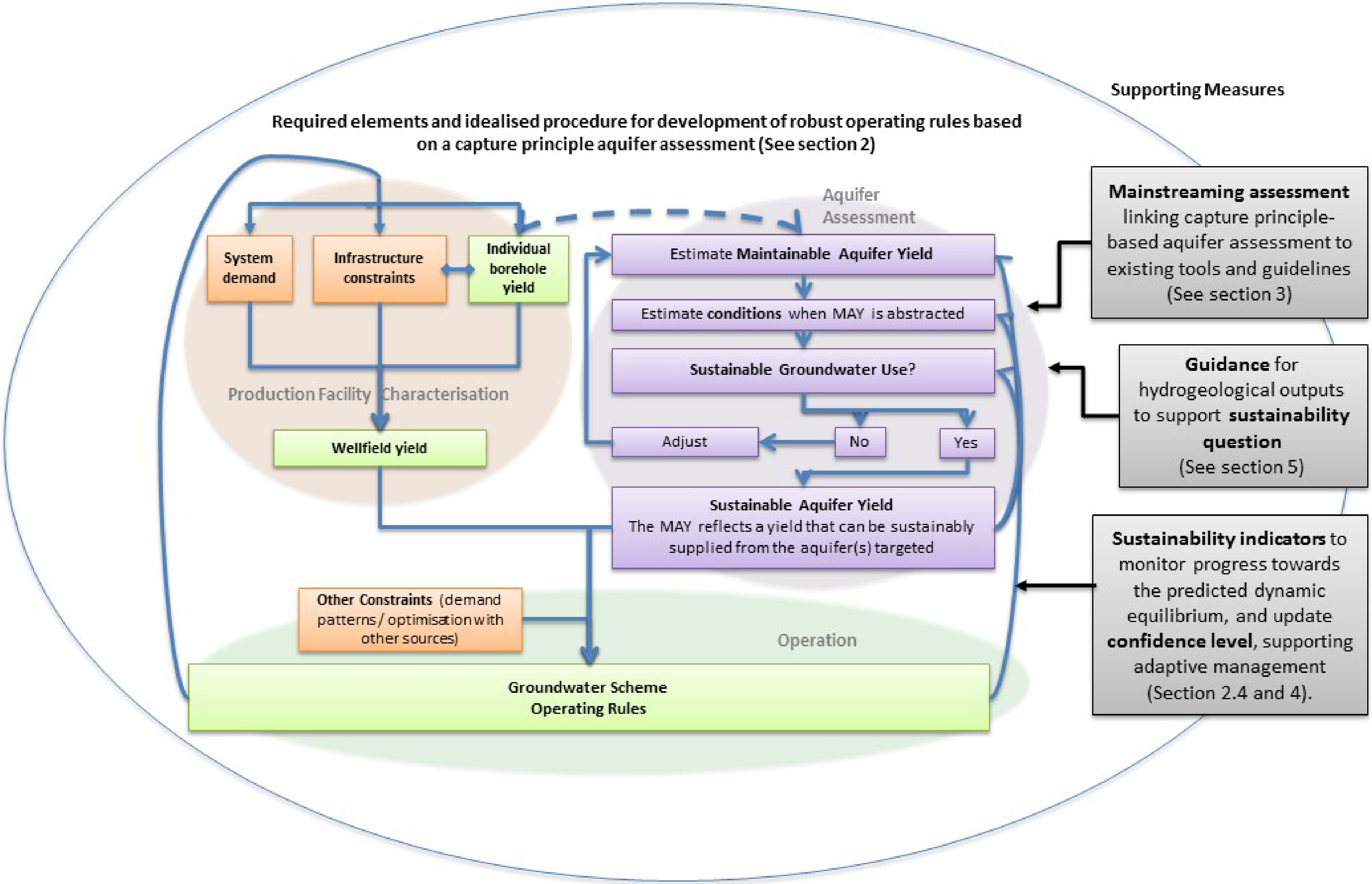


Figure 6-1 **Decision framework for sustainable groundwater use**

Part 2: Illustration of the Capture Principle using two case studies

7. CASE STUDY 1: WEST COAST AQUIFERS

7.1. BACKGROUND

The Cenozoic sand aquifers on the west coast of the Western Cape of South Africa (Figure 7-1) have for a long time been recognised as significant groundwater resources. In 1976 the Saldanha Subterranean Government Water Control Area was established to protect the aquifer units in the region of Saldanha for future urban and industrial use. Since then, the various aquifer units in the area have been the subject of significant hydrogeological investigation including: analysis of monitoring data; groundwater isotope studies; assessment of the interaction of the aquifer systems with surface water; several numerical modelling yield assessments; and artificial recharge investigations. In 1999 the West Coast District Municipality (WCDM) commissioned a wellfield at Langebaan Road (Langebaanweg in Figure 7-2), for supply to Saldanha Bay Local Municipality (and primarily the town of Saldanha), and this abstraction is ongoing (although recently interrupted by vandalism-related infrastructure problems).

7.2. APPLICATION OF DECISION FRAMEWORK TO CASE STUDY

Part 1 of this report documents that an assessment of aquifer yield that is in line with the capture principle would determine the following:

1) The (maximum) maintainable aquifer yield.
2) The source of abstracted water at this yield, once a new dynamic equilibrium is established (i.e. the steady-state mass balance across the aquifer). As input to a decision by the relevant authority and/or stakeholders on whether the abstraction is socially, environmentally and economically acceptable, quantification of the relationship between yield abstracted and aquifer conditions (storage impacts, mass balance). This could be in the form of a graph of changing aquifer flows (y-axis) against abstraction yield (x-axis), as exemplified in Figure 5-1.
3) The response time (the time taken to reach this new dynamic equilibrium).

Despite the volume of work conducted on the Cenozoic sand aquifers in the West Coast region (termed West Coast aquifers (WCA)), the above components of a capture principle-based yield assessment have not been determined (Table 7-1, section 7.12). Abstraction continues by the WCDM without a predefined or estimated future dynamic equilibrium, hence the potential future impacts of abstraction are not known. Whether aquifer storage is still currently being harvested, or whether the natural discharge to Saldanha Bay has decreased, or will decrease (and the associated ecological impacts of this), has not been estimated. The maximum yield before saline intrusion becomes a risk has not been estimated. There is also a real risk of *under*-exploitation of available groundwater resources, given that assessments of the 'sustainability' of groundwater abstraction have revolved mostly around considerations of reduction of water level.

Without an estimation of the future state of the aquifer (flow regime once dynamic equilibrium reached), the sustainability indicators developed in section 4 cannot be tested here. The natural place to apply the decision framework (or more broadly the recommendations from section 6) is with a determination of the maintainable aquifer yield, and the response of the aquifer to current and potential increased abstraction.

Water demand in the West Coast region (supplied by the Withoogte scheme) is greater than allocated supply, and various investigations have been carried out to assess augmentation options. Desalination is currently the favoured intervention by the Saldanha Bay Local Municipality but has significant cost and energy implications for the region. Additional water from the Berg River is the preferred intervention by the WCDM (GreenCape, 2015). However, the DWS has advised that the WCDM's licence application for additional allocation from the Berg River should not be awarded until additional resource interventions have been brought online for the Western Cape Water Supply Scheme (DWS, 2015b). A capture principle-based yield assessment of the WCAs could, if the results show that more water is available than is currently considered, contribute to renewed consideration of groundwater for augmentation to supply.

Given the current situation, the use of the WCAs as a case study in this project was suggested by DWS (meeting held on 12 August 2014), and was agreed to at the inaugural reference group meeting (26 November 2014). The case study endeavours to test the central aquifer assessment section of the decision framework and provide the following recommended hydrogeological outputs to support sustainability decisions (described in section 5):

- Source of water when maintainable or recommended yield is abstracted (i.e. impact on natural flow regime)
- Quantified relationship between aquifer flows and increased abstraction.

A numerical model was established for this purpose (section 7.14 to 7.17), based on the conceptual model described in sections 7.3 to 7.13.

Figure 7-1 Regional location map

Table 7-1 List of key hydrogeological studies of West Coast aquifers

Reference	Year of completion	Type of assessment	Summary content
Timmerman (1985b and c), Timmerman (1988)	1985; 1988	Groundwater resources assessment	1985b and c: Detailed regional assessment including all key elements of a hydrogeological analysis (geology, hydrostratigraphy, delineation of aquifer units, piezometry, water quality). 1988: Numerical modelling carried out to estimate yield. Short-term yields were determined, which were based on recharge and avoiding significant decline in water levels. Long-term aquifer yields were determined which allowed for a certain degree of aquifer 'mining' (or simply loss of storage: which all abstraction causes initially). Not all elements of a capture principle-based assessment were carried out, and the definition of yield is misaligned with the capture principle.
Woodford (2002)	2002	Groundwater resources assessment	Detailed assessment of water level responses to abstraction, building on all aspects of the regional hydrogeology from Timmerman. Included significant GIS mapping of geological layers, and of the basement elevation.
Woodford & Fortuin (2003)	2003	Groundwater resources assessment	Builds on Woodford (2002) to assess aquifer potential. Although numerical modelling for aquifer yields was not carried out, detailed assessments of aquifer potential (based on recharge and hydrogeological characteristics) were provided, along with proposed target zones.
Saayman et al (2004), Saayman et al (2005)	March 2004; March 2005	GW/SW interaction	Includes appendices with various technical papers including: borehole drilling at Langebaan; use of geophysics to detect GW discharge to lagoon; drilling logs of boreholes in West Coast National Park.
WCDM (2005)	February 2005	Monitoring/ data report	Detailed analysis of monitored water level in wellfield and surrounding boreholes for 1998–2005. Includes a summary and some update of aquifer information from Woodford (2002) (basement elevation, thickness of Cenozoic deposits), and an update on the conceptual model of the aquifer (especially hydraulic connectivities of compartments) based on water level responses.
DWAF (2007b)	Study report 2007	Groundwater resources assessment	Summarised recommendations from the resources assessment (Woodford and Fortuin, 2003) for Elandsfontein Aquifer System, Adamboerskraal Aquifer System, and Papkuils Aquifer System were: • Elandsfontein Aquifer System: pre-feasibility results were positive, therefore target zones were delineated and costs provided for a recommended pilot abstraction scheme. However, a potential aquifer yield was not provided. • Adamboerskraal Aquifer System: pre-feasibility stage hydrocensus detected salinity of groundwater 'unacceptably high' and, based on this, the option wasn't assessed further • Papkuils Aquifer: screening phase showed positive results (potentially high yield) but there was little information on which to base the pre-feasibility study, as a hydrocensus was not conducted.
WCDM (2008)	April 2008	Monitoring /data report	Documents monitoring results (abstraction, water level, water quality) between November 2007 and April 2008, for boreholes focused around the Langebaan Wellfield

Reference	Year of completion	Type of assessment	Summary content
DWAF (2008c)	August 2008	Groundwater resources assessment	Included development of conceptual and numerical model, with the aim to characterise the groundwater resource, and provide a quantitative basis for resource assessment into the future. The model included one scenario for utilising the Elandsfontein Aquifer System to augment the water supply to the WCDM. In this hypothetical wellfield, the impact of abstraction on water levels is determined. The conceptual model and numerical model design is detailed, however, only the water level change is presented and the steady-state mass balance under this abstraction scenario is not provided (i.e. the conditions at the future dynamic equilibrium), nor the response time. As only one abstraction scenario is tested, the numerical model is not used to determine a (maximum) maintainable aquifer yield, nor develop a relationship for (change in) aquifer flux compared to abstraction yield.
WCDM (2009)	Volume 2: June 2009	Groundwater resources assessment	A critical review and summary of previous hydrogeological reports is provided (excluding DWAF, 2008c; it appears these two studies were carried out in isolation). The hydrogeological information from pre-feasibility study (DWAF, 2007b) was incorporated into a numerical model, with the aim of 'assessing the long-term yield of the Langebaan Road Aquifer System (LRAS) and Elandsfontein Aquifer System (EAS)' (aquifer systems described in section 7.7). Following model simulations, conclusions were drawn on exploitation potential. The model results show: • Abstraction at the existing WCDM wellfield at LRAS can sustainably be increased from 1.45 million m^3/a to 1.9 million m^3/a, and expansion of the wellfield is recommended. • The EAS could supply 1.45 million m^3/a to 1.9 million m^3/a and development of a pilot wellfield is recommended. However, these groundwater recommendations are not taken forward into the study options analysis (Volume 4), and groundwater is not listed as a potential source in the summary document. These recommended yields are based on a 10-year projection of water levels and change in flow regime under abstraction. The steady-state mass balance under the abstraction scenarios is not established, nor the response time. As only one abstraction scenario is tested, the numerical model is not used to determine a (maximum) maintainable aquifer yield, nor develop a relationship for (change in) aquifer flux compared to abstraction yield.
Tredoux & Engelbrecht (2009)	December 2009	Artificial recharge & resources assessment	Summarises previous hydrogeological investigations, and contributes to the understanding of hydraulic connectivity between aquifer systems with analyses of water level responses to abstraction. Includes numerical modelling with several aims including defining the sustainable yield of the LAU & UAU. The model appears the same as that of WCDM (2009), ran for a 9-year time period. Five abstraction scenarios were tested to determine the 'sustainability of abstraction and injection'. The model scenarios are assessed only in terms of the extent of drawdown and speed of recovery.
DWA (2010b)	November 2010	Artificial recharge report	Repeats key information from Tredoux & Engelbrecht, and suggests a way forward. Also includes a summary of key hydrogeological unknowns for Langebaan (recharge area, recharge rates), and lists required investigations into these unknowns.

Reference	Year of completion	Type of assessment	Summary content
WCDM (2011), **WCDM (2012)**	2012	Monitoring / data report	Quarterly and annual monitoring reports from 2010 to June 2013. Reports include all monitoring data (abstraction yields, water level, water quality, rainfall), and water level trend analysis for Langebaan Road Wellfield, and abstraction at several surrounding farms
Parsons & Associates (2006), **Parsons & Associates (2014)**	July 2006; May 2014	Monitoring / data report	Includes water level and pump test results, hydraulic property estimation, water quality test results, for boreholes drilled in West Coast National Park (Paaikamp borehole and R27 borehole)

7.3. TOPOGRAPHY, DRAINAGE AND LAND USE

The topography is dominated by the underlying geology, with sand dune along coastal areas reaching up to 100 mamsl, relatively flat-lying sandy plains across most of the inland area, especially in the flood plain of the Berg River, and intrusive granite plutons generating koppies reaching up to 300 mamsl (Figure 7-2).

The Berg River is the dominant perennial river in the region, lying in a broad flat plain with elevation less than 20 mamsl. The Berg River drains north-westwards into the Atlantic Ocean at St Helena Bay. The Sout and Groën Rivers (and their tributaries) drain northwards into the Berg River, with their sources in the higher relief Malmesbury Group outcrop areas of the G10L catchment.

According to the national land use mapping classification (NLC, 2000, cited in DWAF, 2008c), the land use is dominated by shrubland and low fynbos, and commercial, cultivated land (large commercial farms). Built-up and industrial areas occur in the small towns of the area, which include Saldanha, Langebaan, Velddrif and Hopefield.

7.4. CLIMATOLOGY

The climate in the region is considered Mediterranean, with mean annual evaporation (MAE) exceeding mean annual precipitation (MAP), and most of the rainfall occurring between the months of May to August (DWAF, 2008c). Rainfall distribution maps based on Vegter (1995)and GRA II are shown in Figure 7-3, and indicate that the MAP varies across the region from < 100 mm to around 500 mm.

Four weather stations exist across the area, with significant long-term data available at each. Data for these stations was sourced from the regional office of the DWS, from the WCDM, and harvested from previous literature (WCDM, 2009). The currently available rainfall data is documented in Table 7-2, and the location of the weather stations is included in Figure 7-3. A further station at Cape Columbine has data available since 1937 from the South African Weather Service (SAWS), however this data is not available from any other sources.

The annual precipitation per weather station, according to the currently available rainfall data, is shown in Table 7-3. Where multiple data sources exist for the same time period per station, yet values within the data differ, priority was given to the original SAWS data (if available), followed by monthly data, followed by annual data. The mean average of the annual rainfall totals represents the MAP, and is shown in Table 7-3. The distribution of MAP is shown in Figure 7-3 (Vegter, 1995) and Figure 7-4 (GRA II, DWAF, 2006).

The inter-annual variability and the variation between stations is significant (Table 7-3, Figure 7-5, Figure 7-6). The geographical setting for the Langebaan South African Police Service (SAPS) weather station (0040/035/8) is similar to the Geelbek weather station (0040192 4); both are at low elevation, along the Langebaan Lagoon, around 15 km from each other. However, the Langebaan SAPS station has an MAP that is 30% higher than the MAP at Geelbek. The data is questionable, however Figure 7-6 indicates that several high rainfall years contribute to this MAP (rather than one or two erroneous readings), such as 2001, 2002, 2007 and 2009 with around 500 mm/a. Without the original daily SAWS data this cannot easily be interrogated. Figure 7-7 shows that this higher MAP is based on a consistently higher monthly average.

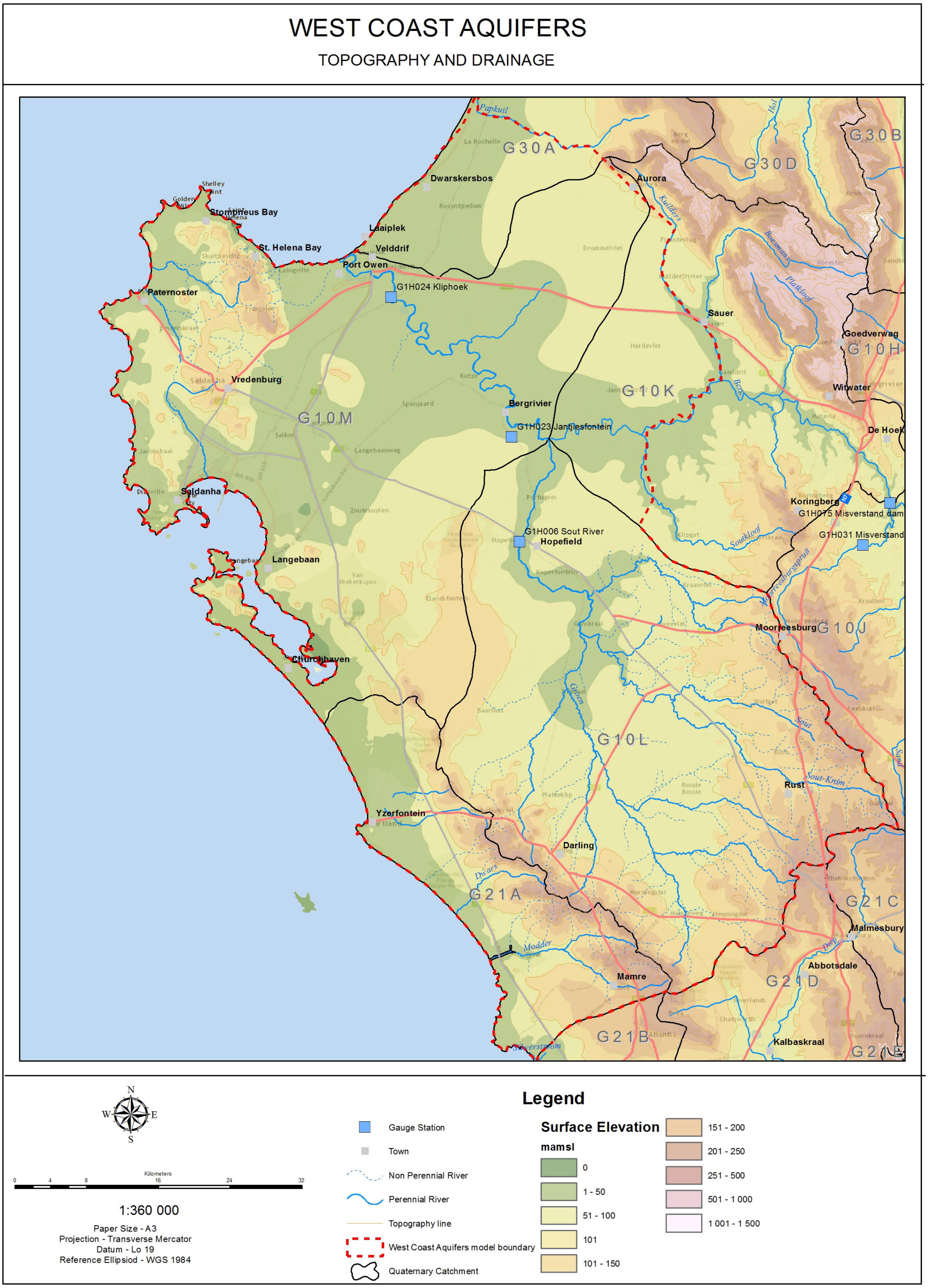


Figure 7-2 Topography and drainage

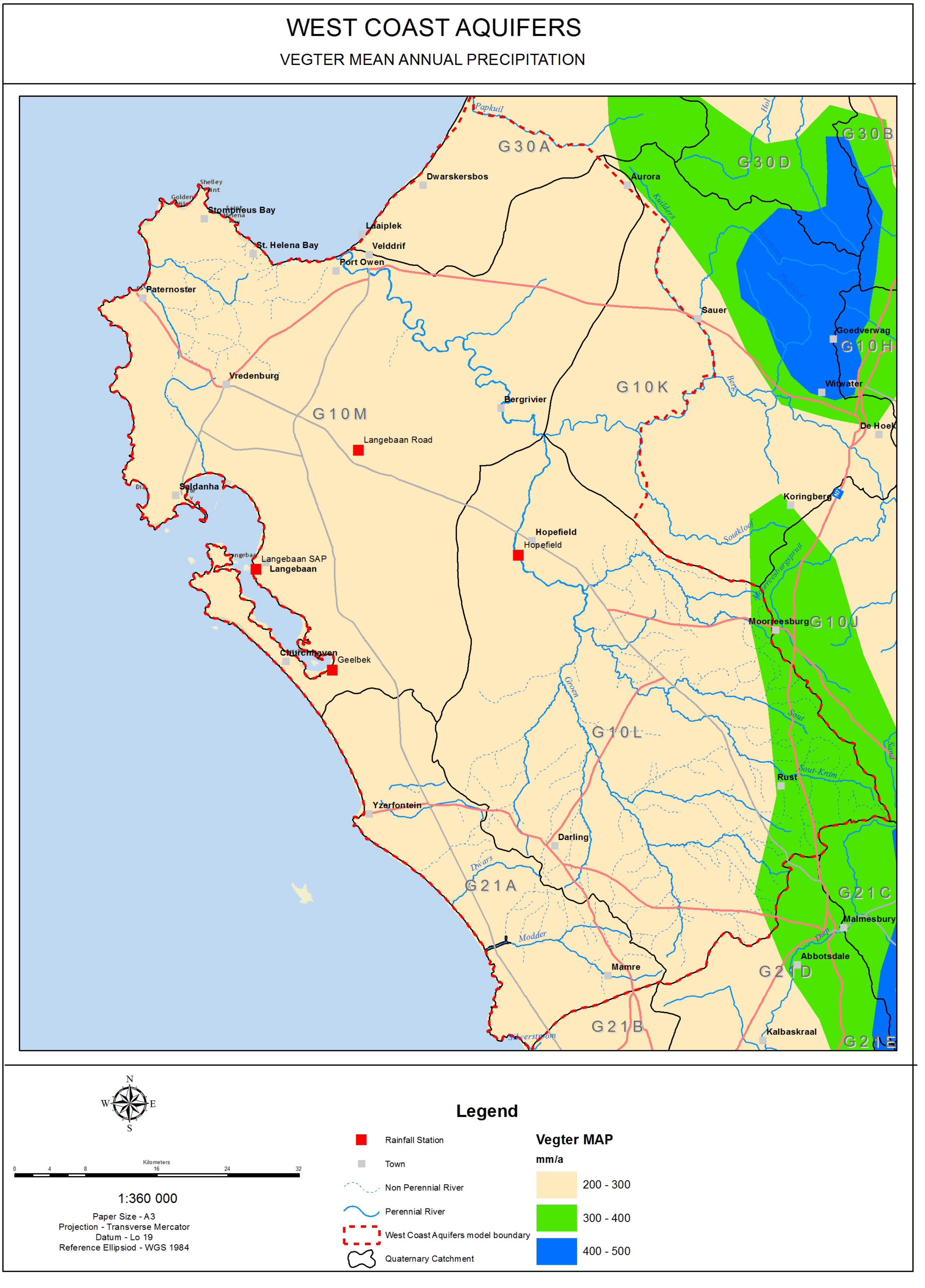


Figure 7-3 **Mean annual precipitation distribution (Vegter, 1995)**

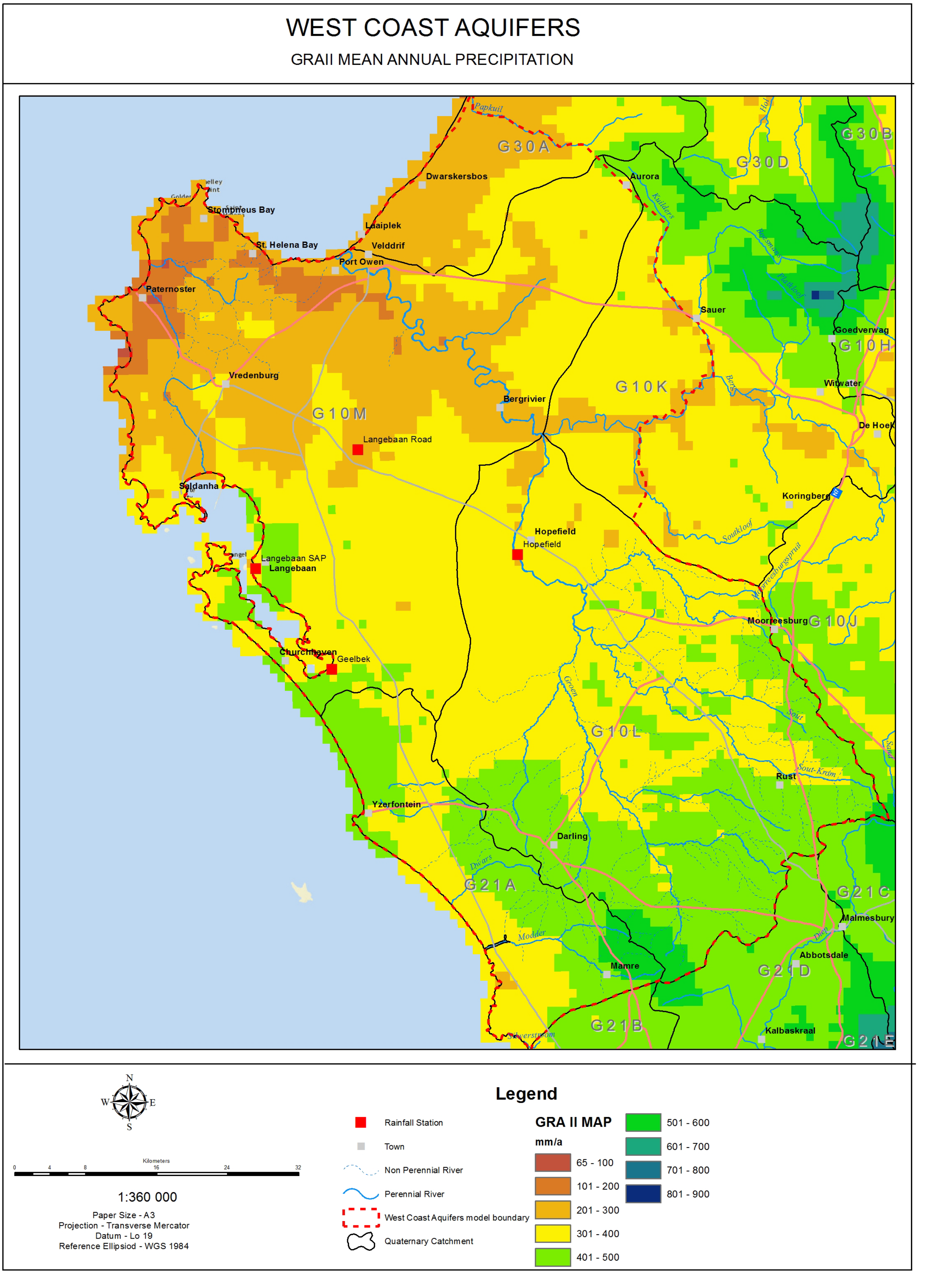


Figure 7-4 Mean annual precipitation distribution (GRA II, DWAF, 2006)

Table 7-2 Available weather station rainfall data

Station name	DWS, WCDM name	Hopefield 178-2R	Langebaan Road 185-1R	Geelbek	Langebaan SAPS
	SAWS name	**Hopefield 0040/604/3**	**Langebaanweg Aws - 0061298 8**	**Geelbek - 0040192 4**	**Langebaan 0040/035/8**
Data sourced	**SAWS (daily)**	(none)	1980–2015	(none)	(none)
	DWS (monthly)	1973–2002[6]	1973–2002	(none)	(none)
	WCDM (monthly)	2000–2009	2000–2012	2000–2009	2000–2009
	Element (annual)	1973–2008	1973–2008	1997–2008	1997–2008
Data available but not sourced	SAWS (daily)	1907–2015	1974–2015	1991–2015	1912–2015

[6] All sources for Hopefield weather station are missing 1985-1997

Table 7-3 Annual precipitation (mm) from monthly weather station data, and calculated MAP[7]

SAWS name	Hopefield 0040/604/3	Langebaanweg Aws - 0061298 8	Geelbek - 0040192 4	Langebaan 0040/035/8
MAP calculated:	**322.9**	**272.2**	**270.6**	**386.5**
1973	235.4	185.0		
1974	433.0	395.1		
1975	264.1	253.7		
1976	369.5	270.6		
1977	415.6	365.6		
1978	241.7	164.2		
1979	254.2	195.7		
1980	290.0	261.2		
1981	266.6	234.8		
1982	289.6	262.0		
1983	382.6	258.9		
1984	409.0	322.5		
1985		320.2		
1986		311.5		
1987		334.7		
1988		266.7		
1989		313.0		
1990		310.7		
1991				
1992		220.6		
1993		271.3		
1994		246.6		
1995		256.5		
1996		370.0		
1997		186.0	260.0	240.3
1998		210.9	172.6	223.0
1999	232.5	311.8	210.4	
2000	232.4	164.9	167.6	229.1
2001	383.8	372.6	373.6	557.8
2002	323.9	344.7	317.6	528.0
2003	261.6	179.0	242.6	301.9
2004	323.7		273.8	438.8
2005	279.0		227.4	267.1
2006	296.9		274.6	422.9
2007	468.5	384.8	376.0	488.2
2008	383.2	278.4	351.0	406.5
2009	389.8			534.9
2010		232.8		
2011		240.8		
2012		228.6		
2013		254.2		
2014		289.8		

[7] Blanks are shown where there is no data available or where incomplete data precludes provision of annual precipitation

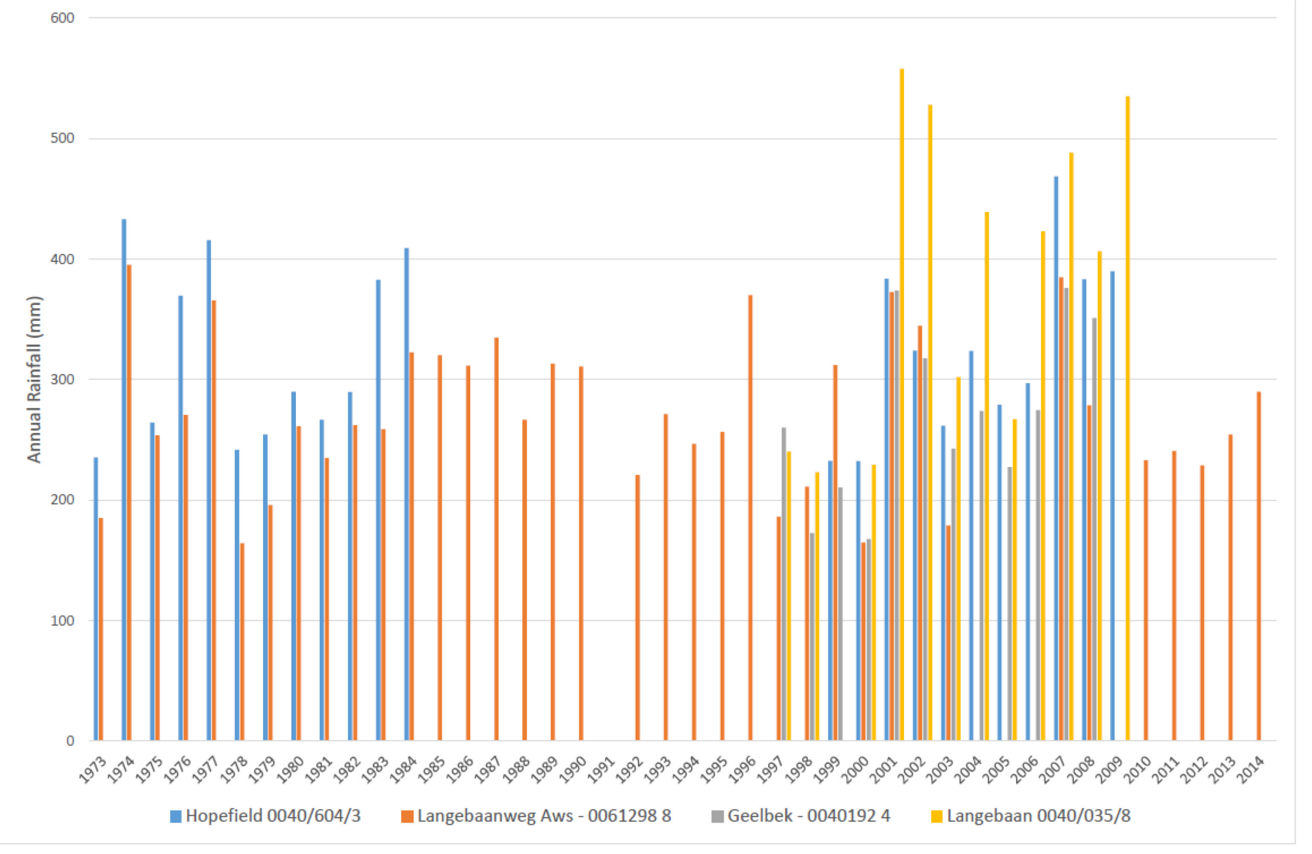


Figure 7-5 **Annual rainfall per weather station (bar), 1974–2014**

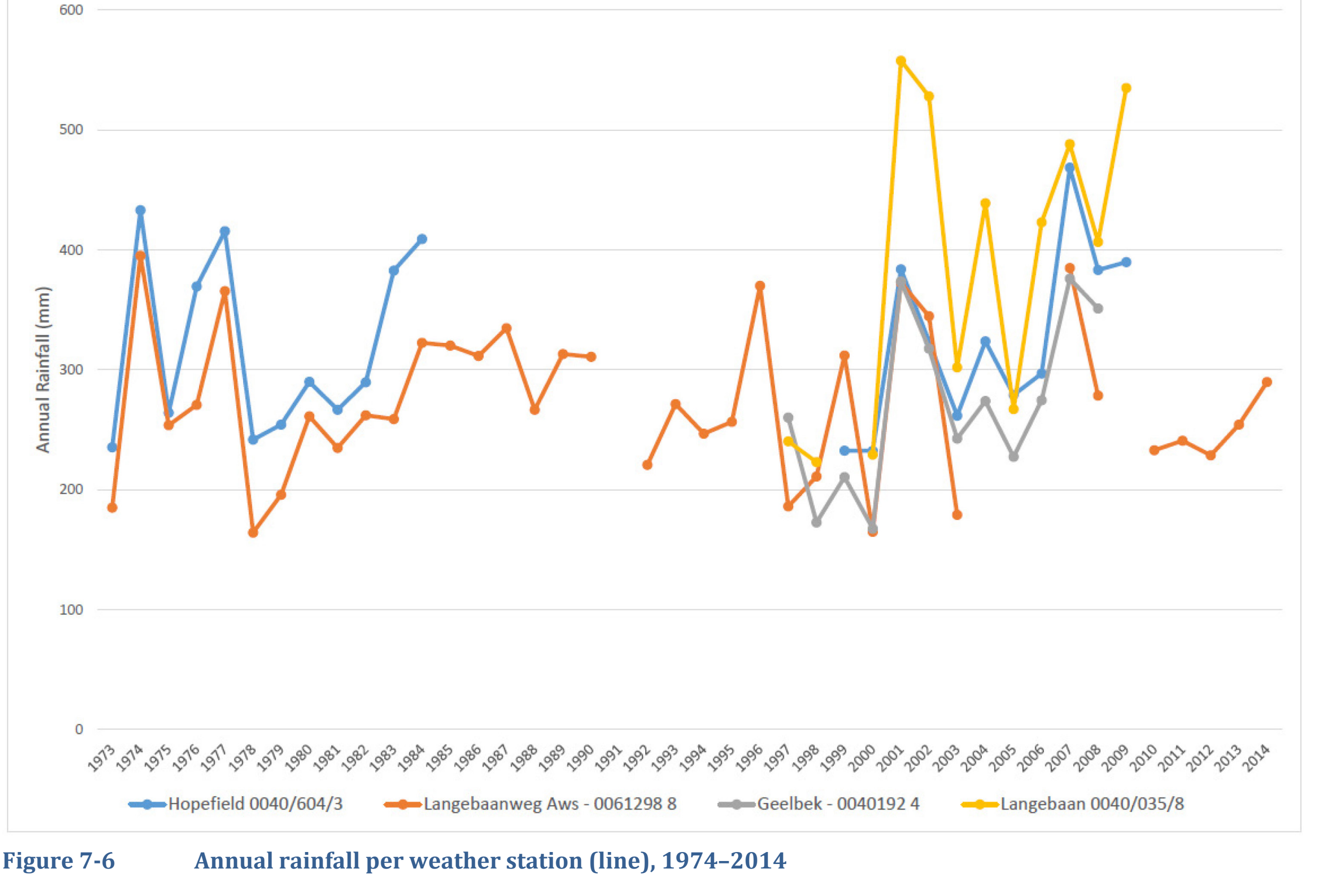


Figure 7-6 **Annual rainfall per weather station (line), 1974–2014**

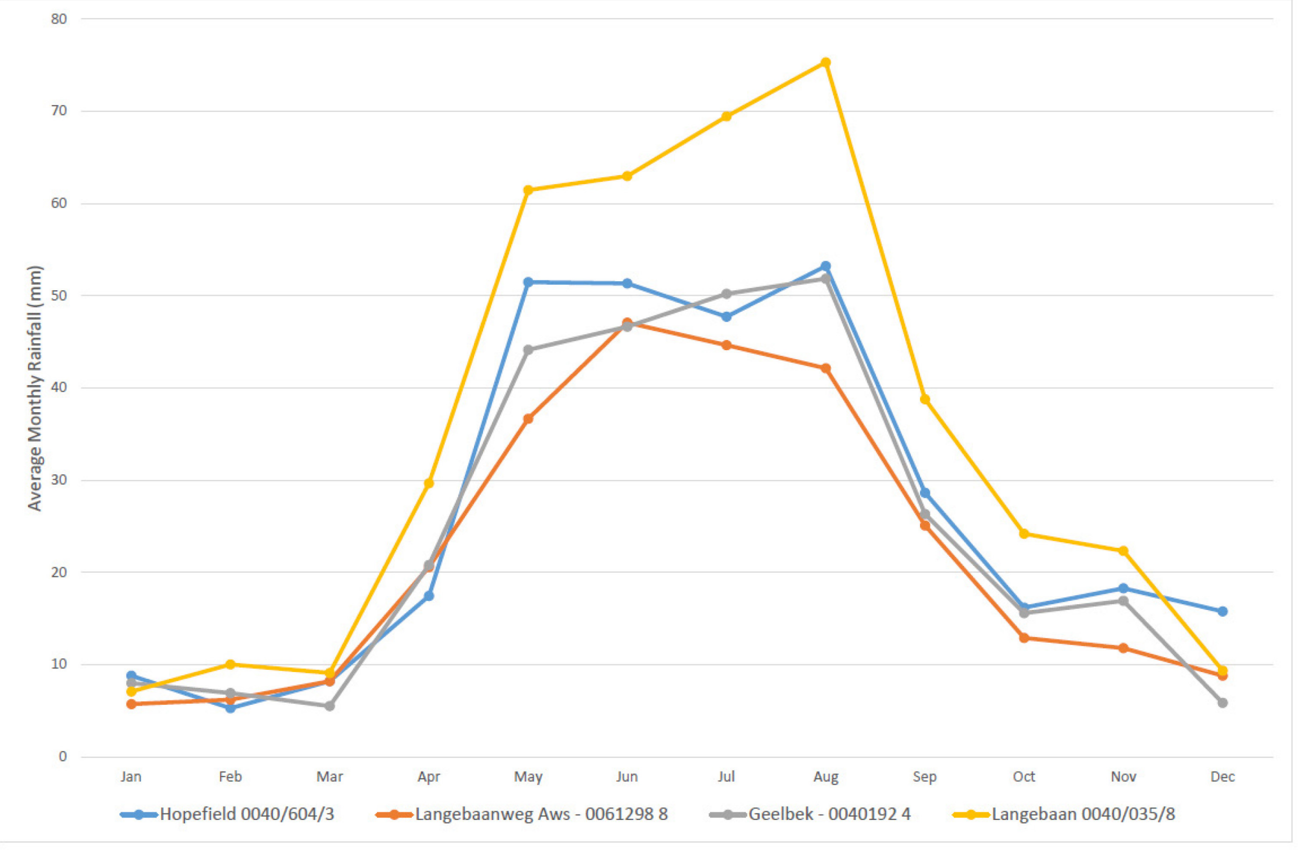


Figure 7-7 Average monthly rainfall per weather station

7.5. HYDROLOGY

Data from DWS for several surface water flow gauges has been collated. (The flow gauges are indicated on Figure 7-2). However, the data is captured in metres above base plate, hence can only be used to illustrate typical annual fluctuations. It is documented in DWAF (2008c) that 'Data from a DWAF flow gauging station at Jantjiesfontein shows typical annual fluctuations of 1.6 m between summer and winter levels... The data also shows peak river flows are up to 2 m above the annual average stage'. MAR data at quaternary catchment scale (Table 7-4) gives an indication of the variability in runoff between the catchments, with G10L (Groën and Sout Rivers), and G21A having the higher runoff, related to the higher rainfall in the area and the steep slopes formed by basement geology.

No rating curve was available from DWS to convert the flow gauge data from metres above base plate to flow, and no survey data for the base plate was available to convert the data to mamsl. This data limitation impacts on the representation of surface water in the model, and the model calibration (section 7.14.3.2 and 7.14.4).

Table 7-4 MAR and MAE from WR90 (from DWAF, 2008c)

Quaternary catchment	Area (km^2)	MAP: Berg WAAS (mm)	MAR: WR90 (mm)	Runoff efficiency: WR90
G10L	1755	305	29	0.07
G10M	2005	225	9	0.03
G21A	523	345	32	0.08
G30A	761	309	6	0.02
G10K	1176	318	21	0.05

7.6. Geology

The predominant geology of the region is the semi- to unconsolidated Cenozoic (65 Ma to present) sediments of the Sandveld Group[8] and the underlying basement rocks of the Malmesbury Group Shales in the northeast, and the Vredenburg and Darling Plutons of the Cape Granite Suite in the southwest (Figure 7-8, Figure 7-9). The Cenozoic Sandveld Group unconformably overlies the basement. The geology of the Sandveld Group in the area has been described in detail, most recently by Roberts and Siegfried (2014).

At the base of the Sandveld (Table 7-5), the Elandsfontyn Formation is a poorly-sorted, angular, fine-to-coarse-grained quartzose sand and gravel, with variable proportions of sandy clay, carbonaceous clay and lignite, with fluvial origin (Roberts and Siegfried, 2014). Its distribution is closely linked with the basement topography, occurring only in bedrock depressions. The Elandsfontyn Formation does not outcrop in the region, and is only known from deep boreholes (Roberts and Siegfried, 2014). The upper layers of the Elandsfontyn Formation include significant thickness of clay. The Elandsfontyn Formation is overlain by the variable Varswater Formation, whose distribution is associated with that of the Elandsfontyn Formation, but its extent is even more limited (Roberts and Siegfried, 2014). The overlying sands of the Langebaan, Springfontein and Witzand Formations cover the area, forming the palaeo and current dunes.

