

Development of a Groundwater Resource Assessment Methodology for South Africa: Towards a Holistic Approach

Report to the
Water Research Commission

by

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TECHNICAL BRIEF

Intro

With growing water demand in South Africa we have nearly reached the limits of what surface water from our rivers and dams can supply. Luckily we still have unused reserves of groundwater, and many experts call for more groundwater use and better groundwater management. However, it can be surprisingly difficult to estimate just how much groundwater can be safely taken from a particular area or “catchment” without harming the environment and other water users. Rivers rely on groundwater flow from the riverbed or “baseflow” to sustain them over most of the year – and over-abstraction of groundwater can reduce river flow. One of the main problems is deciding how much of a river’s flow is due to groundwater, and what proportion is due to rainfall and surface run-off. Over the years, scientists have developed ways or “models” of determining this proportion. The various models all rely on available data about the river systems and groundwater environments that they are studying – data that can be very limited in some areas.

The problem of quantification

Surface water-groundwater interactions take place via different mechanisms on varying scales and are influenced by numerous processes. The complexity of these interactions makes the quantification of the actual volume moving between the two water resources problematic. There are numerous methods available for the quantification of the amount of groundwater contributing to a rivers baseflow, lakes or wetlands as well as methods for quantifying the loss of water from a losing stream. However, surface water-groundwater interaction is still poorly understood and difficult to quantify due to the inherent heterogeneity of aquifers, variable influencing factors, different time scales of surface water and groundwater, and the fact that groundwater is a hidden resource that cannot be directly measured in most cases.

A new approach

In light of the persisting lack of understanding of surface water-groundwater interactions, the importance of the groundwater contribution to streamflow and the increasing use of groundwater, a new approach to the quantification of this is proposed. Although multiple methods exist for the quantification of the groundwater contribution to streamflow, the

addition of the proposed method will be advantageous. The method would be beneficial in terms of using a different dataset comprising water quality data and as part of a multi-method approach which has been suggested by numerous authors.

The method of quantifying the groundwater contribution to streamflow currently used in the latest Groundwater Resource Assessment (GRA2) of South Africa is based on a water balance approach alone, while the proposed new method combines the water balance with solute mass balances. The incorporated solute mass balances serve to better constrain the water balance used to quantify the groundwater baseflow. However, the concept of using two sets of mass balance equations simultaneously is not a novel idea. The use of the basic principal and the Mixing Cell Model (MCM) are also fairly common, but the use of the MCM to quantify the groundwater component of streamflow is an innovative application.

Mixing Cell Model

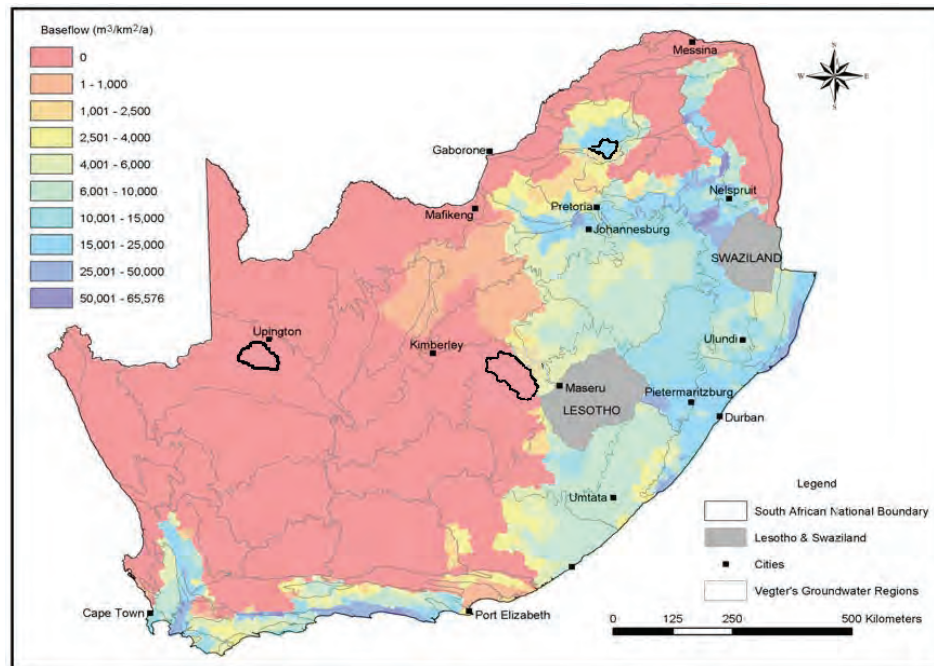
The concept of a mixing cell is essentially based on the continuity equation. The one-dimensional continuity equation states the amount of inflow to a system will equate the amount of outflow with no change in storage, for the considered time step.

The Mixing Cell Model (MCM) builds on this foundation by sub-dividing a system into one or more mixing cells. A water balance equation is expressed for each cell to describe the movement into and out of the cells. The MCM requires that each of the inflows, present in the water balance equation, are chemically defined by a set of tracer concentrations. This water quality data is then used to describe a chemical mass balance equation for each cell. The chemical mass balance equation serves to constrain the water balance equation in order to produce better estimates of the various unknown inflows to the system, than estimates made from the sole use of a water balance equation.

Pilot study of the Mixing Cell Model

The MCMsf programme developed by Adar (2012), implementing the Mixing Cell Model principals and the Wolf algorithm, was successfully applied in the current pilot studies to estimate groundwater baseflow components for the UFS surface water-groundwater interaction test site just outside of Bloemfontein on a site-specific scale, quaternaries C52A-C52H on a large scale along the middle Modder River section, to a set of quaternary

catchments (A42A, A42B and A42C) within the Limpopo Province as well as a quaternary catchment in the Northern Province, South Africa.



A map of the groundwater baseflow zones of South Africa showing the location of the three pilot study areas.

Pilot Study findings

The application of the MCM in South Africa has both advantages and disadvantages. However, the MCM was found to give more accurate results than the chemical hydrograph separation method applied. When there is sufficient data for a MCM run, the MCM groundwater baseflow volume tends to be in-between the Sami and Hughes model estimates. Considering that the Sami model was found to under-estimate the groundwater baseflow and the Hughes model to over-estimate, the MCM is a good indication of the amount of groundwater contributing to a river.

The natural environment can hardly be described by linear, homogeneous expressions or with the assumption of conservative behaviour of solutes, especially in the geochemically very active hyporheic zone separating surface and groundwater. While the model allows for a partial compensation of such violations, the infringement of assuming conservative tracers is still likely to result in errors. We thus have to accept a certain level of inaccuracy in the quantification of natural phenomena such as the groundwater contribution to baseflow using the mixing cell or other methods. Similarly, data paucity and ungauged catchments

limit the applicability of baseflow estimation models based on empirical data in large parts of the country.

Recommendations

The Mixing Cell Model is recommended as an additional tool used to quantify the groundwater baseflow volume in South Africa, and the following scenarios are suggested: validation, low confidence groundwater baseflow areas, physical-parameter data scarce and complex geology areas, ungauged quaternary catchments, site-specific scale and in a multiple method approach.

Further Review and improvement of methodologies applied for the GRA2 project are recommended for the sections: quantification, planning potential map, recharge, aquifer classification and groundwater use.

Better primary data is needed in the following areas: Existing groundwater use in South Africa (actual quantities), Groundwater quality variations, both natural and anthropogenic, Groundwater levels, including relationship to geology and topography, Rainfall (both volumes and intensity) and rainwater quality, and River/stream flows.

Recommendations on groundwater information collection and accuracy in South Africa have been incorporated as policy recommendations.

- Finalisation and expansion of the National Groundwater Archive (NGA)
- Registration of drillers and capturing of driller's groundwater data
- Support water services institutions in the development and implementation of asset registers on groundwater infrastructure, monitoring of groundwater use, and groundwater quality
- Engage with relevant authorities to maintain hydrological and environmental monitoring programmes necessary for groundwater management.

Re-assess the funding required by DWA head- and regional offices for groundwater monitoring, data capture as well as the operation and maintenance of groundwater infrastructure

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1. INTRODUCTION

A movement towards better and more accurate determination of groundwater resources is seen on an international scale and is driven by increasing competition for water resources, a better appreciation of the ecological role of groundwater, and an expanding awareness of possible water supply disruption or variation due to climate change. South Africa is no exception. Planners in South Africa require estimates of how much groundwater is available so that they can incorporate these quantities into their resource plans. The question of how much groundwater is available is more complicated than it might first appear – factors such as groundwater quality, accessibility, the effect of abstractions on ecosystems, and other considerations must be taken into account. It is also easier to estimate regional averages of groundwater availability than it is to say with confidence how much groundwater is available at any particular location.

Examining available groundwater data can give a good picture of regional groundwater availability and quality, and a good average idea of borehole characteristics. Aquifers often vary in their hydraulic properties over short distances causing local scale groundwater prediction or assessment to be more complicated – at these scales the variability in local aquifer properties can make regional “average” assessments meaningless. The time scale is important too – an assessment conducted on the basis of one year’s data may not be representative of a longer term average. The challenge therefore is to make the best use of regional groundwater data to predict and assess local groundwater resources, whilst at the same time being aware of the limitations of the data. Issues of scale become more and more important as a smaller area is selected for assessment. As the United States Geological Survey’s Circular 1323 of 2008 on groundwater availability in the United States puts it: “On a national scale, we know quite a bit about the Nation’s ground-water resource; however, much of the information is generalized and has limitations when attempts are made to plan for the future” (Reilly et al., 2008:7).

South Africa’s groundwater resources are increasingly recognised as being of great importance to the environment, to basic human needs, and to sustainable national development. This follows many decades in which groundwater was relegated to an

essentially “local” resource, regarded in law as the property of the landowner, and accorded far less attention than surface water. This is despite the total renewable volumes of groundwater in South Africa being of equivalent magnitude to our surface water resources. Groundwater has many acknowledged advantages over surface water, such as its resistance to hydrological droughts, its generally good natural quality and the fact that it can usually be found close to where it is needed.

Previous estimates for “available groundwater resource potential” in South Africa for a typical year range from about 7.5 000 million m³/a to as much as 47.7 000 million m³/a, taking various factors such as the Reserve, transmissivity and water quality into account (Rosewarne et al., 2006). The recent National Groundwater Strategy (DWA, 2011) states that: “The total volume of available, renewable groundwater in South Africa (the Utilisable Groundwater Exploitation Potential or *UGEP*) is 10 343 million m³/a (or 7 500 million m³/a under drought conditions)”. These figures were taken from work done by Middleton and Bailey (2009) for the Water Research Commission. According to these authors, *UGEP* is a reasonable estimate of groundwater availability, in that it represents a management restriction on the volumes that may be extracted based on a defined “maximum” allowable water level draw down. The *UGEP* volume, in turn, is based on estimates of groundwater availability derived during the Groundwater Resource Assessment Phase II (GRA2) Project.

The original GRA2 process was an ambitious and commendable endeavour to quantify South Africa’s groundwater resources. Due to a shortage of reliable groundwater data, and an uneven distribution of data across the country, some of the methods used in the GRA2 process are questionable. The GRA2 groundwater data is not frequently used because of a perceived lack of accuracy due to the lack of primary data coverage in some areas of South Africa as well as deficiencies in the algorithms used. The perceived lack of accuracy due to data limitations can be addressed by better groundwater data collection, and by incorporating more of the existing data into the GRA2 calculations. However, addressing the perceived lack of accuracy in the algorithms used is more complicated. This perceived lack of accuracy was found to be greatest for the quantification of surface water-groundwater interaction. In light of this, the main focus has been assigned to the review and improvement of the GRA2 surface water-groundwater interaction methodology.

Surface water and groundwater resources are both components of the larger, interconnected hydrological system and the development of either will affect both the quantity and quality of the other. It stands to reason that the quantification of the amount of groundwater feeding the surface water system at the river – aquifer interface is an essential component of understanding the whole hydrological system to ensure the responsible use of both these water resources.

Surface water-groundwater interactions take place via different mechanisms on varying scales and are influenced by numerous processes. The complexity of these interactions makes the quantification of the actual volume moving between the two water resources problematic. There are numerous methods available for the quantification of the amount of groundwater contributing to a rivers baseflow, lakes or wetlands as well as methods for quantifying the loss of water from a losing stream. However, surface water-groundwater interaction is still poorly understood and difficult to quantify due to the inherent heterogeneity of aquifers, variable influencing factors, different time scales of surface water and groundwater, and the fact that groundwater is a hidden resource that cannot be directly measured in most cases (Sophocleous (2002); Eijkelenburg (2004); Kirk (2006); Kalbus *et al.* (2006); Hughes *et al.* (2007); Levy and Xu (2012)).

In light of the persisting lack of understanding of surface water-groundwater interactions, the importance of the groundwater contribution to streamflow and the increasing use of groundwater, a new approach to the quantification is proposed. Although multiple methods exist for the quantification of the groundwater contribution to streamflow, the addition of the proposed method will be advantageous. The method would be beneficial in terms of using a different dataset comprising water quality data and as part of a multi-method approach which has been suggested by numerous authors (Oxtobee and Novakowski (2002); Environment Agency (2005b); Rosenberry and LaBaugh (2008); Allen *et al.* (2010); Levy and Xu (2011); Sophocleous (2002); Kalbus *et al.* (2006)).

The current GRA2 method is based on a water mass balance approach alone, while the proposed new method combines the water mass balance with solute mass balances in order to better constrain the water balance quantification of groundwater baseflow. The concept of using two sets of mass balance equations (water and solute) simultaneously is not a novel idea. However, while the basic principal and the mixing cell model are fairly common, the

application of the mixing cell model to quantify the groundwater component of streamflow is an innovative application. The concept and mathematical principals of the mixing cell model are covered in detail, including a short history of the use of the MCM. The model is applied to datasets from the surface water-groundwater interaction test site developed by the University of the Free State, as well as additional fieldwork data collected along the middle Modder River. The MCM is then applied to a set of quaternary catchments in the Limpopo Province for which there are available calibrated estimates of the groundwater component of baseflow for the Sami and Hughes models. The Limpopo quaternary application is equivalent to a desktop study. The MCM is also applied to the quaternary catchment D73F, located in the semi-arid Northern Cape, to assess the applicability of the mathematically based MCM to a flow system located within a regionally determined zero groundwater baseflow zone. The MCM results for each study area are assessed in terms of groundwater baseflow volumes determined by the Pitman, Sami and Hughes models. A chemical hydrograph separation method which also incorporates water quality data, namely the Tracer method, is also reported for the study areas to further validate the MCM.

The quantification of surface water-groundwater interaction is important, but it is also important to keep the bigger picture in mind. Surface water-groundwater interaction forms only a small part of a larger groundwater resource assessment that is required to make management decisions to assist with both the protection and responsible use of South Africa's water resources. Additionally, it is important to be aware of the fact that groundwater data forms the building blocks of any method used to determine how much groundwater South Africa has and if these building blocks are insufficient or not available, the selection of a method to determine the amount of groundwater becomes meaningless.

2. LITERATURE REVIEW

Groundwater Resource Assessment methodologies are reviewed on an international and national perspective. The international review of groundwater resource assessment methodologies includes methods implemented in Australia, United Kingdom and California. The national review consists of a critical review of the South African Groundwater Resource Assessment Phase II (GRA2). The critical review of GRA2 highlights the main shortcomings of each methodology applied and it was found that the perceived lack of accuracy in the GRA2 methodology is greatest in the quantification of surface water-groundwater interaction. Following this, a review of surface water-groundwater interaction methods has also been conducted. Numerous methods for quantifying surface water-groundwater interactions exist, ranging from the simple and site-specific to complex and extensive. A number of these available methods are described and discussed on an international and national perspective. The methods currently used in the latest South African Groundwater Resource Assessment are reviewed and described in detail. The application history of the proposed method, the Mixing Cell Model (MCM), is investigated and discussed. Lastly, the groundwater dataset situation in South Africa is reviewed.

Groundwater Resource Assessment Methodologies

2.1. Review of Selected International Groundwater Resource Assessment Methodologies

Approaches for groundwater assessment and management differ world-wide, and depend to some extent on the legal and institutional mechanisms which underpin them. A variety of factors, not all of them “technical” or even directly related to groundwater (for instance legal history, national institutional structures, or disinclination towards groundwater use) all play a role. Three international examples have been selected to give an overview of how groundwater is assessed and managed elsewhere, and to draw conclusions relevant to South Africa. These examples are the United Kingdom, Australia and California. They were chosen on the basis of available data, and are also countries where big efforts are being made to understand groundwater resources at present.

2.1.1. United Kingdom

Introduction

Groundwater makes up about a third of water for public supplies in England and Wales, about 11% in Northern Ireland, and about 3% in Scotland (Grey et al., 1995). This reflects the different aquifer potentials across the UK and variations in the availability of surface water resources. If non-consumptive uses (such as cooling) are taken into account, groundwater makes up about 15% of water used. Although these figures show the importance of groundwater in the UK, they mask regional variations. For example, in the south-east of England, the most heavily populated part of the country, groundwater makes up most of the drinking water supply. Even in Scotland, where total groundwater use is low, there are around 30 000 private groundwater supplies, located particularly in rural areas where no other source may be available. Scottish groundwater is also used for public water supply and is economically important to industries such as breweries, agriculture and mineral water bottlers (SEPA, 2008).

The most important aquifers in the UK are the Cretaceous Chalk, the Permo-Triassic Sandstone, and the Jurassic Limestones. All of these aquifers show both fracture and intergranular groundwater flow, with the relative importance of each depending on the lithology and on the location. Borehole yields in all three aquifers can exceed 40 L/s.

Legal setting and institutions

Groundwater has been exploited in the United Kingdom for thousands of years, but laws specifically relating to national groundwater management date back to the 1945 Water Act, which defined national water policy and made some provisions for abstraction control and data collection (Grey et al., 1995). Further laws followed: the 1963 Water Resources Act recognised the unity of the hydrological cycle, and the river basin as the basic management unit. The 1989 Water Act led to the formation of the National Rivers Authority (NRA) as an independent regulatory body with responsibility for water resources. The Water Resources Act of 1991 obliged the state to “monitor the extent of pollution in controlled waters”, which includes groundwater (Koreimann et al., 1996). The NRA was succeeded by the Environment Agency (EA), which is today the primary public body mandated to protect and improve the environment in England and Wales (Scotland and Northern Ireland have similar organizations). The EA has a head office, eight regional offices and twenty two area offices,

and is responsible for the management of the UK's groundwater resources through information, education and the enforcement of regulation (such as licences). The EA carries out its own groundwater research, and also commissions research from other organizations such as the British Geological Survey (BGS) and private consultants. Groundwater quality is protected legally both at a European Union level (Groundwater Directive 80/68/EEC) and by UK law (Groundwater Regulations 1998), although these are set to evolve as new EU legislation (especially the Water Framework Directive) comes into force in the next few years. Groundwater levels (or abstraction quantities) are controlled through a licensing system enforced by the Environment Agency. The EA relies on an extensive monitoring network of both groundwater and surface water resources (the two are considered to be part of the same resource), supported by a programme of research including conceptual and numerical modelling of major aquifers. Both quantity and quality of water are measured. The EA works with groundwater users in industry (such as water companies), farmers, private users and research organizations to refine groundwater policy at the local level.

Groundwater management resources – maps, databases and publications

Hydrogeology maps at a scale of 1:625 000 cover England, Wales and Scotland, and are intended for broad planning and conceptual purposes. A series of more detailed regional hydrogeological maps at scales of around 1:125 000 exist for most major aquifer areas and some areas of lower groundwater potential (Figure 2-1-1). The information depicted on the regional maps varies from map to map, but typically includes potentiometric contours, annual rainfall, expected groundwater fluctuations, aquifer base levels, aquifer thicknesses, groundwater quality variations, locations of major abstractions and typical borehole hydrographs. The extent of saline water intrusion is shown on some of the maps. These maps give a reasonably good indication of expected borehole prospects, but for detailed local planning or for borehole siting and drilling further information is usually sought. Some of the information (e.g. the piezometric contours) is available as digital vector files for use in GIS systems, but the maps were compiled before the common use of GIS and are normally used as paper copies. They are available for sale from the BGS, the EA, and other outlets.

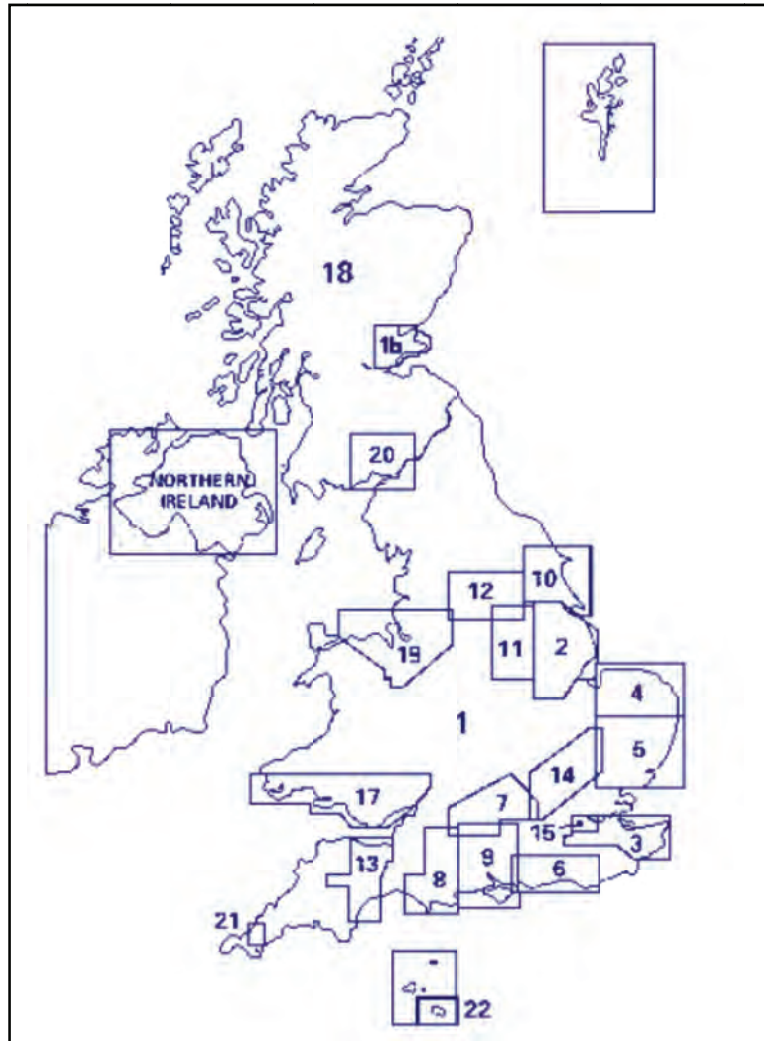


Figure 2-1-1 Availability of UK regional hydrogeology maps

A series of Groundwater Vulnerability maps covering the whole of England and Wales are published at a scale of 1:100 000, showing the vulnerability class of the underlying aquifers, based on type and yield of aquifer and the nature of the overlying material. The maps are based on the underlying aquifer classification, together with the soil leaching potential, depth to groundwater, presence of made ground and other factors. These are mainly used for planning purposes and for regulation, and further information is usually sought for local developments. The maps were prepared by the Soil Survey and Land Research Centre, and the British Geological Survey (BGS).

A large number of research reports and books are also available for British aquifers, including manuals on the properties of the major and minor aquifers produced by the BGS and series of reports on baseline groundwater quality for a number of aquifer regions produced by the BGS and the Environment Agency. The aquifer properties manuals show

the geographic extent of each aquifer, summarize its hydraulic properties based on all available data, and provide information on groundwater quality.

The British Geological Survey (BGS) holds data on over 105 000 water wells and boreholes in its "WellMaster" database. By law, information on all boreholes in the UK should be passed to the BGS for their records, although in practice this does not always happen. In common with South Africa, the lack of information in some records is often a problem. Copies of most of these records, some dating back to the nineteenth century or earlier, can be obtained by anyone for a small fee. The information contained in each record varies, but as a minimum would give the exact location of the borehole and usually some indication of the lithology. Some records include detailed information on water level changes, water quality, and pumping test results. Any investigation at local scale of groundwater prospects would normally consult this database. The Environment Agency keeps a database of all licensed and domestic groundwater users, together with information on groundwater fluctuations and quality changes from a network of monitoring sites. This information is used by the Environment Agency in its planning and regulatory functions.

Assessment methodologies

Early estimates of groundwater volumes and sustainable use by the Environment Agency (EA) were based on recharge calculations, or on baseflow separation methods (Burgess, 2002). Both of these methods are problematic however, and today the EA considers that conceptual and numerical models of groundwater areas are necessary to manage groundwater, including the impact of abstractions on river flows and on dependent ecosystems. The EA is currently working on a programme aimed at deriving numerical models of all the major aquifer units in England and Wales, which will be used to support groundwater management decisions (Burgess, 2002). By 1998, 30% of major aquifers were covered by numerical models, although not all of these models were updated or in use. Currently a good proportion of this modelling work is contracted out by the EA to the private sector, but in 2002 the EA's long-term strategy was to bring more of the modelling work in-house as the work brings peripheral benefits to EA staff (Hulme et al., 2002). This groundwater modelling work is aligned with national and regional water resource strategy documents published and updated by the EA. The basis for the UK's assessment

methodology for groundwater therefore appears to be numerical models of aquifers, based on sufficient high-quality data.

Exploitation of groundwater for commercial purposes (including public supply) in the UK would normally be preceded by a hydrogeological study by the company concerned, which would inform their decisions and also support their licence application to the EA. Currently many aquifer units in England and Wales are considered to be over-exploited, and licences for large abstractions are not available. (An exception to this is the London Basin, where rising water levels in the Chalk aquifer are a concern). Abstractions for private domestic use, up to a maximum of 20 m³/day, are however exempt from licensing but the EA must still be notified and a record of the borehole provided to the BGS (similar to a General Authorisation in South Africa).

In an attempt to align and coordinate research into groundwater, including groundwater resource estimation and sustainability, the UK Groundwater Forum was established in 1994. The Forum has published a document which lays out issues and research needs in UK groundwater (Grey et al., 1995), and continues to act to draw together partners from the research and regulatory communities and from industry.

Groundwater assessment and management in the UK has grown from a situation in the 1970s where groundwater management has been described as “very poor” (Lloyd, 1994:39) to one in which a greater recognition of the role of groundwater in national water security, environmental health, flood prevention and other factors has led to greatly improved management based on increasingly accurate data and more sophisticated modelling techniques.

2.1.2. Australia

Introduction

Australia is mainly arid to semi-arid, and has a land area of more than 7.6 M km². About 80% of the area (i.e. not population) of the Australian continent is mainly dependent on groundwater for consumptive use, and use of groundwater exceeds surface water in both Western Australia and the Northern Territory (Brodie, 2002). Concerns over security of water supply in Australia have been growing in recent years, particularly following the worst hydrological drought ever recorded (which began in 2005/6 and still hadn't broken in some

areas at the time of writing) and predictions of long-term climate change (Guardian, 2006). “Reform” of water policy and law is a strong theme in modern Australian policy and environmental direction, with an acknowledgement that certain modes of water use are not sustainable.

The most productive aquifers in Australia tend in general to be surface sedimentary aquifers, such as alluvium associated with rivers. For example, yields from irrigation wells in the Hunter River Valley in New South Wales can be up to 40 L/s (UNESCO, 2004). The Great Artesian Basin covers an area of about 1.7 M km², and underlies parts of four states. It is up to 3 000 m thick, and artesian flows of more than 100 L/s have been recorded (UNESCO, 2004). It supplies more than 600 000 ML of groundwater per year for various uses, and is currently the subject of a 15-year plan to improve management (GABCC, 2000). The representation and assessment of groundwater resources over such a large and relatively sparsely populated area magnifies a number of problems which occur world-wide, such as the difficulty of collating a sufficient density of data for assessments to be meaningful. Australia is relatively far advanced in making groundwater data available digitally and freely (i.e. no charge) available.

Legal setting and institutions

National Level

At national level in Australia the Department of the Environment, Water, Heritage and the Arts and the Department of Climate Change have Divisions dealing with water, including groundwater. Policy is implemented at State or Territory level via the state level departments, which vary in name and mandate from state (or territory) to state. For example in New South Wales the Department of Water and Energy delivers the New South Wales Government’s policy on water, whilst Queensland has a Department of Natural Resources and Water and an Environmental Protection Agency. Laws relating to water can be passed at both national and state/territory level.

A major “cross cutting” initiative at national level, known as the Australian Government National Water Commission, contributes to water (including groundwater) policy and assessment in all states or territories. The Commission is an independent statutory authority within the national Department of the Environment, Water, Heritage and Arts. The chair of the commission reports directly to the Minister. Following the 2005 National Water

Commission Act, the National Water Commission began to compile a baseline picture of Australia's water management and resource issues, known as "Australian Water Resources 2005". This picture has been compiled with the help of the Water Resources Observation Network (WRON), a network comprising of several Australian institutions with expertise in the water sector including the CSIRO, the Bureau of Rural Sciences, the Australian Bureau of Statistics, and the National Land and Water Resources Audit (see below).

In 2007 the Australian National Water Commission through its Water Science Group began a programme (with a budget of A\$ 82 million) to improve national knowledge and understanding of groundwater. The motivation behind the formation of the Commission included concerns over Australia's water resource management, a lack of data in some areas, and a skills shortage in the water sector. The groundwater programme's three main components are:

- The National Groundwater Assessment Initiative – the main part of the Action Plan, the Initiative funds groundwater work and research into areas where it is needed, ranging from harmonisation of groundwater terms and standards to the management of risks to groundwater quality. Proposals are solicited from eligible organizations.
- National Centre for Groundwater Research and Training – the Centre will train postgraduate scientists in areas of groundwater expertise.
- Knowledge and Capacity Building component – this initiative will develop groundwater guidelines and promote good practice to assist in groundwater sustainability.

The National Water Commission has just completed a study of the state of water planning in Australia, including an examination of case studies (Hamstead et al., 2008). The Australian National Land and Water Resources Audit was established in 1997 following the National Heritage Trust Act, and has just concluded its operations. The audit was a collaborative programme between the Australia's government and states to provide data, information, and nationwide assessments of Australia's natural resources (NLWRA, 2008). An early recommendation of the Audit was the need for more strategic data collection to ensure data is accessible, collated and provided to the community and all levels of government as information. The Audit worked closely with other national government agencies such as the

Office of Spatial Data Management, the Spatial Sciences Institute and the National Water Commission.

The Australian Bureau of Rural Sciences provides scientific advice to government on agriculture and related topics, and employs a multi-disciplinary team of scientists. Under their Integrated Water Sciences Program the Bureau carries out research into groundwater. The Bureau holds digital coverage of the 1:5 000 000 scale Hydrogeology of Australia map compiled by Jacobson and Lau (1987), and publishes reports on groundwater such as “An Overview of Tools for Assessing Groundwater-Surface water Connectivity”. The Bureau is currently engaged with a project called Water 2010 (see below) which seeks to model national catchment water balances, a part of which is an assessment of groundwater recharge.

The CSIRO (Commonwealth Scientific and Industrial Research Organization) of Australia conducts research into a wide range of environmental problems (amongst other research areas). Its Hydrology Research Programme includes a Groundwater and Surface Water Hydrology Group, whose expertise includes the development of conceptual and numerical catchment models.

Headquartered in Canberra, Geoscience Australia (formerly the Australian Geological Survey Organisation) is Australia’s national geological science organisation, producing geoscientific information and knowledge. Geoscience Australia also carries out research into groundwater resources via its Groundwater Group, part of the Geospatial and Earth Monitoring Division. Geoscience Australia works in partnership with state-level geological organisations, such as the Northern Territory Geological Survey or the Queensland Department of Mines and Energy.

The Australian Bureau of Meteorology (BoM) is headquartered in Canberra, and has state offices in each state capital. Its main focus is on climate, but it collaborates with other organizations in water resource assessment. The BoM has a Water Resources Group, which is involved with a number of projects aimed at assessing Australia’s water resources, including groundwater. These include the Australian National Land and Water Resources Audit and the National Water Initiative (both mentioned above). Data and information about these programmes can be downloaded from the BoM website at <http://www.bom.gov.au/hydro/wr/>.

State Level

The day-to-day practicalities of groundwater management are generally carried out at state level or lower and state level institutions normally regulate groundwater abstractions – although the exact policy depends on the state or territory. For example, in New South Wales “Available Water Determinations” are made by the (state) Minister for Water under the *Water Management Act (2000)*. These “determine the volume of water available for extraction for the various categories and subcategories of access licences in relation to those water sources covered by water sharing plans throughout the State” (NSW, 2008). In many parts of the state, new water licences are “embargoed” (no new ones available), and prospective water users must purchase existing licences. Furthermore, the state can (and does) seek to reduce some licence allocations when negative impacts occur – for example a new groundwater plan for the Lower Gwydir area of New South Wales required large cuts in entitlements, although final figures were only agreed after extensive consultation with (and campaigning by) affected parties (Hamstead et al., 2008). Licence allocations may be reviewed annually, in line with predictions for the “water year” ahead. At the level of individual boreholes, owners need to obtain consent from the New South Wales Department of Water and Energy to drill the borehole, but the requirement for a licence depends on what the water will be used for (e.g. domestic and stock use normally does not require a licence). The Department of Water and Energy operates a “Groundwater Drilling Unit” which carries out drilling tasks not normally tackled by ordinary drilling contractors (e.g. deep artesian boreholes, unstable formations, etc.). The Unit has several specialised drilling rigs, a pumping test rig and a geophysical logging truck. Drillers in Australia are required to be licensed, and must submit their drilling records to the appropriate authorities where they are kept as a database.

Maps and resources

The earliest published groundwater maps in Australia were outputs of the state geological surveys in the nineteenth century, but there is evidence that Aboriginal peoples used diagrams showing water sources in prehistoric times (Brodie, 2002). A major output of the National Land and Water Resources Audit was the Australian Natural Resources Atlas, which includes mapping with a hydrogeological basis such as the mapping and categorization of groundwater flow systems at a national scale, and estimates of relevant parameters such as soil hydraulic conductivity and soil water content. Digital data is available on-line from the

Australian Natural Resources Data Library at <http://adl.brs.gov.au/anrdl/php/>, much of it free of charge. The Atlas is intended to be used by managers and community groups for planning and management purposes.

In addition to hydrogeological maps published at a small scale by Australian government agencies, other hydrogeological maps include those published by state groundwater agencies at a larger scale covering important state groundwater resources (Figure 2-1-2). These maps include:

- The 26 maps at 1:250 000-scale of the *Murray Basin Hydrogeological Map Series*
- The 1:5 000 000 scale map of the *Hydrogeology of the Great Artesian Basin* (Habermehl and Lau, 1997), which covers roughly a fifth of Australia (about 1.7 Mkm²)
- Specialised hydrogeological maps (e.g. groundwater vulnerability, salinity hazard/risk and groundwater dependent ecosystems) developed in the 1990s
- State-level mapping covering priority groundwater management areas

Most of these maps are available as digital GIS coverages, and much of this data is free to download from the websites of institutions such as the Bureau of Rural Sciences and the Australian Natural Resource Data Library. Efforts are being made to standardize digital hydrogeological data, and to fill data gaps. The aim is to allow the production of maps customised for area and for theme, based on a variety of “underlying” hydrogeological and other datasets in digital format.

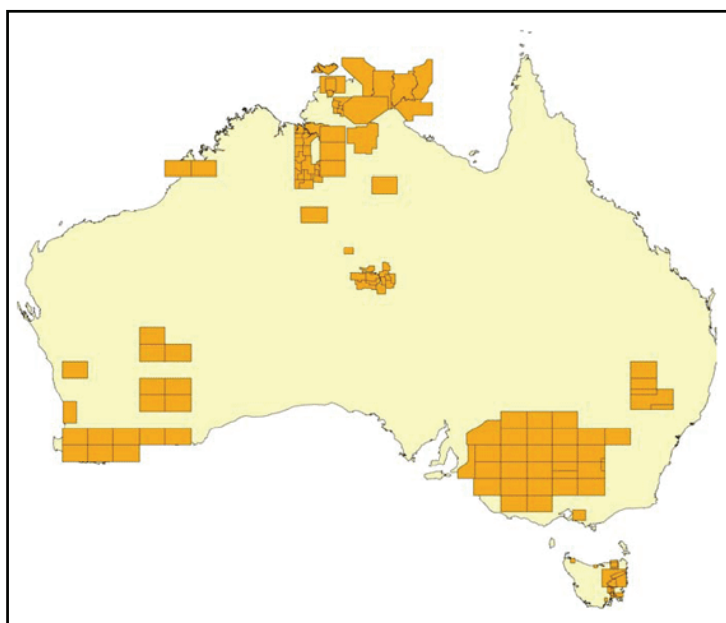


Figure 2-1-2 Extent of published hydrogeological maps at 1:250 000 or larger, after Brodie (2002)

Efforts are being made to understand the requirements of groundwater-dependent ecosystems (GDEs), and to incorporate these into planning and regulation. According to the National Water Commission website (<http://www.nwc.gov.au/www/html/225-groundwater-dependent-ecosystems.asp>) six types of GDEs are conventionally recognised in Australia:

- Terrestrial vegetation that relies on the availability of shallow groundwater
- Wetlands such as paperbark swamp forests and mound springs ecosystems
- River base flow systems where a groundwater discharge provides a baseflow component to the river's discharge
- Aquifer and cave ecosystems where life exists independent of sunlight
- Terrestrial fauna, both native and introduced, that rely on groundwater as a source of drinking water
- Estuarine and near-shore marine systems, such as some coastal mangroves, salt marshes and sea grass beds, which rely on the submarine discharge of groundwater.

Assessment methodologies

Better groundwater data is recognised as a big factor in improving groundwater assessment and management in Australia, and the shift from paper or map-based data to digital GIS coverages has been taking place since the mid-1980s. Data formats are also being standardized (Brodie, 2002). These developments make it easier to tailor data to a specific use, such as planning and managing new abstractions, and also make data more accessible. A working group, with representatives from Australian groundwater institutions, is responsible for drafting national groundwater data standards (The Australian National Groundwater Data Transfer Standard). At the same time, there appears to be growing concern in Australia over a number of groundwater related issues:

- The country is semi-arid to arid, and highly vulnerable to droughts and to the effects of climate change. Severe droughts have endangered economic output in recent years. There is now an Australian national Minister for Climate Change and Water, and the Department of Climate Change was established in December 2007.
- Water resources in Australia, including groundwater, are over-allocated in some areas, and are also threatened by pollution. Different users compete for water.

- Groundwater does not receive the recognition that it merits in terms of its strategic or economic importance.
- Environmental flows and groundwater-dependent ecosystems are imperfectly understood.
- Current management arrangements related to groundwater are likely to be inadequate.
- There are gaps in the scientific understanding of Australia's groundwater resources, and a lack of skilled groundwater scientists.
- Groundwater and surface water should be seen as interdependent, but this is not always the case.
- There are inadequate legal and management instruments (such as trading of entitlements) to facilitate better groundwater management.

Estimates of national groundwater recharge, runoff and evapotranspiration in Australia depend mainly on water balance modeling (conceptual, analytical and numerical), with data derived from a variety of sources and organizations. The Australian Water Resources 2005 Water Availability Assessment (part of the National Water Commission's work) depended on water balance assessments undertaken for 51 priority geographic areas. This methodology is known as the "water accounting approach", a method being developed in Australia which takes the dynamic nature of water resources into account (i.e. different time-scales may apply to surface water and groundwater, and a single year may well not be representative of the resource) (see Figure 2-1-3). The work is still underway, as part of the Water 2010 Project and Australian Water Availability Project being carried out by the Bureau of Rural Sciences (2008). The project aims to release reports summarising average annual water availability and use for every River Basin and Drainage Division in Australia. Essentially, the aim is "to develop an operational system for estimating soil moisture and other components of the water balance, at scales ranging from five kilometres (km) to all Australia, over time-periods ranging from daily to decades" (Bureau of Rural Sciences, 2008). This will help with future planning, decision making and risk assessment. Reports released to date for the Australian Water Availability Project can be downloaded for free from the Bureau of Rural Sciences shop at <http://affashop.gov.au/>.

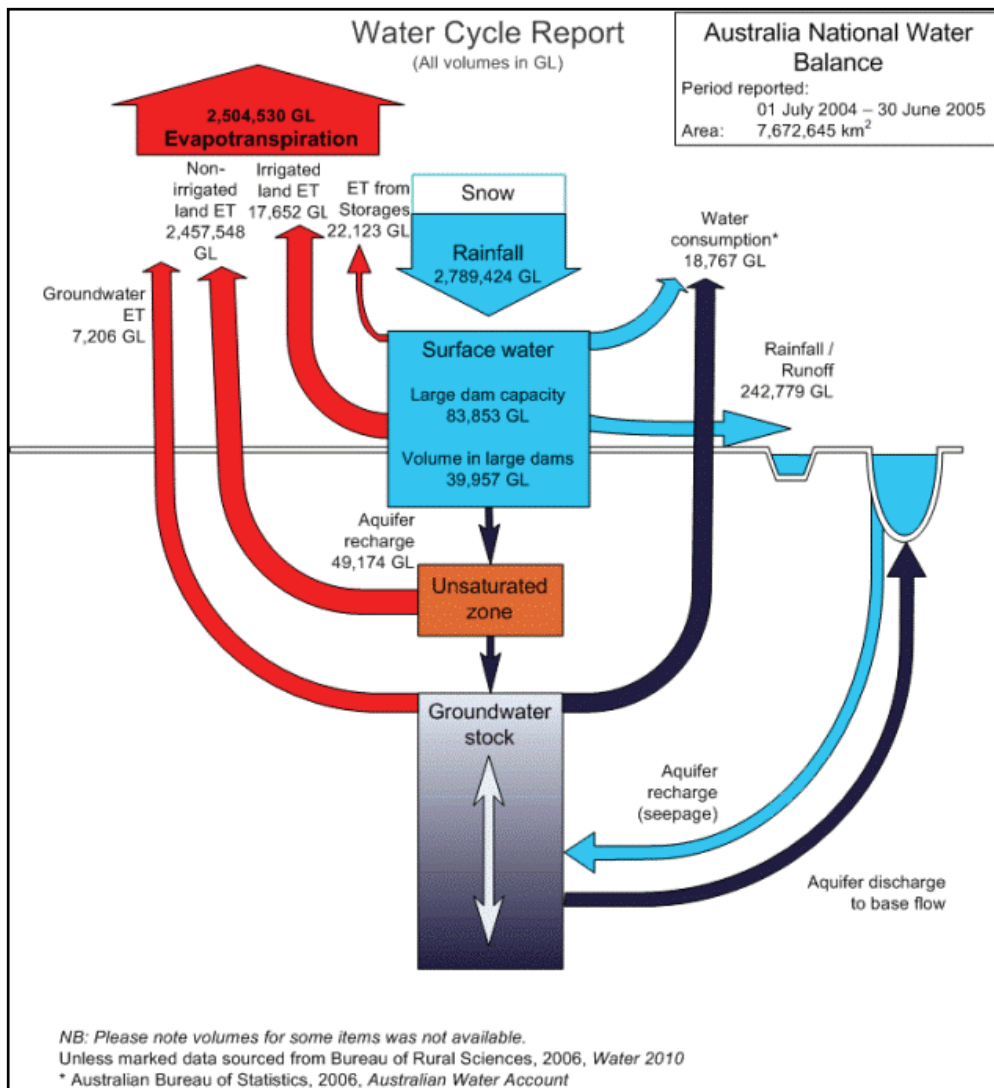


Figure 2-1-3 Diagram of Australian approach to water balance assessment (National Water Commission website)

It is acknowledged that there is a great variation in groundwater recharge, depending on topography, surface morphology, vegetation, geology and other factors. The National Land and Water Resources Audit (NLWRA) estimated total national water inflows at 291 953 gegalitres (1 gegalitre = 0.001 km³) for the 2004-05 year, which includes a total surface water runoff of 242 779 gegalitres and groundwater recharge of 49 174 gegalitres. The volume stored in large dams (wall height greater than 15 m) was 44 164 gegalitres as at 1 July 2004, making a total water resource estimate for Australia of 336 117 gegalitres (or just over 336 km³). The NLWRA did not recognise the close interaction of surface water and groundwater resources. As a result, 'double accounting' of the water resource (i.e. counting a parcel of water as contributing to both surface water yields and groundwater yields) was recognised as an issue that was not adequately addressed in NLWRA, which may have led to over-estimating of the quantity of the resource in that report. The total storage *capacity* of

large dams across Australia in 2004-05 was 83 853 gigalitres (84 km³). This figure excludes small dams (such as farm dams) which are much more difficult to assess.

Local groundwater management in Australia should therefore be seen against a background of growing concern over national water resources, and a move towards more efficient and conjunctive prediction and management of water amidst predictions of worsening shortages. Greater transparency and consultation is also desirable. Depending on the location, a plan to exploit groundwater locally would start with an interrogation of existing data (and/or maps), but should ultimately rest on a conceptual (or better a numerical) model of local water resources, taking into account climate variability and the local and regional environmental requirements. Groundwater policy at both national and state level would need to be followed, and all necessary permissions and licences obtained. Continued use of the groundwater resource would be based on a broadening understanding of the state of the resource, based on better data collection and focused research. Meeting of demand for water would depend not only on better exploitation of the resource (taking into account sustainability concerns), but also on demand control (limits to entitlements) through education, water-saving measures and pricing. These measures would be enforced at state/territory water planning level.

2.1.3. California

Introduction

California is the most populous state in the United States of America, and the third largest by land area (414 000 km²). Geographically it ranges from deserts to high mountains, with deserts making up about a quarter of the surface area, and forests covering another third. The most important aquifer is the basin-fill Central Valley Aquifer, which is extensively used for water supply and irrigation. On less than 1 percent of the total farmland in the United States, the Central Valley supplies 8 percent of the national agricultural output by value – mainly due to irrigation, part of which is groundwater from the Central Valley aquifer (Reilly et al., 2008). Most of the other major aquifers in California are also basin-fill, consisting of sediments which have filled structural depressions. Volcanic rock and carbonate aquifers provide smaller, local groundwater resources.

Groundwater today meets about 30% of California's urban and agricultural water needs on average, rising to about 40% in drought years (DWR, 2003). Many small to medium size

towns such as Fresno (pop. > 400 000) or Lodi (pop. > 55 000) in California are entirely dependent on groundwater for their water supplies, and nearly half of Californians rely on groundwater for at least part of their water supply (DWR, 2003). In many parts of the State there is evidence of a steady decline in groundwater levels, and in other areas declines are expected if current management strategies are continued. Good data is available for the most heavily used aquifers, but in many other areas data is scarce.

Legal setting and institutions

Law in the United States operates at both national (Federal) and State level. States have their own constitutions and state governments, and pass laws on a wide range of issues not covered by Federal law (which is based on the US Constitution). Water law principles differ depending on the state. California adheres to the “prior appropriation” system of surface water rights, which means that a right to water is not necessarily owned by the property owner, but can be sold or mortgaged like a piece of property. Each surface water right has an appropriation date (date first used) and a yearly quantity. The oldest or “senior” appropriation has prior right to the water, in cases where not all allocations can be met. In terms of groundwater, the owners of the land overlying an aquifer in California have a right to a reasonable amount of that groundwater for their own use. The amount depends partly on the surface area of the land owned. This is similar to the riparian system for surface water use. It is not known how interaction between groundwater and surface water is accommodated legally in California. Water allocations and regulation are overseen by the state government and its agencies, and this can devolve to county level – for example Sonoma County has a Water Agency which is responsible for water planning in the county.

The United States Geological Survey (USGS) is the main water, earth, and biological science and civilian mapping agency in the United States, and employs around 10 000 people. The USGS head office is in Reston, Virginia, and it has a Region Office in Colorado and another in California. Offices which are part of or affiliated to the USGS exist in every state.

The United States Environmental Protection Agency (USEPA) is headquartered in Washington DC, has ten regional offices and employs around 17 000 people. The USEPA implements environmental law by writing and enforcing regulations (for example the Clean Water Act), provides grants, works in partnerships with other environmental organisations, and conducts scientific research into environmental issues. The USEPA also publishes

information and teaches people about the environment. The USEPA's Office of Ground Water and Drinking Water works to ensure safe drinking water, and to protect groundwater. The USEPA also implement the comprehensive environmental response, compensation, and liability act (Figure 2-1-4).

The Superfund

In 1980 the Comprehensive Environmental Response, Compensation, and Liability Act (known as "Superfund") was passed in the United States, beginning a programme of cleaning up hazardous waste sites, including groundwater contamination, across all 50 states. Superfund gives the USEPA authority to find parties responsible for pollution of the environment, and compel them to cooperate in cleaning up the pollution. In the case of sites where the owner or responsible party cannot be found, the USEPA is enabled to direct the clean-up themselves, and recover costs through a variety of mechanisms where possible. The USEPA works closely with the relevant state environmental agencies and other parties in the work. The Superfund legislation has been responsible for several notable groundwater pollution remediation projects at "superfund sites" across the United States.

Figure 2-1-4 Box introducing Superfund legislation in the USA

The California Environmental Protection Agency was formed in 1991, and has six boards or departments under it employing about 5 000 people. It is headed by the Office of the Secretary, who is responsible for coordinating and overseeing the activities of the boards, for strategic planning and for budget review. The Board tasked with water regulation and protection is known as the California State Water Resources Control Board. The State Water Board has primary responsibility for balancing the needs of various water users, including industry, agriculture, domestic users, and the environment. The Board was created in 1967 by merging two former boards. The Board allocates water rights, adjudicates water right disputes, develops state-wide water protection plans, and establishes water quality standards. One of the stated goals of the State Water Board is that "groundwater is safe for drinking and other beneficial uses". The Board has five full-time, salaried members. Nine Regional Water Quality Control Boards are located in the major watersheds of California, and are the main implementing agencies for both state and federal water pollution laws. Each Regional Board has nine part-time members. State and Regional Water Board members are appointed by the State Governor and must be confirmed by the State Senate. A Basin Plan exists for each watershed and provides a scientific and regulatory basis for basin water protection work. The regional Boards are semi-autonomous and are responsible

for setting water quality standards, issuing waste-discharge permits, and checking and enforcing compliance. The State Water Resources Control Board recently formed the Groundwater Resources Information Sharing Team (GRIST) consisting of several State and federal agencies with groundwater-related programs, in an attempt to better coordinate data exchange and avoid duplication of effort (DWR, 2003).

Maps and resources

The Ground Water Atlas of the United States (Miller, 2000) is published by the United States Geological Survey (USGS) and describes the location, extent and hydrogeological characteristics of major aquifers in the United States. The atlas has 14 chapters – an introductory chapter (Figure 2-1-5) and 13 chapters covering regional areas called “segments”, which together cover the land area of the United States. The segment covering California (Segment 1) also includes the state of Nevada. The atlas is written so that it would be accessible to non-specialists and it avoids technical jargon where possible. Each segment begins with an overview of the climatic, geological and hydrological conditions. Aquifers are described in terms of their location, extent, thickness, water level conditions and water quality (Miller, 1994). The atlas was intended to improve public information and awareness regarding groundwater as one of its functions, as well as providing a useful planning and overview document. It is freely available on the USGS website.

The USGS Ground-Water Resources Program’s Circular 1323 “Ground Water Availability in the United States” (Reilly et al., 2008) describes groundwater in the USA, including major aquifers, water level and water quality trends, and a description of regional scale resource assessment.

The USGS and its affiliates maintain a network of boreholes to provide statistics on groundwater levels across the United States. The USGS also has a large database of borehole and groundwater information which is managed at state or local level. It includes surface water, groundwater and water quality data, and the groundwater database alone contains records from about 850 000 boreholes or wells. Some of these records are a century old. Water levels at about 20 000 of these sites are monitored annually, for a variety of reasons including state monitoring programs or as part of local and regional research projects (e.g. the High Plains Aquifer Monitoring Program). Data and information about the records are available to the public over the web at the National Water Information System Web

Interface site at <http://groundwaterwatch.usgs.gov/>. In the State of California it is estimated that there are about 10 000 boreholes or wells which are monitored for water levels. The distribution of these tends to be biased towards the main aquifers or basins (DWR, 2003)

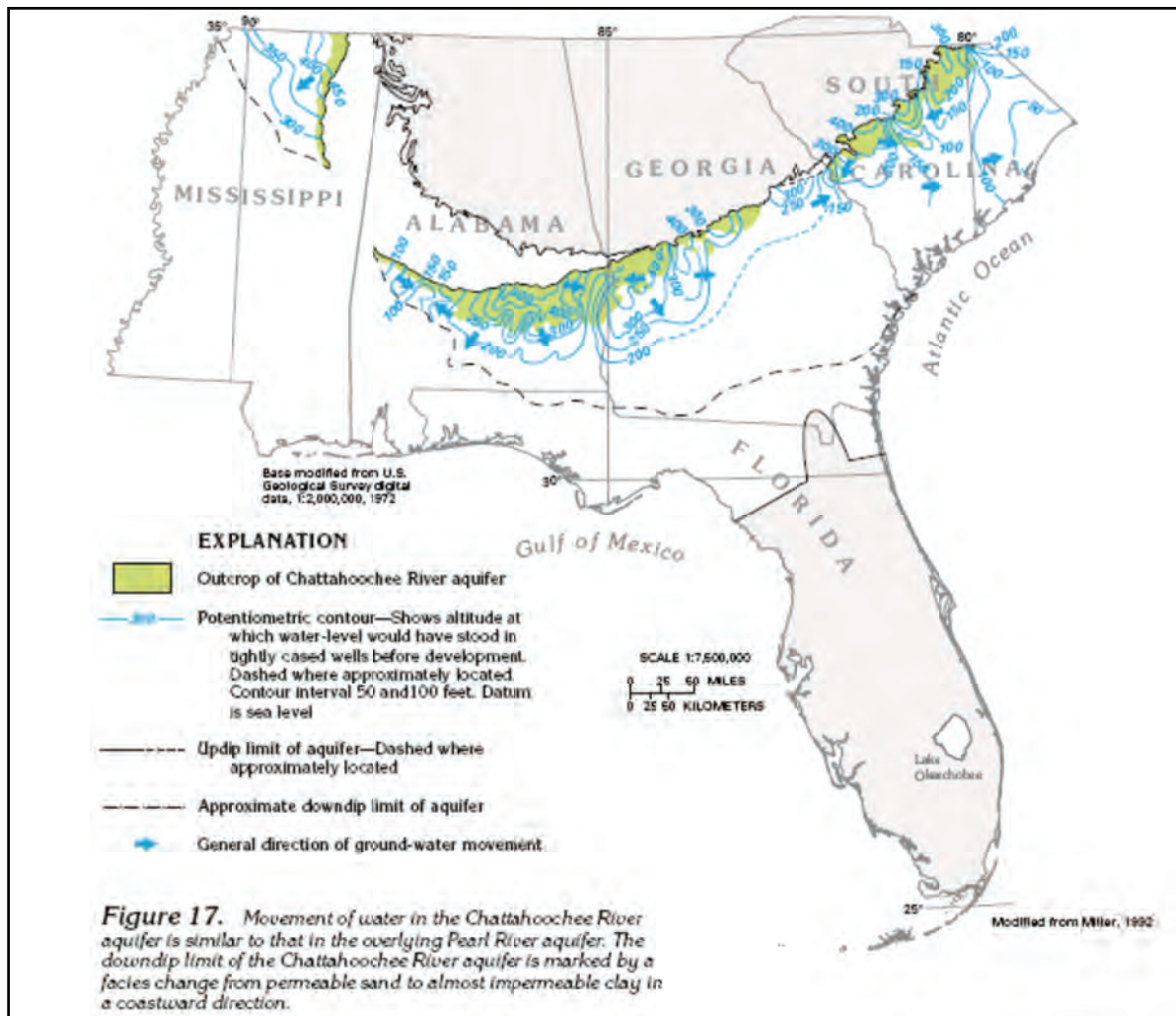


Figure 2-1-5 Detail from the Introductory Chapter of the Ground Water Atlas of the United States (Miller, 2000)

Assessment methodologies

No single “assessment methodology” for the whole state of California appears to be accepted state-wide. Certain trends can however be identified. There appears to be a shift towards more local (county level) groundwater assessment and management in California, following legislation passed in the early 1990s. There are efforts being made towards making or improving local groundwater management plans, and integrating these plans with other agencies such as water suppliers. These plans often call for more data, and local governments are becoming involved in data collection and interpretation. However there is no obligation on the part of local planners to submit groundwater management plans to the

State Department of Water Resources. Cooperation between local (county) level agencies sharing an aquifer is sometimes not adequate for effective management purposes. Local groundwater management plans ideally need to be integrated with regional or basin scale aquifer assessments, such as are conducted by the USGS (e.g. their “RASA” or Regional Aquifer-System Analysis Program, begun in 1978), although it is not known whether there is a formal mechanism for this at present. Regional scale aquifer assessments would typically aim to develop a numerical model of all or part of the regional aquifer system, based on as much data as possible.