Table 7-5 Stratigraphy of units present in the West Coast area (from DWAF, 2008c, Roberts et al, 2006, Roberts and Siegfried, 2014)

Age range (million years ago)	Group	Formation	Origin	Description
0–2.5	Sandveld	Witzand	Aeolian	Semi-consolidated calcareous dune sand
		Springfontyn	Aeolian	Clean quartzitic sands, a decalcified dune sand. Dominates in the coastal zone
		Langebaan	Aeolian	Consolidated calcareous dune sand. The Aeolian deposit accumulated during the last glacial lowering of sea level when vast tracks of un-vegetated sand lay exposed on the emerging sea floor
		Velddrif	Marine	Beach sand. Associated with the last interglacial sea level rise to 6–7 m above present level
2.5–25		Vaarswater	Multiple sedimentary settings: shallow-marine, estuarine, marsh and fluvial.	Deposits include a coarse basal beach gravel member, peat layers, clay beds, rounded fine-to-medium quartzes sand member and palatal phosphate-rich deposits.
		Elandsfontyn	Fluvial	Coarse fluvial sands and gravels, deposited in a number of palaeochannels filling depressions. The upper sections of the formation include clays and peat.
		Major unconformity		
>495	Cape Granite Suite			Granites
	Malmesbury Group			Metamorphosed shales

There is divergent opinion on certain key features of the palaeotopography. The literature converges in the interpretation that distinct incisions occur in the basement topography which can be correlated with palaeo-courses of the Kuilders, Berg and Groën rivers (Timmerman, 1985c, Woodford et al, 2003, DWAF, 2008c, Roberts and Siegfried, 2014). However, interpretation of the lateral extent and continuity of the palaeochannels (and therefore the lateral extent and continuity of basal gravel of the Elandsfontyn Formation) differs in the literature:

[8] Previously termed Bredasdorp Group, the Sandveld Group is equivalent to the Bredasdorp of the Southern Cape (Roberts and Siegfried, 2014).

- Woodford et al (2003, cited in WCDM, 2005) define a southern palaeochannel (the Elandsfontein palaeochannel) that is continuous towards the Langebaan Lagoon. A northern palaeochannel (the Langebaan palaeochannel), beneath the WCDM wellfield, is not continuous towards the Saldanha Bay. These two palaeochannels are separated by a (-20 to 0 mamsl) basement high. There is no basement depression extending beneath the Berg River (Figure 7-10). The basement map was generated through use of previous maps, and additional borehole data.
- The argument is put forward in DWAF (2008c) that if the palaeochannels are palaeo-courses of previous rivers, they would form continuous channels rather than isolated depressions (suggesting these are contouring errors only). The southern and northern palaeochannels both extend to the south-west coastline. These two palaeochannels are separated by a (-10 to 0 mamsl) basement high, which is stated to be based on a number of borehole records. There is no basement depression extending beneath the Berg River (Figure 7-11). The basement map was generated using borehole depths from the (then) National Groundwater Database (NGDB), using only those boreholes that were certain to have reached basement, and also using borehole depths from offshore drilling in Saldanha Bay.
- Roberts and Siegfried (2014) define a southern palaeochannel that is continuous towards the Langebaan Lagoon. A northern palaeochannel, beneath the WCDM wellfield, is not continuous towards the Saldanha Bay. These two palaeochannels are separated by a (-20 to 0 mamsl) basement high. The basement beneath the Berg River sits between -20 mamsl and 0 mamsl, with a -20 mamsl trough extending to the northeast from the northern palaeochannel (Figure 7-12). The basement map is generated through use of previous maps (Rogers, 1980; Timmerman, 1988; Cole and Roberts, 1996, cited in Roberts and Siegfried, 2014) and additional borehole data.

It appears that the original basement contouring showing enclosed basins (Rogers, 1980, Timmerman 1988) may have been perpetuated by taking these contours into account in later efforts (Woodford et al, 2003, Roberts & Siegfried, 2014). The reasoning applied, and therefore the resulting basement topography generated, by DWAF (2008c) is considered most appropriate (i.e. particularly the representation of palaeochannels that are continuous to the coast). Nevertheless, given the control the basement has on aquifer distribution and groundwater flow, it was deemed necessary that this study use all available information (including raw data) to form conclusions on the basement topography, the resulting aquifer distribution (section 7.7), and numerical model layering. Lithology data in the National Groundwater Archive database (NGA, replacement of NGDB) was processed to generate datasets of points (location and elevation) that represent the basement elevation (i.e. NGA geological logs that penetrated sand and then penetrated basement lithologies with the first entry for basement lithology being used as the top of the basement). A total of 8401 lithology entries (representing 2340 boreholes or geosites with lithology data) were processed resulting in 503 basement points. This data was used in combination with Roberts & Siegfried (2014), and DWAF (2008c) to generate the basement topography used in this study, through the following procedure:

- The basement contours of Roberts & Siegfried (2014) agreed closely with the NGA basement points; however the points showed significant local variation and need for smoothing, which is to be expected. Therefore the contours of Roberts & Siegfried (2014) were adopted but augmented with NGA data where the numerical model boundary is greater than the area covered therein, and where the contours were perhaps overly smoothed. For example, around the WCDM wellfield borehole, logs show the basement elevation to be -40 mamsl, yet the -40 contour in Roberts & Siegfried (2014) lies northwest of the wellfield, hence the contour was extended south. The data in Roberts & Siegfried (2014) doesn’t extend to the southwest coastline (G21A), which was assumed to lie around -20 mamsl based on some NGA data points.
- The reasoning from DWAF (2008c) regarding continuous palaeochannels was adopted, hence the basement elevation was smoothed around the mouths of the palaeochannels.

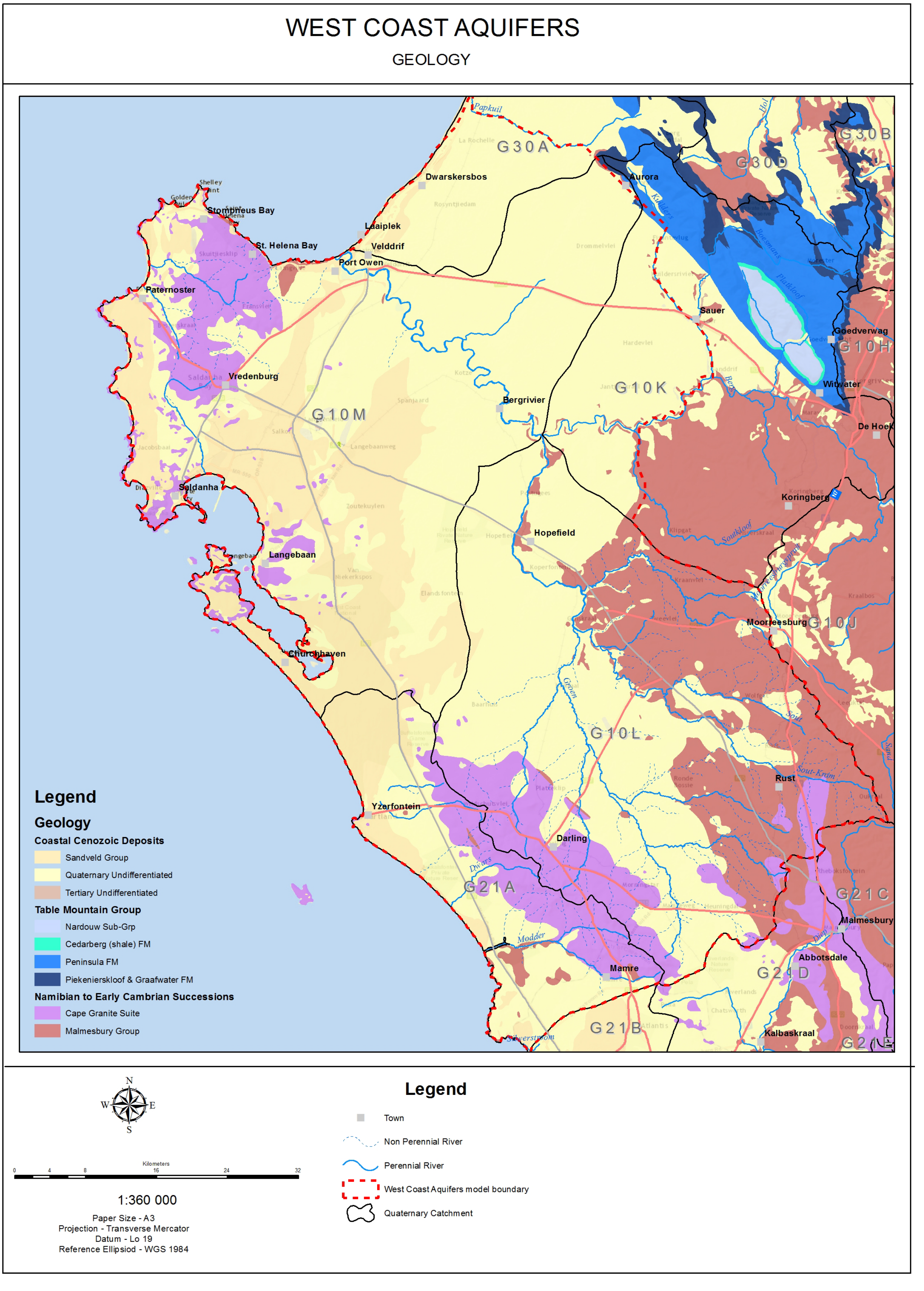


Figure 7-8 Geological map (simplified)

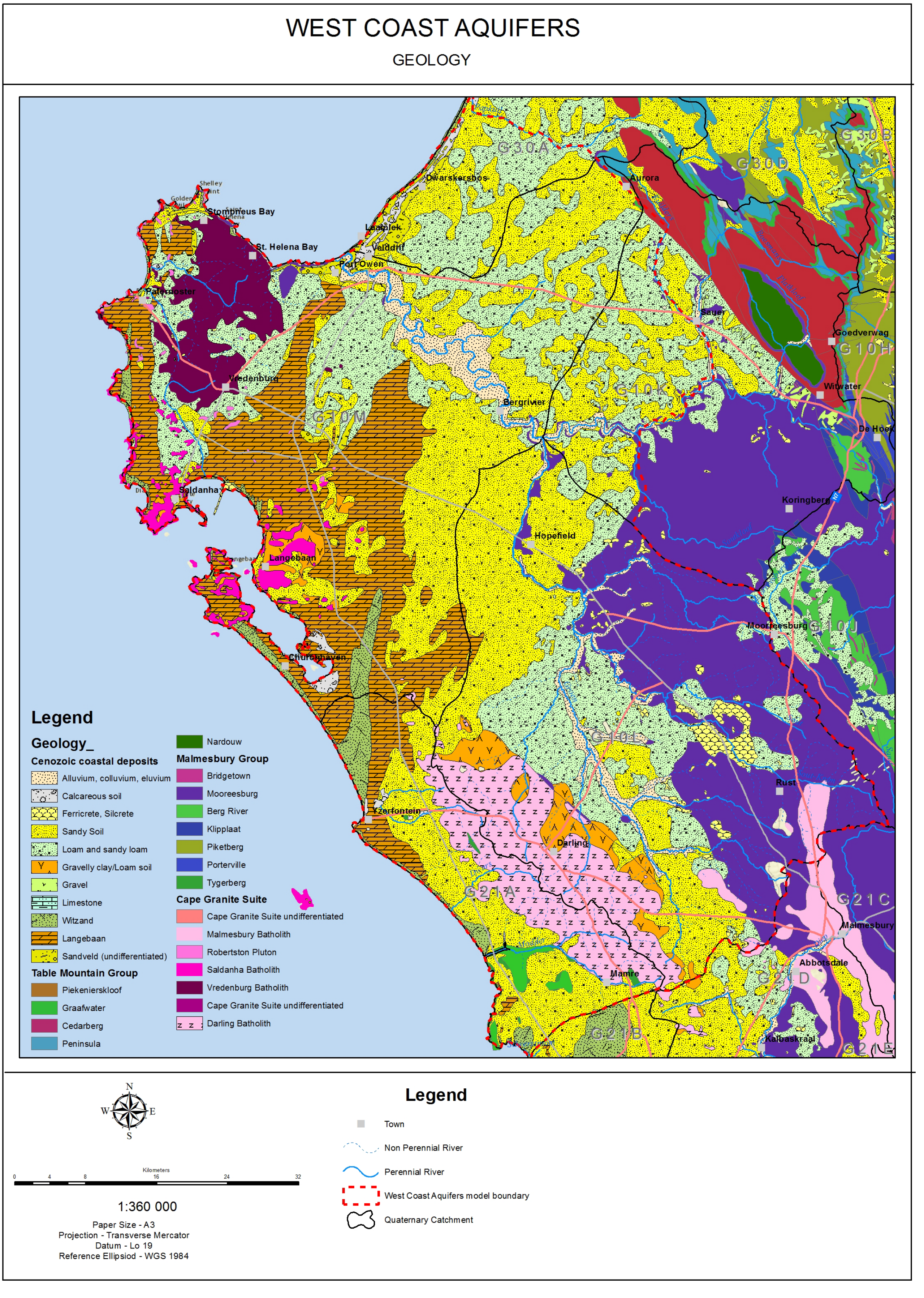


Figure 7-9 Geological map (detailed)

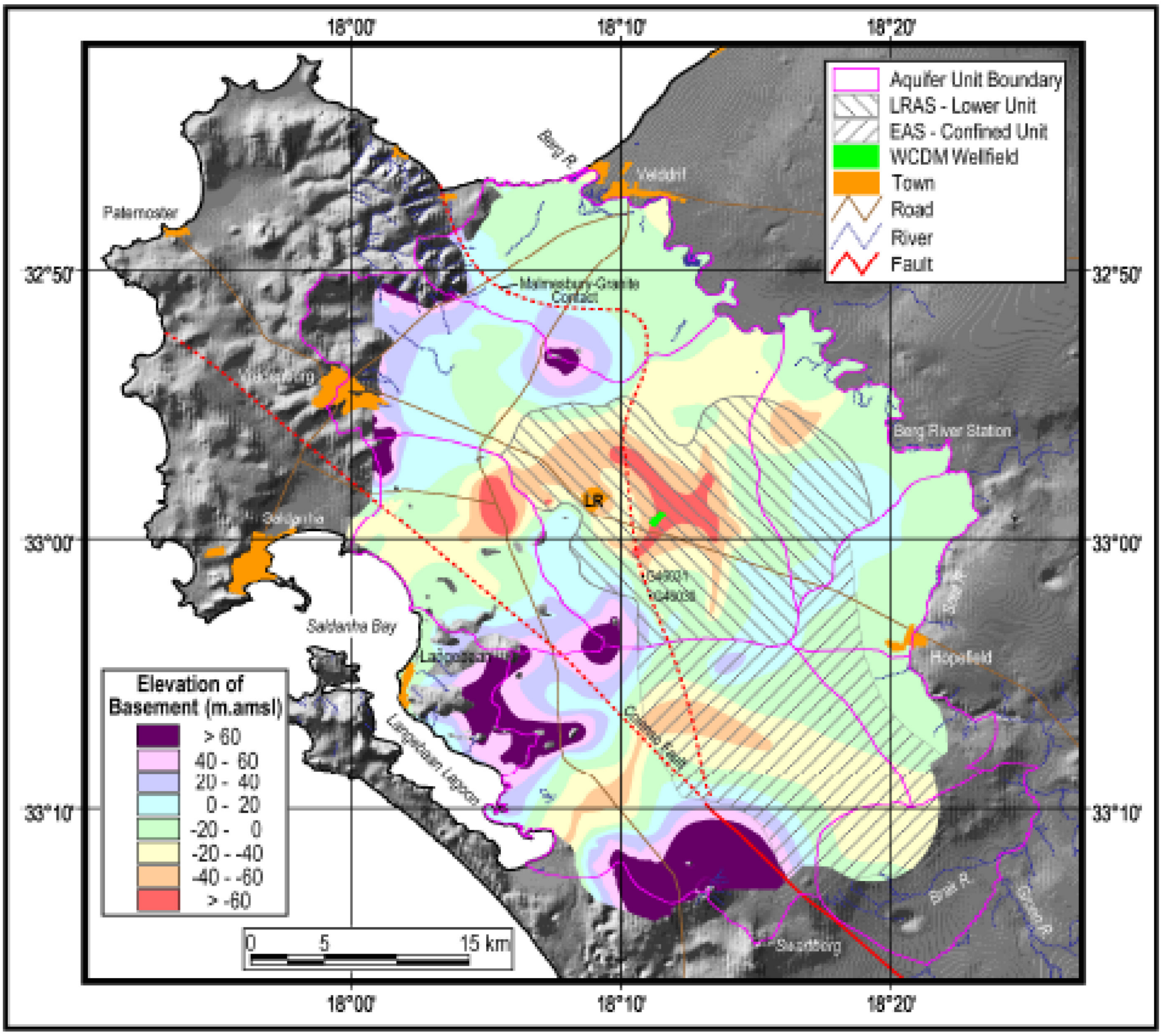


Figure 7-10 Palaeotopography (WCDM, 2005)

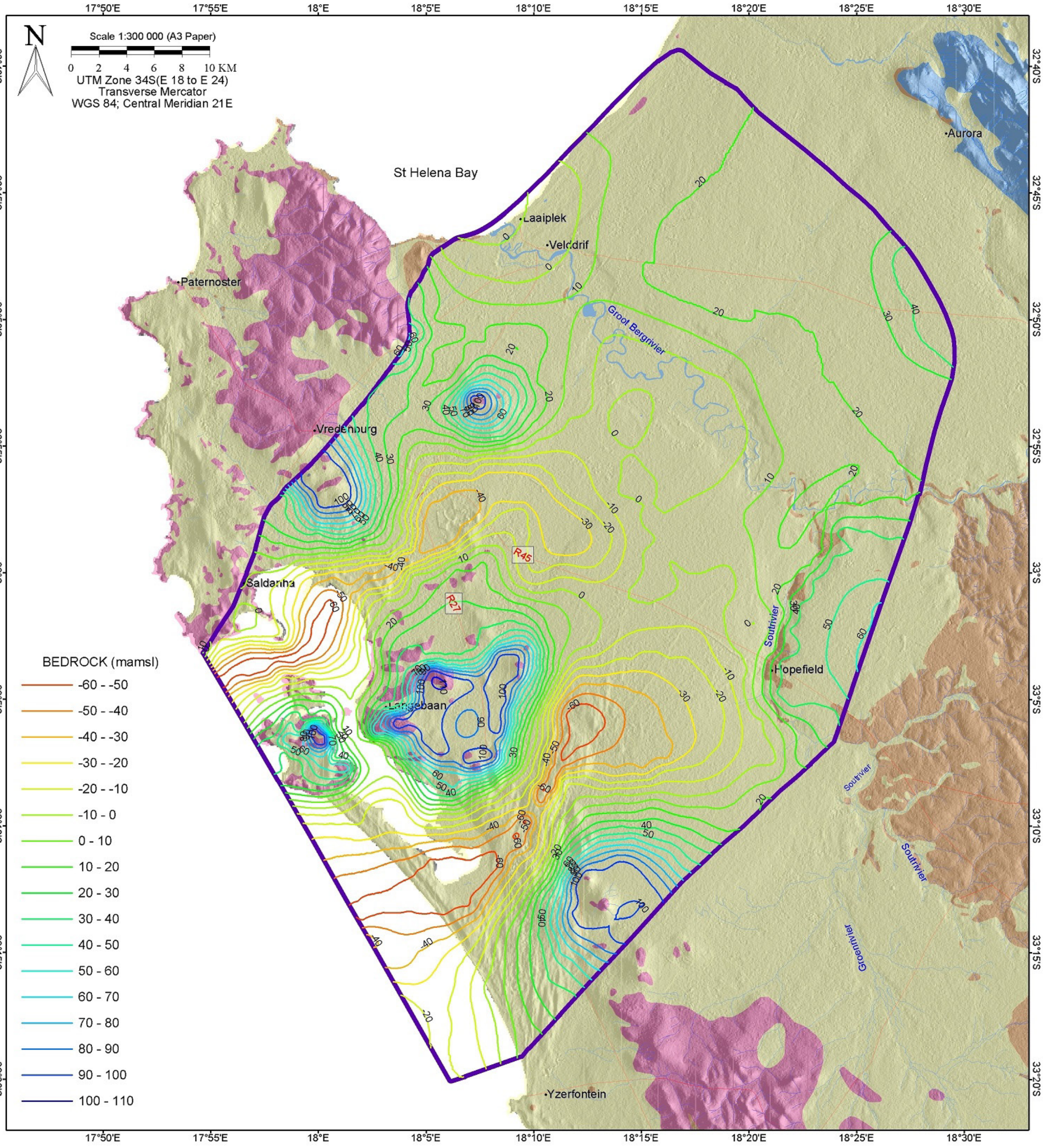


Figure 7-11 Palaeotopography (DWAF, 2008c)

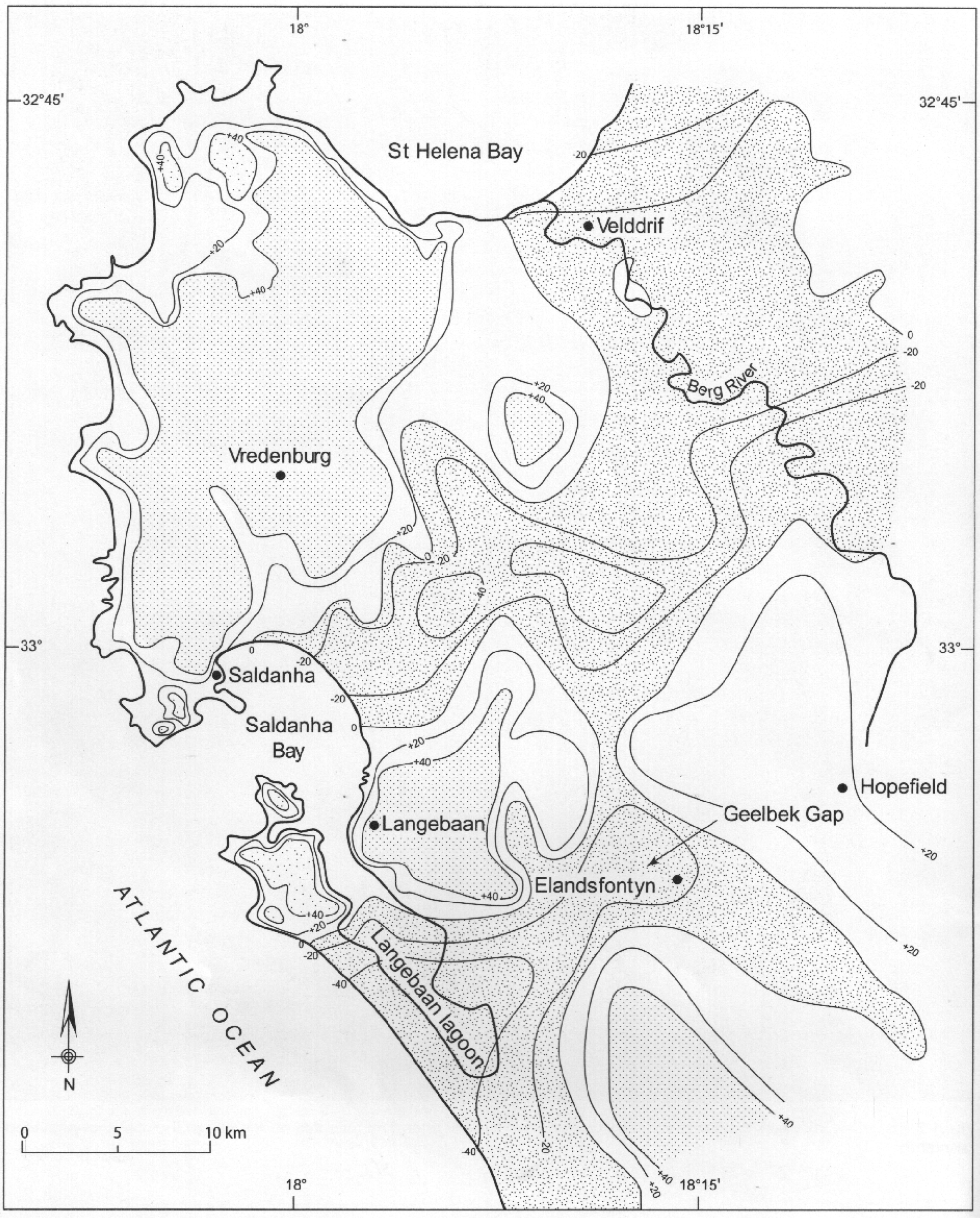


Figure 7-12 **Palaeotopography (Roberts & Siegfried, 2014)**

7.7. AQUIFER SYSTEMS

7.7.1. Aquifer Distinctions

The area of the Langebaan palaeochannel (the northwest palaeochannel) and the area of the Elandsfontein palaeochannel (the southeast palaeochannel) coincide with thick water-bearing sedimentary sequences. They have been named the Langebaan Road Aquifer System (LRAS) and the Elandsfontein Aquifer System (EAS), respectively (by Woodford et al, 2003). In addition, the aquifers northeast of the Berg River have been named the Adamboerskraal Aquifer System (AAS).

Woodford and Fortuin (2003) further subdivided the LRAS and EAS into 17 groundwater management units (GMUs), which were considered zones of similar groundwater quality, hydraulic properties, flow regime, etc. within, but different to each other and can therefore be managed separately. In several reports that use these subdivisions, the following disclaimer is given 'It must be noted that, due to the complex and multi-layered nature of both the primary and the underlying fractured-rock aquifer systems, groundwater may flow from one management unit to another via preferential flow paths (i.e. faults)' (WCDM, 2009). These subdivisions are not utilised in this study, as they are not considered aquifer boundaries from a hydraulic point of view. The water level and piezometric map show the hydraulic continuity between the so-called GMUs (section 7.8), and to consider each one separately confuses the consideration of the natural and impacted total aquifer flow balance. The boundaries of the LRAS, EAS and AAS are also not considered hydraulic boundaries, but these names are useful to distinguish the geographic areas and are utilised in this report.

The stratigraphy presented (Table 7-5) translates to a complex hydrostratigraphy of sand aquifers of varying permeability, and semi-confining peat and clay layers. Previous literature shows that this complex hydrostratigraphy can be represented on the regional scale by three key units, each present in each of the three aquifer systems (geographic areas):

1. The basal gravels of the Elandsfontyn Formation form the semi-confined lower aquifer unit, (LAU). This is the most important aquifer, most significantly in the area of Langebaan Road where it is thick (up to 60 m) and has larger aerial extent. Due to its depositional history, the aquifer is restricted to palaeochannels.
2. The aquitard clay layer of the upper Elandsfontyn, which acts to (semi) confine the basal aquifer.
3. The variably consolidated sands and calcretes, with interbedded peat clay, of the remaining Sandveld Formations, which form the upper unconfined aquifer unit (UAU) (DWAF, 2008c).

Groundwater flow between the Cenozoic aquifers and the underlying basement (weathered and fractured aquifers) has been assumed to be minimal, due to the weathered, clay-rich, upper layers of the basement (Timmerman, 1988, DWAF, 2008c). However, WCDM (2009) documents exploration drilling (i.e. G46030, GG46105 in the wellfield) that specifically targeted the bedrock aquifer and has 'shown that this assumption is incorrect'. The municipal production boreholes in Hopefield apparently target high-yielding fractures in the Malmesbury rocks which 'drain' groundwater from the overlying primary aquifer (WCDM, 2009).

Although high groundwater yields (greater than 10 ℓ/s) have been documented in the underlying fractured bedrock (WCDM, 2009), these areas of potential higher permeability are considered locally relevant (possible conduits to lateral flow, and likely to be harvesting water from the sedimentary aquifers above), but regionally insignificant for the flow regime in the Cenozoic aquifers.

The assumption that the complex geological layering can be represented by three aquifer (or hydraulic) layers is supported by several numerical modelling studies. Each of these achieved a representation of the natural system reasonably well with a three-layered numerical system (Timmerman, 1988, WCDM, 2009, DWAF, 2008c).

Woodford (2002) and Woodford et al (2003), mapped the thickness of the various aquifer units. They indicated that the Cenozoic Sands reach a maximum thickness of > 100 m at Anyskop, and the Elandsfontyn Formation gravels (LAU) reach a maximum thickness of 60 m in the deepest sections of palaeochannels, with their overlying clays (aquitard) reaching a maximum thickness of 50 m (Woodford et al, 2003). The interpreted extent of each geological unit, and therefore aquifer, is closely linked however to the interpretation of the basement topography.

7.7.2. Distribution of LAU

According to DWAF (2008c), the basal gravels of the Elandsfontyn Formation reach a stratigraphic height of up to -5 mamsl. Because of the reported basement rise to 0 mamsl between the EAS and the LRAS, DWAF (2008c) considers the LAU to be discontinuous between the two palaeochannels, and discontinuous beneath the Berg River. It was this basement rise and discontinuity of the LAU that led Timmerman (1985c) to propose a 'no-flow' boundary between the LRAS and EAS (at least for the LAU). However, recent exploration drilling (G46030 and G46031) has indicated that this basement high is either absent or localised, and a palaeochannel may have developed during the early Miocene along the granite-shale contact (WCDM, 2009). If so, the LAU of the two aquifer systems are in some degree of hydraulic connection.

Timmerman (1988) showed that the LAU is discontinuous towards the mouth of the LRAS palaeochannel (as per his basement interpretation). However, DWAF (2008c) questioned this, pointing to NGDB boreholes within 4 km of the coast that intersected the basal clay and gravel of the Elandsfontyn. In line with the arguments in support of continuous LRAS and EAS palaeochannels towards the coastline, it seems reasonable to assume that the basal gravels of the LAU are also continuous.

7.7.3. Distribution of Clay Aquitard

DWAF (2008c) considered that the clays underlie a wide area over the centre of the LRAS and EAS and north of the Berg River, having been deposited over all areas of the basal gravel (i.e. the LAU is confined everywhere). The clay formation has been eroded along the lower course of the Berg River itself, and within approximately 5 km of the southwest coastline, according to Timmerman (1988), DWAF (2008c) – although DWAF (2008c) documents a borehole approximately 4 km from the coast that contains basal gravel and overlying clay. Given the assumption that the palaeochannel and LAU is continuous towards the coast, the LAU and UAU must be in hydraulic connection (no longer confined) within approximately 4–5 km from the coast. DWAF (2008c) assumed in their numerical modelling that the clay layer could be represented by one layer of 20 m average thickness.

More detailed mapping of the uppermost clay unit of the Elandsfontyn Formation shows the clay unit varies in thickness and in places exceeds 40 m (Figure 7-13, WCDM, 2005). WCDM (2005) does caution, however, that although the map suggests continuity of the clay layer over palaeochannel areas and hence over basal gravel, the clay layer is not always present and in these cases the LAU and UAU would be indistinguishable (and in hydraulic connection). This statement requires that either the LAU is present outside of palaeochannels (not likely given its depositional history), or that, contrary to the map, there are localised areas where the clay is absent. WCDM (2005) also suggest that some degree of hydraulic connection exists between the Upper and Lower Aquifer Units where the aquitard thickness is less than 10 m, and WCDM (2009) suggest the same but for thicknesses of less than 5 m.

The map of the clay distribution maintains some uncertainty around the extent of the clay, with the thinnest colour flood indicating '< 5 m' thickness. This uncertainty appears to have been removed in a later isopach map (WCDM, 2009, shown in Figure 7-14); however, there is no supporting information for the map (how it was generated/updated, use of newer drilling information, or conceptual interpretation). The clay layer appears to have been eroded for distances much greater than 5 km from the coastline.

The municipal wellfield at Langebaan Road targets the LAU, and agricultural users generally abstract from the UAU. Landowners have been concerned about a potential connection between the units, relating this to potential impact on their available groundwater resources. However, monitoring results to date have shown the clay layer is a competent aquitard and no transmission of impact or hydraulic connection has been detected (WCDM, 2008, 2009, 2012).

Similar to the basement topography discussion, given the potential control that the position and thickness of the clay layer has on the hydrogeology of especially the LAU, it was deemed necessary to use all available information (including raw data) to determine the thickness of the clay layer. In addition, the WCDM (2009) data shows clay thickness only, not elevation, hence original raw borehole information is required for model input. A total of 8401 NGA lithology entries (representing 2340 boreholes or geosites with lithology data) were processed to identify boreholes that penetrated a clay layer thicker than 4 m (chosen in order to sift out potentially discontinuous clay lenses and isolate the upper Elandsfontyn formation clay), resulting in 341 boreholes identifying the top of the clay layer, and 212 boreholes identifying the base of the clay layer. This NGA data was used in combination with WCDM (2009) to generate the clay elevation top, and clay thickness, through the following procedure:

- The clay thickness, calculated from the NGA analysis (top elevation minus bottom), agreed remarkably closely with the WCDM (2009) map, with some local variation and need for smoothing (as would be expected). Therefore it is assumed that the thickness of WCDM (2009) was developed through a similar analysis, and it was therefore adopted, but augmented with NGA data where the model boundary is greater than the area covered therein.
- The elevation of points from the NGA analysis was used to set the elevation of the top of the clay layer, with the thickness used to set the elevation of the base.

7.7.4. Distribution of UAU

The UAU can, at the regional scale, be considered a single unconfined aquifer. Timmerman (1985c) cautions that, at the local scale, the UAU, especially in the EAS, is a very complex succession of up to four aquifer-aquitard layers.

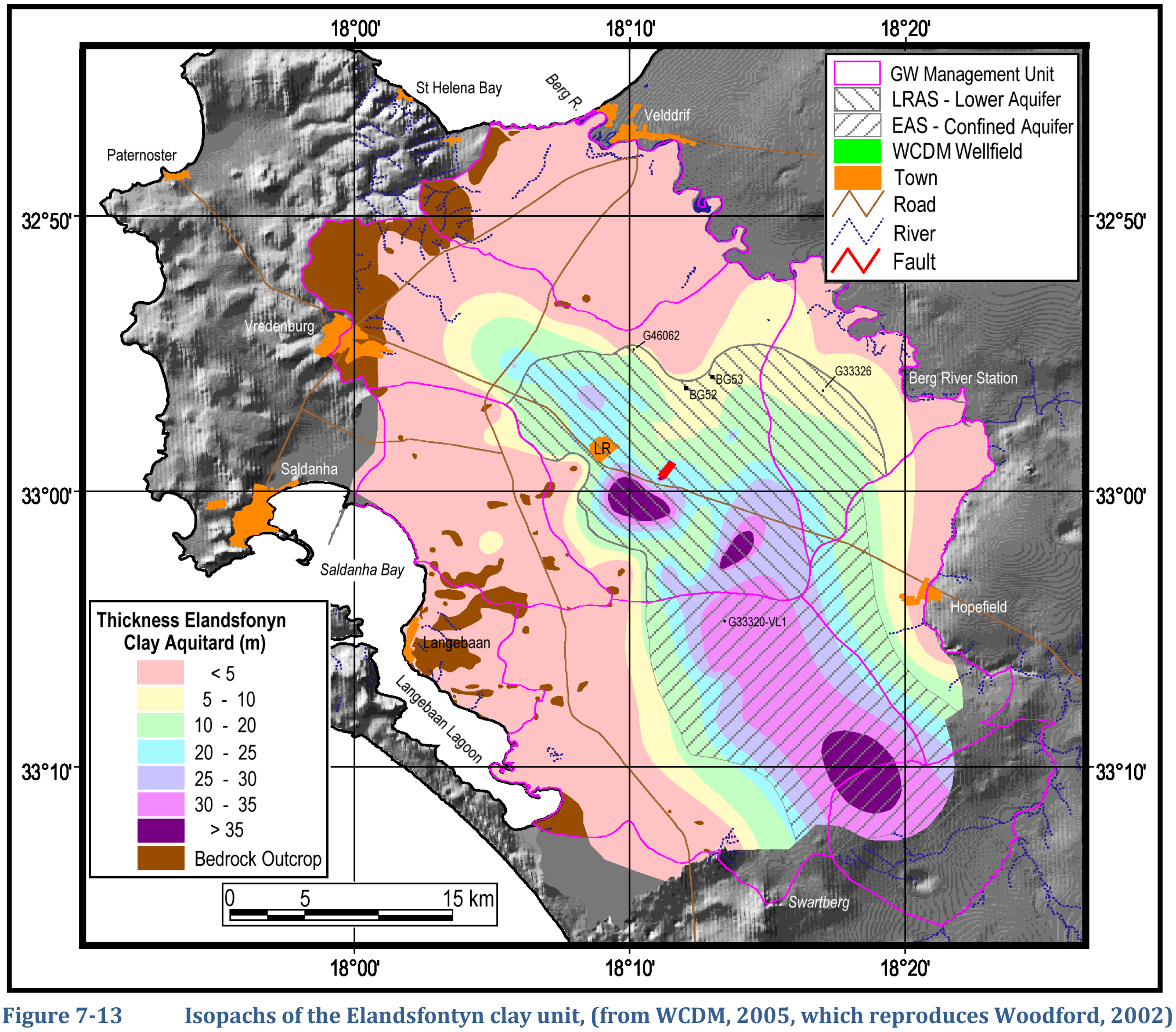


Figure 7-13 Isopachs of the Elandsfontyn clay unit, (from WCDM, 2005, which reproduces Woodford, 2002)

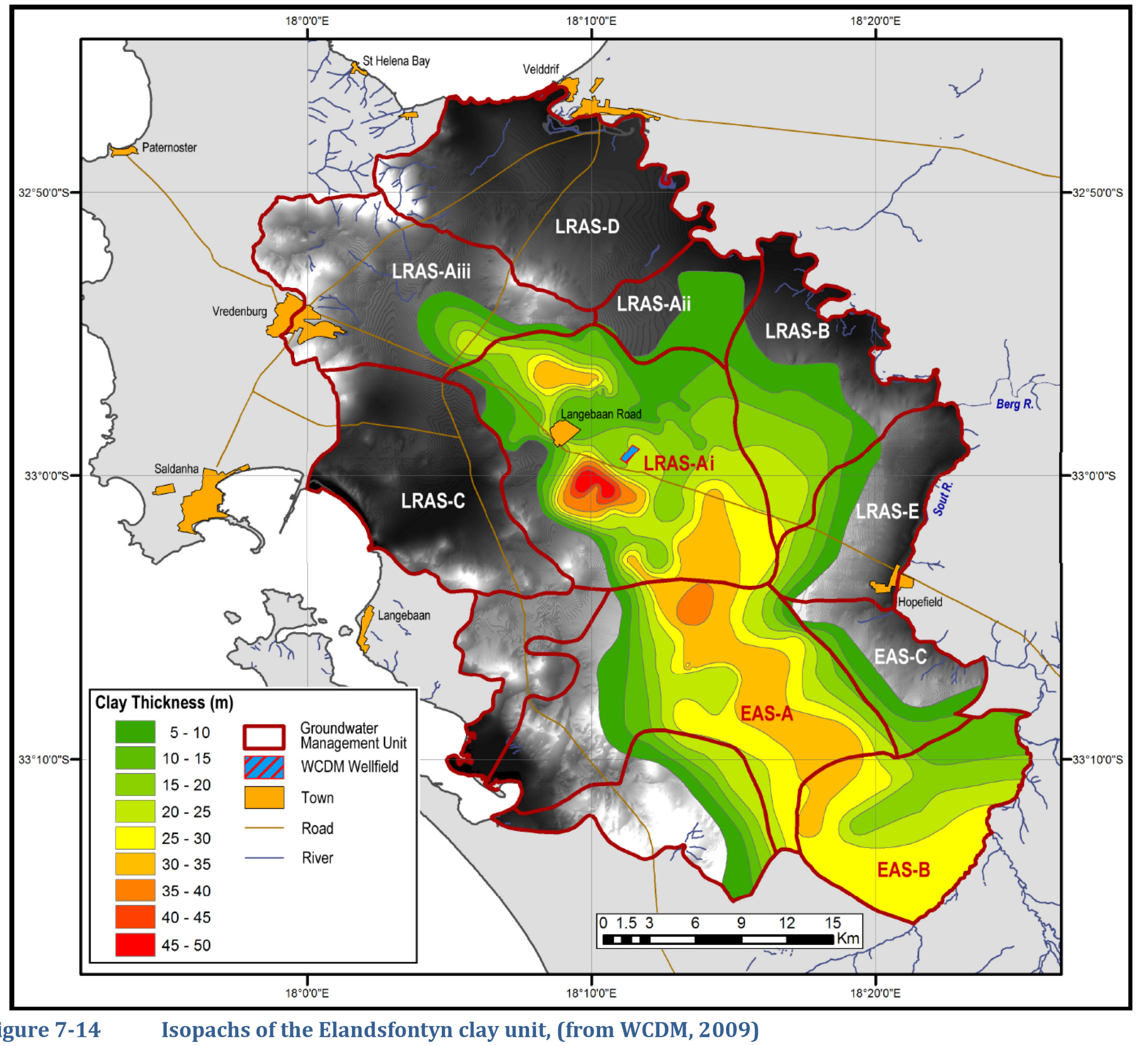


Figure 7-14 Isopachs of the Elandsfontyn clay unit, (from WCDM, 2009)

7.8. Groundwater Levels and Flow directions

Although groundwater was used in the region prior to the start of municipal abstraction in November 1999, this abstraction was spatially distributed, and as the aquifer has a high hydraulic diffusivity, it is considered to be at dynamic equilibrium prior to November 1999. A water level dataset of approximately 1000 water level points, representing dynamic equilibrium or steady-state, was developed based on the following datasets:

- Water levels from 1950 to November 1999, contained in the National Groundwater Archive (NGA):
  - Boreholes with only a single measurement prior to 1999 were considered lowest confidence.
  - For those boreholes with more than one water level record prior to 1999, the water level data was averaged. Those points with more than 12 readings were considered highest confidence.
  - All boreholes with a high range in water levels were investigated and several of these discarded as erroneous data, or impacted by local pumping.
- A borehole inventory containing 174 boreholes with 'baseline' water level data is included within Appendix 1 of DWAF (2003). This baseline reflects the 'monitored water level in November 1999 or earlier, prior to commencement of pumping in the WCDM Wellfield' (DWAF, 2003). Of these boreholes, 32 were not included in the NGA (or could not be matched to entries in the NGA through coordinates or names), and hence were added to the water level dataset.
- Data from all boreholes routinely monitored by DWS was sought (i.e. HYDSTRA database). Three of these boreholes had data prior to 1999 and were not incorporated in the NGA dataset, hence were added to the dataset.

- Data from the WCDM was sought, however all of this data was also captured in the NGA.

Both DWAF (2008c) and WCDM (2009) present water level maps using combined data from the UAU and LAU, due to a reported lack of information with which to attribute the water levels to separate aquifers, and because the assumed degree of hydraulic disconnection is small (< 0.5 m). However, the NGA lithology dataset developed for the clay layering enables some boreholes in the water level dataset to be ascribed to an aquifer, through the following procedure:

- Comparing the borehole locations with the geological map, those boreholes located in basement outcrops were ascribed as such.
- Those boreholes (with water levels and lithology data in NGA), whose lithology data showed they penetrated the base of the clay layer and then penetrated sand/gravel, were ascribed to the LAU.
- Those boreholes that did not pass through the clay base, but were located in the Cenozoic sand according to their NGA lithology data, were ascribed to the UAU.
- Various previous reports also ascribe boreholes to particular aquifers (DWAF, 2003, WCDM, 2008, 2011, 2012) and this information was also used.
- The remaining boreholes (i.e. located within sand, but without any lithology information in NGA or in previous reports) were listed as unknown, given there is no lithology information available to show they penetrated through a clay layer to the basal gravels. The majority of the database falls in this category.

Uncertainties remain with regard to assigning an aquifer per borehole, as, although the available lithology data may show that the borehole penetrates to the LAU, there is no information on the borehole construction. Nevertheless it is possible to detect slight differences in the datasets, as indicated below. Although the WCDM wellfield penetrates the LAU, it is considered likely that most private boreholes would target the UAU, hence in the analysis below boreholes that have 'unknown' aquifer are grouped to the UAU.

Table 7-6 indicates that the water table is deepest in the basement aquifers (9.1 mbgl), and similar in the UAU and LAU, with the LAU being slightly deeper (6.4 mbgl as compared to 6.1 mbgl or to 5.2 mbgl, depending on inclusion of unknown boreholes). The range of water levels is greatest in the basement aquifers (2.5 m) and relatively low in both the LAU (0.6 m) and UAU (0.7m or 1.1 m).

Water levels (in mamsl) strongly correlate with surface topography in all aquifers (Figure 7-15, Figure 7-16, Figure 7-17), with almost identical behaviour between the LAU and UAU. The correlation coefficient is highest in the UAU (0.9951), followed by the LAU (0.9631), and lastly by the basement which shows greatest variability (0.9444), with some points falling far below the linear correlation (i.e. deeper water levels). Nevertheless, the correlation coefficients of above 0.9 are considered high. Because of the similar behaviour between each aquifer (all correlating with topography), and the similar gradients in Figure 7-16, (similar depth to water), it can be assumed that at regional scale, there is some hydraulic connectivity between the different units.

The piezometric map developed (using all data, Figure 7-18) illustrates the correlation with topography. Groundwater (at least in the UAU) flows from the water level high located in the south (around the junction of G10M, G10L, and G21A) in a semi-radial direction towards the northwest, towards the surface water drainage systems in the northeast, and towards the Langebaan Lagoon in the southwest. The flow to the north crosses from the area of EAS to the LRAS, hence the two systems are in hydraulic connection for at least the UAU. This flow regime is interrupted by the several basement koppies which form local groundwater highs. Groundwater northeast of the Berg River flows southwest, towards the Berg and the coastline. A groundwater flow divide roughly coincides with the quaternary catchment boundary between G10L, and G10M. This catchment/groundwater flow divide was used as a model boundary in previous modelling efforts (WCDM, 2009), however, the G10M area is included here to incorporate flow within the LAU which may derive from G10M, and because the flow divide may move with abstraction.

DWAF (2008c) developed a piezometric map (for all data, not aquifer-specific), and considered it possible to detect the behaviour of two aquifer units, suggesting that the flow in the LAU is 'controlled by the bedrock topography', with flow concentrating along the axis of the LRAS and EAS palaeochannels towards their mouths on the southwest coastline.

Deep groundwater will preferentially flow within the high hydraulic conductivity sediments of the basal gravels which form the LAU, and which are only found in the palaeochannels; hence the ‘control’ from basement topography (essentially the control is from the sediments within the basement depressions). The map shows a low in water levels in the region of the LRAS palaeochannel, interpreted by DWAF (2008c) to reflect the higher hydraulic conductivity LAU, and potentially the abstraction from the LAU.

Woodford and Fortuin (2003) point out that the degree to which the LAU is confined is likely to be highly variable across the area, relating to the thickness of the clay layer. There are no known multi-level piezometer measurements available. DWAF (2008c) makes the following observations on the degree of confinement:

- Water levels 11 m higher in the LAU than UAU, close to the palaeochannel mouth in the EAS, an observation from two NGDB (now NGA) boreholes in close proximity
- Water levels 0.86 m higher in the LAU than UAU, 4 km inland from Saldanha Bay in the LRAS, also an observation from two NGDB boreholes in close proximity
- An interpreted ~3 m water level difference (LAU higher than UAU) at the West Coast Fossil Park, located at Langebaan Road in the centre of LRAS.

Figure 7-16 indicates that, on average, there are slightly lower water levels in the LAU compared to the UAU (0.8 m lower), which would suggest that on average, there is a downward gradient from UAU to LAU. The above-listed positions with higher water levels in the LAU may be related to areas of thicker clay and not representative of the entire aquifer. (Although the data for clay thickness towards the LRAS and EAS mouth is insufficient to support this, this is true for the third location at Langebaan Road). If on average there is a downward gradient, this provides a recharge mechanism (section 7.10.1).

Piezometric maps using only data from the LAU or UAU are shown in Figure 7-19 and Figure 7-20 respectively, for the area with sufficient borehole information. Furthermore, Figure 7-21 shows the difference in piezometric heads between the UAU and the LAU. In the south and southeast of the area covered, the heads in the UAU are approximately 10 m higher than in the LAU. The gradient reverses to the north, with heads in the LAU up to 10 m higher than in the UAU. The aquifer-specific water level data is generally insufficient to be conclusive. Degree of separation aside, the response to abstraction has shown that the clay layer is at least a competent aquitard (section 7.13).

Table 7-6 Median depth to water and range of water level readings, per aquifer

Aquifer/group of water level data	Median depth to water (m)	Number of records for depth to water	Median range[9] of water levels (m)	Number of records for range
All data	6.0	1017	1.08	194
Basement	9.1	207	2.5	3
LAU	6.4	92	0.6	26
UAU	6.1	30	0.7	9
UAU including unknown	5.2	718	1.1	165

[9] Range is calculated from the maximum and minimum of all records per borehole, and thus does not represent an average annual variation

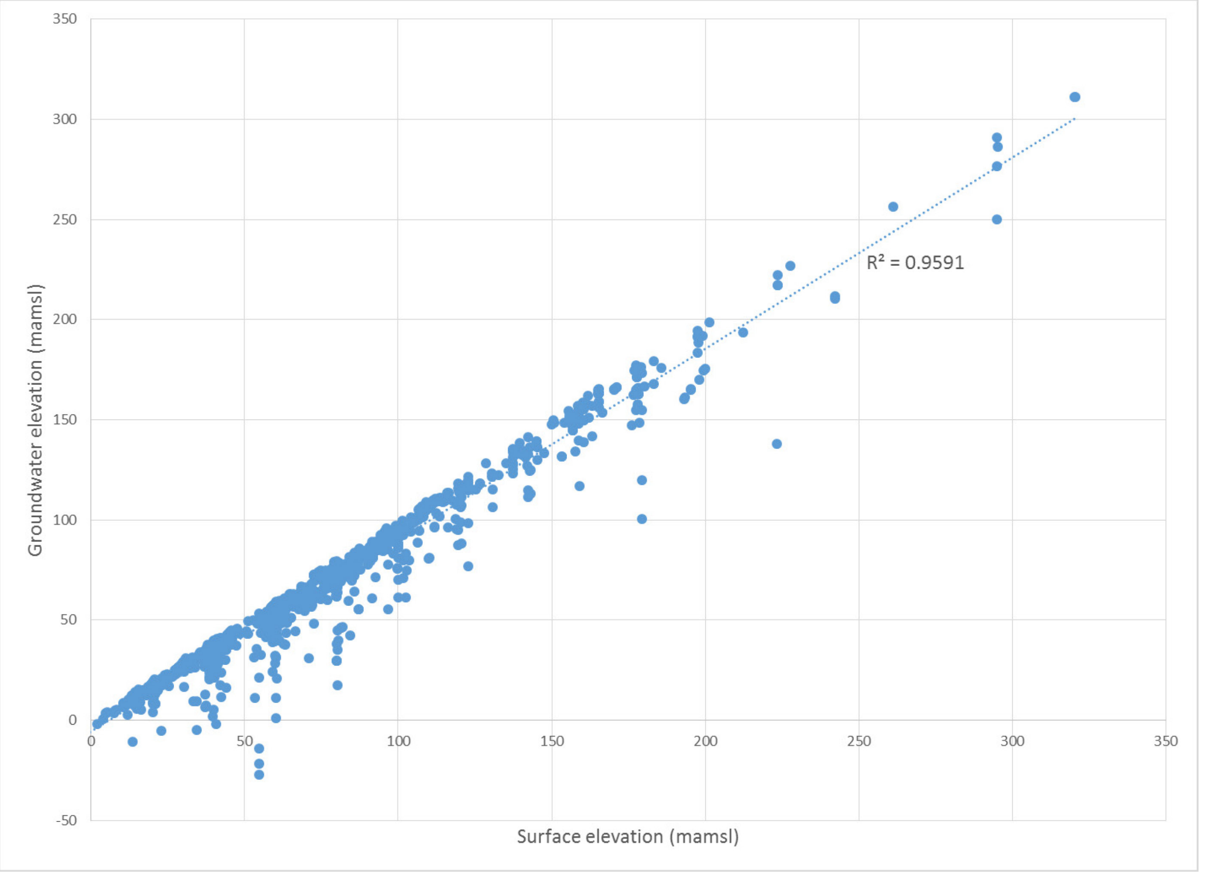


Figure 7-15 **Graph of groundwater level and surface elevation (all boreholes)**

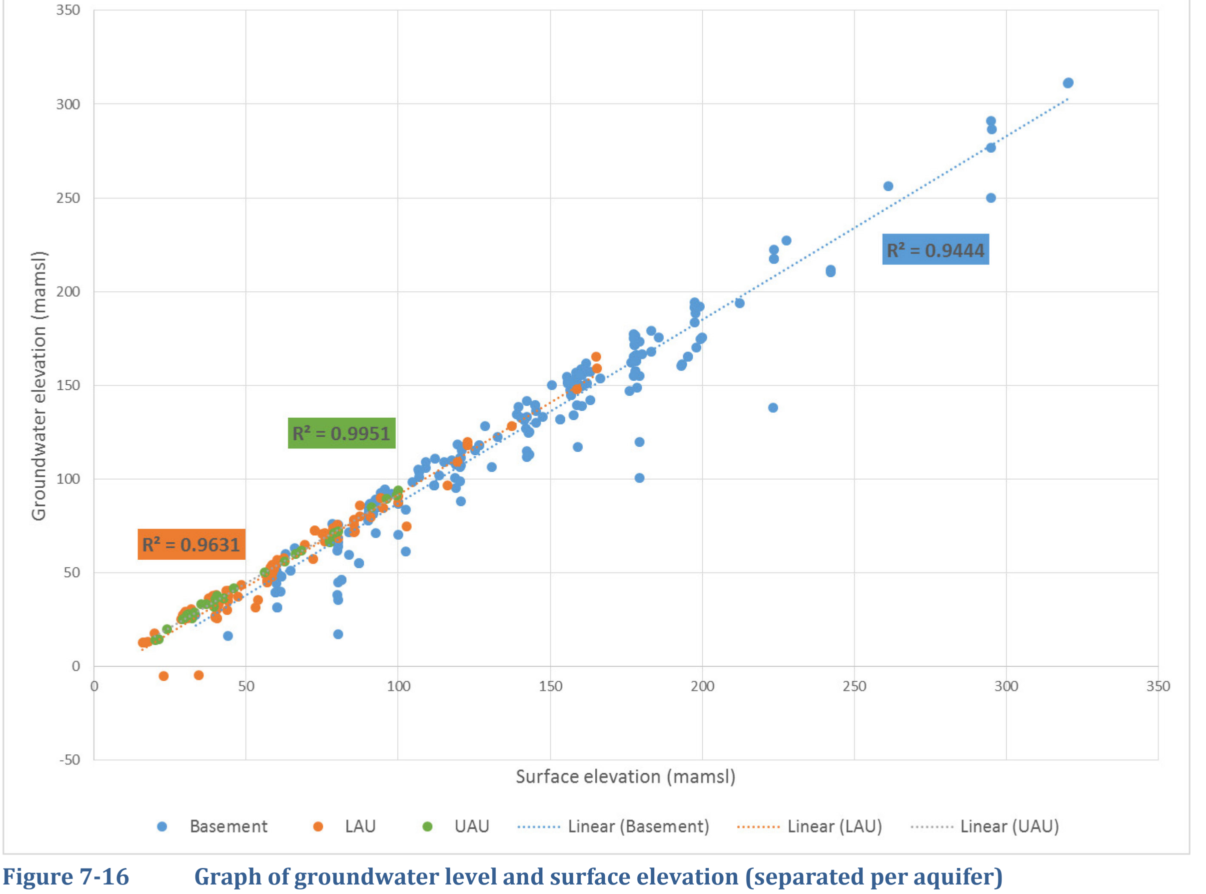


Figure 7-16 **Graph of groundwater level and surface elevation (separated per aquifer)**

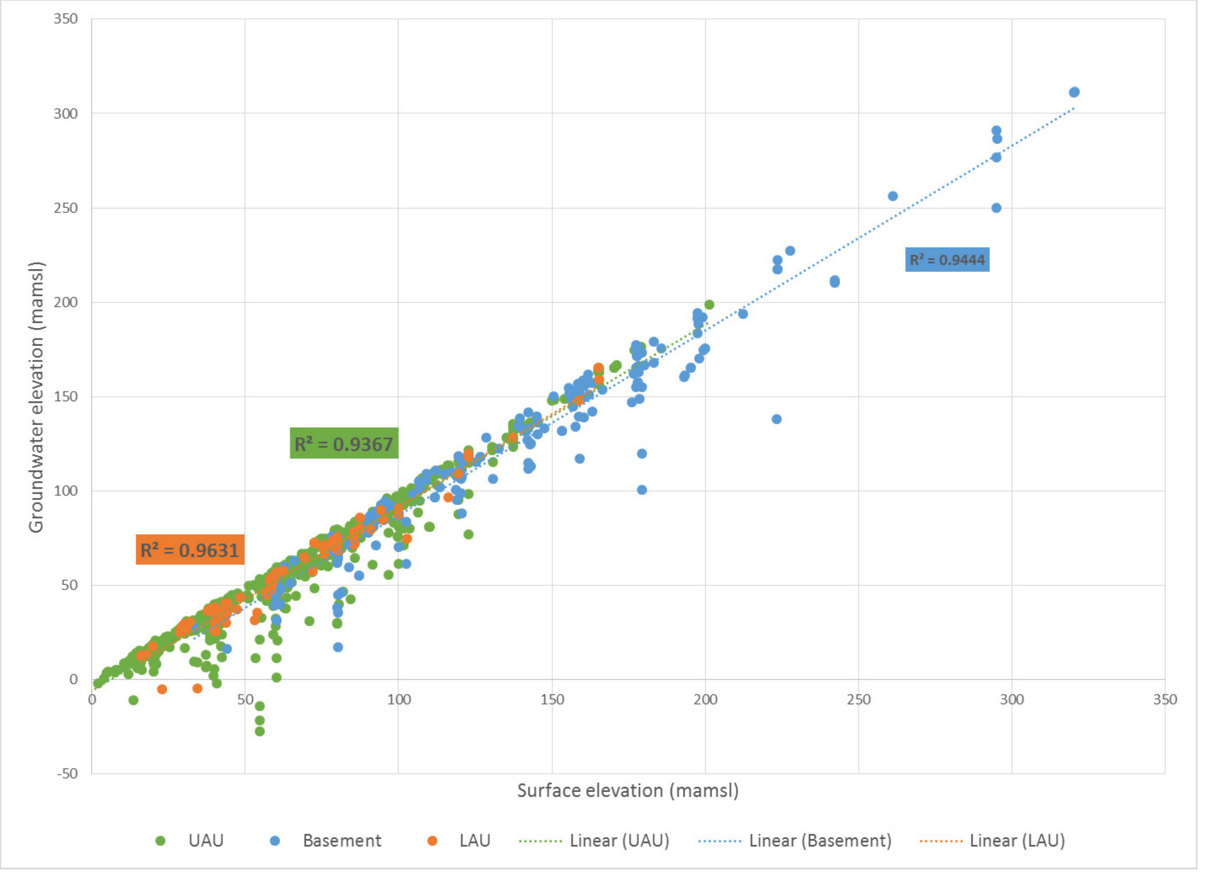


Figure 7-17 Graph of groundwater level and surface elevation, separated per aquifer. (Points with unknown aquifer are grouped here under UAU.)

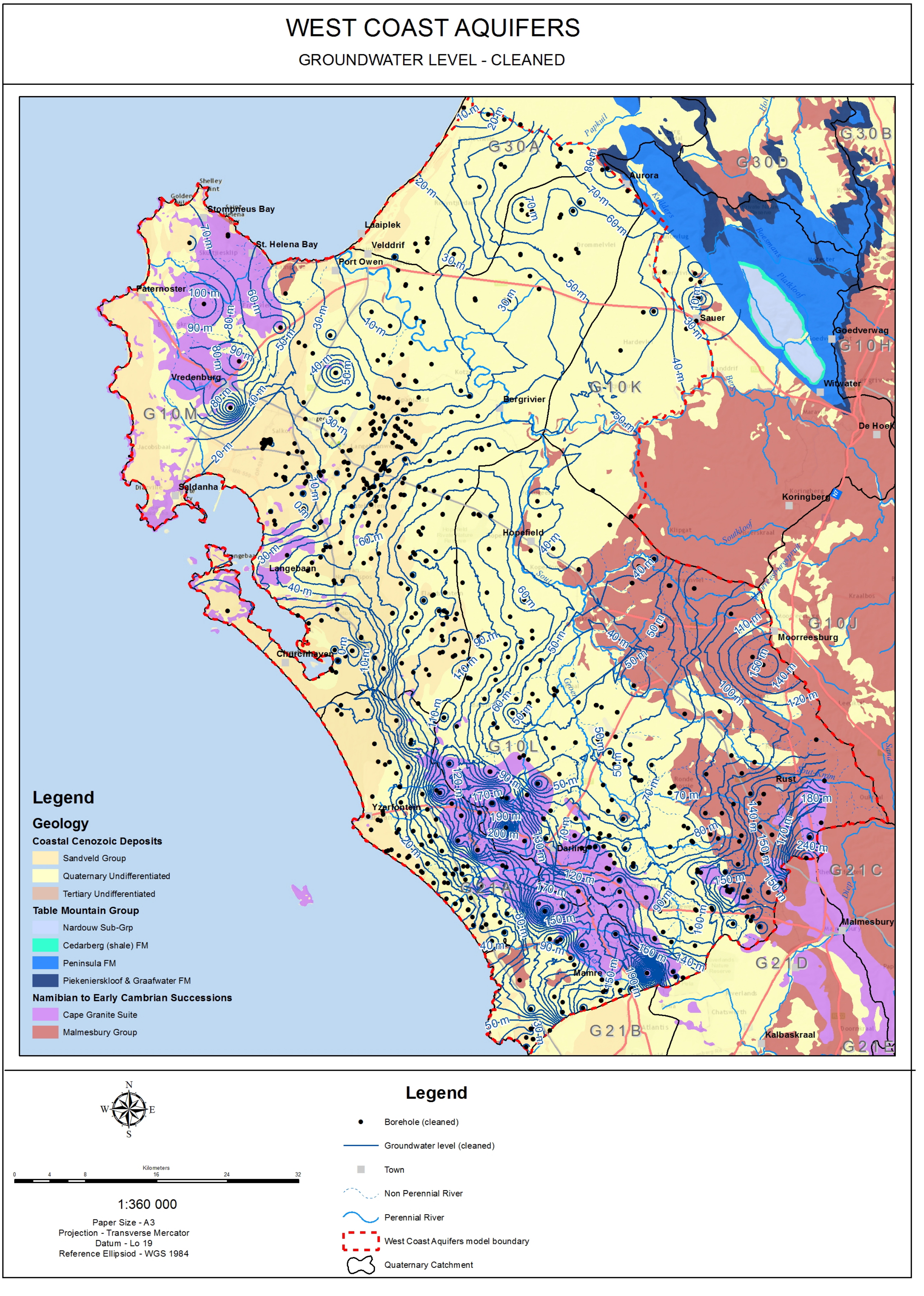


Figure 7-18 Water level map (all boreholes)

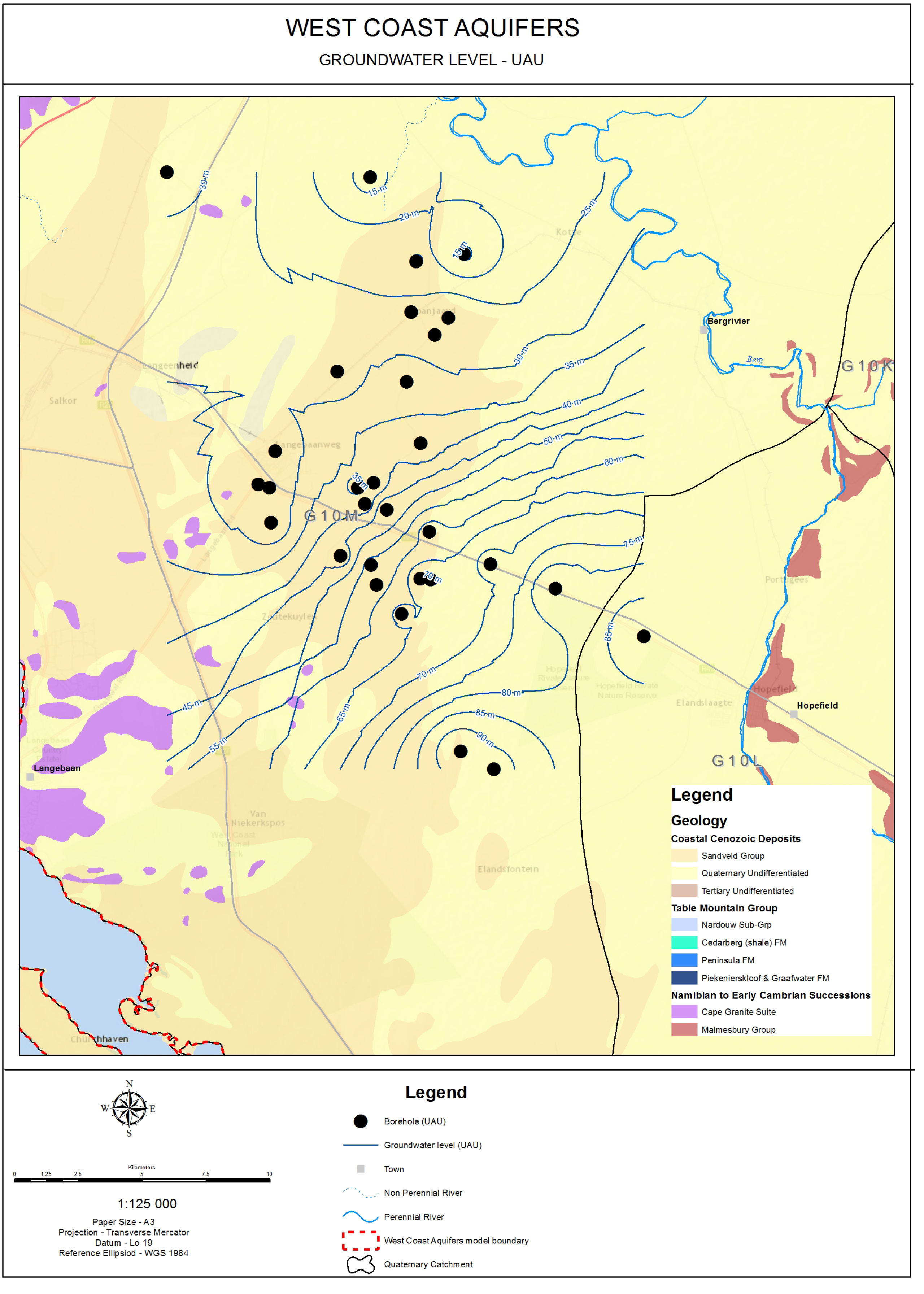


Figure 7-19 **Water level map (UAU)**

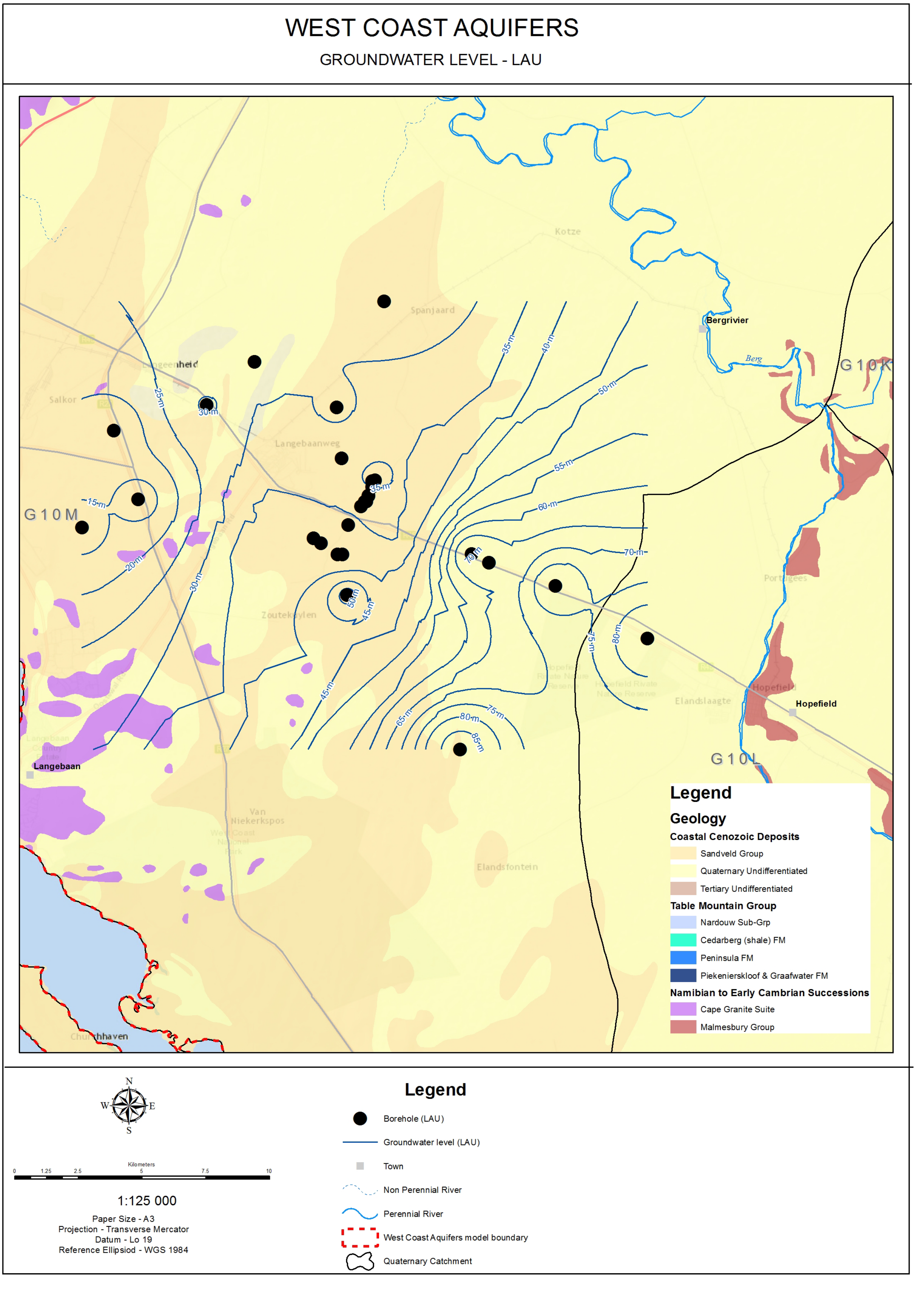


Figure 7-20 Piezometric map (LAU)

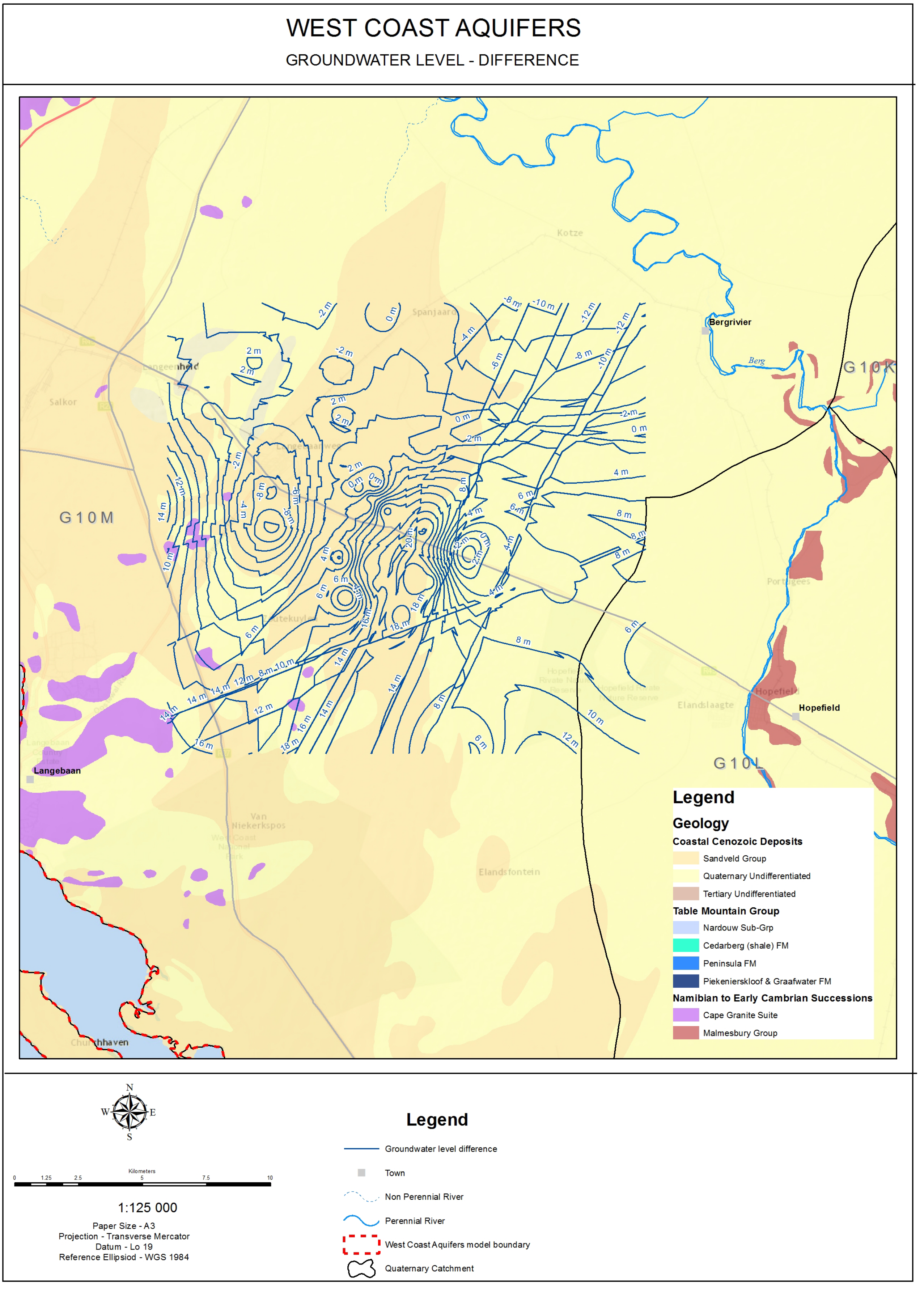


Figure 7-21 Map showing UAU water level minus LAU piezometric level

7.9. HYDRAULIC PARAMETERS

Selected previous estimates of hydraulic properties, from a combination of pump tests and numerical model outputs, are summarised in Table 7-7. The pump test results from Timmerman (1985b) should be taken as indicative only, as Timmerman (1985b) reports that several issues affected the tests (incorrectly installed gravel packs and sand ingress). Pump tests carried out by Parsons and Associates (2006 and 2014), show hydraulic conductivity ranging from 8 to 24 m/d, and also up to infinite (or rather greater than can be detected by the relatively low-stress test, at 6 ℓ/s), as a stable horizontal drawdown almost immediately established.

The model of WCDM (2009) allocates a high hydraulic conductivity (i.e. a distribution of the LAU) only in a central depression under the WCDM wellfield and not extending to the coast at Saldanha Bay, motivated as so because this distribution includes the boreholes that respond rapidly to pumping at the wellfield. In contrast, the model of DWAF (2008c) calibrated with a hydraulic conductivity of 10 m/d in the basal layer, everywhere across the model (i.e. no high hydraulic conductivity layer only existing in palaeochannels). It is proposed that a high hydraulic conductivity across the region 'may represent addition of water to the layer that in reality may occur through discrete positions where the clay is absent, areas which aren't modelled' (DWAF, 2008c).

DWAF (2008c) notes that the modelled hydraulic conductivities for the LAU are not as high as one would expect from the geology (although other model results do calibrate with higher hydraulic conductivities). Furthermore, several of the modelled hydraulic conductivities are lower than those from pump tests. Timmerman (1988) commented that the modelled hydraulic conductivity is 5 m/d compared to a pump test-derived hydraulic conductivity of 100 m/d at the same location. These results highlight that a numerical model operates on numerical layers that cannot necessarily translate directly to geological ones.

Table 7-7 Hydraulic property estimations

	Kx Ky (m/d)	Kz (m/d)	S (-)[10]	Sy (-)	Source
UAU	<5 - 60				Timmerman (1985b, cited in DWAF, 2008a)
	100			0.13	Timmerman (1988)
	8 - 24				Parsons and Associates (2006, 2014)
	0.5 - 7	0.005 - 0.0005		0.07 - 0.2	Timmerman (1988)
	1 - 10	0.1 - 1	0.02		DWAF (2008a)
	3.0	0.3	0.0004 - 0.001	0.13	WCDM (2009)
	17	16.67	0.00036	0.15	GEOSS (2014, cited in SRK, 2015)
	1.8 and 4	0.85 and 0.52	0.00057	0.13	GEOSS (2015)
	1.08	1.08		0.2 - 0.25	SRK (2015)
	2.6 - 5.1				Seyler et al (2016) (for Cape Flats)
Clay	0.01 - 2.0	0.005 - 0.0005	Ss= 1E-5 - 1E-3		Timmerman (1988)
	0.01	0.001	0.02		DWAF (2008a)
	1E-4 to 0.01	1E-5 - 0.001	Ss 1E-3		WCDM (2009)
	1.66E-3	1.66E-3	0.0007	0.02	GEOSS (2015)
LAU	12.5 - 60		1E-3 - 3.1E-3		Timmerman (1985b, cited in DWAF, 2008a)
	0.5 - 70	0.0005 - 0.005	Ss = 3E-4 - 5E-3		Timmerman (1988)
	10	0.1	0.02		DWAF (2008)
	2.0 or 30	0.01 - 0.25	Ss = 6E-6 - 3E-4		WCDM (2009)
	20	60	0.001		GEOSS (2014, cited in SRK, 2015)
	2	1			GEOSS (2015)
Malmesbury Fm	0.061 - 0.691	0.0061 - 0.0691			Seyler et al (2016) (for Cape Flats)

KEY	
(grey)	Non-differentiated result i.e. representative of grouped Sandveld
(orange)	Pump test result
(blue)	Numerical model calibration result

[10] S refers to Storage parameter (Ss x b (-)) unless where specified as Specific storage (Ss, m^{-1})

7.10. Sources and Sinks

7.10.1. Recharge

Recharge to at least the unconfined UAU is derived from rainfall. Rainfall is greatest in the higher lying areas, and in the south of the area of interest (Figure 7-3), and thus a recharge mound has been postulated in the south of the area. This recharge mound may also be related to lower permeability sediments in the region (as suggested by Timmerman, 1985c). Recharge will also be concentrated around the contacts with granite koppies, due to enhanced runoff over these less permeable surfaces (DWAF, 2008c, Timmerman, 1988).

Due to lack of outcrop of the Elandsfontyn Formation and its separation from the UAU by the clay layer across most of the interior, the recharge mechanism to the LAU is more complex, and more open to interpretation. Timmerman (1985c) suggests that the low permeability UAU sediments around the recharge mound (southwest of Hopefield) would facilitate the downward percolation of the recharged water into the LAU, and then the lateral movement of this water, under increasing confining pressures to the north and northeast, in a north-westerly direction towards the LRAS via a 'piston-flow' mechanism. This suggestion is broadly supported by the mapped clay thickness, i.e. it is thin around the water level high in the south (at the junction of G10L, G10M, and G21A), and thicker to the north and northeast (Figure 7-14). For downward percolation to occur, the water levels in the UAU must be greater than those in the LAU, which is supported by the sparse dataset (section 7.8). Timmerman (1988) suggests that during times of high rainfall, recharge to the LAU can percolate down, which is recorded in response to the high rainfall of 1977, after which water levels in the LAU responded with a recharge signal and then declined until another high rainfall year in 1983. DWAF (2008c) conclude that recharge to the LAU must simply occur through leakage via the UAU in areas where the head difference between upper and lower aquifer is large enough to drive vertical recharge downwards. Water quality is notably different between the LAU and UAU (section 7.11) and the water quality difference has been cited as a reason that this recharge mechanism may not occur. However, the cation exchange capacity of the clay has not been assessed and could easily be responsible for altering the water quality after vertical seepage. The groundwater quality of the LAU is known to deteriorate towards the west, suggesting recharge from the overlying UAU where the clay becomes thinner (Woodford et al, 2003). The conceptual interpretation of recharge to the LAU, in areas where clay is thin or missing and there is a high head in the UAU, is illustrated in Figure 7-22.

The WCDM (2009) indicate that mean groundwater residence times in the UAU are between 30–60 years from tritium analysis, and from recent to 9 000 years from Carbon-14. Residence times in the LAU range from 5 000 to 28 000 years (Weaver & Talma, 2000, 2003, Tredoux & Talma, 2009, cited in DWA, 2010b and WCDM, 2009). It has been suggested therefore that flow rates in the LAU are exceptionally low, despite the relatively high hydraulic conductivity values (30–40 m/d), 'due to a flow constriction at the outlet of the system' (Waver & Talma, 2000, cited in WCDM, 2009).

Estimates for recharge rates (from areal infiltration to the UAU) vary from 3% to 10% MAP:

- Woodford et al (2003) concluded an average effective recharge of 3.9% of annual rainfall, using a GRID-based GIS modelling technique.
- The GRA II, BRBS, and Water Balance Model (DWAF, 2008c) methods all generate recharge estimates that are ~5% of the rainfall, and a maximum of 8%.
- Timmerman (1988) estimated recharge rates of 5 to 10% of rainfall for the UAU based on unsaturated zone modelling to estimate infiltration rates from rainfall, and calculated potential evapotranspiration rates using the Penman Method.
- Woodford and Fortuin (2003, cited in WCDM, 2009) estimated the mean annual effective recharge from rainfall for both the LRAS and EAS at 13.84 million m^3/a, using GRID-based GIS modelling. This equates to an average recharge rate of 3.1% of the MAP.

The recharge distribution developed by DWAF (2006a) is shown in Figure 7-23. Comparing the recharge distribution to the MAP surface, also developed by DWAF (2006a), (Figure 7-3), suggests a recharge rate of between 6% and 7% is applied.

The possibility and extent of leakage from the bedrock into the sand aquifers is raised as an unknown by DWA (2010b). However, groundwater from the bedrock is unlikely to have the good water quality signal of the LAU. Isotope and geochemical studies concluded that recharge is a local phenomenon, rather than occurring via deep-seated flow through the bedrock from distant mountain ranges (Weaver & Talma, 2000, cited in WCDM, 2009).

The area has a significant number of boreholes, and the casing details of these are generally unknown. It is plausible that uncased boreholes may allow a hydraulic connection between aquifers (hydraulic shortcut), allowing local downwards or upwards leakage depending on the local degree of confinement.

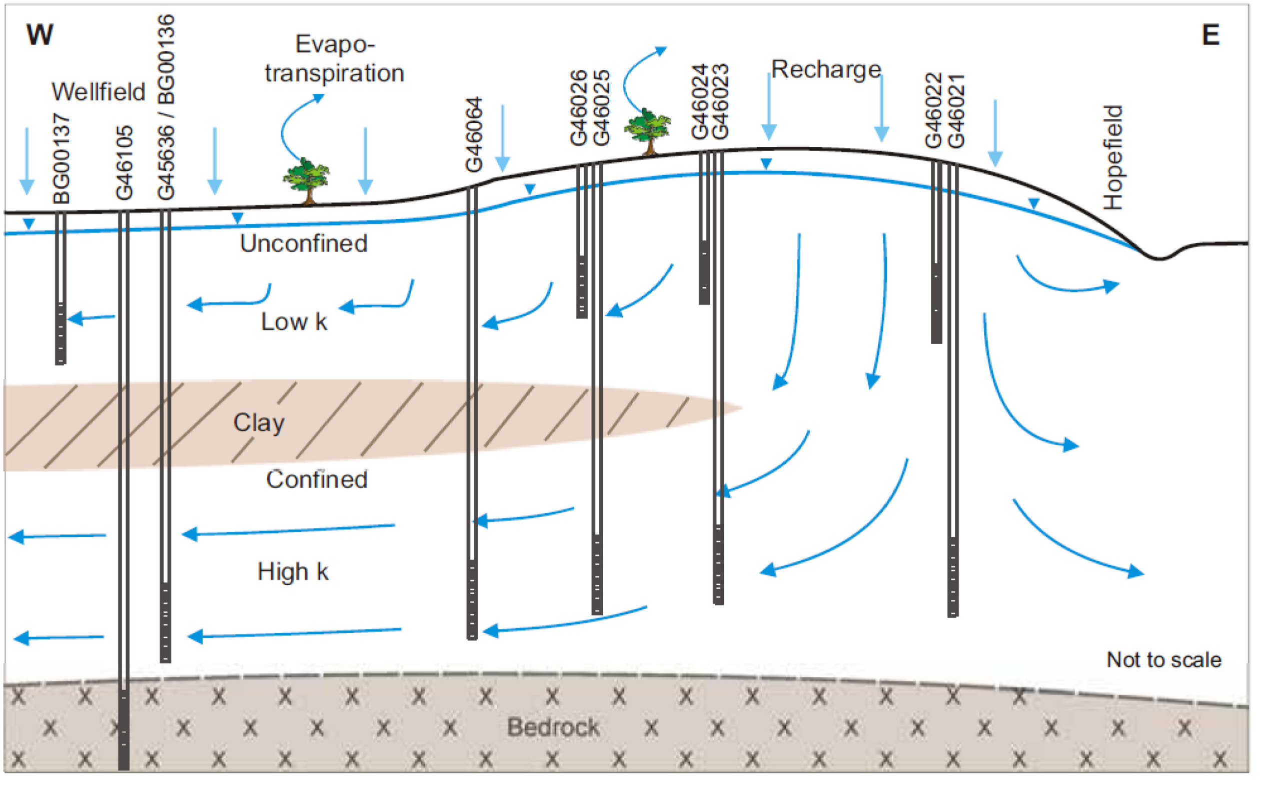


Figure 7-22 **Conceptual cross section for recharge mechanisms (from DWA, 2010b)**

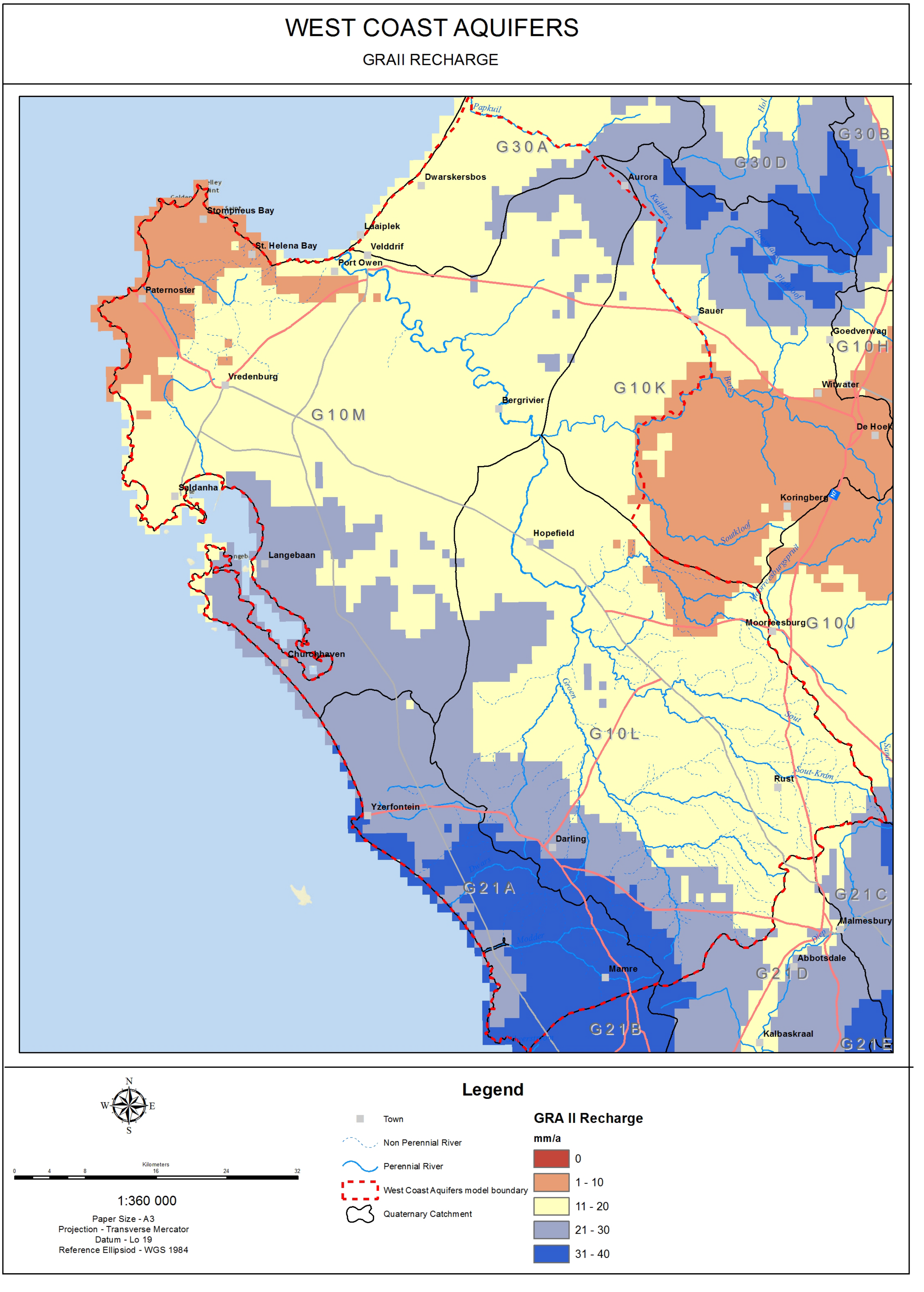


Figure 7-23 Recharge (GRA II, DWAF 2006a)

7.10.2. Interaction with Surface Water

The available baseflow information at quaternary catchment scale is shown in Table 7-8. The table indicates significant variability between different estimates; for example, the G10L catchment (Sout and Groën Rivers) is quoted as having a baseflow ranging between 0 and 2.83 mm (up to ~10% runoff). Groundwater contribution to baseflow is considered zero for almost all catchments in the region. This is considered a misrepresentation, caused by the simplified methods used for generation of the data. The relatively shallow depth to water of ~5–6 m (section 7.8), along with the piezometric maps showing flow (at least in the upper aquifer) towards the Berg, Sout and Groën Rivers (Figure 7-18, Figure 7-19) indicates that groundwater discharge to surface water is to be expected.