2.2. Review of the South African Groundwater Resource Assessment Phase II (GRAII)

The Groundwater Resource Assessment of South Africa – Phase Two (GRA2) initiated by the Department of Water Affairs in 2003 was aimed at building on the short-comings of Phase One (GRA1) and more accurately quantifying the groundwater resources in South Africa on a national scale. The project produced a methodology for the quantification of the country’s groundwater resources, which includes algorithms for the estimation of storage, recharge, baseflow and the impact on the reserve as well as present groundwater use. Several datasets were produced as the methodology was applied to the production of a set of maps which can be used on various levels of planning and management (DWAF, n.d.). The assessment methodology is presented in a series of reports: Quantification, Planning Potential Map, Recharge and Groundwater/Surface Water interaction, Aquifer Classification and Groundwater Use.

The following critical review of the applied methodologies is accordingly structured into the five topics. It must be emphasized that the review does not entail any verification or review of the derived data on e.g. groundwater recharge itself. Each review topic consists of a summary of key outputs and a description of the methodology used to arrive at the outputs (including critical review comments).

2.2.1. Groundwater quantification

Key outputs

- Country-wide 1 km x 1 km grid showing average groundwater storage.
- Country-wide 1 km x 1 km grid showing current groundwater storage based on previous year's input data (especially recharge and abstraction).

Methodology:

The quantification of the national groundwater resources is based on a water balance approach, i.e. what flows in must flow out (including abstractions) or cause a change in storage:

$$(Q_{in} + \Delta Q_{in}) - (Q_{out} - \Delta Q_{out}) + Q_{ab} = \Delta V \quad (2.1)$$

where,

Q_{in} is the flux into the system,

ΔQ_{in} is the change in the flux into the system,

Q_{out} is the flux out of the system,

ΔQ_{out} is the change in the flux out of the system,

Q_{ab} is the total abstraction from the system, and

ΔV is the change in the total volume of the system.

Groundwater storage is calculated on a 1 km x 1 km grid as the sum of water volumes stored in the weathered and fractured zones of South African aquifers (product of specific yield (S_y), specific storage (S_s), saturated thicknesses and surface area) and aggregated to a quaternary catchment scale.

The storage is subdivided into a static storage zone S_{Static} (volume of groundwater available in the permeable portion of the aquifer below the zone of natural dynamic water level fluctuation) and a dynamic storage zone $S_{Dynamic}$ (volume of groundwater available in the zone of natural dynamic water level fluctuation). The static and dynamic storage zones are related to five "physical" aquifer levels defined in separate GIS layers as follows (Figure 2-2-1):

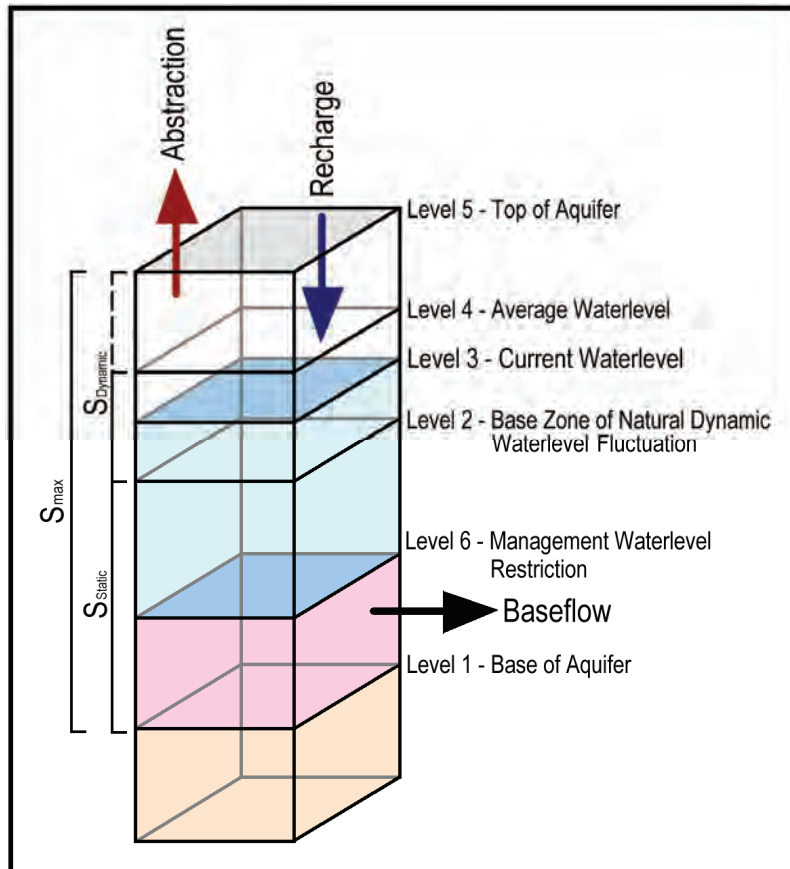


Figure 2-2-1 Aquifer levels used to assess stored volumes of groundwater

Critical review of input parameters:

The base of the aquifer (Level 1) is in this context defined as the depth of readily available and exploitable groundwater, determined as the depth where the water strike frequency (density) graphs approach a maximum (depth of weathered zone, absent for TMG) or zero (depth of fractured zone). The water strike frequency plots are essentially a modification of Seymour's (1996) approach, using an extended database of 152 569 NGDB records for Vegter's 64 groundwater regions. If too few data is available for a specific region, depths of geologically similar, adjacent regions are used. Following the initial determination based on water strike frequencies, the thickness of the weathered zone is corrected for the topographical setting, i.e. reduced by e.g. 5% for slope angles of 5°.

While the approach obviously makes best use of available data, it is generally a subjective approach and fits of strike density graphs for some regions appear arbitrary or based on other (unsubstantiated) information. Furthermore the often insufficient number of deeper water strike data causes a statistical bias in the density curves. A correction of deeper water strikes densities as a function of the borehole depth distribution (to avoid diminishing

maxima in weathered zone) appears necessary. Additional shortcomings of the approach include the unrealistic assumption of a completely (up to base of weathered zone) saturated fractured zone and the assumption of homogeneous regions. Vegter's groundwater regions are geologically heterogeneous and single derived parameters of strike densities, which are in the following assigned to quaternary catchments; result in crude approximations of aquifer depth. Subsequently the determined average depth to the base of the weathered zone for Vegter's 64 groundwater regions is a very deep 56 m below ground level. The average base of the fractured zone on the other hand is a rather shallow 116 m below ground level. For selected regions the fractured zone is extremely thin (thickness below 20 m, e.g. regions 12, 20, 24, and 38) and in 4 regions (regions 51, 54, 55 and 63) the weathered zone is apparently deeper than the fractured zone, indicating clearly conceptual errors.

The base zone of natural dynamic water level fluctuations (Level 2) is defined in the methodology report (not defined in the final report) as the lower depth range of the maximum of strike-density curves for Vegter's (2001) 64 groundwater regions (and included quaternary catchments). As stated above for level 1, the determination is subjective, sometimes arbitrary and assumes homogeneous regions. Additionally, topographic effects (recharge-discharge areas) on water level fluctuations are neglected and instead a geologically determined parameter (water strikes in boreholes determined over a period of decades!) is used to define the base of water level fluctuations. This approach neglects systematic fluctuations of water levels due to drought cycles.

The current groundwater elevation (Level 3), primarily used to estimate available groundwater storage for the following year, is calculated from recent NGDB elevation values. Using water levels measured at different times, the approach neglects seasonal water level fluctuations. Furthermore the spatial distribution of boreholes captured in the NGDB is not uniform, and a bias may exist towards boreholes potentially influenced by abstractions. However, correcting regional water levels for abstractions based on (for example) the WARMS database is virtually impossible.

The determination of the average groundwater elevation (Level 4) is not given in the final report, but apparently defined in the methodology report as the centre of Vegter's (1995) groundwater level interval map. It therefore represents average water levels under the

climatic conditions and abstractions up to the 1990's, without consideration of more recent conditions. As before the heterogeneity of the regions is neglected and a secondary subdivision of water levels into quaternary catchments used.

The top of the aquifer (Level 5) is also not defined in the final report, but given in the methodology review as the minimum elevation of the 1 km x 1 km grid used for the resource assessment. While the top of the weathered aquifer is related to the surface elevation, it does not coincide exactly with it due to the occurrence of soil horizons, calcrete or ferricrete layers, etc. Furthermore the accuracy of the surface elevation depends on the digital terrain model (DTM) resolution. In this case the DTM from the 90 m x 90 m Shuttle Radar Topography Mission was used.

Level 6 is used in the calculations to represent management water level restrictions based on environmental, legal or other constraints placed on the volumes of water that may safely be abstracted. While it is incorporated into the model, it is not used in the regional determination of available groundwater resources.

Additional important data requirements for the methodology include the specific yield and storage coefficient of the weathered and fractured zones respectively. Both parameters are not readily available on a regional scale and an exponential decrease (within reasonable limits) of the parameter with depth is assumed. The decrease function is altered for each region to arrive at reasonable values at required depths, e.g. zero at the base of the fractured zone. The derived average yield for Vegter's 64 regions is around a relatively low $8E-03$ and the average specific storativity is approximately $3E-04$.

The applied methodology is essentially a continuation of the GRA I methodology with a larger NGDB database available. A major shortcoming is that the aquifer information is inferred from non-randomly distributed data (e.g. the NGDB boreholes are typically sited on anomalies around population centres (Figure 2-2-2) and it is assumed that the data is representative of each groundwater region.

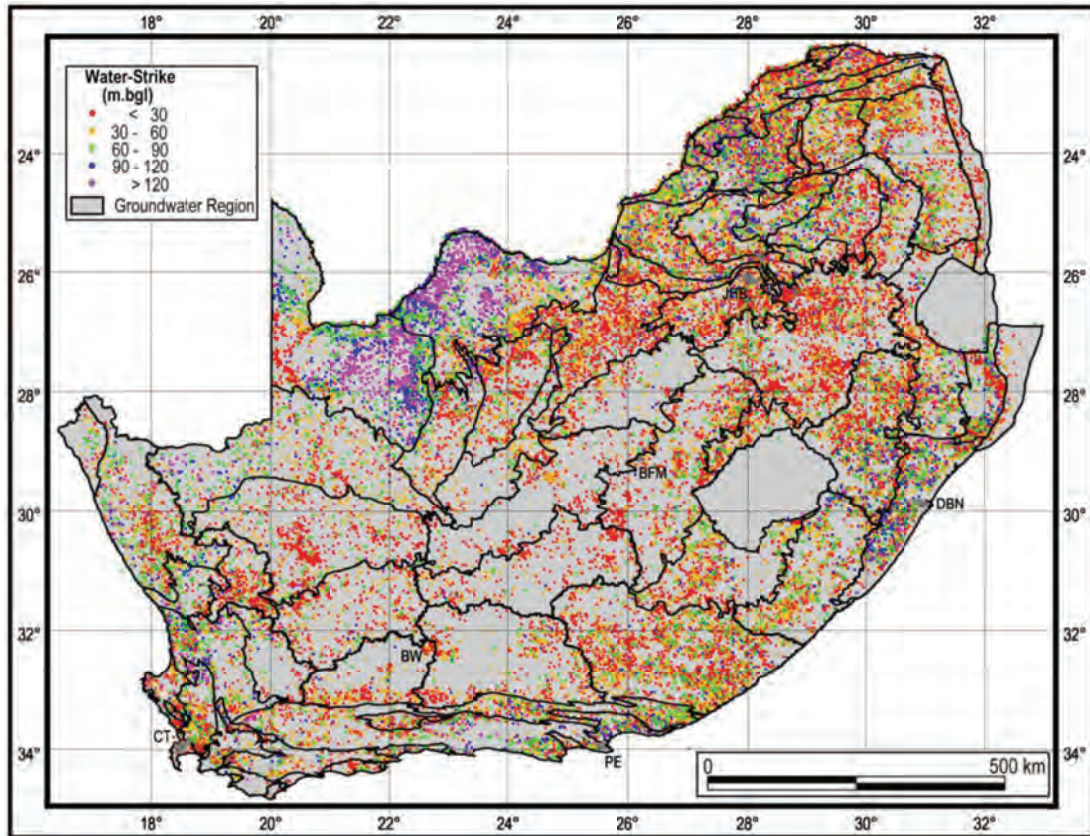


Figure 2-2-2 Example of a mean water strike map showing spatial distribution of the NGDB data

2.2.2. Groundwater recharge

Key outputs

- Country-wide 1 km x 1 km grid showing GIS based recharge estimates aggregated to quaternary catchment scale.
- Annual and mean annual recharge values per quaternary catchment.
- Recharge threshold values (RTV = monthly rainfall below which no direct groundwater recharge occurs) per quaternary catchment. Note that this neglects preferred recharge.
- National recharge volume estimated at 30.52 km³/a or 5.2% of mean annual precipitation (Vegter (1995) estimated 33.82 km³/a or 5.8% of MAP)

Methodology:

National recharge values are calculated on a quaternary catchment scale using a chloride mass balance (CMB) approach, i.e. the ratio of chloride content in rain and groundwater respectively. In order to apply the CMB method despite limited spatial coverage of chloride measurements in precipitation, theoretical linear relations between mean annual

precipitation (MAP) and chloride content, elevation and chloride content as well as distance to coast and chloride content (only for coastal regions) were derived for coastal and inland regions. The relations were used to derive grids of chloride content in precipitation, before these were averaged for coastal and inland regions and combined to a national grid, which was subsequently smoothed. It is obvious that the various mathematical operations limit an assessment of the reliability of derived data, but the initial poor to almost absent correlations between chloride content in precipitation and MAP or elevation (r^2 only around 27%) for the inland station question the applied approach. The apparently better correlation between the final smoothed calculated dataset and observed values ($r^2 = 79\%$) is a direct result of the predominance of coastal chloride values with high concentration ranges. The inland values show a completely different (linear) relation in comparison to coastal values, and calculated correlation coefficients are therefore not representative. The general underestimation by the model of chloride contents in inland precipitation should theoretically result in an overly pessimistic estimation of inland recharge figures.

Chloride contents in groundwater were interpolated (kriging) for a 1 km x 1 km grid based on 28 465 measurement locations extracted from the NGDB (a harmonic average was used for stations with time series data, or if multiple stations fell into one grid cell). Interpolated values were overwritten with measured values for the cells containing such.

In view of cyclic variations of chloride content or even linear trends for impacted stations the calculation of harmonic averages for stations with time series data is generally questionable. Furthermore the data should have been classified into annual data sets and subsequently used to estimate annual recharge values. Such an approach would have enabled the identification of potential cyclical effects and trends, which currently average each other out. Excluding pre-1980 data from the dataset furthermore results in a potential time difference of up to 26 years for determined chloride values, a time span during which land use changes and climate changes could have occurred and impacted on chloride concentrations. Another major shortcoming of the method refers to the applicability of the CMB method itself, which originally uses the chloride content in soil water and/or the groundwater surface, while the NGDB data represent mostly pumped samples to achieve a sample representative of the aquifer and not necessarily the unsaturated zone.

The two datasets (grids) are subsequently used to estimate “raw” recharge percentages based on their ratio, neglecting dry deposition of chloride (especially in coastal regions and an important contribution to the salt load). The derived grid is again smoothed before it is “calibrated” (using multiple linear regression) against known recharge values by introducing rating (influencing) factors accounting for the depth to water table, soil drainage rate, rainfall seasonality, geology, land cover, topography as represented by slope and the coefficient of variation of annual precipitation. While the correlation coefficient (r^2) between calculated recharge values and point values from literature improves from 20% for the “raw” estimates to around 45% for the “calibrated” estimates, it must still be considered as a very poor (or even non-existent) correlation. After a final smoothing of the recharge grid it is used to determine the recharge values per quaternary catchment as well as for the country. The derived national recharge figure of 52.7 km³/a (equal to an average recharge of 9% of MAP) appears rather high.

The smoothed “GIS calibrated” recharge grid values are again “calibrated” (using linear regression) against recharge values determined in the GRA2 GW/SW interaction project (reducing the national recharge estimate to 27.2396 km³/a), before the recharge values of 413 quaternary catchments are “adjusted” to match minimum baseflow values (per quaternary catchment) as determined by the GRA2 GW/SW interaction project. The final “adjusted”, “calibrated”, “GIS-calibrated and smoothed” recharge grid estimates the national recharge volume as 30.5187 km³/a, or 5.2% of MAP. Correlation (r^2) between the final estimated recharge values and point values from literature improves to a still disappointing 56%. It is obvious that the various smoothing and calibration or adjustment procedures do not help to improve the reliability or scientific defensibility of CMB recharge figures, and a further discussion is deemed unnecessary. The recharge figures should essentially be recalculated with more chloride data, or different methods (e.g. the Cumulative Rainfall Departure method or the Water Table Fluctuation method). It is furthermore interesting to note that the correlation between the “final estimates” and Vegter’s recharge estimates show a clearly non-linear relation between the two with a calculated correlation coefficient (for a linear relation) of 67% with the “adjusted, calibrated, GIS-calibrated and smoothed” recharge estimates appearing to limit recharge values to threshold values like 100 mm.

The groundwater recharge project team furthermore estimated precipitation threshold values (PTV) for recharge to occur per quaternary catchment. The PTV was determined for each quaternary catchment by calculating the cumulative distribution of estimated monthly recharge values based on monthly rainfall data for the period 01/2000 to 08/2004. A 98% non-exceedence criterion (assuming a normal distribution) for recharge was used to set the PTV at the corresponding precipitation value. The short time series of rainfall data considered obviously questions the application of statistics based on large data sets and influenced by cyclic climate variations. Beyond this short-coming the setting of PTV at a percentage of mean monthly precipitation is an over-simplification of the underlying processes, as noted by the authors themselves, and renders determined values questionable and difficult to defend. Accordingly within the GRA2 project itself the PTV was contradicted by the surface-groundwater interaction group, which set it at 200 mm.

2.2.3. Groundwater surface water interactions

Key output

- Quantification of stream flow depletion by groundwater abstractions.

Methodology:

The project team used two alternative methods to estimate the soil moisture content:

- Hydrograph separation (Herold method) to determine groundwater baseflow (defined as baseflow from a regional aquifer), interflow (defined as baseflow from perched aquifers) and storm runoff on a monthly basis (data base: WR90, observed flow data / stochastic hydrograph). “Back calculated” sub-surface storage is then used to estimate recharge.
- The catchment soil moisture time series S determined by WRSM 2000 is directly used to calculate a time series for recharge. If the “aquifer capacity” (defined as the product of aquifer thickness and storativity) is reached, any excess recharge contributes to interflow. Groundwater storage is depleted by evapotranspiration and groundwater outflow (function of gradient) and abstractions (also reduces baseflow).

The “reverse engineering” of subsurface storage and recharge from hydrograph/baseflow separation, which is then used to assess impacts of groundwater abstractions on baseflow is

obviously highly problematic for impacted and regulated catchments. The hydrograph separation is considered as a subjective method, with the hydrograph being influenced by stream regulations (e.g. dams or weirs), direct abstractions, induced recharge or diversion of water from a river as well as discharges or return flows into a river. In other words, baseflow in a regulated and impacted catchment might only originate to a minor degree from groundwater, as assumed in the separation method.

Recharge is in either case calculated as a fraction of the maximum recharge at maximum soil moisture content. The fraction is determined by the weighted ratio of the difference between actual soil moisture content and the soil moisture below which there is no recharge and the difference between maximum soil moisture storage and the soil moisture content below which there is no recharge.

The potential recharge from the soil moisture contributes to actual aquifer recharge until the “aquifer capacity” is reached, when the “excess recharge” begins to contribute to interflow. Since groundwater abstractions reduce aquifer storage and prolong or prevent the time needed to reach the “aquifer capacity”, it increases recharge and reduces interflow. For catchments without groundwater abstraction interflow is calculated as the difference between baseflow obtained from the hydrograph separation and calculated groundwater baseflow. Further losses may occur by evapotranspiration demand by zones of shallow groundwater, which is calculated as the product of mean annual evaporation, monthly distribution and Acocks veld type crop factor minus precipitation (all data from WR90). Evapotranspiration from this shallow groundwater occurs as long as the demand exceeds precipitation and decreases with decreasing groundwater storage. It reaches zero once the groundwater storage drops below the stream level (static water level), i.e. it does not allow the vegetation to induce recharge from the river. Additional losses from aquifer storage arise as a result of regional groundwater flow, which is calculated using gradients derived from the aquifer storage (of quaternary catchments) and Darcy’s law. Following the consideration of these losses groundwater baseflow is then calculated as a non-linear function of the head difference between groundwater and surface. Fitting parameters allow the modelled rate of groundwater baseflow to be limited or prevented.

Generally groundwater abstraction depletes groundwater storage and groundwater baseflow as a function of the aquifer diffusivity (ratio of transmissivity and storativity), the distance from the river, and time.

Groundwater baseflow and transmission losses due to prevailing head gradients in the aquifer are calculated using what appears to be the Rushton & Tomlinson (1979) nonlinear relationship, while Glover's (1974) method (based on Theis's (1941) solution) is apparently used to quantify effects of groundwater abstractions on stream flow. In the latter method the authors omit the complementary error function of the original solution by introducing two new fit parameters, both without any physical meaning. Beyond this deficiency the quantification suffers from the same conceptual shortcomings as the original solution by Glover/Theis, i.e. the assumptions of fully penetrating wells and rivers and a perfect hydraulic connection between the river and the aquifer.

2.2.4. Groundwater use

Key outputs

- National sectoral (municipal, rural, agriculture/irrigation, agriculture/livestock, industry and mining sectors) and total groundwater use maps (low confidence estimates).
- Comparison of current and previous groundwater use estimates and observations drawn from validation of datasets and sources.

Methodology:

While the project team outlines the Principle Method to achieve mid-term groundwater use estimates of a medium to high confidence, due to scarcity of groundwater use measurements a simplified method (Figure 2-2-3) is necessary to achieve groundwater use estimates of a low confidence. Different available data sources are used to estimate sectoral groundwater use before they are summed up to give the total national groundwater use figure of 1.88623 km³/a.

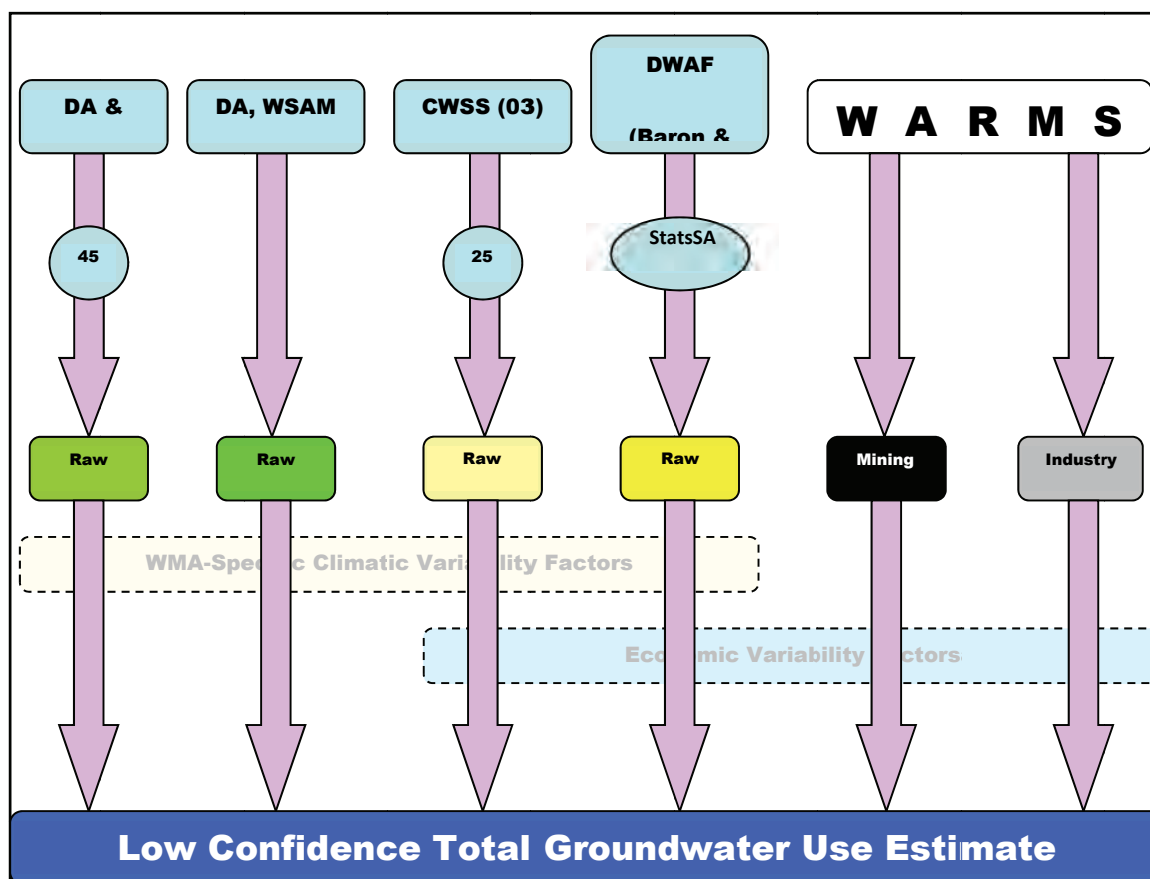


Figure 2-2-3 Schematic representation of applied current method

It is interesting to note that the WARMS database is only used to estimate groundwater use by the mining and industrial sectors.

Agriculture / Irrigation Groundwater Use

The determination of groundwater use for irrigation per quaternary catchment follows essentially the Baron and Seward (2001) method. The spatial distribution of irrigation throughout a catchment is based on the irrigated land cover classes from the National Land Cover dataset (NLC 1998). Total irrigation requirements per catchment are derived from the WSAM (2001) and divided according to the area given in the NLC. The percentage groundwater dependence of irrigation is finally derived from the DA Development Survey, which relates to the Broad Homogenous Agricultural Areas (BHLG), and irrigation requirements are again assigned according to the size of the sub-areas. Finally the sub-areas are aggregated to a quaternary scale. It can be assumed that the applied method generally gives a relatively accurate estimate of the spatial distribution of groundwater use for irrigation within a catchment, but the authors themselves question the reliability of the groundwater reliance information given in the BHLG.

Agriculture / Livestock groundwater Use

As above the method of Baron and Seward (2001) is followed. Livestock units watering requirements per catchment were obtained from the WSAM, multiplied by 45 L per stock unit per day and finally multiplied by percentage groundwater dependence from the BHLG.

Groundwater use by aquaculture as extracted from the WARMS database is included in the final groundwater use map.

Mining Groundwater Use

Groundwater use by the mining sector is extracted from the WARMS database, where mine lease holders have to register their groundwater use as part of their EMPR. Not all mines registered in WARMS are reported as active mines to the Council for Geoscience and vice versa, questioning the accuracy of both databases.

Industrial Groundwater Use

Industrial groundwater use is extracted from the WARMS database and aggregated on a quaternary catchment scale. The authors state that double accounting with Municipal groundwater use is likely and that several regions appear to lag behind with registration (Eastern Cape and KwaZulu-Natal), questioning the confidence of derived groundwater usage data.

Rural Communities Groundwater Use

Rural groundwater usage was aggregated from the updated Water Services communities' data; the database of service delivery to rural areas (includes the 2003 population and water source information). Since the percentage of water supply by groundwater was unknown for a large number of villages in the latter database, the percentage of groundwater reliance for the specific Water Management Area (WMA) was assigned to these villages. If surface and groundwater are conjunctively used for water supply to a village, a reasonable estimate of 50% groundwater reliance was used. Multiplying the percentage reliance on groundwater by the population of the village gives the estimated number of people using groundwater per village. Aggregation of the village information to quaternary catchment level and multiplication by 25 L daily requirement and 365 days per year yields the annual figures. The authors note that the assumption of 25 L per person per day appear to be an overestimate and that seasonal variations in groundwater dependence could not be quantified. It is

obvious that the applied method relies on a number of questionable (e.g. population census and percentage groundwater reliance) and outdated (2003) data, but represents a reasonable low confidence estimate based on available data.

Municipal Groundwater Use

The municipal groundwater use determination is based on scaling the town groundwater use data of Baron and Seward (2001) using a projection of population for June 2004. While the town groundwater use data (per person) rely only on limited data from the Eastern Cape regional DWA office and are therefore regionally biased, the projection of the 2004 population relies on 1996 and 2001 Statistics SA census data and assumed growth factors (taking into account declining fertility, increasing mortality and migration).

The large number of underlying assumptions and projections in the method renders it questionable and of low confidence, but in the absence of a complete WARMS database (with regard to municipal groundwater use) or other accurate data it provides at least a crude figure.

Comparison of current and previous groundwater use estimates

The comparison of these current estimates with previous national, regional (GRIP) and local studies highlights the low confidence groundwater use figures derived using the current method (as stated by the authors themselves). Major deviations between current estimates and validated studies are related to agricultural and rural groundwater use, though similar large deviations can be expected if validated municipal groundwater use data become available.

2.2.5. Groundwater resource classification

Key output

- National classification of (ground-)water resources

Methodology

The classification evaluates the contributions of different surface and groundwater resources within a catchment to the downstream water resources along with the ecological condition of the resources to rate the ecological health of and to develop a sustainability baseline configuration for a catchment.

The management class (MC) of a resource is determined by a six-step process as outlined in Figure 2-2-4. Each MC is associated with a set of economic, social, hydrological and ecological characteristics that relate to the ecological integrity of the resource respectively its capacity to deliver Ecosystem Goods and Service Attributes (EGSAs), and subsequently to the degree of acceptable impacts.

Situation assessment

The situation assessment includes the description of the water resource infrastructure in a catchment (well fields, boreholes etc.), the delineation of groundwater resource units including probable aquifer dependent ecosystems (ADEs, with linked EGSAs) and a hydro-geomorphological classification of surface-groundwater interactions according to Xu *et al.*, (2003).

Aquifer vulnerability with regard to contamination and over-exploitation are assessed using the DRASTIC method (with an unrealistic derivation of hydraulic conductivities of aquifers and impacts of the vadose zone from geological data on a 1:1 million scale) and a simplified score system based on aquifer recharge, storage (both based on GRA2 assessments discussed above) and coefficient of variation of precipitation.

The present status category of the groundwater resource units is then spatially characterised using 3 categories, which assess quantity impacts (abstractive use as a percent of recharge), natural groundwater quality (using, unfortunately, DWA water quality guidelines instead of the legally binding SANS 241) and impacts on groundwater water quality (using the product of the contaminant hazard rating from the National Land Cover data and the DRASTIC vulnerability index). It appears that the proposed groundwater resource assessment is of a similar detailed scale as a comprehensive Reserve determination, with resultant time and financial commitments. It is furthermore not clear why the authors propose a hydro-geomorphic classification of surface groundwater interaction in contradiction to the method proposed for GRA II above. Due to data scarcity (see discussion of the groundwater use assessment above) the delineation and description of socio-economic communities and their sectoral use of water and EGSAs will be one of the most challenging tasks in the national water resource classification process and will need future alignment to the methodology currently developed by Statistics SA. In its current form it appears only applicable in a research environment or on a very limited spatial scale.

Integration

The integration of the different information layers is essentially an overlay of the information gathered during the situation assessment, but unfortunately only new maps of surface groundwater interaction (as percent of baseflow and with a new categorisation) are given. In the absence of available required detailed data, the integration step appears mostly theoretical, especially with regard to “integrated units of analysis”.

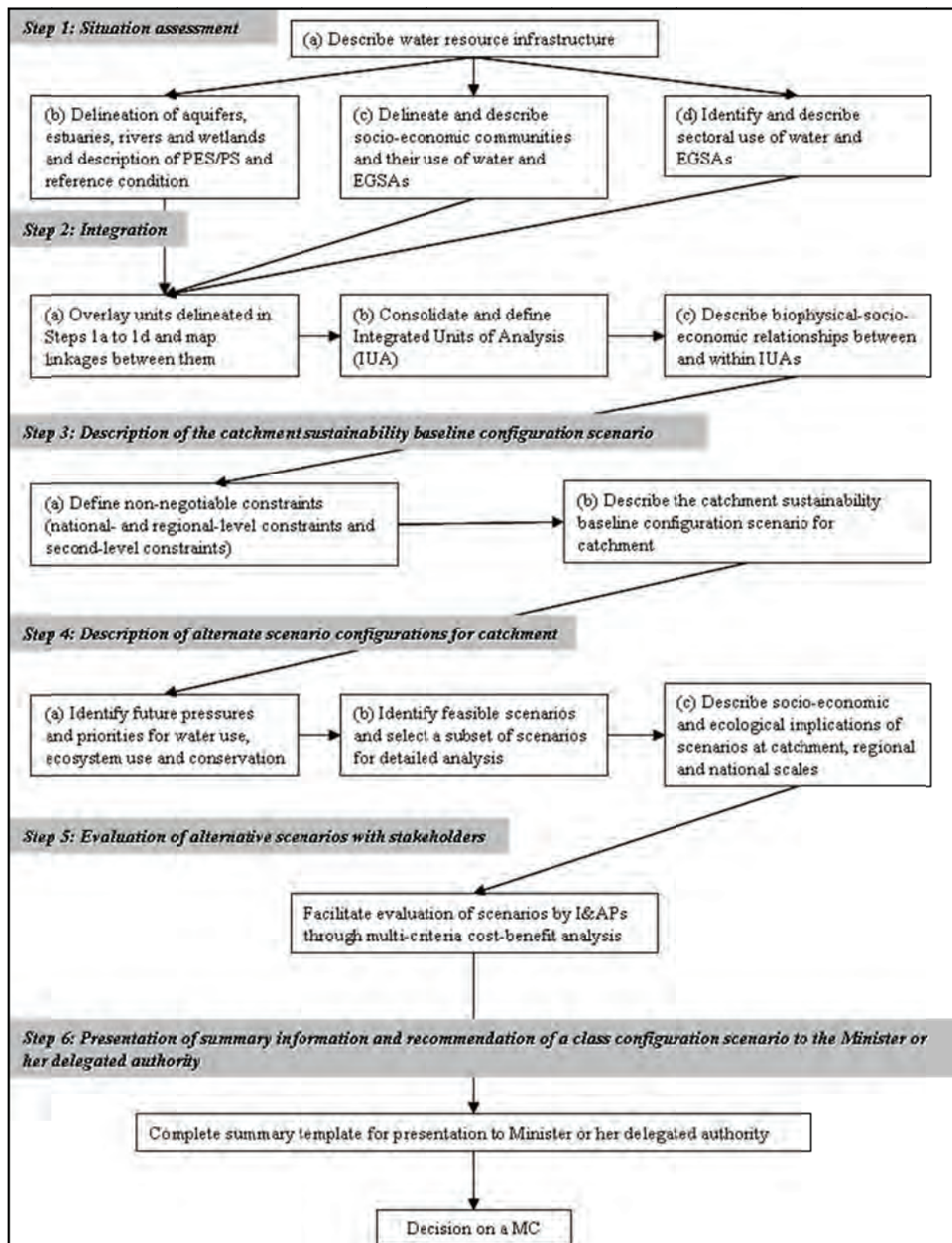


Figure 2-2-4 Outline of national water resource classification process

Description of the catchment sustainability baseline configuration scenario

The catchment sustainability baseline configuration scenario gives the maximum extent to which a catchment's water resources can be used for water supply or waste removal by referring it to a health class D under consideration of the groundwater Reserve non-allocable water or water allocated for strategic use of water supply schemes. The latter constraints are supposed to be also given in a separate definition of non-negotiable constraints. In other words; the description of the catchment sustainability baseline essentially entails a very ambitious reserve determination (above desktop level) for the entire country. For surface but also groundwater resources and aquifer dependent ecosystems the assessment should also consider the downstream dependence on upstream conditions in river systems or aquifers (e.g. passive saltwater intrusion), which would require even more detailed and site specific studies on a national scale. The proposed "permitted changes in groundwater levels from present status level accounting for selected national scale sustainability constraints" assigns a water level change of zero to large parts of the country (especially around population centres like Gauteng or Cape Town), i.e. eliminates any future groundwater use if strictly followed.

Description of alternate scenario configurations for catchment

The description of alternate scenarios for the catchment entails the consideration of future pressures and priorities for water and ecosystem use as well as an evaluation of the scenarios (which is not further specified). However, the socio-economic and ecological impacts of the scenarios should be evaluated with different models, with the yield model being the only model specified by the authors. Furthermore generic rules associated with the proposed management classes are presented but are of no further assistance for a specific case.

While the consideration of alternate scenarios is, for example, best practice within the EIA process, further standardised guidelines with regard to the evaluation of the scenarios should be given. In the current form the proposed evaluation is too generic and vague for general application.

Evaluation of alternative scenarios

The evaluation of alternative scenarios with stakeholders and presentation of summary information and recommendation of a class configuration scenario to the Minister are the

final steps of the resource classification. These follow essentially legal requirements and require no further discussion.

General remarks

The proposed national classification of (ground-) water resources is in most parts scientifically defensible and laudable, but might be too ambitious if the data availability is considered. The classification follows in major parts a comprehensive to intermediate groundwater Reserve determination with the additional consideration of downstream water resources and aquifer dependent ecosystems, and hence proposes a very resource intensive assessment methodology which also has to incorporate different spatial levels.

Deviations from surface-groundwater assessment methods proposed in a separate GRA2 report, as well as the established GRDM methodology, lack integration of methods and this needs to be rectified.

2.2.6. Groundwater planning potential

Key output

- Planning potential maps of Groundwater Quantification, Recharge and Surface Water Interactions, Aquifer Classification and Groundwater Use

Methodology

The planning potential maps for South Africa consolidate essentially the data output of the 4 project reports above into national estimates of the maximum volume of groundwater that can be abstracted per quaternary catchment on a sustainable basis without depleting the aquifer system. The proposed methodology to determine the groundwater harvest potential is a revision of the method by Baron et al. (1998), which considered groundwater storage, recharge and recharge frequency. The revised methodology applies a simple steady-state (with or without abstractions) water balance approach. Transient algorithms are provided to determine the current status of a water resource based on input data from the previous year, though the simplicity of the algorithms does not allow for the transfer of water between neighbouring 1 km x 1 km grid cells and renders the methodology rather inadequate.

The steady-state and transient algorithms are used to derive not only the harvest potential, but a series of different water volumes potentially stored in South African aquifers under varying planning constraints.

Groundwater resource potential

The groundwater resource potential (GRP) is the maximum volume (m³) of groundwater that can be abstracted without ‘mining’ the aquifer system. The average GRP for pristine aquifer conditions (no abstractions) is calculated based on the mean annual recharge, aquifer storage (level 1-level 4 in Figure 2-2-1), a drought index as given by Seymour and Seward (1997) and the mean annual contribution of groundwater to river baseflow. An average dry season GRP is calculated based on Schultze’s (1997) coefficient of variation of mean annual precipitation. Calculation of the current transient GRP follows the average GRP calculation, but considers additionally abstractions and current volumes of stored groundwater.

It is obvious that GRP determination is influenced by the shortcomings in the determination of recharge, aquifer storage and baseflow contribution discussed above. Considering potential error propagation, recharge and baseflow estimates appear to be the most sensitive and critical parameters.

Groundwater exploitation potential

The exploitation potential of an aquifer is the portion of the groundwater resource potential, which can be practically abstracted. The exploitation factor used for downscaling is simply derived from Vegter’s (1995) exploitability – accessibility dataset (average probability of drilling a borehole with a yield > 1.0 and > 2.0 L/s) for fractured aquifers or set to 0.7 for all primary aquifers. While a single value for all primary aquifers appears unrealistic (e.g. alluvial versus weathered aquifers), Vegter (1995) assumed an adequate number of randomly spaced boreholes in the NGDB to derive the exploitation factor for fractured aquifers. However, NGDB boreholes target typically any form of discontinuity in an aquifer and cannot therefore be seen as randomly distributed. The assumption of randomness for systematically drilled boreholes should theoretically result in an overestimation of the groundwater exploitation potential.

The groundwater exploitation potential is further scaled down to a potable groundwater exploitation potential with a potability factor, which considers TDS, NO₃, K, Na, SO₄ and Ca concentrations with regard to DWA's domestic water quality guidelines (marginal quality as lowest acceptable class). The applied potability factor neglects very important microbiological parameters (e.g. E. coli), important trace element concentrations (e.g. F), which can render groundwater in many rural areas unfit for human consumption. Also neglected are cyclic variations in water quality during droughts (despite being used for such calculations).

While the factor that is currently used is therefore too optimistic, the application thereof is generally questionable as it neglects potential water treatment – much groundwater rejected as being of poor quality could be used given suitable treatment. If a similar approach was used for surface water this would probably designate almost all currently used surface water resources as non potable.

Another subset of the groundwater exploitation potential is defined as the utilisable groundwater exploitation potential, which considers the ecological Reserve in the definition of aquifer storage (level 6 in Figure 2-2-1). While the authors claim that the basic human need component of the reserve is considered in the annual abstractions (for transient calculations), their approach neglects a potentially large part of the reserve which has to be catered for despite the population being supplied already. Again, the utilisable groundwater exploitation potential will be an overestimate and does not help in planning. It should be scrapped and the Reserve (both components) incorporated into the groundwater resource potential.

2.3. Discussion of Groundwater Resource Assessment methods

International Groundwater Assessment Methods

The three international examples on groundwater resource assessment are summarised in Table 2-3-1 and compared to South Africa.

Table 2-3-1 Comparison of the three international examples with South Africa

Case study country	Land area (km ² x 1000)	Pop. (M)	Groundwater dependence (domestic supply) estd.	Digital groundwater data availability	Notes
United Kingdom	244	61	30%	Descriptions and data normally for sale. Incomplete digital availability.	High groundwater dependence in populated SE of country. Moves towards numerical modelling (hydrodynamic methods) for regional assessment.
Australia	7 600	21	15%	Descriptions and data are free to download, much available digitally. Moves towards common digital data standards.	Higher priority now given to groundwater. Efforts to limit abstractions. "Water accounting" approach developed for groundwater assessment.
California	414	36	30%	Descriptions and data are free to download. Good digital coverage.	Groundwater very important for domestic supply and agriculture. Local or county management. Assessment appears to be done on an ad-hoc basis, preferably based on numerical model results.
South Africa	1 200	49	60%	Some descriptions free to download, state data currently available free on request (NGDB) – plans for web access to groundwater data.	Process of devolution of groundwater management to water management areas from national level. Serious efforts being made to increase groundwater resource assessment. Available resource potential is between 7.5 km ³ /a and 47.7 km ³ /a depending on what factors taken into account.

The concept of “safe yield” is often used when planning groundwater abstractions – i.e. the volume of water that can be extracted without undesirable consequences such as loss of baseflow to rivers, ingress of saline water or excessive lowering of the water table. Undesirable consequences need to be defined first of course, and the definition of these can change with time. There are several groups of methodologies available for assessing “safe yield” (UNESCO, 2004):

- Hydrodynamic methods – these involve analytical and numerical calculations, and include numerical modelling. These methods require sufficient data, such as knowledge of boundary conditions and initial water levels, to be successful.
- Hydraulic methods – based on empirical observations and measurements of pumped boreholes, surface water features, etc. These methods have the advantage in that they constitute real observations of the system, but it is difficult to extrapolate test data beyond what is observed – which is a common requirement.
- Method of hydrogeological analogy – this refers to the transfer of knowledge from an aquifer system which is well known, to another where information is scarcer. Success depends on the similarity of the two systems, and the degree of accuracy required.
- Method of expert assessments – this is basically the ability of an experienced groundwater professional to give estimates for a system where data is scarce. Success will depend on the skill (and any unconscious bias) of the professional, and the complexity of the system.

In all three case studies discussed above, hydrodynamic methods appear to be favoured for all but the simplest systems. Other methods are also used however, if only to provide first approximations and in the construction of conceptual models. On-going monitoring of water levels, surface water flows, etc. will normally be carried out, and management decisions made according to the flow of information. Hydrodynamic methods need good data, at sufficient density, and therefore can be expensive to implement. Decisions about groundwater development are often taken in the context of growing pressure on the resource, and better knowledge of the consequences of over-abstraction. It appears from the case studies that there is an increasing concern about groundwater resource depletion or degradation, and a growing demand for greater certainty in hydrogeological prediction

and management. This is likely to imply a further shift towards numerical modeling of local and regional aquifers, with all of the attendant data requirements and conceptual modeling that this implies. The following conclusions can be drawn from the case studies, with relevance to our situation here in South Africa:

- Increasing moves towards better and more accurate determination of groundwater resources is driven by increasing competition for water resources, a better appreciation of the ecological role of groundwater, and an expanding awareness of possible water supply disruption or variation due to climate change.
- Awareness of the importance of groundwater outside of a relatively small community of experts needs to grow, partly to help ensure funding and political/institutional support for assessment programs.
- There does not appear to be a single methodology for groundwater assessment, although numerical modelling of one sort or another is often seen as desirable. The Australian “accounting” system may be the most ambitious and technically demanding. Integration with surface water assessment is necessary.
- Assessment methodologies should ideally be able to predict what might occur in future, as well as providing a snapshot of the present. This is necessary for planning and the mitigation of risk.
- Institutional cooperation is vital in groundwater assessment, since almost all methodologies are very data intensive. This requires strong coordination at the national level.
- Appropriate legislation is likely to greatly facilitate national and regional assessments, with the institutional collaboration and coordination which is required. The large increase in Australian efforts to quantify and manage groundwater in recent years is a result of specific legislation. In all the three cases reviewed there is a clear link between the planning and legal framework and groundwater assessment requirements.

The Australian case study in particular presents a lot of lessons for South Africa on the strategic nature of groundwater. It demonstrates that a high level of financial investment into groundwater management is supported by robust institutional arrangements at national to state then regional and local levels.

It is necessary to reach a better quantitative understanding of South Africa's groundwater resources, including that portion which can be used sustainably, for reasons that include:

- More effective implementation of national environmental legislation, including the Water Act.
- Better water resources planning and more efficient service delivery
- Greater recognition of groundwater as a reliable and sustainable national water resource – groundwater needs to take its “rightful place” along with surface water as a key national resource, in the eyes of decision makers as well as scientists.
- The facilitation of new developments in the most streamlined and cost-effective ways possible (industrial, agricultural and others).

Earlier efforts in South Africa, up to and including GRA2, have relied mainly on existing data (such as held in the NGDB), applied in various ways and with a variety of quality control techniques and modes of filling data gaps. Better quantification of the nation's water resources will inevitably rely on more data, as well as better data assessment techniques.

International examples indicate that numerical techniques (hydrodynamic methods) are currently most favoured for estimating local groundwater availability, including the impact of abstractions on adjacent groundwater systems and users. These are often done on an ad-hoc basis, with more attention and resources being given to those areas where pressures are greatest on the resource, and comparatively little attention paid to less pressured areas. There is no “one size fits all” groundwater assessment methodology, and methodologies appear to have developed organically against a changing background of data, human resources, legal requirements, growing demand, and water restrictions. Indeed, changes in groundwater assessment methodologies over the years in the international examples suggest that fixation on a single methodology for this country may not be appropriate. In South Africa, with our historical bias towards surface water, there may be a subconscious desire to choose a single groundwater assessment methodology (such as the surface water system currently in use) and adapt data collection, staff deployment and other policies towards that methodology. This should be resisted.

In Australia it is recognised that the different time scales or variations inherent in water resource assessment (e.g. between groundwater and surface water, or due to unpredictable drought cycles) demands some kind of annual accounting system which begins each year

with a “statement” of the resource carried over from the previous year. It is likely that South Africa would benefit by moving towards such a “dynamic” system – although it is acknowledged that this would be very data intensive. The case of Australia also demonstrates the role of robust and well-funded groundwater institutions in assessing groundwater and driving groundwater policy.

All of the international examples show a move towards more data-intensive groundwater assessment methodologies, and increasing data density and availability in South Africa should be a core part of GRA3. This need not mean collecting new data only – much data that is already generated in South Africa (e.g. by private consultants and drillers) is currently difficult to access. It is likely that a process to centralize “private” data would be a very cost effective and rapid way of expanding national groundwater data archives in South Africa.

Finally, the coordination of databases, implementation of legislation and policy, initial assessment of promising regional groundwater resources and many other functions all depend (to an extent) on a capable and well-resourced Department of Water Affairs, with a strong groundwater capacity in particular. At present concerns exist as to the capacity of the Department in this regard, and support needs to be given to the various initiatives aimed at remedying the situation.

National Groundwater Assessment Methods

The original GRA2 process was an ambitious and commendable endeavour to quantify South Africa’s groundwater resources. Due to a shortage of reliable groundwater data, and an uneven distribution of data across the country, some of the methods used in the GRA2 process are questionable. GRA2 emphasises the urgent need for more data on South African groundwater, and better quality data.

Partly due to the lack of data on which the GRA2 process is based, and the consequent potential to misrepresent certain catchments, the GRA2 datasets are not presently freely available from DWA. It appears that the GRA2 data is also not routinely used by water resource planners, no doubt partly due to difficulties in obtaining the data as well as issues with accuracy and reliability.

GRA2 has lessons for any future Groundwater Resource Assessment process – mainly that some caution needs to be exercised when extrapolating from only a small amount of

data. There is also a need for any outputs of a GRA process to be made widely available, so that planners may use them and so that other scientists may review the methodologies involved.

The following issues have stopped the GRA2 groundwater data being used more frequently in South Africa:

- The perceived lack of accuracy in some catchments due to the lack of primary data coverage in some areas in South Africa
- The perceived lack of accuracy in some catchments due to deficiencies in the algorithms used, particularly in how surface water is incorporated into GRA2

The first issue can be addressed by better groundwater data collection, and by incorporating more of the existing data into the GRA2 calculations. In this way, the accuracy of the GRA2 predictions can be slowly expected to increase.

The second issue is more complicated. The critical review of the GRA2 methodology, described per section, highlights the main shortcomings of each methodology applied. However, this perceived lack of accuracy is the greatest for the quantification of surface water-groundwater interaction. In light of this, the main focus has been assigned to the review and improvement of the GRA2 surface water-groundwater interaction methodology.

Groundwater-surface water interaction is handled in different ways. There are currently two models in use in South Africa, which attempts to model groundwater-surface water interaction at a quaternary scale. The assessment methodology for the quantification of groundwater-surface water interaction in GRA2 is given in DWAF (2006a) and referred to herein as the Sami model. The WRSM2000 programme currently used by surface water planners in the Department of Water Affairs includes this groundwater-surface water interaction methodology. However, DWA (2009) highlights that this methodology has been reviewed by hydrogeologists and its applicability has been brought into question (Dennis, 2005; Sami and Witthüser, 2006). An additional method which is used to estimate the groundwater contribution to baseflow is the Hughes model, a modified version of the Pitman model developed by Hughes (2004) and based in the SPATSIM system. Both the Sami and Hughes model methodology will be described in detail.

Surface water-groundwater Interaction methodologies

A review of surface water-groundwater interaction methods was conducted which briefly describes available methods for surface water-groundwater interaction investigations, presented on an international and national perspective. The methods for quantifying the groundwater contribution to streamflow in current South African Groundwater Resource Assessments are the Sami and Hughes models, which are described in detail.

2.4. Review of Selected International SW-GW Interaction Methodologies

2.4.1. Australia

Guidelines

The Australian government initiated the Water for the Future – Water smart Australian programme in order to aid in integrated water management and the quantification of double accounting in water resource assessments. The project's objectives were to develop a practical and moderately priced methodology for assessing the different connections between groundwater and river systems. The project compared estimates of surface water-groundwater interaction using flow differences, hydraulic gradient analysis, hydrograph baseflow separation and geochemical comparisons in ten representative catchments. A method of quantifying the groundwater-surface water interaction is then recommended in the 2012 final report based on the predefined level of importance of the water resources within a catchment. For low importance groundwater and surface water systems, a groundwater balance method is recommended. For catchments with moderate importance groundwater and surface water resources, baseflow separations using the Tracer method and Lyne and Hollick Filter method are recommended. For high importance water resource systems, baseflow separations using the Tracer and Lyne and Hollick Filter methods complimented with run of river sampling methods would be the minimum recommendation. However, it is important to note that the higher the accuracy of a surface water-groundwater interaction assessment is, the higher the cost will be. The indicative costs per catchment for a poor to moderate, moderate to high, moderate to high (instrumentation), high to excellent and high to excellent (instrumentation) are \$10 000, \$20 000, \$85 000, \$150 000, \$500 000 respectively. It was concluded that the chemical hydrograph separation method (Tracer method) is sensitive to the groundwater and surface water end members applied but the method has the best potential for providing catchment

scale estimates of groundwater inflow to a river over time (Australian Government, 2012a and 2012b).

The National Water Initiative (NWI) is the Council of Australian Governments (COAG) principal water policy agreement, Australia's enduring blueprint for water reform. One of the objectives of the NWI is the conjunctive management of surface water and groundwater and in light of this the Groundwater Project of the eWater Cooperative Research Centre is developing modeling tools which will incorporate a groundwater-surface water interaction capability for the new RiverManager© and WaterCast© products (Australian Government, 2004). Rassam and Werner (2008), in a comprehensive review of surface water-groundwater interaction modeling approaches and their applicability to Australia, found that GW-SW interactions are poorly handled in existing surface water and groundwater models. In river models the interaction volume is simply modeled as a loss term whereas in groundwater models the river is simply modeled as a boundary condition. In more sophisticated models, able to account for the interaction more explicitly, more data and a higher degree of modeling expertise is usually required. In light of these conclusions, Rassam and Werner (2008) suggest that groundwater-surface water interaction processes that are most relevant to the Australian landscape be identified in order to choose a modeling tool which will incorporate an appropriate balance between surface water and groundwater processes. It follows that this balance can only be achieved through the use of custom-built, special-purpose models developed to answer particular management questions. Jolly, and others (2008) summarise the research done by the eWater Cooperative Centre and describe three simplified modelling approaches that are currently in development, namely a reach scale model, 'Groundwater-Surface Water Link', which operates as a groundwater link to river models and accounts for interactions at the river-reach scale; a sub-reach scale model, 'Floodplain Processes', which dynamically models bank storage, evapotranspiration, and floodplain inundation enabling a more refined modelling of groundwater-surface water interactions, and can be linked to ecological response models; and a catchment scale model that estimates the surface and sub-surface flow components to streams (Jolly *et al.*, 2008).

Integrated Groundwater-Surface Water Models

A review of groundwater- surface water interaction modelling in arid/semi-arid floodplains was performed as part of the Australian Hydrological Modelling Initiative-Groundwater Surface Interaction Tool Project. Jolly and Rassam (2009) state significant advances have been made over the last fifteen years in the modelling of groundwater-surface water interactions, progressing from relatively simple 1D and 2D analytical and empirical approaches to highly refined 3D spatially distributed integrated models. However, these large progressive steps in modelling power have been accompanied with the scaling down of routine hydrological and hydrogeological monitoring networks (Silberstein, 2003). The advances within the modelling discipline will be limited by computing power or solution methods, but rather by the availability of suitable data to parameterize, calibrate and validate the current models. It is thus seen that simple analytical options are still a useful option in data scarce situations. Jolly and Rassam conclude that the use of sophisticated numerical models should be limited to data-rich situations, where calibration and validation may be performed (Jolly and Rassam, 2009).