Table 7-8 Baseflow and groundwater contribution to baseflow data (mm) (from DWAF, 2008c)

Quaternary catchment	Baseflow: GRDM	Baseflow: Hughes	Baseflow: Pitman	Baseflow: Schulze	Groundwater contribution to baseflow
G10L	0.0	2.83	0.0	0.0	0.00
G10M	0.0	0.56	0.0	0.0	0.00
G21A	0.0	2.28	0.0	0.0	0.58
G30A	0.0	0.07	0.0	0.0	0.00
G10K	0.0	2.14	0.0	0.0	0.00

Coastal discharge will be the main mechanism for discharge from the LAU, if the LAU is considered continuous along the palaeochannel. Ecologically significant wetlands exist around Geelbek in the southern point of Langebaan Lagoon, which are known to be groundwater dependent (Saayman et al, 2004, Saayman, 2005). Whether the LAU also has (direct) interaction with surface water, also discharging to the Berg River, is dependent on whether the basement depression (and associated Elandsfontyn Formation) extends beneath the Berg River, and if it does extend, whether the clay confining layer is present. It also depends (as pointed out by DWAF, 2008c) on the depth of incision of the riverbed. It seems likely, based on the geological history (described in DWAF, 2008c), that if the palaeochannel was a palaeo-course of the Berg River, its course would have progressed from a mouth at Geelbek or Saldanha swinging clockwise to a mouth at Velddrif. There is no mechanism to explain the north-eastern extent of the LRAS palaeochannel extending towards the northeast beneath the Berg, as it does in Figure 7-12.

DWAF (2008c) points out that in times when the river is in flood, the groundwater gradient in the UAU will be reversed, and surface water will recharge the aquifer, at least locally to the river. Flow gauging data from a station at Jantjiesfontein shows that annual river stage fluctuations are around 1.6 m and that peak river flows are up to 2 m above the annual average stage (DWAF, 2008c).

The pressure effects of the fluctuations in river stage are detectable in water levels in the UAU. WCDM (2009) documents that water levels in holes near to the Berg River show yearly cyclical fluctuations of up to 0.8 m, which are not related to rainfall, but are a result of pressure variations caused by fluctuating water levels in the Berg River. The effects of this pressure wave in the aquifer are 'damped' further away from the river. This is well illustrated in boreholes G33323 and G33325 which are situated ~1.5 and 3.5 km from the Berg River respectively (WCDM, 2009).

7.10.3. Abstraction

Municipal abstraction

The WCDM operates a wellfield which abstracts from the LAU, within the LRAS palaeochannel. The abstracted groundwater augments the surface water-dominated Withoogte Water Supply Scheme, supplying a broad region slightly beyond the boundaries of Langebaan Local Municipality. The wellfield is licensed to abstract 4000 m^3/day or 1.46 million m^3/a, and between commissioning in December 2009 and January 2005, the WCDM utilised 80% of this allocation (WCDM, 2005). Over this ~5-year period of abstraction, the piezometric level in the LAU reduced by around 10 m, and the UAU water table remained stable (WCDM, 2005).

During 2005, concern was raised over the 10 m reduction in piezometric level in the LAU, leading to a reduction in the abstraction rates by 10% — a reduction which remains in place currently. The most detailed description available of this decision to reduce abstraction is given in WCDM, 2008:

> On the 23rd November 2005, the monitoring committee agreed upon a 10% reduction in the volume abstracted from the WCDM Wellfield to 1.314 million m3/a (109,500 m3/month or 3,600 m3/day). **This step was taken as a precautionary measure after the water levels** in a number of monitoring holes in the Lower Aquifer of the LRAS had **declined to just above their 'early-warning' trigger-levels (as set in the AquiMon monitoring system**). This reduced rate of abstraction will be maintained until water levels have recovered and stabilised above their trigger-levels (WCDM, 2008).

What the early-warning trigger-levels refer to is not documented and is not known to DWS or WCDM, but may relate to a desire to maintain the LAU as an artesian system. With the reduction of 10 m the LAU remains confined in the area of the wellfield. Abstraction during April 2006–March 2007 totalled just over 1 million m^3/a (76% of revised allocation). WCDM (2009) recommended that abstraction rates remain at the reduced rate until 'the next Committee meeting to be held in July 2008'. Recent annual abstraction values, however, remain around 1 million m^3/a (2007–2012, WCDM 2012), and the decision to reduce the rates by 10% does not appear to have been revisited. Abstraction has significantly reduced in 2014–2015 due to wellfield pump and infrastructure vandalism.

Groundwater is also abstracted from a wellfield at Hopefield and no detailed information could be found on this. WCDM (2009) refers to the wellfield as a 'recently revived municipal wellfield', for which they could not obtain data. The All Towns Strategy (DWS, 2016) states that the town has seven boreholes which are in poor condition, with a yield of 0.16 million m^3/a, but that these are not currently in use.

Private abstraction

There are 78 registrations in the WARMS database for the study area, with a total registered abstraction of 6.9 million m^3/a. The sum of registered abstractions per water use sector is shown in Table 7-9. The majority of groundwater use is for irrigation.

Table 7-9 Registered groundwater abstraction in WARMS (WARMS extract dated August 2015).

Water use sector	Registered abstraction (m^3/a)
Agriculture: irrigation	5 800 547
Agriculture: watering livestock	224 980
Water supply service	341 300
Industry (urban)	532 870
All	**6 899 697**

Comparing the distribution and sum total of WARMS abstraction with the distribution and sum total of abstractions in WCDM monitoring reports (WCDM, 2005, 2009, 2011, 2012), it is clear that a number of private abstractions are not captured in the WARMS database for the area. An additional challenge with the WARMS database is that there is no information as to which aquifer is targeted (LAU, UAU). The majority of private groundwater use is assumed to be from the UAU; however, some private groundwater users do access the LAU (DWA, 2010b).

The private abstraction data incorporated in the WCDM monitoring reports (WCDM, 2005, 2008, 2011, 2012), is generally based on results from DWAF hydrocensuses carried out in 1990 and 2002 (the results of which are reported in WCDM, 2005). The earlier hydrocensus was more extensive and Woodford and Fortuin (2003) used this to calculate a 'weighted' estimate of total private groundwater abstraction. Furthermore, recent WCDM monitoring reports include data from monitoring at four neighbouring farms, at least one of which also abstracts from the LAU. Water level data is available, yet abstraction rates are not monitored (WCDM 2012).

Raw data (location or borehole name, and abstraction) from the original 1990 and 2002 DWAF hydrocensus is not available.
Figure 7-24 shows the available information regarding borehole locations, and Appendix 2 of DWAF (2003) contains a list of total abstraction per cadastral unit (not per borehole). The abstraction data from the DWAF hydrocensus has been re-created using the table of total abstraction per cadastral unit, digitising the points shown in
Figure 7-24, then distributing the total abstraction per cadastral unit across the boreholes indicated. This method is not ideal as individual high-yielding boreholes may be misrepresented if their abstraction is divided between several boreholes on a cadastral unit. Nevertheless it is the only available method to re-create the hydrocensus information.

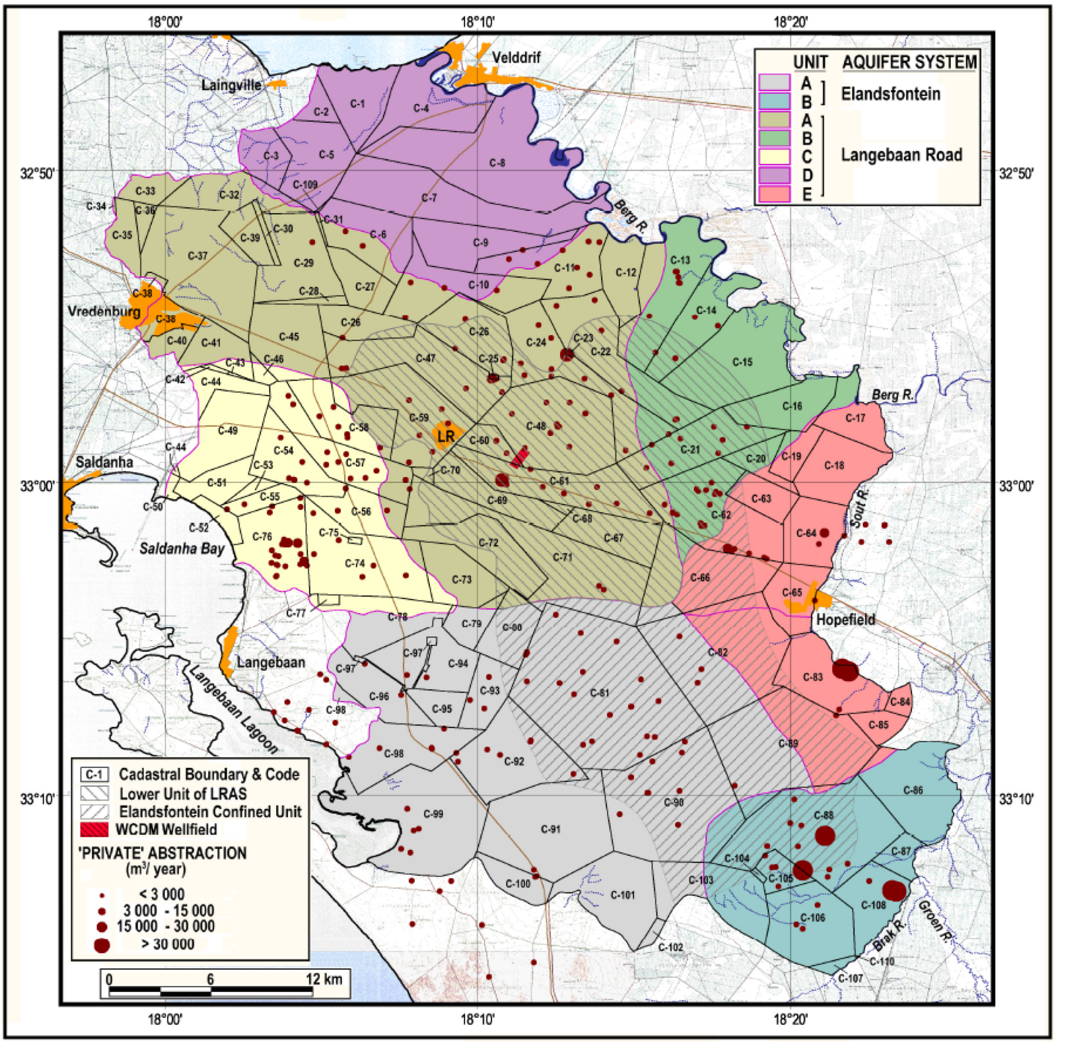


Figure 7-24 Private groundwater abstraction based on DWAF 2002 hydrocensus (WCDM, 2009)[11]

The current WARMS registrations were compared to the hydrocensus data. Where a WARMS registration existed on a farm where there was already hydrocensus information, the WARMS point was discarded in favour of hydrocensus data (assuming WARMS may be an over-registration and the hydrocensus would be more accurate, potentially based on interviews with users or pump specification). The hydrocensus data was also favoured because it contains borehole coordinates. Where there was no hydrocensus information on a cadastral unit where there was a WARMS registration,

[11] Unit subdivisions within the aquifer systems (A to E) should be disregarded

the registration was retained. This is true for all abstractions initiating after ~2003, and for areas of the model domain outside of that shown in

Figure 7-24 covered by the hydrocensus.

The resulting abstraction for the whole area was 6.53 million m^3/a. This total is less than the total in WARMS likely due to the process of merging with the hydrocensus information, where per borehole abstractions were significantly lower.

7.11. Groundwater quality

A Durov diagram presented in DWAF (2008c) highlights that groundwater in the region has a Na Ca\Cl alkaline character, with an average pH of 8. There is a marked difference in salinity between the upper and lower aquifer units. In the LAU, electrical conductivities (EC's) are commonly less than 120 mS/m, whilst that in the WCDM wellfield is in the order of 80 mS/m (WCDM, 2005). In WCDM (2005) it is noted that ECs in the UAU are often > 250 mS/m, and often exceed 500 mS/m close to the Berg River, and close to Saldanha Bay. An EC map is presented in WCDM (2005), which shows that ECs reach > 1 000 mS/m (reproduced in Figure 7-25). The available NGDB records (both aquifers) provide an average groundwater EC of 342 mS/m and range between 24 and 4 548 mS/m (DWAF, 2008c).

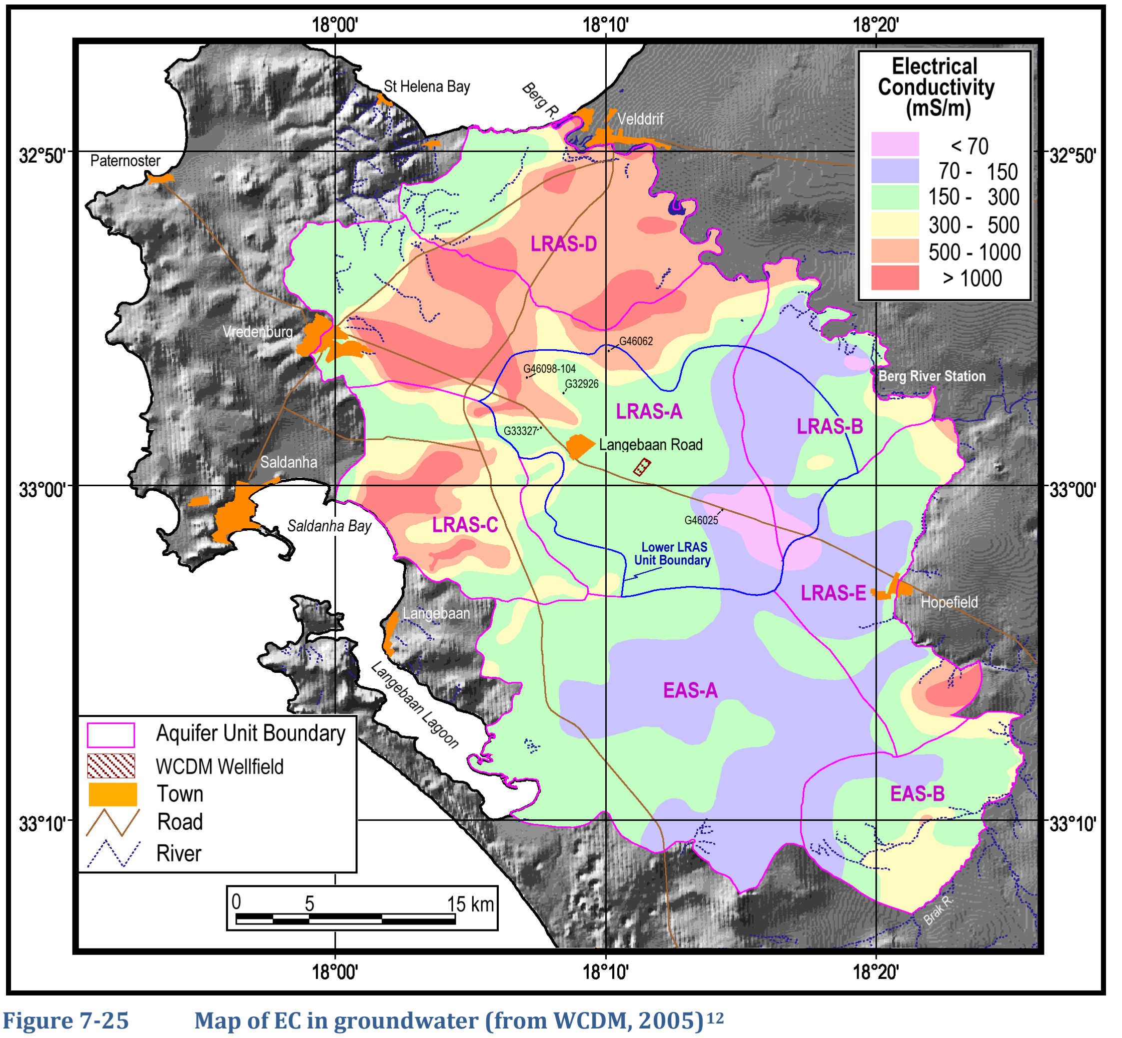


Figure 7-25 Map of EC in groundwater (from WCDM, 2005)[12]

DWAF (2008c) suggests that the high EC in the UAU around the Berg River may be the result of mixing with high-salinity surface water (i.e. when the hydraulic gradient is reversed and the river recharges the aquifer). This, however, doesn't explain the high EC in the west at Saldanha Bay, and DWAF (2008c) discounts saline intrusion. WCDM (2009) points to the low rainfall and shallow groundwater table, coupled with high rates of evapotranspiration and low rates of

[12] Aquifer unit boundaries shown should be disregarded

groundwater movement, as the cause of an accumulation of salts in the aquifer over time. It is also suggested that mixing with connate groundwater, and salt loading from mist and windborne sea spray, may have contributed to the high salinities in the UAU around Saldanha Bay. In addition to these mechanisms, dryland agriculture is predominant in the area, hence land management/tillage approaches may also enhance salt accumulation.

7.12. Previous estimations of Aquifer Yield

Some previous estimates of the aquifer yield are listed below (with varying definitions for, and hence approaches to, estimation).

Estimates based on a proportion of recharge include:

- Rainfall recharge to the **LRAS** has been estimated at ~13 million m^3/year, assuming a recharge rate of 8%. Weaver et al (1997, cited in WCDM, 2009) concluded that approximately **7.8 million m^3/a** could be safely abstracted from the aquifer system.
- Woodford et al (2003, cited in DWAF, 2008c) suggested that the sustainable yield from the **LAU** in the **LRAS** was **1.5 million m^3/a**, which is equivalent to the current abstraction at the WCDM wellfield.
- Woodford (2003, cited in WCDM, 2005) determined a combined exploitation potential for the **LRAS and EAS** of approximately **14.0 million m^3/a**. Groundwater abstraction from these aquifer systems was estimated at 4.27 million m^3/a, of which the WCDM is licensed to pump 1.46 million m^3/a.

Estimates based on the monitored response to abstraction include:

- The sustainability of the WCDM abstraction, which between 1999 and 2005 had caused 10 m drawdown, is widely considered in literature. Almost all discussions revolve (only) around the 10 m reduction in water levels without consideration of the source of abstracted water, the impact on the flow regime, and the acceptability of these impacts. Wellfield abstraction was reduced in 2005 by 10% based on this 10 m drawdown (section 7.10.3), and because the water levels were approaching pre-set early-warning levels. What the pre-set early-warning levels are, and how they were determined, is not documented and is not known to DWS.
- The water level response to abstraction for 1999–2005 is shown in Figure 7-26. WCDM (2009) considers the wellfield LAU water levels in 2004 and 2005 to have stabilised, and applied a multiple linear regression analysis to derive a linear equation of the water level changes as a function of recharge and abstraction, using data after the initial 7.5 m water level reduction. Their assumption is that a dynamic equilibrium is reached after this 7.5 m reduction, at which point water is no longer derived from storage, and as such the water level can be described as a function of abstraction and recharge in the capture zone. WCDM (2009) managed a 0.82 correlation coefficient between water level change and annual rainfall. Their conclusion from the resulting equation is that in order to maintain a 'stable' piezometric level in the Lower Aquifer of the LRAS around the wellfield, **the wellfield should be operated at 1.113 million m^3/a**. They are able to predict that abstracting 1.3 million m^3/a from the wellfield would result in the piezometric level declining to the base of the clay layer in the vicinity of the production holes after approximately 30 years. This approach, however, implicitly assumes a linear relationship between rainfall and recharge, and that the aquifer is homogeneous over its depth (or drawdown), but neither assumption is met. Furthermore, although the rate of reduction of water levels has certainly reduced in early 2005, water levels do continue to decline.

Estimates based on numerical modelling include:

- DWAF (2008c) modelled a hypothetical wellfield abstracting 0.7 million m^3/a, distributed between four boreholes, in the LAU of the EAS. The impact on the water levels and modelled aquifer fluxes are presented, with the interpretation that the abstracted water is derived from storage. The model results are for a transient model simulation after an unspecified duration, and no steady-state (dynamic equilibrium) was established, hence no long-term assessment of the impact (and therefore sustainability) of this abstraction is possible.
- Timmerman (1988, summarised in WCDM, 2009), simulated several abstraction scenarios to support an estimation of the resource potential. The scenarios included abstraction of:
  - 3.15 million m^3/a from the LAU during drought (no recharge during simulation)
  - 3.15 million m^3/a from the LAU during normal climate (recharge exists)
  - 9.5 million m^3/a from the LAU and 4.7 million m^3/a from the UAU during normal climate (recharge exists).

  From these scenarios, Timmerman (1988) provided 'short-term' and 'long-term' yields of exploitable groundwater. These appear to be based on there either being no significant impact on water levels (short-term yields), or on allowing some decline in water levels which was equated to aquifer mining (long-term yield).

There is some link to the capture principle, as the short-term yields were to be met by existing recharge, with an acknowledgement that the long-term yields would reduce water levels and allow more 'space' for subsequent recharge. The conclusion is that the development potential of the LAU is limited, despite high hydraulic conductivity. **The 'long-term sustainable yield' of the LRAS and EAS combined was concluded to be 6.32 million m^3/a.** It is assumed that this conclusion applies to the LAU in the LRAS and EAS.

- WCDM (2009) established a numerical model to simulate groundwater flow conditions between 26 August 1998 and 3 September 2008. The purpose of the model was to provide estimates of the groundwater resource potential of the LRAS and EAS at the current levels of development, as well as under a scenario where the current WCDM wellfield in the LRAS is expanded and a new wellfield is established in the EAS. The (then) current abstraction was simulated and the aquifer fluxes were presented for each GMU (section 7.7.1). This makes interpretation of the model results challenging as groundwater flows between GMUs, and no summary of exchanges between external sources and sinks is provided (i.e. total flow between upper/lower aquifer and particular stretches of surface water). The model shows an average rate of 1.5 million m^3/a reportedly leaks through the clay layer into the LAU in the central LRAS area around the wellfield.
- The WCDM (2009) assessment of potential additional aquifer yield revolves around discussion of pre-abstraction virgin recharge, the locations of other users, hydraulic properties, and of good quality groundwater and high recharge areas. Based on this discussion, it is considered possible to extend the current WCDM wellfield by developing an additional two production boreholes and 'it is estimated that **~1.9 million m^3/a of the net annual average recharge can be 'harvested' by developing a wellfield that taps the Lower Aquifer in the central portion of the EAS**'. This increased abstraction was tested in the numerical model under the 10-year simulation. The scenarios are evaluated based on water level change during the 10-year period, and source of abstracted water (flow regime change). However, only the flow regime between neighbouring GMUs is reported, rather than aquifer boundaries and external sources. The impact of abstraction on the natural aquifer fluxes is not discernible. Some comments are made about water level changes but they are considered insignificant, hence the schemes are considered acceptable and recommended.

None of the existing yield estimates fully meet the requirements of a capture principle-based assessment of yield (summarised in Table 7-1).

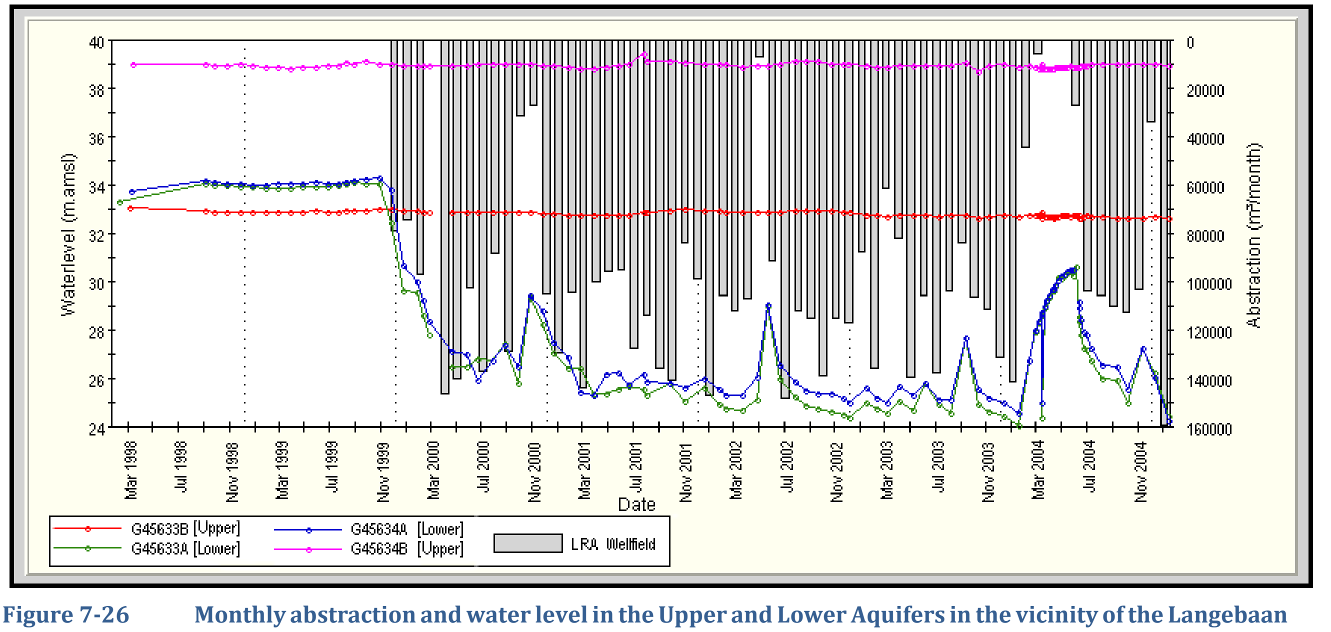


Figure 7-26 Monthly abstraction and water level in the Upper and Lower Aquifers in the vicinity of the Langebaan Road Wellfield (up to Jan 2005) (WCDM, 2005).

7.13. Current status of abstraction

The response of the LAU in the LRAS to abstraction is documented in several monitoring reports over time (WCDM, 2005, 2008, 2009, 2011, 2012). These reports show:

- An asymmetrical pumping 'cone of depression' has developed in the LAU that extends 4 to 11 km from the wellfield.

- The clay layer acts as a competent aquitard as no water level impact has been found in the UAU.
- The wellfield underwent a recovery test in 2005 (WCDM, 2005), during which piezometric-levels in the LAU responded up to 10 km to the northwest of the wellfield (broadly downstream), and those monitoring points lying further than 5 km to the east and southeast of the wellfield did not respond (broadly upstream).
- Since implementing the 10% reduction in abstraction rates, piezometric-levels in the LAU have risen by ~2 m since April 2007, likely due to the abstraction reduction and also potential recharge influences.
- In general across the area, water levels in both the Upper and the Lower Aquifers of the LRAS have risen significantly since April 2007. The water table in the Upper Aquifer has risen in response to widespread rainfall recharge that took place between May and September 2007.
- The water levels over time for wellfield boreholes is shown in Figure 7-27. Certainly the abstraction since 2005 (and before) appears to be (less than) a maintainable aquifer yield.
- Although several of the previous investigations recommended the EAS as a future bulk water supply option for large scale/municipal abstraction (section 7.12), these recommendations have not been implemented.

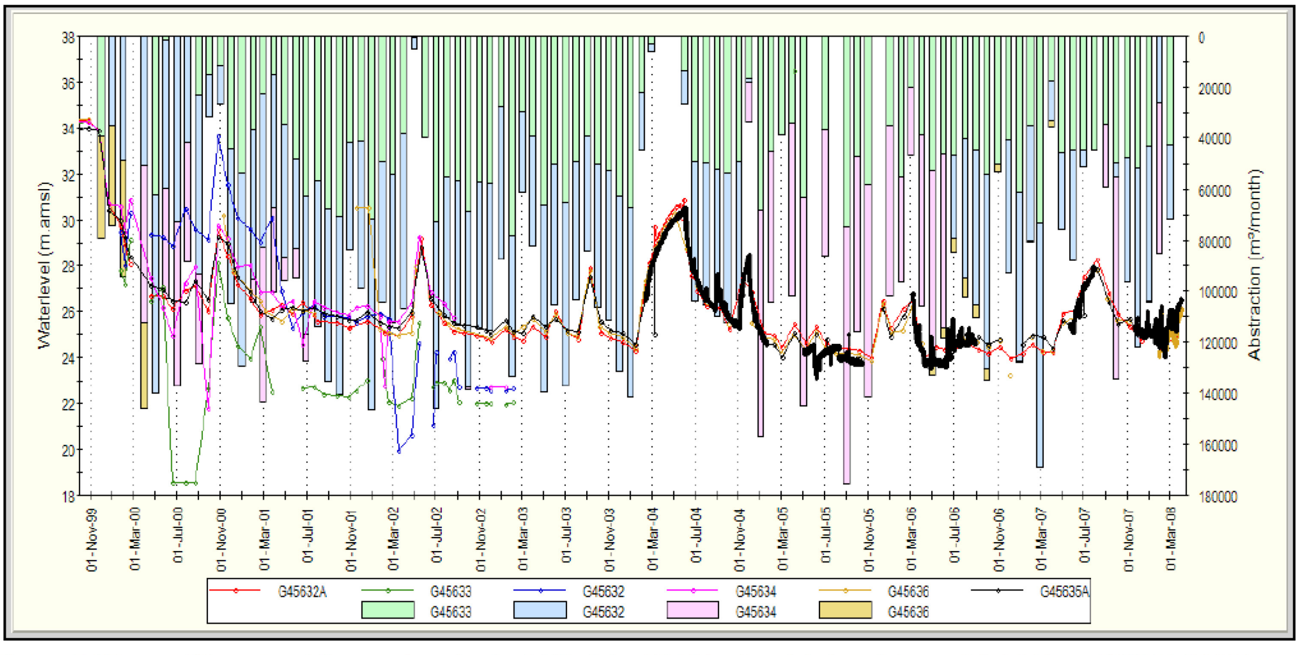


Figure 7-27 Water level fluctuations in the LAU of the LRAS in the vicinity of the wellfield (from WCDM, 2009)

7.14. NUMERICAL MODEL SETUP

7.14.1. Computer Code

The software code chosen for the numerical finite-element modelling work was the 3D groundwater flow model, SPRING, developed by the delta h Ingenieurgesellschaft mbH, Germany (König, 2011). The program was first published in 1970, and since then has undergone a number of revisions. SPRING is widely accepted by environmental scientists and associated professionals. SPRING uses the finite-element approximation to solve the groundwater flow equation. The model area or domain is represented by a number of nodes and elements. Hydraulic properties are assigned to these nodes and elements and an equation is developed for each node, based on the surrounding nodes. A series of iterations are then run to solve the resulting matrix problem utilising a pre-conditioning conjugate gradient (PCG) matrix solver. The model is said to have 'converged' when errors reduce to within an acceptable range. SPRING is able to simulate steady and non-steady flow, in aquifers of irregular dimensions.

SPRING solves the stationary flow equation independent of density for variable saturated media as a function of the pressure, according to:

$$-\nabla(K_{ij}\nabla h) = -\nabla\left(K_{perm}\frac{\rho g}{\mu}\nabla h\right) = q = -\nabla\left[\frac{K_{perm}\cdot k_{rel}}{\mu}(\rho g\nabla z + \nabla p)\right]$$

∇ $\left(\frac{\partial}{\partial x},\frac{\partial}{\partial y},\frac{\partial}{\partial z}\right)$
q Darcy flow
K_{ij} Hydraulic conductivity tensor
ρg Density $\cdot$ gravity
K_{perm} Permeability
μ Dynamic viscosity
k_{rel} Relative permeability
p Pressure

The relative hydraulic conductivity is calculated as a function of water saturation, which in turn is a function of the saturation:

$$k_{rel}(S_r) = (S_e)^l\left[1-\left(1-(S_e)^{\frac{1}{m}}\right)^m\right]^2$$

$$S_e = \frac{S_r(p)-S_{res}}{S_s-S_{res}} = \left[1+\left(\frac{p_c}{p_e}\right)^n\right]^{\frac{1-n}{n}}$$

$S_r(p)$ Relative saturation dependent on pressure
S_e Effective saturation
l Unknown parameter, determined by van Genuchten to 0.5
m equal to $1-(1/\mathrm{n})$
n Pore size index
S_{res} Residual saturation
S_s Maximum saturation
p_c Capillary pressure
p_e Water entry pressure

Solving these equations for the relative saturation as a function of the capillary pressure $S_r(p_c)$ results in the capillary pressure- saturation function according to the Van Genuchten (1980) model as used in SPRING:

$$S_r(p_c) = S_{res} + (S_s - S_{res})\cdot\left[1+\left(\frac{p_c}{p_e}\right)^n\right]^{\frac{1-n}{n}}$$

The water entry pressure is a soil specific parameter and defined as the inverse of a = $1/p_e$ in the saturation parameters. The density independent, instationary flow equation for variable saturated media as a function of the capillary pressure is given as follows:

$$\rho\left(S_r(p_c)S_{sp} + \theta\frac{\partial S_r(p_c)}{\partial p}\right)\frac{\partial p}{\partial t} + \theta S_r(p_c)\frac{\partial \rho}{\partial t} - \nabla\left[\rho\frac{K_{perm}k_{rel}}{\mu}(\nabla p + \rho g\nabla z)\right] = q$$

The specific pressure dependent storage coefficient S_{sp} is hereby given as

$$S_{sp} = \alpha(1-\theta) + \beta\theta$$

α Compressibility of porous media matrix
β Compressibility of fluid (water)
θ Aquifer porosity

The transport equation for a solute in variably saturated aquifers is given as follows:

$$\theta S_r(p_c)\frac{\partial c}{\partial t} + \theta S_r(p_c)v\nabla c - \nabla\left(\theta S_r(p_c)\left(D_m\bar{\bar{1}} + D_d\right)\nabla c\right) = qc^* + R_i$$

qc^* Volumetric source/sink term with concentration $\mathrm{c}*$
D_m Molecular diffusion
$\bar{\bar{1}}$ Unit matrix
D_d Hydrodynamic dispersion
R_i Reactive transport processes (sorption, decay, etc.)

The software is therefore capable of deriving quantitative results for groundwater flow and transport problems in the saturated and unsaturated zones of an aquifer.

7.14.2. Model Spatial and Temporal Domain

The model domain covers an area of 4 791 km^2 and is bounded by the coastline in the northwest and southwest. The southeast model boundary follows the quaternary catchment divide separating G21A and G10L (within the model) from G21B, C and D and G10J (Figure 7-2). In order to avoid subdivision of the Adamboerskraal Aquifer System along quaternary catchments which may not represent groundwater divides, the model boundary diverts from catchment divides and follows the contact between Sandveld and the basement outcrop, along which the surface water courses of the Soutkloof and Berg River also flow. The model boundary in the northeast then follows either surface water courses or the geological contact with the Table Mountain Group Formation, which in the region, flows to the northwest and is considered a hydraulic boundary to the Sandveld Formation aquifers (DWAF, 2008c).

The finite-element model was originally set up as a three-dimensional, four-layer, steady-state groundwater model. In view of the capabilities of the software to simulate the unsaturated zone and layers that pinch out, the layers were arranged to represent the conceptual model (Table 7-10). While the model can essentially be simulated as a four-layer model, enhanced numerical stability is required to support consideration of the unsaturated flow component, hence the model layer representing the UAU was divided into two (Table 7-10). The Elandsfontyn Formation (LAU and overlying clay layer) does not outcrop in the area, hence the LAU layer 4 pinches out against rising basement, or against the clay layer where thick; and in turn the clay layer also pinches out where basement rises. The UAU also pinches out against basement where basement outcrops.

The model domain was spatially discretised into 96 409 nodes on six node layers, which make up five element layers with 105 394 elements (triangles and quadrangles) per layer. The element size varies from a minimum of 10 m at areas with expected steep groundwater gradients (abstraction holes, geological contacts, rivers) to a maximum of 250 m (Figure 7-28) further away from areas of interest. The spatially variable discretisation of the (finite-element) model domain allows the sufficiently accurate incorporation of the wellfield in a regional groundwater flow model.

The aquifers in the region are considered to be at a state of dynamic equilibrium prior to 1999 (section 7.8). Therefore, data prior to 1999 (abstraction and water levels) was used in the model to represent steady-state conditions and enable a steady-state calibration.

Table 7-10 Layer arrangement for the groundwater model

Element layer (colour in Figure 7-29)	Representing	Thickness	Data source
1 (red)	Upper 2 m of UAU / upper 2 m of basement aquifer. Established for numerical stability	2 m	Layer top: DEM Layer bottom: DEM-2 m
2 (pink)	UAU (discontinuous)	0-169 m	Layer bottom: clay top as defined through NGA lithology data (section 7.6)
3 (yellow)	Clay layer (discontinuous)	0-47 m	Layer bottom: clay top minus clay thickness based on WCDM (2009) (section 7.6)
4 (blue)	LAU (discontinuous)	0-65 m	Layer bottom: basement elevation (NGA lithology data and Roberts & Siegfried, 2014), (section 7.6)
5 (red)	Basement aquifer	20 m	Layer bottom: basement elevation – 20 m

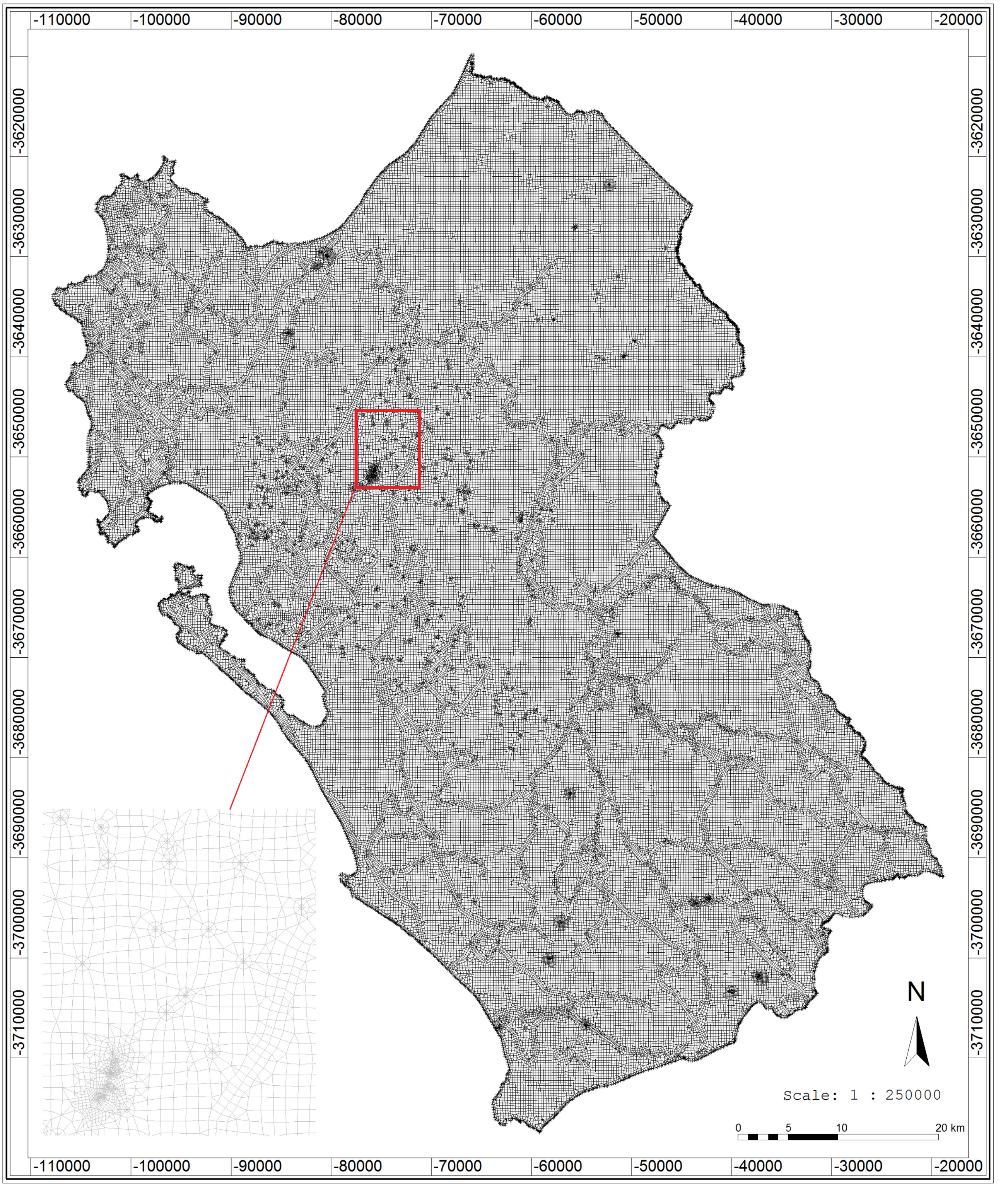


Figure 7-28 Finite-element mesh of the groundwater model

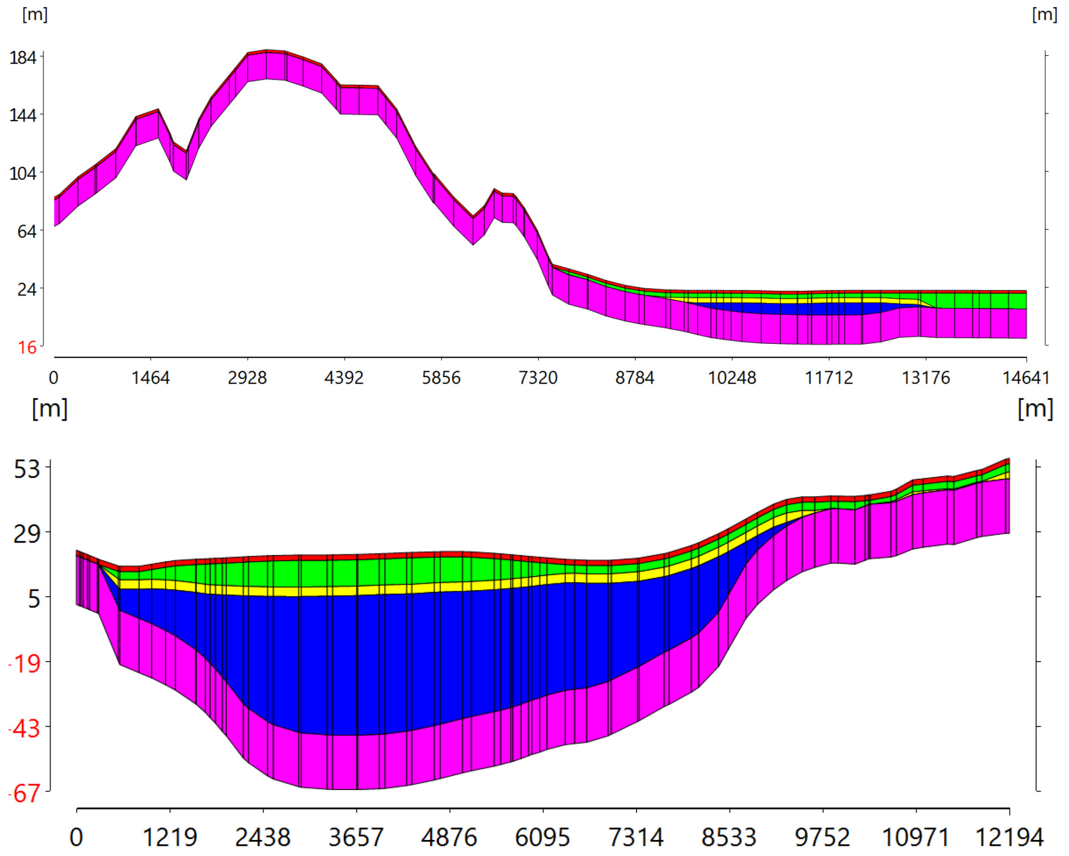


Figure 7-29 Example of the vertical grid layout with sections from Vredenburg koppie towards the southeast (top), and across the LRAS palaeochannel (bottom). Colours indicate numerical model layers only.

7.14.3. Boundary Conditions

7.14.3.1.Recharge

Recharge was set according to four zones as illustrated in the GRA II data for the region (Figure 7-23, using the mid-point of the range shown per zone). Over the model domain, the recharge rate varies from 5.5 to 35.5 mm/a, and equates to a total of 88.5 million m^3/a. The lower rates (equivalent to 3 to 5% MAP) applied in WCDM (2009) were tested in the model and appeared to require hydraulic conductivities out of range of other studies and reasonable estimates. Furthermore, when compared to the MAP derived for the four rainfall stations in the area (section 7.4), the GRA II recharge is equivalent to 5 to 9%, which seems reasonable for the sandy subsurface.

Irrigated agriculture is limited in the area and enhanced recharge via irrigation return flows is not accommodated in the model setup. Nor does the model allow for a potential increase in aerial recharge caused by pumping. Given the average depth to water is around 6 mbgl, the possibility of rejected recharge due to high groundwater tables is likely to be negligible.

7.14.3.2.Interaction with surface water

A river or 3rd type (Cauchy) boundary condition was assigned to the ephemeral and perennial rivers within the model domain, whereby the leakage of groundwater into the surface water course depends on the prevailing gradient. Furthermore, using the river network flow balance approach built into SPRING, losses of downstream river stretches are limited to upstream groundwater baseflow or artificial discharge into the river network. By preventing unrealistic water losses from individual river stretches, the approach ensures a dependable flow balance for surface water courses in the model domain. A uniform incision of 5 m below the surrounding topography was assumed for the hydraulically active river bed for all rivers in the model domain. Although stage information (in metres above baseplate) was sourced (section 7.5), without the rating curve or survey heights it is not possible to specify surface water level in greater detail.

7.14.3.3.Abstraction

The abstraction dataset available for modelling contained WARMS data and the 2002 DWAF hydrocensus (section 7.10.3). The steady-state model was constructed to simulate conditions prior to November 1999, hence the abstraction dataset was separated by dates. A pre-November 1999 abstraction dataset included all WARMS registrations initiated prior to November 1999, plus data in the 2002 DWAF hydrocensus, under the assumption that there is insignificant growth in groundwater abstraction between November 1999 and 2002. A post-November 1999 abstraction dataset includes all WARMS registrations after this date, for use in the predictive scenarios.

Borehole abstraction from the model domain was simulated using the withdrawal well boundary condition. Outflow rates were assigned to single nodes (at the appropriate elevation) within the model mesh. For fully penetrating wells, equal heads were prescribed for all applicable well nodes and the contributions from the different model (or aquifer) layers computed by the software.

7.14.4. Selection of Calibration Targets and Goals

The water level database established for the region (section 7.8) contained a significant number of points (~1000), and several were closer to each other than the model mesh (i.e. closer than 250 m). Furthermore, the dataset showed some irregularities (highs or lows in water level compared to surrounding points); to be expected given the range of time periods covered by the measured data. The water level dataset was therefore manually amended prior to modelling to remove points that were very close together (keeping one of a cluster), and removing points that were significantly dissimilar to neighbouring points, if that point was based on a single measurement. Priority was given to maintaining points that were based on multiple measurements, and points that are continually measured up to today (HYDSTRA holes). All remaining water level points (530 in total) were utilised throughout the calibration.

Out of the remaining 530 points, 27 were assigned to the UAU, 30 to the LAU, 100 to basement aquifers (in areas of outcrop), and the remainder were listed as unknown. The dataset is insufficient to form conclusions over the degree of hydraulic separation (difference in piezometric head) between the UAU and LAU (section 7.8), and insufficient for a separate calibration of the UAU and LAU. All water level data was therefore grouped together for calibration. The lack of a rating curve also precludes the use of surface water flow data as a secondary calibration target.

Since the modelled groundwater levels are directly related to the assigned recharge rates and hydraulic conductivities, an independent estimate of one or the other parameter is required to arrive at a potentially unique solution of the model. The applied recharge rates were therefore considered fixed during the calibration and only hydraulic conductivities varied.

7.14.5. Initial Conditions

The initial conditions specified in the numerical model were as follows:

- Initial heads for the model were interpolated from the (full) steady-state water level dataset using Bayesian interpolation, i.e. co-kriging using the established correlation between surface topography and groundwater elevation.
- Hydraulic conductivities were based on the existing estimates for the area (Table 7-7).
- Vertical hydraulic conductivities were set at 10% of the horizontal conductivities, and not varied.

7.14.6. Numerical Parameters

SPRING uses an efficient preconditioned conjugate gradient (PCG) solver for the iterative solution of the flow and transport equation. The closure criterion for the solver, i.e. the convergence limit of the iteration process was set at a residual below 1e-06. The Picard iteration, used for the iterative computation of the relative permeability for each element as a function of the relative saturation, used a damping factor of 0.3 and was limited to five iterations. The relative difference between the two computed potential heads or capillary pressures after five iterations was usually below an acceptable 0.1 m.

7.15. NUMERICAL MODEL CALIBRATION

7.15.1. Model Refinement

To maintain numerical stability, the uppermost model layer was vertically refined for the calibration, with the final model layering shown in section 7.14.2. The vertical refinement enables an accurate calculation of the relative permeabilities according to the Van Genuchten model as well as an accurate representation of the seepage surfaces. A thin layer with higher hydraulic conductivity (reflecting a soil zone) is often necessary to enable infiltration of recharge through the unsaturated zone, where underlying layers have lower hydraulic conductivity (i.e. basement outcrops).

7.15.2. Steady-State Calibration

The model was run with the initial conditions, and the hydraulic conductivities were adjusted using sensible boundaries until a best fit between measured and computed heads was achieved. Figure 7-30 shows the resulting correlation between observed and modelled water levels. The root mean square error (RMSE) was used as quantitative indicator for the adequacy of the fit between the 530 (=*n*) observed (h_{obs}) and simulated (h_{sim}) water levels:

$$RMSE = \sqrt{\frac{\sum(h_{obs} - h_{sim})^2}{n}}$$

$$NRMSE = \frac{RMSE}{h_{max} - h_{min}}$$

An RMSE of 8.6 m was achieved, with a normalized RMSE (NRMSE) of 2.8%, and a mean error of 6.4 m. An RMSE and NRMSE of less than ten is generally considered acceptable, and with such a large model domain with a significant number of observation points, the calibration is considered good, and adequate for the intended model purpose.

The steps taken during calibration highlight the following:

- The model comprises a large area, across which there is significant topographical variability (and in many areas not a significant recharge variability), and yet depth to water is fairly constant in the observation data. Furthermore, there is a good spread of 530 observation points across all terrains. A heterogeneous hydraulic conductivity was required in various layers to represent the water level variability.
- Once a set of hydraulic conductivities per layer, that generated a generally acceptable calibration, was found, (i.e. an RMSE of ~14), the remaining points or areas of the model where large errors occurred, were mainly related to irregular layer thicknesses. In places sudden changes in layer thicknesses were noted, which translate to sudden changes in hydraulic conductivities. Therefore during calibration layers were smoothed to maintain a more constant hydraulic conductivity in a region, rather than generate a highly varying hydraulic conductivity within a single layer.
- The water level high in the south of the EAS (i.e. at and to the north of the junction between G21A, G10L and G10M) was a challenging area in the calibration. Large differences in model errors occurred in close proximity to each other, suggesting a high heterogeneity that is not replicated in the model. The UAU is thick in the area, and the heterogeneity in the UAU in the EAS is known in literature (section 7.7). To replicate the observed water levels, the hydraulic conductivity of the UAU is lower in the EAS than other regions, which corresponds with the description that up to four aquifer-aquitard layers may be present within the UAU in the EAS (Timmerman, 1985c).
- Certain observation points could not be replicated through application of hydraulic conductivities that generally worked across the remainder of the model and were left as is. Hydraulic conductivity combinations that enable a fit to each individual observation point could have been applied in small areas local to an observation point, but this would result in an over-fitted solution. Those observation points with the highest errors were generally on the edges of the model boundary, in the southeast, and northeast. No observation points were removed from the calibration.

The range in modelled hydraulic conductivities for each model layer is shown in Table 7-11. As only the upper layer was calibrated to point data, these hydraulic conductivities are representative of an equivalent hydraulic conductivity required to match the observed, and are not directly representative of each aquifer. The resulting modelled piezometric contours are shown in:

- Figure 7-31 for node layer 1 (i.e. model surface). The contours can be considered representative of basement where basement outcrop occurs, and the UAU outside of basement outcrop.
- Figure 7-32 for node layer 1 wherever element layer 2 is present (i.e. the UAU).
- Figure 7-33 for node layer 4 (i.e. the LAU), shown with basement topography.

Comparing the piezometric contours for the UAU and LAU shows that a separation of piezometric head between the two layers has been achieved. Although the data on the degree of separation of piezometric level is inconclusive (section 7.8), some hydraulic disconnection of the layers is required given that the UAU has not responded to pumping in the LAU. Although a formal calibration to point data has not been carried out for the LAU, comparison of the modelled contours to the information presented in section 7.8 suggests that the modelled heads are too low around the EAS. Further investigation would be required to improve on the model calibration (section 7.17), but the result is considered adequate for the purpose of the study.

The steady-state model aquifer flows are shown in Table 7-12. This illustrates that ~60% of the 88.5 million m^3/a recharge coming into the model domain leaves via discharge to surface water (53.9 million m^3/a), and much of the remainder is discharged to the ocean. Considering the aquifer discharge at each of the major bays, the majority is contributed by the LAU rather than the UAU (5.1 million m^3/a from the LAU to Langebaan Lagoon, compared to 0.6 million m^3/a from the UAU). This is an intuitive result given the thickness of the LAU at the coastline and the higher hydraulic conductivity. This result is reported cautiously however, due to the lack of data on which to base the representation of the aquifers and the coastline. The thickness of the clay layer overlying the LAU at the coastline is not known, and the direct connectivity between LAU and coastline as represented in the model may be an exaggeration (discussed in recommendations, section 7.17).

Table 7-11 Calibrated hydraulic conductivities

Model layer	Hydraulic conductivity (m/s)	Hydraulic conductivity (m/d)
1	5E-6 – 1E-3	0.43 – 86.4
2	1E-6 – 5E-5	0.09 – 4.32
3	5E-10 – 1E-7	4.3E-5 – 8.6E-3
4	5E-5 – 4E-4	4.3 – 34.6
5	5E-8 – 3E-6	4.3E-3 – 0.26

Table 7-12 Calibrated steady-state aquifer flows (inflows to the aquifer as positive, outflows as negative)

	Steady-state aquifer flow (million m^3/a)
Recharge	**88.5**
Ocean net	**-29.7**
Langebaan Lagoon UAU net	-0.6
Langebaan Lagoon LAU net	-5.1
Saldanha Bay UAU net	-0.8
Saldanha Bay LAU net	-5.9
St Helena Bay UAU net	-1.1
St Helena Bay LAU net	-4.2
Rivers net	**-53.9**
Berg net	-11.1
Sout, Groën & Tributaries	-29.5
Abstraction	**-4.9**

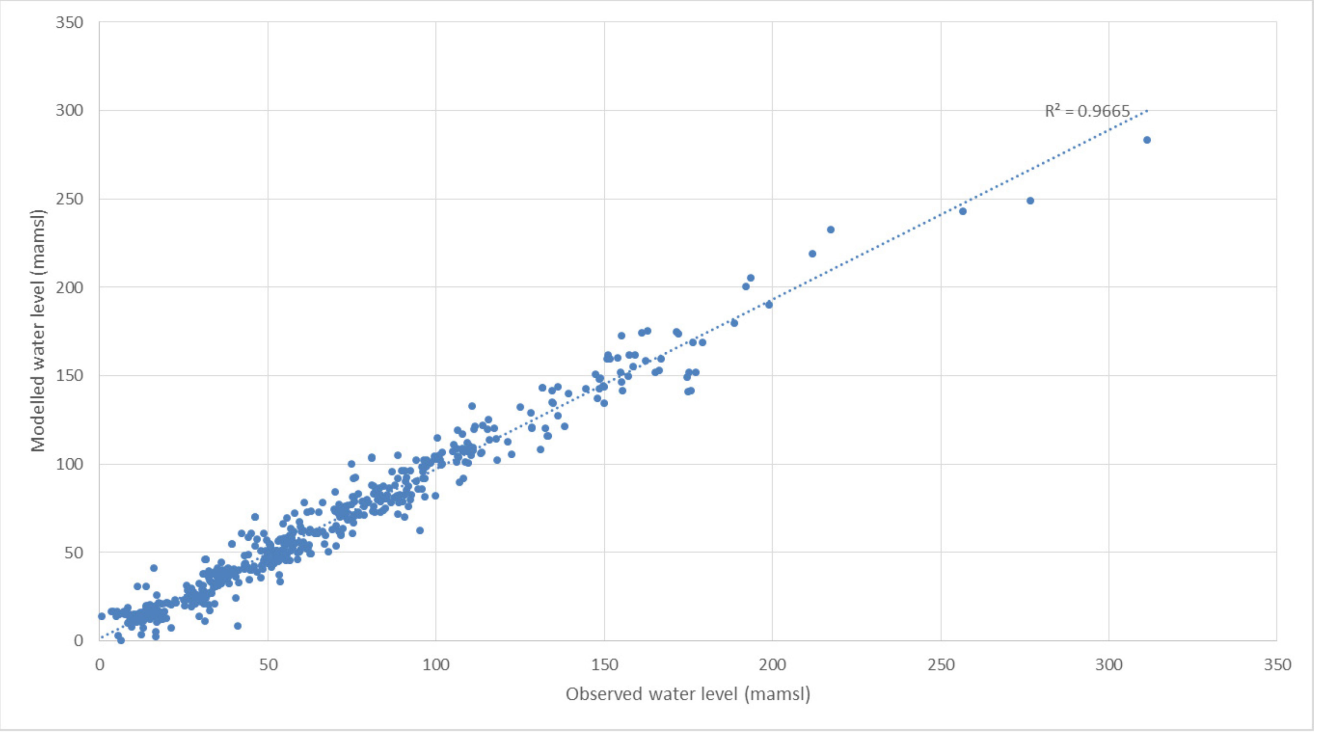


Figure 7-30 Modelled versus observed groundwater levels of the steady-state calibration

Figure 7-31 Simulated groundwater contours (with simplified location map) for model node layer 1 across the model domain

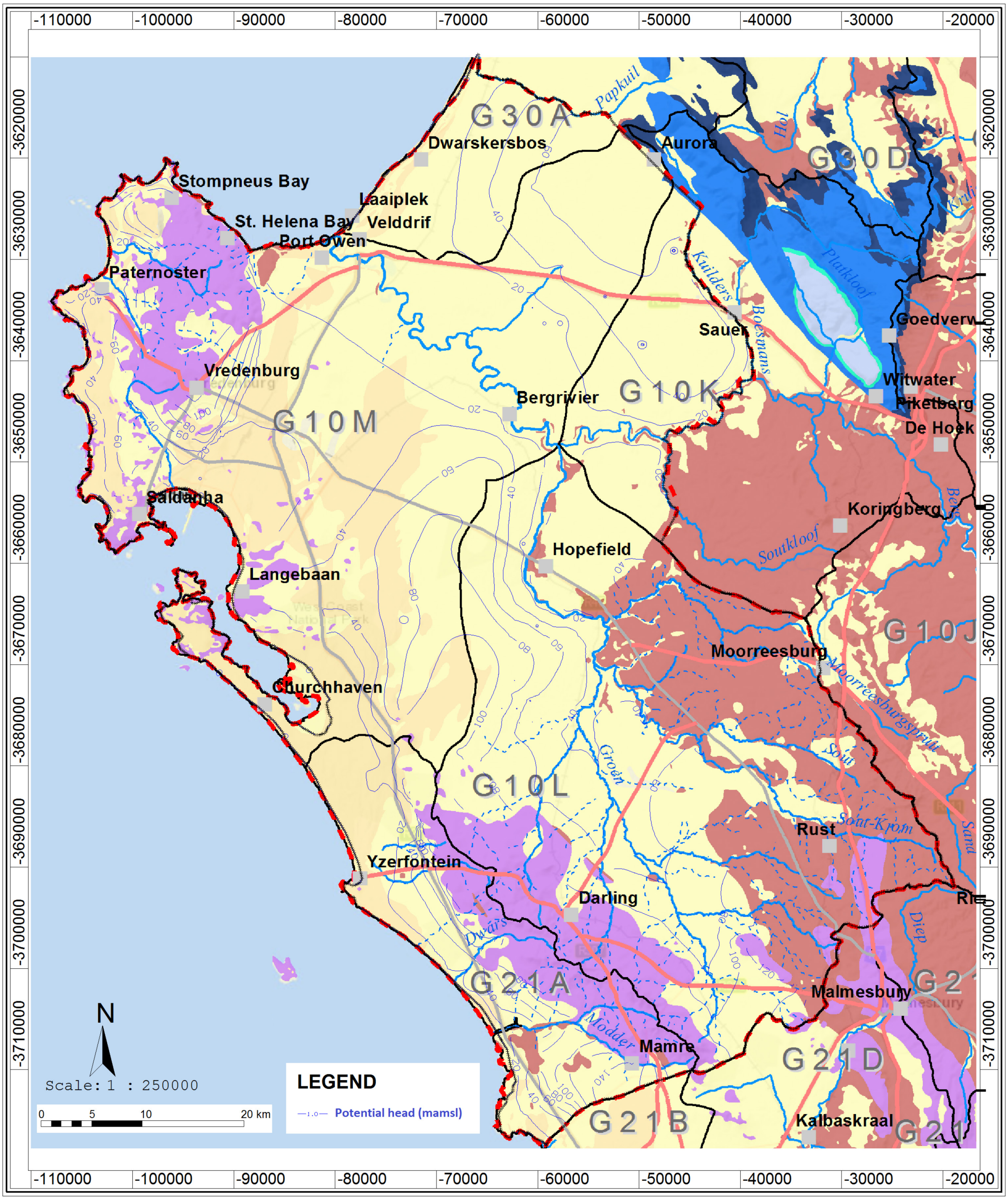


Figure 7-32 Simulated groundwater contours (with simplified geological map) shown for UAU only (model node layer 1, at locations where element layer 2 is present)

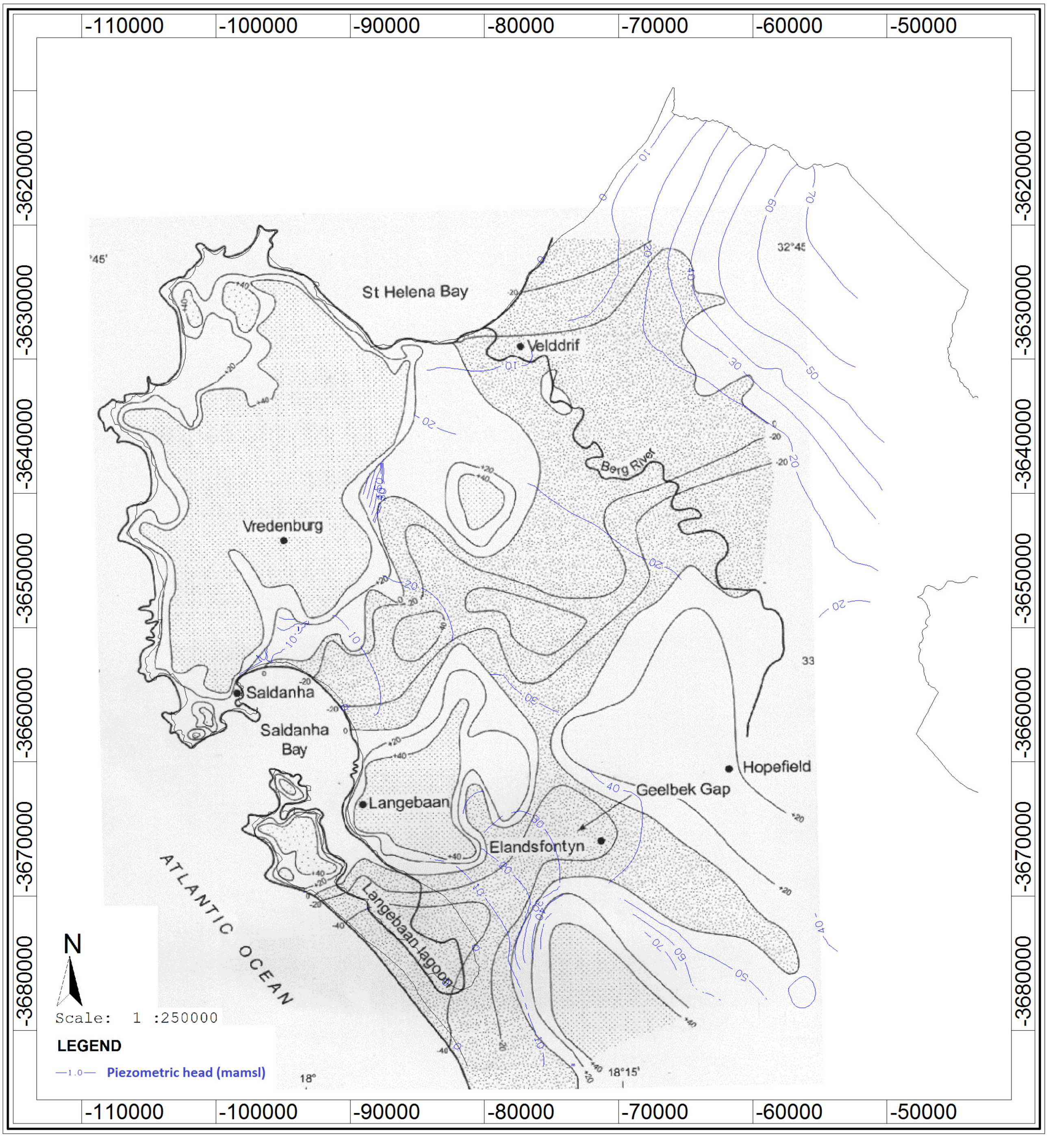


Figure 7-33 Simulated groundwater contours (with basement topography, Roberts and Siegfried, 2014) shown for model LAU only (node layer 4)

7.16. Predictive Simulations

7.16.1. Scenarios

The steady-state model replicates a base case of dynamic equilibrium prior to initiation of abstraction by the WCDM and the Langebaan Road wellfield, in November 1999. The predictive scenarios are modelled in steady-state, and replicate a future state of dynamic equilibrium under the specified abstraction regime at the WCDM wellfield (Table 7-13). The impacts illustrated (drawdown and change in flow regime) would only be reached once the equilibrium is established.

The status of the WCDM boreholes has changed over time, with some more in use than others, and some having been vandalised. For the purposes of the model scenarios, the abstraction rates listed in Table 7-13 were divided equally between the four potentially available abstraction boreholes at the WCDM wellfield. In addition to including municipal abstraction at the wellfield, the scenarios also incorporate the increased private (dispersed) abstraction, post November 1999 (section 7.14.3.3).
The aims of the case study assessment included generating the hydrogeological outputs that are required to inform a sustainability assessment of abstraction from the LRAS at the Langebaan Road wellfield (section 7.2):

- Source of water when maintainable or recommended yield is abstracted (i.e. impact on natural flow regime)
- Quantified relationship between aquifer fluxes and increased abstraction.

The scenarios listed in Table 7-13 are adequate for illustrating the maintainable aquifer yield from this wellfield, and illustrating the capture principle concepts central to this project. The aquifer's response to abstraction is related to number of boreholes (impact on drawdown), and distance between abstraction and discharge point (impact on flow regime). As such, the results of these scenarios can only be taken as an illustration of abstraction from this wellfield rather than an indication of the yield of the whole aquifer should wellfield(s) be optimally sited across it. Several additional scenarios are possible with the model, and would be beneficial for answering other resource questions and to support aquifer management. Additional scenarios are listed in the recommendations (section 7.17).

Table 7-13 Details of predictive scenarios

Scenario	WCDM wellfield abstraction (million m^3/a)	Dispersed abstraction (million m^3/a)
Base case	0	4.94
1	1.35	6.53
2	3.50	6.53
3	5.50	6.53
4	7.00	6.53
5	12.00	6.53

7.16.2. Results

The impact of the simulated abstraction on water levels is shown as drawdown (base case piezometric level minus the piezometric level result of the scenario) in Figure 7-34, Figure 7-35, Figure 7-36, Figure 7-37 and Figure 7-38 for model scenarios 1 to 5 respectively. Due to model accuracy, only drawdown above 2 m is shown. The modelled drawdown at the WCDM wellfield is shown in Table 7-14. The modelled aquifer flows are shown in Table 7-15 compared to the base case flows, and are shown graphically with increased abstraction in Figure 7-39.

Table 7-14 Modelled drawdown at WCDM wellfield

Borehole:	G45633	G45636	G45632	G45634
Scenario	Drawdown (m)			
Available drawdown (rest water level minus clay base)	37	38	33	31
Scenario 1	8	8	8	8
Scenario 2	21	22	22	22
Scenario 3	34	35	36	36
Scenario 4	51	54	56	54
Scenario 5	164	180	186	178

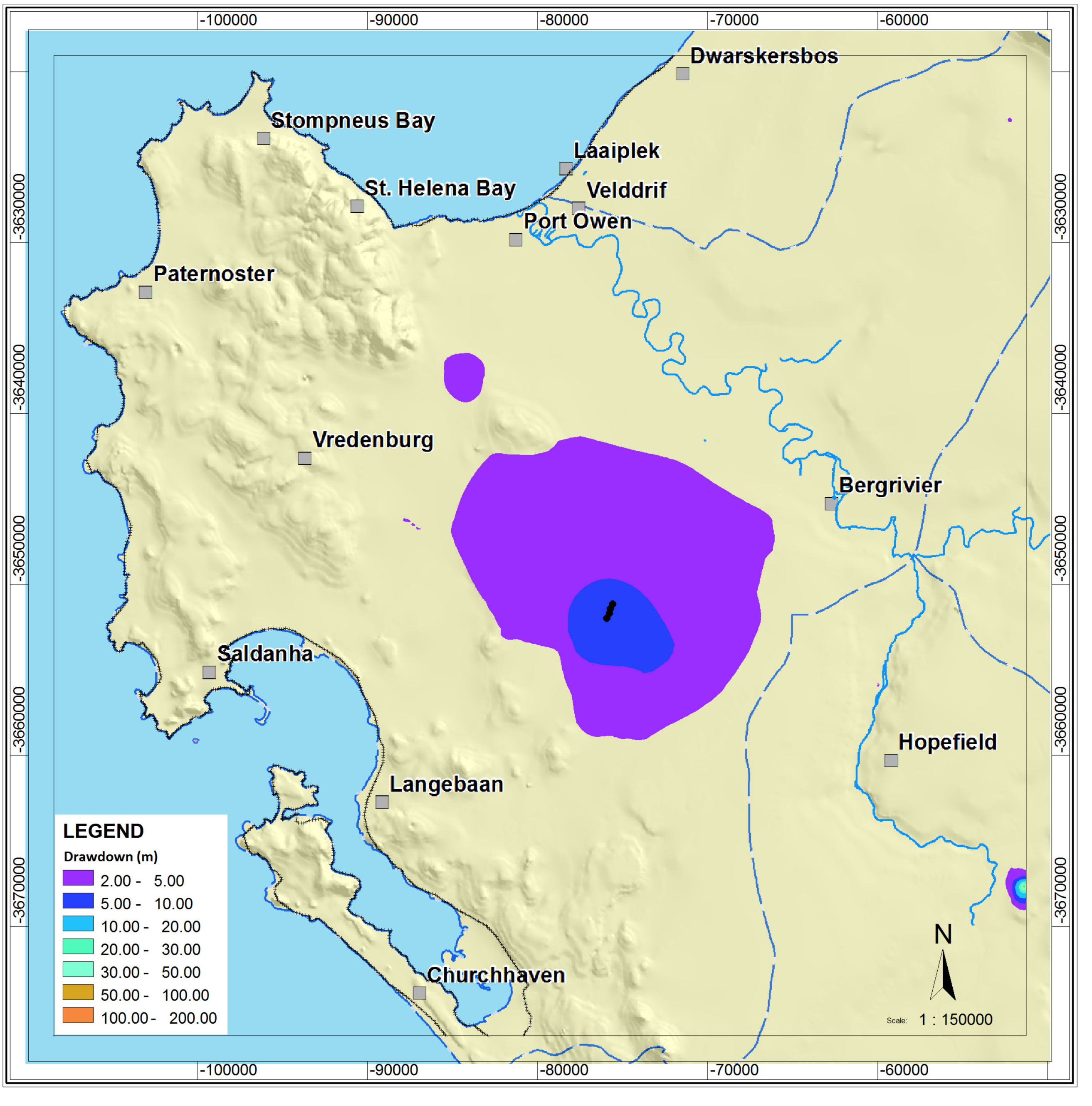


Figure 7-34 Simulated drawdown for scenario 1 for model node layer 4, LAU

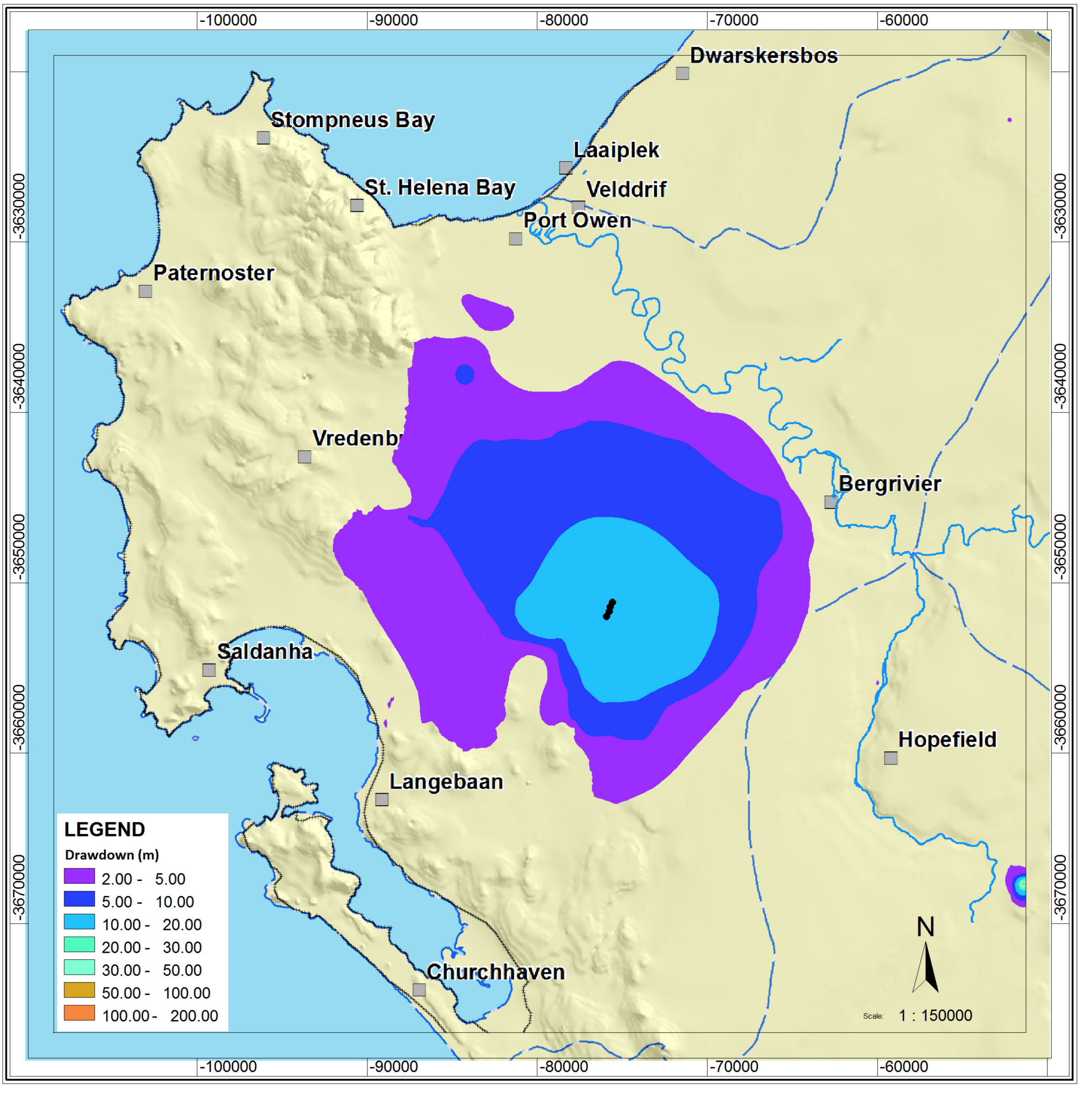


Figure 7-35 Simulated drawdown for scenario 2 for model node layer 4, LAU

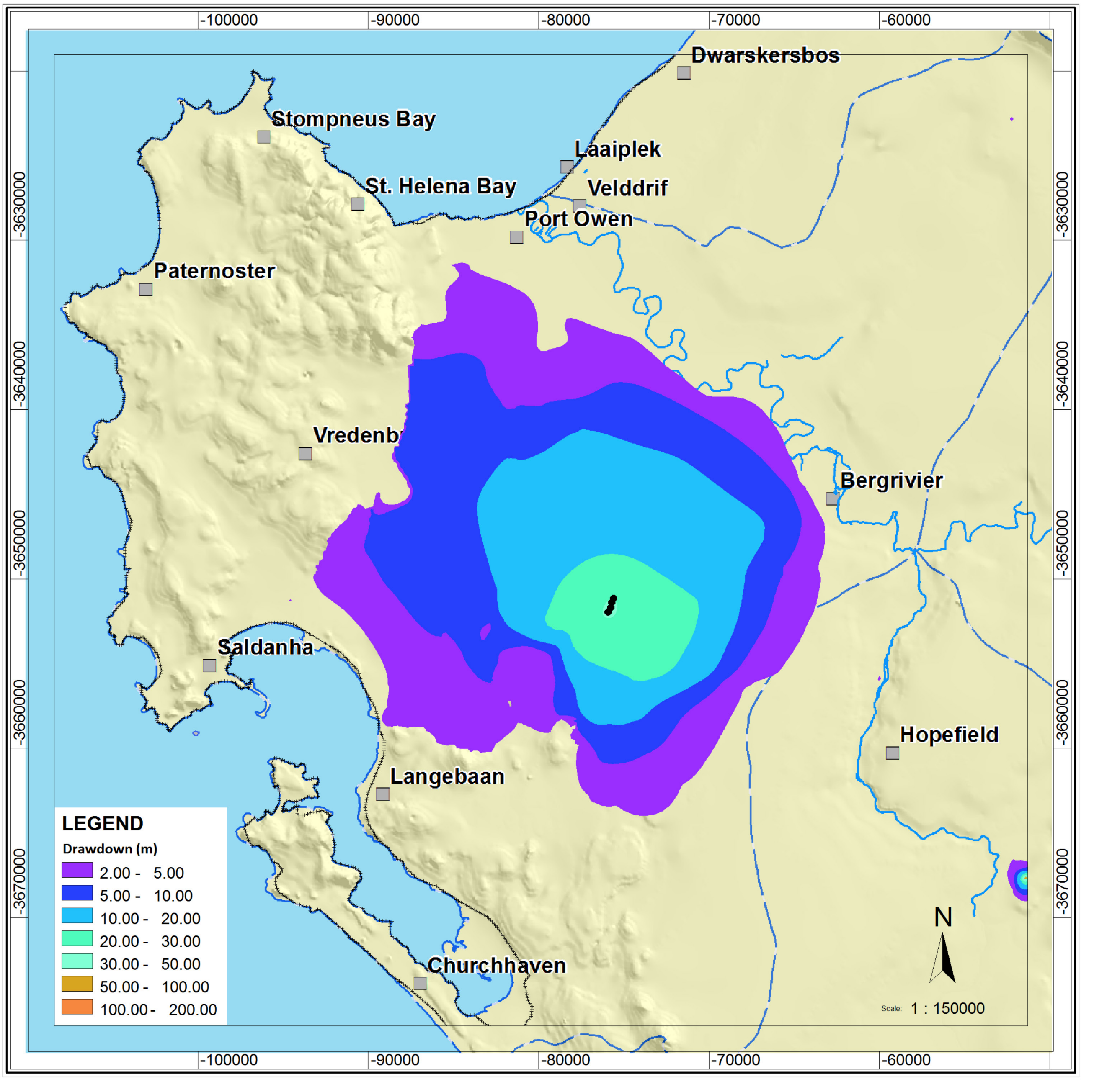


Figure 7-36 Simulated drawdown for scenario 3 for model node layer 4, LAU

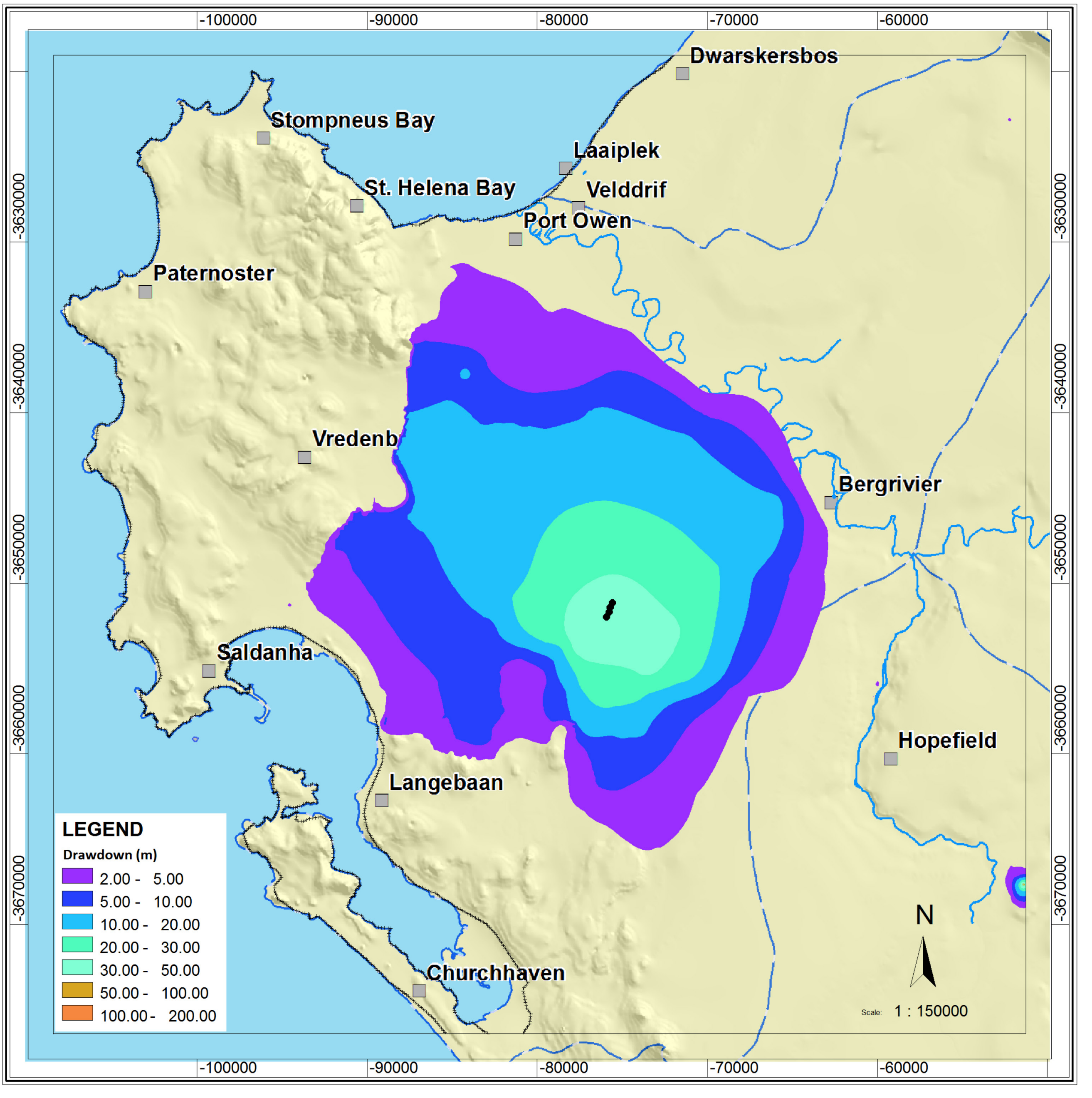


Figure 7-37 **Simulated drawdown for scenario 4 for model node layer 4, LAU**

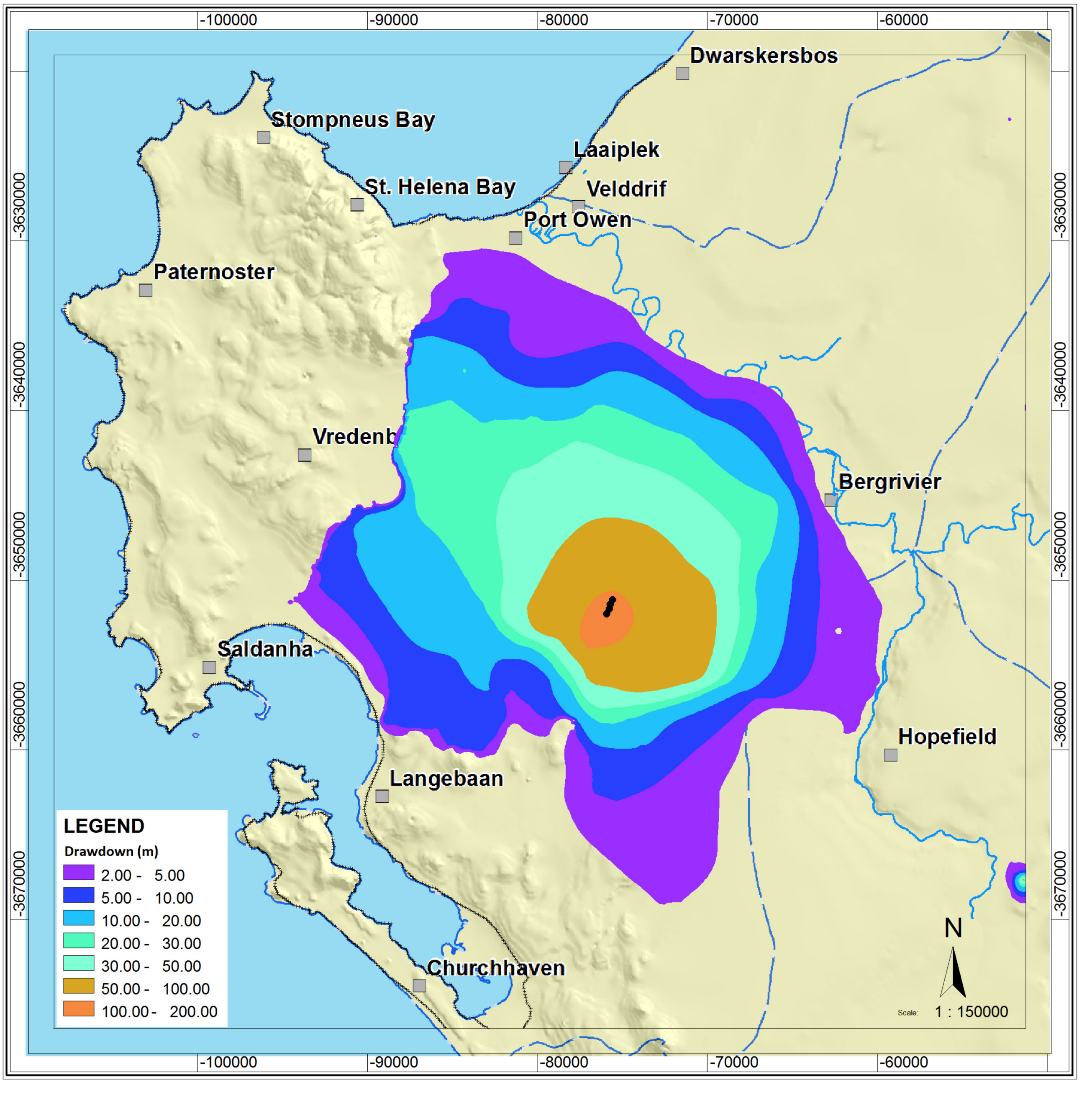


Figure 7-38 **Simulated drawdown for scenario 5 for model node layer 4, LAU**

Table 7-15 Model flow results for scenarios

Aquifer Flow	Base case	Scenario 1		Scenario 2		Scenario 3		Scenario 4		Scenario 5	
	million m^3/a	million m^3/a	Change %	million m^3/a	Change %	million m^3/a	Change %	million m^3/a	Change %	million m^3/a	Change %
Recharge	**88.5**	**88.5**	**0%**	**88.5**	**0%**	**88.5**	**0%**	**88.5**	**0%**	**88.5**	**0%**
Ocean net	**-29.7**	**-28.8**	**-3%**	**-27.9**	**-6%**	**-27.0**	**-9%**	**-26.2**	**-12%**	**-23.9**	**-19%**
Langebaan Lagoon UAU net	*-0.6*	*-0.6*	*0%*	*-0.6*	*-1%*	*-0.6*	*-1%*	*-0.6*	*-1%*	*-0.6*	*-2%*
Langebaan Lagoon LAU net	*-5.1*	*-5.1*	*-1%*	*-5.0*	*-2%*	*-5.0*	*-3%*	*-5.0*	*-3%*	*-4.9*	*-4%*
Saldanha Bay UAU net	*-0.8*	*-0.8*	*-2%*	*-0.7*	*-5%*	*-0.7*	*-8%*	*-0.7*	*-9%*	*-0.7*	*-15%*
Saldanha Bay LAU net	*-5.9*	*-5.4*	*-9%*	*-4.6*	*-23%*	*-3.8*	*-36%*	*-3.1*	*-48%*	*-1.0*	*-83%*
St Helena Bay UAU net	*-1.1*	*-1.0*	*-12%*	*-0.9*	*-17%*	*-0.9*	*-21%*	*-0.8*	*-25%*	*-0.7*	*-32%*
St Helena Bay LAU net	*-4.2*	*-4.2*	*-1%*	*-4.2*	*-1%*	*-4.1*	*-1%*	*-4.1*	*-1%*	*-4.1*	*-2%*
Rivers net	**-53.9**	**-51.8**	**-4%**	**-50.6**	**-6%**	**-49.5**	**-8%**	**-48.8**	**-10%**	**-46.0**	**-15%**
Berg net	-11.1	-10.3	-8%	-9.2	-18%	-8.2	-26%	-7.5	-33%	-5.1	-55%
Abstraction	**-4.9**	**-7.9**	**59%**	**-10.0**	**103%**	**-12.0**	**143%**	**-13.5**	**174%**	**-18.5**	**275%**
Abstraction at WCDM wellfield	**0.0**	-1.4	n/a	-3.5	n/a	5.5	n/a	-7.0	n/a	-12.0	n/a

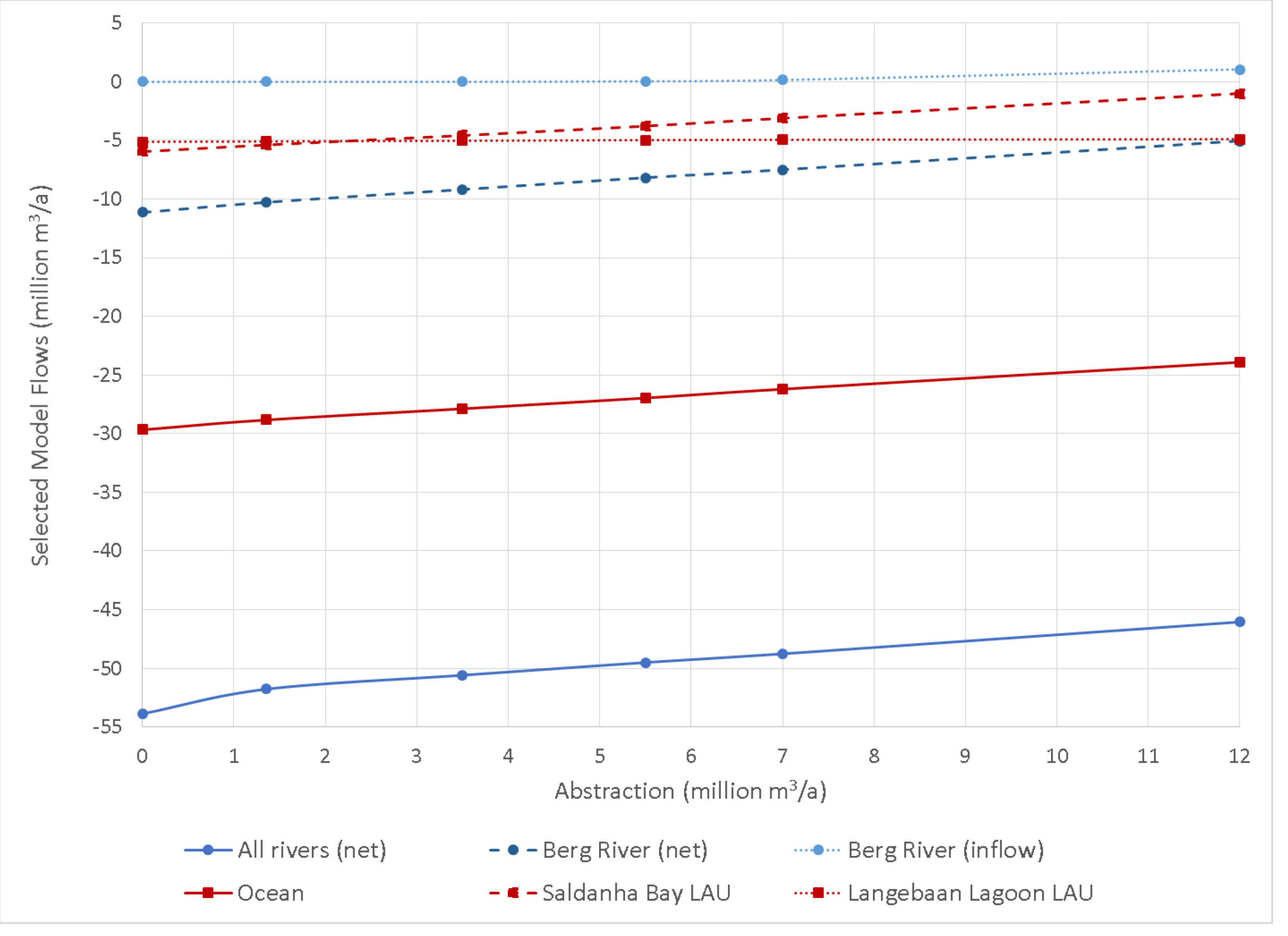


Figure 7-39 **Simulated changes in model flows with increasing abstraction**[13]

7.16.3. Discussion of Results

Impacts of abstraction (water level, flow regime) are related to the abstraction location (distance to discharge points such as ocean and river). The number of boreholes over which the abstraction is divided also has a large influence on the induced drawdown. Therefore the results can only be interpreted as an indication of abstraction from the four boreholes located at the WCDM wellfield.