The currently available fully integrated surface water-groundwater flow and transport models are a reflection of the increasing complexity of the existing hydrological model. Partington and others (2011) state accurate quantification of streamflow generation mechanisms are still not possible within these advanced models, in that the groundwater component of baseflow at a particular point along the stream cannot be specified. Partington *et al.* (2011) developed a hydraulic mixing-cell (HMC) method, as part of the Linkage Scheme supported by the Australian Research Council, which uses only the flow solution from fully integrated surface water-groundwater flow models to accurately determine the groundwater component of streamflow. The method is based on the modified mixing cell (Campana and Simpson, 1984). The model only requires hydraulic information which eliminates the need for the simulation of tracer transport, which could be advantageous if tracer concentration data was not available. It can be seen that the popular trend in quantifying the groundwater component of streamflow has become integration into groundwater flow numerical models. These methods as discussed by Harington *et al.* (1988) and Partington *et al.* (2010) are adequate methods for determining the groundwater component of streamflow; however the data required to set up the models are often not widely available, especially in a South African context.

Hydrograph Separation Techniques

The unit hydrograph separation method distinguishes between streamflow originating from surface runoff and groundwater. The method is popular as it requires only readily available streamflow data (Australian Government, 2012b). The widespread method of estimating fluxes to and from groundwater aquifers using streamflow data traditionally starts with using the measured rainfall at the surface and then estimate infiltration, redistribution, evapotranspiration, percolation of residual water through the unsaturated zone and discharge of groundwater to streams, respectively (Wittenberg and Sivapalan, 1999). Wittenberg and Sivapalan (1999) refer to this approach as reductionist or “bottom-up” approaches in a report developed through the Centre for Water Research, University of Western Australia. However, these approaches are not suitable for arid or semi-arid conditions where only a small fraction of precipitation reaches the groundwater because the relative errors in the measurement of precipitation can exceed both groundwater recharge and discharge. Wittenberg and Sivapalan (1999) thus suggest a holistic or “top-down” approach which is based on the analysis of measured streamflow. Observed total streamflow is separated into quick flow and baseflow by following previous applications of this approach (Chapman, 1997; Chapman and Maxwell, 1996; Fröhlich et al., 1994; Nathan and McMahon, 1990) with the exception that a nonlinear reservoir algorithm is used. The results from the application of this method compare reasonably well to response functions estimated by other authors based on theoretical, bottom-up approaches and lysimeter measurements (Wittenberg and Sivapalan, 1999).

Environmental Tracer Methods

Environmental tracer methods have been used to quantify the groundwater discharge to rivers for the past few decades as they offer advantages over physically-based methods such as seepage meters and river flow gauging in that they can potentially provide more accurate information on the spatial distribution of groundwater inflows with less costly resources. Cook, Favreau, Dighton and Tickell (2003) make use of ^{222}Rn , CFC-11, CFC-12, major ions and temperature of river water and springs are used to quantify rates of groundwater discharge to a tropical lowland river in Northern Australia. The method makes use of a numerical model which simulates concentrations of a number of different tracers allowing most parameters to be constrained and the method was found to produce more accurate estimates of groundwater inflow to the river than the simple mass balance method

conventionally used. The method concludes that CFC-11 and CFC-12 are suitable to infer rates of groundwater inflow to streams, where ^{222}Rn and major ion tracers are traditionally used.

2.4.2. Canada

Isotopes

In the report Progress in isotope tracer hydrology in Canada, Gibson and others (2005) argue that Canadian researchers have played an important role in the development and refinement of isotope hydrology techniques. Fritz, Cherry, Weyer and Sklash (1976) define the pre-event and event water components of watershed runoff in one of the earliest applications of stable isotopes, with multiple subsequent applications in various physiographic regions of Canada. Cey, Rudolph, Parkin and Aravena (1998) quantifying groundwater discharge to a small perennial stream in southern Ontario by performing hydrograph separations using $\delta^{18}\text{O}$ and electrical conductivity on two large rainfall events with different antecedent moisture conditions in the catchment. Both events indicated that pre-event water was dominated by groundwater, with a 64-80% contribution towards discharge added by pre-event water. The study also investigates three other techniques to estimate the contribution of groundwater to the stream, namely streamflow measurements using the velocity-area technique, mini-piezometers measuring hydrometric measurements and seepage meters directly measuring the water flux into or out of the stream. Cey *et al.* (1998) conclude that large-scale measurements provided a better estimate of groundwater discharge than point-scale measurements, due to the heterogeneous nature of the site. Techniques which can incorporate spatial averaging on a relatively small scale are recommended for proposed new approaches.

Site Specific Scale

The Natural Science and Engineering Research Council of Canada funded a study to facilitate a better understanding of groundwater-surface water interaction between a fractured rock aquifer and a bedrock stream. Oxtobee and Novakowski (2002) made use of air-photo interpretation, detailed stream surveys, electrical conductivity, temperature and isotopic surveys, mixing calculations and point measurements from mini-piezometers, seepage meters and weirs to identify and quantify the interaction between the creek and local aquifer. Groundwater and surface water could easily be distinguished within the study area

on the basis of differences in electrical conductivity, temperature and isotopic signatures. Oxtobee and Novakoski (2002) conclude that groundwater discharge in fractured bedrock stream environments mainly occur as discrete point sources related to open fractures which differs from the diffuse, continuous seepage observed in alluvial aquifer environments. Techniques which conventionally are applied to studies in porous media, namely electrical conductivity, temperature and hydraulic head surveys, were found to produce reasonable estimates of groundwater discharge to a stream in a fractured bedrock situation.

2.4.3. United Kingdom (UK)

Guidelines

The Environment Agency is an executive, non-departmental public body which aims to protect and improve the environment in England and Wales (House of Commons, 2006). The Agency has a legislative duty to manage the sustainable development of groundwater resources. Conceptual and numerical model development is the main objective of the Agency in order to efficiently meet their regulatory responsibilities. The Agency currently invests £3 million per year on groundwater resource assessments and modelling (Environment Agency, n.d (a)). The need for a regional groundwater conceptual or numerical model has been identified for selected areas in England and Wales, mainly in major aquifers. Groundwater resource assessment and modelling is an iterative process beginning with the development of a conceptual model which is used as a basis for testing ideas and to identify data and knowledge gaps. The conceptual model is then refined when new data or understanding of the area improves. If there is sufficient data and a need, the groundwater modelling process can be taken further by developing a numerical model which is a computer-based representation of the conceptual model. The numerical model is then used to make predictions which aid in making decisions regarding the management of groundwater resources (Environment Agency, n.d (a)). The Lowland Catchment Research (LOCAR) and Catchment Hydrology and Sustainable Management (CHASM) programmes have resulted in considerable field-based activity investigating groundwater-surface water interactions, which form sixteen field test sites (Environment Agency, 2005a).

Resources Assessment Methodology (RAM) and Impact of Groundwater Abstractions on River Flows (IGARF) are two of the tools utilized by the Environment Agency to support their management and protection of groundwater. RAM sets the resource availability status for

river reaches and associated groundwater. IGARF evaluates the effects of groundwater abstraction on surface water flows (Environment Agency, n.d (b)).

Site Specific Scale

In order to better characterise the hyporheic zone, the measurement of groundwater flow on a small- scale is vital. High-resolution methods for the estimation of groundwater-surface water interaction are described and tested in a report presented by the Environment Agency. Borehole-based, buried flow meters, direct measurement of the flux at the surface water-groundwater interface, geophysical and thermal techniques are investigated. The report concludes that none of the devices are ideal for all situations and thus a combination of the methods would provide the best results (Environment Agency, 2005b).

The Lambourn River in the United Kingdom is used as a case study for a detailed surface water-groundwater interaction investigation. Allen *et al.* (2010) states a variety of techniques are available to identify and quantify groundwater-surface water interaction processes at a site scale, i.e. hydrochemistry (Tetzlaff and Soulsby 2008; Mencio and Mas-Pla 2008), fluorescence properties of organic matter (Lapworth et al. 2009), physical parameters (Keery et al., 2007; Schmidt et al., 2007; McGlynn et al., 1999) and process-oriented modelling approaches (Krause et al., 2007). However, each method has its own advantages and limitations which complicate the selection of only one particular method for a specific site. The conjunctive use of more than one method would increase the overall confidence and understanding of the complex hydrological processes taking place at this scale. An extensive network of boreholes, piezometers and water quality sampling sites were utilized in order to apply a combination of geological, hydraulic and hydrochemical approaches to investigating the surface water-groundwater interactions. These multiple methods have facilitated the development of a comprehensive conceptual model of the study area which according to the authors is clear in certain respects but more ambiguous in others (Allen *et al.*, 2010). This ambiguity in spite of sufficient data illustrates some of the problems faced when considering surface water-groundwater interactions in a hydraulically complex, site scale system. The study has shown that even a seemingly simple groundwater-surface water system can be hydrologically complex at the site scale. Due to chemically similar groundwater in different components of the system and the heterogeneity of the alluvial aquifer, the hydraulic relationship between the river, the

alluvial aquifer and underlying aquifer are still only partially understood in spite of the extensive available physical and geological data. Allen *et al.* (2010) mention recent studies which have emphasized the complexity of groundwater-surface water exchange processes (Krause *et al.*, 2007; Grapes *et al.*, 2005; Griffiths *et al.*, 2006). The realisation of this complexity has implications in how these exchanges are investigated and managed. Methodologies need to be developed which can encompass detailed site scale knowledge into decisions applied at the larger catchment scale and monitoring and sampling extents would need to be carefully considered to ensure an appropriate density.

2.4.4. United States of America (USA)

Guidelines

Rosenberry and LaBaugh (2008) compiled a comprehensive overview of available techniques and methods to describe and quantify groundwater-surface water interaction. The reports objectives are to create an awareness of the scope of the approaches available as well as to serve as a guide to surface water-groundwater interaction studies for water-resource investigators. The report covers scale appropriate methods and an in-depth description of most methods. LaBaugh and Rosenberry (2008) suggest watershed-scale modelling, groundwater flow modelling, flow-net analysis or dye and geochemical tracer tests for catchment scale studies, defined as larger than a kilometre or more in length or width. The measurement of streamflow at two places over an intermediate scale (ten to hundreds of meters) which enables the calculation of gains and losses in that river reach is recommended for the identification of interaction zones. Tools such as seepage meters, mini-piezometers and buried temperature probes are more appropriate for local, small scale studies.

MODFLOW

The U.S. Geological Survey's MODFLOW three-dimensional, groundwater flow model was released as a versatile simulator of groundwater flow within an aquifer almost thirty years ago and the programme is still in widespread use (Swain, 1994). The modular design of the MODFLOW model allows for the addition of new packages to both expand the capacity of the model and improve the accuracy. A number of additional packages have been developed to allow for the simulation of the effect of artificial recharge to the interaction between river water and groundwater. Swain (1994) describes three add-on packages

designed to simulate the interaction of groundwater-surface water, namely the channel stage River package, the flow-routing Stream package and the unsteady open-channel flow model BRANCH using the MODBRANCH coupling programme. The River package assumes a constant river stage and computes the surface water-groundwater interaction as leakage across a confining riverbed but the flow in the river is not simulated and thus acts as an infinite source or sink. A programme developed by Swain (1994) allows direct-flow connections to be simulated between MODFLOW and the three available groundwater-surface water interaction packages. The programme thus facilitates the modelling of different sections of a river using the various interaction packages simultaneously within MODFLOW.

MODFLOW solves the three-dimensional groundwater flow equation by means of finite difference approximations (McDonald and Harbaugh, 1988). The aquifer is divided into cells which have dimensions x , y and z and the aquifer properties within are assumed uniform as this is required for the finite difference equations. The head is assigned and calculated at the centre of each cell by iterating the finite difference equations for all nodes until the maximum head change in any cell within the previous and current iteration is less than the user-specified value and then the process is repeated. However, the appropriate time step for this iteration will vary for surface water and groundwater because the response time of surface water systems is usually faster than groundwater systems. Jobson and Harbaugh (1999) couple the surface water flow model DAFLOW to the modular groundwater flow model MODFLOW to improve the model's ability to simulate surface water-groundwater interaction. DAFLOW has been structured to include subroutines which allow multiple time steps to be run iteratively within one MODFLOW time step. The subroutines within DAFLOW can compensate for the difference in response times between surface water and groundwater.

Jobson and Harbaugh (1999) mention but do not compare DAFLOW to the other available groundwater-surface water interaction packages available for MODFLOW. It is concluded that DAFLOW provides a highly stable solution scheme which is easy to run and requires a minimum of field data and calibration.

The increased awareness of the interconnection between surface water and groundwater lead to the development of a new Stream-Routing (SFR1) package for the MODFLOW

groundwater flow model (Prudic, Konikow and Banta, 2004). The SFR1 package replaces the older Stream (STR1) package written for earlier versions of MODFLOW. The new SFR1 package has several improvements from the previous STR1 package with the main difference between the two packages being that the river depth is computed at the midpoint of the river reach in SFR1 rather than at the beginning of the reach as in STR1. The main limitation of the new SFR1 package is that it is not suited for modelling the transient exchange of water between the river water and groundwater on a short-term time scale of days or minutes. This is due to the assumption that streamflow and associated dissolved solutes are routed between the stream reaches only on the basis of continuity. The package also makes the assumption that solutes are completely mixed within each stream reach which limits the applicability for large rivers and simulating short-term effects on surface water-groundwater interaction (Prudic *et al.*, 2004).

Hydraulically disconnected streams found in semi-arid regions are becoming increasingly prevalent as groundwater abstractions lower groundwater levels in valley aquifers beneath rivers (Niswonger and Prudic, 2005). In order to investigate the connection of surface water and groundwater through an unsaturated zone another streamflow-routing (SFR2) package for MODFLOW was developed which has the capability to simulate unsaturated flow beneath rivers. A kinematic wave approximation is used to simulate unsaturated flow beneath rivers which assumes that the downward flow beneath the river is purely due to the force of gravity (Niswonger and Prudic, 2005). The SFR2 package still has all of the SFR1 package capabilities and has a simulation for a time delay in recharge, while maintaining the applicability of MODFLOW-2000 to catchment-scale situations. The package has a number of associated assumptions which can limit the applicability of the model, but in two test simulations the magnitude and downward progression of the wetting front were in agreement with results from the U.S. Geological Survey's Variably Saturated Two-Dimensional Flow Transport (VS2DT) model (Niswonger and Prudic, 2005).

GSFLOW

The U.S. Geological Survey (USGS) Groundwater and Surface water Flow model (GSFLOW) is an integrated hydrological model developed to simulate coupled groundwater and surface water resources. The model is based on the integration of the USGS Precipitation-Runoff Modelling System (PRMS) and the USGS groundwater flow model MODFLOW. The coupled

approach towards integrated hydrologic modelling used in GSFLOW partitions the surface and subsurface systems into separate regions and the governing equations which describe flow in each of these regions are then integrated or coupled using iterative solution methods (Markstrom, Niswonger, Regan, Prudic and Barlow, 2008). GSFLOW thus provides a robust modelling system for simulating flow through the entire hydrological cycle and can be used to evaluate the effects of land-use changes, climate variability, groundwater abstraction on surface and subsurface flow and many more. A numerical algorithm is made use of to simulate the most important processes affecting surface water and groundwater flow systems. The interaction between surface water and groundwater can be simulated in catchments ranging from a few square kilometres to several thousand square kilometres and allow for simulation periods that range from months to decades. However, the model has a large number of inherent assumptions. There are assumptions and limitations associated with each of the modules and packages contained within GSFLOW. GSFLOW is a non-linear model and is thus limited by the possibility of non-convergence among any or all coupled dependent variables, or due to inappropriate input data or parameters. The GSFLOW model has a number of limitations in terms of the discretization of time and space, in that the model has a computational time step of one day resulting in all flow and storage data are required in mean daily format. The small time step might also lead to errors as flow near the land surface tends to occur faster than flows in the subsurface. The size of finite difference cells are constrained by the relative width of cells compared to the width of the river, where a large cell relative to a stream that flows over a cell will result in model errors and misrepresentation of groundwater-surface water interaction or where a cell width is equal to or less than the width of a river the model may not converge (Markstrom *et al.*, 2008). There are also addition assumptions associated with the canopy zone, land-surface precipitation and temperature, the soil zone, streams, lakes, groundwater and the unsaturated zone functions (Markstrom *et al.*, 2008).

SWAT

The Soil and Water Assessment Tool (SWAT) is a conceptual, continuous time model developed to assess water supplies and nonpoint source pollution on watersheds and large river basins using readily available data (Arnold, Srinivasan, Muttiah and Williams, 1998). Daily precipitation is the main input to the model and the groundwater flow is computed in the model by creating shallow aquifer storage, namely a water balance for the shallow

aquifer. The return flow from the shallow aquifer is then calculated using an empirical equation described by Arnold *et al.* (1993) as well as a relationship for the water table height in response to recharge. The model was however originally developed to predict the impacts of agriculture management on erosion and sedimentation rates. The model has sub-basin components which are categorised into hydrology, weather, sedimentation, soil temperature, crop growth, nutrients, pesticides and agricultural management (Arnold *et al.*, 1998). Arnold, Muttiah, Srinivasan and Allen (1999) conducted a comparison study of the estimates of regional recharge and discharge produced by the SWAT model and a combination of hydrograph techniques, namely a digital recursive filter used to separate baseflow from total streamflow and a modified hydrograph recession curve displacement method to estimate recharge to the shallow groundwater system. The baseflow estimates from both methods were compared to measured baseflow for three watersheds in Illinois and to results from another separation technique. The comparison between the models and to the measured baseflow values indicated that both methods followed the same regional trends (Arnold *et al.*, 1999). However, the model seems to estimate total baseflow to the river instead of the groundwater component of baseflow.

Hydrograph Separation Techniques

A hydrograph separation technique described by Moore (1992) was applied to Boulder Creek, USA using extensive groundwater elevation and streamflow data in order to determine the groundwater discharge component during storm conditions (Hannula, Esposito, Chermak, Runnells, Keith and Hall (2002). The estimates of groundwater discharge which resulted from the study were found to be reasonable based on the facts that the estimates did not exceed total flow in the stream, the estimates followed both storm and seasonal trends and the parameters entered into the calculations are physically based (Hannula *et al.*, 2002).

Moore (1991) describes a simple method for hydrograph analysis that is based on relationships of storage depletion to aquifer properties and flow rates during water-level and streamflow recessions which can be utilised in a fractured rock terrane. The method was applied in the headwaters of the Melton Branch basin, USA where traditional methods assuming a constant transmissivity do not produce reasonable results for groundwater discharge. Analysis of the streamflow hydrograph and water level hydrographs during the

non-growing season of the area indicates that storm runoff constitutes most of the stream flow after the end of overland runoff, but that discharge from groundwater dominates streamflow again after eight days of recession (Moore, 1991). There is however a contrasting opinion on hydrograph separation techniques where Halford and Mayer (2000) argue, from an analysis of 13 sites in the USA, that these methods can be unreliable if used alone and Arnold and Allen (1999) claim to have had good success with correlation between a separation technique and catchment mass balances for six USA streams.

Site Specific Scale

Rosenberry, LaBaugh and Hunt (2008) describe three of the more commonly used methods applied at the local scale for investigation groundwater-surface water interaction. The methods include water-level measurements and flow-net analysis, hydraulic potentiometer (mini-piezometer) and seepage meters methods. The water-level measurements and flow-net analysis method involves the measurement of water levels in a network of wells in combination with measurement of the river stage to calculate gradients and then water flow. The Hydraulic Potentiometer method makes use of multiple mini-piezometers to measure gradients. The Seepage Meters method makes use of seepage meters to directly measure flow across the sediment-water interface at the bottom of the surface-water body. Rosenberry *et al.* (2008) conclude that all three of the methods have specific advantages and disadvantages that results in the selection of a method for a local study area a unique one which depends on the characteristics of that specific site (Rosenberry and LaBaugh, 2008).

2.5. Review of selected South African SW-GW Interaction Methodologies

Guidelines

The Department of Water Affairs and Forestry developed the Guidelines for Groundwater Resources Management in Water Management Areas of South Africa in 2004. The Guidelines were aimed at the integration of coordinated groundwater management into Integrated Water Resources Management (IWRM) at different levels within Catchment Management Agencies (CMA). The guideline contains a step by step plan for groundwater resource assessments in South Africa, including initial/conceptual planning, water balance calculation, strategic environmental assessment, characterisation of the aquifer and detailed planning and reconnaissance. The guideline elaborates on the Water Balance Calculation in that a conceptual model of groundwater and surface water resources should be developed on which a water balance equation is based. It is important to gain a good understanding of the groundwater discharge from the system in order to ensure a balance between inflows and outflows. Groundwater discharge may present itself in the following forms: abstraction from boreholes, baseflow to rivers, baseflow to springs, baseflow to wetlands, discharge to the sea, transpiration from vegetation and evaporation from shallow groundwater. Methods which are available to directly measure or infer groundwater discharge volumes to rivers are summarised in the report. The use of streamflow hydrographs can be used to separate streamflow into its components based on the assumption that these different components will appear at different time intervals. The graphical method of hydrograph separation separates quick flow from slow flow purely based on graphical properties, but this method tends to inadequately describe stream chemistry during storm runoff events. Another method available is the isotope/chemical hydrograph separation where stable environmental isotopes are used in conjunction with the stream hydrograph to estimate the groundwater baseflow. The report continues that the hydrograph separation technique used will depend on the available data. The guideline suggests Bokuniewicz and Zeithin's (1980) method of directly measuring the groundwater inflow to surface water by means of a simple drum and plastic bag and Paulsen *et al.* (1997) ultrasonic groundwater seepage meter method which takes continuous measurements of groundwater seepage into the surface water body, as small scale methods for estimating the groundwater-surface water exchange. The report also documents best practices and step by step methodologies which can be used to implement the protection strategies as

stated in the National Water Act 1998. The National Water Act 1998 gave rise to Resource Directed Measures which includes resource classification, determination of resource management classes, reserve determination and setting of resource quality objectives. The guideline describes a comprehensive step by step implementation of the Resource Directed Measures. Furthermore, suggested methods of determining the groundwater-fed baseflow are given under the quantification of the Reserve. The guideline suggests the Smatkin or Herold hydrograph separation methods, Darcy's law, chemical investigations or numerical modelling for the quantification of groundwater-surface water interaction. However, the DWAF policy with regards to this step in the Resource Directed Measures is still in the formulation stage (DWAF, 2004).

Moseki (2013) states that there is no "one fits all" method to assess surface water-groundwater interactions and a number of factors along with the conceptual understanding of the area and the applicability of the data, play an important role. Moseki (2013) recommends the use of a framework to choose a particular method for a surface water-groundwater investigation at a particular location. A framework created by Moseki (2013) to inform decision processes in the selection of appropriate methodologies for the evaluation of surface water-groundwater interaction in the context of South Africa's environmental conditions is shown in Figure 2-5-1. Gaining a conceptual understanding of the area under investigation is recommended by Moseki (2013) as the first step in the process of evaluating the surface water-groundwater interaction, followed by whether exchange is possible, the degree of interconnectivity and subsequently the type of interaction occurring. Once this has been determined then an appropriate method of quantification can be selected. Moseki (2013) strongly recommends the use of more than one method to increase the confidence levels in the results and decrease the associated uncertainties.

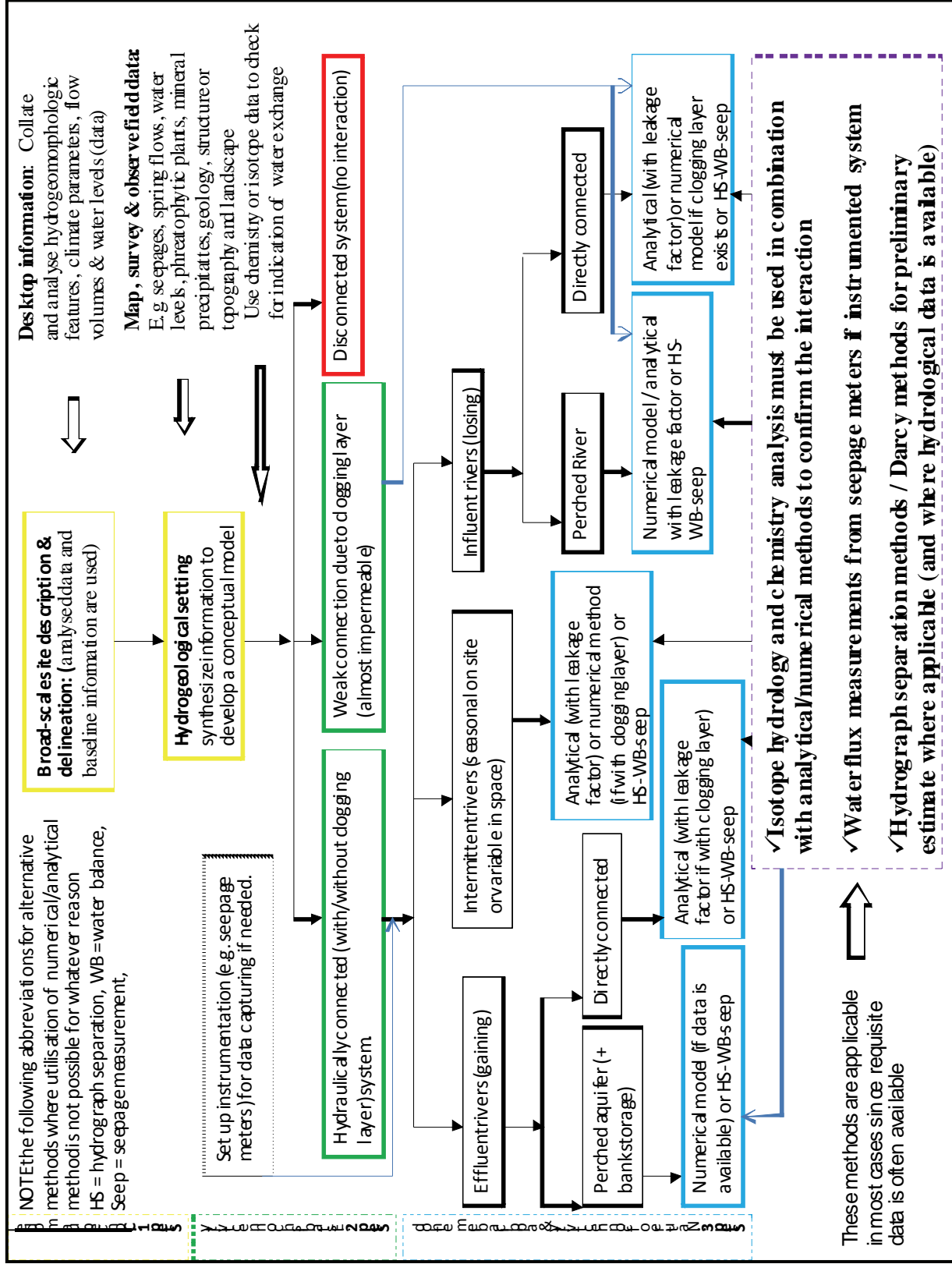


Figure 2-5-1 A framework for the selection of an appropriate method for the assessment of surface water-groundwater interactions (Taken from Moseki, 2013).

Hydrograph Separation techniques

Hydrogeomorphological approach

Xu *et al.* (2002) presented a geomorphologic framework on which the quantification of groundwater baseflow from a streamflow hydrograph can be discussed. The geomorphologic framework was developed in order to supplement the hydrograph separation techniques used in South Africa to quantify groundwater-surface water interaction. Xu *et al.* (2002) list the various methods of hydrograph separation utilised in South Africa, namely the RCD, Concentration ratio, Herold, SARES and Smakhtin methods. According to Xu *et al.* (2002) the RCD method is not frequently applied, the Concentration method is favoured for interflow investigations, the SARES method is favoured for ecological reserve investigations and the Herold and Smakhtin methods are the most acceptable and popular methods for the quantification of groundwater-surface water exchange. Halford and Mayer (2000) however found that hydrograph separation techniques are insufficient tools when used unaccompanied by additional methods to determine the interaction between groundwater and surface water. Hydrogeological investigations are traditionally qualitative and aimed at understanding the groundwater flow occurrences, where numerical solutions have been favoured for quantitative investigations (Xu *et al.*, 2002). Numerical simulation techniques tend to be costly and require additional calibration data. Xu *et al.* (2002) proposes an alternative approach where geomorphic characteristics of rivers are used to create hydrogeologic rules aimed to increase the consistency of the separation of streamflow by hydrograph techniques. Rivers are geomorphologically classified into upper catchment areas, middle river courses and lower river courses. The rivers are then further classified on the hydrogeomorphological type, namely constantly losing or gaining streams, intermittent streams, gaining streams with or without storage and interflow-dominant streams. Four different relationships between rivers and groundwater are defined based on geomorphologic typing, interaction scenarios, hydraulic connection and baseflow separation. Xu *et al.* (2002) propose an algorithm for the estimation of the monthly groundwater discharge which incorporates qualitative knowledge. The algorithm estimates the groundwater contribution to baseflow through a summation of the decay of the previous groundwater contribution and a rainfall-induced flow increment, where each different relationship between groundwater and surface water will result in different parameter values. The proposed approach was applied to the Sabie River, South Africa and Xu *et al.* (2002) report reasonable estimates which are

comparable to estimates presented by Vegter (1995). Xu *et al.* (2002) conclude that the proposed approach can add meaning to simple hydrograph separation techniques, but should be applied with caution as it is based on an hydrogeomorphological understanding and is subjective in nature (Xu *et al.*, 2002).

Recession Curve Displacement (RCD) Method

The RCD (Recession Curve Displacement) method developed by Rorabaugh (1964) is a recession analysis hydrograph separation technique which is based on the upward displacement of the recession curve during a rainfall event. The total recharge to the groundwater system during the rainfall event is shown to be approximately double the total potential discharge to the stream at a critical time (T_c) after the hydrographic peak, by means of an algorithm. The total volume of groundwater recharge due to the rainfall event (R) can be estimated from the stream hydrograph by (Brodie and Hostetlet, 2005):

$$R = \frac{2(Q_2 - Q_1)K}{2.3026} \quad (2.2)$$

where,

Q_1 is the baseflow at critical time (T_c) extrapolated from the pre-event recession curve,
 Q_2 is the baseflow at critical time (T_c) extrapolated from the post-event recession curve, and
 K is the recession index.

The recession index (K) is estimated from the stream hydrograph record, and then used to determine the critical time (T_c) from the relationship $T_c = 0.2144K$. Figure 2-5-2 graphically describes the various parameters in Equation 2.2. When the recharge to groundwater is known, Equation 2.2 can be rearranged to solve for the groundwater baseflow from the pre-event recession curve.

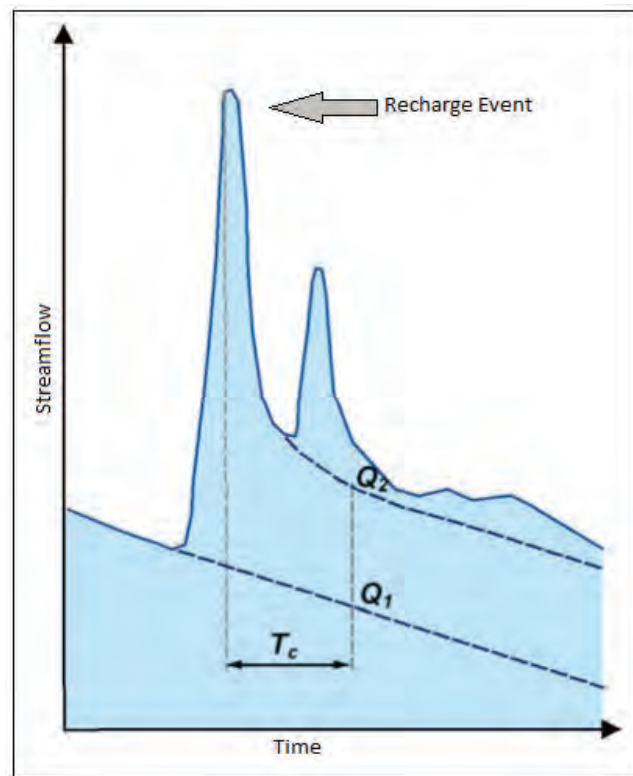


Figure 2-5-2 Graphical representation of the RCD method parameters T_c , Q_2 and Q_3 for a stream hydrograph recharge event (Modified from Rutledge and Daniel (1994)).

Concentration Ratio Method

The Concentration Ratio method referred to by Xu *et al.* (2002) is a chemical hydrograph separation method. The method is also referred to as the Tracer method in Australian Government (2012a) and as a chemical method in DWAF (2006c). End-members of streamflow, runoff and groundwater are chemically defined with recommended tracers including chloride (Cl), silica (Si), hydrogen-2 isotope (^2H) and oxygen-18 isotope (^{18}O). Thus, at any point along the river, the proportion of river flow supplied by groundwater discharge is calculated based on a chemical mass balance equation (DWAF, 2006c):

$$Q_g = Q_T \left(\frac{C_r - C_d}{C_g - C_d} \right) \quad (2.3)$$

where,

Q_g is the baseflow volume,

Q_T is the total measured streamflow, and

C_d , C_g and C_r are the concentrations of surface runoff, groundwater and streamflow.

Herold Method

The Herold hydrograph separation method was developed by Herold (1980). The method is used in the Water Resources 1990 project to separate monthly flows into surface and groundwater components. The method is based on the following equation (DWAF, 2006a):

$$Q_g = Q_{gi-1} \cdot Decay + Q_{i-1} \cdot PG \quad (2.4)$$

where,

- Q_{gi} is the groundwater contribution of the current month,
- Q_{gi-1} is the groundwater contribution of the preceding month,
- Q_{i-1} is the total streamflow of the preceding month,
- Decay* is a groundwater factor ($0 < Decay < 1$), and
- PG* is a groundwater growth factor (%).

The calculated groundwater baseflow (Q_{gi}) is thus the combined effect of previous groundwater after decay ($Q_{gi-1} \cdot Decay$) and rainfall induced recharge ($Q_{i-1} \cdot PG$). The Herold method has four parameters which it requires for computation, namely *DECAY*, *PG*, *GGMAX* and *QGMAX*. *GGMAX* is varied on a monthly time step in response to *DECAY* (Groundwater decay factor) and *PG* (Groundwater growth percentage). *DECAY*, *PG* and *QGMAX* are selected on the basis of a realistic division of groundwater and surface water (Levy and Xu, 2011). The calibration of these parameters is facilitated by a graphical output of total and groundwater hydrographs, but the selection remains completely subjective as what is the definition of a “realistic” division.

SARES method

SARES is a computer programme forming part of the Decision Support System (DSS) software developed by Hughes and Münster (1999). The programme was developed as a tool to rapidly access an initial low-confidence estimate of the quantity component of the Reserve for rivers at the outlet of any quaternary catchment. The final result of the programme is a time series of monthly flow volumes recommended for the quantity component of the Ecological Reserve (Hughes and Münster, 1999).

In order to estimate the Ecological Reserve the in-stream flow requirement needs to be determined, which consists of four components in the SARES programme, namely low and high flow maintenance quantities and the high and low flow drought quantities. The low flow

maintenance quantity can be assumed to the groundwater contribution to stream flow (Hughes and Münster, 1999).

The maintenance low flow requirements are determined based on the study area's Ecological Management Class (EMC) and the defined CVB index, which is a combination of the variability index and the Baseflow Index (BFI). The equation used to determine the maintenance low flow requirement is (Hughes and Münster, 1999):

$$MLIFR = LP4 + \frac{(LP1 \cdot LP2)}{(CVB^{LP3})^{(1-LP1)}} \quad (2.5)$$

where,

MLIFR is the maintenance low flow total as % natural MAR, and LP1-4 are parameters associated with the EMC set (Table 2-5-1).

Table 2-5-1 Parameters LP1-P4 for each Ecological Management Class option (Taken from Hughes and Münster, 1999).

Parameter	Ecological Management Class						
	A	A/B	B	B/C	C	C/D	D
LP1	0.9	0.905	0.91	0.915	0.92	0.925	0.93
LP2	79	61	46	37	28	24	20
LP3	6	5.9	5.8	5.6	5.4	5.25	5.1
LP4	8	6	4	2	0	-2.0	-4.0

Smakhtin Method

The Smakhtin method is a recursive digital filter method for monthly baseflow separation developed by Smakhtin (2001). The method is based on daily separation technique developed by Nathan and McMahon (1990). A filter parameter (α) is used to separate quick flow from baseflow. Quick flow is assumed to be made up of interflow and storm runoff, and the remaining flow is assumed to be baseflow.

The monthly Smakhtin groundwater baseflow equation is (Smakhtin, 2001):

$$QB_m = Q_m - q_m \quad (2.6)$$

and,

$$q_m = \alpha q_{m-1} + 0.5(1 + \alpha)(Q_m - Q_{m-1}) \quad (2.7)$$

where,

q_m and q_{m-1} are the current and previous monthly flow attributed to high-flow events,

Q_m and Q_{m-1} are the current and previous months total monthly flow,
 QB_m is the part of the total monthly flow which could be attributed to baseflow, and
 α is the filter parameter.

The determined baseflow is further constrained to ensure that the volume does not become negative or exceed the original total monthly streamflow in any month.

Statistical Analysis

River heterogeneity signatures, derived from a combination of geomorphological province, eco-region and an index of river flow variability (i.e. the Hydrological Index (HI)), is used to prioritise conservation of South African rivers (Le Maitre and Colvin, 2008). The Hydrological Index is a general index of flow variability calculated from two standard flow statistics, namely the Coefficient of Variation Index (CVI) and the Baseflow Index (BFI). A similar principal is applied by Le Maitre and Colvin (2008) to assess the effectiveness of river flow statistics in characterising the contribution of groundwater to a river flow system. Flow statistics, extracted from the SPATSIM modelling system, were used to estimate the contribution of groundwater to streamflow and subsequently compared in terms of principal aquifer types in South Africa. The analysis found that the river flow statistics commonly used in river investigations (CVI, BFI and HI) and flow concentration statistics, in relation to the percentage zero flows in a catchment, are complex and variable on a national scale. When the relationships were investigated on a smaller scale, catchments within the Crocodile-Marico Water Management Area, they were still found to be complex. Nel *et al.* (2004) suggest that there might not be a statistic which can be used as an indication of the groundwater contribution to baseflow as the Hydrological Index (HI) is used for river conservation planning. However, Le Maitre and Colvin (2008) found that zero flows might be a useful indicator of groundwater baseflow, especially if combined with groundwater flow concentrations. Zero flow statistics and groundwater flow concentrations are recommended for further testing (Le Maitre and Colvin, 2008).

Hydrological Modelling

Hydrological models are used as simple, conceptual representations of the hydrological cycle or parts thereof to better facilitate understanding the processes occurring therein and consequently predicting what may happen in the future. South Africa is a semi-arid to arid area which has a number of different climate, rainfall and vegetation zones. There have been

a number of successful developments and applications of hydrological models in the country despite the limitations of variable climatic conditions and data scarcity (Hughes, 2008). Hughes (2008) reviews hydrological models which have been specifically developed for the arid southern region of Africa. The hydrological model which has been the most extensively applied within southern Africa is the Pitman monthly time-step model. The model has however undergone numerous revisions since its development in the 1970's (Hughes, 2008). The Pitman model is an explicit soil moisture accounting model which represents interception, soil moisture and groundwater storages with model functions to allow inflow and outflow from these components (Hughes, 2008). Most model versions use a semi-distributed system where each sub-area has its own hydrometeorological inputs and parameter set, including components to simulate abstractions from distributed farm dams and direct flow from the river or major dam at the outlet of each sub-area. The modified Pitman model developed by Hughes (2004) incorporates a groundwater component to estimate groundwater baseflow. The Hughes version of the Pitman model has 24 model parameters, but there are guidelines available for parameter estimation, provided by the WR90 study (Midgley and others, 1994). Hughes (2008) includes that the model does however perform better in humid and temperate areas than in the more arid regions which is a consequence of the poorly defined real spatial variations in rainfall input, limitations in the temporal distribution of rainfall within a single month and the relatively simplistic approach to simulating runoff generation.

The Agricultural Catchments Research Unit (ACRU) is a multi-purpose model originating from an evapotranspiration study conducted by the University of Natal (Schulze, 1989). The model integrates water budgeting and runoff components of the terrestrial hydrological system with risk analysis. The model can be applied in crop yield modelling, design hydrology, reservoir yield simulation, irrigation water demand and supply, planning of optimum water resources and regional water resource assessment among other applications (Schulze, 1989). One of main outputs of the model is a water balance, but the model is not directly geared towards the quantification of the exchange between groundwater and surface water. Hughes (2008) highlights that the model has a large number of parameters that require quantification and has been applied in mostly temperate and humid areas in South African to assess the impacts of various land use modifications.

The Variable Time Interval (VTI) model, developed at the Institute for Water Research (IWR), Rhodes University, South Africa has been applied to a large number of basins under the Southern African FRIEND programme (Hughes, 2008; Hughes, 1997). The VTI model is basically a daily time step model which can use smaller modelling intervals during periods of higher activity. The model requires short interval rainfall data for increased modelling interval times and the main moisture accounting routines are complex resulting in a large number of parameters (Hughes, 2008). Hughes (2008) concludes that the successful use of the VTI model would require a detailed understanding of the models structure, a good conceptual understanding of the main runoff generation mechanisms in the catchment and good quality climate data.

The Spatial and Time Series Information Modelling (SPATSIM) system was developed by the Institute of Water Research to replace the outdated integrated modelling environment package (HYMAS) used by the SA FRIEND project. The SPATSIM system has since been adopted as the main modelling environment to be used for the update of the South African water resources information system, WR90.

Hughes (2008) concludes his review of available hydrological models by highlighting that the success of any modelling study depends on the quality and appropriateness of the model as well as the quality of the input data and the level of experience of the user of the model. The better use of available data and the development of comprehensive guidelines for these models will serve to greatly improve model estimates, while only limited changes to existing models would be required (Hughes, 2008).

GRA2: Surface water-groundwater interaction methodology

The assessment methodology for the quantification of groundwater-surface water interaction is given in DWAF (2006a) and referred to herein as the Sami model. DWA (2009) highlights that this methodology has been reviewed by hydrogeologists and its applicability has been brought into question (Dennis, 2005; Sami and Witthüser, 2006). An additional method which is used to estimate the groundwater contribution to baseflow is the Hughes model, a modified version of the Pitman model developed by Hughes (2004) and based in the SPATSIM system. Both the Sami and Hughes model methodology will be described in detail.

2.5.1. Sami Model

Overview of Sami model initiative

Double accounting, groundwater underflow and the simplistic linear approach incorporated by the widely-used MODFLOW modelling program were highlighted in DWAF (2006a) as problems to consider when developing a SW-GW interaction methodology. The methodology used by MODFLOW to determine surface water-groundwater interaction is considered simplistic due to the assumption therein that the relationship between head difference and water exchange is linear. DWAF (2006a) states surface water-groundwater interaction is not linear because hydraulic resistance will result in non-linearity as streamflow increases. The method used by MODFLOW is further critiqued on the grounds that it cannot process changes in streamflow over time and erroneously implies that the rate of flow from the river into the groundwater system will equal the rate of flow to that river from the groundwater system. DWAF (2006a) suggests that these problems could be overcome by using a non-linear equation to simulate the interaction as this would be a more realistic approach.

The main aim of the Sami model was the development of a methodology which could determine the impacts of abstraction on baseflow without the necessity of modelling. The simulation of interactions under abstraction conditions is important as the related decline in groundwater levels can capture ambient groundwater that would have otherwise discharged as baseflow and in extreme cases induce streamflow into the groundwater system. Groundwater abstraction upsets the natural, steady-state condition of the water table by increasing recharge or decreasing discharge until a new equilibrium is reached. However, DWAF (2006a) adds that until the new equilibrium is reached, where pumping is balanced by

baseflow depletion, the abstraction results in aquifer storage depletion. Groundwater abstraction calculations should thus include both aquifer storage depletion and baseflow depletion components. The transition from aquifer storage depletion to streamflow depletion is a slow process and depends on the rate at which discharge can be captured (aquifer diffusivity), the location of pumping wells and time. DWAf (2006a) concludes that determining the magnitude of potential groundwater abstraction should be aimed at developing relationships between abstraction and baseflow depletion, instead of simply on projected drawdown.

Methodology summary

The Sami model is based on an eight-stepped methodology, including determining the amount of groundwater discharging to the surface water body in question. The eight steps are hydrograph separation, estimation of recharge, groundwater storage increments from recharge, evapotranspiration from shallow groundwater, groundwater outflow, groundwater baseflow and transmission losses, interflow, and groundwater abstraction.

Hydrograph Separation

1.1.1.1.1 Method:

The Herold hydrograph separation method is performed, using monthly streamflow data, in order to separate the total baseflow (groundwater and interflow) contributing to the river from the total runoff.

The Herold hydrograph separation equation:

$$Q_{gi} = (Q_{gi-1} \cdot Decay) + (Q_{i-1} \cdot PG) \quad (2.8)$$

where,

- Q_{gi} is assumed to be the groundwater contribution,
- Q_{i-1} is the total streamflow of the preceding month,
- $Decay$ is a groundwater factor ($0 < Decay < 1$), and
- PG is a groundwater growth factor (%).

From Herold's equation it can be seen that the groundwater baseflow (Q_{gi}) is defined by previous groundwater after decay ($Q_{gi-1} \cdot Decay$) and rainfall induced recharge ($Q_{i-1} \cdot PG$), which includes interflow.

Inputs:

- Monthly streamflow data (WR90 data, observed gauging weir data or stochastic hydrographs)
- Previous months groundwater contribution (Q_{gi-1})
- Groundwater *Decay* factor
- Groundwater growth factor (PG)

Assumptions:

The general assumptions of a hydrograph-separation include:

- Hydraulic characteristics of the contributing aquifer can be estimated from stream-discharge records,
- Periods of exclusively groundwater discharge can be reliably identified, and
- Stream-discharge peaks approximate the magnitude and timing of recharge events

The assumptions in the Herold method hydrograph separation methodology are:

- Streamflow below a certain, pre-defined parameter ($GGMAX$) is groundwater flow,
- This upper limit of groundwater ($GGMAZ$) can be correctly varied month to month based on the surface water runoff from the preceding month and calibrated parameters defining groundwater decay ($DECAY$) and groundwater growth (PG),
- Parameters $DECAY$, PG and $QGMAX$ can be appropriately selected on the grounds of a “realistic” division between surface water and groundwater, and
- The selection of $DECAY$, PG and $QGMAX$ is facilitated by a graphical output of total and groundwater hydrographs.

Estimation of Recharge

Method:

It is required to first estimate the recharge by calculating subsurface storage by reverse engineering of the Pitman model, in order to subdivide baseflow into groundwater baseflow and interflow.

1. Calculating soil moisture storage (S):

The Pitman Runoff-soil moisture relationship equation:

$$Q = FT \left(\frac{S-S_L}{S_T-S_L} \right)^{POW} \quad (2.9)$$

where,

S is the actual soil moisture storage (mm),

S_L is the minimum soil moisture storage below which no runoff occurs,
 S_T is the maximum soil moisture storage,
 POW is the power function of the runoff-soil moisture curve, and
 FT is a parameter of the maximum baseflow depth at S_T .

The Sami model then reverse engineers this relationship to calculate the soil moisture storage (S), by using parameters S_L , S_T , FT and POW from WR90 and the total baseflow volume from the hydrograph separation for Q .

2. Calculating monthly recharge (Re):

Once soil moisture is calculated, or input from WRSM2000 obtained, potential monthly recharge is calculated using the Hughes Recharge-soil moisture relationship:

$$Re = GW \left(\frac{S - S_L}{S_T - S_L} \right)^{GPOW} \quad (2.10)$$

where,

Re is the potential recharge,

S is the actual soil moisture (mm),

S_L is now the soil moisture, below which there is no recharge (mm),

GW is the maximum amount of recharge at maximum soil moisture (S_T) in mm,

$GPOW$ is the power function of the storage-recharge relationship.

DWAF (2006a) state parameters GW and $GPOW$ can either be calibrated to achieve a fit with long term mean annual measurements obtained from other methods, or initial values could be chosen equal to the FT and POW parameters of the Pitman Runoff-soil moisture relationship.

Inputs:

- Baseflow value from hydrograph separation (Q), or time series of the Pitman S variable
- Parameters S_L , S_T , FT and POW
- Parameters S_L , GW and $GPOW$

Assumptions:

Runoff-soil moisture relationship:

- The assumption is made that the Pitman runoff-soil moisture relationship estimates runoff comprised of interflow and groundwater baseflow, which under another assumption that the Herold method separates baseflow and interflow

from total flow, allows baseflow from the Herold method to be used in the Pitman runoff-soil moisture relationship.

- The assumption is made that the same maximum soil moisture value (ST) can be used in both the Pitman runoff-soil moisture relationship and the Hughes recharge-soil moisture relationship.
- The parameter POW is assumed to represent the relationship between total basin moisture and the spatial distribution of this moisture.

Recharge-soil moisture relationship:

- The surface characteristics can be represented by a single storage, given that direct recharge can occur where there are bare rock areas.
- The depth of recharge can be estimated as a non-linear relationship with the ratio of current storage to the maximum storage.

Groundwater Storage Increments from recharge

Method:

Direct recharge from soil moisture is incremented to the groundwater aquifer storage. However, if the aquifer has reached its calculated capacity, the excess becomes interflow. As a result, aquifer recharge may be somewhat less than the calculated potential recharge. Actual recharge is calculated by subtracting the excess (interflow) from the potential recharge.

Thus, if the sum of groundwater storage and incremented recharge is greater than the aquifer capacity (CAP), then the potential recharge will equal the actual recharge and if less than the aquifer capacity (CAP), then the potential recharge minus the excess recharge ($EXCESS_1$) will be the actual recharge.

It should be noted that under abstraction conditions, the aquifer capacity could be increased by reducing groundwater storage which would lead to less excess recharge. The capacity of the aquifer is calculated as:

$$CAP = b \cdot S \quad (2.11)$$

where,

CAP is the capacity of the aquifer,

b is the aquifer thickness estimated from the recommended drilling depth, and

S is Storativity.

Inputs:

- Aquifer thickness and storativity (aquifer capacity)
- Groundwater storage

Assumptions:

- Aquifer capacity can be estimated by the product of aquifer thickness and aquifer storage.
- Once the aquifer capacity has been reached, any additional recharge would become interflow.
- Aquifer thickness can be estimated by the recommended drilling depth below groundwater level.

Evapotranspiration from shallow groundwater

Method:

Evapotranspiration from groundwater is calculated using:

$$ET = ((MAE \cdot MDIST \cdot CROP) - RAIN) \cdot \left(AREA \cdot \left(\frac{STORE - SWL}{CAP - SWL} \right) \right) \quad (2.12)$$

where,

MAE is the mean annual evaporation,

MDIST is the monthly distribution of evaporation (%),

CROP is the monthly A-pan crop factor for an appropriate cover,

RAIN is the monthly rainfall variable,

AREA is the area where Evapotranspiration from groundwater can take place,

STORE is the variable of groundwater storage,

CAP is the capacity of the aquifer, and

SWL is the static water level.

The static water level is calculated as:

$$SWL = (CAP - \Delta \bar{h}) \cdot S \quad (2.13)$$

where,

$\Delta \bar{h}$ is the degree of annual groundwater level fluctuation,

CAP is the aquifer capacity, and

S is storativity.

Rainfall is subtracted from monthly evapotranspiration ($MAE \cdot MDIST \cdot CROP$), to obtain the evapotranspiration demand from groundwater. The evapotranspiration demand from groundwater is then multiplied by an aquifer storage factor $\left(AREA \cdot \left(\frac{STORE - SWL}{CAP - SWL} \right) \right)$, to allow

evapotranspiration to be decreased as groundwater storage is depleted. The calculated evapotranspiration is then decremented from groundwater storage.

Inputs:

- Parameters *MAE*, *MDIST*, *CROP*, *RAIN*, *AREA*, *STORE*, *SWL* and *CAP*
- Aquifer thickness and Storativity (*CAP*)
- Degree of annual groundwater level fluctuation ($\Delta\bar{h}$)

Assumptions:

- Evapotranspiration from groundwater only takes place when the evapotranspiration demand is not met by the amount of rainfall that month.
- Monthly evapotranspiration can be estimated by the product of mean annual evapotranspiration, monthly distribution of evapotranspiration and crop factor.

Groundwater Outflow

Method:

A Darcian approach is used to calculate groundwater outflow (underflow). DWAF (2006a) state “groundwater outflow is calculated using the Darcian approach of the product of parameters transmissivity and hydraulic gradient oriented out of the catchment”. The maximum hydraulic gradient, defined by a parameter *HGRAD* (channel gradient), is decremented as groundwater storage approaches the static water level, by multiplying the gradient with an aquifer storage factor:

$$i = HGRAD \left(\frac{STORE-SWL}{CAP-SWL} \right) \quad (2.14)$$

where,

HGRAD is the maximum hydraulic gradient, and
i is the hydraulic gradient.

The calculated groundwater outflow is then decremented from groundwater storage.

Inputs:

- Transmissivity
- Parameters *HGRAD*, *STORE*, *SWL* and *CAP*

Assumptions:

Darcy’s assumptions:

- The groundwater discharge is directly proportional to the transmissivity, hydraulic gradient.

- The flow dimensions are assumed to be one-dimensional, as this form of Darcy's law describes one-dimensional, pipe flow.
- The groundwater flow is slow and the Reynolds number is less than 10, where resistive forces of viscosity are dominant and laminar flow occurs.

Groundwater baseflow and transmission losses

Method:

Groundwater storage has been decremented by both the calculated evapotranspiration and outflow. Groundwater baseflow is now calculated as a function of the head difference between the new decremented groundwater level and the surface water level. The groundwater head is calculated as the difference between storage and the static water level ($STORE - SWL$), while the surface water head is calculated as the monthly runoff volume divided by the catchment area ($Runoff/AREA$).

Effluent conditions are simulated when the groundwater head is greater than the surface water head (groundwater lost to the river as baseflow). Inflow conditions are simulated when the groundwater head is less than the surface water head (transmission losses into the groundwater system). The model allows for the calculation of both groundwater and surface water head differences on a monthly time step, thus overcoming the short coming highlighted in MODFLOW of the unrealistic assumption of constant head conditions in the river.

Once the direction of the interactive water flow is determined, the groundwater baseflow or transmission losses are calculated using a non-linear equation to account for the effects of hydraulic resistance:

$$GW_{baseflow} = (1 - e^{HEAD \cdot (-0.05)}) \cdot BFMAX \quad (2.15)$$

where,

$GW_{baseflow}$ is the groundwater baseflow,

$HEAD$ is the difference between the calculated groundwater and surface water heads, and

$BFMAX$ is parameter of the maximum rate of groundwater baseflow.

Parameters $BFMAX$ and SWL can be calibrated on the principal that groundwater baseflow approximately equals, but doesn't exceed the total streamflow at the lowest flow period on a hydrograph. If there is no interaction between groundwater and surface water, then groundwater baseflow is set to zero and if the calculated groundwater baseflow exceeds the total baseflow, the groundwater baseflow is defaulted to the total baseflow volume.

Inputs:

- Groundwater storage less evapotranspiration and outflow (*STORE*)
- Static water level (*SWL*)
- Total monthly runoff
- Catchment area
- Maximum rate of groundwater baseflow (*BFMAX*)

Assumptions:

- The decremented groundwater storage parameter (*STORE*), resulting from the series of steps in the methodology is representative of the groundwater storage of the catchment
- The surface water head is sufficiently estimated by the monthly runoff volume divided over the catchment area.
- The assumptions from each calculation which is incorporated into the estimation of groundwater baseflow.

Interflow

Method:

DWAF (2006a) state the interflow of a catchment (under virgin conditions) is calculated as the difference between the total baseflow and the calculated groundwater baseflow. However, abstraction decreases groundwater storage which increases aquifer capacity, implying that interflow is expected to be less under abstraction conditions because of the potential recharge volume can become the actual recharge volume due to the increased capacity of the aquifer.

The depletion of interflow is calculated by:

$$Interflow = Q_g - GW_{baseflow} - EXCESS_1 + EXCESS_2 \quad (2.16)$$

where,

Q_g is the total baseflow,

$GW_{baseflow}$ is the groundwater baseflow,

$EXCESS_1$ is the recharge in excess of aquifer capacity under virgin conditions, and

$EXCESS_2$ is the recharge in excess of aquifer capacity under abstraction conditions.

Inputs:

- Total baseflow (Q_g)
- Groundwater baseflow ($GW_{baseflow}$)
- Aquifer capacity at virgin and modified conditions

- Potential recharge

Assumptions:

- Interflow is the difference between total baseflow and groundwater baseflow.
- Groundwater abstraction decreases groundwater storage which is assumed to increase aquifer capacity.

Groundwater abstraction

Method:

DWAF (2006a) assume that groundwater abstraction depletes groundwater storage and groundwater baseflow in a non-linear manner. This non-linear relationship is dependent on the transmissivity and storativity of the aquifer, the distance from the stream and the time since abstraction started. The streamflow depletion solution is an analytical solution of the Glover and Balmer (1954) stream-depletion method (DWAF, 2009).

The Sami model groundwater abstraction equation:

$$\%GW = \frac{100}{(1+e^{(k_3+(k_2 \cdot t')})} \quad (2.17)$$

and,

$$t' = \frac{4Tt}{x^2S} \quad (2.18)$$

where,

- %GW is the percentage of groundwater abstraction derived from groundwater storage,
- t' is a dimensionless time parameter calculated as,
- T is the transmissivity,
- S is the storativity,
- x is the distance to the stream,
- t is the time since pumping began, and
- k_3 and k_2 are parameters used to ensure that the percentage of abstraction from groundwater storage (%GW) is 100% when pumping is commenced.

Inputs:

- Transmissivity (T), storativity (S), time since pumping started (t), and the distance to the stream from the abstraction point (x)
- Calibrated parameters k_3 and k_2

Assumptions:

- Groundwater abstraction is assumed to deplete groundwater storage and groundwater baseflow in a non-linear fashion depending on transmissivity, storativity, the distance to the stream and the time since pumping started.

Assumptions associated with the Glover and Balmer (1954) streamflow depletion method, which was based on the earlier work of Theis (1941) (Contor, 2011):

- The river is infinitely long.
- The aquifer is semi-infinite; the only boundary to the aquifer is the connected river.
- The results aggregate the effect upon the entire length of the river.
- The river is straight.
- The river fully penetrates and is in full communication with the aquifer.
- The aquifer is homogeneous and uniform.
- Saturated thickness of the aquifer is constant over time.

Review

The Herold method of hydrograph separation used in the Sami model is widely accepted and used in South Africa due to the fact that the only data requirement is readily available streamflow data. The method is an improvement with respect to earlier hydrograph separation techniques, but remains subjective according to Xu *et al.* (2002). The general technique was criticised by Halford and Mayer (2000) on the grounds that the major assumptions of the method are commonly violated making hydrograph separation techniques poor tools for estimating groundwater discharge or recharge. Levy and Xu (2011) note that the hydrograph separation methods are indeed informative, but when applied to a single downstream hydrograph, were not able to account for spatial heterogeneity. On the other hand, Arnold and Allen (1999) reported to have found a good correlation between a separation technique and catchment mass balances for six USA streams and Wittenberg and Sivapalan (1999) successfully analysed streamflow to determine all the main components of groundwater balances, for a catchment in the humid part of Western Australia (Xu and Beekman, 2003). Levy and Xu (2011) refer to Parsons and Wentzel (2007) describing the Herold hydrograph separation method as estimating the groundwater contribution to streamflow by assuming that during each month the groundwater contribution will not drop below a certain amount. This statement holds a certain degree of uncertainty in the fact that

the “groundwater contribution” could be interpreted as either total baseflow to the stream (groundwater + interflow) or as the actual groundwater contribution to baseflow (only groundwater). Subsurface baseflow was originally considered to be comprised of only groundwater discharge, but it is now understood to be comprised of both groundwater discharge and interflow. The question is whether Herold refers to the former or latter understanding of groundwater baseflow.

The Sami model does allow for an alternative route to be taken, in that instead of estimating the soil moisture value from the Pitman runoff-soil moisture relationship, that one could directly utilize time-series Pitman S-values from a WRSM2000 model run. This alternative method would compensate for the possible misconception in the hydrograph separation step. The time-series Pitman S-values from WRSM2000 (using default values) would be acceptable for a desktop or initial estimation of recharge for the quaternary.

The Sami model assumption that aquifer capacity can be estimated from the product of aquifer thickness and aquifer storage (Storativity) is reasonable. The determination of the aquifer thickness and storativity for the calculation however has uncertainties. The aquifer thickness is estimated from the *Recommended Drilling Depth Below Groundwater level* from the Map of National Groundwater Resources Map of South Africa. Vegter (1995) considered aquifer storage as groundwater stored in the upper weathered and fractured zone. Vegter’s *Optimum Drilling Depth* was determined by the addition of the depth to the fresh bedrock, estimated by statistical analysis of the data within the National Groundwater Database (NGDB) at the time, and the mean depth to groundwater level (DWAF, 2003). The use of this *Optimum Drilling Depth* is not the optimum situation as the data used to determine these depths is outdated and includes the assumption that the groundwater capacity is determined from the weathered subsurface (main water bearing depth) and a further distance of half the weathered depth (Figure 2-5-3). The aquifer capacity is based on a fixed groundwater level, but as groundwater is recharged, the water level will rise. In reality the depth for interflow and groundwater baseflow is variable. In times of high rainfall the groundwater level can reach the surface, thus excess recharge would result in direct surface runoff. Figure 2-5-4 indicates that if the aquifer capacity is based on a static water level, when the water table does rise, groundwater would then be accounted for as interflow. This highlights that there is ambiguity when determining the upper limit of the aquifer, namely is it defined by the static

water level or by the ground surface. The methodology does not state where the storativity values are obtained from, but are probably obtained from the WR90 database. The methodology also does not explain how the groundwater storage is estimated. The method and inherent uncertainties associated with determining this initial groundwater storage will affect the final calculation of groundwater baseflow because this initial value will be the base from which the other water balance components are subtracted from to determine the final decremented groundwater storage.

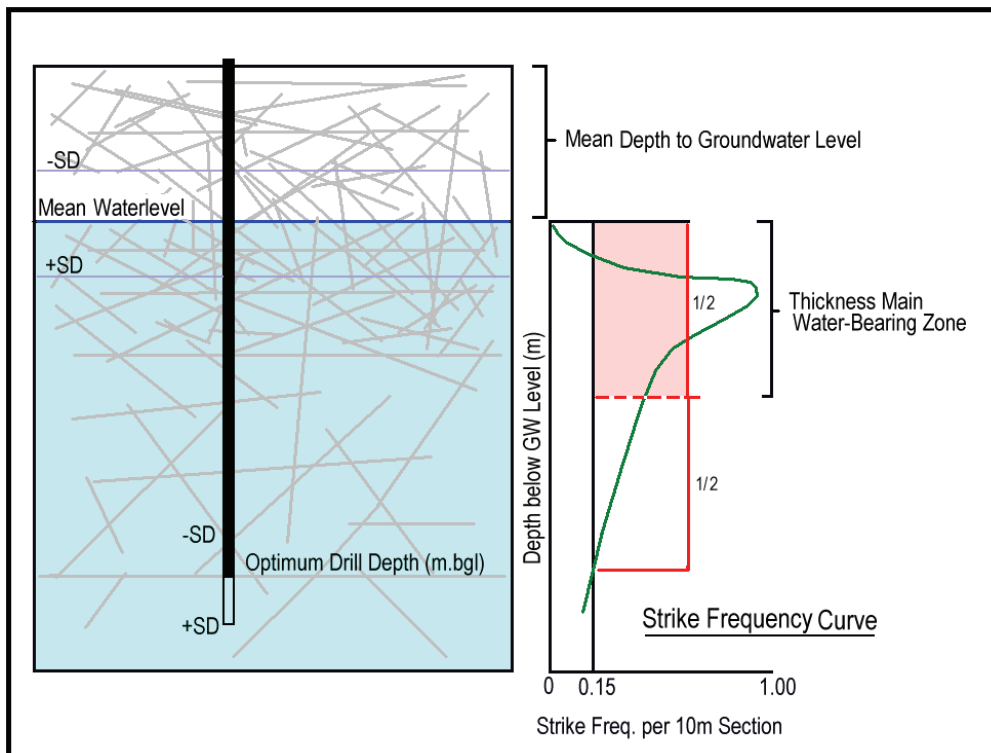


Figure 2-5-3 A schematic representation of the thickness of Vegter's main water-bearing zone and optimum drilling depth (Taken from DWAF, 2003)

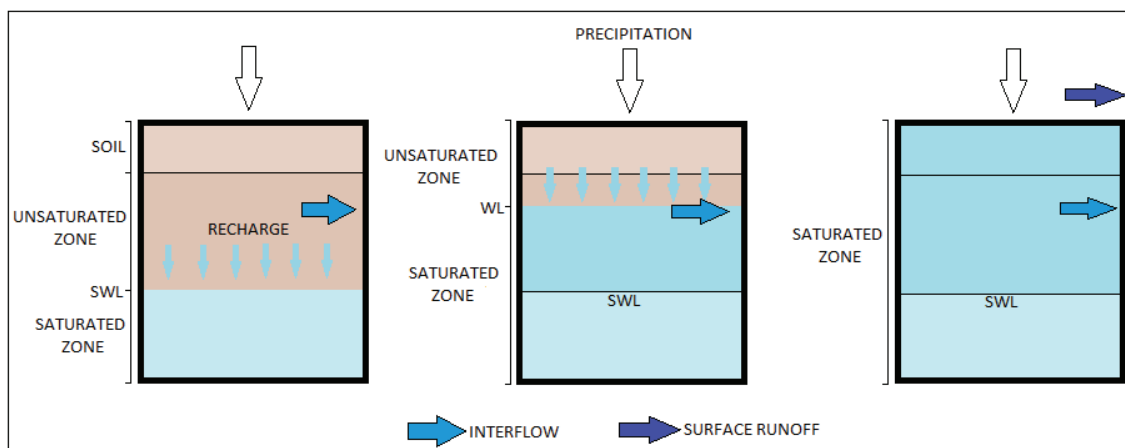


Figure 2-5-4 A graphical representation of the subsurface, indicating the variability of the groundwater level in relation to recharge.

The proposed equation for calculating evapotranspiration from groundwater is acceptable when one considers the difficulty in estimating evapotranspiration from groundwater. The assumption that evapotranspiration will only occur from groundwater if the evapotranspiration demand is not met from rainfall is incorrect, as this depends on the depth of the water level. There is an *evapotranspiration extinction depth*, the maximum depth at which water can move upwards under the forces of evapotranspiration (Figure 2-5-5). This depth is determined by the type of plants in the area. It can thus be seen that even if the evapotranspiration demand is not met by the rainfall, if the water level is below the *evapotranspiration extinction depth* than no evapotranspiration from the groundwater will occur.

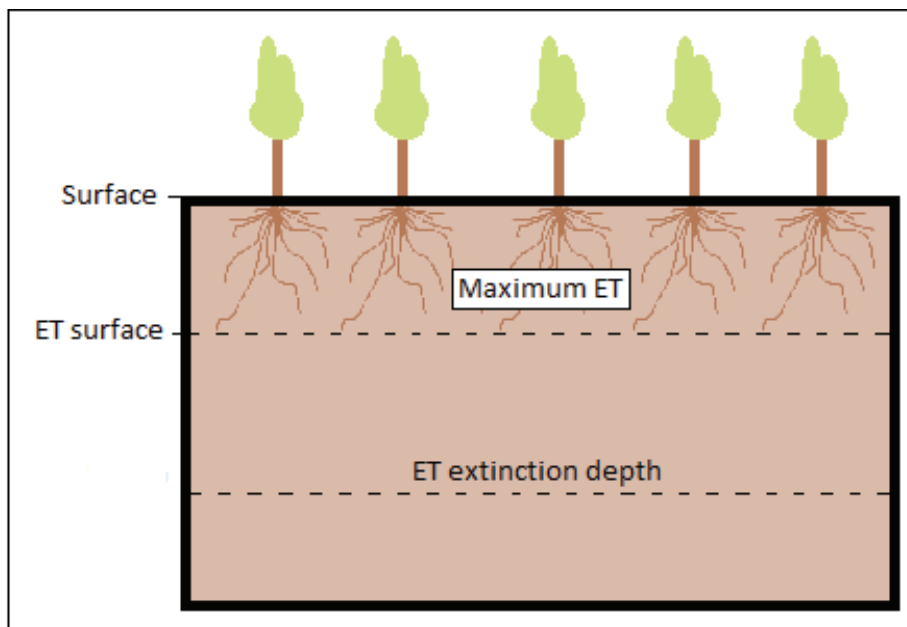


Figure 2-5-5 A schematic representation of the evapotranspiration extinction depth, the depth below which no evapotranspiration can occur.

The use of the Darcy equation in the Sami model to determine the groundwater outflow could lead to an over-simplification as the rate of groundwater outflow is directly proportional to the transmissivity and the hydraulic gradient. Phillips and Ingersoll (1998) report on the estimation of lateral subsurface outflow estimation by Lines (1979), where the Darcian approach was used.

The equation used in this study is:

$$Q = TiL \quad (2.19)$$

where,

Q is the groundwater flow (m^3/d),

T is the transmissivity (m^3/d),

$HGRAD$ is the maximum hydraulic gradient, and

L is the length of the section perpendicular to the direction of flow.

However, in the Sami model methodology groundwater outflow is stated to be the product of parameters transmissivity and hydraulic gradient only, and does not mention the parameter L , the length of the section perpendicular to the direction of flow. The two components of this equation with the most uncertainty are transmissivity and the horizontal hydraulic gradient according to Phillips and Ingersoll (1998). Transmissivity values are often estimated at specific sites and then extrapolated over a larger area and the horizontal hydraulic gradient can vary along the length of the cross-section under investigation (Phillips and Ingersoll, 1998). DWAF (2006a) does not report on where the transmissivity values were obtained from.

The Sami model assumption that interflow is equal to the difference between the total baseflow and groundwater baseflow volumes is correct, if other factors such as bank storage are ignored. The assumption that groundwater abstraction decreases groundwater storage and thus increases aquifer capacity is valid, but the loss of interflow and increased recharge would only be seen if the aquifer regularly reached its capacity. The extension of the Sami model to include the use of time-series of Pitman S variable to calculate recharge allows the user to remove the subjective nature of the hydrograph separation. The two routes for calculating recharge also differ in the manner in which interflow is calculated. The hydrograph separation technique calculates interflow as the residual total baseflow after groundwater baseflow has been deducted. The time-series of Pitman S-values technique calculates the groundwater baseflow and interflow without the catchment hydrograph separation volume. The calculation of interflow under virgin and modified conditions is covered in the Sami model methodology, but the independent calculation of interflow if the time-series Pitman S-values were used is not covered. Equation 2.16 could be used in the Pitman S method, but DWAF (2006a) does not state where the total baseflow value would be obtained from.

The groundwater abstraction equation in the Sami model is an analytical solution of the Glover and Balmer (1954) streamflow depletion method. The Glover and Balmer method is useful, but has a number of restrictive assumptions. The Sami model equation only indicates relative amounts of abstraction influence on the two available storages (groundwater storage and groundwater baseflow). This relative determination implies that in order to determine the quantitative amount of groundwater that can be abstracted would require the setting of an acceptable influence on the storages. The simplified water balance approach using averaged catchment parameters, results in the model not being able to accurately quantify the surface-groundwater interactions for single abstraction points.

Several shortcomings were found by Witthüser (2006) when reviewing the Sami model. Each quaternary catchment in the Sami model is discretized into two compartments allowing only vertical flow between the two layers instead of additional horizontal lagging of water movement as seen in other models such as the SHE model (Witthüser, 2006). Additionally, the global nature of the parameters is a limitation leading to Witthüser (2006) not recommending the use of the Sami model. Moseki (2013) also reviewed the Sami model and did not recommend the model for use in South Africa, as the model had not been validated for fractured-rock aquifers. Moseki (2013) also stated the model as a work in progress because it has the potential to make a valuable contribution, but the initial success of the pilot test in the Schoonspruit Catchment (Mare et al. (2007), cited by Moseki, 2013) has not been replicated in subsequent tests using different datasets and/or study areas.

In conclusion, the Sami model is able to simulate the interactions between surface and groundwater, and the effects of abstraction in a realistic manner for the quaternary catchment scale. However, Levy and Xu (2011) report on Seward *et al.* (2006) and Xu *et al.* (2002) suggesting that the estimation of groundwater discharge rates on a regional scale might not be an acceptable approach towards implementing the National Water Act (1998). The groundwater baseflow determination does indeed meet the initial criteria set by DWAF (2006a) of overcoming the problem of constant surface water head, inherent in the MODFLOW program, by allowing the groundwater and surface water heads to differ from month to month, and taking into account the hydraulic resistance of the river bed by incorporating the natural log function into the estimation of groundwater baseflow. The groundwater storage used in the calculation of groundwater baseflow is influenced by the

uncertainties associated with either the hydrograph separation and Pitman runoff-soil moisture relationship value or the times series Pitman S-values from WRSM2000 and the calculated initial groundwater storage, recharge, evapotranspiration, and groundwater outflow volumes. Hughes, Kapangaziwiri and Barker (2010) suggest that the overall uncertainty associated with the Sami model is more likely due to the estimation of parameters using the scarce data available in South Africa rather than the model structure. However, the model structure is not adequate for detailed groundwater investigations.

2.5.2. The Hughes Model

Overview

The procedure for determining the groundwater component of the ecological reserve is not as well established as the river component, in South Africa (Hughes, 2004). The main reason for this is the lack of quantitative information regarding the contribution of groundwater to surface water. Hughes (2004) state that the problem with quantifying the groundwater contribution to streamflow is in finding a method which is able to estimate recharge and groundwater discharge from available data, while allowing the integration with a surface water estimation approach which would be acceptable for both hydrologists and geohydrologists in South Africa. The widely-accepted Pitman model was determined by Hughes (2004) as a reasonable starting point, as the model is extensively utilized for the simulation of stream runoff. Hughes (2004) set to solve the problem of quantifying the groundwater contribution to streamflow by incorporating a recharge and groundwater discharge component into the existing Pitman model. The various components of the Pitman model as well as the components incorporated by Hughes (2004) are discussed in detail.

Pitman Model

The Pitman model has two main inputs, monthly precipitation expressed as a mean annual percentage and monthly potential evapotranspiration. Additional compulsory data includes basin area, a time series of basin average rainfall, seasonal distributions of evaporation, irrigation water demand, other water demands and monthly parameter distribution factors. Optional input data consists of time series basin average potential evaporation, upstream inflow and transfer inflow. The original Pitman model flow diagram and structure is illustrated

in Figure 2-5-6. Following this flow diagram the main algorithms or functions used to simulate the flow are described (Hughes, 2004).

Rainfall distributed function (RDF)

The input precipitation data is distributed by the rainfall-distributed function over time, using the RDF parameter in SPATSIM, which uses a cumulative mass curving over four iterations. The lower the RDF parameter the more evenly distributed the rainfall will be represented in the model. See Table 2-5-2 for a full list of parameters used in the Pitman model (Hughes, 2004).

Interception function

The Interception function is based on the parameter *PI*, which has the ability to vary seasonally and allows for two different vegetation types to be defined. The depth or amount of rainfall intercepted is calculated based on an empirical relationship between the set parameter *PI* and the monthly rainfall depth or amount. The amount of water subtracted from the total rainfall to account for interception (interception storage) is used to satisfy the potential evaporation rate. The rain water which is not intercepted then forms the input into the surface water runoff function (Hughes, 2004).

Surface Runoff function

The Pitman model allows for a parameter *AI* to be set, which represents the portion of the basin that is impervious, allowing for surface runoff to be directly generated from this portion of the basin (Hughes, 2004). The infiltration of precipitation on the remaining pervious section of the basin is calculated by means of the Surface Runoff function. The Surface Runoff function was originally a symmetrical triangular distribution of basin absorption rates based on a minimum (*ZMIN*) and a maximum (*ZMAX*) absorption rate. The amount of rainfall contributing to surface runoff is represented by the area under the triangle between *ZMIN* and the rainfall rate. In the SPATSIM version of the Pitman model, a parameter *ZAVE* has been incorporated to permit an asymmetrical triangle, allowing for a distribution of runoff (Hughes, 2004).

Soil Moisture Storage and Runoff function

The precipitation that has not been intercepted or contributed to surface runoff will increment the soil moisture storage (*S*). If the soil moisture storage exceeds the maximum soil

moisture storage capacity (ST) the surplus becomes runoff from the Upper zone (Figure 2-5-6). Runoff from soil moisture storage is controlled through a non-linear relationship between runoff and storage through a parameter POW , controlling the power function of the relationship (Hughes, 2004; DWAF, 2006a):

$$Q = FT \left(\frac{S-S_L}{S_T-S_L} \right)^{POW} \quad (2.20)$$

where,

FT is the runoff rate at full storage (mm/month) i.e. ST is exceeded,

S is the soil moisture storage (mm),

ST is the maximum soil moisture storage capacity,

SL is the minimum soil moisture storage below which no soil moisture runoff occurs, and

POW is the relationship between the runoff and soil moisture storage.

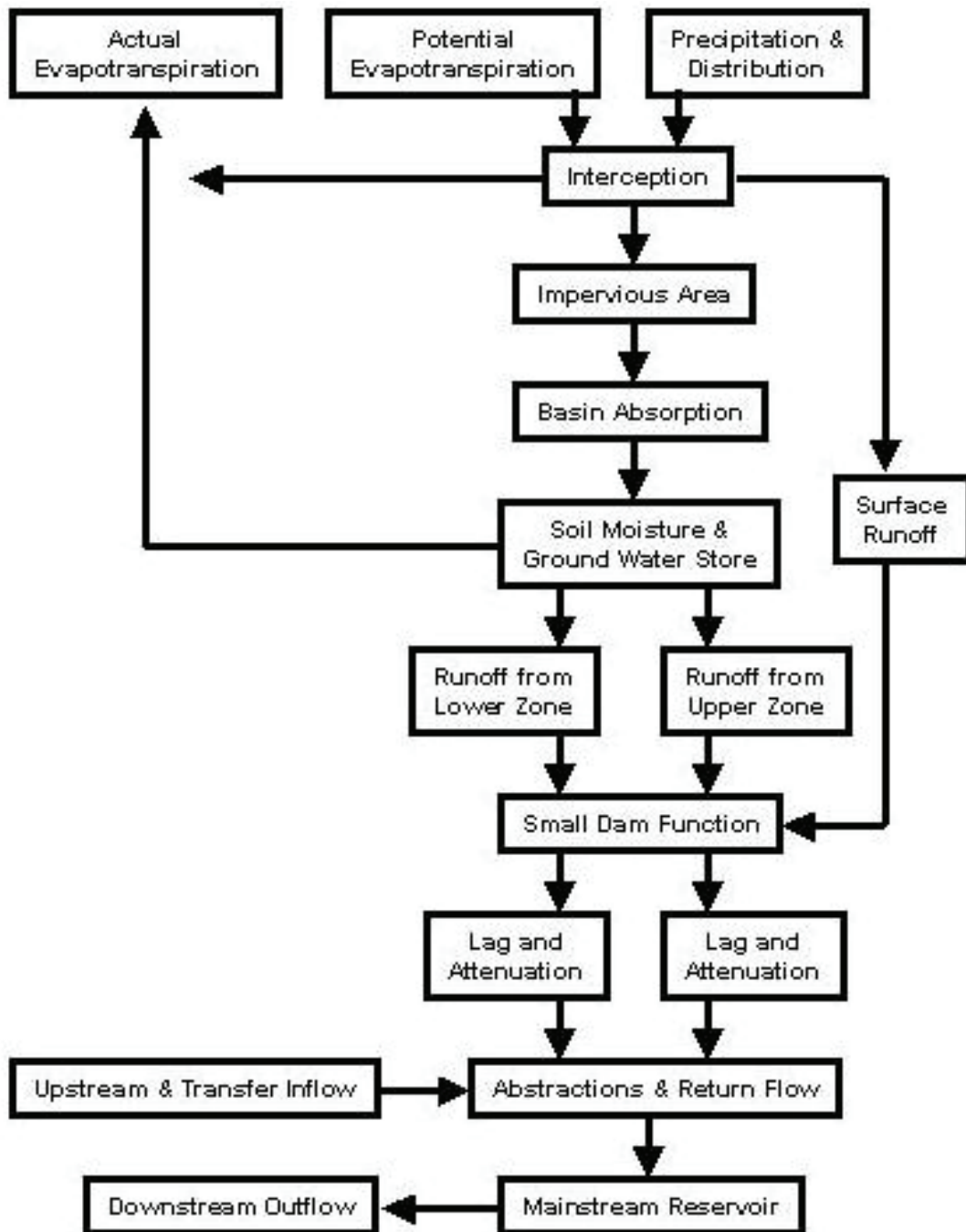


Figure 2-5-6 The basic structure and flow distribution of the Pitman model (Taken from Hughes, 2004).

The runoff from groundwater is determined by a parameter GW , which is the maximum groundwater runoff, but Hughes (1997) highlights that there is no theoretical background to set this parameter value.

Evaporation from Soil Moisture Storage function

The Evaporation function is based on the parameter R ($0 < R < 1$) and the potential evaporation volume. A low R value indicates an effective evaporation loss which will continue to take place even at low soil moisture storage levels, where a high R value will indicate an

evaporative loss which will cease at a higher soil moisture storage level. The value of R can be related to vegetation types, in that a low R value would indicate deeper-rooted vegetation. The model also allows for different rates of evaporation to be set by means of the parameter FF , which scales the potential evaporation for certain areas in order to consider different vegetation types within the basin.

Runoff Delays and Lags function

The runoff from the Upper zone (ST is exceeded), and the runoff from the Lower zone (groundwater) are lagged at different rates, controlled by parameters TL and GL respectively. The runoff considered to be groundwater by the model is lagged longer than the remaining runoff from soil moisture storage using the Muskingum equation (Hughes, 2004).

Artificial Modification functions

There are a number of additional components of the model which allow artificial modifications to the hydrological system to be simulated. These include a small dam routine, direct abstraction from river water and return flow functions.

Table 2-5-2 A list and description of all parameters in the original Pitman model (Taken from Hughes, 2004).

Table 1 Pitman model parameters.		
Parameter	Units	Description
RDF		Rainfall distribution factor. Controls the distribution of total monthly rainfall over four model iterations
AI	Fract	Impervious fraction of sub-basin
PI1 and PI2	mm	Interception storage for two vegetation types
AFOR %	%	area of sub-basin under vegetation type 2
FF		Ratio of potential evaporation rate for Veg2 relative to Veg1
PEVAP	mm	Annual basin potential evaporation
ZMIN	mm month ⁻¹	Minimum basin absorption rate
ZAVE	mm month ⁻¹	Mean basin absorption rate
ZMAX	mm month ⁻¹	Maximum basin absorption rate
ST	mm	Maximum moisture storage capacity
SL	mm	Minimum moisture storage below which no runoff occurs
POW		Power of the moisture storage-runoff equation
FT	mm month ⁻¹	Runoff from moisture storage at full capacity (ST)
GW	mm month ⁻¹	Maximum runoff from groundwater
R		Evaporation-moisture storage relationship parameter
TL and GL	months	Lag of runoff (surface and groundwater respectively)
AIRR	km ²	Irrigation area
IWR	Fract	Irrigation water return flow fraction
EFFECT	Fract	Effective rainfall fraction
RUSE	m ³ × 106 year ⁻¹	Non-irrigation demand from the river
MDAM	m ³ × 106	Small dam storage capacity
DAREA %	%	sub-basin above dams
A and B		Parameters in non-linear dam area-volume relationship
IRRIG	km ²	Irrigation area from small dams

Hughes Components

The Pitman model has been re-coded by both the original author and others, resulting in a number of subsequent versions and additional components. However, the basic form of the Pitman model has been preserved. Hughes (2004) has incorporated two new functions into the Pitman model in order to more efficiently quantify the interaction taking place between groundwater and surface water. These two new components consist of a Recharge function and a Groundwater Discharge function.

Recharge function

The new recharge function is based on the assumption that recharge will occur where there are rock outcrops and that the surface characteristics of the area can be represented by a single storage parameter. By defining a parameter below which no recharge will occur (when soil moisture capacity has been reached), the depth of recharge can be estimated by a non-linear relationship between the current storage and the maximum storage (Hughes, 2004). Hughes (2004) adapts the original Pitman soil moisture-runoff relationship (Equation 2.20) to now estimate recharge by redefining the SL , GW and POW parameters. SL is redefined as the soil moisture threshold below which no recharge occurs, GW is redefined as the maximum amount of recharge, and POW is redefined as $GPOW$ and now represents the relationship between recharge and current storage (S). Figure 2-5-7 is a graph of the two different power parameters, POW and $GPOW$ indicating their relative differences (Hughes, 2004).

The newly defined recharge-soil moisture function:

$$RE = GW \left(\frac{S - S_L}{S_T - S_L} \right)^{GPOW} \quad (2.21)$$

where,

RE is the estimated recharge depth or amount (mm),
 GW is the maximum rate of recharge (mm/month), and
 $GPOW$ is the new relationship between recharge and soil moisture storage.

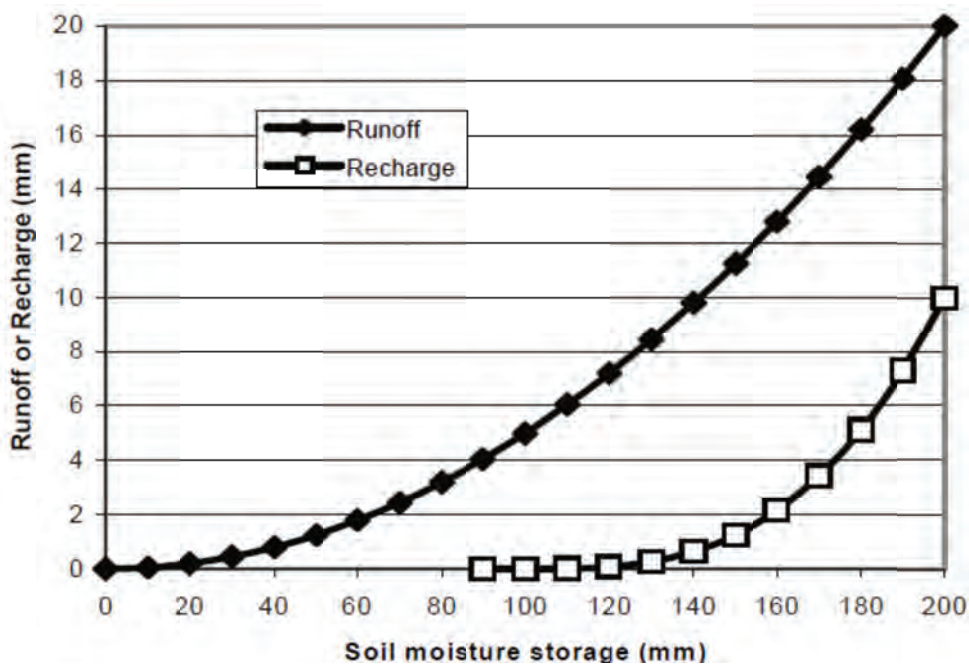


Figure 2-5-7 A graphical representation of the original soil moisture-runoff relationship (POW) and the redefined recharge-moisture relationship ($GPOW$) (Taken from Hughes, 2004).

Groundwater Discharge function

The Groundwater Discharge function aims to reduce the complexity of the spatial geometry of the basin to apply simple groundwater discharge principals. Hughes (2004) states the first step is to represent the basin as a square and the rivers as parallel lines separated by drainage slopes (Figure 2-5-8). The discharge is considered one-dimensional to further simplify the system. The number, length and width of the separating drainage slopes as well as the effective drainage density can thus be calculated from the modified basin area. However, the drainage density is a model parameter which can be deduced from maps and an understanding of the basin and channels (Hughes, 2004).

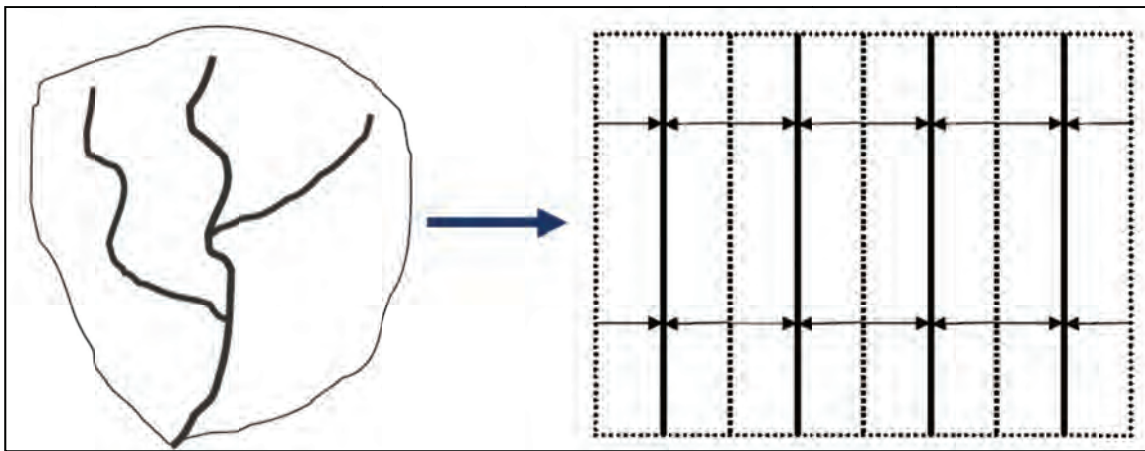


Figure 2-5-8 Conceptual simplification of a drainage basin as a square and rivers as parallel lines separated by drainage slopes. Solid lines are rivers, dotted lines are drainage divides and arrows indicate the direction of drainage on the eight drainage slopes (Taken from Hughes, 2004).

The number of channel lines is calculated first (Hughes, 2004):

$$\text{Total channel length} = \text{Drainage Density} \times \text{Area}$$

From Figure 3-11, it follows that:

$$\text{Total channel length} = \text{No. drainage slopes} \times \text{Area}$$

and,

$$\text{No. drainage slopes} = \text{Integer even value of} \left[\text{Drainage density} \times \frac{\sqrt{(\text{Area})}}{2} \right]$$

where,

$$\text{Drainage slope width} = \frac{\sqrt{(\text{Area})}}{\text{No. of drainage slopes}}$$

The volume of the “wedge” of groundwater stored under the drainage slope, as shown in Figure 2-5-9 for a single drainage slope, can be calculated as follows if the lower boundary is the river at the bottom of the slope (Hughes, 2004):

$$\text{Wedge volume} = (\text{Drainage width})^2 \times \text{Gradient} \times \frac{\text{Drainage length}}{2}$$

and,

$$\text{Volume of water in wedge} = \text{wedge volume} \times \text{Storativity}$$

Finally, outflows from this wedge representing the groundwater discharge to a river within a single slope element, can be calculated as (Hughes, 2004):

$$\text{Discharge} = \text{Transmissivity} \times \text{Gradient} \times \text{Time step} \times \text{Channel length}$$

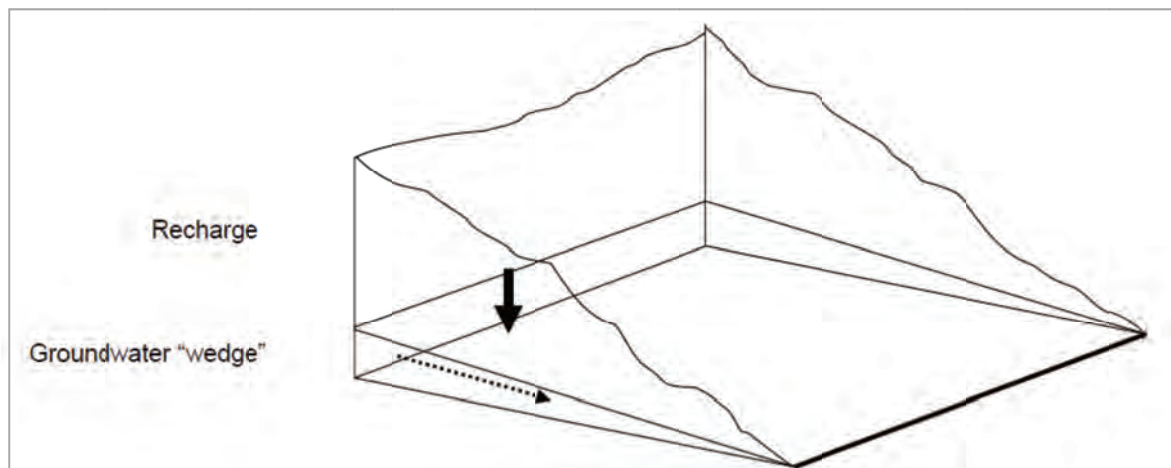


Figure 2-5-9 A single drainage slope element with the corresponding “wedge” representing the groundwater body that is above the conceptual river (Taken from Hughes, 2004).

Hughes (2004) performed several checks to ensure that the water balance was still achieved after the modifications to the Pitman model were done. The modified Pitman model is referred to as the Hughes model. The Hughes model was applied to two river basins in South Africa to test the modifications. The river basin examples showed that the modifications were a small improvement on the results from the original model. Hughes (2004) includes that from a perspective of representing processes involved in the runoff generation, the modifications and the modified parameter set are considered an improvement, but there is insufficient field data to confirm the runoff generation processes.

Review

The representation of the investigated basin as a square and the corresponding geometric representation of groundwater flow towards a river is an extremely simplistic view and ignores the complexities associated with groundwater flow. Hughes (2004) argues that the representation is sufficient as the calculations used are simple geometric equations. Following the new groundwater discharge function there is no longer a need to lag the groundwater component or Lower zone runoff, which renders the parameter GL as well as the lag routine unnecessary.

It is noted by Hughes (2004) that the model formulation would not be appropriate where groundwater flow did not follow the surface water flow. The fact that groundwater flow does not follow the surface water derived quaternary catchments in certain areas has been a contentious issue regarding groundwater resource assessments, but due to the fact that groundwater water cannot be directly observed there is not enough information to identify these locations. There are a number of situations listed by Hughes (2004) where the current modified version of the Pitman model application is not recommended. These situations include groundwater abstractions, evaporative losses of groundwater discharge from riparian areas, groundwater discharge to aquifer compartments in adjacent sub-basins and where the groundwater level is below the river level.

Hughes (2004) conclude that it is not suggested that the new components incorporated into the Pitman model are a completely realistic representation of groundwater flow to a river, but they are still effective and the parameters should be quantifiable from currently available data. The problem with these simplifications is, at what point the increasing degree of simplification will begin to misrepresent the groundwater flow processes occurring within a basin.

2.6. Discussion of Surface water-groundwater Interaction methods

International SW-GW Interaction methods

There is a trend in international methods of quantifying surface water-groundwater interaction towards numerical modelling solutions, as seen in the United States of America (USA), United Kingdom (UK) and Australia. However, a great expense accompanies the use of these numerical solutions as seen in the extensive monitoring infrastructure in the UK (Environment Agency, 2005a). There are a number of site specific study which have been done (Oxtobee and Novakowski (2002); Environment Agency (2005b); Allen *et al.* (2010); Tetzlaff and Soulsby (2008); Mencio and Mas-Pla (2008); Lapworth *et al.* (2009)), but Allen *et al.* (2010) conclude that the groundwater-surface water interactions can still only be partially understood even after extensive study at a site scale due to the inherent heterogeneity of the groundwater system. The advantages and disadvantages of each method are different and it has been suggested by numerous authors that a combination of methods would result in the best estimate of groundwater-surface water interaction (Environment Agency (2005b); Allen *et al.* (2010); Oxtobee and Novakowski (2002); Rosenberry and LaBaugh (2008)).

Hydrograph separations are extensively used all over the world to determine surface water-groundwater interactions because stream flow records are widely available. However, the use of a hydrograph separation technique alone has been highlighted as a poor determination method for the groundwater component of baseflow (Halford and Mayer (2000); Wittenberg and Sivapalan (1999); Xu *et al.* (2002)). A number of improvement techniques have been suggested for the hydrograph separation techniques, along with the incorporation of chemical data (Australian Government (2012a)).

National SW-GW Interactions methods

In South Africa there is a clear trend towards the use of hydrological models (Pitman model, Hughes model, Sami model) for the estimation of groundwater-surface water interaction at a catchment scale, while hydrograph separation techniques are seen to be favoured for local scale investigations. There are guidelines for selecting a method that is more suitable for specific investigation or area, but comprehensive governmental guidelines specifically aimed at the quantification of the surface water-groundwater interaction are lacking.

2.7. Mixing Cell Model methodologies

2.7.1. Historical applications

The earliest mention of the mixing cell concept in relation to a hydrological system appears to be when Wentworth (1948) suggested the use of an array of cells with perfect mixing to explain a transition zone at a moving interface between fresh and salt water in a coastal aquifer. The use of mixing cell models and discrete reservoirs to model hydrologic systems has continued for decades, seen in the use by Craig (1957), Dooge (1959), Eriksson (1971), Simpson (1988), Harrington *et al.* (1998) and Partington *et al.* (2010). The mixing cell model has also been extensively used within the chemical engineering field to investigate the movement of and chemical changes within and among reactor vessels. The names given to the mixing cell model within the chemical engineering field are varied, where Levenspiel (1972) refers to tank-in-series models, Deans and Lapidus (1960) refer to finite-stage models and Himmelblau and Bischoff (1968) refer to population-balance models. These models differ slightly in algorithms used and assumptions, but all make use of a cell of some sort as the basic sub-division of the system. The chemical engineering models differ to the hydrologic models in that they are applied on different scales (Campana, 1975).

Limitations on some of the earlier mixing cell models were highlighted by Campana (1975). Mixing cells models developed by chemical engineers allow for complex network configurations, but real-world hydrologic systems would not conform to the models geometric configurations due to scale differences. Earlier models also limited the user to a specific number of input and outputs that could be utilized per cell.

The Discrete State Compartment (DSC) model, developed by Simpson in 1972, overcame these limitations in that it consists of a set of interconnected cells of any desired size through which the transport of an incompressible fluid and dissolved matter is represented by a sequence of finite states and in theory these states could assume an infinite number of values. The model obtains solutions by iterating a recursive equation derived from the continuity equation and fluid and tracer transport can be modelled simultaneously (Campana, 1975).

The basic equation for each cell in an assigned network of a DSC model is (Simpson, 1973):

$$S(N + 1) = S(N) + (BRV(N + 1) \cdot BRC(N + 1)) - (BDV(N + 1) \cdot BDC(N + 1)) \pm R(N + 1) \quad (2.22)$$

where

$S(N+1)$ is the cell state or amount of substance in cell at iteration $N+1$,

$S(N)$ is the cell state or amount of substance in cell at iteration N ,

N is the iteration number,

$BRV(N+1)$ is the boundary recharge volume at iteration $N+1$,

$BRC(N+1)$ is the boundary recharge concentration at iteration $N+1$,

$BDV(N+1)$ is the boundary discharge volume at iteration $N+1$,

$BDC(N+1)$ is the boundary discharge concentration at iteration $N+1$, and

$R(N+1)$ is the source/sink term for iteration $N+1$.

Equation 2.22 is a discrete form of the continuity equation and states that the amount of a substance in a cell at iteration step $N+1$ will equal the amount of substance in the cell at iteration step N , plus the amount that entered the cell at iteration $N+1$, minus the amount that leaves the cell, plus or minus any amount that was added from the external environment or subtracted from the cell to the external environment.

Two different algorithms were described by Simpson (1973), namely the Simple Mixing Cell (SMC) and the Modified Mixing Cell (MMC). The SMC is equivalent to the conventional mixing cell, while the MMC is somewhere between the perfect mixing of the SMC and pure piston flow (displacement only, no mixing). The SMC is also described as the “in-mix-out” algorithm in that the inflow tracer mixes with the cell contents and is then discharged, whereas the MMC is described as the “in-out-mix” algorithm in that the cell discharges before the inflow tracer mixes with the cell contents (Campana, 1975). The finite-state mixing cell model was unique due to the fact that it modelled mass transport in an aquifer system without the usual requirement of a dispersion coefficient and it is not a black box model (Campana, 1975).

Woolhiser *et al.*, (1982) developed a method to quantify the inflows from several sources to a stream reach. The method requires that the chemical characteristics of the inflows are known and further assumes that water moving through a unique environment will have a characteristic chemical composition. Thus, the water in the stream would consist of a mixture of water from the different sources, with each representing a unique environment. Pinder and Jones (1969) estimated the proportion of stream discharge from groundwater and surface water using the differences in Total Dissolved Salts (TDS) content between the two.

This method is limited to two sources and is thus of limited use. Visocky (1970) and Hall (1970) also made use of the differences in chemical composition of different sources.

Mathematically the Woolhiser *et al.* (1982) method consists of a water mass balance equation and a mass balance equation for each of the selected ionic species, which are each equated to an error term. The unknown inflow rates are then estimated by minimizing the square percentage errors in each of the mass balance equations using quadratic programming. The method has inherent assumptions, of which the most important is that each ionic species is conservative within the reach. Woolhiser *et al.* (1982) found that the method is less sensitive to errors in the chemical analyses if the concentration of each ion is divided by the concentration of that ion in the mixture, which is equivalent to minimizing the sum of squared percentage errors. Errors in estimates of a particular inflow are related to the proportion of the total ionic load contributed by that inflow relative to the total ionic load contributed to the river reach. If a significant inflow is absent from the calculation, the related error is shifted to the inflows with the most similar chemical composition (Woolhiser, 1982).

Another mixing cell model was developed by Adar (1984) to estimate the recharge rates from various sources into an aquifer by means of chemical and isotopic data. The model would be of the greatest use in areas with complex hydrogeological structures for which there is limited hydrologic information. The model's approach is a combination of two of the previously discussed mixing cell models, namely the Simpson (1973) and Woolhiser *et al.* (1982) models. The Adar model makes use of the idea by Woolhiser *et al.* (1982) for estimating unknown flows and the interconnected mixing cell concept from Simpson (1973). The model divides the investigated aquifer into mixing cells and mass balance equations are written for each cell expressing the conservation of water, dissolved chemical constituents and stable environmental isotopes. The mixing cell model, developed by Adar (1984), estimates recharge rates by simultaneously solving the mass balance equations using quadratic programming (Adar, 1984).

The use of calculated transient groundwater fluxes from MODFLOW as the input data to a Compartmental Mixing Cell (CMC) model to simulate the transport of hydrochemical and isotopic species in regional groundwater systems is described by Harrington *et al.* (1998). The main advantage of their integrated modelling approach is that quantitative estimates of aquifer processes can be obtained with greater confidence than if they were determined

using only one of the approaches. Harrington *et al.* also highlight that the main disadvantage of the CMC is the inherent assumption of complete mixing. Attempts have been made to allow for varying degrees of mixing within cells (Allison and Hughes, 1975), but this creates a large amount of parameterisation (Harrington *et al.*, 1998).

Partington *et al.* (2010) highlight the fact that tools for quantifying the groundwater component of streamflow are not readily available in the latest generation of fully integrated spatially distributed models. Partington *et al.* (2010) thus developed a Hydraulic Mixing Cell method (HMC) for quantifying the groundwater component of streamflow in fully integrated spatially distributed models. The mixing cell is based on the Modified Mixing Cell (MMC) developed by Campana (1975), but it differs in that it requires only hydraulic data.

2.7.2. Discussion of MCM applications

The popular trend in quantifying the groundwater component of streamflow is towards numerical modelling and more recent applications of the MCM have been found to follow the same trend, with the integration of mixing cell modes into numerical groundwater flow models. Harrington *et al.* (1988) and Partington *et al.* (2010) are examples of this and are both adequate methods for determining the groundwater component of streamflow. However, the data required to set up the models is substantial and often not available in certain areas of South Africa. Considering the fact that South Africa has large areas of data paucity, where it is not feasible to setup numerical models, the usage of the mixing cell model developed by Adar (1984) could be advantageous in that it only requires water quality and minimal flow data.

Hydrochemistry and environmental isotope data have been traditionally limited to a qualitative geohydrological tool, used to support or reject hypotheses on prevailing flow regimes. However, there is a great potential of this data in quantitative geohydrology. Adar (1984) states accordingly that valuable information may be lost by excluding hydrochemical and stable isotope data from a quantitative study. Harrington *et al.* (1998) state to increase the confidence in a model, the incorporation of additional information is required. Environmental tracers and isotopes have the potential to provide such additional data to facilitate such an increase in confidence. The Adar mixing cell model is simple enough to be applied to data scarce areas while the incorporation of environmental tracer data to a water balance can increase the confidence in the estimated fluxes.

2.8. Review of Groundwater Data

2.8.1. Data Paucity theme in South Africa

Adelana and MacDonald (2008) investigated the complexity and variety of issues surrounding the development of groundwater resources across Africa. Development of groundwater resources can reduce poverty and increase economic growth, but the sustainable development of the resource depends on an accurate understanding of the groundwater system and the availability of capable people to make the best, informed decisions. However, Adelana and MacDonald (2008) continue that despite the obvious need for groundwater data, there has been no specific attention paid to the systematic collection of groundwater data. The result is that groundwater data is unevenly distributed; knowledge is limited; and investment is often poorly targeted. According to Adelana and Meyer (2008), in order to strengthen groundwater management in Africa, there must be systematic collection of information on both the quality and quantity of groundwater resources and where the data allows groundwater models should be developed to test the understanding of that system.

Jonck and Meyer (2002) stated that importance of groundwater is rapidly increasing in South Africa, but the relevant groundwater information is not reaching planner, decision-makers and users in sufficient quantities. In 2002, the hydrogeological map series of the Republic of South Africa was completed. The countrywide groundwater characterisation programme was initiated in the 1990's to address this problem of insufficient groundwater knowledge. Today these maps are still in wide use and provide essential general hydrogeological information, but the data used to create these maps are now outdated. The database used to create the maps has grown which should provide the grounds to an update of these maps.

The Groundwater Resources Assessment Phase III (GRA3) process was initiated in 2008 (DWA, 2009). DWA (2009) highlights the main concern for the GRA3 project – the limited data currently available for groundwater resource assessment in South Africa. An international trend of growing recognition that numerical quantification and depiction of groundwater resources at a local scale is an important management tool was noted by DWA (2009). This approach however requires large amounts of good quality data. Thus, for the GRA3 project to build on the previous groundwater resource assessments as well as meet international trends,

groundwater collection and accuracy will need to be improved. This improvement in data collection and accuracy will most likely be the main challenge for the GRA3 project.

In a National Groundwater Strategy (2010) Newsletter titled *The Monitoring of Groundwater Levels*, the National Groundwater Strategy was quoted to motivate groundwater monitoring for assessment, planning and management. The motivation stated reliable groundwater data can only be achieved through continuous monitoring and only reliable groundwater data will allow for accurate assessment of the availability and abstraction potential of groundwater resources. NGS (2010) indicated that South Africa does have a national network of monitoring boreholes and a National Groundwater Database (NGDB), but states this is still inadequate for reliable groundwater resource assessments, especially on a quaternary catchment scale. In order to overcome this limitation, a culture of individual groundwater monitoring needs to be fostered and become an important target of the National Groundwater Strategy (NGS, 2010).

Better data collection and improved sharing of groundwater data are central to improving data quality, accessibility and exchange. Means by achieving these goals were given by Pietersen *et al.* (2011):

- Finalisation and expansion of the National Groundwater Archive (NGA),
- Registration of drillers and capturing of driller's groundwater data in public databases such as the NGA,
- Support water services institutions in the development and implementation of asset registers on groundwater infrastructure and monitoring of groundwater use and groundwater quality,
- Engage with relevant authorities to maintain hydrological and environmental monitoring programmes necessary for groundwater management; such as the rainfall monitoring systems maintained by the South African Weather Service,
- Develop and implement an integrated groundwater information system to support water services provision at municipal level. Improve the compatibility (and/or integration) of existing groundwater databases / information systems maintained by different institutions (including water quality databases and municipal groundwater asset registers), and

- Re-assess the funding required by DWA head and regional offices for groundwater monitoring, data capture as well as the operation and maintenance of groundwater infrastructure.

Pietersen *et al.* (2011) recommend an adaptive management approach to groundwater resources, whereas the collection of groundwater data increases, the technical and conceptual model of a system is refined, and the systematic incorporation of management interventions such as irrigation abstraction restrictions can be implemented (Pietersen *et al.*, 2011).

However, there are challenges to implementing sustainable and adaptive groundwater management in South Africa. In a study conducted by Knüppe (2011) where interviews were conducted with experts in order to determine the challenges impeding sustainable and efficient groundwater use in South Africa. Knüppe (2011) found that the development and future application of groundwater management tools is hindered by an insufficient appreciation of the resource, shortcomings in knowledge and information, centralised system structures and an inadequate recognition of the significance of aquifer-dependent ecosystems and services. Knüppe (2011) highlights that the lack of adequate knowledge and physical data on aquifer characteristics such as recharge, discharge, baseflow and aquifer-dependent ecosystems result in challenging and unpredictable resource planning and management. The lack of physical data can also be attributed to the misallocation of roles and responsibilities where monitoring data is incomplete and evaluation of data is lacking. The existing governance systems, and their performance, are characterised by knowledge gaps and a scarcity of the necessary data pertaining to aquifers.

Pietersen *et al.* (2012) evaluated the effectiveness of the existing governance provisions in South Africa as well as the capacity to implement adequate groundwater governance. The evaluation found that basic technical provisions such as hydrogeological maps and aquifer delineation were in place, but other governance provisions were weak or non-existent, such as (Pietersen *et al.* 2012):

- Groundwater monitoring is weak and assessment of groundwater resources is poor, both in terms of quantity and quality (e.g. lack of numerical groundwater model).

- There are fair provisions for water-well drilling and groundwater use rights but provisions to control groundwater abstraction and pollution are weak (poor compliance monitoring).
- Provisions for establishment of an aquifer-management organisation are non-existent.
- Cross-sector policy coordination is weak or non-existent.

National and local governments need to undertake coordinated and interdisciplinary action to improve the national monitoring network and implement an adaptive management strategy for groundwater resources. However, Knüppe (2011) concludes that moving from scientific knowledge to legislation and finally to implementation requires a paradigm shift in mindsets, and often this is not related to financial shortages or a lack of human capacity. Groundwater needs to be perceived as an integral part of water management and attributed a status equal to that of surface water (Knüppe, 2011).

2.8.2. Review of South African groundwater datasets

A brief review of groundwater datasets held or maintained by the Department of Water Affairs is provided below. Some of these datasets have been discontinued.

NGDB and NGA

The NGDB data set is the most comprehensive borehole data set in South Africa and incorporates an estimated 225 000 boreholes. However, the quality of the data is variable – e.g. earlier records held in the NGDB have estimated locations based on the cadastral farm name on which the borehole is found, leading to inaccuracies in position of up to several kilometres. Modern pumping rates are rarely available. Many boreholes have lengthy sets of monitoring data, recording water levels taken by DWA personnel annually or more frequently, whilst others consist of a single point and date. The NGDB has now been replaced by the online National Groundwater Archive (NGA). The NGA allows access via the internet and also allows users to upload borehole information directly, thus augmenting the database.