The spatial extent of the radius of influence is fairly stable. (As dictated by flow theory, the position of the radius of influence depends on duration pumped not on abstraction rate, so it will actually be identical between scenarios although it appears to grow because of displaying drawdown above 2 m). But the magnitude of drawdown clearly increases with increased abstraction rate; from 8 m and 22 m in scenario 1 and 2 respectively, to ~35 m, ~50 m and ~180 m in scenarios 3, 4 and 5.

The maintainable aquifer yield (as defined in section 2.1.2) relates to a yield that can be maintained without runaway drawdown (such as drawdown next to a hydraulic barrier), and met by reduced discharge and or enhanced recharge, with a reduction in storage. Each of the scenarios tested reaches a new equilibrium in which abstraction is met by reduced discharge to surface water and oceans, and in some cases enhanced recharge from surface water, hence reflect maintainable aquifer yields. However, in the case of abstraction from the Langebaan Road wellfield, two other factors are important and could limit the yield:

- Inducing recharge from the Berg River is deemed unacceptable, due to increased salinity in the lower reaches of the Berg River

[13] Abstraction at WCDM is plotted rather than total abstraction per sceanrio.

- Maintaining the LAU in a confined state (not necessarily artesian) has been deemed important in previous investigations, on the assumption that this would prevent ingress of poorer quality water from the overlying UAU to the LAU.

Abstraction of 3.5 million m^3/a (scenario 2) induces drawdown that is well within the available drawdown limits (Table 7-14), whereas abstraction of 5.5 million m^3/a (scenario 3) appears to be at the limit of abstraction (from four boreholes) as the induced drawdown exceeds the available at two of the four boreholes. Again, the drawdown is dependent on number of boreholes: 5.5 million m^3/a distributed over six wells in the vicinity of the WCDM wellfield may allow the LAU to remain confined.

Table 7-15 and Figure 7-39 reveal significant information regarding the flow regimes of the aquifers:

- Wellfield abstractions of 1.35, 3.5 and 5.5 million m^3/a cause reduced discharge to the ocean and reduced discharge to surface waters. The reduction in discharge to surface waters, plus the drawdown information, (and the fact that only the UAU is in direct hydraulic connection with surface water in the model), indicates that recharge from the UAU to the LAU is likely enhanced under abstraction, and flow from the UAU to surface waters reduces to compensate.
- Although net discharge to surface water courses remains negative (i.e. net discharge occurs), Figure 7-39 shows that inflow to the model (UAU) from the Berg River occurs under a wellfield abstraction of 7 million m^3/a (and increases further under abstraction of 12 million m^3/a). The results suggest that these abstraction rates (regardless of distribution across more boreholes in the region of the WCDM wellfield) exceed acceptable impacts of abstraction from this location in the LAU. This does not necessarily mean that abstraction of 7 or 12 million m^3/a dispersed across several wellfields would also be unacceptable. The result illustrates the importance of considering local impacts when setting available yields; a mass balance on the aquifer scale does not illustrate this change in flow.
- Discharge from the LAU to Saldanha bay decreases by 9% under scenario 1, increases to 23% under scenario 2 and increases to 36% under scenario 3 (wellfield abstraction of 5.5 million m^3/a).
- The discharge from the LAU to the Langebaan Lagoon is also impacted (although by an essentially undetectable amount), decreasing by 1%, 2% and 3% for abstractions of 1.35, 3.5 and 5.5 million m^3/a respectively. The cone of depression elongates along the basement depression that connects the aquifer systems. The result confirms the hydraulic connection in the LAU across the LRAS and EAS aquifer systems, which is dictated by the model input data for basement topography being below 0 mamsl in the Geelbek gap, hence model layer 4 is continuous between the two aquifer systems, and as such a transmission within the high hydraulic conductivity layer is possible (also illustrated with contours in Figure 7-33).
- The discharge from the UAU to the ocean (at Saldanha Bay and at St Helena Bay) is also impacted, the impact increasing with increasing abstraction. This result suggests that under abstraction, recharge from the UAU to the LAU is enhanced, decreasing discharge from the UAU to compensate.
- Discharge from the LAU to St Helena Bay also reduces with increased abstraction, but only slightly, suggesting that abstraction at the WCDM wellfield intercepts water flowing to Saldanha Bay. This result is also related to the thickness and distribution of the LAU.

7.17. CASE STUDY SUMMARY

7.17.1. Key Results and Comparison to Other Studies

The results suggest that the maximum yield from the Langebaan Road wellfield which maintains confined conditions and prevents ingress from the Berg River is greater than 3.5 million m^3/a and less than 5.5 million m^3/a. Furthermore, abstraction of 5.5 million m^3/a does not induce ingress from the Berg River, but the drawdown at the wellfield reaches the available drawdown limit. However, if this abstraction rate was spread between more boreholes in the same location, the aquifer flow results would be similar (i.e. no ingress from the Berg river), and the drawdown would be lower and therefore likely to be acceptable.

The case study analysis has achieved its aim of quantifying impacts (overall aquifer flow balance and impact on water levels) for varying abstraction rates at the WCDM wellfield, which had not yet been documented (section 7.2, 7.12). The source of the water currently abstracted under the rate of 1.35 million m^3/a is reduced discharge; natural discharge to Saldanha Bay and St Helena Bay reduces (quantities given in Table 7-15). Recharge or leakage from the UAU to the LAU is likely enhanced, and flow from the UAU to surface waters reduced to compensate. The maximum yield from the Langebaan Road wellfield (in its current arrangement with four boreholes) which maintains confined conditions and prevents ingress from the Berg River is greater than 3.5 million m^3/a and less than 5.5 million m^3/a. Whether this abstraction rate can be considered sustainable depends on assessment of the ecological impact of the reduced groundwater discharges to the receiving environments. The response time has not been quantified as the model has been constructed in steady-state (see recommendations, section 10.3). The findings suggest much higher abstraction rates are available than previous estimations of yield from the WCDM wellfield (section 7.12):

- Woodford and Fortuin (2003, cited in DWAF, 2008c) suggested that the sustainable yield from the LAU in the LRAS was 1.5 million m^3/a, which is equivalent to the current abstraction at the WCDM wellfield. DWS (2016) also refer to this restriction, and go on to suggest that for further groundwater development, artificial recharge using surplus water from the Berg River in the winter months is recommended.
- WCDM (2009) predicted, based on an empirical relationship, that abstracting 1.3 million m^3/a from the wellfield would result in the piezometric level declining to the base of the clay layer after approximately 30 years, and recommended that the wellfield be operated at 1.113 million m^3/a.

Other estimates of yield are not directly comparable as they are for larger areas (LRAS and EAS), or for both UAU and LAU together. For example, Timmerman (1988) concluded that the maximum long term yield of the LAU in the LRAS and EAS was 6.32 million m^3/a. Woodford (2003, cited in WCDM, 2005) determined a combined exploitation potential for the LRAS and EAS of approximately 14.0 million m^3/a. These estimates appear more in line with estimates generated here.

The findings, specifically the aquifer flow balance, provide information towards some previously uncertain aspects of the conceptual understanding of the aquifer:

- There is concern within the scientific community interested in the region, about inducing recharge from the UAU to the LAU if the aquifer becomes unconfined, as this has the potential to cause ingress of poor quality water. However, abstraction is likely to be already increasing recharge from the UAU to the LAU, by increasing the head difference. In turn this will reduce discharge from UAU to surface waters (Berg River). Maintaining the LAU in a confined state is supported, but increased flow from UAU to LAU is unavoidable if the groundwater resources are to be used. Geochemical studies are required to quantify the current influence of the clay on water quality, and the influence of the clay when the pressure gradients are impacted by abstraction (see recommendations, section 10.3).
- Distant recharge mechanisms have previously been postulated for the LAU, given the competence of the clay aquitard in preventing a response in the UAU to pumping in the LAU. The broad behaviour of the LAU is replicated in the model, without a distant recharge source for the LAU. The model supports that the LAU is recharged via vertical leakage through the clay.
- If the basement data is correct, and the basal Elandsfontyn Formation is present everywhere where basement is <0 mamsl, then there is hydraulic connection in the LAU between LRAS and EAS.
- There is currently concern over potential impact on Langebaan Lagoon from developments in the EAS region that dewater the UAU. The aquifer flow results suggest that the Langebaan Lagoon predominantly receives water (90%) from the LAU. This result is however a maximum discharge from the LAU, given the way that the LAU is represented at the coastline, and requires further testing (see recommendations, section 10.3).

7.17.2. Case Study-specific Recommendations

Increased abstraction

The initiation of this case study recognised the potential for previous under-estimation of available groundwater resources, particularly at the WCDM wellfield, given that assessments of the 'sustainability' of groundwater abstraction had to date mostly revolved around considerations of reduction of water level. The above results suggest that the current abstraction at the WCDM wellfield may indeed be an under-exploitation of available groundwater resources.

DWA (2010b) also suggest that reducing the abstraction below 1.5 million m^3/a may have been over-cautious, and that there would be benefit in 'further stressing the aquifer by increasing abstraction from the wellfield over the short term'.

Based on the results presented here, it is recommended that abstraction be increased at the wellfield, prior to further consideration of artificial recharge, and the response closely monitored.

The numerical model developed is considered low confidence, based on the calibration undertaken and the predictive simulations including stresses greater than those applied in calibration (Barnett et al, 2012). It is recommended that this aquifer assessment be combined with current information on the wellfield performance to enable development of Low-Confidence Operating Rules (Table 2-2).

The case study has provided useful insights for the overall project and decision framework, which are described in section 9. Furthermore, the analysis has also contributed information applicable for the management of the West Coast aquifers, specifically the LRAS. The established numerical model has, to date, only been used for a fraction of the available (and recommended) analyses, and it provides a valuable tool for ongoing groundwater management in the area. In support of this, the following recommendations are made:

Improved monitoring and data sources to enable model update

Geochemical analysis: Recharge from the UAU to the LAU via seepage through the clay has generally been questioned in the scientific community based on water quality differences between the aquifers. However, there has not been an assessment of the cation exchange capacity of the clay, and geochemical interactions could be responsible for altering the water quality after seepage. A geochemical analysis is required to investigate this (under natural conditions and increased hydraulic gradient with pumping), which, if shown to be correct, would have implications for available abstraction yield given that the motivation for preventing ingress from UAU to LAU would be removed.

Aquifer-specific water level data:

- A more detailed analysis of existing available water level records should be completed (in association with DWS regional office) in order to ascribe additional boreholes to the LAU or UAU, based on a response to rainfall (for boreholes with enough records). This analysis could increase the number of dedicated observation points for the LAU, enabling a separate calibration.
- The installation of multi-level water level monitoring points in various locations, screened in the UAU and LAU separately, would greatly benefit understanding of hydraulic connectivity between the two.

Data on clay layer: The thickness and hydraulic conductivity of the clay layer has dominant control on the vertical transmission of water between the UAU and LAU (recharge and impact of pumping). Furthermore, the model representation of discharge from the LAU to Saldanha Bay and Langebaan Lagoon is simplified, assuming direct hydraulic connection between LAU and coast. If significant thickness of clay is present at the coastline, with the LAU remaining confined some distance to the west, then the discharge from the LAU to the Bay and Lagoon is exaggerated. More detailed analysis of existing data (sourcing borehole logs, etc.) is recommended to further develop data on the top and bottom elevation and continuity of the clay layer. In addition, various model representations of the coastline should be constructed to establish a range of possible discharge rates.

Private abstraction rates: Current groundwater use rates are poorly known for the area, which is a common problem in water resource assessments. WCDM monitoring reports detail neighbouring users of the LAU, but abstraction rates are not monitored. The WARMS database has significant uncertainties, the existing hydrocensus information is outdated, and the extent of unregistered use is unknown. A validation and verification process is underway by DWS, and the model should be updated with the latest groundwater use data, once the data is available.

Recharge: Modelled groundwater levels are directly related to the assigned recharge rates and hydraulic conductivities. Different datasets for recharge, combined with different hydraulic parameters can calibrate equally well to the same observed water level dataset, yet these two models would respond differently to the abstraction scenarios tested here. Modellers typically assume available recharge rates are fixed, and only vary hydraulic conductivity. However, recharge is particularly poorly quantified for the West Coast aquifers, with several GIS-based estimates (GRA II, and others), but a lack of (known) point estimates of recharge to supplement. DWA (2010b) highlights the same uncertainty as a key priority area for future research in the area.

Updated model

It is recommended that the model be updated. Some updates rely on the above additional analyses being completed and/or additional data being available:

- It is recommended that the LAU be independently calibrated. Some dedicated observation points do exist already; more aquifer-specific water level data would enable an accurate calibration.
- Following independent calibration of the LAU and UAU, the vertical connectivity between the aquifers should be further interrogated.
- Detailed information on the basement elevation north of the Berg River (Adamboerskraal Aquifer System) was sourced post model calibration (Timmerman, 1985a). Basement data in this region of the model was based largely on sparse NGA data. A preliminary assessment of the basement data shows the model calibration would benefit from incorporating the data.
- The EAS region proved challenging to calibrate likely due to local heterogeneities in the UAU. Data recently gathered for private developments in the EAS area (pump test and water level monitoring) should be sourced and used to improve calibration in that region.
- It is recommended that the model undergo a transient calibration. Several datasets are available to support a transient calibration, including the response to pumping at the WCDM wellfield from 1999 to 2005, and EAS pump test data at private developments. The later pumping period at the WCDM wellfield (2006–2012) could be used as a validation dataset. Once calibrated in transient, the response time should be determined, as a critical input to the aquifer management approach.

Additional predictive simulations

It is recommended that an additional scenario be tested under constant head philosophy (rather than constant abstraction rate), to determine the abstraction rate when the piezometric head in the LAU is maintained some metres above the bottom of the clay layer. The yield will be between 3.5 and (closer to) 5.5 million m^3/a.

The model abstraction scenarios establish the available yields from the existing WCDM site only. The maintainable yield of the whole LRAS system (UAU and LAU) is likely to be significantly higher, especially if targeted through dispersed abstraction. A model scenario is recommended whereby abstraction boreholes are spaced along the existing pipeline route from Misverstand dam to Saldanha, in order to contribute to bulk supply. Various abstraction rates should be tested, up to the required future projected supply shortfall for the Saldanha Bay Local Municipality. It is recommended that this relatively simple test be conducted as a matter of priority to contribute to the considerations over future supply interventions, which focus on desalination and water re-use, and do not currently consider enhanced groundwater use.

Although the LAU is connected across the EAS and LRAS, an additional wellfield in the EAS would be feasible, if it is suitably sited to harness natural discharge to Langebaan Lagoon with minimal impact on the existing LRAS wellfield. It is recommended that the relationship between abstraction and reduced discharge to Langebaan Lagoon is quantified, the potential ecological impact assessed and maximum groundwater abstraction established.

The DWS is currently considering injection of 14 million^3/a of Berg River flood water to the LRAS and/or EAS to enhance available yields. The model can be used to test injection scenarios, testing the feasibility of various locations and direct injection versus seepage. However, given the current under-utilisation of the WCDM wellfield, and the groundwater resources as a whole, the necessity of injection is uncertain. Furthermore, injection (direct or via seepage) needs to be implemented in conjunction with significant abstraction to generate the ‘space’ (illustrated in another case study in Seyler et al, 2016). It is recommended that abstraction at the WCDM wellfield be increased, and the above mentioned scenarios be tested in the model before investment in injection infrastructure or testing.

The vehicle for undertaking the above recommendations varies. Some should be internalized by DWS regional office, in their monitoring and management of the aquifers in the area. Some will be accommodated under the PhD currently being completed as part of this WRC project. The assessment of future abstraction scenarios is also part of Water Resources Classification, hence the above recommendations should be considered by the concurrent DWS project to establish water resources classes and resource quality objectives in the Berg (ex) WMA.

8. CASE STUDY 2: MALONEY'S EYE, STEENKOPPIES COMPARTMENT

8.1. BACKGROUND

The Maloney's Eye is the only naturally discharging spring located in the Steenkoppies Dolomitic Compartment. In March 2007, eight out of nine of its constituting springs stopped flowing, which is an incident that has drawn much attention. This incident caused major concern to the downstream users as the spring discharges form part of the flow of the Magalies River. At the time of this incident the flow measured at a record low of 0.05 m^3/s (or 1.58 million m^3/a). The long-term average of the flow from 1908 to 2015 is estimated to be around 14.13 million m^3/a. The management and sustainability of the aquifers underlying the catchment area was brought into question, as the Steenkoppies Compartment has been widely used for agricultural purposes. Barnard (1997) as well as Holland et al (2009) suggested that there has been a noticeable decline in the Eye's discharges due to the utilisation of the compartment's groundwater.

When the discharging springs stopped flowing in 2007, the Magalies River Crisis Committee requested that all groundwater abstraction from the compartment be stopped temporarily, to offer sufficient time for the flow of the Eye to recover. This however did not materialize as cessation of groundwater use would have impacted the agricultural sector, which is a major contributor to revenue and employment in the area. The surrounding farmers established the Steenkoppies Aquifer Management Association in 2007 with the aim of furthering the interests of groundwater users of the Steenkoppies Compartment.

Considering the ongoing conflict between up- and downstream water users in a heavily utilised system, the Maloney's Eye is an interesting case study for sustainability challenges and application of the decision framework.

8.2. APPLICATION OF DECISION FRAMEWORK TO CASE STUDY

The decision framework documented in section 6 shows the various aspects required for development of operating rules that accommodate a capture principle-based assessment of yield. The purpose of the decision framework is essentially to enable maximised groundwater use in a sustainable manner by incorporating a capture approach. This is important for this specific case study as the Maloney's Eye flow (natural aquifer discharge) is vulnerable due to the poorly understood abstraction of groundwater from the Steenkoppies Compartment by groundwater users. Similarly to the West Coast aquifers case study, there has not been a previous capture-based assessment of the aquifer yield. Although groundwater abstraction is assumed to be the cause of reduced flow at the Eye, the source of the currently abstracted water has not been assessed (i.e. the impact on, or departure from, the natural flow regime) and the relationship between abstraction and reduced discharge has not been quantified. Again, the most appropriate application of the decision framework (or, more broadly, the recommended approach that it represents) is with a determination of the maintainable aquifer yield and the response of the aquifer to abstraction. The development of a numerical model is required to understand the maintainable aquifer yield, as input to stakeholder decisions over sustainability and ultimately to form the basis of an aquifer management plan. The numerical model is described in sections 8.12 to 8.14, based on the conceptual model described in sections 8.3 to 8.10 and summarised in section 8.11.

8.3. LOCATION, TOPOGRAPHY AND DRAINAGE

The Maloney's Eye, shown in Figure 8-3, is located approximately 40 km northwest of Johannesburg and falls within the local municipal area of Mogale City and Randfontein. The Steenkoppies Compartment's western boundary stretches to the west of Tarlton and the eastern boundary is the lower Zwartkrans Compartment. From previous studies (Holland et al, 2009) the total area of the Maloney's catchment has been estimated at 311 km^2 and the Steenkoppies Dolomite Compartment estimated at approximately 212 km^2.

The Steenkoppies Compartment is characterised by a nearly even, slightly undulating plain. The compartment is confined between the steep slopes of the Witwatersrand Supergroup to the south and the Pretoria Group in the north. Virtually no surface drainage features exist in this area. The elevation is in the range of 1700 mamsl along the Witwatersrand watershed and Magaliesberg mountain range, while the Magalies River flood plains, from west to east, vary in elevation between 1400 mamsl to 1200 mamsl.

The upper part of the Magalies River catchment (with a total drainage area of nearly 1000 km^2) hosts the Steenkoppies Compartment. The Magalies River catchment forms part of the upper Crocodile River subsystem, which is located within the Crocodile (West) and Marico WMA. The tertiary drainage region is A21 and the Steenkoppies Compartment forms part of the upper regions of quaternary drainage region A21F.

8.4. HYDROLOGY AND CLIMATOLOGY

The climate in the area is typical of the South African 'Highveld', characterised by warm summers, with 80% of the rainfall experienced as thunderstorms, and cool, dry winters with cold nights. Climatic data from six meteorological stations was compiled into a single, representative time series of rainfall from 1908 to 2011 (Holland et al, 2009, Wiegmans et al, 2012). For this investigation, that data was augmented with data from the Deodar rainfall station, maintained by the Agricultural Research Council (in Pretoria), and the time series thereby extended to 2015. The compilation of the 107-year time series of rainfall records is shown in Figure 8-1.

The MAP was calculated at 670 mm based on a 107-year period (Table 8-1). The distribution of mean monthly rainfall is shown in Figure 8-2. Although it is evident that over the last 10 years rainfall was above the long-term average (675 mm versus 670 mm, Table 8-1), there is a recent lack of significant intense rainfall events (above 850 mm). The highest monthly measured rainfall for the last decade (2006 to 2015) was 246 mm compared to 349 mm for the previous decade (1996 to 2005). Monthly rainfall only exceeded 180 mm in two months of the last decade, compared to the five months of the previous decade. Above average rainfall events trigger exceptional groundwater recharge events and lead to an increase in water levels and the recovery of flow of the Maloney's Eye (Holland et al, 2009, Wiegmans et al, 2012). The lower rainfall over the last two decades can be an explanation of the lower flows at the Maloney's Eye, but increasing or continued groundwater abstractions in the eye's catchment are likely to contribute as well.

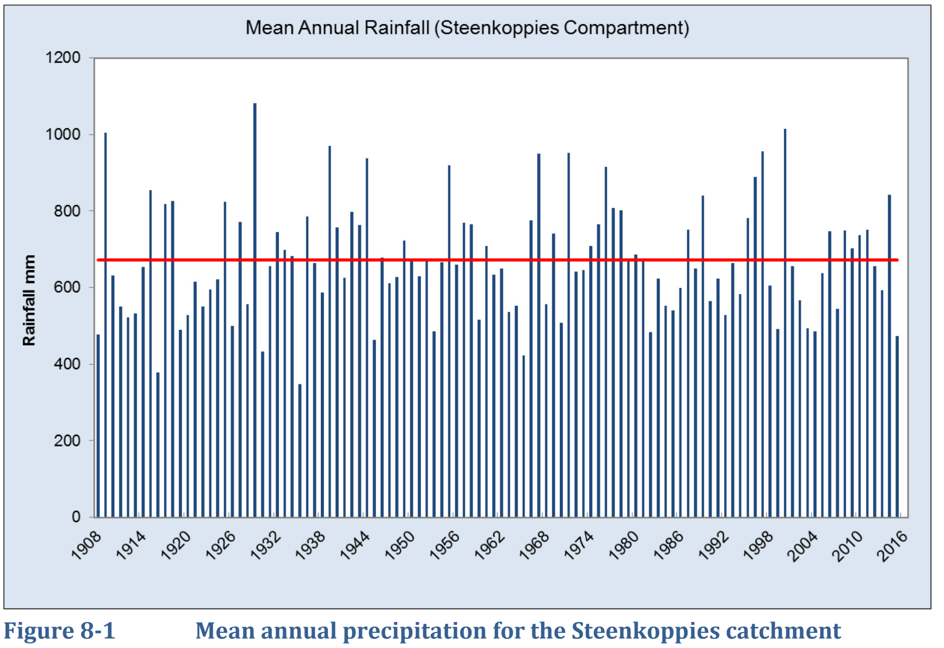


Figure 8-1 Mean annual precipitation for the Steenkoppies catchment

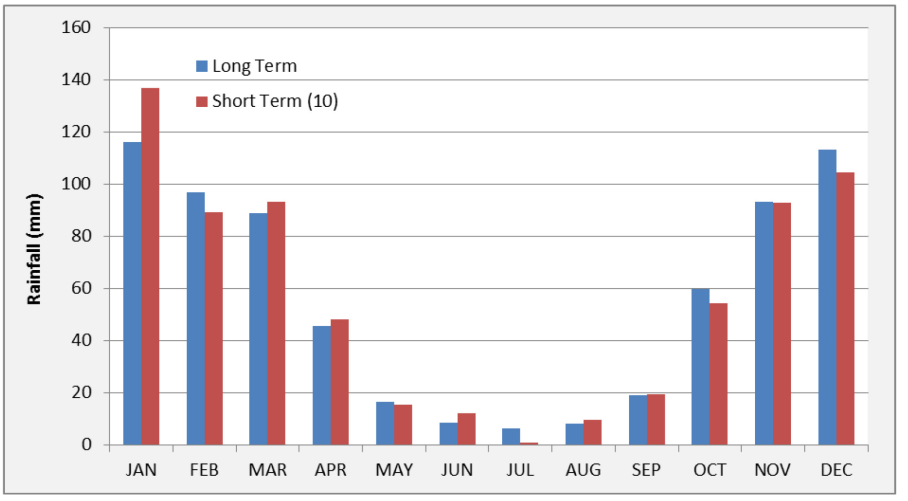


Figure 8-2 Mean monthly precipitation for the Steenkoppies catchment

Table 8-1 MAP for the Steenkoppies Compartment

Year (record)	MAP (mm)	MAP (mm)	
		Min	Max
Long Term (1908 to 2015)	670.2	348	1081
Short Term (2005 to 2015)	675.7	474	842

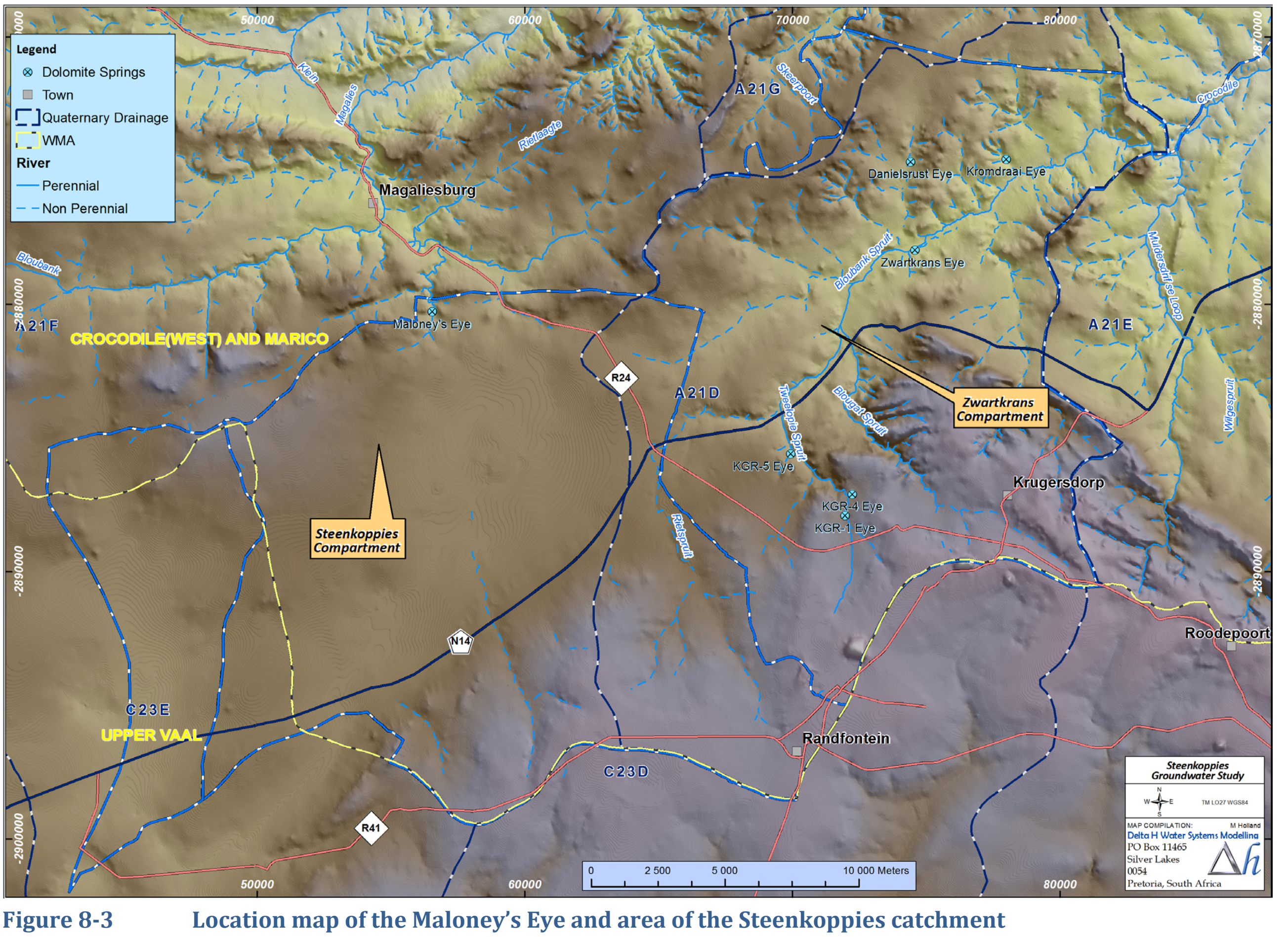


Figure 8-3 Location map of the Maloney's Eye and area of the Steenkoppies catchment

8.5. GEOLOGY

The up-tilting Pretoria Group strata make up the northern boundary of the Steenkoppies Compartment. The southern boundary is characterised by numerous rock types which include igneous basement rocks and the sedimentary succession of the gold-bearing Witwatersrand formations which forms the faulted rim of the Witwatersrand basin. Dipping off the western border of the Johannesburg Dome with an unconformable contact is the Black Reef Quartzite Formation (basal formation of the Transvaal Supergroup) which underlies the Malmani Subgroup dolomites of the Chuniespoort Group.

Based on the abundance of chert, the Malmani Subgroup has been subdivided into five dolomitic formations based on the abundance of chert. These formations are depicted in Table 8-2. Numerous intrusive dykes occur in the area. These dykes subdivide the dolomites into compartments and smaller sub-units.

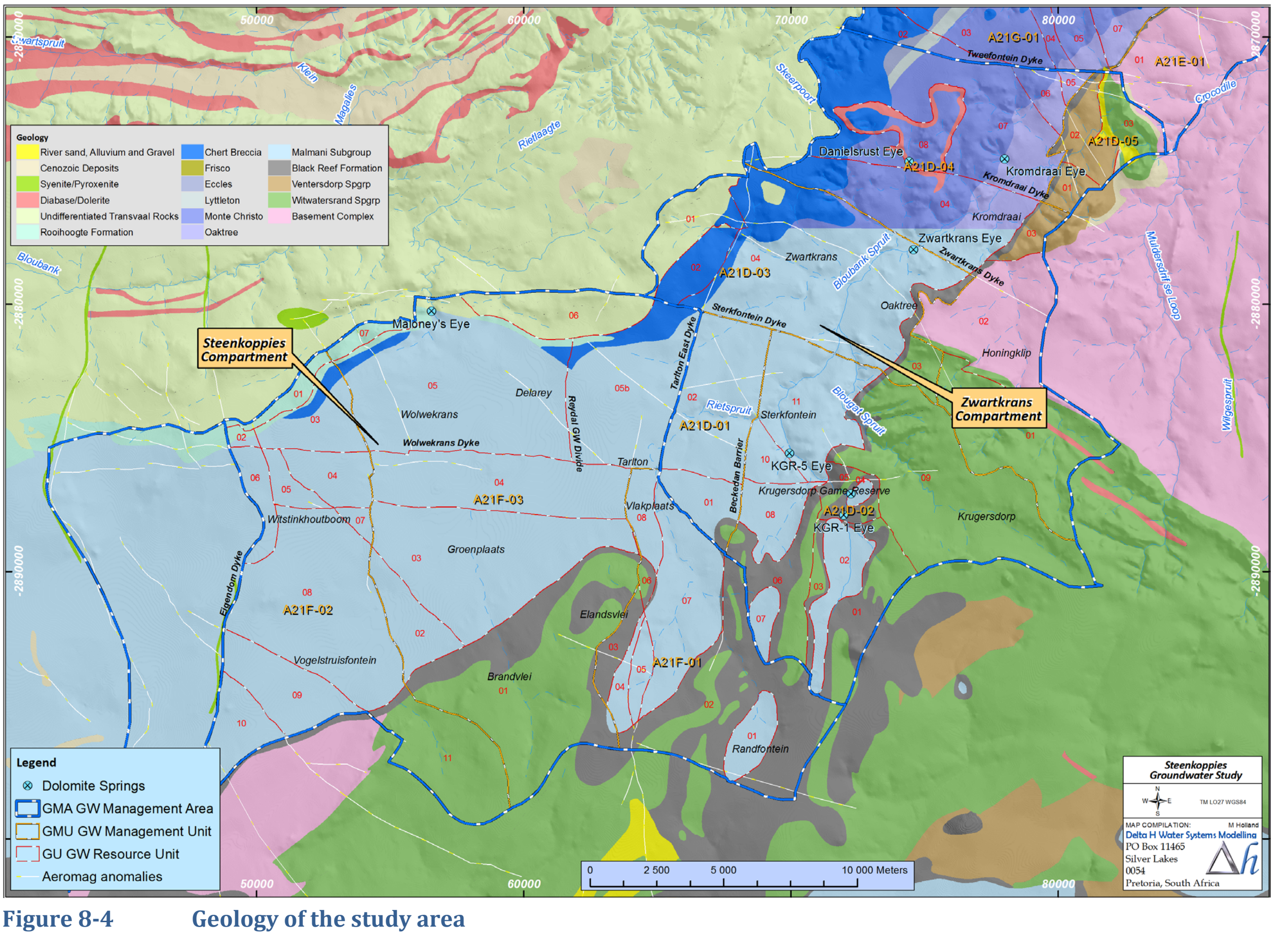


Figure 8-4 Geology of the study area

Table 8-2 Stratigraphic column of the study area (After SACS, 1980, Foster, 1984, Obbes, 2001)

Super Group	Group	Formation	Thickness (in m)	Lithology
TRANSVAAL	PRETORIA	Rayton	120	Shale, quartzite
		Magaliesburg	300	Quartzite
		Silverton	600	Shale
		Daspoort	80-95	Quartzite
		Strubenkop	105-120	Slate
		Hekpoort	340-550	Andesite
		Timeball Hill	270-660	Shale, diamictite, Klapperkop quartzite and ferruginous quartzite
		Rooihoogte	10-150	Quartzite, Shale, Bevets Conglomerate Member and Breccia
	CHUNIESPOORT	Frisco	30-158	Chert-free dolomite with some primary limestone and carbonaceous shale at the base
		Eccles	490	Chert-rich dark dolomite with stromatolitic and oolitic bands. Chert increases to the top
		Lyttelton	220-290	Chert-free dark dolomite with large stromatolites and sometimes with wad (complex residual soil mantle)
		Monte Christo	740	Alternate layers of chert-rich and chert-poor light coloured dolomite with stromatolites and oolites
		Oaktree	190-330	Chert-poor dark dolomite with interbedded layers of carbonaceous shale at the base
		Black Reef	11-30	Shale and quartzite. Arkosic Grit
WITWATERSRAND	CENTRAL RAND	-	2 880	Arenaceous, rudaceous rocks
	WEST RAND		5 150	Quartzite, reddish and ferruginous magnetic shales
	DOMINION		Unknown	Quartzite, conglomerate, shale, interbedded lava
BASEMENT COMPLEX				

8.6. AQUIFER SYSTEMS

The aquifers underlying the study area consist mainly of hard rock aquifers associated with the Witwatersrand Supergroup (quartzite), the Ventersdorp Supergroup (lava), Chuniespoort Group (dolomite), the Pretoria Group (quartzite and shale) and Karoo Sequence (Ecca Group) sandstone and shale. Post-Karoo dolerite dykes and diabase dykes traverse the study area to divide the dolomite into 'compartments'. It is generally accepted that very large transmissivities and storativities are associated with these dolomites (especially within the Monte Christo and Eccles Formations) where karstification is present. These chert-rich formations are generally favourable for large scale development of groundwater. The deep weathering of the dolomite can be indicated by gravity lows (Figure 8-5). A gravity survey along the Rietspruit valley and in the eastern and central part of the Steenkoppies Compartment was done initially by Bredenkamp et al (1986) and Kuhn (1986). Wiegmans et al (2012) extended the gravity survey to include the northern part of the compartment. Linear and lateral gravity lows were found, where the linear gravity features may have resulted from weathering of dykes, dissolution along faults and fissures, and the leaching of the dolomite at the contact zones with dykes or sills.

The width of the linear lows varied from 50–300 m depending on the degree of leaching of the dolomites and the lateral extent of the weathering of the dykes and fracture zones. Lateral lows are associated with the dissolution of dolomite formations, paleo-valleys filled with residual sediments and the intersection of more than one fracture or fault zones subjected to weathering processes. The weathered part of the dolomite, underlain with fractures and fissures, is highly heterogeneous. These karst aquifers are often characterised by a dual or triple porosity, comprising of solutional voids, fractures and the rock matrix (intergranular pores). While the fractures and the rock matrix provide most of the storage potential (low permeability), the conduits act additionally as (high permeability) drains.

The overlying Pretoria Group has very low primary permeabilities and weakly developed secondary permeabilities along faults and fractures. Once the dolomite is exploited, it is expected that the Pretoria Group will contribute groundwater to the dolomite (Kuhn, 1986). The hydraulic connection between the dolomites and the underlying Witwatersrand and Basement rocks is generally poor due to the low permeability values in these underlying rocks. As a result of the large permeability that dolomites exhibit and the presence of surface dissolution features like sinkholes, groundwater recharge in dolomites is significantly higher than in other hardrock aquifers.

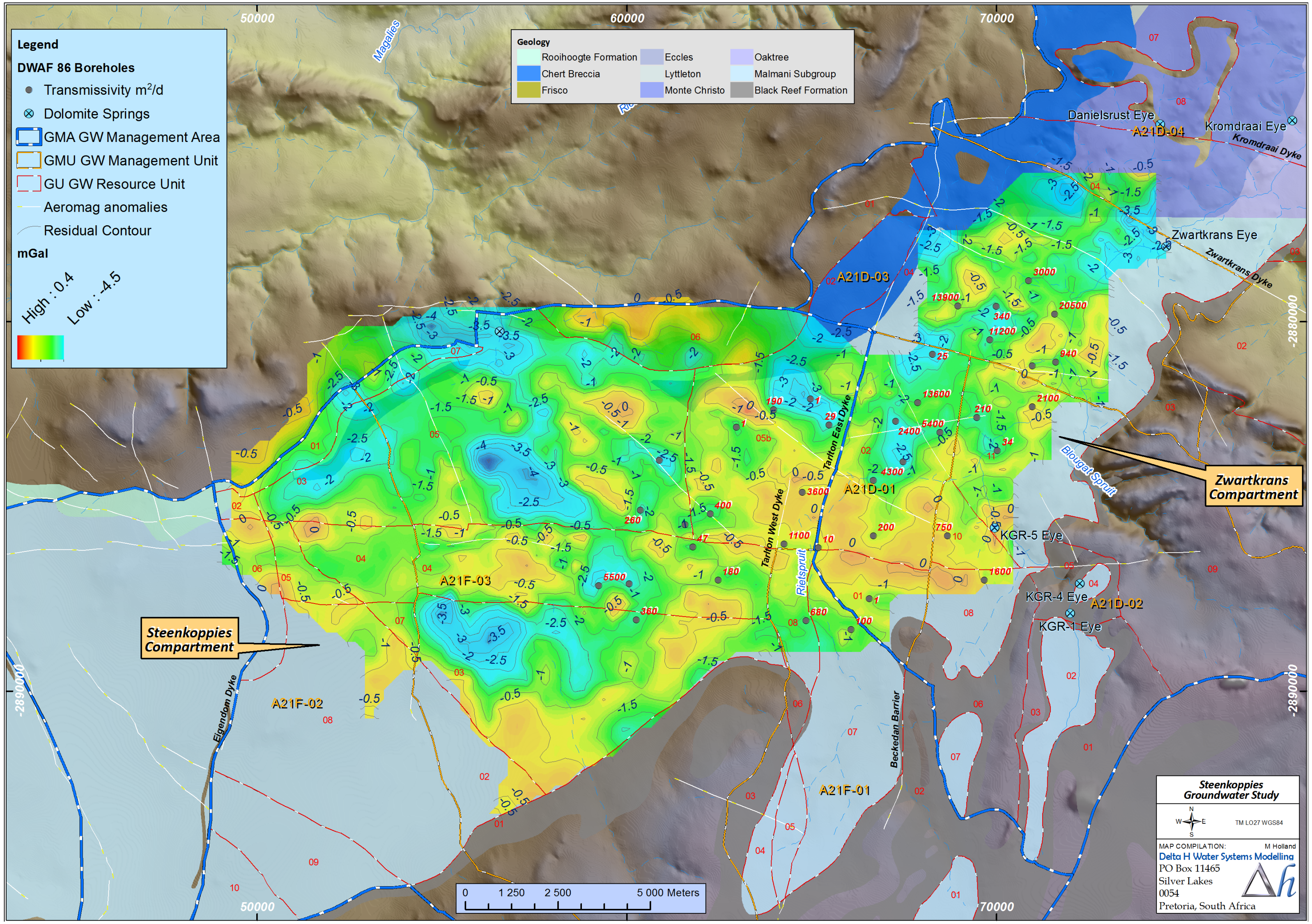


Figure 8-5 Gravity map of the Steenkoppies Aquifer (Wiegmans et al, 2012)

8.7. GROUNDWATER LEVELS AND FLOW DIRECTIONS

Vegter (1986) established that dolomites are effectively divided by dolerite or syenite dykes into 'compartments'. Other features such as brecciated faults and topographic divides may also form compartment boundaries. Groundwater conditions within each compartment may be relatively uniform and the water table surface fairly flat, whilst large differences in water levels between compartments may be found. Flow does occur between compartments, but usually on a smaller scale compared to flow within a compartment. The compartment boundaries are also often associated with spring lines and seepages as groundwater is forced to the surface, as is the case with the Maloney's Eye.