Water Management System (WMS)

The water quality database holds the most accurate point-source information. The WMS does not contain any borehole or aquifer information, but the quality of the chemical analyses results is excellent. It is not linked to the NGDB or other hydrogeological databases, but operates as a “stand alone” chemistry database. The WMS database resides at the Institute for Water Quality Studies (IWQS) within the Department of Water Affairs and Forestry.

WARMS Database

DWA also maintains the Water Authorisation and Use (WARMS) database, which gathers together information provided by licence holders on groundwater use (i.e. pumping rates). Municipalities also collect groundwater use information, and some of this is submitted to DWA for incorporation into WARMS. The WARMS database tends to record licensed groundwater pumping rates (as opposed to the actual rates verified in the field), but is nevertheless useful for measuring pumped quantities.

GRIP

The Groundwater Resource Information Project (GRIP) was first introduced in the Limpopo Region to collect all groundwater related data, verify this in the field, and present it to planners and engineers in a format that is sensible and easy to incorporate in planning studies (Botha, 2005). Groundwater Resource Information Projects (GRIPs) have also recently been introduced in the Eastern Cape and in Kwazulu-Natal. The overall aim of the GRIP is a properly developed and implemented groundwater information system to generate information and/or knowledge necessary for integration of groundwater into the holistic management of our water resources. The GRIP project has the following aims:

- To deliver borehole information and hydrological/geological information to all institutions involved in water supply and management to enable real-time decision-making for planning, operational and management purposes.
- To support the full implementation of regional and national groundwater databases. The data can be manipulated to develop a series of planning maps, develop site-specific water supply business plans and to assist planning engineers with groundwater development and management programmes.

- Effective dissemination of information through the development of a database accessible through the World Wide Web.
- To support future specialist groundwater studies addressing key priorities.

GRIP outcomes:

- The development and maintenance of a regional groundwater database.
- The development of groundwater potential and protection maps with supporting documentation.
- The development of maps and documentation in support of future resource development.
- The establishment of a Groundwater Service Centre with supporting information.

It is the intention of DWA that the GRIP data eventually be incorporated into the National Groundwater Archive. At present this process is hampered by a lack of resources. Most specialists agree that a GRIP (or something very similar) is a desirable thing to have in every province in South Africa. There is far less agreement on where funds and human resources are to come from to make this a reality. The GRIP project in the Eastern Cape has been badly hampered by lack of resources, and this calls into question whether it is realistic to expect GRIP to proceed in its current format across the country. GRIP is also administered by the regional DWA offices, which are themselves currently in a state of flux (transition to Catchment Management Agencies) and which lack a clear mandate for medium to long-term data collection. For a national GRIP to succeed, more support will need to be given to the programme. Implementation is the main challenge GRIP faces, together with the planning and resources that this implies. An advantage of the GRIP projects is that they bridge the gap between the national and regional scale.

2.8.3. Details on high confidence groundwater datasets

A summary of groundwater datasets introduced or augmented since the first GRA2 project is given in Table 2-8-1. This list is a work in progress, to be added to as more information becomes available.

Table 2-8-1 Details on groundwater datasets

Dataset	Availability	Notes	Comment on accuracy of the dataset
Limpopo GRIP	The DWA contact for GRIP Limpopo is Willem du Toit, Limpopo Regional Office. This data is currently available to the project team.	Started in about 2001. Is continuously updated. Currently contains about 23 000 entries. Was not integrated into original GRA2. Are compatibility issues between GRIP (recorded in Aquabase) and the NGA.	Reasonably accurate, since collected recently for specific purpose.
GRIP NW Province	No GRIP has yet been established in NW. However, in 2000 DWA obtained all available reports (about 200 reports) which were captured onto the NGDB and then migrated to NGA.	This dataset has been migrated to NGA. DWA is currently trying to collect reports generated since 2000	Likely to be fairly accurate.
GRIP E Cape Province	There was never a “stand-alone” GRIP for the Eastern Cape. Eastern Cape borehole data is available from DWA E Cape (possibly Jane Baron and Wendt Petzer)	Available data has been added to NGA	No new data collected?

GRIP KZN Province	Contact in the KZN Regional Office (Durban) is Ms Nthuthu Zingithwa, who has provided the project team with the data.	GRIP for KZN has not yet been added the NGA. Dataset consists of 31 073 entries at present, of which 8 535 have latest static water level entries.	Reasonably accurate, since collected recently for specific purpose.
GRIP Mpumalanga Province	Initial work has started on this. The groundwater consultancy AGES has a copy of this data. The project team has some of this data, obtained during work for the Gert Sibande District Municipality.	Was supposed to have been captured onto NGA directly. (Need to confirm this.)	Reasonably accurate, since collected recently for specific purpose.
DWA "Clearing House" Data	Get from E Cape DWA office in King Williams town. Ms Bathiswe Djantji (or assistants Thuthuzela or Hloni); 043 604 5545 or 082 806 1518; dyantiv@dwa.gov.za Currently in discussion with this office to obtain the data.	This dataset covers only the Eastern Cape province. Has details of water infrastructure, includes boreholes. Will be little detail on the boreholes, such as depth, Q, etc.?	Not accurate from a hydrogeological point of view, since the dataset was originally intended to be an infrastructure survey.
Blue Drop/Green Drop	Mariette Swart at DWA. Currently in discussion with Ms Swart re a meeting to view / obtain data.	May be the best indication of actual municipal groundwater use, and any current groundwater quality issues.	Accuracy likely to be highly variable, since depends on submissions from municipalities. Is at least recent, however.

Water Availability Assessment (WAA) studies	DWA website for reports	Umvoto did the WAA study for the W Cape	No new data collected?
All-Towns studies	DWA website for reports	No new groundwater data was collected for this project, but towns using groundwater were identified	No new data collected?
Internal Strategic Perspectives (ISPs)	Via DWA. Mainly on web, although we may be missing a few updates (e.g. Limpopo Province, done by Golder).	No new groundwater data was collected for this project. Need to summarise what each ISP says about groundwater.	No new data collected?
NGDB / NGA	Ernst Bertram/DWA NGA Support	The original GRA2 dataset was calculated based on 175 000 geosites in the NGDB. When the NGDB was closed in 2009 there were more than 251 000 geosites. As of July 2011 the NGA has a metadata list of 254 238 items.	Some highly accurate data, but mostly of moderate accuracy only. Some data is of considerable age.
Groundwater Master Plans	Available on DWA website. Team already has KZN, E. Cape and Mpumalanga.	No new data was collected for the Groundwater Master Plans. These probably need to be reviewed and summarised.	No new data collected?

In summary, the following “new” groundwater data (i.e. groundwater data collected or captured since the original GRA2 project) is available:

- Approximately 79 000 extra data points on the NGA
- The GRIP datasets for Limpopo, KZN and Mpumalanga
- Groundwater reports (likely to contain groundwater data) for NW Province, and other provinces.

Two further important sources of groundwater data are the municipalities and the private sector, both of whom collect groundwater data for their own purposes. Much of this data does not find its way onto the NGA or into the public domain, and is consequently difficult to obtain. Some of the municipal data finds its way into the Blue/Green Drop reports, even though the collection of groundwater data is not the main aim or function of this system.

3. METHODOLOGY

The mixing cell model and Tracer method applied for the study are presented within this chapter. The methodology of the mixing cell model developed by Adar (1984) is discussed in terms of the basic principal of the method, how this principal is mathematical applied, the software available and how the MCM has been slightly adapted for the quantification of groundwater-surface interactions. The methodology of the additional chemical hydrograph separation method (Tracer method) is briefly covered.

3.1. Mixing Cell Model

3.1.1. Basic Concept

The concept of a mixing cell is essentially based on the continuity equation. The one-dimensional continuity equation states the amount of inflow to a system will equate the amount of outflow with no change in storage, for the considered time step (Figure 3-1-1).

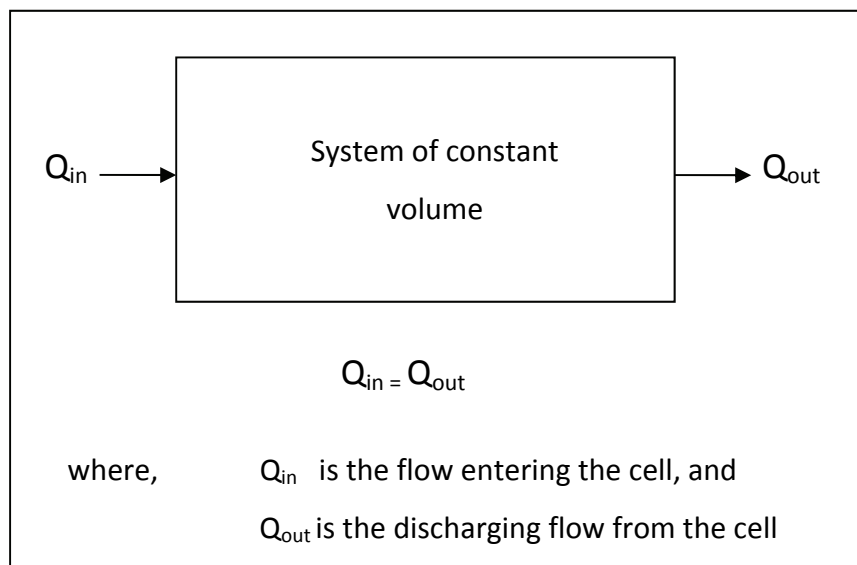


Figure 3-1-1 Basic principal of the mixing cell model

The mixing cell model builds on this foundation by sub-dividing a system into one or more mixing cells. A water balance equation is expressed for each cell to describe the movement into and out of the cells. The MCM requires that each of the inflows, present in the water balance equation, are chemically defined by a set of tracer concentrations. This water quality data is then used to describe a chemical mass balance equation for each cell. The chemical mass balance equation serves to constrain the water balance equation in order to produce

better estimates of the various unknown inflows to the system, than estimates made from the sole use of a water balance equation.

The main assumption of the mixing cell model is that any quantity or chemical property entering the cell is instantaneously dispersed throughout that cell. This implies that there is perfect mixing within the cell and the concentration of the entire cell and the discharging flow are the same. The “walls” of each mixing cell are assumed impermeable except for allocated connections to either an adjacent cell or the external environment (Campana, 1975).

3.1.2. Methodology

The mixing cell model developed by Adar (1984) was intended for the identification and quantification of multiple recharge sources, subsurface fluxes and physical aquifer parameters based on easily obtainable natural tracer concentration data.

The model relies on three types of conceptual models commonly applied in hydrology:

- 1) The evaluation of the motion of water and solutes with a multi-compartmental mixing cell model as suggested by Simpson (1988).
- 2) The solution of a set of water and dissolved constituents mass balance equations via a quadratic programming optimization scheme used by Woolhiser *et al.* (1982) and Adar (1984).
- 3) A mathematical model combining an inverse process to estimate compartmental conductances and storage coefficients distributed in a multi-compartmental model for a non-steady flow as described by Adar and Sorek (1989) and Adar (1996).

The application of the MCM to estimate the groundwater component of streamflow only makes use of the first two conceptual models because a steady-state approach is used and determining the physical parameters of the system is not a priority. The mathematical principals used to express and solve the mass balance equations for each cell are discussed below. The mathematical model presented here is taken from the “Quantitative evaluation of flow systems, groundwater recharge, and transmissivities using environmental tracers” paper by Adar (1996) presented in the *Manual on Mathematical models in isotope hydrogeology* and therefore not separately referenced.

The assumptions associated with this model as stated by Adar (1996):

- 1) Tracers are considered conservative, all reactions including dissolution and precipitation are considered negligible and the spatial change in the concentration of the solute is solely the result of dilution.
- 2) The assumption inherent in all mixing cell models is that any quantity entering a cell is instantaneously dispersed throughout the cell, implying complete mixing.
- 3) Seasonal pulsation of fluxes for each cell can be represented by mean values covering a time interval in which the hydraulic head may be regarded as a constant.
- 4) Transport of dissolved constituents is dominated by advective forces, i.e. the compartmental Peclet number is infinite.
- 5) Concentrations of solutes, which are constant within each cell over a specific time step, are measured and known together with the concentration of the same solute or tracer in the inflow and outflow components.
- 6) Flows entering the aquifer system are known qualitatively, where most of the source-sink and discharge flow components are known quantitatively and qualitatively.

A set of mass balance equations for the water and solute fluxes over a given time period are written for each cell. The water balance for a fluid with constant density within the n -th compartment (cell) is expressed as:

$$Q_n - W_n + \sum_{i=1}^{I_n} q_{in} - \sum_{j=1}^{J_n} q_{nj} = S_n \cdot \frac{dh_n}{dt} \quad (3.1)$$

where,

I_n is the number of sources which flow enters the n -th compartment,

J_n is the number of leaving flows from the n -th compartment,

q_{in} and q_{nj} is the the fluxes from the i -th source or compartment into the n -th one and from the n -th into the j -th cell, respectively

Q_n and W_n is the the fluid sources and sinks, respectively

S_n is the the storage capacity within cell n , and

h_n is the the hydraulic head associated with that compartment.

If one identifies two points in time (t_1 and t_2) where the hydraulic head is the same, for example at the beginning and end of a season, the total magnitude of the derivative (dh_n/dt) has not changed over that time period.

Hence, we obtain Equation 3.2:

$$\frac{1}{\tau} \int_{t_1}^{t_2} S_n \cdot \frac{dh_n}{dt} dt = 0 \quad (3.2)$$

By integrating the water balance (Equation 3.1) over the same time period, the following quasi steady-state equation is obtained:

$$\bar{Q}_n - \bar{W}_n + \sum_{i=1}^{I_n} \bar{q}_{in} - \sum_{j=1}^{J_n} \bar{q}_{nj} = 0 \quad (3.3)$$

All the parameters in Equation 3.3 have the same meaning as in the water balance (Equation 3.1), but now represent average values over the time period. A schematic diagram of a compartmental system is represented in Figure 3-1-2 to illustrate the flow parameters within the water balance equation (Equation 3.1).

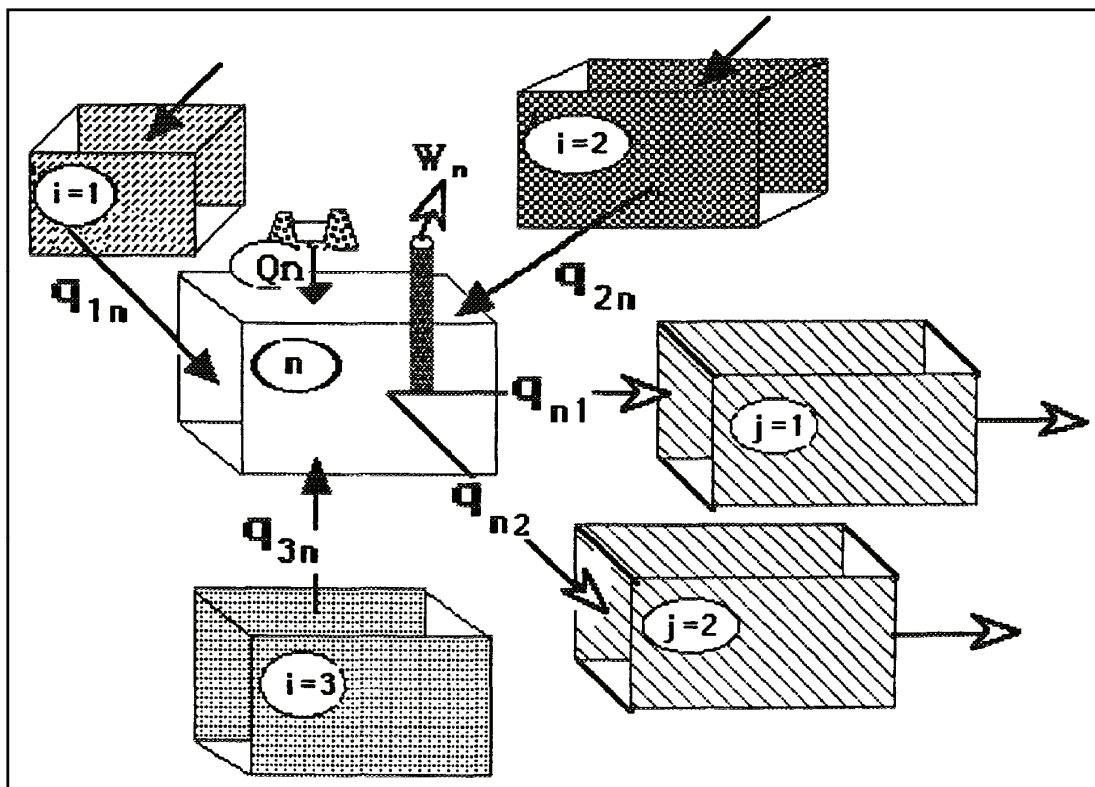


Figure 3-1-2 A schematic diagram explaining the parameters used in the mixing cell mathematical model. The parameter q indicates an unknown flux; i and j denote the direction of the flux, in and out respectively; Q_n denotes the known inflow to cell and W denotes the known out flow from the cell (Taken from Adar, 1993).

When the mixing cell concept is applied for quasi steady-state variations of concentrations and taking into account assumption 1 and Equation 3.3, the mass balance for a dissolved constituent k , in cell n is:

$$\tilde{C}_{nk}\bar{Q}_n - \bar{C}_{nk}[\bar{W}_n + \sum_{j=1}^{J_n} \bar{q}_{nj}] + \sum_{i=1}^{I_n} \bar{q}_{in} \bar{C}_{ink} = 0 \quad k = 1, 2, \dots, K \quad (3.4)$$

where,

\bar{C}_{ink} is the average concentration of solute k entering cell n with the incoming flux from cell i ,

\bar{C}_{nk} is the average concentration of the k -th constituent within cell n , and

\tilde{C}_{nk} is the average concentration of k associated with the source Q_n .

The mass balance equations, Equation 3.3 and Equation 3.4, should not be expected to close (without error) if real data is used, due to a number of possible sources of error. The water balance may not close due to the inadequacy of the assumptions that the annual change in storage is zero within the cell or errors in the measurement of Q_{nj} , C_{ink} and C_{nk} , or both. An error in identifying and measuring fluxes or rates of pumpage will also cause the water balance not to close. The chemical balance may not close due to one or more of the assumptions not holding for several of the solute species, or sampling and analytical errors. Mass balance errors may also be caused by incorrect quantifications of cell concentrations (Adar, 1984).

In order to account for the above mentioned inconsistencies, an error term is introduced to Equation 3.3 and Equation 3.4, to obtain Equation 3.5 and Equation 3.6, respectively:

$$\bar{Q}_n - \bar{W}_n + \sum_{i=1}^{I_n} \bar{q}_{in} - \sum_{j=1}^{J_n} \bar{q}_{nj} = e_n \quad (3.5)$$

$$\tilde{C}_{nk}\bar{Q}_n - \bar{C}_{nk}[\bar{W}_n + \sum_{j=1}^{J_n} \bar{q}_{nj}] + \sum_{i=1}^{I_n} \bar{q}_{in} \bar{C}_{ink} = e_{nk} \quad (3.6)$$

where,

e_n and e_{nk} are the deviations from the water and solute mass balances in cell n , respectively.

Equation 3.5 and Equation 3.6 are combined and expressed in matrix form:

$$\underline{C}_n \mathbf{q}_n + \underline{D}_n = \underline{E}_n \quad (3.7)$$

where \underline{C}_n is a matrix with known concentrations in cell n , of the form:

$$C_n = \begin{bmatrix} 1 & ,1 & , \dots & , -1 & , -1 & , \dots & , -1 \\ C_{1n1} & , C_{2n1} & , \dots & , C_{I_n n1} & , -C_{n1} & , \dots & , -C_{n1} \\ C_{1n2} & , C_{2n2} & , \dots & , C_{I_n n2} & , -C_{n2} & , \dots & , -C_{n2} \\ \vdots & \vdots & \vdots & \vdots & \vdots & \vdots & \vdots \\ \vdots & \vdots & \vdots & \vdots & \vdots & \vdots & \vdots \\ C_{1nK} & , C_{2nK} & , \dots & , C_{I_n nK} & , -C_{nK} & , \dots & , -C_{nK} \end{bmatrix} (K + 1)(I_n + J_n)$$

where the first row accounts for the water balance,
the other K rows express the solute mass balances,
 K is the total number of tracers used in the analysis,
– denotes the coefficients which are associated with outgoing fluxes, and
 C_{ink} is the concentration of the k -th species flowing into cell n from cell i .

\mathbf{q}_n is vector matrix of the unknown fluxes through the boundaries of cell n , of the form:

$$q_n = [\bar{q}_{1n} \quad , \bar{q}_{2n} \quad , \dots \quad , \bar{q}_{I_n n} \quad , \bar{q}_{n1} \quad , \bar{q}_{n2} \quad , \dots \quad , \bar{q}_{nJ_n}] (I_n + J_n) \cdot 1$$

\underline{D}_n is a vector containing elements that are measured and known quantitatively in cell n , of the form:

$$D_n = [(\bar{Q}_n - \bar{W}_n), \quad (\check{C}_{n1} \bar{Q}_n - C_{n1} \bar{W}_n), \quad (\check{C}_{n2} \bar{Q}_n - C_{n2} \bar{W}_n), \dots \quad , (\check{C}_{nK} \bar{Q}_n - C_{nK} \bar{W}_n)] (K + 1) \cdot 1$$

\underline{E}_n is the error vector matrix, of the form:

$$E_n = [e_n, \quad e_{n1}, \quad e_{n2}, \dots, \quad e_{nK}] (K + 1) \cdot 1$$

The unknown flux components in the aquifer can now be estimated by minimizing the square error sums J . By assembling the square error terms over all the cells the following equation is obtained:

$$\begin{aligned} J &= \sum_{n=1}^N (C_n q_n + D_n)^T W (C_n q_n + D_n) \\ &= \sum_{n=1}^N [E_n^T W E_n] \end{aligned} \quad (3.8)$$

where,

$()^T$ is a transpose matrix function, and

W denotes the diagonal matrix comprised of weighting values about estimated error expected for each of the terms building the mass balance for the fluid and the dissolved constituents.

The weighting matrix (W) can also reflect the level of confidence in a tracer's degree of conservation. The lower the confidence in the conservation of the tracer, the lower the weighting factor ($0 < W < 1$) assigned to that tracer. Adar (1996) states q_n is deconstructed into linear and non-linear components allowing the square error sums (J) to be minimized in order to estimate q_n . The quadratic optimization scheme used by Adar (1984) to minimize J was developed by Wolf (1967).

3.1.3. MCM Software

Adar and Küll (n.d.) developed an Excel® Add-in, based on the mathematical model presented by Adar (1984), for running a compartment or mixing cell model under in a steady state. The *Mixcel* software includes a special Mixing-cell Input Generator (MIG). The MIG serves to simplify the procedure of preparing data, as it quite literally builds an input file in Excel for the Wolf solver algorithm program which is also included in the *Mixcel* downloaded. The MIG computer code provided here is however restricted to a steady flow and steady hydrochemical flow system. The software can be downloaded free of charge from the website [<http://www.uhydro.de/doku/en/models/compartmentmodels>] and includes explanations and examples.

The latest version of the MCM developed by Adar (2012) is based in Microsoft® Access and was made available for this research project. Two codes were developed by Adar, one for steady state (MCMsf – A Mixing Cell Model for Steady Flow solver code) and one allowing for transient state (MCM_FTS – A Mixing Cell Model for Transient Flow solver code). The simpler steady state code was made use of to keep the data required and model complexity to a minimum. A step-by-step guide to using the MCMsf programme based on an example flow system is provided in Appendix A.

3.1.4. MCM adaption for the quantification of SW-GW Interactions

The Mixing Cell Model (MCM) developed by Adar (1984) is aimed at the identification and quantification of multiple recharge sources, subsurface fluxes and physical aquifer parameters, by means of representing the aquifer as mixing cells and assigning chemically-defined recharge sources to these cells to determine the recharge flux into that aquifer. To apply the MCM concept in quantifying the groundwater baseflow to a river, the main cell focus was changed from the aquifer to the river. The river section under investigation is

represented as mixing cells, instead of traditionally defining the aquifer as the main mixing cells. Chemically-defined groundwater inflow sources are then assigned to the river cell to determine the groundwater flux into that river.

The conceptual representation of the surface water-groundwater system for the application into the MCM is shown in Figure 3-1-3. The example quaternary catchment is conceptually represented as a single box model flow system, comprising of one mixing cell. The box model flow diagram then forms the basis for both the water and chemical mass balance equations. The main river within the area is presented by a single mixing cell, the River Cell (Figure 3-1-3). The River Cell is assumed to have a uniform chemical composition for the considered time step (the assumption appears reasonable if the different time scales of surface and groundwater flow are considered). Water quality data from within this river segment is used to chemically-define this River Cell in the MCM model. The upstream inflow into the river (IN), the three tributary inflows and the groundwater source along the river stretch are assigned chemical compositions from water quality data, so their flow rates to the River Cell can be estimated by the MCM. The conceptual model neglects evaporation from the stream and the adjacent channel aquifer as well as surface water use, but these parameters can be easily incorporated as constant loss term in a specific MCM application.

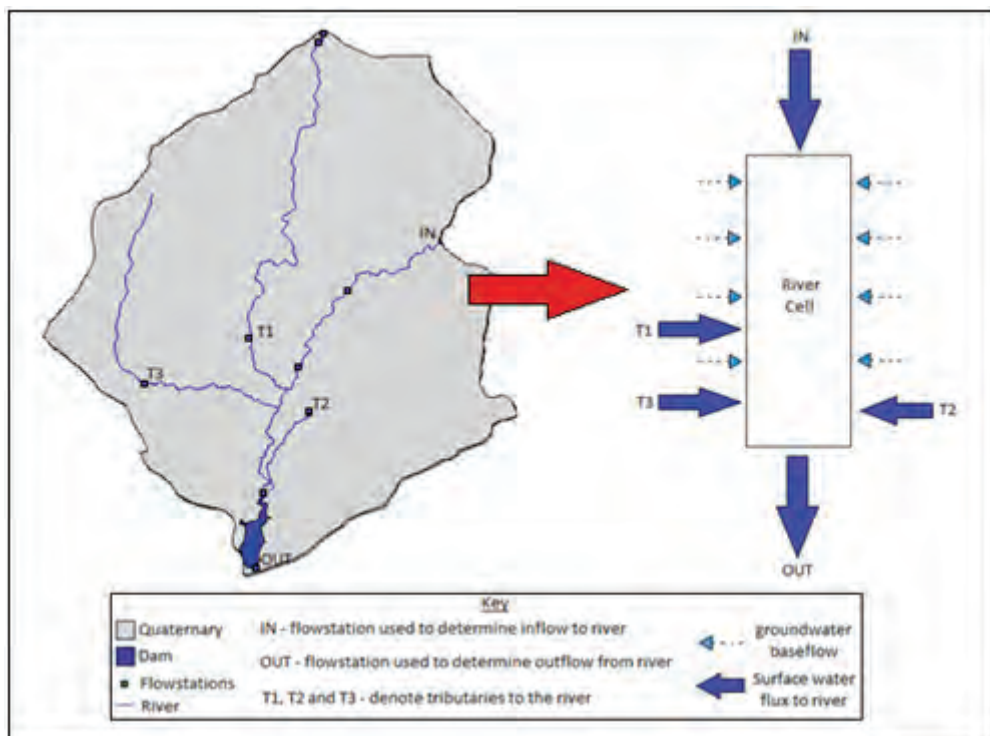


Figure 3-1-3 Conceptual representation of a catchment for the SW-GW interaction MCM application.

The water balance equation based on the box model flow representation of the example quaternary catchment shown in Figure 3-1-3, is:

$$Outflow(Q_{OUT}) = Inflow(Q_{IN}) + Tributary\ inflow(Q_T) + Groundwater\ inflow(Q_{GW})$$

The chemical mass balance equation based on the box model flow representation of the example quaternary catchment shown in Figure 3-1-3, is:

$$Q_{OUT} \cdot C_{OUT} = Q_{IN} \cdot C_{IN} + Q_T \cdot C_T + Q_{GW} \cdot C_{GW}$$

where,

C_{OUT} is the tracer concentration of the River Cell,

C_{IN} is the tracer concentration of the defined inflow,

C_T is the tracer concentration of the defined tributary inflow, and

C_{GW} is the tracer concentration of the defined groundwater source.

3.2. Tracer Method

3.2.1. Methodology

The Tracer method is a chemical hydrograph separation method described in an Australian Government document on surface water-groundwater interaction quantification methods (Australian Government, 2012a and 2012b). The method makes use of tracer concentrations for river flow, runoff and groundwater inflow and the total measured river flow at the point of investigation to determine the unknown groundwater discharge. The Tracer method assumes that the total outflow volume is made up of runoff and groundwater inflow, and the total tracer concentration at this point is controlled by the ratio of input from each of these two sources. The proportion of river flow interpreted as sourced from groundwater discharge is calculated using a chemical mass balance equation (Australian Government, 2012b):

$$\frac{Q_g}{Q_t} = \frac{c - c_r}{c_g - c_r} \quad (3.9)$$

where,

c , c_r and c_g are the tracer concentrations for the river, runoff and groundwater,

Q_t is the measured total river flow, and

Q_g is the volume of groundwater inflow.

The data collection of this method is similar to that of a traditional hydrograph separation method, where measurements of river water quality are made at regular time intervals along with the traditional flow measurements at a single point. A continuous record of flow and

water quality data allows for a continuous record of groundwater baseflow fluxes to be determined. The accuracy of the groundwater baseflow volumes determined by this method is dependent on the adequate differentiation of the source water and the assignment of the end-member concentrations, namely the river, runoff and groundwater tracer concentrations. The runoff end-member concentration can be defined by sampled rainfall or canopy throughfall. The runoff member could also be defined by the lowest tracer concentration measured in the river. Similarly, the groundwater end-member could be defined by the maximum tracer concentration measured in the river, but this is not recommended as the groundwater baseflow would tend to be over-estimated. The mean tracer concentration of groundwater sampled in boreholes is recommended for defining the groundwater end-member as it would give more accurate results (Australian Government, 2012a and 2012b).

The Tracer method can be used to determine the relative or volumetric proportion of groundwater inflow at any point along a river, where sufficient tracer data is available. The estimated groundwater baseflow at this point then includes the entire catchment upstream of this point. The method thus cannot provide specific information on where the calculated groundwater baseflow is occurring within the catchment. Considering this limitation the method would be best suited to smaller catchments were the groundwater baseflow volume could be more accurately determined (Australian Government, 2012a and 2012b).

4. PILOT STUDY OF THE MIXING CELL MODEL

The Mixing Cell Model (MCM) is applied to three study sites to investigate the applicability of the model to quantify the groundwater component of streamflow. The first study area is situated in the Free State Province, South Africa (Figure 4-1) and comprises quaternary catchments C52A-C52H; including the UFS surface water-groundwater interaction test site. The area is located on the edge of a regionally-defined zone of zero groundwater baseflow (Figure 4-2). The second study area is located in the Limpopo Province, South Africa and comprises quaternary catchments A42A-A42C, where calibrated Sami and Hughes model groundwater baseflow estimates are available for additional comparison and validation. The quaternary catchments are located within a regionally-defined high groundwater baseflow zone (Figure 4-2). The third study site is situated within the Northern Cape Province, South Africa and comprises of the quaternary catchment D73F, which falls within a semi-arid region (Figure 4-1). The quaternary catchment D73F is located in the middle of a regionally-defined zero groundwater baseflow zone (Figure 4-2).

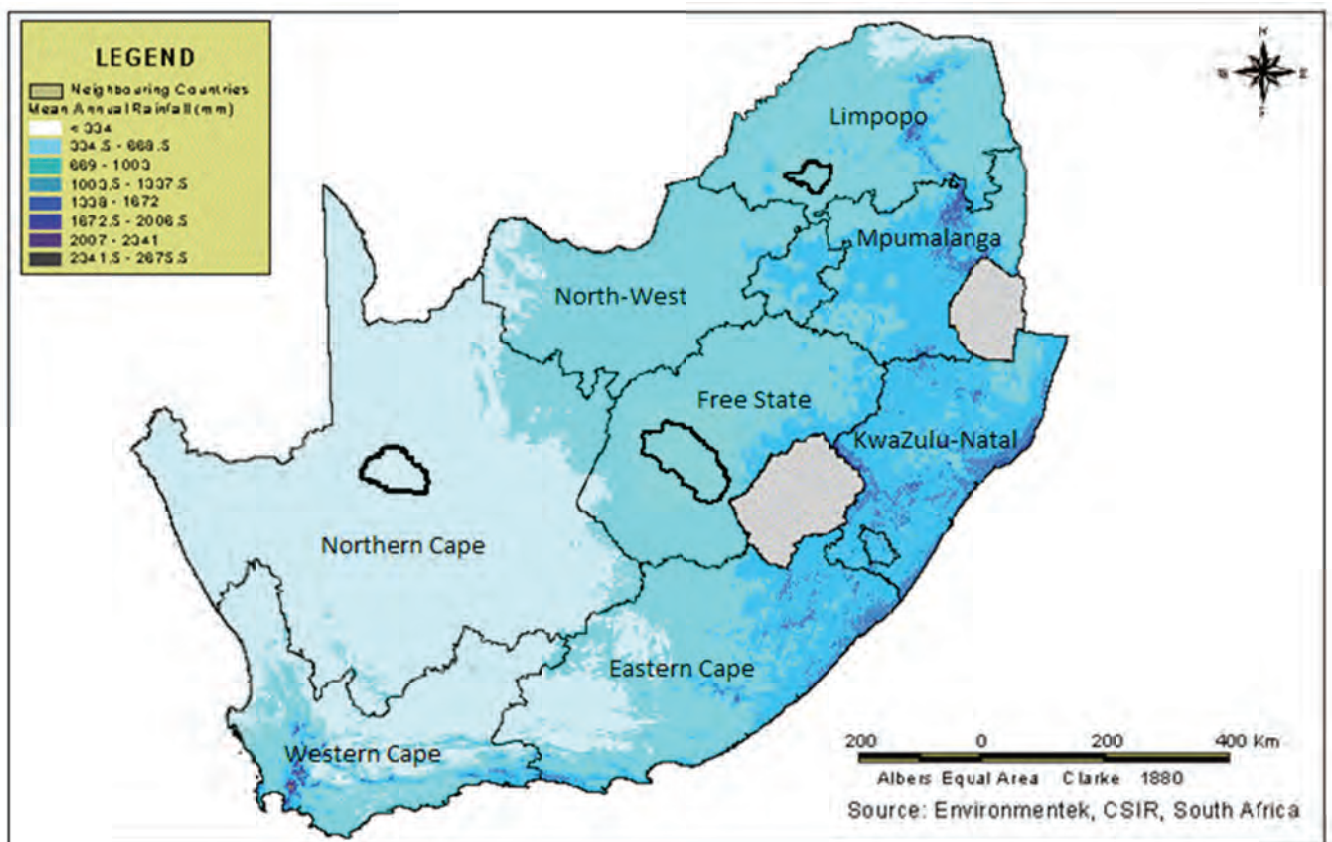


Figure 4-1 A rainfall distribution map of South Africa indicating the location of the three pilot study areas (Modified from Department of Environmental Affairs and Tourism, 1999).

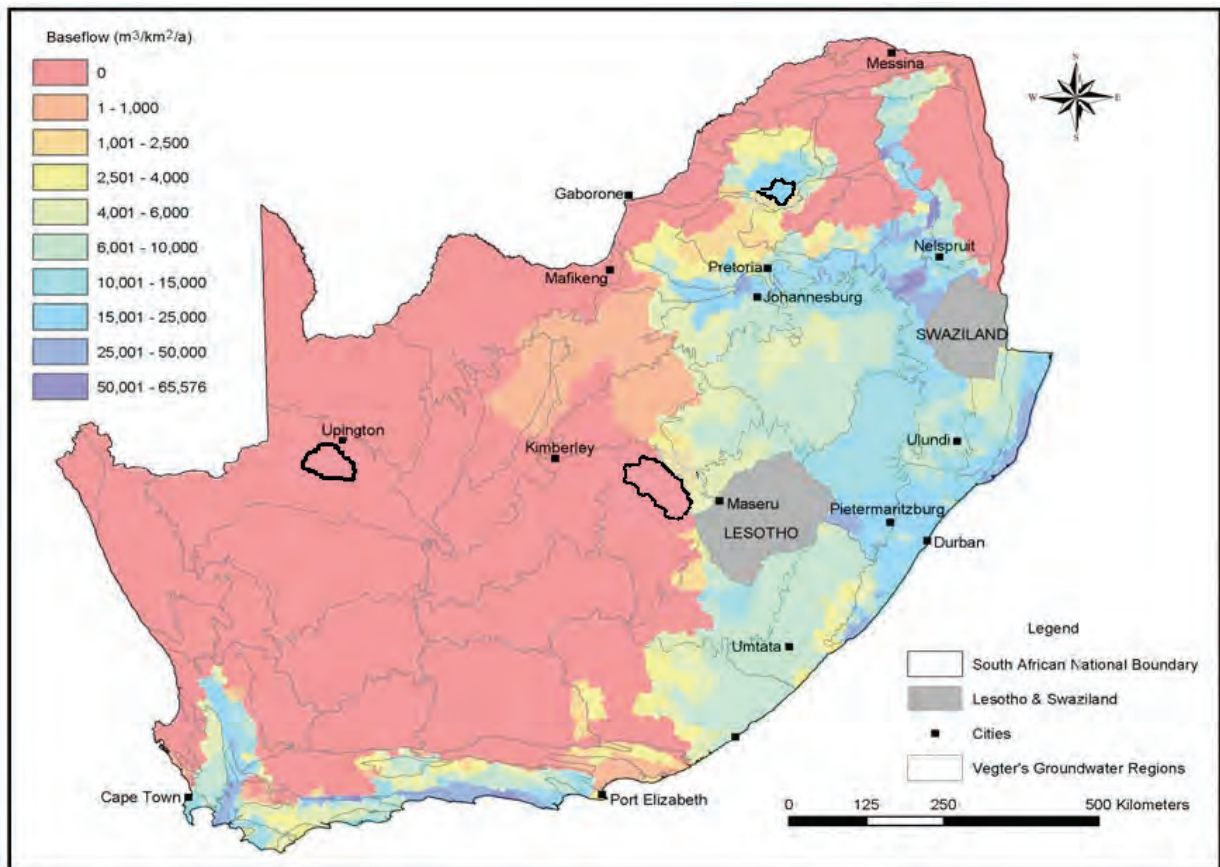


Figure 4-2 A map of South Africa indicating the GRA2 determined groundwater baseflow zones of South Africa, showing the location of the three pilot study areas (Modified from DWAF, 2006b).

4.1. Pilot Study 1: Free State quaternary catchments C52A-C52H

4.1.1. Overview

The Water Research Commission (WRC) and the University of the Free State (UFS) have established a surface water-groundwater interaction test site on the Modder River just outside of Bloemfontein at the base of the Krugersdrift Dam, South Africa (Figure 4-1-1). River water and groundwater quality samples collected along the middle Modder River, from the Rustfontein Dam to the base of this test site formed part of a further investigation aimed at a better understanding of the surface water-groundwater interaction taking place on a larger scale.

The middle reach of the Modder River is situated within the Upper Orange Water Management Area of the Free State Province, South Africa. The Modder River is a perennial river (Welderufael and Woyessa, 2010) and is an important source of water for domestic, agricultural and industrial use to Bloemfontein, Botshabelo and Thaba N'chu areas (Seaman *et al.* 2003). The Middle Modder River covers quaternaries C52B-C52G (Figure 4-1-1), with the

main tributaries being the Korannaspruit, Osspruit, Renosterspruit, Stinkhoutspruit and Doringspruit Rivers. The area has an arid to semi-arid climate, characterised by long periods of low rainfall events in the winter months and intense thunderstorms in the summer, rainy season which promote surface runoff. The average annual rainfall is estimated at 567 mm and the average annual evaporation is estimated at 1943 mm, based on data from meteorological stations at the Rustfontein Dam and the Krugersdrift Dam. The average annual temperature ranges from 16°C to 26°C. The area is characterised by open flat plains with gradual hills, and koppies. The topography ranges from around 1300 mamsl to 1700 mamsl over most of the area, with a general decrease towards the north-west. There is an isolated area in the south east where topography rises to 2000 m. The vegetation types in the area are Moist Cool Highveld Grasslands, Dry Sandy Highveld Grassland and Eastern mixed Nama Karoo (Department of Water Affairs GRDM, 2010). The warms database for the quaternaries in the study area indicate that the abstraction from surface water resources per annum is 39.7 Mm³.

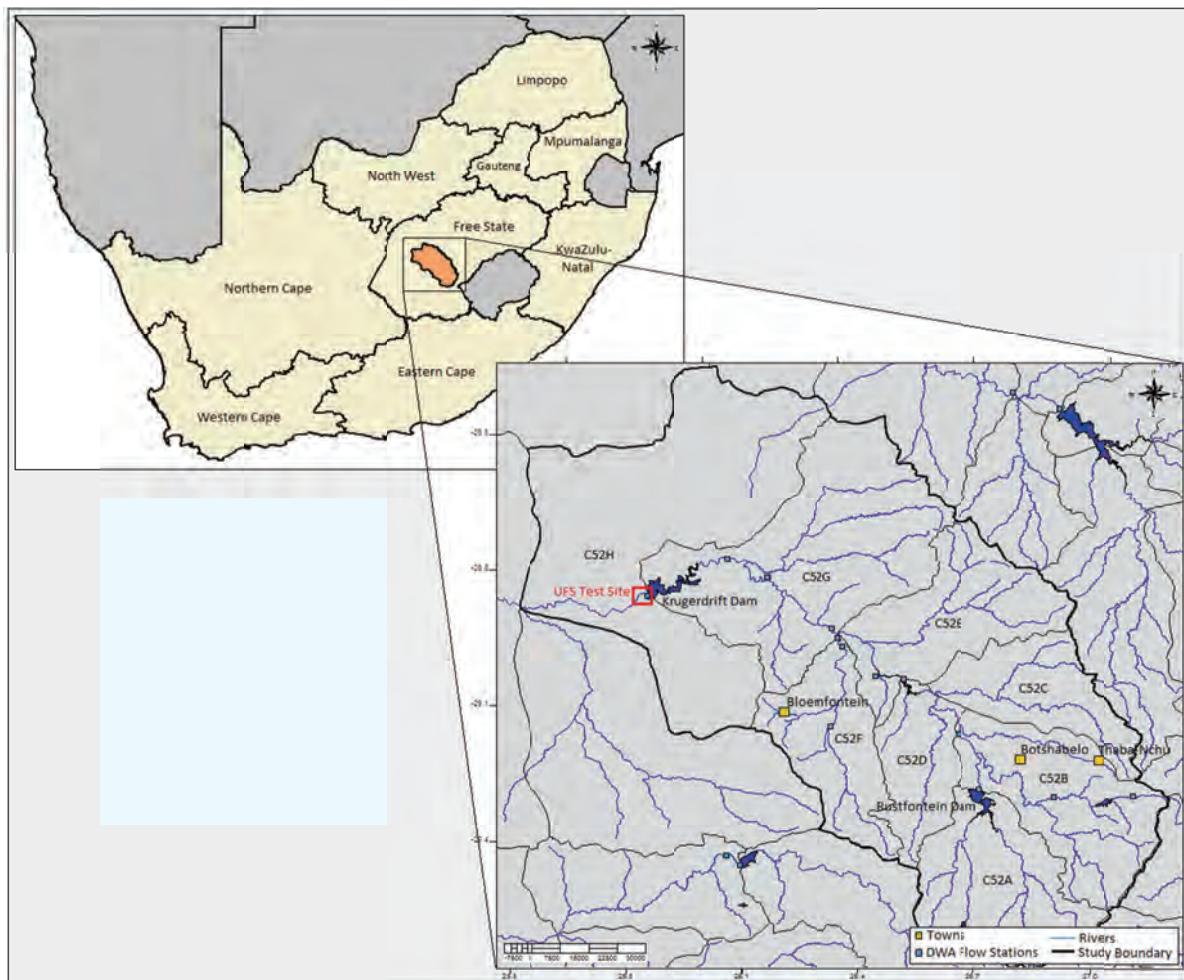


Figure 4-1-1 The location of the middle Modder River with the associated quaternaries and the UFS Surface water-groundwater interaction Test Site within South Africa.

4.1.2. Geology

The general outcrop geology of the middle Modder River area is comprised of Transvaal Supergroup rocks (Figure 4-1-2). The area is extensively intruded with dolerite dykes and sills. The Transvaal Supergroup rocks that are present in the study area form only a portion of the complete sequence, namely from the uppermost formation of the Eccca Group to the Elliot Formation of the Stormberg Group (Table 4-1-1). The overlying Drakensberg Group and underlying Dwyka Groups are not seen in the outcrop geology. The Volksrust Formation of the Eccca Group, the last of the group's sixteen formations, is a predominantly argillaceous layer that interfingers with the overlying Beaufort Group. The Volksrust Formation consists of silty shale with thin siltstone and sandstone lens. The overlying Adelaide Sub-group of the Beaufort Group consists of alternating layers of mudrock and sandstone, with sandstone becoming dominant towards the base. The Tarkastad Sub-group is characterised by a larger abundance of sandstone and red mudstone than in the Adelaide Sub-group. The boundary between these sub-groups is the only one in the Beaufort Group which can be traced throughout the Karoo Basin. The Molten Formation of the Stormberg Group consists of alternating layers of sandstone and grey mudstone. The overlying Elliot Formation is also an alternating sequence of red, green-grey mudrock and sandstone. The Elliot Formation is a typical "red bed" fluvial deposit (Johnson, Anhaeusser and Thomas, 2006).

Based on the outcrop geology of the area, a conceptual geological cross-section along the course of the middle Modder River (Rustfontein Dam to the Krugersdrift Dam) is created (Figure 4-1-3). The cross-section shows that the investigated section Modder River initially flows over the alternating sandstone and mudstone lithologies of the Adelaide Sub-group in the south-east where the topography is slightly higher, while further downstream the river flows over the lacustrine lithology of the Volksrust Fm.

The UFS surface water-groundwater interaction test site consists of 15 boreholes drilled near the base of the Krugersdrift Dam. Borehole logs from five of the test site boreholes were utilised to create a geological cross-section perpendicular to the river (Figure 4-1-4). The geological cross-section shows the conceptual understanding of the geology within 300 meters of the river. The lithology underlying the river is shale, presumably the Volksrust Shale of the Eccca Group. The surrounding river valley geology consists of this shale overlain by layers of calcrete, clay, sand and gravel.

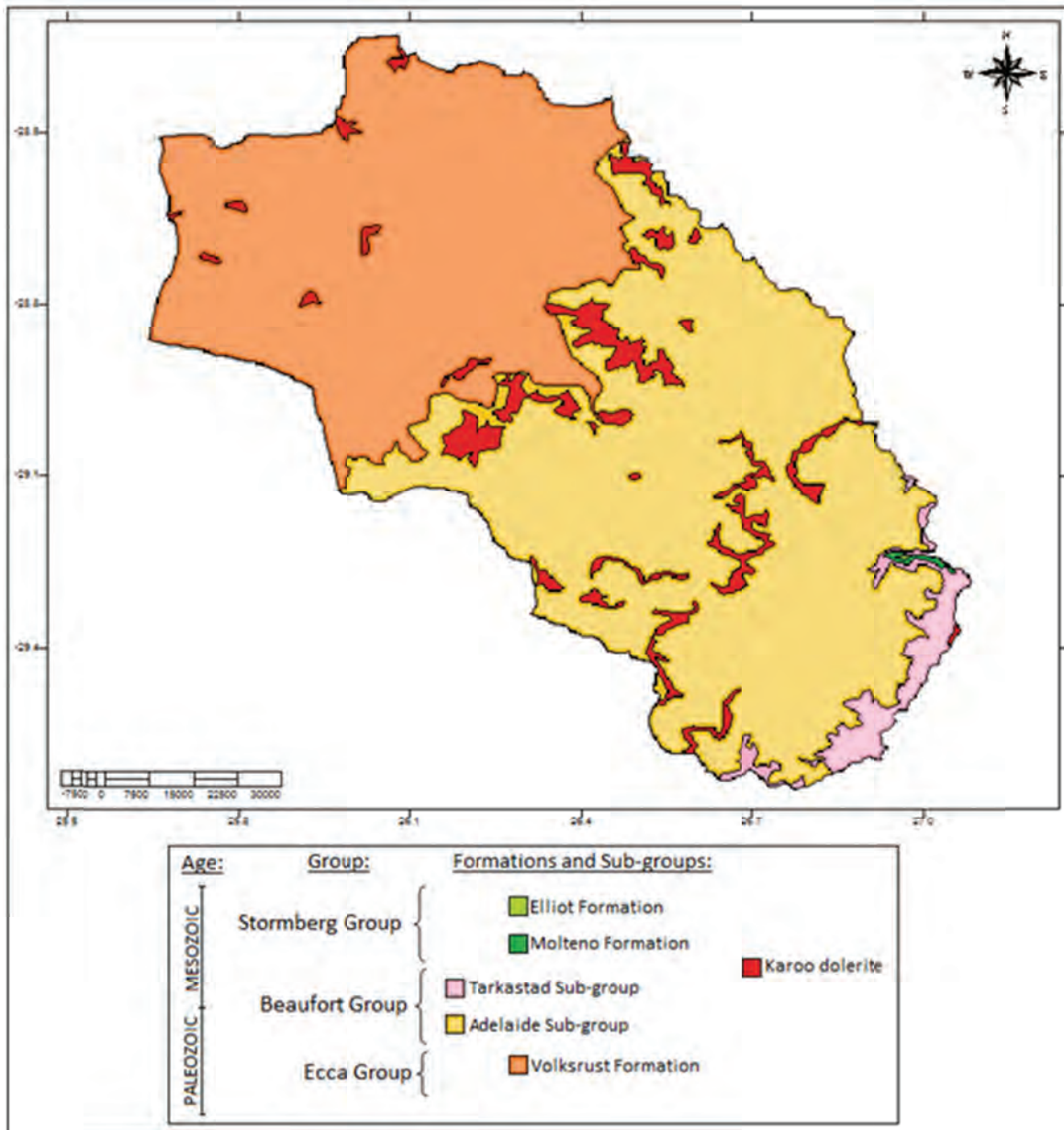


Figure 4-1-2 The outcrop geology of the C52 study area (based on the GRDM programme, 2010)

Table 4-1-1 The stratigraphic sequence present in the C52 study area

Era	Group	Sub-group	Formation
MESOZOIC	Drakensberg		
	Stormsberg		Clarens
			Elliot
PALEOZOIC	Beaufort	Tarkastad	
		Adelaide	
	Ecca		Volksrust

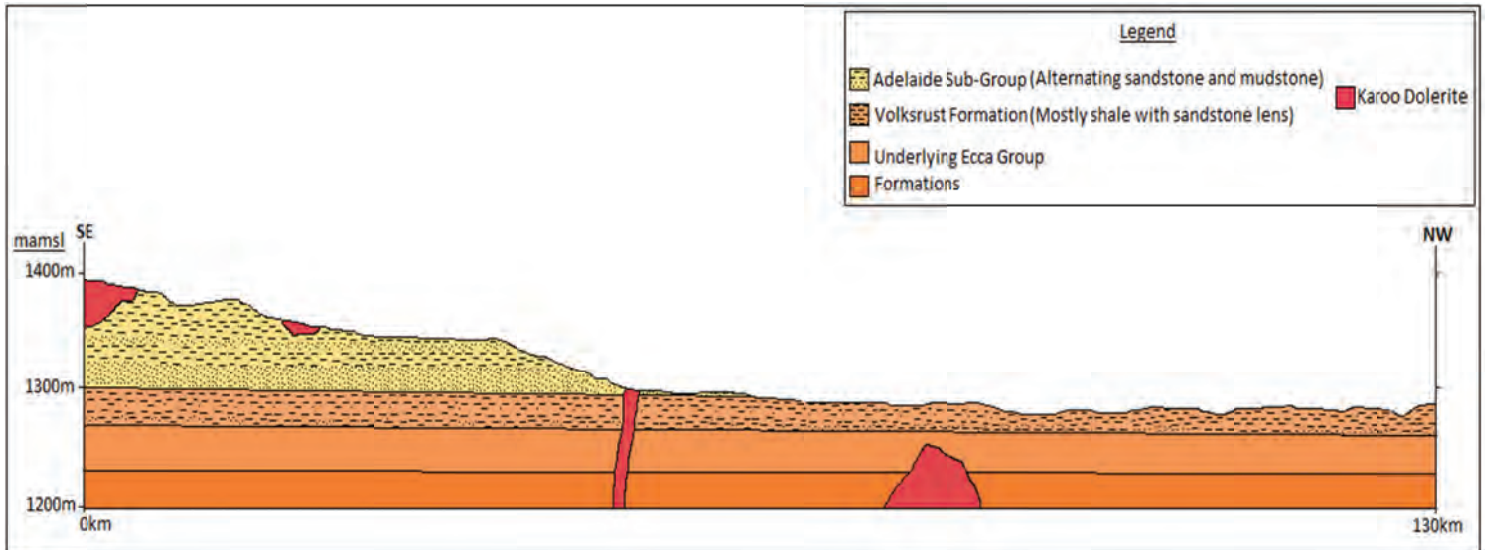


Figure 4-1-3 A conceptual geological cross-section along the Modder River from the Rustfontein Dam in the South-east to the Krugersdrift Dam in the north-west.

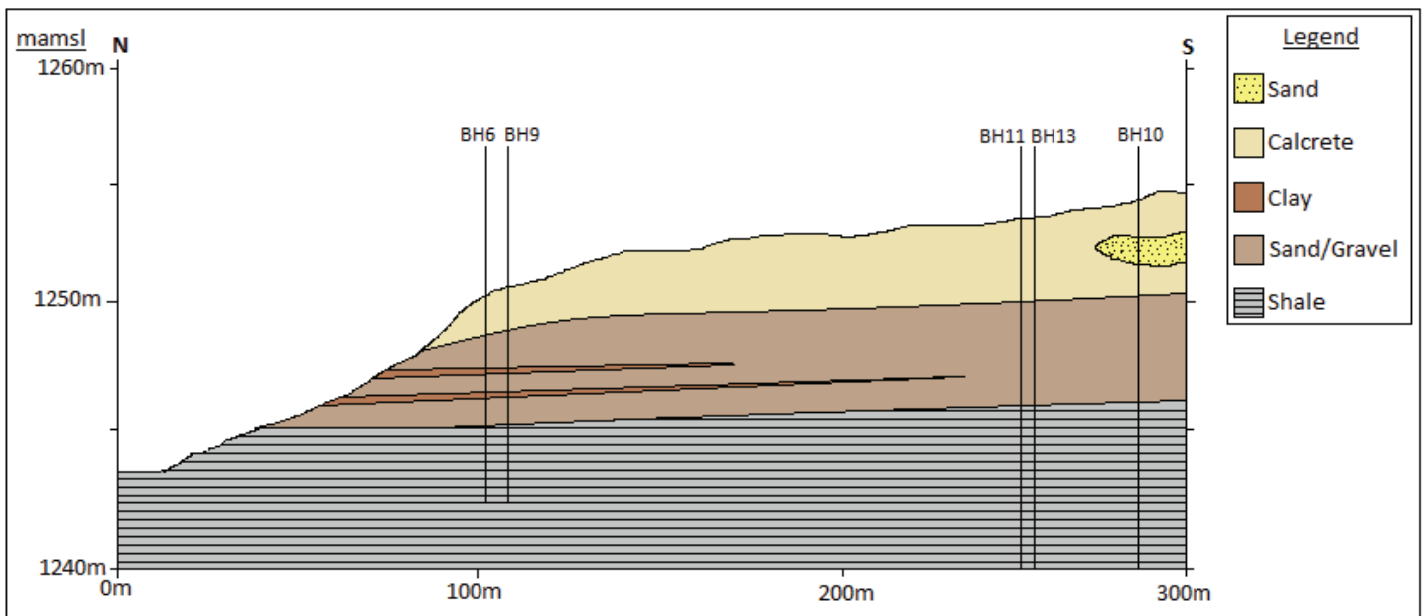


Figure 4-1-4 A conceptual geological cross-section perpendicular to the Modder River at the UFS Surface water-groundwater interaction site based on geological logs from BH10, BH13, BH11, BH9 and BH6.