Based on the high-level delineation of the groundwater resources within the larger A21F quaternary catchment and the Steenkoppies Compartment, 14 GMUs have been delineated. The Steenkoppies Compartment consists of 6 GMUs, which can be seen in Figure 8-4, and represents an area of 332 km^2. Table 8-3 shows the groundwater level statistics of each GMU within the Steenkoppies Compartment.

Groundwater flow is from areas of higher piezometric values to areas of lower piezometric values, and its relation to topography is indicated by the fair correlation between groundwater levels and the surface elevation (Figure 8-6). The locality of the groundwater monitoring boreholes as well as groundwater level measurements (2009 to 2016) is shown in Figure 8-7.

Table 8-3 Groundwater Levels of the GMUs of the Steenkoppies Compartment

Groundwater management unit (GMU)	Area (km^2)	Number of measurements	Water levels (metres below surface)			Water levels (mamsl)		
			Min	Max	Mean	Min	Max	Mean
A21F-01	62.6	45	1.9	75.6	32.4	1505.0	1655.5	1565.9
A21F-02	78.5	37	11.3	107.9	63.1	1494.2	1603.1	1523.7
A21F-03	94.5	77	6.1	98.2	64.8	1483.0	1707.2	1508.9
A21F-04	12.9	50	3.5	84.9	54.5	1483.9	1576.6	1514.1
A21F-05	27.5	15	7.2	126.5	52.8	1479.8	1580.1	1536.1
A21F-06	56.2	107	3.8	178.9	63.4	1437.5	1571.3	1498.2

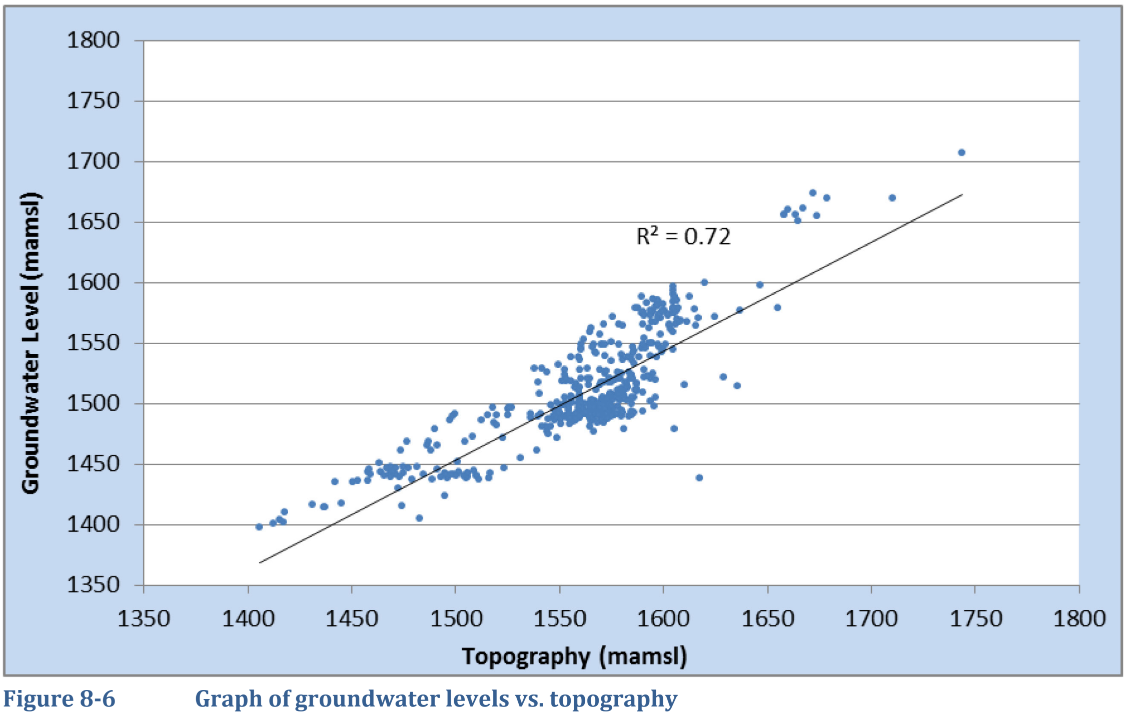


Figure 8-6 Graph of groundwater levels vs. topography

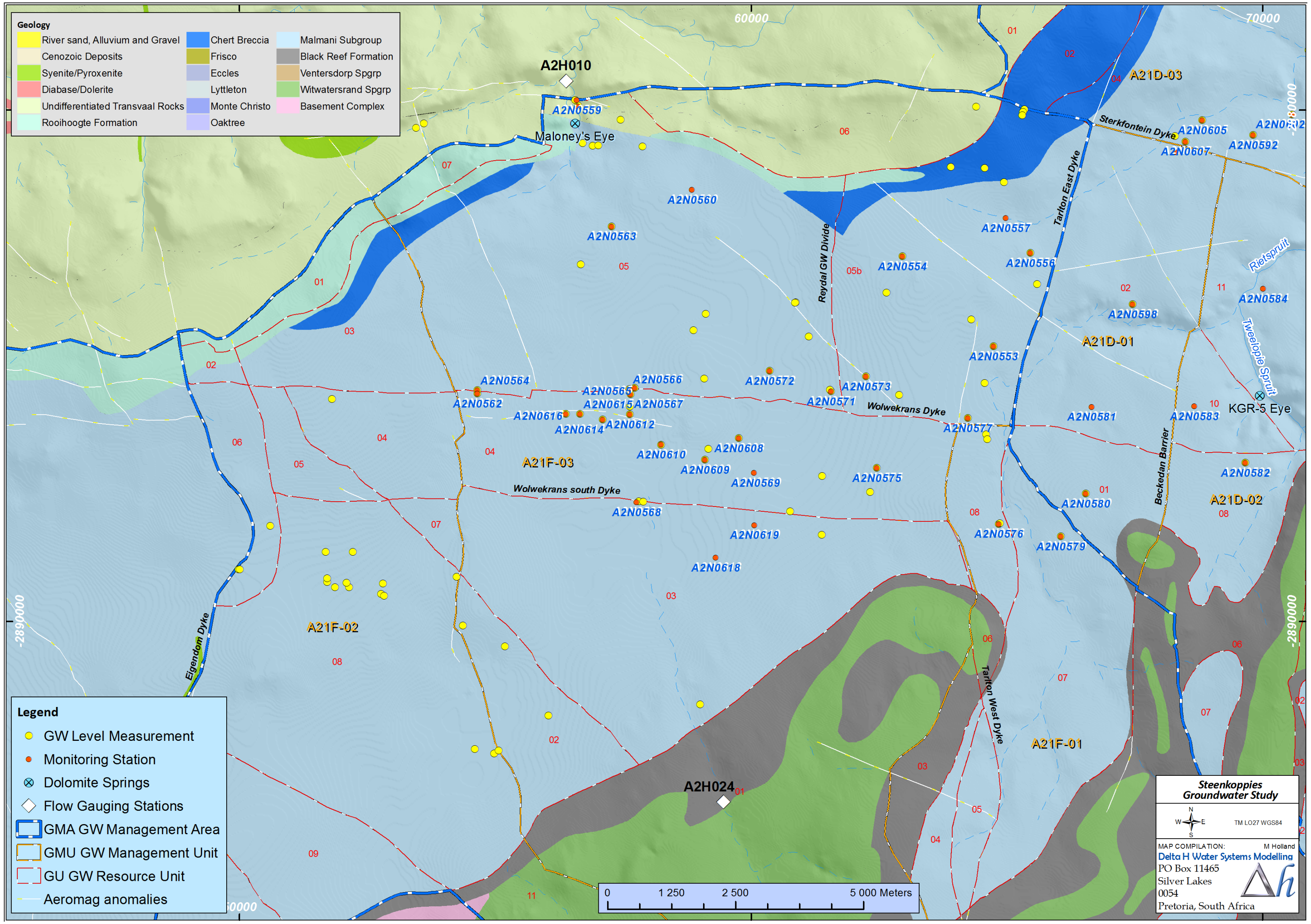


Figure 8-7 Location of monitoring boreholes within the Steenkoppies Compartment.

8.8. HYDRAULIC PARAMETERS

No recent aquifer test data is available for the study area, but aquifer tests in the 1980s confirmed very high transmissivity values across the dolomite aquifers associated with the karstification of dolomite. The distribution of transmissivity values collated for the area is shown in Figure 8-8 and summarised in Table 8-4.

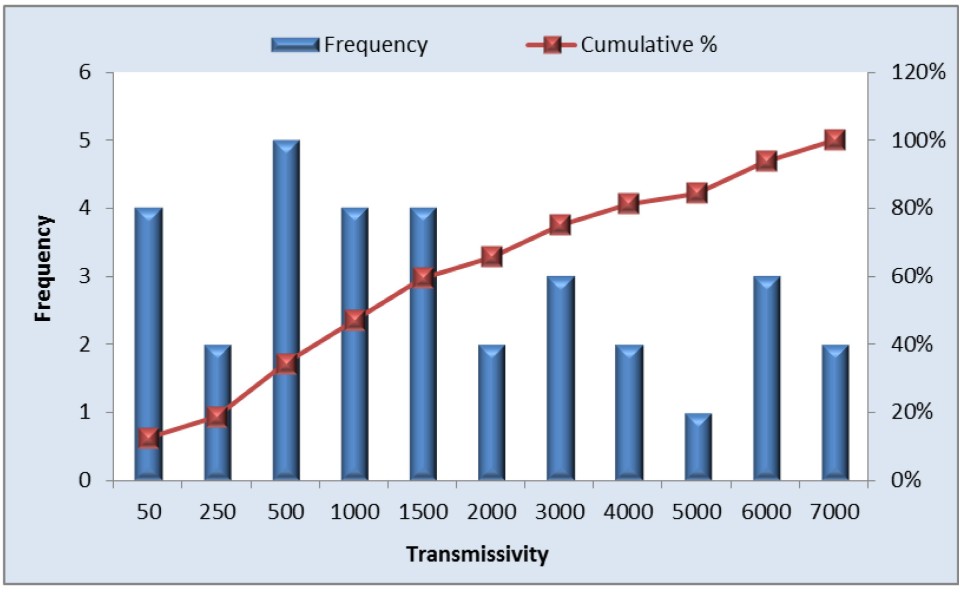


Figure 8-8 Transmissivity cumulative frequency plot for aquifer tests within the Steenkoppies Compartment

Table 8-4 Transmissivity values within the Steenkoppies Compartment

Parameter	Transmissivity	Conductivity	
	m^2/d	m/d	m/s
Arithmetic mean	1 923	12.8	1.5E-04
Median	1 145	7.6	8.8E-05
Geometric mean	751	5.0	5.8E-05
Harmonic mean	73	0.5	5.6E-06

8.9. SOURCES AND SINKS

Infiltration of precipitation is the main recharging mechanism in this area and most of the recharge of the Steenkoppies Compartment occurs within the study area itself. Recharge of the aquifer occurs during the rainy season's precipitation. A summary of the recharge values (estimated with different methods), average precipitation and catchment area for the Steenkoppies Compartment compiled from different reports, is given in Table 8-5. It was established that the average recharge of the Steenkoppies Compartment is nearly 24 million m^3/a (Wiegmans et al, 2012). The high permeability of the dolomites in the area contributes to the considerably high recharge rate of the aquifer.

Smaller inflows from the surface drainage system, i.e. Brandvlei non-perennial tributary in the upper Steenkoppies Compartment, and the upper Rietspruit, also recharge the Steenkoppies Compartment. Both these drainage channels flow for some distance, but irrigation dams and leakage from the river bed into the underground network reduce the flow to virtually zero. Flow of the upper Rietspruit is not gauged, but it receives between 2.3 and 4.6 million m^3/a treated sewage effluent from the Randfontein Sewage Works facility. The DWS maintains a hydrological gauging station in the non-perennial tributary, Brandvlei (station A2H024, Figure 8-7) where records commenced in 1967. The average flow at the Brandvlei gauging station is around 0.4 million m^3/a.

The natural discharge of this area occurs at the Maloney's Eye and to the base of the Lower Magalies river drainage systems. The DWS maintains a hydrological gauging station downstream of Maloney's Eye (station A2H010), with records dating back to 1908. The long-term average flow at the Maloney's Eye to date is 437 ℓ/s (13.8 million m^3/a), but over the last 10 years the average flow rate reduced to 210 ℓ/s (5.8 million m^3/a).

A good correlation between the flow of the Maloney's Eye and the cumulative rainfall departure (CRD) (short-term moving average of 9 months and long-term moving average of 240 months) was obtained, confirming that rainfall is the main 'driving force' behind the groundwater system's dynamics (Figure 8-9). However, since 1997 the actual discharge is lower than the simulated discharge, which can be explained by increased abstraction from the aquifer (not accommodated in the model), especially during the drought periods 2002–2005 and 2007 when farmers relied heavily on groundwater for irrigation.

The earliest known borehole surveys were conducted by Hobbs (1980) and Bredenkamp et al (1986) and provide the earliest indication of the rate of groundwater abstraction. Estimates of the rate of groundwater abstraction are listed in Table 8-6. Crop production on the Steenkoppies Aquifer has increased steadily over the last 35 years, and so has the rate of groundwater abstraction for irrigation (more than 7.5 times since 1980). The crop area has increased 2.6 fold since 1997, while the yield of water abstracted has increased 1.6 fold (Vahrmeijer et al, 2013). Water use generally varies dependent on mean annual rainfall, crop type and crop distribution.

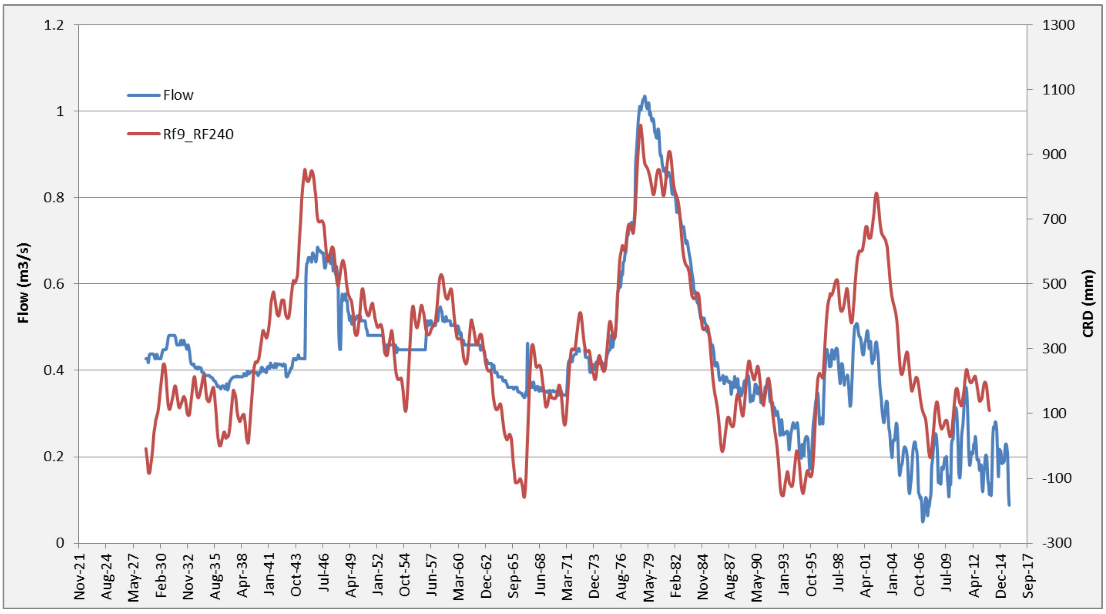


Figure 8-9 **CRD simulated flow (red) vs. Maloney's Eye measured flow (blue)**

Table 8-5 Published recharge values for the Steenkoppies Compartment

Reference	Method	Recharge as a percentage of average rainfall (%)	Average precipitation (mm)	Catchment area (km^2)
Bredenkamp et al (1986)	Estimated from spring flow	13.9	630	177
	Rainfall-recharge relationship	14.5	630	177
	Chloride ratio	32.0	630	177
	Darcy's Law	20.0	630	177
Barnard (1997)*	Chloride ratio	17.5	690	235
	Estimated from spring flow	9.2	690	235
Barnard (1997)	Saturated volume fluctuation method	16.0	690	214
Holland (2007)	CRD and CMB method	17.2	670	311
Wiegmans et al (2012)**	Numerical modelling	12.0	676	332

*Barnard (1997) used recharge values of 10% for dolomite and 6.5% for quartzite and shales

**Wiegmans et al (2012) used recharge of 12% for dolomite and between 2 and 4% for quartzite and shales

Table 8-6 Estimated groundwater abstraction for the Steenkoppies Aquifer.

Reference	Year	Crop area (ha)	Abstraction (million m^3/a)
Hobbs (1980)	1980	n/a	3.95
Bredenkamp et al (1986)	1986	n/a	13.45
Barnard (1997)	1997	1 952	19.00
Schoeman and Partners (2016)	1998	4 525	25.55
Schoeman and Partners (2016)	2015	5 035	29.92

8.10. GROUNDWATER QUALITY

The groundwater quality of the Steenkoppies Compartment is generally within an acceptable range. Some groundwater units, like the A21F-01 and A21F-04, receive effluent discharge water from the Randfontein WWTW (via Rietspruit) and therefore have elevated concentrations of chlorine and sulphate, which results in a somewhat poorer water quality. Previous data collected from a single sample showed groundwater unit A21F-06 had elevated iron and manganese concentrations, but it was concluded that a local source may be responsible for this (Wiegmans et al, 2012).

8.11. AQUIFER CONCEPTUAL MODEL

The Maloney's Eye is situated at the intersection of the Maloney's Eye dyke and the east-west striking fault zone. The Eye is situated within the shales/quartzites of the Rooihoogte/Timeball Hill Formations, therefore, the existence of the Eye could be attributed to a dyke of low permeability and the cross cutting of the fault zone representing the main water conduit from the dolomite into the shales and quartzites.

The recharge area of the spring is expected to include the quartzites and shales of the Witwatersrand Group which extends further south than the Steenkoppies dolomites. The Wolwekrans dyke and the Wolwekrans south dyke show little difference in water elevation and, based on the high transmissivities identified through pumping tests and gravity lows, it is expected that these dykes are perhaps more permeable than initially perceived. Therefore, large scale abstractions in groundwater unit A21F02 will also impact on the Maloney's Eye discharge due to the flat hydraulic gradient and interconnected groundwater units.

There remains uncertainty regarding the integrity of the Steenkoppies Compartment eastern boundary, as not enough data is available to confirm this boundary as a hydraulic barrier. The vertical extent of the groundwater system was determined by drilling results. The boreholes indicate a maximum weathering depth of about 35–60 m and a fracturing depth of up to 140 m below surface. A uniform thickness of 150 m for the aquifer(s) is therefore assumed.

8.12. NUMERICAL MODEL SETUP

8.12.1. Computer Code

The software selected for this case study is SPRING, and is described in section 7.14.1.

8.12.2. Model Domain

The model domain covers a surface area of 332 km^2 and is bounded by a surface water (and assumed groundwater) divide to the south, by the Tarlton Dyke to the east, the Eigendom Dyke to the west and the contact with the Pretoria Group to the north. Groundwater flow directions largely follow topography, and the groundwater basin geometry can be approximated by the surface water drainage geometry. These boundaries are represented numerically by what is referred to as a 'no-flow' boundary condition.

Figure 8-10 Finite-element mesh of the Steenkoppies groundwater model

The model domain was spatially discretised into 93 451 nodes on five node layers, which make up four finite-element layers with 102 442 elements (triangles and quadrangles) each. The model mesh accommodated the delineated karst zones, dykes and irrigation areas. The horizontal element size (side length) varies from a minimum of 10 m to a maximum of 50 m further away from the area of interest. The finite-element model was set up as a three-dimensional, steady-state groundwater model. The layers were arranged to represent the conceptual model, as shown in Figure 8-11 and described in Table 8-7.

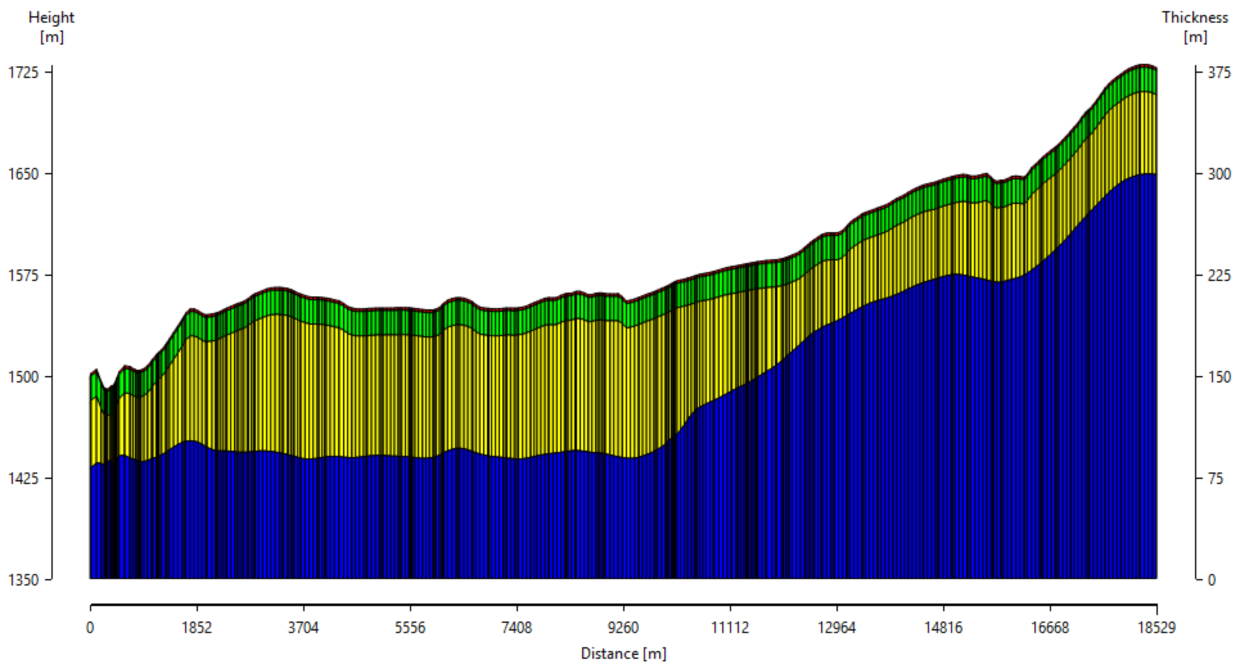


Figure 8-11 **Vertical grid layout across (N-S cross section; colours indicate numerical model layers only).**

Table 8-7 **Model layer arrangement**

Node Layer	Element layers	Aquifer feature	Data used for interpolation
I, top	I, top	Surface elevation	Digital Elevation Model (DEM) 5 m (Contours)
I, bottom	1	Soil zone	DEM – 2 m
II, bottom	2	Weathered aquifer	DEM – 35 m
III, bottom	3	Dolomite aquifer	70 m below water level
IV, bottom	4	Bottom of active flow system	Set at 1350 mamsl (approximately 150 m below water level in dolomites)

8.12.3. Boundary Conditions

8.12.3.1.Recharge

The main source of recharge into the aquifers is direct rainfall recharge that infiltrates the aquifer through the overlying unsaturated zone. Due to the heterogeneous nature of the karst and fracture systems, it is assumed to be highly variable. As a starting point the defined quantity of natural/pre-abstraction recharge is assigned (41 mm/a on the hard rock aquifers and 84 mm/a in the dolomite aquifers) for the steady-state simulation/calibration. This constitutes about 6% of MAP for the hard rock aquifers and 12% of MAP for the dolomite aquifers and is in line with the published recharge values for the different aquifers in the area (Table 8-5).

8.12.3.2.Interaction with surface water

The interaction between groundwater and surface water courses was simulated using a river or 3rd type (Cauchy) boundary condition assigned to the streams and river courses within the model domain, whereby the leakage of groundwater into the river (or vice versa) depends on the prevailing gradient. Both ex- and infiltrating conditions of the river were allowed in the model simulations to represent the possible recharge contribution from Brandvlei and the upper Rietspruit. In the absence of site-specific data, leakage of groundwater into the rivers/streams is assumed to not be further constricted by semi-pervious (e.g. clay) layers in the river bed and a leakage coefficient equivalent to the weathered aquifer permeability is assigned to the river. An incision of 2 m below the surrounding topography is assumed for the hydraulically active river bed. The infiltration rate for the rivers (i.e. surface water infiltration into the aquifer, also referred to as a losing river stretch) was limited to a maximum of 5 ℓ/s per km of river course to prevent unreasonable rates of water, not provided for by the river discharge, entering the aquifer.

8.12.3.3.Abstraction

Borehole abstraction from the model domain was simulated using the withdrawal well boundary condition. Outflow rates were assigned to single nodes (at the appropriate elevation) within the model mesh. For fully penetrating wells, equal heads were prescribed for all applicable well nodes. The irrigation requirement for each crop type and percentage crop distribution is based on the preliminary results from Mr. Theunis Vahrmeijer's PhD research[14] (Figure 8-12). The crop irrigation was linked to the digitised crop area and calculated for each groundwater unit of analysis of the Steenkoppies Compartment. Based on Vahrmeijer's results and other sources of information on groundwater use in the Steenkoppies Compartment, it is evident that, depending on mean annual rainfall, crop type and crop distribution, water use for irrigation purposes ranges from 20 to 30 million m^3/a. Despite attempts by Vahrmeijer et al (2013) and Wiegmans et al (2012) to accurately pin-point large scale abstraction boreholes, only a limited number could be verified. In contrast to a wellfield with known borehole positions and measured abstraction rates, the exact position and actual pumping rate per borehole for the Steenkoppies Compartment is not available. As a result the abstraction rates per crop area had to be applied to nearby known boreholes. A further complicating factor is that many irrigated lands receive groundwater abstracted from neighbouring farms. Due to the lack of actual abstraction rates per borehole and over time (i.e. since 1986) and given the changes in land use, crop type and area, the temporal responses of the aquifers to variable external stresses were not assessed. Therefore, a steady-state approach was followed whereby the boundary conditions were changed in the predictive simulations.

[14] Personal Communication (7 July 2016). Mr. Theunis Vahrmeijer, Steenkoppies Aquifer Management Association.

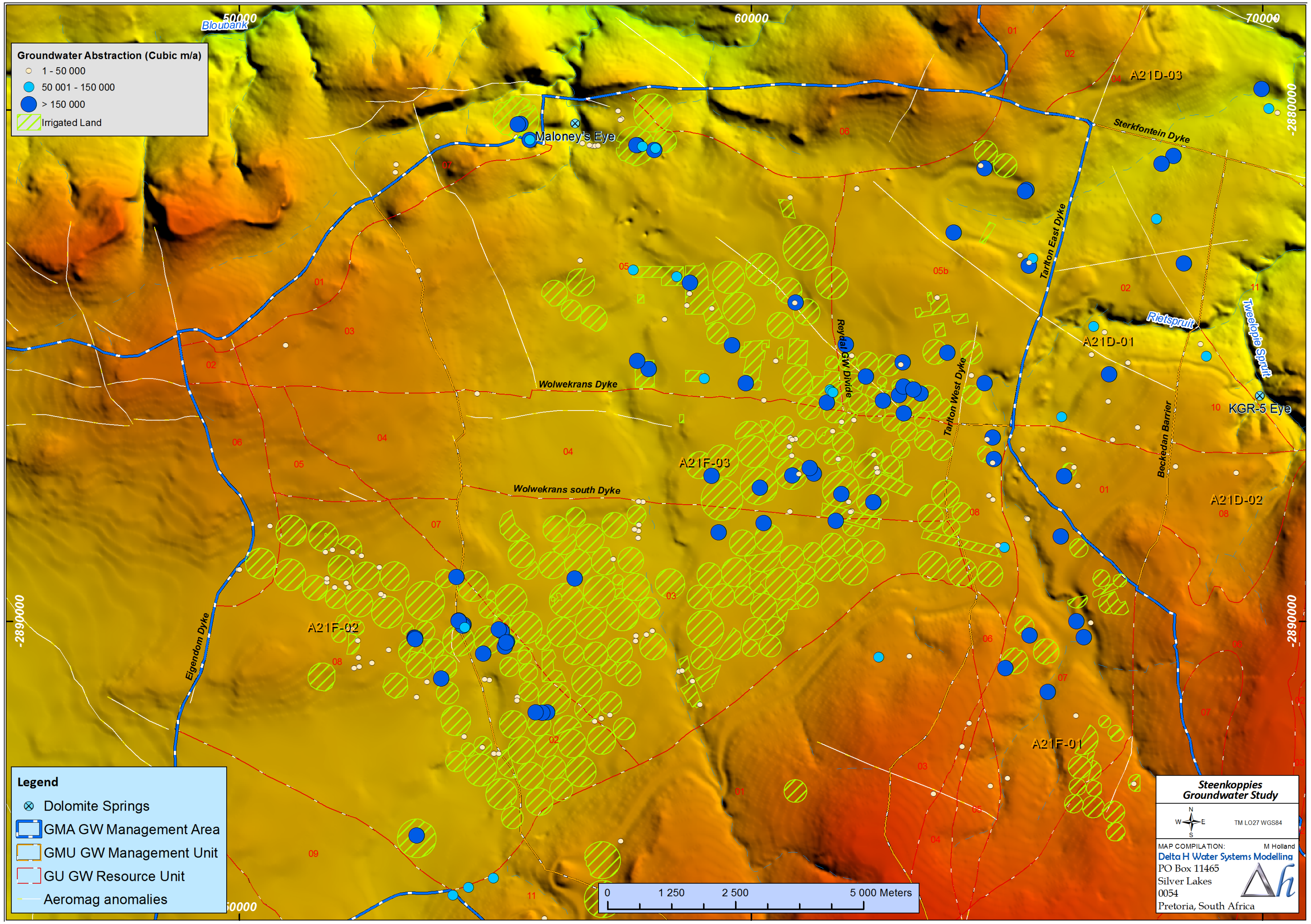


Figure 8-12 Irrigated land and groundwater abstraction locations

8.12.4. Selection of Calibration Targets and Goals

To establish a baseline for the steady-state calibration, the data from Bredenkamp et al (1986) was used as a calibration target. At that time, abstraction was estimated to be in the order of 13.5 million m^3/a (428 ℓ/s), the average flow for the Maloney's Eye was 12.6 million m^3/a (400 ℓ/s), while a recharge rate of 14 % of MAP was established. As a starting point, the model was calibrated against the 70 water level measurements collated during 1985/1986. For future scenarios and for verification of the predicted scenario from 1986 onwards the model was re-calibrated against current abstraction rates, rainfall and observed flow at the Maloney's Eye. A standard trial-and-error approach to calibrate the model was used.

8.12.5. Numerical Parameters

SPRING uses an efficient PCG solver for the iterative solution of the flow and transport equation. The closure criterion for the solver, i.e. the convergence limit of the iteration process was set at a residual below 1e-06. The Picard iteration, used for the iterative computation of the relative permeability for each element as a function of the relative saturation, used a damping factor of 0.5 and was limited to five iterations. The relative difference between the two computed potential heads or capillary pressures after five iterations was generally below an acceptable 0.1 m.

8.13. MODEL CALIBRATION

8.13.1. Steady-State Calibration

Steady-state calibration for the 1986 model was accomplished by varying the hydraulic conductivity values and keeping the recharge rate constant until a reasonable match between the measured groundwater elevations and the simulated groundwater elevations was obtained, in addition to the measured flow of the Maloney's Eye and the simulated flow. The observed groundwater levels are shown against the simulated water levels in a line graph (Figure 8-13).

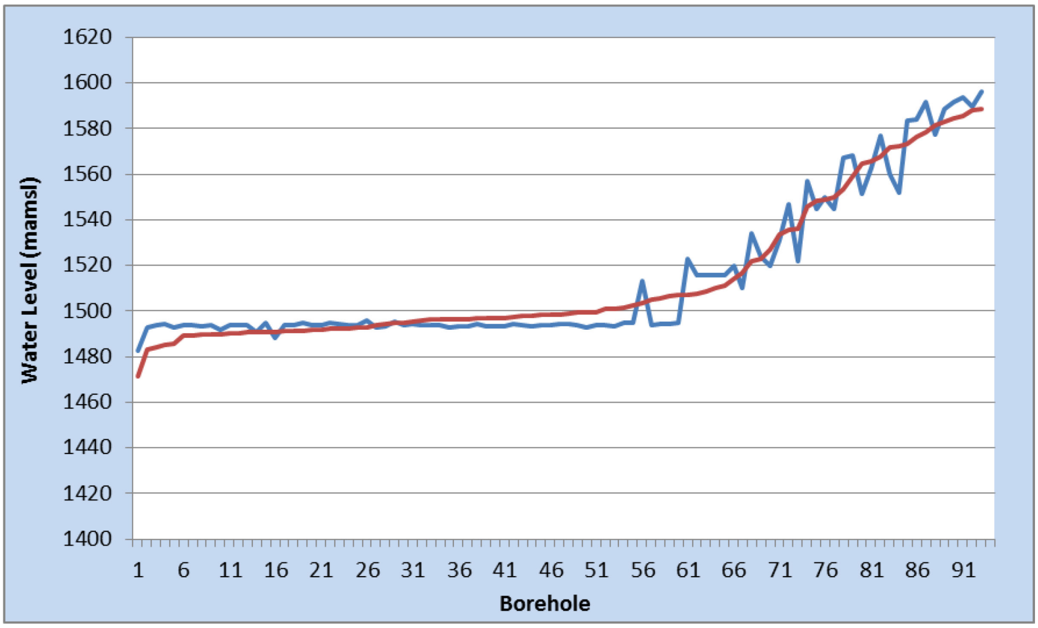


Figure 8-13 Plot of simulated versus observed groundwater levels

The root mean square error (RMSE) was used as a quantitative indicator for the adequacy of the fit between the 95 (=n) observed (h_{obs}) and simulated (h_{sim}) water levels:

$$RMSE = \sqrt{\frac{\sum(h_{obs} - h_{sim})^2}{n}}$$

An RMSE of 7.1 was achieved during the steady-state calibration of the Steenkoppies groundwater flow model and is adequate for the purpose of the study. The simulated flow of the Maloney's Eye is 425 ℓ/s compared to an average flow rate in 1986 of around 400 ℓ/s. The calibrated hydraulic conductivity values are shown in Table 8-8 while the simulated steady-state head contours of the regional model are shown in Figure 8-14.

Table 8-8 Calibrated hydraulic conductivity values

Aquifer	Hydraulic conductivity	
	(m/s)	(m/d)
Pretoria Group	8E-07	0.07
Dolomite (non-karst)	7E-05	6
Dolomite (karst zone)	5E-04 to 7E-04	43 to 60
Witwatersrand Supergroup	7E-07	0.06

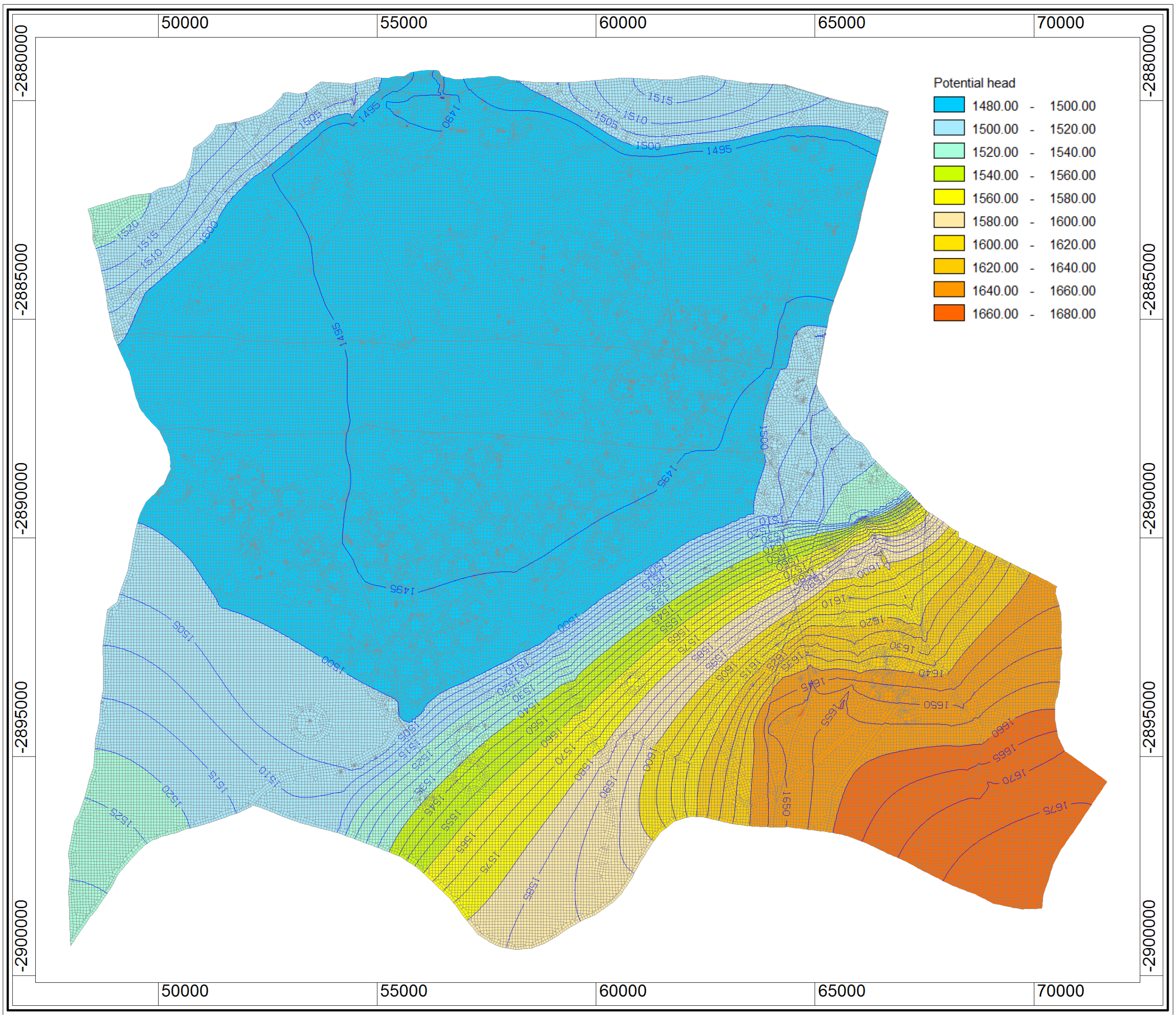


Figure 8-14 Steady-state (1986) water levels of the Steenkoppies groundwater flow model (5 m contours)

8.14. PREDICTIVE SIMULATIONS

8.14.1. Scenarios

The steady-state of base case model (1986) was used as a basis for future (post 1986) or predictive scenarios. The predictive scenarios were assessed in steady-state, representing a state of equilibrium under a specific abstraction or recharge regime. Steady-state modelling generally overestimates the actual flow which means it is conservative and the impacts illustrated (drawdown and change in flow regime) are therefore a maximum and would be reached once the equilibrium is established.

The predictive simulations are based on the reported abstraction rates over time for the Steenkoppies Compartment (Table 7-13). The aim of the first simulation is to determine the impact of an increase in abstraction on the aquifer flows, under the assumption of a constant recharge. A second simulation considered irrigation return flows as an induced recharge mechanism (with the abstraction regime the same as in simulation 1). The current estimated rate of groundwater abstraction from the Steenkoppies Compartment ranges from 25 to 30 million m^3/a, and, taking into account that irrigated land comprises of 4360 ha, this rate is equivalent to an average of 570 to 690 mm of irrigation. In comparison, MAP is 670 mm. Numerous studies (i.e. Arora et al, 1996, Sobowalea et al, 2014, Ebrahimi et al, 2016) have shown an increase in groundwater recharge contribution due to applied irrigation. To account for the potential increase in average groundwater recharge, 10% of the abstraction rate for each scenario was assumed to contribute to groundwater recharge via return flows.

Table 8-9 Details of abstraction and irrigation return flows used for the simulations

Scenario	Simulation 1	Simulation 2
	Abstraction (million m^3/a)	Irrigation return flows (million m^3/a)
Base case	14.96	-
Sc1	17.95	1.79
Sc2	21.54	2.15
Sc3	25.84	2.58
Sc4	28.43	2.84
Sc5	30.13	3.01

8.14.2. Results (Simulation 1)

The impact of abstraction is discussed in terms of the modelled flows with emphasis on sustaining the flow at the Maloney's Eye. The modelled flows, compared to the base case flows, are shown in Table 8-10 and graphically in Figure 8-15. Each of the scenarios tested reaches a new equilibrium in which abstraction is met by reduced discharge to the Maloney's Eye, and enhanced recharge from the upper Rietspruit. The results indicate that under average recharge conditions an abstraction rate of 30 million m^3/a (Scenario 5) will result in a complete loss of flow from the Maloney's Eye. This abstraction rate is 28% more than the recharge rate and the results show that under the current simulation assumptions, induced recharge (from the upper Rietspruit and Brandvlei) is not sufficient to sustain the flow at the Maloney's Eye.

Table 8-10 Model flow results for simulation 1 (in million m^3/a)

Flow	Base case	S1_Sc1	S1_Sc2	S1_Sc3	S1_Sc4	S1_Sc5
Recharge	23.49	23.49	23.49	23.49	23.49	23.49
Upper Rietspruit	4.56	4.67	4.80	4.96	5.07	6.22
Brandvlei	0.28	0.28	0.28	0.28	0.28	0.28
Abstraction	-14.96	-17.95	-21.54	-25.84	-28.43	-30.13
Maloney's Eye	-13.44	-10.56	-7.12	-2.99	-0.53	0.00

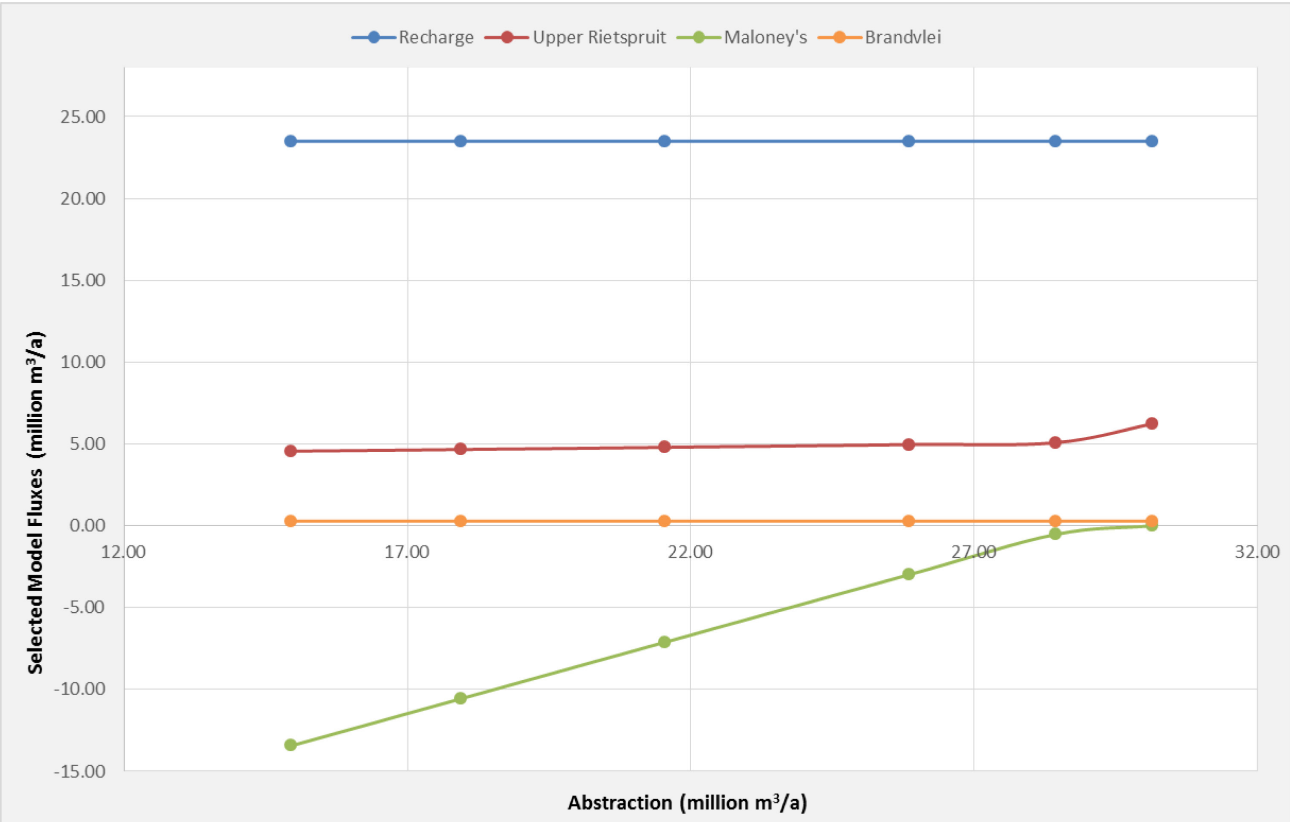


Figure 8-15 Simulated changes in model flows with increasing abstraction (Simulation 1)

8.14.3. Results (Simulation 2)

To show the potential impact of irrigation return flows on the modelled aquifer flows, the recharge rate over the 46.3 km^2 irrigated land area (refer to Figure 8-12) was increased by 10% of their respective mean annual irrigation (Table 8-11). The modelled flows, compared to the base case flows, are shown in Table 8-11 and Figure 8-16. Due to recharge from irrigation return flows, the decline of the Maloney's Eye flow is significantly less compared to the same scenarios under simulation 1.