4.1.3. Hydrogeology

The topography of the middle Modder River, spanning over quaternaries C52A to C52H, is higher towards the south-east with a localised high in the south-west (Figure 4-1-5). The topography gradually decreases towards the north-west, following the course of the Modder River towards the Krugersdrift Dam. The groundwater table tends to follow the topography in the area (Figure 4-1-6).

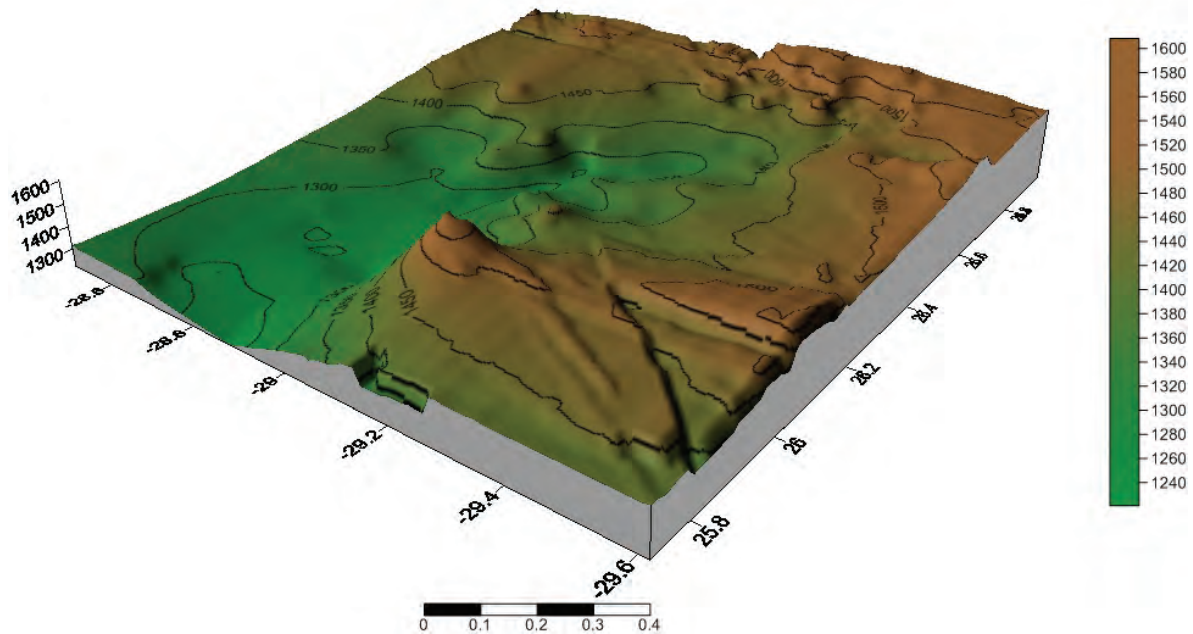


Figure 4-1-5 Generalised topography of the middle Modder River (quaternaries C52A-C52H).

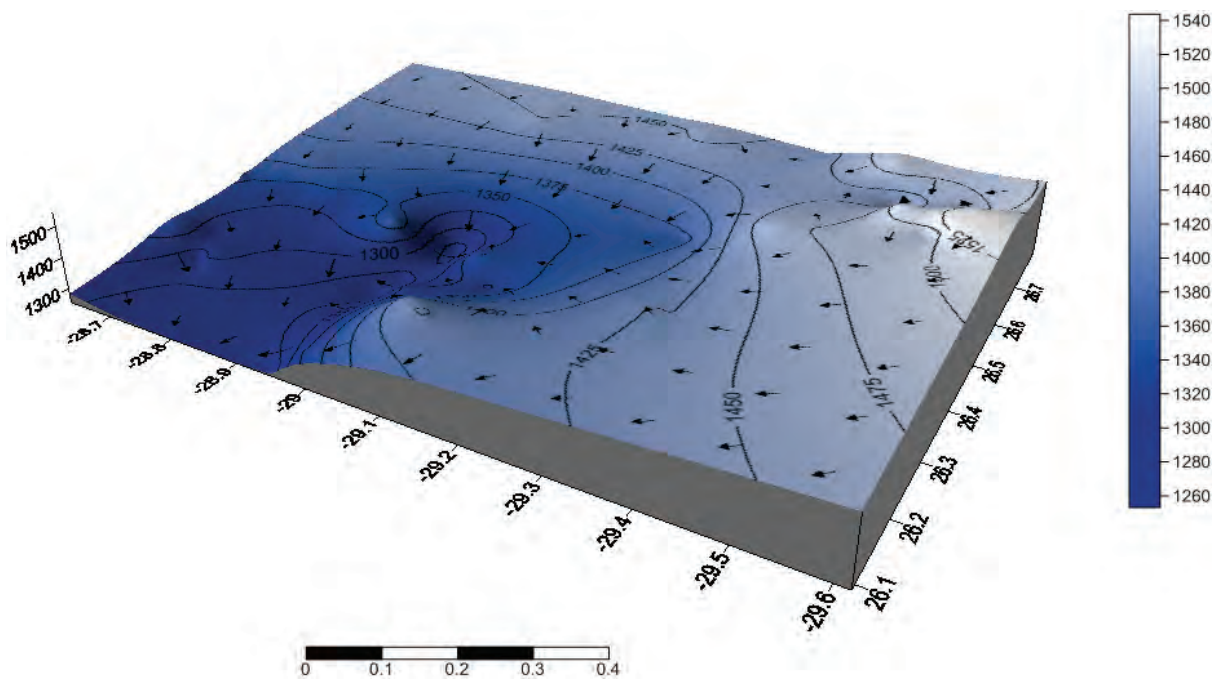


Figure 4-1-6 Generalised groundwater level within the middle Modder River course (1983-1990).

The Modder River flows over the Adelaide sub-group lithologies as well as the Volksrust shales (Figure 4-1-3). The Adelaide sub-group consists of alternating sandstone and mudstone layers. The sandstone layers are a potential source of groundwater to the river, but the mudstone layers would not be a high-yielding source of groundwater to the river due to its lower permeability. However, contact planes between the two layers and secondary features such as fractures could supply the river with groundwater through preferential pathways. The underlying Volksrust shale is an aquiclude which would only allow for slow diffusion of water, and movement along fractures and bedding planes.

The geological cross-section from geological borehole logs at the UFS surface water-groundwater interaction test site indicates that the river at this location is underlain by the impermeable shale of the Volksrust Fm (Figure 4-1-4). The contribution of groundwater to the river through this lithology would be minimal. Groundwater contributing to the river would then have to either be reaching the river through fractures and weathered areas of the shale, or from the overlying alluvial aquifer of calcrete, sand and gravel (Figure 4-1-4). Gomo (2011) first suggested this mechanism of groundwater baseflow to the Modder River along a seepage face between the overlying alluvial aquifer and the impermeable underlying shale. Gomo (2011) suggests that the groundwater discharge along the seepage face is derived from the local groundwater flow system, namely the shallow alluvial cover channel deposits. This mechanism of surface water-groundwater interaction implies that even though there is no direct hydraulic connection between the river and the underlying lithology, large amounts of groundwater can be contributing to the river flow (Gomo, 2011).

4.1.4. Mixing Cell Model (MCM)

The Mixing Cell Model (MCM) is applied on a site-specific and catchment scale. The site-specific scale application is performed using river and groundwater chemical analysis data from the UFS surface water-groundwater interaction test site. River water and groundwater chemical analysis data is available for samples collected on the 24/01/2011 and 05/08/2011, which allows the MCM to be run on two temporal scales. Figure 4-1-7 indicates the position of the 15 boreholes sampled within the test site. The catchment scale application is performed using river and groundwater chemical analysis data collected along the middle Modder River on 29/10/2012-31/10/2012 and 29/01/2013-01/02/2013, also allowing for two temporal scale applications. Figure 4-1-8 indicates the position of the river water and groundwater sample points for the January 2013 catchment scale application, and the active flowstations within the area.

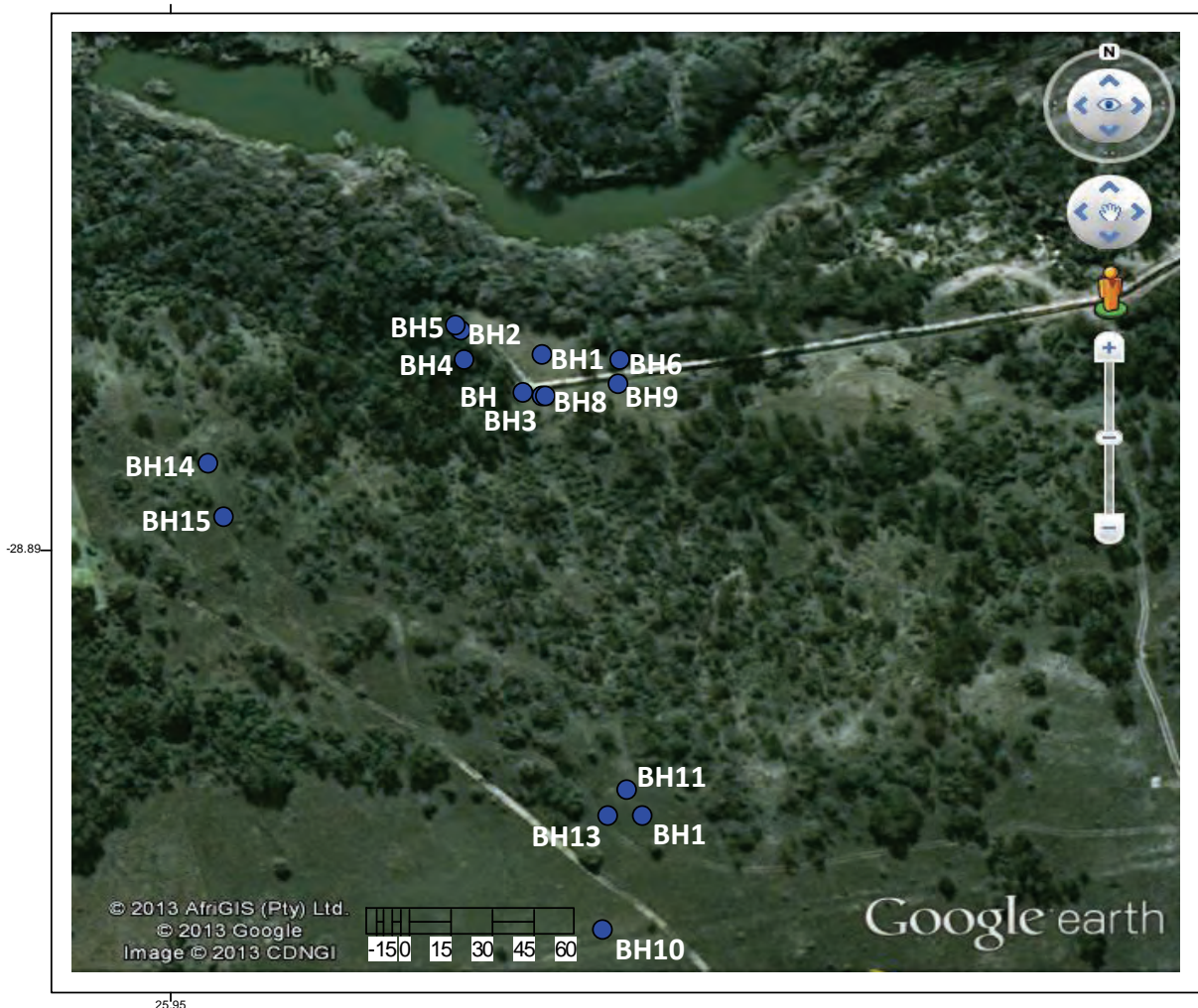


Figure 4-1-7 Location of the 15 borehole samples at the UFS Surface water-groundwater interaction test site.

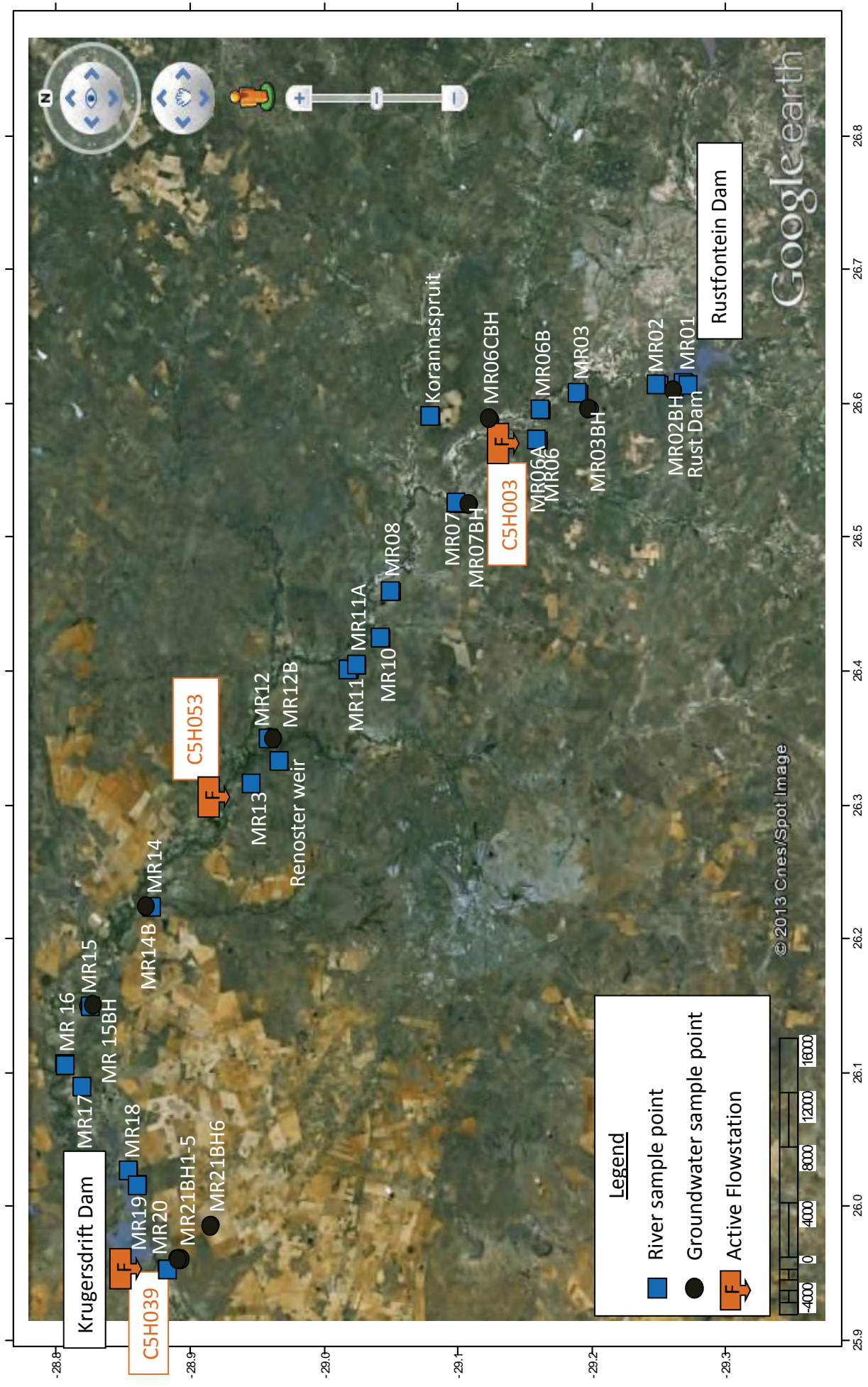


Figure 4-1-8 A Google Earth© Image indicating the location of the river water and groundwater samples collected along the Modder River in January 2013.

Site Specific Scale

4.1.5. Site Specific Scale MCM Conceptualisation and application

The various flows into and out of the river system at the UFS surface water-groundwater interaction test site are conceptualised into a box model, which forms the basis of the water balance equation for the MCM (Figure 4-1-9). The river course between the Krugersdrift Dam wall and the point of flow measurement is conceptualised as the River Cell. The MCM software requires a minimum of two mixing cells to be defined, thus a fictitious cell (o-River Cell) is created with exactly the same tracer concentrations as the River Cell, which is standard procedure for single cell applications of the model. The River Cell is defined by a set of tracer concentrations obtained from the average of two river samples taken within the defined study area (See Appendix B Table B-1). Inflow to the River Cell is defined by the average tracer concentration data from the flowstation C5H039 at Krugersdrift Dam. Groundwater sources to the River Cell are defined by chemical analysis data from each of the 15 boreholes within the study area, and the seepage source is defined by the water quality analysis data from the sampled seepage water. The river flow measurement taken downstream from the study area by Gomo (2011) is utilised in this model run as there is no flowstation at or near this point, other than the upstream Krugersdrift Dam flowstation. Abstraction and evaporation loss volumes from the section of the Modder River under investigation as well as direct rainfall volumes into the system are assumed to be negligible, as chemical data used in the model run covers a short time period and this omission ensures a mathematically simple model run. The representative EC and chloride (Cl) values for each flow represented in the flow box model are shown in Figure 4-1-10. Figure 4-1-10 indicates that the EC and Cl values, used to define the River Cell, are higher than the upstream river inflow source, which could indicate that the poorer quality groundwater could indeed be contributing to the stream.

The MCM was applied using the flow setup discussed in two model runs, each making use of a complete set of chemical data from two different time periods, namely one model run for data from January 2011 and one model run for data from August 2011. Three scenarios regarding tracers and weighting factors were performed for each time period, namely 1) all available tracer data is utilised and all tracers are assumed conservative, and thus assigned a weighting factors of 1 ($\omega = 1$), 2) tracers showing a high chemical mass balance error are assigned a lower weighting factor ($\omega = 0.3$), and 3) lower weighting factors are assigned as in

scenario 2 and additionally, the tracer showing the highest error is omitted (Table 4-1-2). The outflow from the January 2011 run is $6.8 \text{ Mm}^3/\text{a}$ measured by Gomo (2011), while the outflow from the August 2011 run is determined from the upstream flowstation C5H039, because no other direct measurements of flow are available. The chemical data for each time period is given in Appendix B Table B-1 and B-2.

Table 4-1-2 The weighting factor assigned to each tracer for scenarios 1-3.

Weighting Factor Scenario	Weighting Factor (ω)										
	pH	EC	Ca	Mg	Na	K	TAL	F	Cl	SO4	Si
Scenario 1	1	1	1	1	1	1	1	1	1	1	1
Scenario 2	1	1	1	1	1	0.3	1	0.3	0.3	0.3	1
Scenario 3	1	1	1	1	1	0.3	1	0	0.3	0.3	1

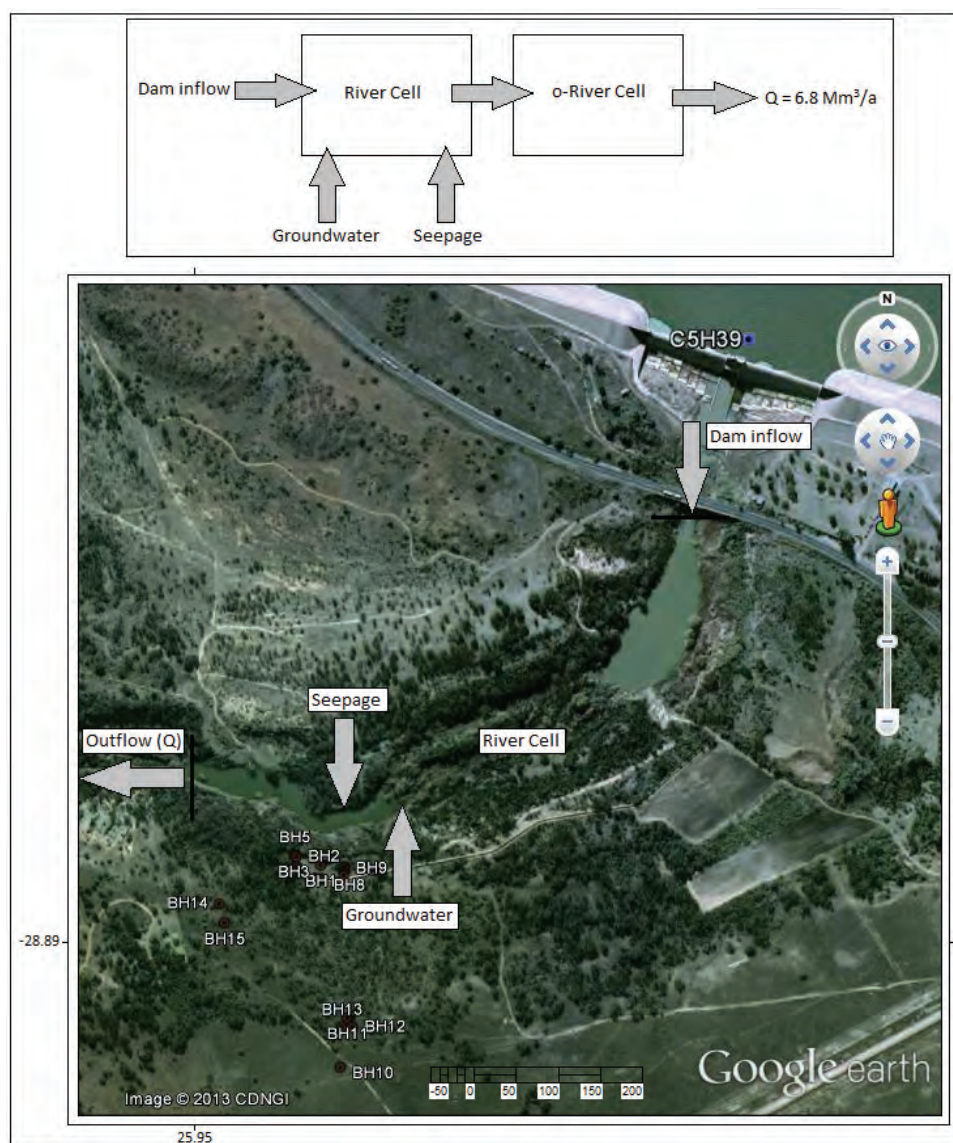


Figure 4-1-9 The box model conceptualisation of the various flows modelled at the UFS Surface water-groundwater interaction test site and a Google Earth© image showing the conceptualisation on a real scale indicating the position of the flowstation (C5H039) and sampled boreholes (BH1-15).

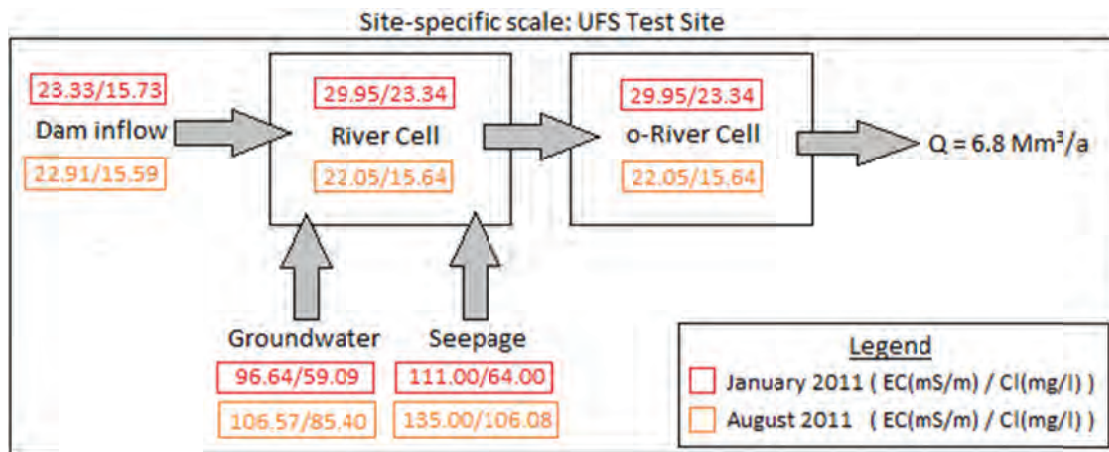


Figure 4-1-10 The representative EC and chloride (Cl) values for the box model conceptualisation of the various flows modelled at the UFS Surface water-groundwater interaction test site. Concentrations indicate that groundwater could be contributing to the streamflow because the EC-value and chloride concentration increase downstream.

4.1.6. Site Specific Scale MCM Results

The MCM run for January 2011, using scenario 1 weighting factors, indicated a groundwater contribution to baseflow from the sampled boreholes BH1-9 of 0.03 Mm³/a, while the contribution from sampled boreholes BH10-15 was 0.84 Mm³/a (Table 4-1-3). No inflow from the seepage source was found. The inflow from the Dam to the River Cell was determined to be 5.23 Mm³/a, while the outflow from the River Cell to the fictitious o-River Cell was determined at 7.32 Mm³/a. The MCM run for January 2011, using scenario 2 weighting factors, indicated a groundwater contribution to baseflow from the sampled boreholes BH1-9 of 0.57 Mm³/a, while the contribution from sampled boreholes BH10-15 was 0.00 Mm³/a (Table 4-1-3). No inflow from the seepage source was found. The inflow from the Dam to the River Cell was determined to be 6.51 Mm³/a, while the outflow from the River Cell to the fictitious o-River Cell was determined at 7.68 Mm³/a. The MCM run for January 2011, using scenario 3 weighting factors, indicated a groundwater contribution to baseflow from the sampled boreholes BH1-9 of 0.47 Mm³/a, while the contribution from sampled boreholes BH10-15 was 0.00 Mm³/a (Table 4-1-3). No inflow from the seepage source was found. The inflow from the Dam to the River Cell was determined to be 6.89 Mm³/a, while the outflow from the River Cell to the fictitious o-River Cell was determined at 7.74 Mm³/a.

Table 4-1-3 Summary of the MCM output for the UFS Test Site (January 2011) for each of the weighting factor scenarios

UFS Test site (January 2011)			
Name of inflow	Rate of inflow (Mm ³ /a)		
	Scenario 1	Scenario 2	Scenario 3
C5H039	5.23	6.51	6.89
BH1-9	0.03	0.57	0.47
BH10-15	0.84	0.00	0.00
Seep	0.00	0.00	0.00

The MCM run for August 2011, using scenario 1 weighting factors, indicated a groundwater contribution to baseflow from the sampled boreholes BH1-9 of 0.03 Mm³/a, while the contribution from sampled boreholes BH10-15 was 1.6 Mm³/a (Table 4-1-4). Inflow to the River Cell from the seepage source along the river was estimated at 0.7 Mm³/a. The inflow from the Dam to the River Cell was determined to be 4.45 Mm³/a, while the outflow from the River Cell to the fictitious o-River Cell was determined at 19.13 Mm³/a. The MCM run for August 2011, using scenario 2 weighting factors, indicated a groundwater contribution to baseflow from the sampled boreholes BH1-9 of 0.37 Mm³/a, while the contribution from sampled boreholes BH10-15 was 0.00 Mm³/a (Table 4-1-4). Inflow to the River Cell from the seepage source along the river was estimated at 0.1 Mm³/a. The inflow from the Dam to the River Cell was determined to be 12.77 Mm³/a, while the outflow from the River Cell to the fictitious o-River Cell was determined at 20.30 Mm³/a. The MCM run for August 2011, using scenario 3 weighting factors indicated a groundwater contribution to baseflow from the sampled boreholes BH1-9 and boreholes BH10-15 of 0.00 Mm³/a (Table 4-1-4). Inflow to the River Cell from the seepage source along the river was estimated at 0.35 Mm³/a. The inflow from the Dam to the River Cell was determined to be 15.29 Mm³/a, while the outflow from the River Cell to the fictitious o-River Cell was determined to be 21.43 Mm³/a.

Table 4-1-4 Summary of the MCM output for the UFS Test Site (August 2011) for each of the weighting factor scenarios

UFS Test site (August 2011)			
Name of inflow	Rate of inflow (Mm ³ /a)		
	Scenario 1	Scenario 2	Scenario 3
C5H039	4.45	12.77	15.29
BH1-9	0.03	0.37	0
BH10-15	1.6	0	0
Seep	0.7	0.1	0.35

The results from the two temporal scale applications, utilizing scenario 2 weighting factors are graphically represented in Figure 4-1-11 and expressed as percentages of the assigned total outflow. The January 2011 MCM application determined the inflow from the Dam to the conceptualised River Cell to be 84.7% of the total outflow from the cell. The groundwater contribution to the stream was estimated at 7.4% of the total flow, while the seepage to the river was estimated at 0.0% of the total flow. The August 2011 MCM application determined the inflow from the Dam to the conceptualised River Cell to be 62.9% of the total outflow from the cell. The groundwater contribution to stream was estimated at 1.8% of the total flow, while the seepage to the river was estimated at 0.5% of the total flow.

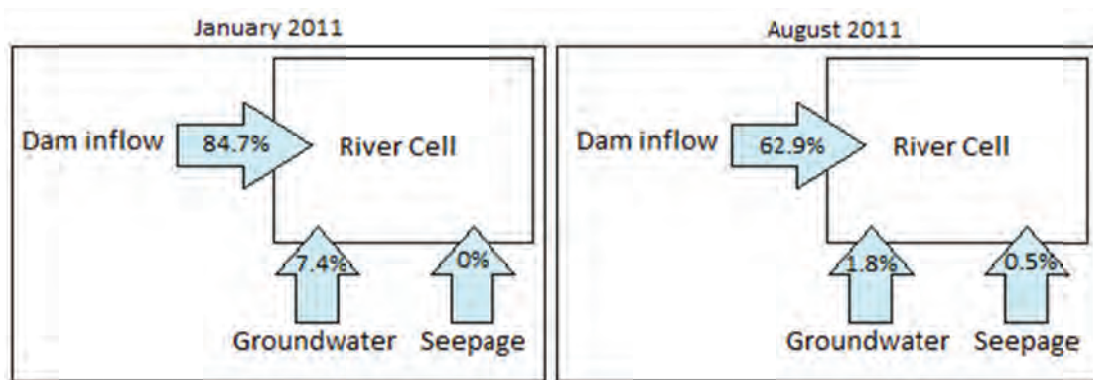


Figure 4-1-11 The unknown fluxes determined by the MCM for the January and August 2011 model runs, making use of the scenario 2 weighting factors, expressed as a percentage of the total flow volume.

4.1.7. Site Specific Scale Discussion and Comparison

The water balance percentage errors for each time period and the associated weighting factor scenario are shown in Table 4-1-6. These errors indicate that there is more water flowing out of the system than flowing in from the identified sources, which could be caused by unidentified runoff, rainfall or river inflow sources to the system. The percentage flow diagram (Figure 4-1-11) highlights this difference where in the January 2011 model run 7.9% of the flow to the system is unaccounted for, while in the August 2011 model run 34.8% of the flow is unaccounted for. These unaccounted flows result in the error seen in the water balance. The chemical mass balance percentage errors are shown in Table 4-1-5a and b. The water balance errors for the MCM run for January 2011 are much lower than the errors in the August 2011 run. The large errors associated with the August 2011 run could be attributed to the outflow volume that is used, because a flow volume from the C5H039 flowstation was utilised as no direct flow measurement was taken. The water balance error

decreases when weighting factors are implemented in both instances and then even further when the tracer F (Fluoride) is omitted (Table 4-1-6). The resulting effect on the estimation of the groundwater contribution to the system in response to the three different scenarios is clear, with the change from scenario 1 to scenario 2 resulting in a drastic decrease in percentage error and overall groundwater contribution to the river. A shift in the main source of groundwater baseflow from BH10-15 to BH1-9 in the January and August model runs are (Table 4-1-3 and Table 4-1-4), and a decrease in the amount of seepage inflow for the August model run (Table 4-1-4) results from the incorporation of weighting factors. The change from scenario 2 to scenario 3 results in a decrease in groundwater contribution from BH1-9 in the January run, whereas the result in the August run is no groundwater contributes to the river and an increase in seepage inflow is seen. The chemical mass balance percentage error drastically decreases from scenario 1 to scenario 2, but slightly increases again from scenario 2 to scenario 3, as seen in the average percentage error for each scenario (Table 4-1-5a and b). Considering these changes in percentage error for the balance equations, it seems that scenario 2 would be a reasonable selection for the final model run, because it allows for a lower water balance and chemical mass balance percentage error.

Table 4-1-5 Chemical mass balance percentage error for each tracer for a) January 2011 for each scenario, and b) August 2011 for each scenario.

a) Ion balance (January 2011)				b) Ion balance (August 2011)			
Tracer	Chemical Mass Balance Error (%)			Tracer	Chemical Mass Balance Error (%)		
	Scenario 1	Scenario 2	Scenario 3		Scenario 1	Scenario 2	Scenario 3
pH	-13.80%	0.30%	4.50%	pH	-67.70%	-35.60%	-23.90%
EC	-15.00%	-5.80%	-9.70%	EC	-47.50%	-32.30%	-27.50%
Ca	-12.40%	-10.30%	-4.30%	Ca	-28.00%	-1.30%	19.60%
Mg	16.00%	11.80%	12.40%	Mg	21.20%	23.90%	28.30%
Na	-21.30%	-12.50%	-10.90%	Na	6.80%	-29.20%	-11.00%
K	-42.80%	-33.30%	-30.60%	K	-52.60%	-38.50%	-12.20%
TAL	-4.00%	1.20%	4.30%	TAL	-37.00%	-26.00%	5.00%
F	46.70%	77.80%		F	34.40%	165.50%	
Cl	-30.90%	-24.50%	-23.00%	Cl	-46.20%	-34.40%	-13.70%
NO3	-10.50%	0.20%	3.50%	NO3	-40.00%	-29.30%	-12.50%
SO4	-38.00%	-28.40%	-26.20%	SO4	-23.30%	-21.40%	-20.20%
Average	-11%	-2%	-8%	Average	-25%	-5%	-7%

Table 4-1-6 Water balance percentage error for each scenario for each time period

Water Balance Error (%)			
Time period	Scenario 1	Scenario 2	Scenario 3
January 2011	22%	9%	6%
August 2011	70%	41%	30%

The final groundwater contribution to streamflow in the study area is estimated by the MCM at 0.57 Mm³/a in January 2011 and 0.37 Mm³/a in August 2011, with an additional seepage inflow of 0.1 Mm³/a in August 2011. The rainfall difference between the two months is substantial (Figure 4-1-12), yet the groundwater contribution to baseflow seems to remain reasonable stable. This constant groundwater contribution to the river flow might be an indication that there is a limiting factor which inhibits the increase of groundwater baseflow with an increase in rainfall, such as a limited or conditional hydraulic connection between the river and groundwater system. An unconventional surface water-groundwater connection is plausible in light of the impermeable shale underlying the river bed. Another component to consider is the seepage inflow to the river sourced from the overlying alluvial aquifer. The MCM run for the high rainfall time period January 2011 found no seepage inflow while the low rainfall period August 2011 model run found a seepage inflow of 0.1 Mm³/a, which is opposite to what would be expected. However, this could be explained by high evapotranspiration rates and high borehole abstraction in the peak of the summer season resulting in little or no groundwater reaching the river from the overlying alluvial aquifer comprising the local groundwater system. The other explanation is that the MCM is essentially a numerical model, which could easily create a fictitious inflow from this source. The three weighting factor scenarios also highlight the inconsistency with regards to input data variation.

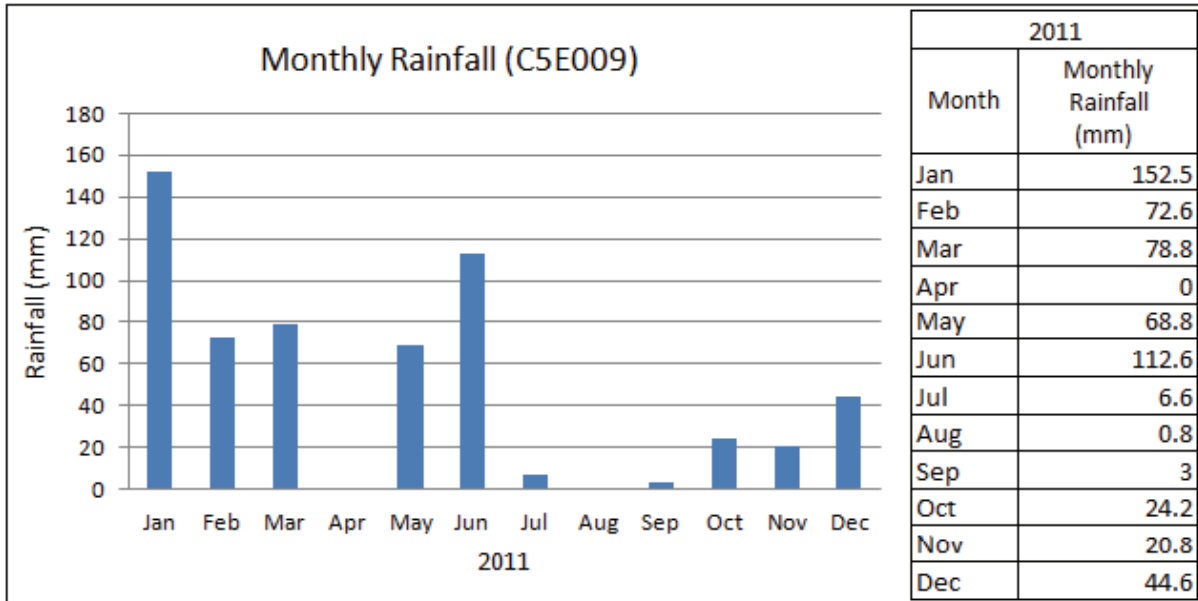


Figure 4-1-12 Rainfall in millimetres per month over the year 2011 in the UFS Test Site area from the meteorological station C5E009 at the Krugersdrift Dam.

The groundwater contribution estimates from the MCM are compared to four other methods in Table 4-1-7. The Pitman, Sami and Hughes methods all estimate the groundwater baseflow volume to be 0 Mm³/a, while the tracer and MCM methods estimate the groundwater baseflow at 0.61 Mm³/a and 0.57 Mm³/a, respectively. The average volume from a number of Tracer method application using different tracers results in a much higher estimate than when only EC is used. In terms of the methods making use of chemical data, namely the tracer and MCM methods, the MCM provides a more acceptable groundwater baseflow estimate, especially if one attempts to use more than one constituent in the tracer method. Comparing the MCM to the Pitman, Sami and Hughes methods, there is a large difference, where the conventional methods found no groundwater contribution to the river baseflow in the area, while the MCM did find that groundwater contributes to the river in small quantities. In this case, the MCM method seems to provide the most reasonable estimate of groundwater baseflow because of the fact that estimating no groundwater baseflow might be an under-estimate. This is further substantiated when considering the results from the work performed by Gomo (2011) on the test site, where the river was found to be a gaining river and groundwater contribution to the river estimated at 4% of the total flow in the river. Gomo (2011) has reported on farmers in the area confirming seepage into the river from the alluvial aquifer along the river.

Table 4-1-7 Groundwater contribution estimates from the Pitman, Sami, and Hughes methods for the quaternary C52H & tracer and MCM methods for a) January 2011 and b) August 2011

a)

Quat	Pitman uncalibrated (Mm ³ /a)	Sami uncalibrated (Mm ³ /a)	Hughes uncalibrated (Mm ³ /a)	MCM (Mm ³ /a)	Tracer Method (Mm ³ /a)	
					EC	Average
C52H	0.00	0.00	0.00	0.57	0.61	1.96

b)

Quat	Pitman uncalibrated (Mm ³ /a)	Sami uncalibrated (Mm ³ /a)	Hughes uncalibrated (Mm ³ /a)	MCM (Mm ³ /a)	Tracer Method (Mm ³ /a)	
					EC	Average
C52H	0.00	0.00	0.00	0.37	0.57	6.72

Catchment scale

4.1.8. Catchment Scale MCM Conceptualisation and application

The catchment scale study area covers approximately 130km of the middle Modder River, stretching from the Rustfontein Dam to the Krugersdrift Dam, passing through the Mockes and Maselspoort Dams. Nineteen river water samples and eleven groundwater borehole samples were collected in October 2012. Twenty three river water samples (three additional tributary samples and one additional Modder River sample) and thirteen groundwater borehole samples were collected in January 2013. The study area has been divided into three sections based on active flowstations (Figure 4-1-8), to allow for the quantification of groundwater baseflow within each section because the MCM requires a defined outflow from the modelled system to quantify the unknown inflows. Section 1 is defined from the Rustfontein Dam to the flowstation C5H003, Section 2 is defined from flowstation C5H003 to flowstation C5H053 and Section 3 is defined from flowstation C5H053 to the flowstation C5H039 at Krugersdrift Dam (Figure 4-1-8). A MCM run for each section is performed for each sample collection time period, namely October 2012 and January 2013.

The various flows into and out of the river system within each section are conceptualised in a box model, which forms the basis of the water balance equation for the MCM. The box model conceptualisation for Section 1 is shown in Figure 4-1-13. The system is represented as a two cell box model, with the river samples MR03 and MR06 defining the cells. Upstream river inflow to the MR03 Cell is defined by the water quality analysis results of the upstream river sample MR02 and the groundwater inflow is defined by the borehole samples, MR03BH and MR02BH. Upstream river inflow to the downstream cell (MR06 Cell)

is sourced from the MR03 Cell and inflow from the Sepane tributary, represented by sample MR06B. Groundwater sources to the MR06 Cell are defined by the chemical analysis data from borehole sample points, MR06CBH and MR07BH. The flow volume recorded at the flowstation, C5H003 is used to define the downstream outflow from the system. The outflow volumes recorded at the flowstation for the month of October 2012 and January 2013 are extrapolated to yearly volumes of 6.24 Mm³/a and 16.05 Mm³/a, respectively. The representative EC and chloride (Cl) values for each flow represented in the flow box model are shown in Figure 4-1-17. Figure 4-1-17 indicates both EC and Cl values increase downstream in October 2012, which could indicate that the poorer quality groundwater could be contributing to the stream. Chemical data collected in January 2013 shows an opposite trend in terms of EC as it decreases downstream, but Cl increases. The lower EC values in January 2013 could be attributed to the increase rainfall during this period. Groundwater quality is shown to remain fairly stable between the two time periods. Data not collected in October 2012 for MR06CBH, MR07BH and MR06B prevent a complete model run.

Figure 4-1-14 is the box model conceptualisation of the flow system in Section 2 as a two cell box model, defined by river water quality samples MR12 and MR13. Upstream river inflow to the MR12 Cell is defined by the water quality analysis results of the upstream river sample MR11 and the groundwater inflow is defined by the borehole sample MR12BH. Upstream river inflow to the downstream cell (MR13 Cell) is sourced from the MR12 Cell and the groundwater inflow is defined by water quality data from the borehole sample MR12BH. The flow volume recorded at the flowstation, C5H053 is used to define the downstream outflow from the system. Outflow volume data at the flowstation C5H053 is 19.05 Mm³/a based on the monthly outflow data from October 2012. There is no monthly flow data for January 2013 at this flowstation. The representative EC and chloride (Cl) values for each flow represented in the flow box model are shown in Figure 4-1-18. Figure 4-1-18 shows a trend in EC and Cl values to increase downstream as well as the groundwater quality to be of a much poorer quality, which could indicate that groundwater is indeed contributing to the stream. Groundwater quality is once again seen to be fairly constant between the two time periods.

The box model conceptualisation for Section 3 is shown in Figure 4-1-15. The system is represented as a single cell box model, with the river cell defined by water quality data from the flowstation point at the Krugersdrift Dam, C5H039. The MCM requires at a minimum of two cells to be defined, thus a fictitious cell (o-C5H039 Cell) is created with exactly the same tracer concentrations as the river cell, C5H039 Cell. Inflow is defined by the average tracer concentration set of river samples MR18-MR20, and the groundwater inflow is defined by the average tracer concentration set of borehole samples MR21BH1-BH6. The box model for the January 2013 model run is slightly different from the one shown in Figure 4-1-19 because chemical data from the flowstation C5H039 sampling point is not available for this time period and thus follows a setup similar to the box flow model in Figure 4-1-16. The flow volume recorded at the flowstation, C5H039 is used to define the downstream outflow from the system. Outflow volume data at the flowstation C5H039 is 114.12 Mm³/a based on the monthly outflow data from October 2012. There is no monthly flow data for January 2013 at this flowstation. The representative EC and chloride (Cl) values for each flow represented in the flow box model are shown in Figure 4-1-19. The October 2012 setup shows a trend of decreasing EC and Cl values downstream. This trend could indicate that no groundwater is contributing to the stream, but the water damming up in the Krugersdrift Dam could also be the cause for the reverse trend seen here. The January 2013 setup shows an expected trend of EC and Cl values increasing downstream. The shift in trend could be attributed to the different box model setup for this time period, where the January 2013 setup does not make use of chemical data from the C5H039 flowstation. Groundwater quality is found to be fairly constant between the two time periods.

The middle Modder River study area is additionally conceptualised as one continuous flow system to show a large scale application of the MCM and the functionality in ungauged systems (Figure 4-1-16). The continuous model run makes use of the January 2013 dataset. The area between the river samples MR15 and MR16 is insufficiently defined to allow for one continuous model, resulting in these two sections being modelled individually. Each sample point along the course of the sampled river areas is conceptualised as a river cell, optimizing the number of mixing cells, to minimize the infringement on the basic assumption that total mixing occurs within each cell. Groundwater inflows are defined by borehole samples taken within the cell's vicinity.

The MCM was applied using the discussed flow system setups for Sections 1 to 3, where for each Section, two model runs were performed making use of a complete set of tracer concentration data from the two different time periods, namely one model run for data from October 2012 and one model run for data from January 2013. The water quality dataset for each section and each time period model run therein is shown in Appendix C Table C-1-C-6. Two different scenarios regarding weighting factors for individual tracers are performed also for each individual model run. Scenario 1 utilises all available tracer data and all weighting factors are assigned a value of 1, where scenario 2 assigns a lower weighting factor to tracers that show a high chemical mass balance error. The weighting factors assigned to the individual constituents for scenario are shown in Table 4-1-8. The ungauged continuous flow model setup was also run for each of the weighting factor scenarios. The water quality data used is shown in Appendix C Table C-7.

Table 4-1-8 Weighting factors assigned to each tracer for scenarios 1 and 2.

Weighting Factor Scenario	Tracer																			
	pH	EC	Ca	Mg	Na	K	MAIk	F	Cl	Br	SO4	TDS	Al	Fe	Mn	Si	Ba	Cu	Se	Zn
Scenario 1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Scenario 2	1	1	1	0.3	1	0.3	1	0.3	0.3	1	0.3	1	0.3	0.3	0.3	0.3	0.3	0.3	1	1

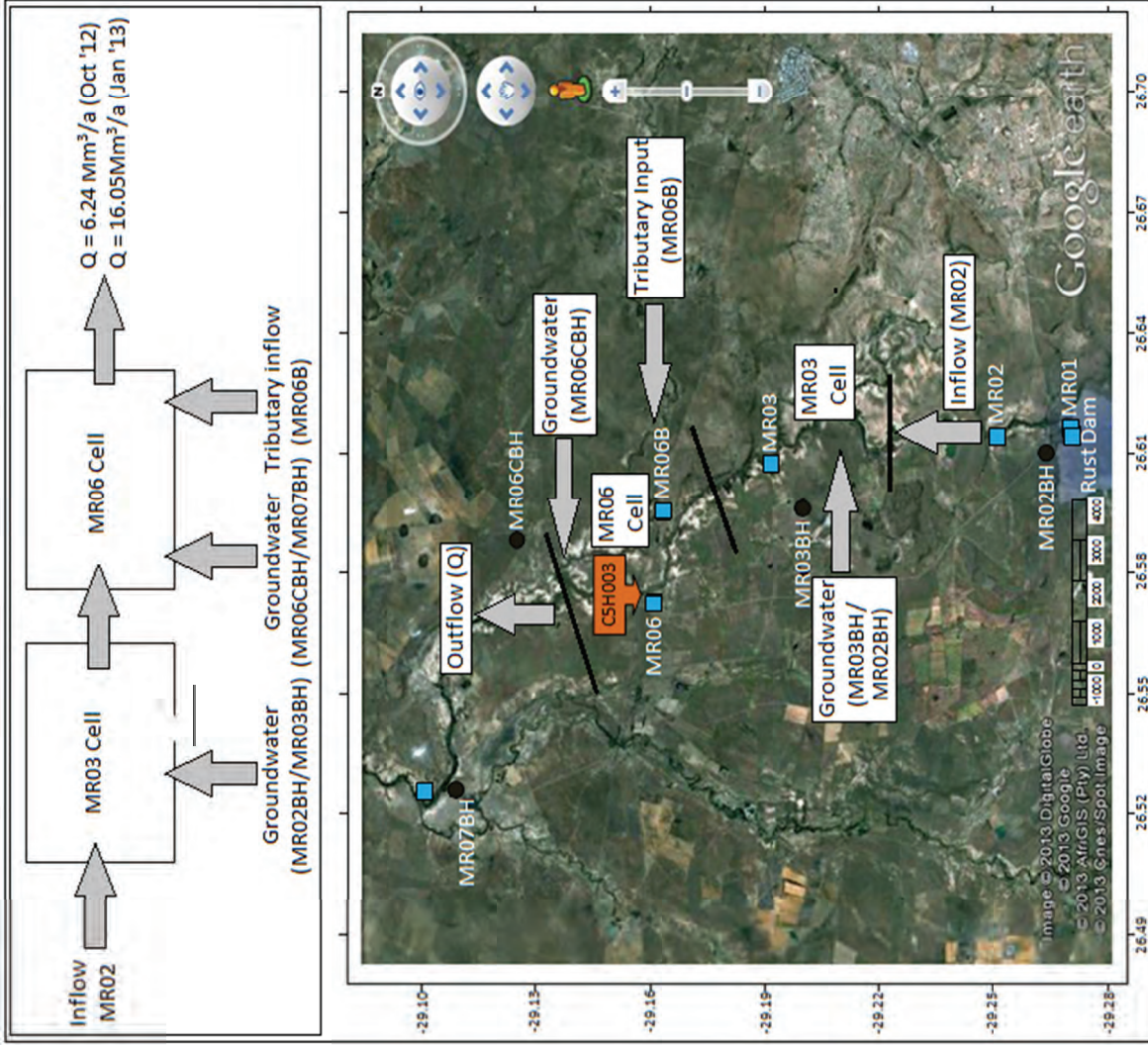


Figure 4-1-13 The box model conceptualisation of the various flows modelled for Section 1 and a Google Earth© image showing the conceptualisation on a real scale, indicating flowstation C5H003 and the various sample points.

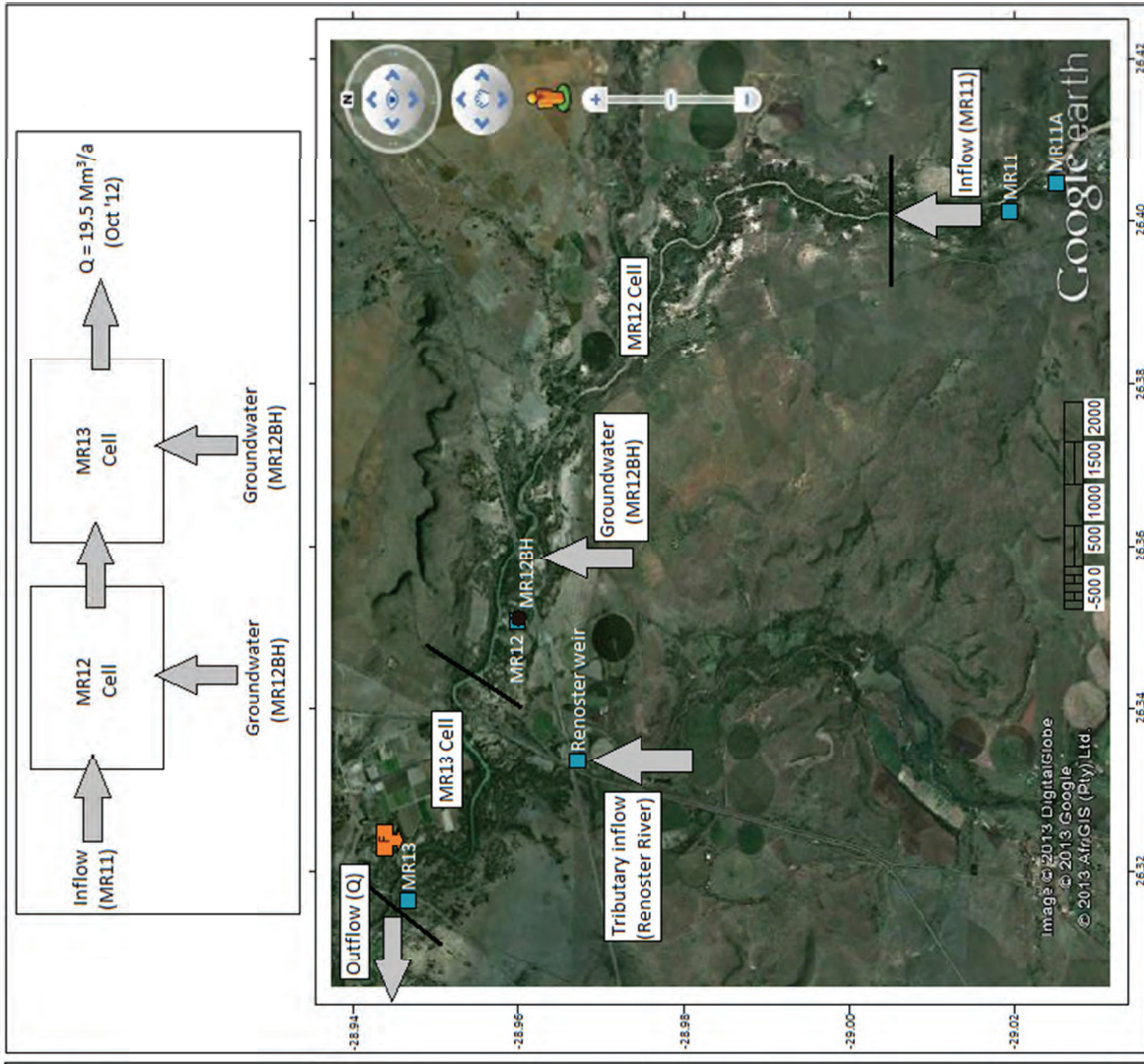


Figure 4-1-14 The box model conceptualisation of the various flows modelled for Section 2 and a Google Earth© image showing the conceptualisation on a real scale, indicating flowstation C5H053 and the various sample points.

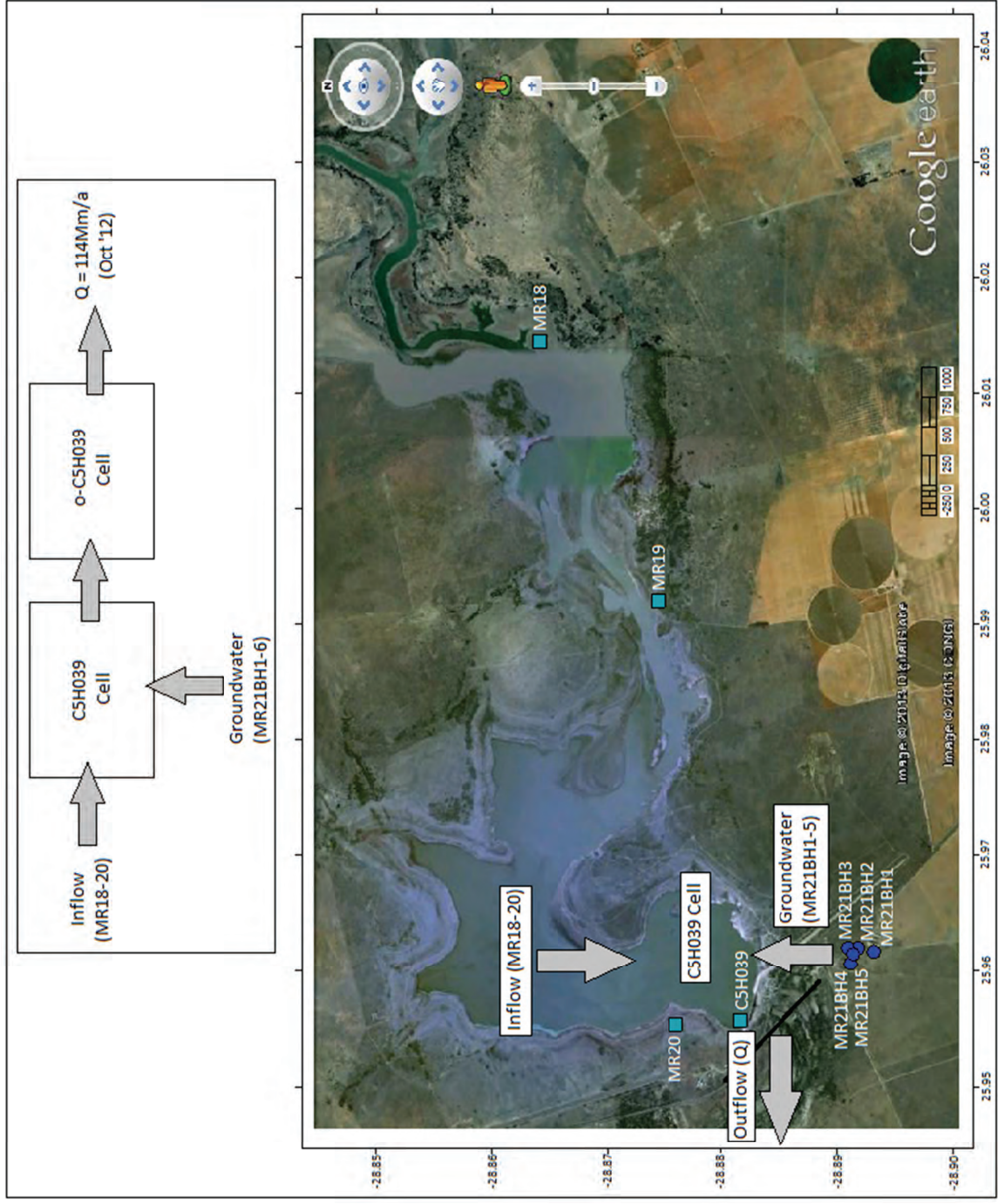


Figure 4-1-15 The box model flow conceptualisation of the various flows modelled for Section 3 and a Google Image© showing the conceptualisation on a real scale, indicating flowstation C5H039 and the various sample

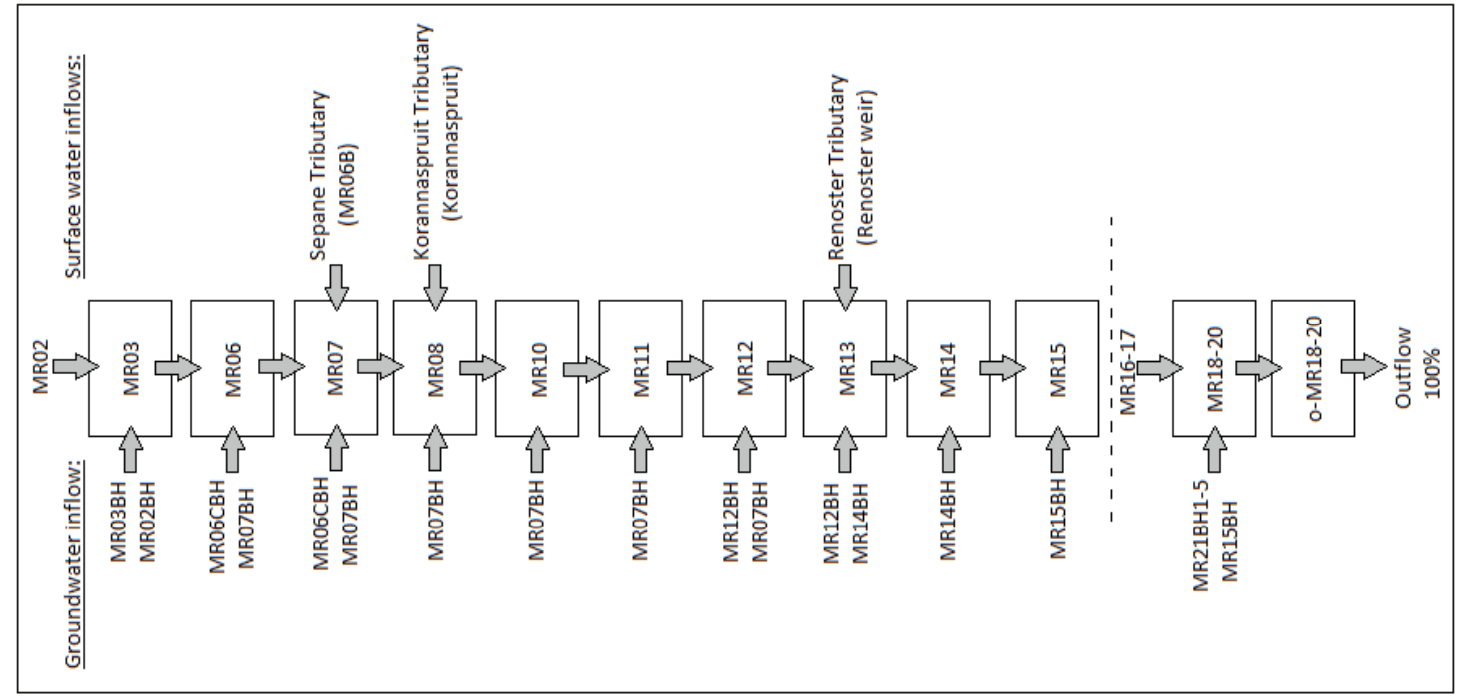


Figure 4-1-16 The box model conceptualisation of the flow system over the entire middle Modder River study area based on sample data over January 2013.

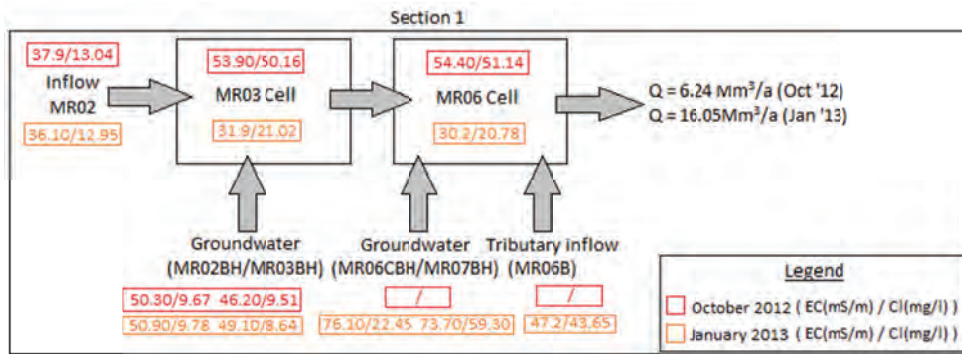


Figure 4-1-17 The representative EC and chloride (Cl) values for Section 1 flow conceptualisation. Concentrations indicate that groundwater could be contributing to the streamflow because the EC-value and chloride concentration increase as moving downstream.

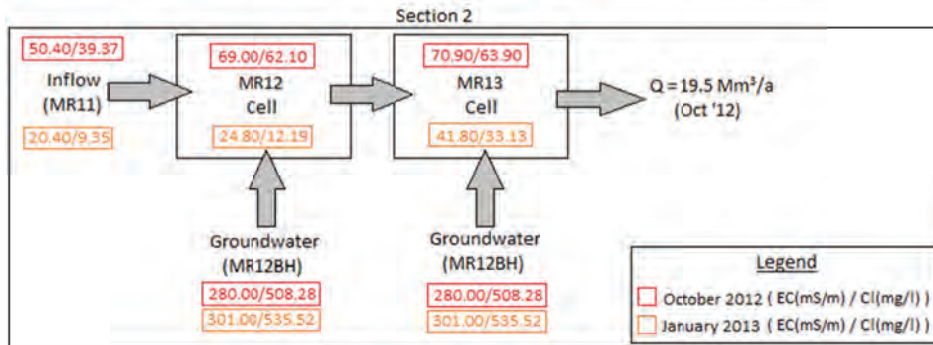


Figure 4-1-18 The representative EC and chloride (Cl) values for Section 2 flow conceptualisation. Concentration trends indicate that groundwater could be contributing to the streamflow.

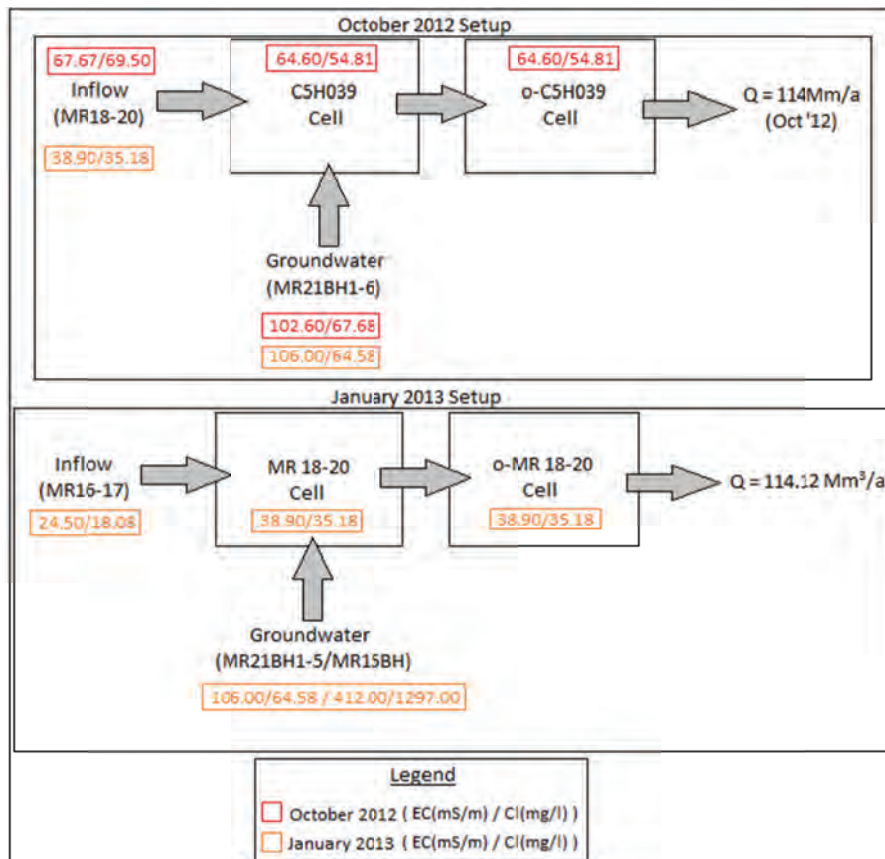


Figure 4-1-19 The representative EC and chloride (Cl) values for Section 3 flow conceptualisation, which varies for each time period due to data restrictions. Concentration trends in both conceptualisations could indicate that groundwater is contributing to streamflow.

4.1.9. Catchment Scale MCM Results

The MCM run for Section 1 using chemical data from October 2012 was unable to produce an output because the inflow sources to the defined cells were insufficient. The data collected in the January 2013 sample run did sufficiently define the inflow sources to ensure a complete MCM run. Using scenario 1 weighting factors and the January 2013 data set, the MCM estimated the groundwater inflow from the defined groundwater sources MR02BH, MR03BH, MR06CBH and MR07BH as 0.00 Mm³/a, 0.04 Mm³/a, 0.00 Mm³/a and 0.15 Mm³/a, respectively (Table 4-1-9). The sum of all the groundwater baseflow volumes estimated is 0.19 Mm³/a for the Section 1 area of the Modder River. The upstream river inflow defined by the sample MR02 to Cell 1 (MR03 Cell) was determined to be 8.09 Mm³/a, while the outflow from this Cell to the downstream MR06 Cell was determined to be 8.54 Mm³/a. The tributary inflow from the Sepane River to the MR06 Cell was determined to be 5.49 Mm³/a. Outflow from the MR06 Cell was defined at 16.05 Mm³/a from monthly flow data from flowstation C5H003. The water balance error for Section 1 using scenario 1 is 19.26% (Table 4-1-9). Using scenario 2 for the same flow setup and data, results in groundwater inflow from the defined groundwater sources MR02BH, MR03BH, MR06CBH and MR07BH as 0.00 Mm³/a, 0.08 Mm³/a, 0.00 Mm³/a and 0.01 Mm³/a, respectively (Table 4-1-9). The sum of all the groundwater inflow volumes estimated is then 0.09 Mm³/a for the Section 1 area of the Modder River. The upstream river inflow defined by the sample MR02 to Cell 1 (MR03 Cell) was determined to be 8.93 Mm³/a, while the outflow from this Cell to the downstream MR06 Cell was determined to be 10.33 Mm³/a. The tributary inflow to the MR06 Cell was determined to be 4.33 Mm³/a. Outflow from the MR06 Cell was defined at 16.05 Mm³/a from monthly flow data from flowstation C5H003. The water balance error for Section 1 using scenario 2 is 21.71% (Table 4-1-9).

Table 4-1-9 Summary of the MCM output for Section 1 for the weighting factor scenarios 1 and 2, indicating the quantified inflows and the associated water balance error.

Section 1				
Sample period	Cell	Name of inflow	Rate of inflow (Mm ³ /a)	
			Scenario 1	Scenario 2
Jan '13	Cell 1	MR02	8.09	8.93
		MR02BH	0.00	0.00
		MR03BH	0.04	0.08
	Cell 2	MR06B	5.49	4.33
		MR06CBH	0.00	0.00
		MR07BH	0.15	0.01
Water balance error (%)			19.26%	21.71%

The results from the Section 1 MCM application making use of the January 2013 dataset and utilizing scenario 2 weighting factors are graphically represented in Figure 4-1-20 and expressed as percentages of the assigned total outflow. The January 2013 MCM application determined the inflow from the upstream river source MR02 to the MR03 Cell at 55.6% of the total outflow from the system. The groundwater contribution to streamflow within the MR03 Cell was estimated at 0.0% from the MR02BH defined groundwater source and 0.5% from the MR03BH defined groundwater source. Flow from the MR03 Cell to the downstream MR06 Cell was determined at 64.3% of the total flow. The groundwater contribution to streamflow within the MR06 Cell was estimated at 0.0% from both the MR06CBH and MR07BH defined groundwater sources, while the tributary inflow to the cell was estimated at 27%.

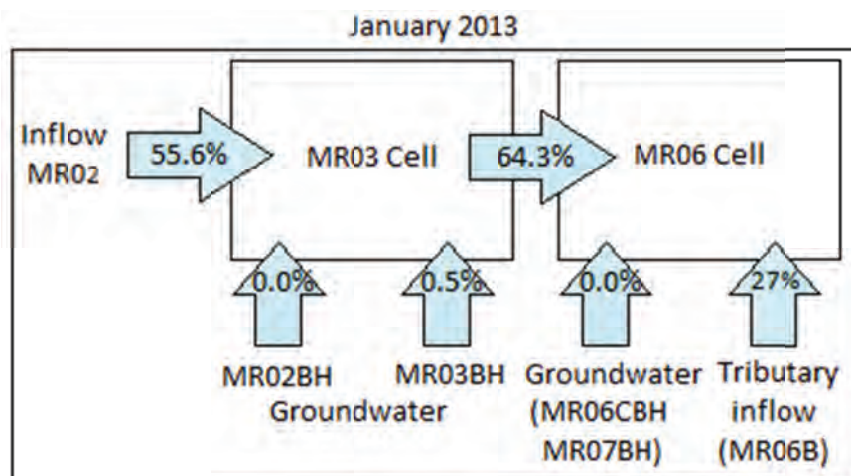


Figure 4-1-20 The unknown fluxes determined by the MCM for the January 2013 application on Section 1, making use of the scenario 2 weighting factors, expressed as a percentage of the total flow volume.

It was possible to run the MCM for both datasets from the two time periods for Section 2 as the inflows were sufficiently defined. Using the weighting factor scenario 1 and the October 2012 dataset for Section 2, the groundwater inflow from the defined groundwater source MR12BH is estimated at 1.56 Mm³/a and 0.07 Mm³/a for cells MR12 and MR13, respectively (Table 4-1-10). The sum of all the groundwater baseflow volumes estimated is 1.63 Mm³/a for the Section 2 area of the Modder River. The upstream river inflow defined by the sample MR11 to Cell 1 (MR12 Cell) was determined to be 15.99 Mm³/a, while the outflow from this Cell to the downstream Cell 2 (MR13 Cell) was determined at 19.50 Mm³/a. Outflow from the MR13 Cell was defined at 19.50 Mm³/a from monthly flow data from flowstation C5H053. The water balance error in October 2012 model run for Section 2 using scenario 1 is 12.11% (Table 4-1-10). Using the weighting factor scenario 2 and the October 2012 dataset for Section 2, the groundwater inflow from the defined groundwater source MR12BH is estimated at 1.33 Mm³/a and 0.00 Mm³/a for cells MR12 and MR13, respectively (Table 4-1-10). The sum of all the groundwater baseflow volumes estimated is 1.33 Mm³/a for the Section 2 area of the Modder River. The upstream river inflow defined by the sample MR11 to Cell 1 (MR12 Cell) was determined to be 18.63 Mm³/a, while the outflow from this Cell to the downstream Cell 2 (MR13 Cell) was determined at 19.67 Mm³/a. Outflow from the MR13 Cell was defined at 19.50 Mm³/a from monthly flow data from flowstation C5H053. The water balance error in the October 2012 model run for Section 2 using scenario 2 is 2.63% (Table 4-1-10).

Using the weighting factor scenario 1 and the January 2013 dataset for Section 2, the groundwater inflow from the defined groundwater source MR12BH is estimated at 0.36 Mm³/a and 0.92 Mm³/a for cells MR12 and MR13, respectively (Table 4-1-10). The sum of all the groundwater baseflow volumes estimated is 1.28 Mm³/a for the Section 2 area of the Modder River. The upstream river inflow defined by the sample MR11 to Cell 1 (MR12 Cell) was determined to be 16.64 Mm³/a, while the outflow from this Cell to the downstream Cell 2 (MR13 Cell) was determined at 18.96 Mm³/a. Outflow from the MR13 Cell was defined at 19.50 Mm³/a from monthly flow data from flowstation C5H053. The water balance error in the January 2013 model run for Section 2 using scenario 1 is 7.23% (Table 4-1-10). Using the weighting factor scenario 2 and the January 2013 dataset for Section 2, the groundwater inflow from the defined groundwater source MR12BH is estimated at 0.32 Mm³/a and

0.96 Mm³/a for cells MR12 and MR13, respectively (Table 4-1-10). The sum of all the groundwater baseflow volumes estimated is 1.28 Mm³/a for the Section 2 area of the Modder River. The upstream river inflow defined by the sample MR11 to Cell 1 (MR12 Cell) was determined to be 19.78 Mm³/a, while the outflow from this Cell to the downstream Cell 2 (MR13 Cell) was determined at 20.05 Mm³/a. Outflow from the MR13 Cell was defined at 19.50 Mm³/a from monthly flow data from flowstation C5H053. The water balance error in the January 2013 model run for Section 2 using scenario 1 is 2.75% (Table 4-1-10).

Table 4-1-10 Summary of the MCM output for Section 2 for the weighting factor scenarios 1 and 2, indicating the quantified inflows and the associated water balance error.

Section 2				
Sample period	Cell	Name of inflow	Rate of inflow (Mm ³ /a)	
			Scenario 1	Scenario 2
Oct '12	Cell 1	MR11	15.99	18.63
		MR12BH	1.56	1.33
	Cell 2	MR12BH	0.07	0.00
Water balance error (%)			12.11%	2.63%
Jan '13	Cell 1	MR11	16.64	19.78
		MR12BH	0.36	0.32
	Cell 2	MR12BH	0.92	0.96
Water balance error (%)			7.23%	2.75%

The results from the Section 2 MCM application on both temporal scales utilizing scenario 2 weighting factors are graphically represented in Figure 4-1-21 and expressed as percentages of the assigned total outflow. The October 2012 MCM application determined the inflow from the upstream river source MR11 to the MR12 Cell to be 97.8% of the total outflow from the system. The groundwater contribution to streamflow within the MR12 Cell, from the MR12BH defined groundwater source, was estimated at 6.9% of the total assigned flow. Flow from the MR12 Cell to the downstream MR13 Cell was determined at 103.2% of the total flow. The groundwater contribution to streamflow within the MR13 Cell, from the MR12BH defined groundwater source, was estimated at 0.0%. The January 2013 MCM application determined the inflow from the upstream river source MR11 to the MR12 Cell at 101.4% of the total outflow from the system. The groundwater contribution to streamflow within the MR12 Cell, from the MR12BH defined groundwater source, was estimated at 1.6% of the total assigned outflow. Flow from the MR12 Cell to the downstream MR13 Cell

was determined at 102.8% of the total flow. The groundwater contribution to streamflow within the MR13 Cell, from the MR12BH groundwater source, was estimated at 4.9% of the total assigned outflow.

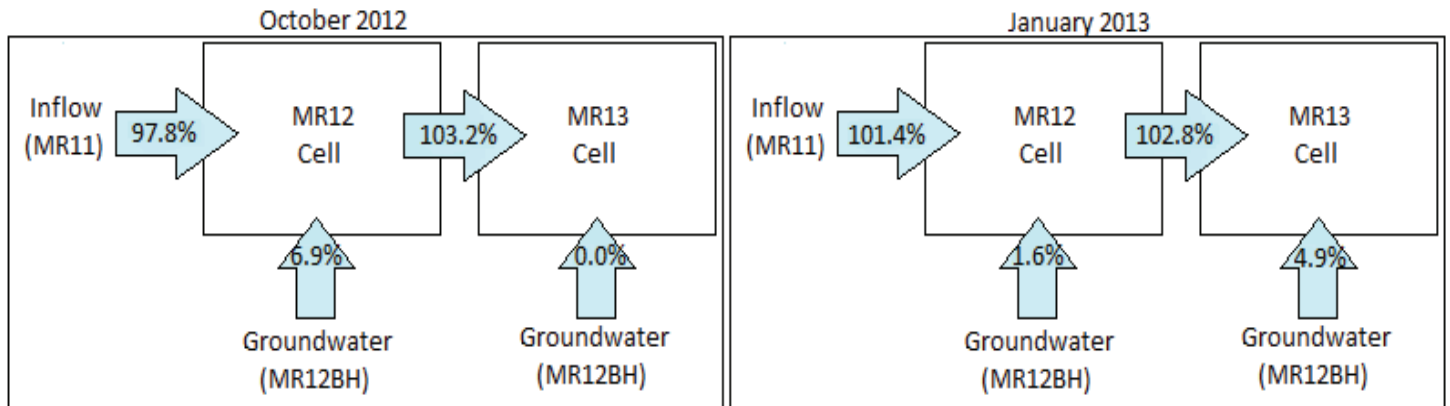


Figure 4-1-21 The unknown fluxes determined by the MCM for both the October 2012 and January 2013 applications on Section 2, making use of the scenario 2 weighting factors, expressed as a percentage of the total flow volume.

The model setup for Section 3 differs for the two time periods because water quality data from the flowstation C5H039 sample point was not available for the January 2013 time period. Using weighting factor scenario 1, the original flow setup and the October 2012 dataset for Section 3, inflow from the groundwater source defined by the average tracer concentration from sampled boreholes MR21BH1-6 is estimated at 2.87 Mm³/a (Table 4-1-11). The upstream river inflow defined by the samples MR18-20, to Cell 1 (C5H039 Cell) was determined to be 100.77 Mm³/a, while the outflow from this Cell to the downstream fictitious Cell 2 (o-C5H039 Cell) was determined at 112.30 Mm³/a. Outflow from the fictitious cell was defined at 114.12 Mm³/a from monthly flow data from flowstation C5H039. The water balance error in the October 2012 model run for Section 3 using scenario 1 is 9.88% (Table 4-1-11). Using the weighting factor scenario 2, the original flow setup and the October 2012 dataset for Section 3, inflow from the groundwater source defined by the average tracer concentration from sampled boreholes MR21BH1-6 was estimated at 2.61 Mm³/a (Table 4-1-11). The upstream river inflow defined by the samples MR18-20, to Cell 1 (C5H039 Cell) was determined to be 102.66 Mm³/a, while the outflow from this Cell to the downstream fictitious Cell 2 (o-C5H039 Cell) was determined at 113.34 Mm³/a. Outflow from the fictitious cell was defined at 114.12 Mm³/a from monthly flow

data from flowstation C5H039. The water balance error in the October 2012 model run for Section 3 using scenario 2 is 8.55% (Table 4-1-11).

Using the weighting factor scenario 1, the flow setup to compensate for the lack of chemical data at C5H039 and the January 2013 dataset for Section 3, inflow from the groundwater sources defined by the average tracer concentration from sampled boreholes MR21BH1-6 and MR15BH, are estimated at 0.00 Mm³/a and 2.38 Mm³/a, respectively (Table 4-1-11). The upstream river inflow defined by the samples MR16-17, to Cell 1 (MR18-20 Cell) was determined to be 24.67 Mm³/a, while the outflow from this Cell to the downstream fictitious Cell 2 (o-MR18-20 Cell) was determined at 79.94 Mm³/a. Outflow from the fictitious cell was defined at 114.12 Mm³/a from monthly flow data from flowstation C5H039. The water balance error in the January 2013 model run for Section 3 using scenario 1 is 76.50% (Table 4-1-11). Using the weighting factor scenario 2, the flow setup to compensate for the lack of chemical data at C5H039 and the January 2013 dataset for Section 3, inflow from the groundwater sources defined by the average tracer concentration set from sampled boreholes MR21BH1-6 and MR15BH, are estimated at 0.00 Mm³/a and 2.13 Mm³/a, respectively (Table 4-1-11). The upstream river inflow defined by samples MR16-17, to Cell 1 (MR18-20 Cell) was determined to be 62.13 Mm³/a, while the outflow from this Cell to the downstream fictitious Cell 2 (o-MR18-20 Cell) was determined at 92.83 Mm³/a. Outflow from the fictitious cell was defined at 114.12 Mm³/a from monthly flow data from flowstation C5H039. The water balance error in the January 2013 model run for Section 3 using scenario 2 is 44.18% (Table 4-1-11).

Table 4-1-11 Summary of the MCM output for Section 3 for the weighting factor scenarios 1 and 2, indicating the quantified inflows and the associated water balance error.

Section 3				
Sample period	Cell	Name of inflow	Rate of inflow (Mm ³ /a)	
			Scenario 1	Scenario 2
Oct '12	Cell 1	MR18-20	100.77	102.66
		MR21BHave	2.87	2.61
Water balance error (%)			9.88%	8.55%
Jan '13	Cell 1	MR15BH	2.38	2.13
		MR16-17	24.67	62.13
		MR21BHave	0	0
Water balance error (%)			76.50%	44.18%

The results from the Section 3 MCM application on both temporal scales utilizing scenario 2 weighting factors are graphically represented in Figure 4-1-22 and expressed as percentages of the assigned total outflow. The October 2012 MCM application determined the inflow from the upstream river source defined by river samples MR18-20 to the C5H039 Cell at 89.95% of the total outflow from the system. The groundwater contribution to streamflow within the C5H039 Cell was estimated at 2.3% from the MR21BH1-6 defined groundwater source. Flow from the C5H039 Cell to the downstream fictitious o-C5H039 Cell was determined at 116.8% of the total flow. The January 2013 MCM setup differs from the October 2012 setup due to data restrictions. This MCM run determined the inflow from the upstream river source defined by the average of samples MR16-17 to the MR18-20 Cell at 54.4% of the total outflow from the system. The groundwater contribution to streamflow within the MR18-20 Cell was estimated at 1.8% from the MR21BH1-6 and MR15BH defined groundwater sources. Flow from the MR18-20 Cell to the downstream fictitious o-MR18-20 Cell was determined at 81.3% of the total flow.

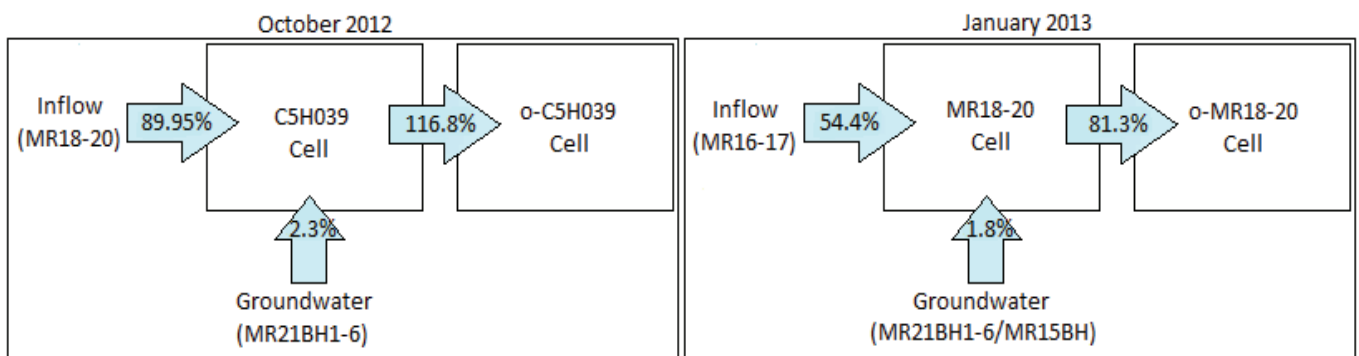


Figure 4-1-22 The unknown fluxes determined by the MCM for both the October 2012 and January 2013 applications on Section 3, making use of the scenario 2 weighting factors, expressed as a percentage of the total flow volume. The two temporal scale applications differ in setup due to data restrictions.

The continuous flow system setup from MR01-MR15 and MR16-MR20 using data from January 2013 was used to perform a MCM run which defined the outflow from the system at 100%, instead of assigning an outflow volume. The quantified inflows to the system are expressed as percentages relative to this 100% outflow. The results from this model run, making use of scenario 1, are graphically displayed along with the flow system diagram (Figure 4-1-23). The Renoster tributary sample is not included in the model run as this sample insufficiently defines an additional inflow to the system and results in an incomplete model run. Cell 1 (MR03 Cell) receives 2.32% inflow from the upstream MR02 source, and

receives 0.02% and 0.00% from groundwater sources MR03BH and MR02BH. Cell 2 (MR06 Cell) receives 2.68% upstream inflow from the MR03 Cell, and receives 0.15% and 0.00% groundwater inflow from MR06CBH and MR07BH, respectively. Cell 3 (MR07 Cell) receives an upstream inflow of 3.20% from the MR06 Cell as well as an additional surface water inflow from the Sepane tributary at 2.3%. The groundwater inflow to the MR07 Cell is 0.00% and 0.32% from MR06CBH and MR07BH, respectively. Cell 4 (MR08 Cell) also receives an additional surface water inflow from the Korannaspruit tributary at 53.76% and an upstream inflow from the MR07 Cell at 6.42%. The groundwater inflow to this cell is 1.47% from MR07BH. Cell 5 (MR10 Cell) receives 63.94% upstream inflow from the MR08 Cell, with a groundwater contribution at 0.00% from MR07BH. Cell 6 (MR11 Cell) receives 69.60% from the upstream cell MR10 Cell and a groundwater inflow of 1.28% from the MR07BH defined groundwater source. Cell 7 (MR12 Cell) receives 74.53% inflow from the upstream cell MR11 and receives groundwater inflow from the defined sources, MR12BH and MR07BH at 1.10% and 0.00%, respectively. Cell 8 (MR13 Cell) receives 77.77% upstream river inflow from the MR12 Cell and groundwater inflow of 3.19% and 2.02% from MR12BH and MR14BH, respectively. The downstream cell 9 (MR14 Cell) receives inflow from the MR13 Cell at 77.99% and 1.48% groundwater inflow from MR14BH defined source. The last cell, MR15 Cell, receives 84.73% of upstream river inflow from the upstream MR14 Cell and 0.41% groundwater inflow from MR15BH, while the outflow from this cell is set at 100%. There is a break in the continuous flow model run at this point as the sources are insufficiently defined for a complete model run. An adjacent model run is thus performed for the remaining sample data (MR16-MR20), with the outflow of this section also set to 100%. Cell 1 of the adjacent model run is MR18-20, which receives an upstream inflow of 54.51% from the MR16-17 defined source and groundwater inflow from MR21BH1-5 and MR15BH at 0.00% and 1.87%, respectively. This cell flows into a fictitious last cell (o-MR18-20) at a rate of 81.44%. The outflow from the fictitious cell is set to 100%.

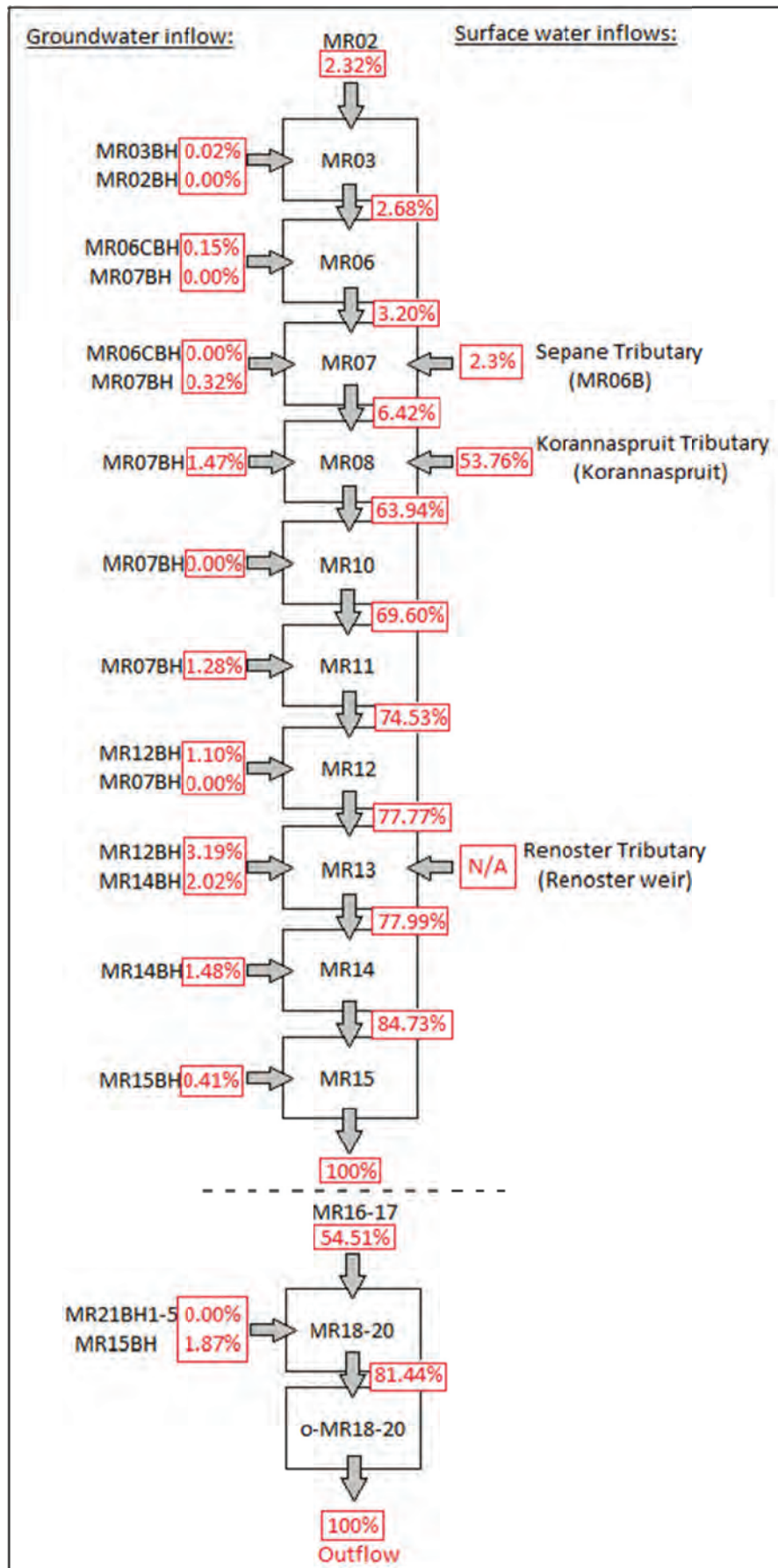


Figure 4-1-23 The box model diagram of the flow system conceptualised for the continuous flow MCM run, with the quantified percentage flows

4.1.10. Catchment Scale MCM Discussion and Comparison

The total sum of the quantified groundwater inflows into each section of the Modder River are summarised in Table 4-1-12 for each weighting factor scenario and time period. Table 4-1-12 includes the water balance and average chemical mass balance errors associated with the quantified groundwater baseflow volume for each model run. Appendix C Table C-8 is a detailed chemical mass balance percentage error report on each chemical constituent for all model runs. There are no results for Section 1 making use of the October 2012 data as the chemical data sampled was not sufficient for a MCM run. The MCM quantification of groundwater baseflow for Section 1 using January 2013 data shows a decrease from 0.19 Mm³/a to 0.09 Mm³/a when changing from scenario 1 to scenario 2. This decrease in groundwater baseflow is however accompanied with an increase in the water and chemical balance percentage error. This increase in both water and chemical mass balances is unexpected and opposite to the response of all other model runs, where a decrease in water and chemical mass balance percentage errors are seen when scenario 2 is implemented. This increase in percentage error could be attributed to the fact that an important tracer constituent has been given a lower weighting factor, or that the flow system has been insufficiently defined with either the omission of an inflow source or inaccurate river cell definition, among other reasons. The MCM quantification of groundwater baseflow for Section 2 using October 2012 data shows a decrease from 1.63 Mm³/a to 1.33 Mm³/a when changing from scenario 1 to scenario 2 accompanied with a decrease in both the water and chemical mass balance percentage errors. However, the groundwater baseflow estimate for this section using January 2013 data does not show a change in the sum of the groundwater inflows when changing from scenario 1 to scenario 2, even though the portion of the inflow from the two different sources changes slightly (Table 4-1-10). The water and chemical balance percentage errors both decrease for this model application though, when scenario 2 is implemented. The MCM quantification of groundwater baseflow for Section 3 using October 2012 data shows only a slight decrease from 2.87 Mm³/a to 2.61 Mm³/a when changing from scenario 1 to scenario 2. The change from scenario 1 to scenario 2 is also accompanied with a slight decrease in both the water and chemical mass balance percentage errors. The groundwater baseflow estimate for this section using January 2013 data also shows a slight decrease from 2.38 Mm³/a to 2.13

Mm³/a, but is accompanied by a large decrease in the water and chemical mass balance percentage errors. The percentage errors are extremely high for this model run, which could be due to the fact that a differently defined flow setup was used, because data used in the October 2012 model run was not available for January 2013. The Section 3 study area mainly comprises the Krugersdrift Dam the Modder River where one would expect estimations of flow volumes to become more variable as natural flow is no longer taking place. Considering the various changes in the groundwater baseflow estimates and the water and chemical mass balance percentage errors, scenario 2 seems a reasonable selection of weighting factors as it results a lower associated error in most cases and by incorporating lower weighting factors attempts to decrease the infringement of the natural system on one of the main assumptions of the MCM, that of conservative tracers.

Table 4-1-12 The total groundwater baseflow estimates for each section and time period using both scenarios with the associated water and chemical mass balance percentage errors.

Section	GW Baseflow (Mm ³ /a)		Water balance error (%)		Average chemical mass balance error (%)	
	Scenario 1	Scenario 2	Scenario 1	Scenario 2	Scenario 1	Scenario 2
Section 1 (Jan '13)	1.49	0.96	19.3%	21.7%	6.2%	11.3%
Section 2 (Oct '12)	1.63	1.33	12.1%	2.6%	13.4%	9.5%
Section 2 (Jan '13)	1.28	1.28	7.2%	2.8%	7.4%	5.0%
Section 3 (Oct '12)	2.87	2.61	9.9%	8.6%	4.2%	3.7%
Section 3 (Jan '13)	2.38	2.13	76.5%	44.2%	60.3%	26.7%

Section 1, the segment of the Modder River from the Rustfontein Dam to the flowstation C5H003, falls within the quaternary catchment C52B. Similarly, Section 2 and Section 3 fall within the quaternary catchments C52E and C52G, respectively. Groundwater baseflow estimates from the Pitman, Hughes and Sami models for each of the quaternary catchments as well as an estimate from another method incorporating water quality data, namely the Tracer method, using the same data as the MCM are shown in Table 4-1-13. The Tracer method estimate is shown for both the use of only the electrical conductivity (EC) tracer as well as an average value from using a number of tracers. The groundwater baseflow estimate from the MCM and the Tracer method is based on water quality data from both October 2012 and January 2013. The Pitman, Sami and Hughes model groundwater baseflow estimates for the quaternary catchment C52B are 0.00 Mm³/a, 0.03 Mm³/a and

5.03 Mm³/a, respectively (Table 4-1-13). The Tracer method groundwater baseflow estimate for Section 1 using data from October 2012 and only EC as a tracer is 0.51 Mm³/a, while for an average from a number of tracers is 1.24 Mm³/a. Similarly, using data from January 2013 results in a groundwater estimate of 2.12 Mm³/a for EC alone and 1.61 Mm³/a for an average from a number of tracers. The MCM groundwater baseflow estimate for section 1 is 0.09 Mm³/a based on data from January 2013. From the various groundwater estimates for Section 1, it can be seen that the Tracer method produces variable results between the two time periods as well as between using EC alone and using an average of a number of tracers. The MCM and the Tracer method found groundwater baseflow to be contributing to the Modder River within this section unlike the Pitman model that estimated zero groundwater baseflow. The MCM and Tracer method found slightly more baseflow than estimated by the Sami model at 0.03 Mm³/a, but much less groundwater baseflow than was found in the Hughes model at 5.03 Mm³/a.

The Pitman, Sami and Hughes model groundwater baseflow estimates for the quaternary catchment C52E are 0.00 Mm³/a, 0.00 Mm³/a and 2.22 Mm³/a, respectively (Table 4-1-13). The MCM estimates the groundwater baseflow for this section at 1.33 Mm³/a in October 2012 and 1.28 Mm³/a in January 2013. The Tracer method estimates the groundwater baseflow for EC alone and multiple tracers at 0.21 Mm³/a and 1.35 Mm³/a, respectively for October 2012 and 1.35 Mm³/a and 2.13 Mm³/a for January 2013. The tracer method proves to be quite variable with the MCM producing a fairly constant value for both time periods. The MCM and the tracer method found groundwater baseflow to be contributing to the Modder River within this section unlike the Pitman and Sami models that both estimated 0.00 Mm³/a groundwater baseflow, but found less groundwater baseflow than the Hughes model that estimates the groundwater baseflow at 2.22 Mm³/a.

Pitman, Sami and Hughes model groundwater baseflow estimates for the quaternary catchment C52G are 0.00 Mm³/a, 0.00 Mm³/a and 5.35 Mm³/a, respectively (Table 4-1-13). The MCM estimates the groundwater baseflow for this section at 2.61 Mm³/a in October 2012 and 2.13 Mm³/a in January 2013. The Tracer method estimates the groundwater baseflow for EC alone and multiple tracers at 0.04 Mm³/a and 0.61 Mm³/a, respectively in October 2012 and 4.24 Mm³/a and 4.25 Mm³/a in January 2013. The Tracer method seems to be more stable between using EC alone and a number of tracers for this section, but quite

variable between time periods. The MCM produces similar groundwater estimates for both time periods. Both water quality methods found groundwater baseflow contributing to the river, while the Pitman and Sami model estimate zero groundwater baseflow to this section of the Modder River. On the other hand, the water quality methods indicate much less groundwater baseflow than the Hughes model at 5.35 Mm³/a.

Table 4-1-13 Groundwater baseflow estimates from the Pitman model, Sami model, Hughes model, Tracer method and MCM for Section 1 (C52B), Section 2 (C52E) and Section 3 (C52G).

Quat/ Section	Pitman uncalibrated (Mm ³ /a)	Sami uncalibrated (Mm ³ /a)	Hughes uncalibrated (Mm ³ /a)	MCM (Mm ³ /a)		Tracer Method			
				Oct '12	Jan '13	Oct '12		Jan '13	
						EC	Average	EC	Average
C52B/ Section 1	0.00	0.03	5.03	/	0.09	0.51	1.24	2.12	1.61
C52E/ Section 2	0.00	0.00	2.22	1.33	1.28	0.21	1.35	1.35	2.13
C52G/ Section 3	0.00	0.00	5.35	2.61	2.13	0.04	0.61	4.24	4.25

Considering the results from these three sections, the trend seen is for the MCM groundwater baseflow volume to seem like an over-estimate when compared to the Sami and Pitman model estimates, but to seem like an under-estimate when compared to the Hughes model estimate. Considering work done by Welderufael and Woyessa (2010) which found that baseflow contributed on average 71% of the total streamflow in the quaternary catchment C52A using four baseflow separation techniques and work by Gomo (2011) also reporting the Modder River is a gaining stream at the base of the Krugersdrift Dam, it seems reasonable to assume that there is some groundwater contributing to the baseflow of the river, even if it is in small quantities.

The continuous flow model run was performed as if the study area was an ungauged catchment. The model run shows an overall groundwater contribution to this approximately 130km section of the Modder River of 11.44% of the total river flow at MR15, and an additional 1.87% of the total flow at the Krugersdrift dam (Figure 4-1-23). The water and chemical mass balance errors associated with these model runs are high at 30.87% and 13.89% for MR01-MR15, and 44.18% and 26.71% for MR16-MR20, respectively. The contribution of the Korannaspruit River tributary seems to have been over-estimated at 53.76% of the total river flow. This over-estimate could be attributed to insufficiently

defined runoff sources which have resulted in a large % being attributed to this single source. However, the MCM does seem to give reasonable groundwater inflow estimates to the river system. The MCM could thus serve as an initial estimate method for the contribution of groundwater to baseflow in ungauged flow systems.

4.2. Pilot Study 2: Limpopo Quaternary Catchments A42A-A42C

4.2.1. Overview

The quaternary catchments A42A, A42B and A42C, fall within the Limpopo Water Management Area (WMA) of the Limpopo Province, South Africa (Figure 4-2-1). The quaternary catchments form the upper most source area of the Moloko River, which is the highest yielding river in the WMA. The Sand River in quaternary A42A, the Grootspuit and Sandspruit Rivers in A42B and the Klein Sand River in A42C are tributaries to the Moloko River. The average annual rainfall within this area is between 400 mm and 700 mm. The total annual rainfall was 647 mm and the average annual evaporation was 1582.9 mm in the year 2005, the selected period of investigation. The average annual temperature ranges from 14°C to 20°C. The area is characterised by a flat open landscape surrounded by mountains, where streams flow through steep, rocky areas. The topography varies from 1200 mamsl in the river valley to 1700 mamsl in the surrounding mountains. The predominant vegetation is Waterberg Moist Mountain Bushveld and Mixed Bushveld (Department of Environmental Affairs and Tourism, 2006). The warms database for the A42 quaternaries indicate that the abstraction from surface water resources in the area per annum is 33.4 Mm³.

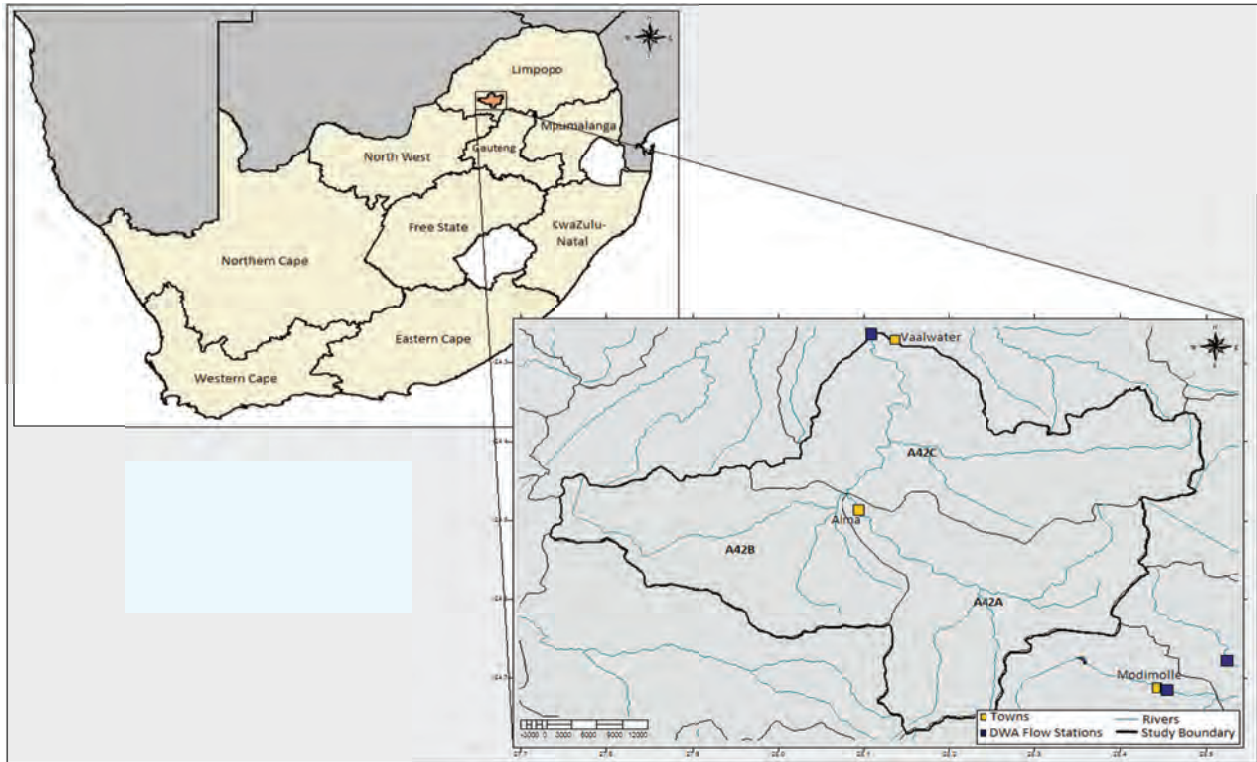


Figure 4-2-1 The position of the quaternaries A42A, A42B and A42C within South Africa.

4.2.2. Geology

The outcrop geology of the area mainly consists of Waterberg and Rooiberg Group lithologies (Figure 4-2-2). Minor intrusions of the Lebowa Granite Suite are also present in the study area along with the Rooiberg Group they make up part of the Bushveld Magmatic Province. The Shrikkloof and Kwaggasnek Formations, representing the volcanic Rooiberg Group, are the oldest rocks in the study area (Table 4-2-1). A quartzite layer forms the base of the Kwaggasnek Formation, overlain by a thick layer of siliceous lavas (rhyolite) and capped by a laterally extensive shale/tuff layer that is underlain by a layer of volcanic breccias. The Shrikkloof Formation consists mainly of siliceous lavas (flow-banded rhyolite), with a layer of ash-flow tuff marking the top of the unit. The Lebowa Granite Suite intrudes above the Rustenburg Layered Suite of the Bushveld Complex, but does not outcrop with the study area. The Lebowa Granite Suite consists of several granite types, namely Nebo, Bobbejaankop, Klipkloof and Makhutso Granite. The small and isolated Glentig Formation outcropping in the study area consists of predominantly argillaceous, clastic sedimentary rocks with interbedded lavas and a basal conglomerate of reworked volcanic material. This formation used to be classed as the uppermost beds of the Transvaal Supergroup, but is now considered as proto-Waterberg deposits (Johnson, Anhaeusser and Thomas, 2006).

The Waterberg Group lithologies which outcrop within the study area form a complete stratigraphic column of the group (Table 4-2-1). The Waterberg Group lies unconformably on the rocks of the Bushveld Complex in the study area. The rocks of the Waterberg Group are usually dark greyish-red in colour. The oldest formation within the Waterberg group is the Swaershoek Formation which consists of mainly fractured arenites and rudites, and thought to have been deposited as a fan-delta, or inter fan-delta tidal flats. The Alma Formation overlies the Swaershoek Formation and is made up of a succession of various arkoses and feldspathic arenites, deposited as a series of alluvial fans. The overlying Skilpadkop Formation consists of thickly bedded immature lithic arenites, pebble rudites and minor quantities of arkose. The Aäsvöelkop Formation coarsens from arenaceous lutites at its base, overlying the Skilpadkop Formation, to arenites higher up, indicating deposition in a shallow inland lake environment. The Sandriviersberg Formation consists of arenite, rudite and interbedded pebble rudites. The Cleremont Formation is thought to have been deposited as arenaceous sediments along a shoreline, consisting of medium-grained well-sorted arenites. The Formation maintains a constant thickness of approximately 125 m. The Vaalwater Formation, the topmost formation of the Waterberg Group, comprises poorly exposed feldspathic arenites and lutites thought to have been deposited in a lower energy shelf setting (Johnson, Anhaeusser and Thomas, 2006).

A conceptual geological cross-section of the area is created based on the outcrop geology map (Figure 4-2-3). A general cross-section from the south to the north of the area is conceptualised as a folded landscape to account for the reverse geological sequence along with a decrease in topography. The outcrop geology also indicates the presence of a large fault to the south. The area is intruded by Lebowa granites and various basic intrusive lithologies, making for a complicated geology.