Table 8-11 Model flow results for simulation 2 (in million m^3)

Flow	Base case	S1_Sc1	S1_Sc2	S1_Sc3	S1_Sc4	S1_Sc5
Recharge	23.49	23.53	23.53	23.52	23.51	23.48
Irrigation return flow	-	0.15	0.18	0.22	0.24	0.25
Recharge total	23.49	25.33	25.68	26.10	26.35	26.49
Upper Rietspruit	4.56	4.50	4.60	4.72	4.80	4.86
Brandvlei	0.28	0.28	0.28	0.28	0.28	0.28
Abstraction	-14.96	-17.95	-21.54	-25.84	-28.43	-30.13
Maloney's	-13.44	-12.22	-9.10	-5.36	-3.11	-1.61

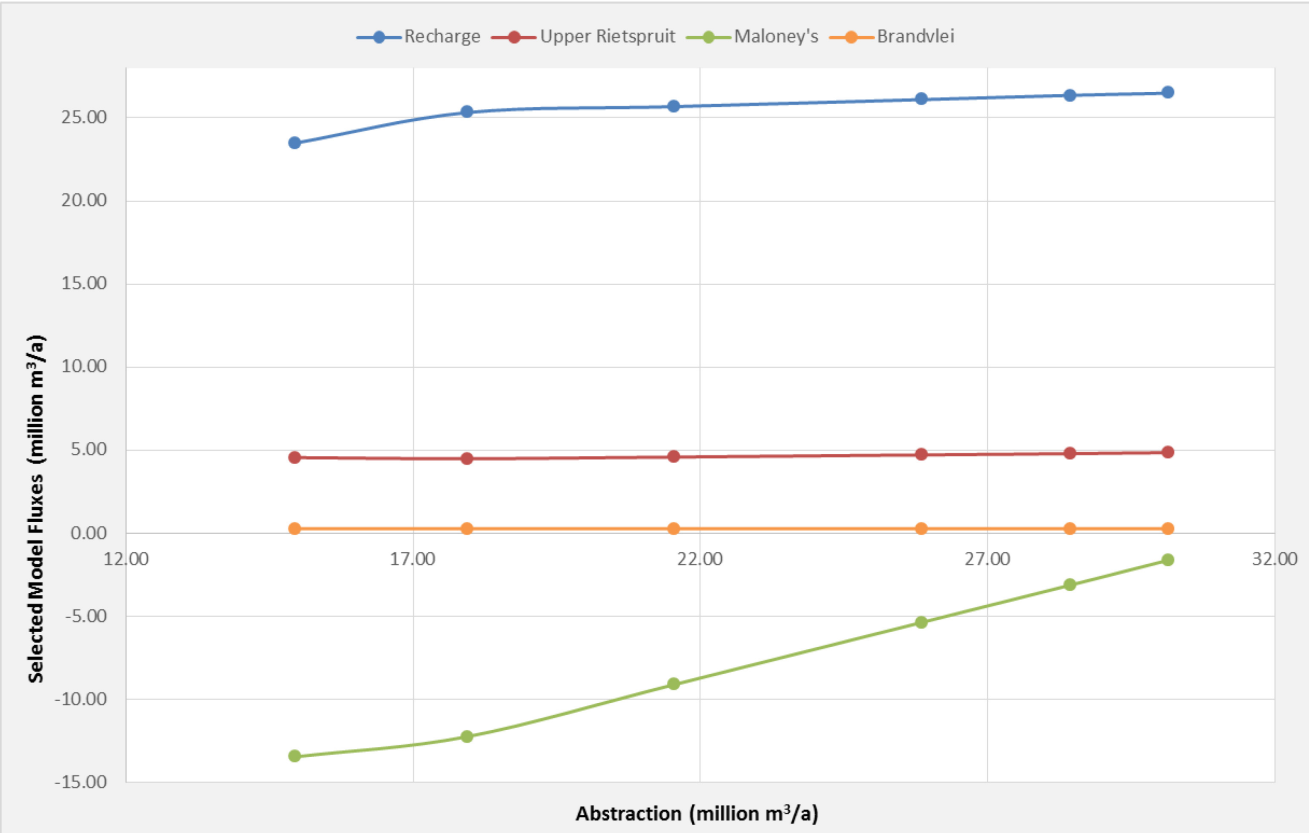


Figure 8-16 Simulated changes in model flows with increasing abstraction (Simulation 2)

8.14.4. Case Study Summary

Estimated groundwater abstractions from the Steenkoppies Compartment amount to between 20 million m^3 and 30 million m^3 per annum, with a likely current abstraction rate of 28.5 million m^3/a.

The discharge of Maloney's Eye is sensitive to below average rainfall events, which lead to below average recharge rates and lower groundwater yields available for abstraction. During drought events (and neglecting groundwater in storage for the time-being), groundwater abstraction rates need to be adapted to prevent complete reduction of the Maloney's Eye flow. The Maloney's Eye has discharged at below 100 ℓ/s (3.1 million m^3/a) continuously for the past 7 months (since December 2015), which is the longest on record since 1908. Based on the long-term rainfall/spring hydrograph (Figure 8-17) and water level/spring hydrographs (appendix, section 13), not only above average rainfall is required, but significant (above 125 mm/month) and frequent rainfall events are required for the flow of the Eye to recover.

With an abstraction rate of 28.5 million m^3/a, model simulation 1, that does not take into account irrigation return flows, calculates flow at Maloney's Eye to be significantly less than observed. Model simulation 2, with irrigation return flows incorporated, generates discharge at Maloney's Eye that is closer to the currently observed discharge. The model flow results illustrate that the Steenkoppies Compartment is benefitting from enhanced recharge from Upper Rietspruit, Brandvlei and irrigation return flows, which assist in maintaining flow at Maloney's Eye. Based on the modelled flows under average groundwater recharge conditions and assumed (10%) irrigation return flows, it appears that 25 million m^3/a is an optimal abstraction rate while maintaining a flow of around 5 to 6 million m^3/a from the Maloney's Eye.

The case study has achieved a quantification of the relationship between abstraction and discharge to Maloney's Eye. The model flow results also suggest that enhanced recharge is likely to be occurring, and contributing to maintaining some spring flow.

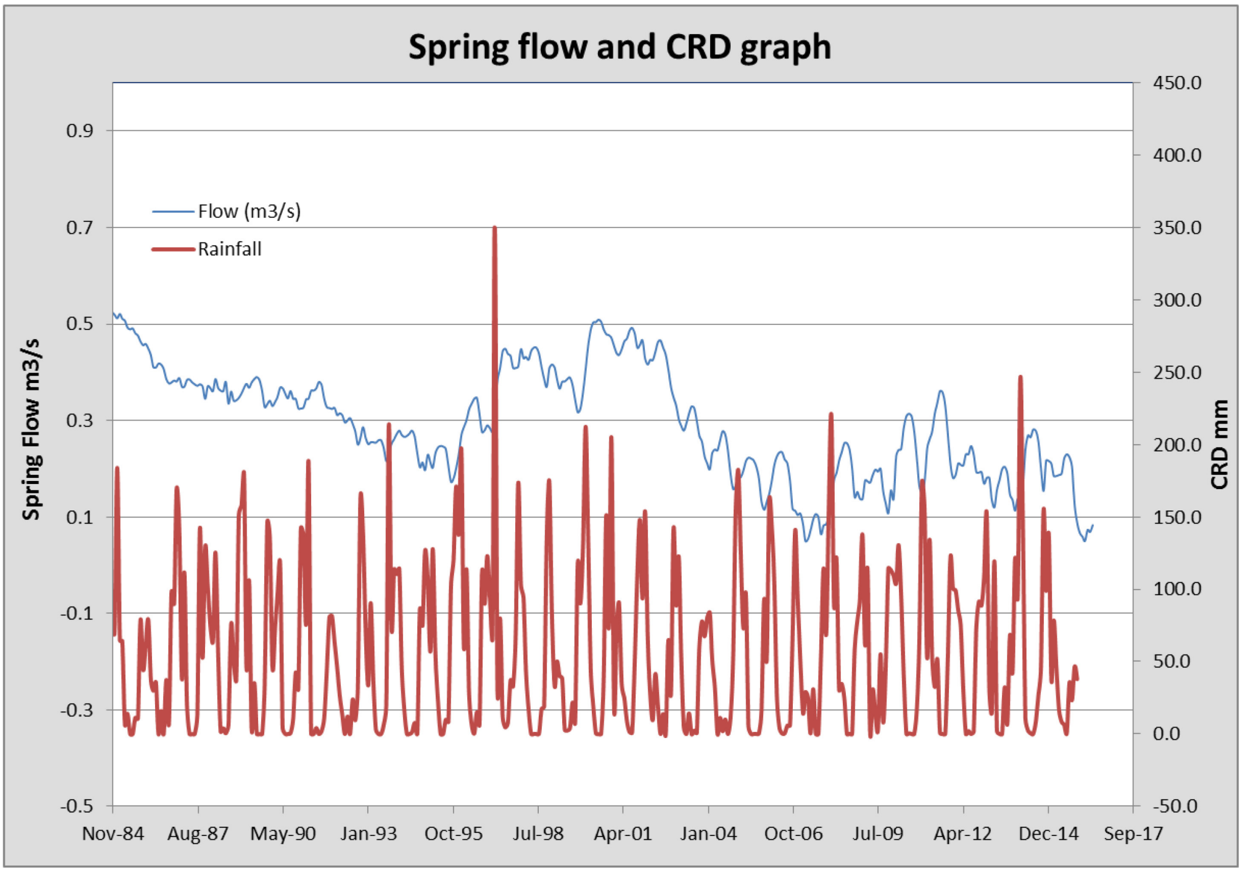


Figure 8-17 **Monthly rainfall vs. monthly spring flow at Maloney's Eye**

Part 3: Integration, Conclusions, Recommendations

9. SUMMARY OUTCOME OF CASE STUDIES

The case studies illustrate some key aspects of the capture principle approach to groundwater assessment.

Both case studies demonstrate the trade-off between groundwater abstraction and reduced discharge:

- In the West Coast aquifers case study, the maintainable aquifer yield at the WCDM wellfield, derived from the capture principle-based assessment, is significantly higher than previous estimates (3.5 to 5.5 million m^3/a, compared to 1.1 million m^3/a) whilst taking the following limitations into account: preventing ingress from the Berg River; preventing dewatering of the confined aquifer.
- In the case of the Maloney's Eye Steenkoppies aquifer, the relationship between abstraction and discharge at Maloney's Eye has been quantified, and as there are direct users of the aquifer (i.e. borehole abstractions), and users of the discharge at Maloney's Eye, this relationship can be used to determine allocation to each user group.

Both case studies illustrate the inability of a water balance approach to accurately determine available groundwater yields:

- The recharge to the West Coast aquifers is significantly higher than the combined private and WCDM abstraction, yet combined abstraction rates equivalent to only 30% recharge[15] would induce ingress from the Berg River (due to the position of the wellfield), which would likely be considered unacceptable. Application of water balance calculations to estimate yield without taking impacts into account, could overestimate available yields, and lead to failure of the groundwater supply.
- In the case of the Maloney's Eye Steenkoppies aquifer, the aquifer is heavily utilised (and critically stressed according to the current abstraction/recharge ratio), yet the model illustrates that the flow from Maloney's eye is likely being sustained (albeit at a lower rate) by irrigation return flows and induced recharge from surface water. A water balance-based calculation of available yields would have limited abstraction to significantly lower than 23 million m^3/a (current natural recharge estimate), whereas model results show that abstraction of 25 million m^3/a can occur while still permitting Maloney's Eye discharge to remain at 5–6 million m^3/a.
- Therefore, in one case study, rates significantly less than virgin recharge can be harnessed (at the abstraction location tested), and in the other case study, abstraction rates greater than virgin recharge are feasible supported by induced recharge.

Capture principle-based assessments, which invariably require numerical models, are often assumed only appropriate or necessary for moderately (20–65%) to heavily used aquifers (> 65%) (based on the Water Resource Classification System management classes, Dennis et al, 2013, DWA, 2013a). Steenkoppies would be classified as heavily used and critically stressed (> 95%). However, the West Coast aquifers (specifically the LRAS) would be classified as minimally used[16], yet the application of the capture principle-based approach to groundwater assessment is still deemed necessary due to the limits on yield that need to be considered. Although capture principle-based groundwater assessments are recommended as applicable in all cases (section 6), they are certainly critical wherever there are sensitive receptors (related to potential risks of abstraction), and wherever the natural discharge is local to the abstraction point.

The decision framework is largely an idealised theoretical approach, developed based on literature and application of groundwater flow theory. The case studies revealed some complexities that are not obvious in the idealised theoretical approach, which are:

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[15] Calculation based on recharge/(current private abstraction + 7 million m^3/a) in quaternary catchment G10M

[16] A calculation of recharge/current private and WCDM abstraction in quaternary catchment G10M results in 8%

- The timing of impact is linked to the hydraulic diffusivity of the aquifer and the distance from abstraction to discharge point.
- Because the distance between abstraction and discharge point impacts on the flow regime under abstraction, the maintainable aquifer yield (MAY) cannot be considered in isolation of abstraction location. There is a MAY *per wellfield*. To consider the (maximum) MAY of the aquifer, optimal position(s) and design(s) of wellfield(s) must be established.
- Full inclusion of a capture principle approach requires enhanced recharge to be modelled (or allowed to occur in the model). In many model designs enhanced recharge cannot be fully incorporated.
- Although the capture principle is relatively easy to illustrate numerically in models, it is challenging to prove in measurement. While water levels are often measured, discharges to receiving environments are not. Where discharge is measured, (i.e. gauges in surface water), it is dependent on several factors, and isolating changes in groundwater discharge would be challenging. Furthermore, the response time means that impacts of current abstraction on receiving environments may not be evident and measurable for several years.

10. PROJECT CONCLUSIONS AND RECOMMENDATIONS

10.1. KEY ACHIEVEMENTS

An accessible description of the theoretical principles of groundwater flow that are part of the capture principle has been provided (section 2.1), along with a summarised description of wellfield infrastructure and yields. These two have been linked to provide an idealised approach (as a flowchart, section 2) for the development of wellfield operating rules that would support or incorporate implementation of the capture principle approach to sustainable groundwater use. The operating rule grades provide a mechanism to implement adaptive management cycles necessary to update maintainable aquifer yields and, in turn, operating rules.

A mainstreaming assessment determined the degree to which the identified idealised approach for the implementation of the capture principle approach to sustainable groundwater use is incorporated in selected existing tools and guidelines. The outcome of the mainstreaming assessment is the identification of gaps that impede the implementation of the capture principle approach. Two further products have been generated intending to address (some of) the gaps identified, and are both considered 'supporting measures' in promoting the established idealised approach for the development of wellfield operating rules:

- Sustainability indicators that are designed to support adaptive management; and
- Recommendations for hydrogeological input to the sustainability decision.

The supporting measures are drawn together with the flowchart for the recommended approach for the derivation of operating rules based on the capture principle, and together these make up the decision framework tool presented in Figure 6-1. In essence, the decision framework tool represents an amalgamation of recommended approaches.

Two case studies have been completed, each of which provide estimations of the maintainable yield of the aquifers assessed. The case studies each illustrate different aspects of the capture principle, and part of the decision framework's recommended approach. For each case study, the relationship between abstraction and aquifer flow regime was quantified, which is one of the decision framework's recommended hydrogeological outputs necessary to support a multi-stakeholder decision over groundwater sustainability.

Many other studies and peer-reviewed papers assess and illustrate the capture principle; there is particularly close alignment between the messages in this report and those in Seward et al (2006). Example cases provided in Seward et al (2006) include a discussion of likely recharge/discharge mechanisms and the associated impact of abstraction at two aquifers, going on to provide adaptive management approaches for the aquifers. Several well-known papers illustrate the capture principle with a numerical model of a test case hypothetical aquifer (Bredehoeft et al, 1982, Sophocleous 2000), whilst others use numerical models of real case studies (i.e. Alley and Leake, 2004). This study contributes to the body of work on the capture principle by providing numerical models of two real cases which illustrate the capture principle, and both of which highlight the inapplicability of a water balance approach.

10.2. KEY MESSAGES AND APPLICABILITY

The key message from the study is the decision framework itself (Figure 6-1); this represents the recommended approach to establishment of operating rules that incorporate a capture principle approach. The operating rule grades, and associated sustainability indicators, provide a mechanism to implement the adaptive management cycles necessary to update maintainable aquifer yields, and, in turn, operating rules.

Another supporting measure of the decision framework is a list of recommendations for hydrogeological inputs to the sustainability decision, and can be seen as a subdivision of the overall decision framework message. These are:

1) Estimate the future conditions when the maintainable aquifer yield (or planned yield) is abstracted, including the water level distribution (impact on storage), and the change in discharge and enhanced recharge. The relationship between abstraction and aquifer flows should ideally be quantified, such that an optimal abstraction rate for the acceptable reduced discharge/enhanced recharge can be determined.
2) Estimate the response time. If the response time is significantly beyond a timescale appropriate for consideration of environmental impact, then the relationship between abstraction and aquifer flows should be quantified for a reasonable planning timescale (i.e. not at steady-state).

The decision framework (and all that it entails) can be considered idealistic. The scale, applicability, and necessity of incorporating a capture principle-based assessment, or of providing the above-listed recommended outcomes, may be debated. Quantification of the relationship between abstraction and reduced discharge invariably requires a numerical model for most situations, and incorporation of numerical models is often assumed only necessary for moderately or heavily used aquifers. One of the case studies completed here would be considered a heavily used or critically stressed aquifer, the other a minimally used aquifer. Nevertheless, in both case studies, a capture-based assessment was critical for definition of maintainable aquifer yield. The water balance approach is unable to accurately determine available groundwater yields in these two cases.

Rather than basing the necessity of a numerical model or capture principle-based assessment on extent of aquifer use, the decision framework as an overarching approach is considered applicable in all cases. For example, an aquifer where abstraction is very distant from definable discharge may have a response time so long that for all practical purposes under a planning horizon, abstraction can be considered to be solely derived from storage. In such a case estimation of reduced discharge is less relevant, and water level change becomes the primary criteria upon which to assess the acceptability (sustainability) of abstraction (e.g. Karoo Supergroup aquifers example given in section 6). This situation should still be described as such, and hence the overarching approach of the decision framework (along with the associated definition of terms) is applicable.

In many ways, a capture principle-based groundwater assessment is not radically different to existing approaches. Many studies do assess impact on water levels (storage), and make estimates of stream flow reduction (as illustrated by the mainstreaming assessment showing that elements of the capture principle are fully accommodated in some existing tools). But, implicit to the decision framework, is a recommendation that a fresh orientation to messaging is required (by hydrogeologists in groundwater assessments), in which the capture principle is put at the centre of planned groundwater use:

- Groundwater level decline is **not** an indication (alone) of unsustainability. At the onset of pumping, abstracted water is met by storage, hence water levels **will** decline when an aquifer is pumped. The (rate of) decline in water levels will reduce as a new dynamic equilibrium is established, in which abstraction is met by reduced discharge and/or enhanced storage. At this point, there is no further loss of storage, yet if abstraction continues, the loss of storage is not reversed. This loss of storage/change in water levels must be estimated.
- Natural discharge **will** reduce ***and/or*** recharge **will** increase (after some time), given that the abstracted water must have a source. The reduction in discharge/increase in recharge, and the response time, should be quantified.

10.3. Limitations and Recommendations

The decision framework itself (Figure 6-1) represents the recommended approach to establishment of operating rules that incorporate a capture principle approach. Measures to fully implement or promote uptake of the tool were not included in the proposed project, which would require further dialogue within the groundwater community and with regulators.

This project has a relatively narrow focus on groundwater assessment and sustainability from a capture principle and theoretical approach. Other aspects of groundwater sustainability have not been taken into account, such as consideration of sinkhole formation, subsidence, or potentially unacceptable water quality changes induced by pumping.

Only two case studies were incorporated in this project to illustrate the capture principle and test the decision framework. Several of the recommended approaches in Part 1 of this report remain theoretical, particularly the operating rule grades and associated sustainability indicators, and it was not possible to source case studies that would be suitable to test either of these. As these tools both relate to supporting adaptive management approaches for groundwater abstraction management, full illustration of these tools would require a long-term project (5–10 years) in association with a groundwater user (i.e. a municipal wellfield) to fully demonstrate the approach. Some case studies that already demonstrate the capture principle may exist across South Africa, and ideally a wider search for relevant case studies should be carried out. It is recommended that existing case studies (ideally including monitoring data showing reduction of discharge/enhancement of recharge) be collated into a compendium of capture principle assessments to further demonstrate and promote the capture principle approach to sustainable groundwater use.

Although this project contributes to overcoming the technical challenges associated with the gap between theory and current practice, other significantly wide-ranging issues contribute to this gap. The mainstreaming assessment highlights that the groundwater use authorisation and licensing process, which takes into account the reserve, is to some degree misaligned with the capture principle. As an illustration, the current guidelines for implementation of the Water Resource Classification System recommend, for example, the use of a stress index (Dennis et al, 2013). This misalignment is also compounded by the prevalent application of water balance approaches in any regional to national scale groundwater assessment in which it is often deemed impractical to assess groundwater resources at the aquifer scale, although this is necessary for a capture principle-based assessment (primarily due to project budget and timescale). The project did not aim to address these gaps, yet without change in these spheres, the gap between theory and implemented methodologies will remain.

By way of comparison, groundwater in the United Kingdom is managed by the Environment Agency, through the use of regional numerical groundwater models covering all of the major aquifer systems. A new abstraction licence can be assessed in the relevant model to determine its impact on reduced discharge/enhanced recharge and existing abstractions, and associated licence conditions derived. Although the Environment Agency subcontracts the routine re-calibration and update of the models, the models are ‘owned’ by the Agency, enabling the cumulative effect of each licence application to be assessed. These models are available under licence for direct or indirect use by consultants where required[17] (for example, for the creation of sub-regional models to address local/site scale issues) preventing wasted effort (and national funds) in multiple models being generated for the same aquifer over time by different consultants. It will take time, but a transition to this approach, in which DWS manages numerical models of all the major aquifers for abstraction management and resource protection (i.e. GRDM), is the only way to change the status quo, fully implement the capture principle approach to sustainable groundwater use, and avoid the prevalence of water balance approaches in regional studies.

[17] http://esi-consulting.co.uk/our-expertise/hydrogeology-and-water-resource-management/groundwater-modelling/ accessed 20 September 2016

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12. APPENDIX 1: GLOSSARY

Box 12-1 Capture principle-based yield and sustainability terms to be used in this study

Maintainable Aquifer Yield (Delvin & Sophocleous, 2005)

- The rate that can be maintained without mining an aquifer.
- It is a rate that can be maintained by reduced discharge and/or induced recharge in a new equilibrium, such that the aquifer storage doesn't deplete.
- It does not directly depend on the pre-abstraction recharge rates.
- Long-term application of the maintainable aquifer yield does not necessarily equate to sustainable groundwater use
- It is determined through pumping, and aquifer assessment, and is influenced by landuse, climate and a variety of time-varying factors, hence cannot be considered fixed.

Sustainable Groundwater Use

- Long-term abstraction of the maintainable aquifer yield can be considered sustainable groundwater use if its impact in terms of reduced discharge or enhanced recharge is considered socio-economically-environmentally acceptable.
- This requires that the future equilibrium conditions under pumping have been assessed, and a socio-environmental-economic decision taken as to whether this is acceptable.
- This decision over acceptability will change as the maintainable aquifer yield changes, and as the socio-economic-environmental conditions against which the abstraction is being compared change, due to external factors. Hence, sustainable groundwater use is also a transient concept.

Sustainable Aquifer Yield

- The yield that can be supplied from the aquifer targeted under conditions of sustainable groundwater use.
- It is equivalent to the Maintainable Aquifer Yield ***IF*** long-term groundwater use at this rate is considered sustainable groundwater use.
- It cannot be considered a fixed number, but one that must be updated through adaptive management and will change over time, due to the transient nature of the maintainable aquifer yield and the sustainability decision.

Borehole Yield

- The yield that can be maintained from a borehole without borehole failure (i.e. inability to continue to pump).
- It is influenced by time-varying factors and cannot be considered a fixed number, requiring update through adaptive management.

Wellfield Yield

- The yield that the combined wellfield infrastructure can supply, incorporating hydraulic interference between boreholes and constraints from other infrastructure.
- It is influenced by time-varying factors and cannot be considered a fixed number, requiring update through adaptive management.

A scheme is considered to have robust **operating rules** if: The scheme operator operates the scheme based on a living document, updated through adaptive management cycles, containing a set of rules describing the required operation of the wellfield and any associated infrastructure that makes up the complete scheme. These operating rules are based on, or take into account:

- The sustainable aquifer yield, incorporating natural constraints
- The infrastructure capacity and infrastructure constraints
- The demand on the system and any constraints this may introduce, and optimisation approaches.

13. APPENDIX 2: CASE STUDY 2: WATER LEVEL/SPRING HYDROGRAPHS

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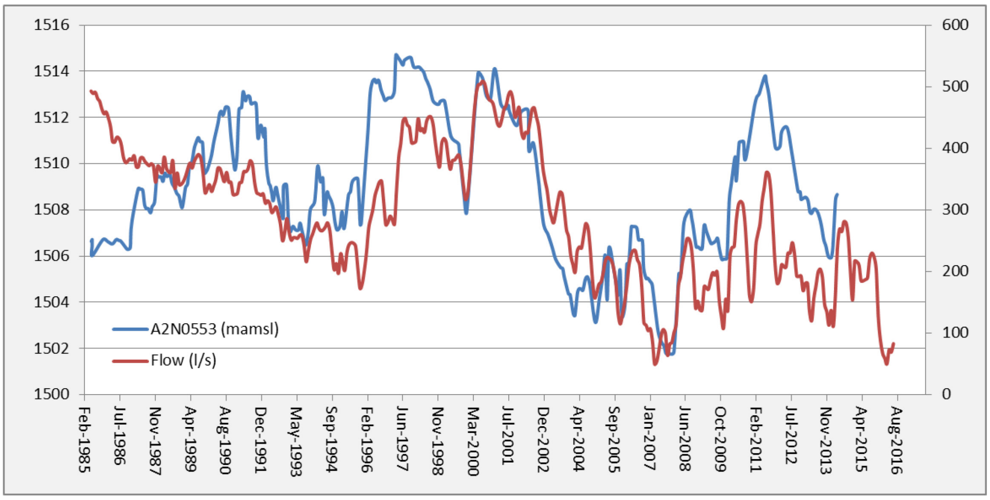

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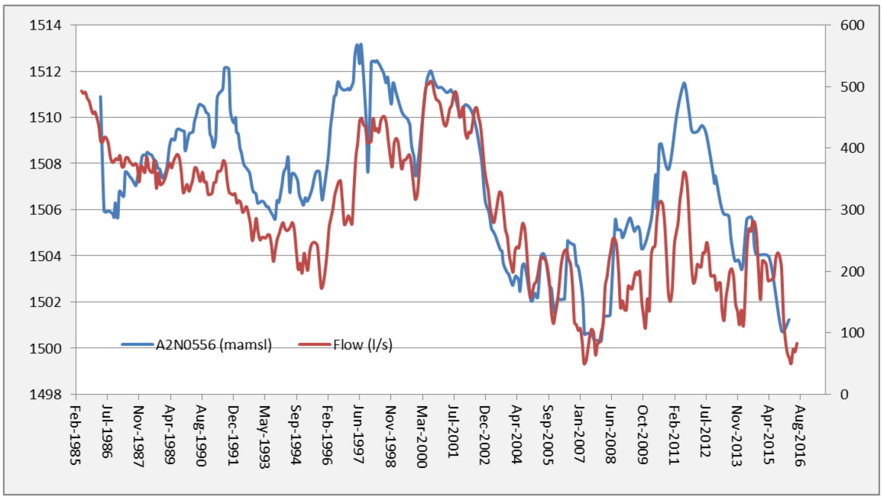

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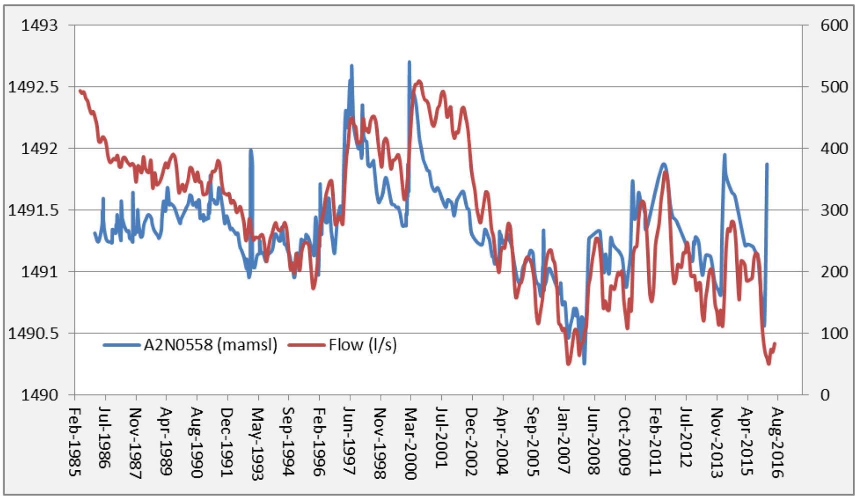

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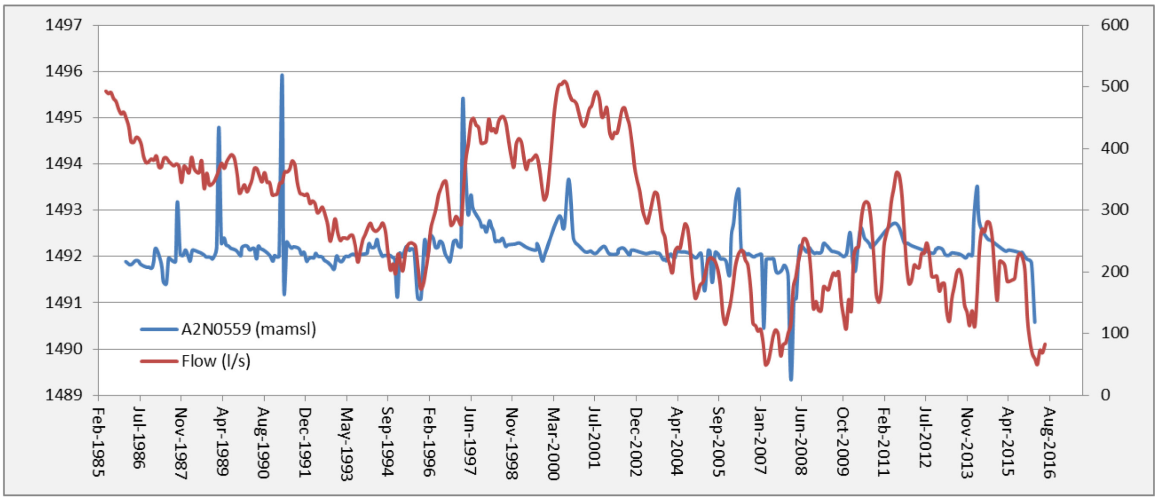

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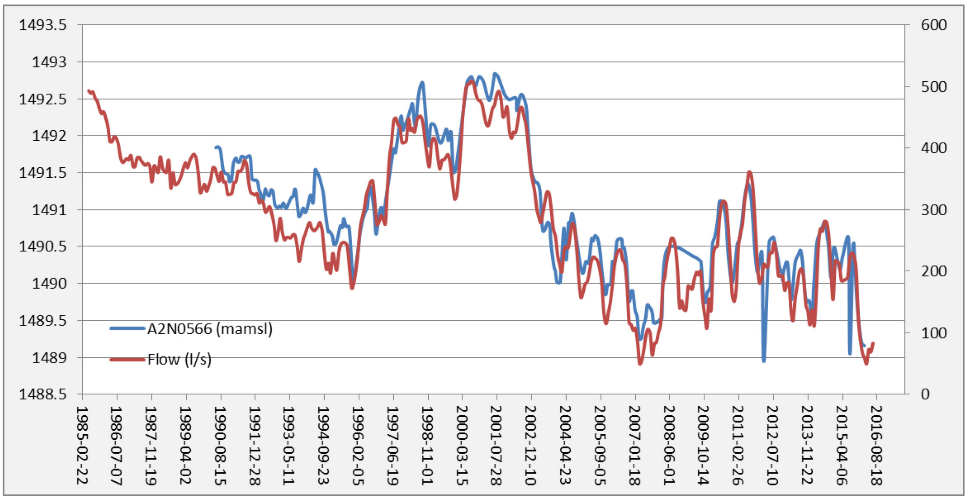

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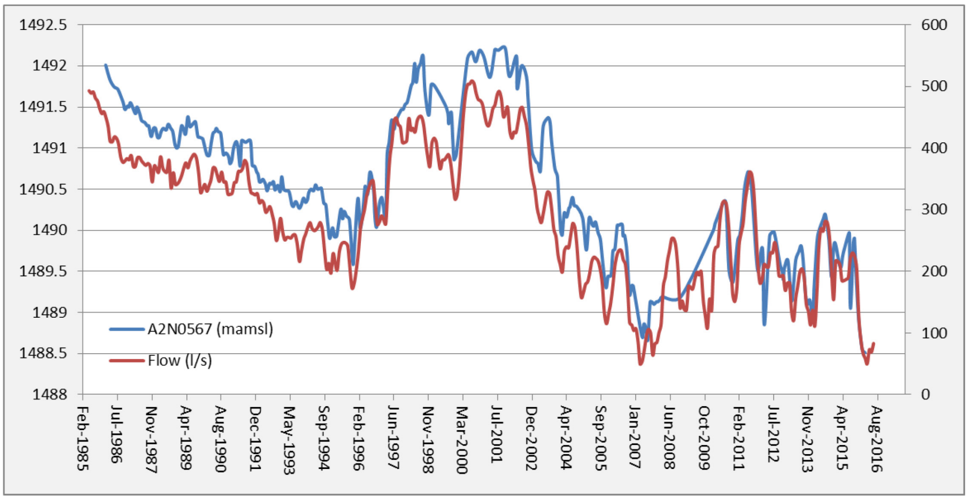

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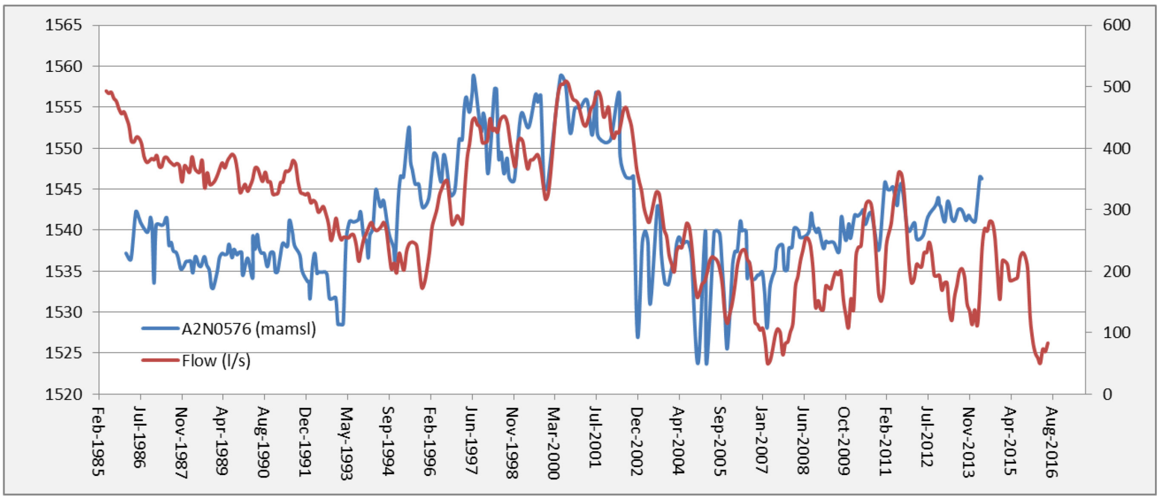

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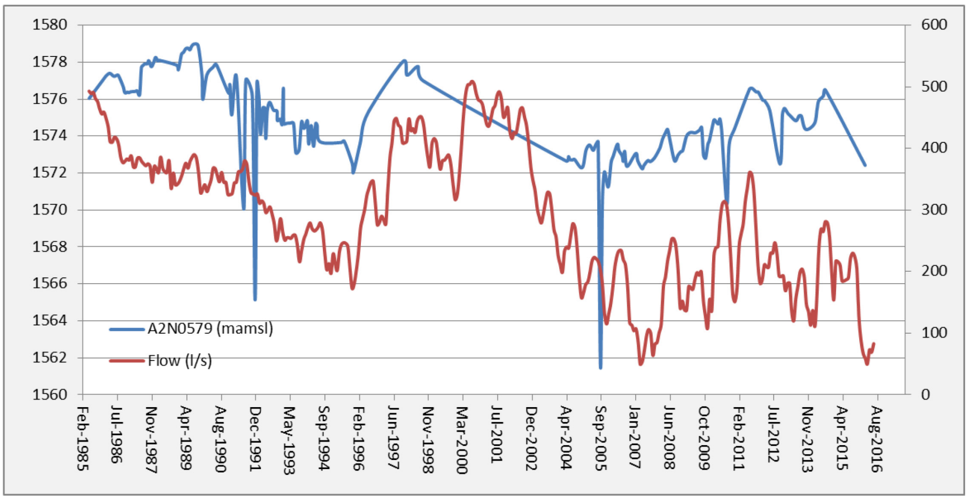

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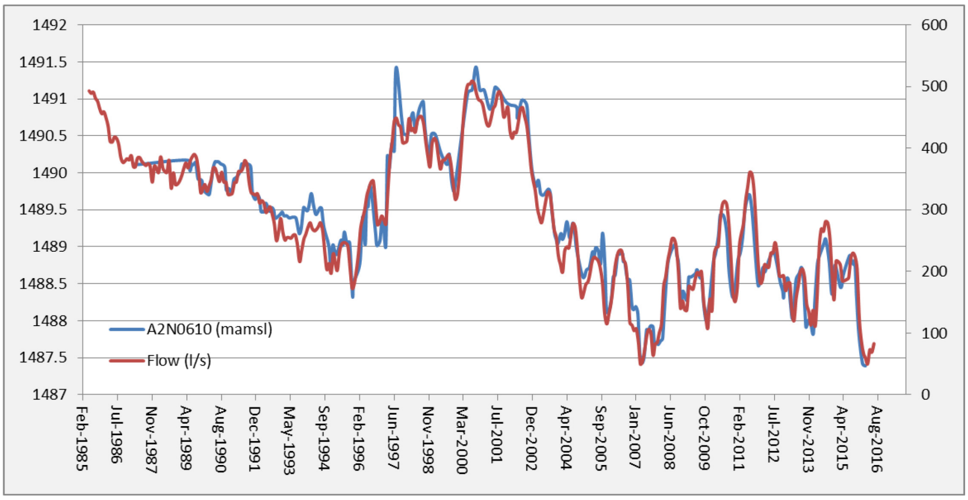

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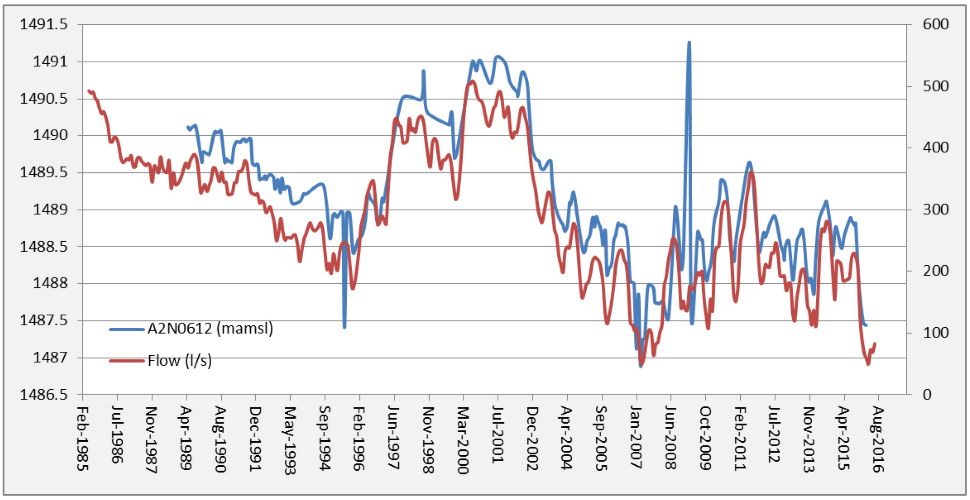

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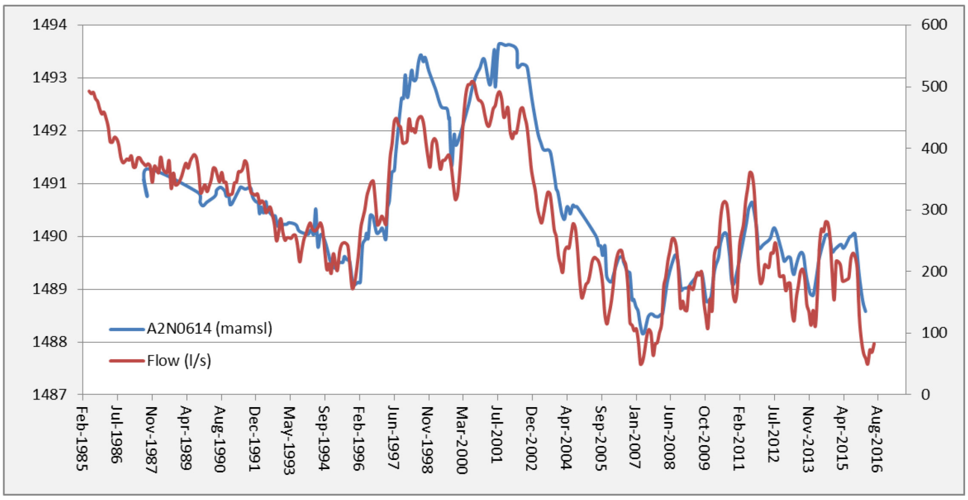

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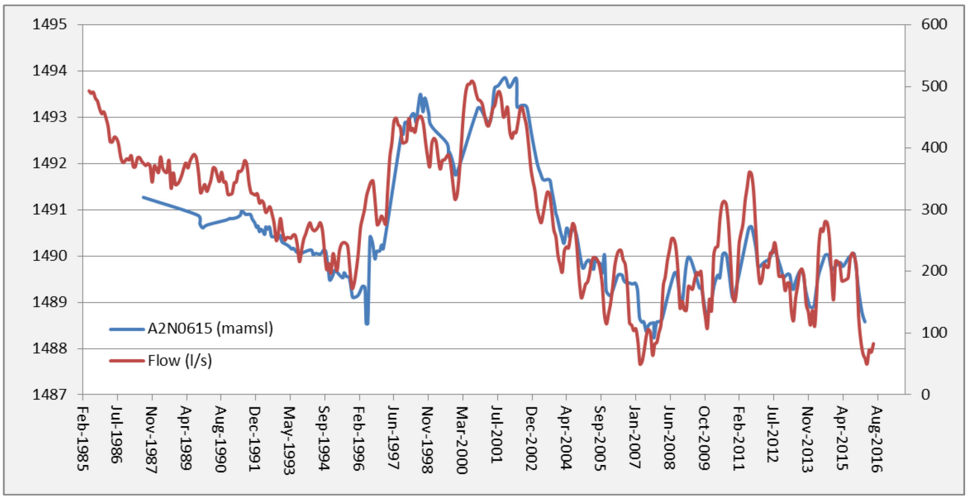

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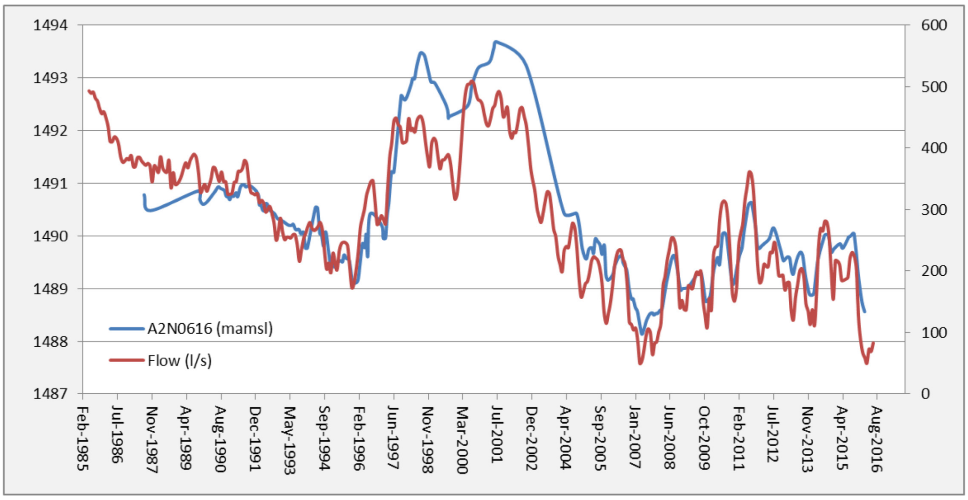
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