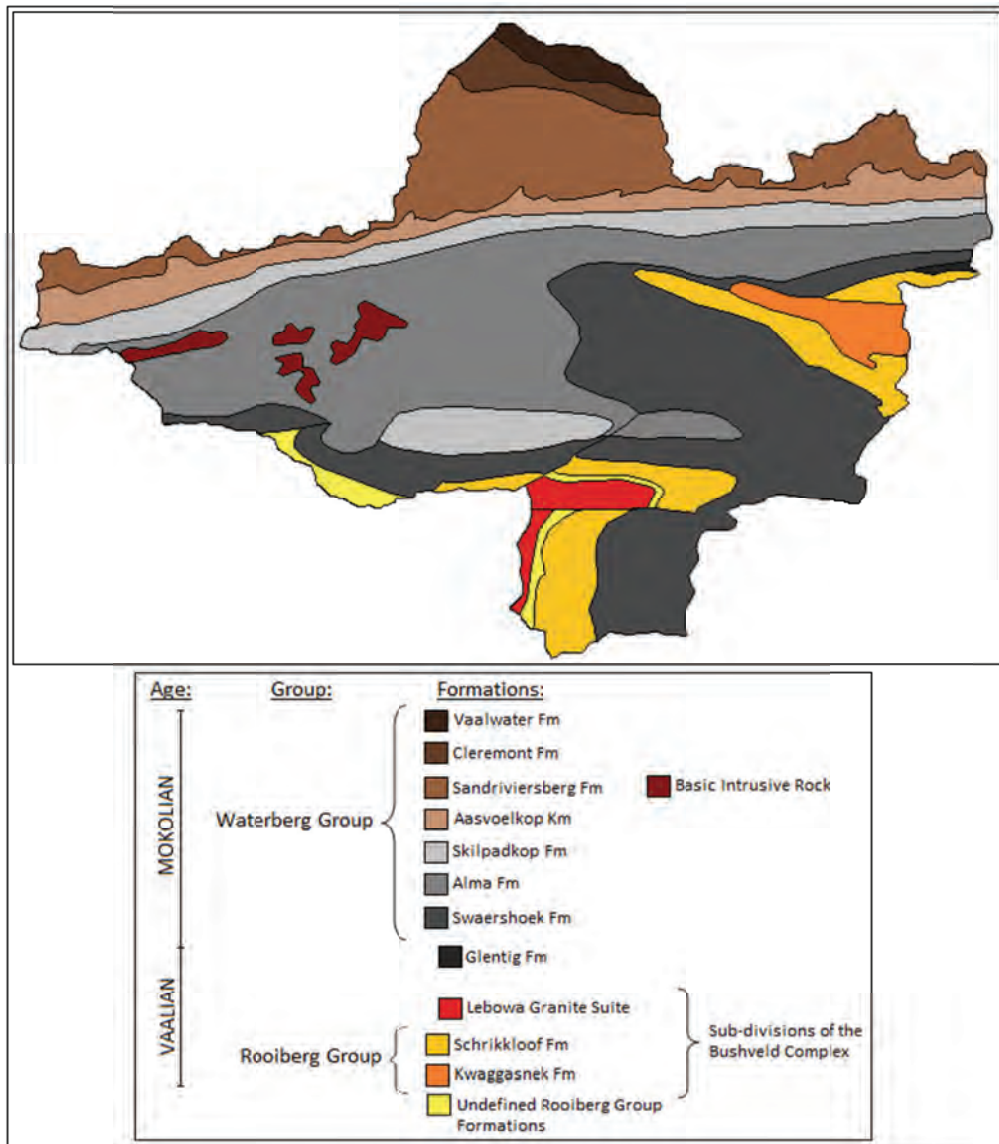


Figure 4-2-2 The outcrop geology of the A42 study area. (Based data from GRDM, 2010).

Table 4-2-1 The stratigraphic sequence within the A42 study area

Era	GROUP	SUB-GROUP	FORMTATIONS	THICKNESS
MESOZOIC	Waterberg	Kransberg	Vaalwater	<475 m
			Cleremont	~125 m
			Sandriversberg	1250 m
		Matlabas	Aasvoëlkop	<600 m
			Skilpad	<600 m
		Nylstroom	Alma	<3000 m
			Swaershoek	<1000 m
PALEOZOIC	Proto-Waterberg		Glentig	
	Rooiberg		Schrikkloof	>1000 m
			Kwaggasnek	>1000 m

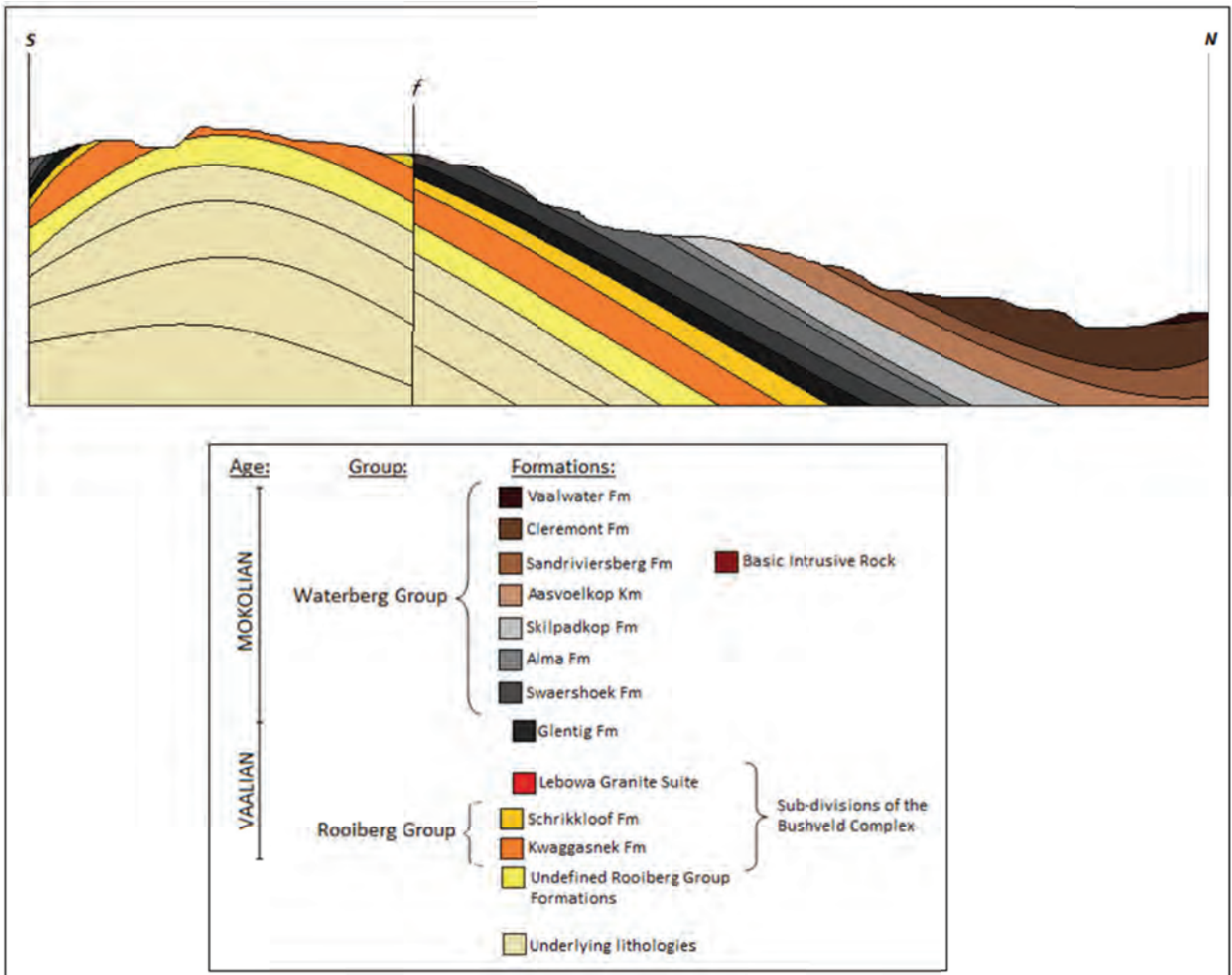


Figure 4-2-3 A conceptual geological cross-section of the A42 study area based on the outcrop geology

4.2.3. Hydrogeology

The topography of the A42 area (quaternary catchments A42A, A42B and A42C) varies between 1100 mamsl to 1680 mamsl above sea level. The western and eastern borders are mountainous with steeper topography that evens out towards the river valley in the centre (Figure 4-2-4). The topography to the southern border of the area indicates three river valleys which form one larger river valley towards the north which can be assumed to be the Moloko River. There is a general trend of decreasing topography from the south to the north. The groundwater level seems to follow the topography and there is a general gradient towards the river valley as well as towards the north (Figure 5-2-5).

The underlying Rooiberg Group lithologies within the study area consist of volcanic rocks, mostly rhyolite. Rhyolite is a fine-grained extrusive rock which would have little primary porosity due to the nature in which the rock is crystallised. Secondary features such as fractures would however allow for the movement of water through this lithology. The proto-Waterberg formation, Glentig, consists of mainly argillaceous sedimentary rocks which would also be an aquiclude or leaky aquifer based on the primary porosity of such lithologies. The overlying Waterberg Group lithologies on the other hand could provide good aquifers because the formations comprise of mostly arenites, rudites and arkoses which are sedimentary in nature.

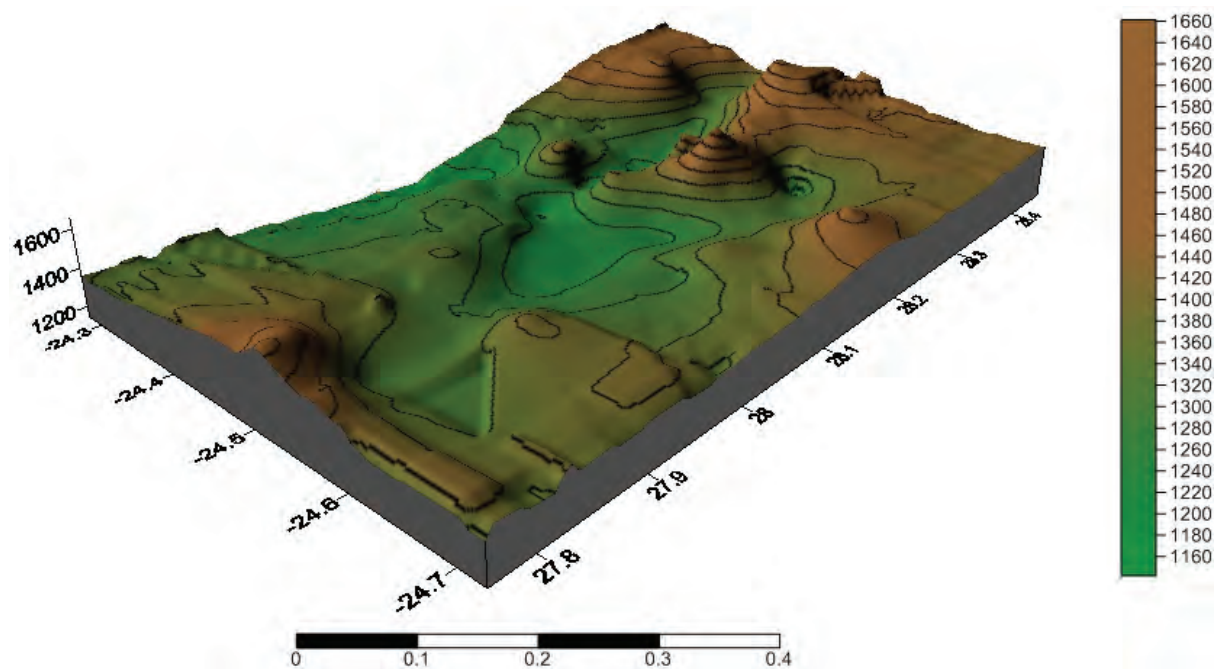


Figure 4-2-4 Generalised topography of the A42 area (quaternaries A42A-A42C).

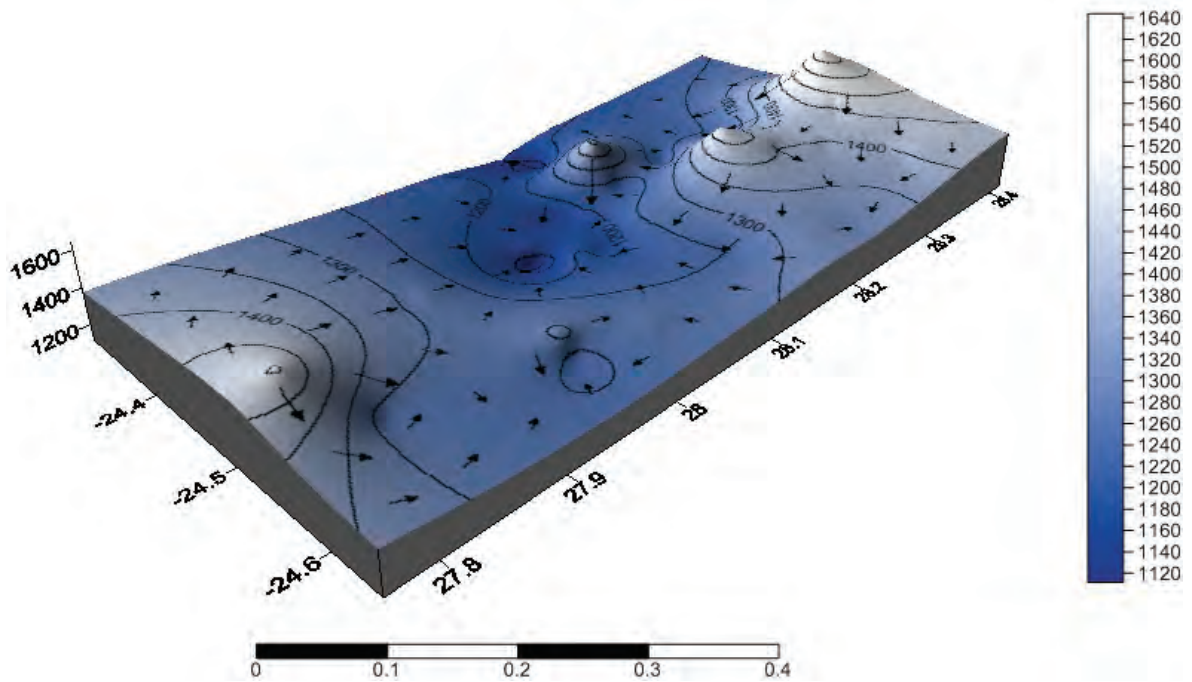


Figure 4-2-5 Generalised groundwater level within the A42 area based on data from 1993-1994.

4.2.4. Mixing Cell Model

The quaternary catchments A42A, A42B and A42C have an uneven distribution of river water and groundwater water quality data. The available groundwater quality data in the quaternary catchment A42B consists of a total of four borehole samples, one sample in 1979 and three in 1983 (Figure 4-2-6). The river water and groundwater quality data available for this time period in the other two quaternary catchments consists of one river sample point in A42C and one borehole sample in A42B. There are additional river water quality samples in the quaternary catchment A42C in 2005 as well as two borehole sample points, but there is no borehole or river water quality data in either A42A or A42B quaternary catchments during this time period (Figure 4-2-7). The MCM application is thus limited to a small area within the A42C quaternary catchment, where there is sufficient surface water and groundwater quality data from 2005.

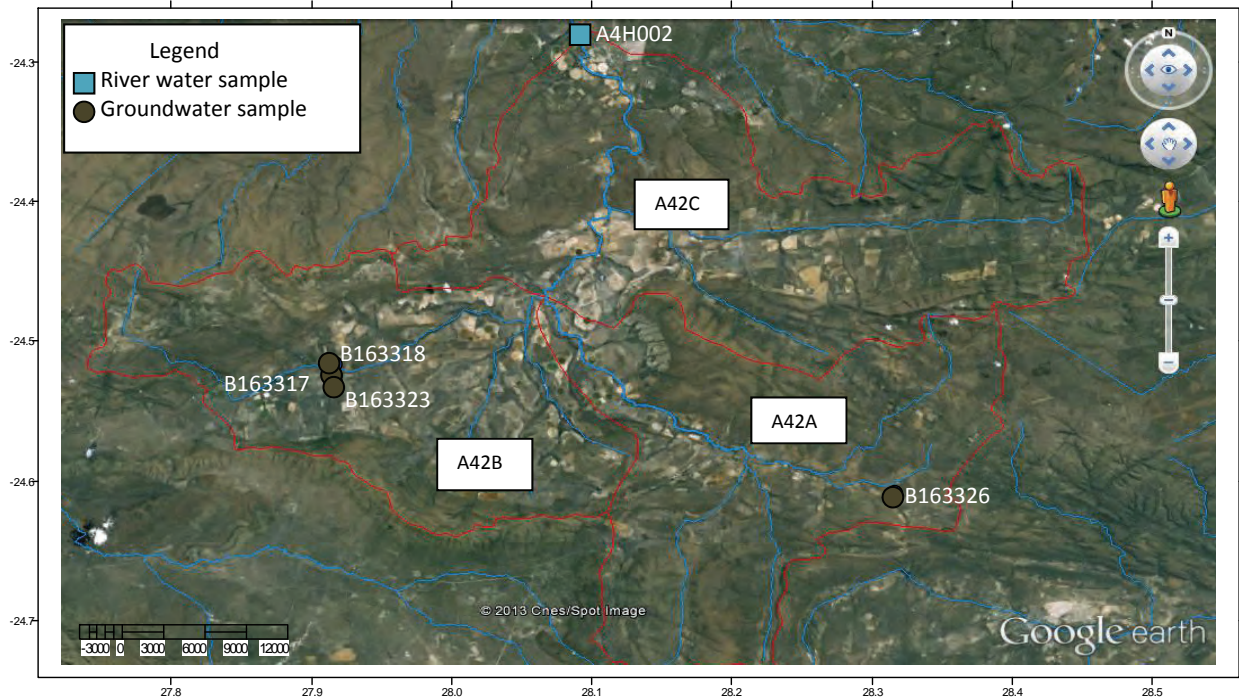


Figure 4-2-6 The river water and groundwater water quality data sample points for 1983 in the A42 area.

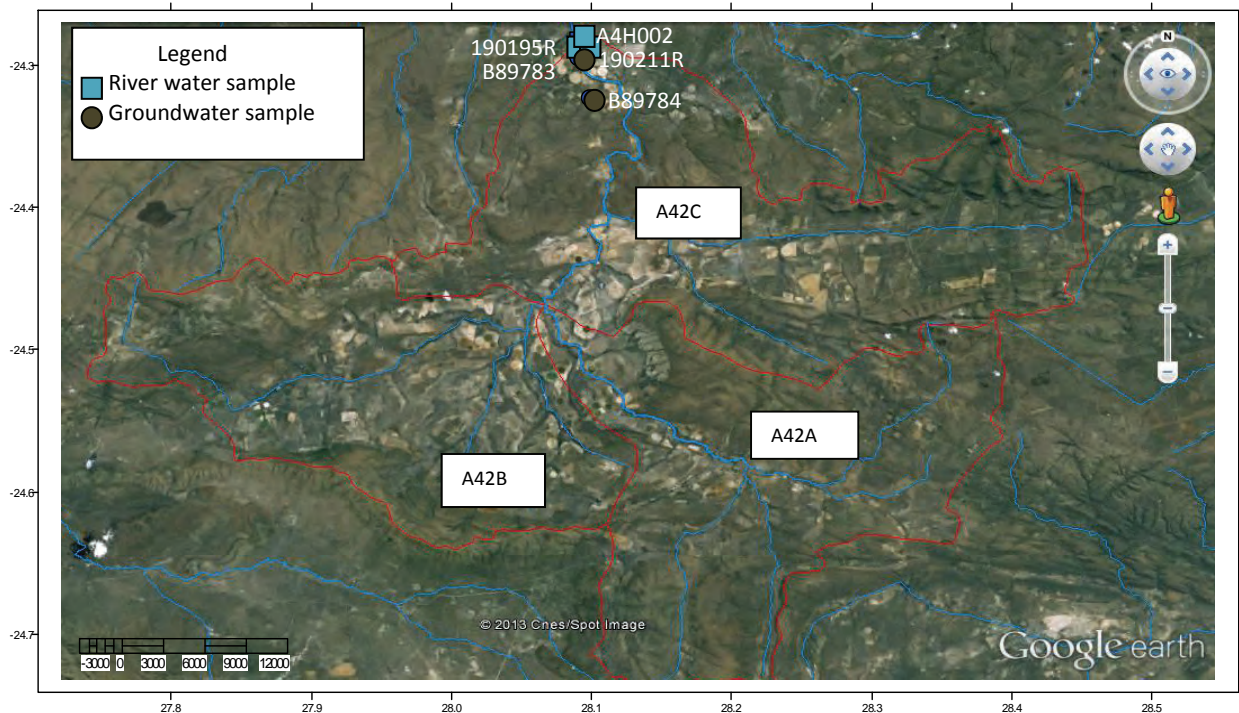


Figure 4-2-7 The river water and groundwater water quality data sample points for 2005 in the A42 area.

4.2.5. MCM Conceptualisation and application

The last segment of the Moloko River within the quaternary catchment A42C at the flowstation A4H002 is conceptualised as the river cell A4H2 Cell for the MCM application. The various flows into and out of this river system are conceptualised into a box model which forms the basis of the water balance equation for the MCM (Figure 4-2-8). A fictitious cell (o-A4H2 Cell) with exactly the same tracer concentrations as the river cell A4H2 Cell was created because the MCM software requires a minimum of two mixing cells to be defined in each model run. The A4H2 Cell is defined by a set of tracer concentrations from water quality data sampled at the flowstation A4H002. Inflow to the river cell is defined by water quality data also sampled at the flowstation due to data restrictions. Groundwater sources are defined by chemical analysis data for each of the 2 boreholes within the study area. The river flow measurements taken at the flowstation A4H002 are used to define the outflow from the flow system. All the water quality data used to define the various cells and inflows are shown in Appendix D Table D-1.

The representative EC and chloride (Cl) values for each flow represented in the flow box model are shown in Figure 4-2-9. Figure 4-2-9 shows a trend for EC and Cl values to decrease downstream. However, tracer concentrations for F, K, NO₃ and TAL increase downstream. The limited surface water quality data for this area has led to a poorly defined flow box model which is most likely the cause of the ambiguous chemical concentration trends. Groundwater is found to be of a much poorer quality when compared to surface water.

Abstraction and evaporation loss volumes from the section of the Moloko River under investigation as well as direct rainfall volumes into the system are assumed to be negligible because chemical data used in the model run covers a short time period and this omission ensures a mathematically-simple model run. A MCM model run is performed for each of the different weighting factor scenarios. Scenario 1 assigns the maximum weighting factor ($\omega = 1$) to all tracers, while scenario 2 assigns a lower weighting factor ($\omega = 0.3$) to tracers indicating a high chemical mass balance error (Table 4-2-2).

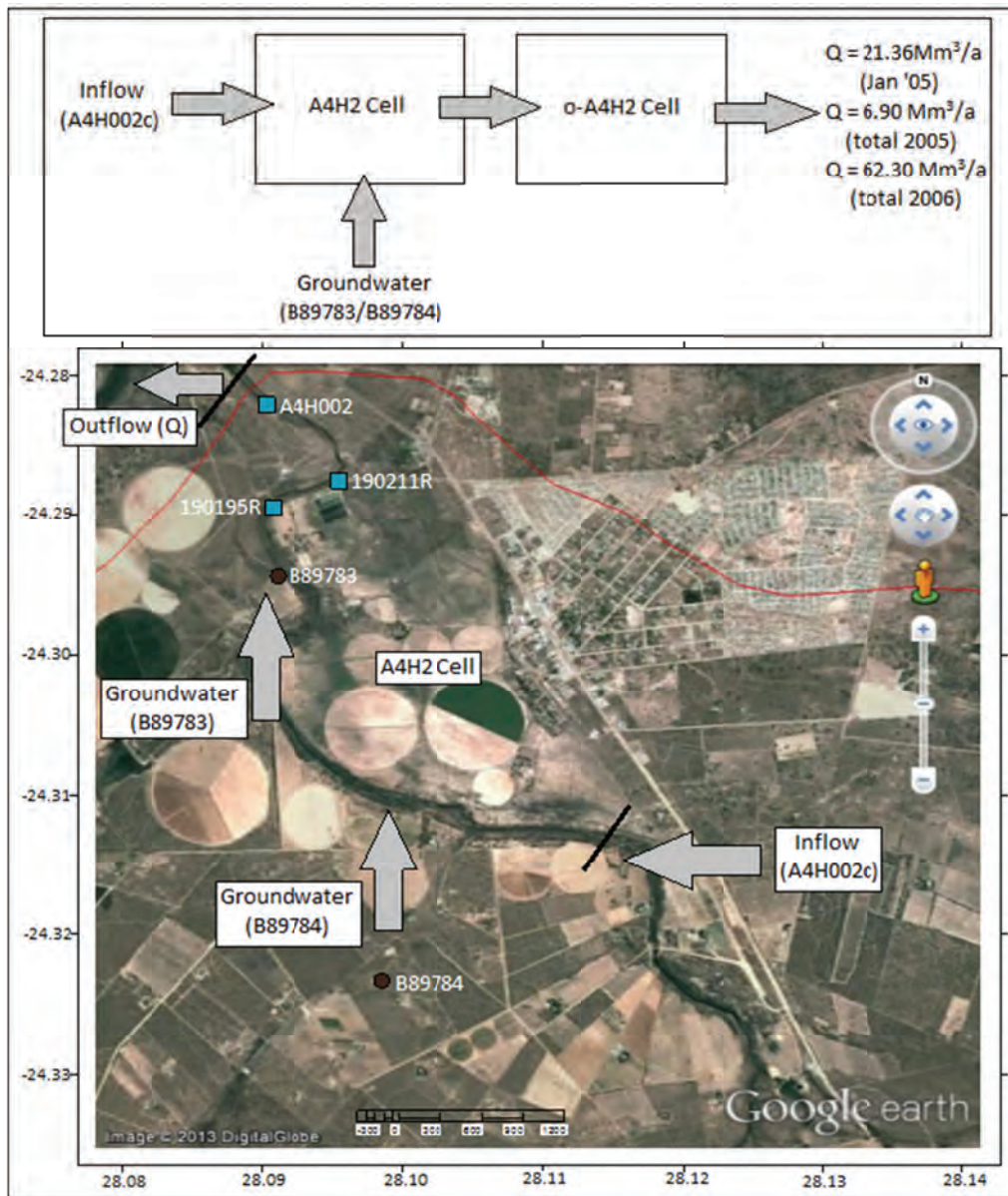


Figure 4-2-8 The box model conceptualisation of the various flows modelled for the A42 area and a Google Earth® image showing the conceptualisation on a real scale, indicating the position of the flowstation and borehole samples.

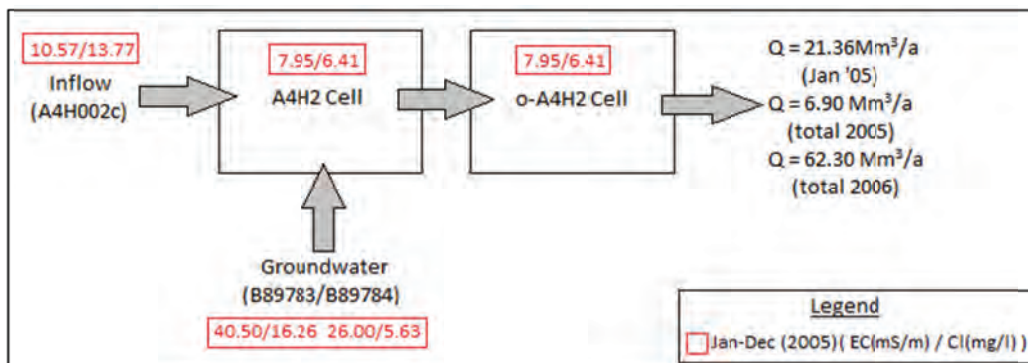


Figure 4-2-9 Representative EC and chloride (Cl) values for the A42 area flow conceptualisation. Concentration trends show a decrease in the EC-value and chloride concentration as moving downstream indicating that groundwater is not contributing to streamflow, but the area is in a high groundwater baseflow zone.

Table 4-2-2 Weighting factor assigned to each tracer for scenarios 1 and 2.

Weighting Factor Scenario	Tracer											
	Ca	Cl	EC	F	K	Mg	Na	NO3	pH	PO4	SO4	TAL
Scenario 1	1	1	1	1	1	1	1	1	1	1	1	1
Scenario 2	1	0.3	1	0.3	0.3	1	1	1	1	1	0.3	1

4.2.6. MCM Results

The MCM run, using scenario 1 weighting factors and defining the outflow from the system with the monthly river flow measured in January 2005 extrapolated to an annual flow volume, estimated the groundwater contribution to baseflow from the groundwater source defined by the borehole sample B89783 at 0.60 Mm³/a, while the contribution from the source defined by the borehole sample B89784 was estimated at 1.31 Mm³/a (Table 4-2-3). The inflow to the A4H2 Cell, defined by the river sample A4H002c, was determined to be 11.69 Mm³/a, while the outflow from the A4H2 Cell to the fictitious o-A4H2 Cell was determined to be 20.72 Mm³/a. The MCM was run for the same setup, but using scenario 2 weighting factors, resulting in a estimation of the groundwater contribution to baseflow from the groundwater source defined by the borehole sample B89783 of 0.58 Mm³/a, while the contribution from the source defined by the borehole sample B89784 was estimated at 0.63 Mm³/a (Table 4-2-3). The inflow to the A4H2 Cell, defined by the river sample A4H002c, was determined to be 13.70 Mm³/a, while the outflow from the A4H2 Cell to the fictitious o-A4H2 Cell was determined to be 21.43 Mm³/a.

The results from the A42 area MCM application, using scenario 2 weighting factors, are graphically represented in Figure 4-2-10. The unknown fluxes determined by the model are expressed as percentages of the assigned total outflow, which is the monthly flow volume measured at the flowstation in January 2005 extrapolated to an annual flow volume. The MCM application determined the inflow from the upstream river source A4H002c to the A4H2 Cell at 63.9% of the total outflow from the system. The groundwater contribution to streamflow within the A4H2 Cell was estimated at 5.6% from the defined groundwater sources. Flow from the A4H2 Cell to the downstream fictitious o-A4H2 Cell was determined at 100.3% of the assigned total flow. The outflow of more than 100% to the fictitious o-A4H2 Cell indicates that that the assigned total outflow volume could have been underestimated.

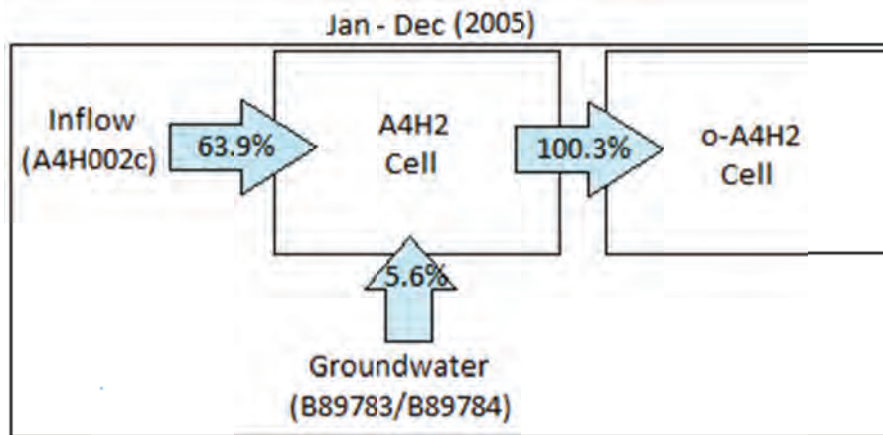


Figure 4-2-10 The unknown fluxes determined by the MCM for the A42 area application, making use of the scenario 2 weighting factors, expressed as a percentage of the total flow volume.

Two additional MCM runs were performed using two different volumes to define the outflow from the modelled system, to demonstrate the variability of the model. The total sum of the monthly river flow volumes measured at the flowstation A4H002 in 2005 and 2006 are used. The total sum of monthly flow volumes for 2005 is 6.9 Mm³/a, while the total sum of monthly flow for 2006 is 62.30 Mm³/a. The MCM run, using scenario 1 weighting factors and defining the outflow from the system with the total sum of monthly flow volumes from 2005, estimated the groundwater baseflow from the groundwater source defined by the borehole sample B89783 at 0.21 Mm³/a, while the contribution from the groundwater source defined by the borehole sample B89784 was estimated at 0.46 Mm³/a (Table 4-2-3). Inflow to the A4H2 Cell, defined by the river sample A4H002c, was determined to be 4.13 Mm³/a, while the outflow from the A4H2 Cell to the fictitious o-A4H2 Cell was determined to be 7.32 Mm³/a. The MCM was run for the same setup, but making use of scenario 2 weighting factors, resulting in the groundwater baseflow from the groundwater source defined by the borehole sample B89783 to be estimated at 0.20 Mm³/a, while the contribution from the groundwater source defined by the borehole sample B89784 was estimated at 0.22 Mm³/a (Table 4-2-3). The inflow to the A4H2 Cell, defined by the river sample A4H002c, was determined to be 4.84 Mm³/a, while the outflow from the A4H2 Cell to the fictitious o-A4H2 Cell was determined to be 7.57 Mm³/a.

The MCM run, using scenario 1 weighting factors and defining the outflow from the system with the total sum of monthly flow volumes in 2006, estimated the groundwater contribution to streamflow from the groundwater source defined by the borehole sample B89783 at 1.70 Mm³/a, while the contribution from the groundwater source defined by the borehole sample B89784 was estimated at 3.72 Mm³/a (Table 4-2-3). The inflow to the

A4H2 Cell, defined by the river sample A4H002c, was determined to be 33.09 Mm³/a, while the outflow from the A4H2 Cell to the fictitious o-A4H2 Cell was determined to be 58.65 Mm³/a. The MCM was run for the same setup, using scenario 2 weighting factors, resulting in a estimation of the groundwater contribution to streamflow from the groundwater source defined by the borehole sample B89783 of 1.63 Mm³/a, while the contribution from the source defined by the borehole sample B89784 was estimated at 1.78 Mm³/a (Table 4-2-3). The inflow to the A4H2 Cell, defined by the river sample A4H002c, was determined to be 38.79 Mm³/a, while the outflow from the A4H2 Cell to the fictitious o-A4H2 Cell was determined to be 60.66 Mm³/a.

Table 4-2-3 Summary of MCM results for the A42 area model run for each scenario using three different outflows.

Name of inflow	Rate of inflow (Mm ³ /a)					
	Outflow (Jan '05)		Outflow (Sum 2005)		Outflow (Sum 2006)	
	Scenario 1	Scenario 2	Scenario 1	Scenario 2	Scenario 1	Scenario 2
A4H002c	11.69	13.70	4.13	4.84	33.09	38.79
B89783	0.60	0.58	0.21	0.20	1.70	1.63
B89784	1.31	0.63	0.46	0.22	3.72	1.78

4.2.7. MCM Discussion and comparison

The water and chemical mass balance errors for each model run and the associated weighting factor scenario are shown in Table 4-2-4. From Table 4-2-4 it can be seen that the use of different outflow volumes has no effect on the associated water and average chemical mass balance errors. A detailed, tracer-specific table of all the chemical mass balance percentage errors can be seen in Appendix D Table D-2. The change in outflow volumes does affect the estimated groundwater inflow, where a larger assigned outflow volume results in a higher groundwater baseflow estimate (Table 4-2-3). The water balance error is 39.2% using scenario 1 and 33.3% using scenario 2, for all the model runs. The chemical mass balance error is 12.7% using scenario 1 and 9.9% using scenario 2, for all the model runs (Table 4-2-4). The groundwater baseflow estimates from the MCM run using the weighting factors from scenario 2 are used for comparison because this scenario results in a decrease in both the water and chemical mass balance errors.

Table 4-2-4 Water and chemical mass balance percentage errors associated with each model run and scenario.

Balance	Outflow 1 (Jan '05)		Outflow 2 (Sum 2005)		Outflow 3 (Sum 2006)	
	Scenario 1	Scenario 2	Scenario 1	Scenario 2	Scenario 1	Scenario 2
Water balance error (%)	39.2%	33.3%	39.2%	33.3%	39.2%	33.3%
Chemical mass balance error (%)	12.7%	9.9%	12.7%	9.9%	12.7%	9.9%

The total groundwater baseflow was estimated at 1.21 Mm³/a, 0.42 Mm³/a and 3.41 Mm³/a for the model runs using outflow 1, 2 and 3, respectively. Groundwater baseflow estimates from the Pitman, Hughes and Sami models as well as an estimate from another method incorporating water quality data are shown in Table 4-2-5. The Tracer method estimate is shown for both the use of electrical conductivity (EC) as the sole tracer, as well as an average value from applying the Tracer method with a number of tracers. The Sami and Hughes models were calibrated for quaternary catchments A42A and A42B, while an estimated A42C calibration volume is inferred from the ratio between calibrated and uncalibrated volumes. The MCM and Tracer method were only applied within the A42C quaternary catchment due to data restrictions. The uncalibrated Pitman, Sami and Hughes models groundwater baseflow estimates for the quaternary catchment A42A are 17.99 Mm³/a, 8.70 Mm³/a and 15.62 Mm³/a, respectively (Table 4-2-5). The calibrated groundwater baseflow estimates for the Sami and Hughes models are 8.99 Mm³/a and 11.90 Mm³/a, respectively. For the quaternary catchment A42B the uncalibrated Pitman, Sami and Hughes models groundwater baseflow estimates are 16.86 Mm³/a, 6.56 Mm³/a and 15.39 Mm³/a, respectively. The calibrated groundwater baseflow estimates for the Sami and Hughes models are 8.50 Mm³/a and 11.72 Mm³/a, respectively. The uncalibrated Pitman, Sami and Hughes model groundwater baseflow estimates for the quaternary catchment A42C are 22.41 Mm³/a, 8.83 Mm³/a and 20.26 Mm³/a, respectively while the inferred calibrated volumes for the Sami and Hughes model are 10.28 Mm³/a and 15.43 Mm³/a, respectively. The Tracer method groundwater baseflow estimate for the quaternary A42C, using only EC is 9.83 Mm³/a, while for an average from a number of tracers is 20.83 Mm³/a. The MCM groundwater baseflow estimate for A42C was 1.21 Mm³/a for outflow 1, 0.42 Mm³/a for outflow 2 and 3.41 Mm³/a for outflow 3, based on data from the 2005 time period.

The trend seen in these results for the Pitman, Sami and Hughes models is for the groundwater baseflow estimated volume to decrease from the Pitman model to the Hughes model to the Sami model, with the difference between the Pitman model and the Hughes model being less dramatic than between the Hughes and Sami model. The smaller difference between the Hughes and Pitman models is expected because the Hughes model is a modified version of the Pitman model. However, the difference between the Sami and Hughes models is fairly substantial. This difference becomes less after calibration with the Sami model groundwater baseflow estimate increasing after calibration and the Hughes model estimate decreasing after calibration. Considering these changes after calibration, the initial Sami model groundwater baseflow volume is an under-estimation of the groundwater baseflow, while the initial Hughes model volume is an over-estimation. Figure 4-2-11 is a graphical representation of the estimates for each method, illustrating the discussed trends. The Tracer method groundwater baseflow estimates seems to be in agreement with the general amount of groundwater contributing to the river, with the EC-only estimate similar to that of the Sami model and the multiple-tracer estimate similar to that of the Hughes model initial estimate.

Table 4-2-5 The groundwater baseflow estimates from the Pitman model, Sami model (calibrated and uncalibrated), Hughes model (calibrated and uncalibrated), Tracer method and MCM for the quaternary catchments A42A-A42C.

Quat	Pitman (Mm ³ /a)	Sami (Mm ³ /a)	Sami Calibrated (Mm ³ /a)	Hughes (Mm ³ /a)	Hughes Calibrated (Mm ³)	MCM (Mm ³ /a)			Tracer Method (Mm ³ /a)	
						Outflow 1	Outflow 2	Outflow 3	EC	Average
A42A	17.99	8.70	8.99	15.62	11.90	/	/	/	/	/
A42B	16.86	6.56	8.50	15.39	11.72	/	/	/	/	/
A42C	22.41	8.83	10.28 (E)	20.26	15.43 (E)	1.21	0.42	3.41	9.83	20.83

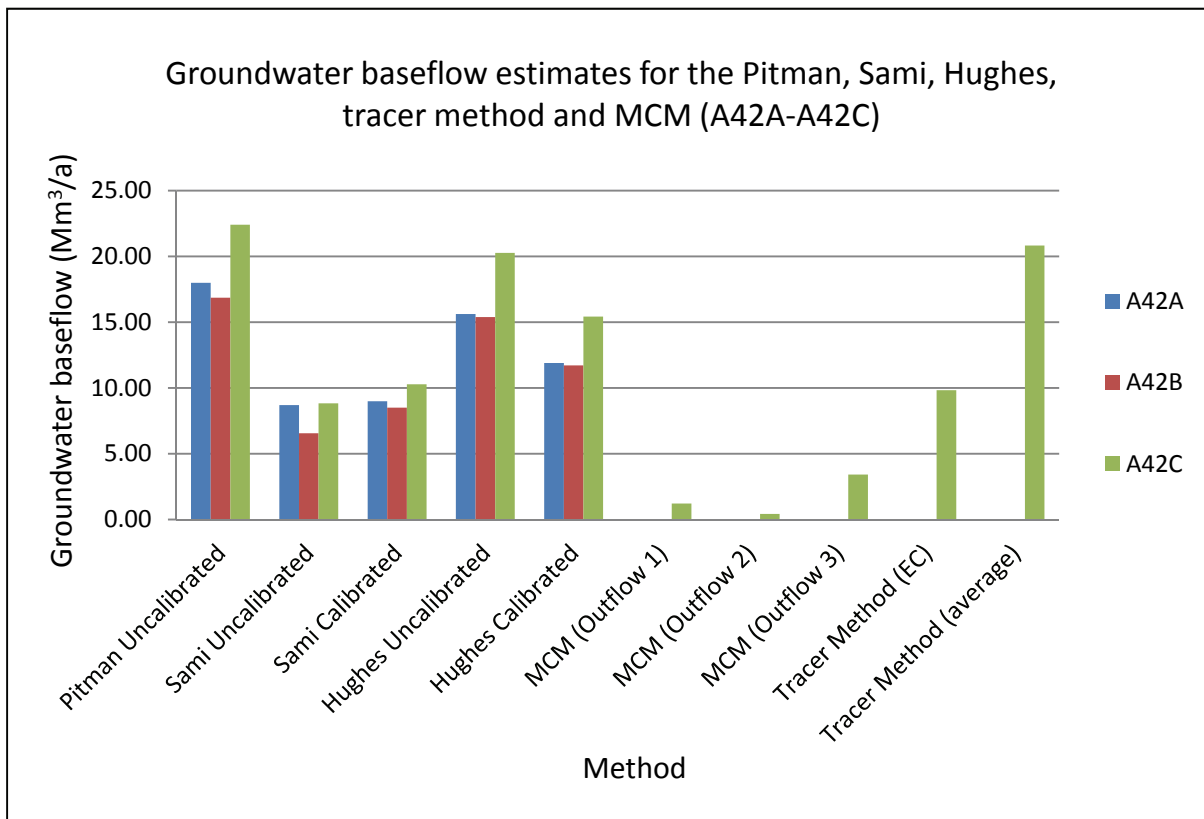


Figure 4-2-11 The groundwater baseflow estimates from the Pitman model, Sami model (calibrated and uncalibrated), Hughes model (calibrated and uncalibrated), Tracer method and MCM for the quaternary catchments A42A-A42C.

The groundwater baseflow estimates from the MCM are considerably lower than all the other methods. One explanation for the lower groundwater contribution estimated by the MCM is that the data used in the model was collected during a particularly dry year with the rainfall being less than that of the corresponding month in 2006. The measured river flow volumes measured at A4H002 in 2005 show a drastic difference when compared with the corresponding monthly flow volumes in 2006 (Figure 4-2-12). The difference in rainfall is not as drastic as the difference in measured flow volumes, which might indicate that the low river flow is not related to the amount of rainfall, but perhaps the amount of water use. If groundwater resources were over utilised along with the surface water resources this would lead to a drastic decrease in the amount of groundwater reaching the river. The MCM run using the outflow volume measured in 2006 still results in a low groundwater baseflow estimate which could mean that the chemical data is restricting, or that the low flow in 2005 is not the reason for the low estimate. Another explanation is the rough definition of the inflow sources and the river cell due to a general lack of water quality data. The MCM results also have a large percentage error associated with the estimates which could

indicate that an important source has been omitted or that the river cell was incorrectly defined.

Considering the results from this study area, the Tracer method seems to be better suited for quantifying the groundwater baseflow volume than the MCM. The Tracer method estimate from the use of EC alone is in agreement with the calibrated Sami model estimate and the estimate from multiple tracer use is in agreement with the initial Hughes model estimate. In this instance, it could be deduced that the average Tracer method estimate from a number of tracers is an over-estimation of the groundwater baseflow, as is seen with the initial Hughes model estimate. The Tracer method estimate from the use of EC alone gives the most reasonable estimate of groundwater baseflow, between the two water quality methods.

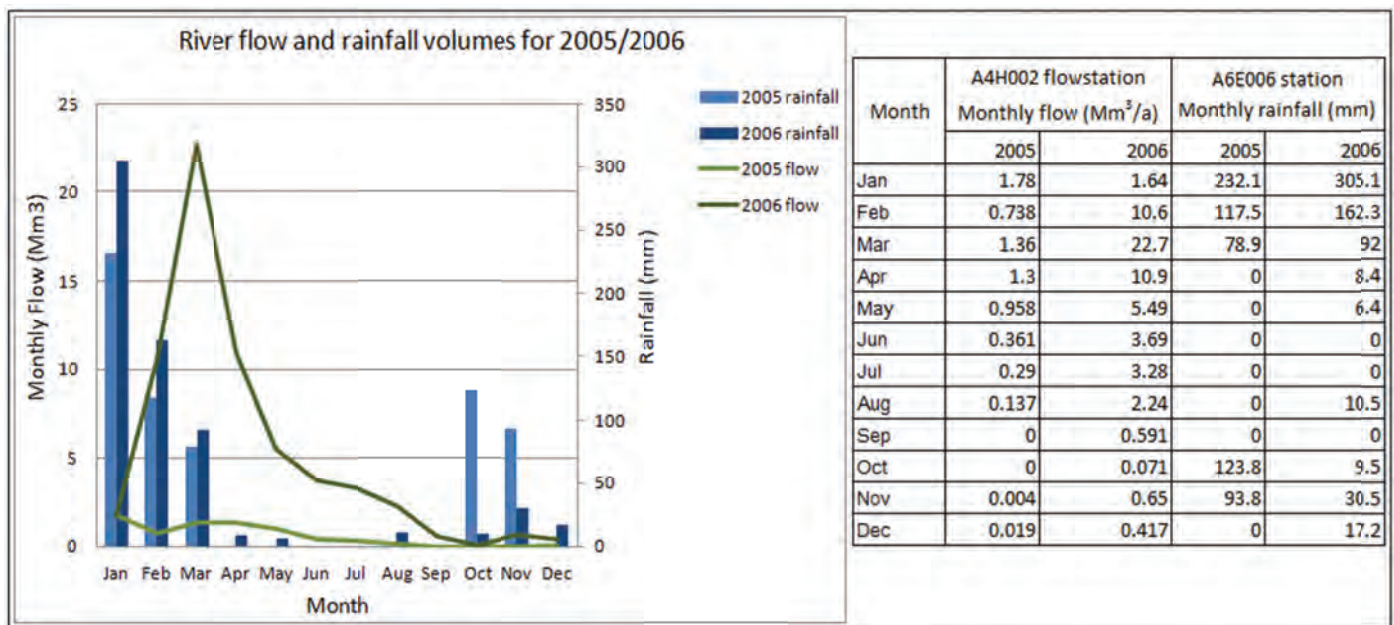


Figure 4-2-12 Rainfall at the meteorological station A6E006 and river flow volumes at the flowstation A4H002 for 2005 and 2006.

4.3. Pilot Study 3: Quaternary Catchment D73F

4.3.1. Overview

The quaternary catchment D73F is situated within the Northern Cape Province of South Africa (Figure 4-3-1). This area falls within the Lower Orange Water Management Area and includes a section of the Orange River between the towns of Upington and Kakamas. The Orange River is a perennial river and the longest in South Africa. It is fed by a number of non-perennial tributaries within the study area, namely the Neuspruit, Kameel, Brak, Vaalputs, Kareeboom, Olienhout and Donkerhoek Rivers. The average annual rainfall, based on data from the Department of Water Affairs meteorological station D7H003, is 190 mm, and the total measured rainfall for the year of investigation (1989) was 136 mm. The average annual evaporation, based on S-pan measurements from the meteorological station, is 2527 mm. The average monthly temperature for the area ranges from 4°C to 35°C. The area is classified as a semi-desert and characterised by a flat landscape. The topography ranges from 700 mamsl to 1000 mamsl, with no drastic changes in topography. The predominant vegetation is Nama Karoo which consists of low shrubs and grasses, with the northern point of the quaternary catchment D73F dominated by Southern Kalahari vegetation. The WARMS database indicates that most of the surface water abstraction is for irrigation schemes and amounts to approximately 226 Mm³/a.

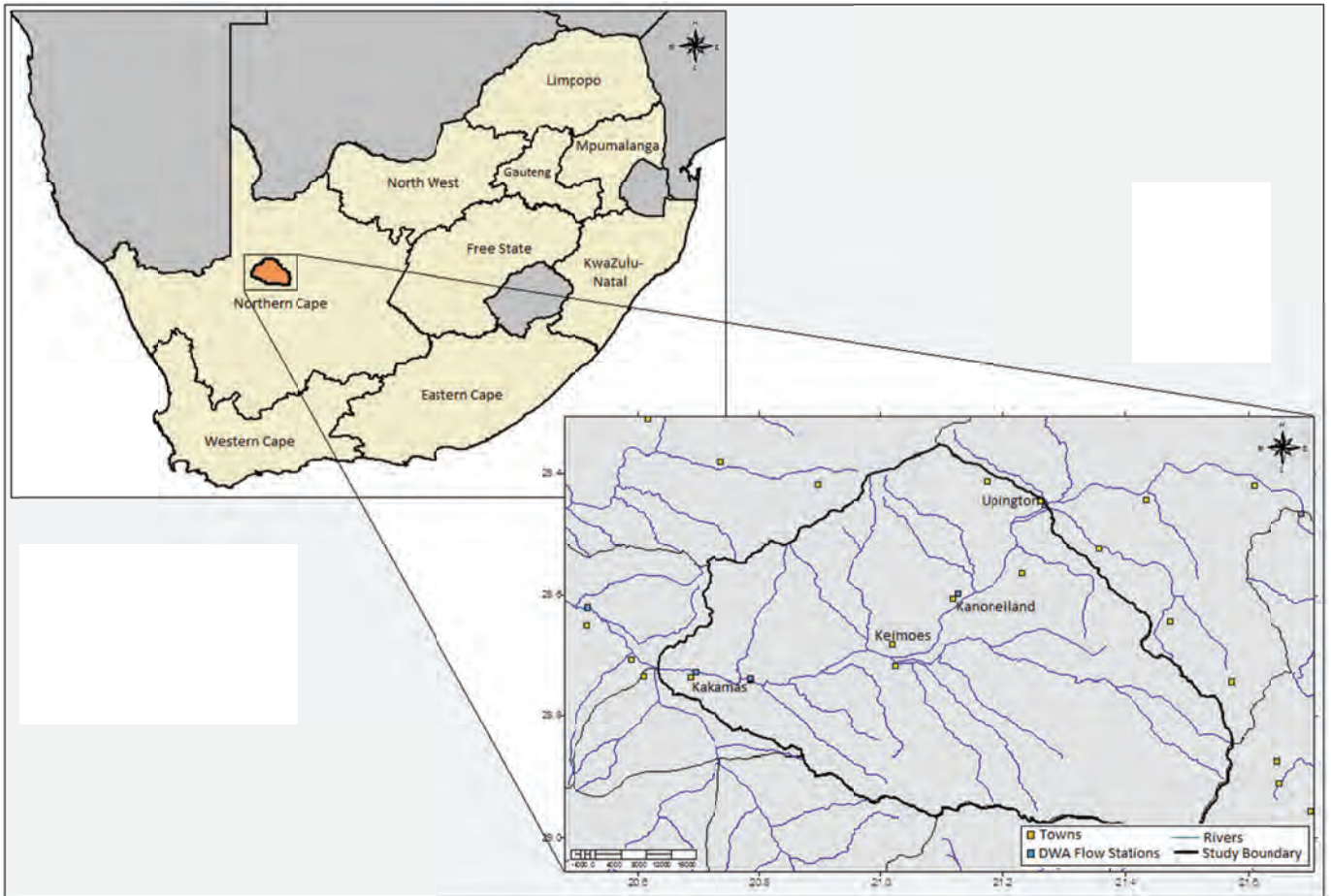


Figure 4-3-1 The position of the quaternary catchment D73F within South Africa.

4.3.2. Geology

The Namaqua-Natal Metamorphic Province covers most of the Northern Cape and the quaternary catchment under investigation falls within the Namaqua sector of this tectono-stratigraphic province. The Namaqua-Natal Province comprises a number of tectono-stratigraphic terranes bounded by shear zones. The five main subdivisions in the Namaqua sector are the Richtersveld Sub-province, the Bushmanland Terrane, Kakamas Terrane, Areachap Terrane and the Kaaien Terrane (Johnson, Anhaeusser and Thomas, 2006). Formations from the Kakamas, Areachap and Bushmanland Terranes are found in the outcrop geology of the quaternary catchment D73F (Figure 4-3-2). The outcrop geology of the area indicates the structural, metamorphic and intrusive complexity of the area which leads to uncertainty in regards to stratigraphic relations. The Bushmanland Terrane is the largest of the crustal blocks in the Namaqua sector, and represented in the quaternary catchment by the supracrustal Hoogoor Suite and intrusive rocks of the Little Namaqualand Suite and the Eendoorn Granite. The Kakamas Terrane lies to the East of the Bushmanland

Terrane and is represented within the study area by the supracrustal Korannaland Group, Witwater Gneiss and Toeslaan Formation and intrusive rocks of the Eendoorn Suite, Keimos Suite and Friersdale Charnockite. The Bethesda and Jannelsepan Formations of the Areachap Terrane are found in the outcrop geology of the area as well as intruded Keimoese Suite granites (Johnson, Anhaeusser and Thomas, 2006).

The Hoogoor Suite of the Bushmanland Terrane, often referred to as the pink gneiss, is a large concordant to semi-concordant body of red-weathering quartzofeldspathic gneisses. The intrusive Little Namaqua Suite consists of igneous lithologies ranging from granite to rocks with sheet-like bodies of quartz-microcline-biotite augen gneiss. The Eendoorn Granite resembles the augen gneiss in the Little Namaqua Suite, but geochronological restraints are currently insufficient to confirm a direct relation. The Korannaland Group of the Kakamas Terrane comprises several lithologies, which can be generally described as high-grade supracrustal rocks consists of mostly arenite and calc-arenite lithologies. Witwater Gneiss rocks are also found within this Terrane. The Toeslaan Formation is included into the Korannaland Group, but its stratigraphic position is uncertain and might be correlated to the Areachap Terrane. The formation consists of a thick succession of quartz-feldspar-cordierite-spinel-garnet meta-pelitic gneisses. The Kakamas Terrane is intruded by syn- to late-tectonic granitoids, including the Eendoorn Suite. The Keimos Suite, a collection of the syn- to post-tectonic granitoids east of the Neusberg Shear Zone, is also an intrusive rock of the Kakamas Terrane. The Friersdale Charnockite consists of late-tectonic bodies of undeformed lithologies which intrude earlier members of the Keimoese Suite and has been interpreted as the intrusive equivalent of the Jannelsepan Formation of the adjacent Areachap Terrane. The Jannelsepan Formation is found in the northern part of the Areachap Group and consists of mainly migmatitic amphibolite and calc-silicate rocks. The Bethesda Formation occurs along the western side of the Group forming metapelitic schist similar to that of the Sprigg Formation, but devoid of a conglomerate component and partly interlayered with amphibolites. The Keimoese Suite and the charnockite intrusive of the Friersdale Charnockite intrude into the Areachap Terrane (Johnson, Anhaeusser and Thomas, 2006).

The Kubis Sub-group of the Nama Group from the Namibian successions, outcrops in a small section of the study area near Upington. The sub-group consists of alternating layers of

sandstone and shale. Cenozoic age sediments of the Kalahari Group are also evident in the outcrop geology of D73F. The Kalahari Group is the most extensive body of Cenozoic age terrestrial sediments and has been divided into a number of formations, ranging from clayey gravels to calcretes. The Kalahari group deposits tend to coincide with the occurrence of Dwyka Group rocks at their thickest parts, which is seen in the study area (Johnson, Anhaeusser and Thomas, 2006).

The geology within quaternary catchment D73F is highly complicated which makes an accurate stratigraphic column or conceptual cross-section impractical and is thus not attempted.

4.3.3. Hydrogeology

The topography of the quaternary catchment D73F ranges from 700 mamsl to 1000 mamsl. There is a gradual decrease in topography towards the river valley as well as a gradual decrease in topography from east to west along the Orange River (Figure 4-3-3). The water table seems to follow the topography, but the groundwater level is far below the terrestrial surface (Figure 4-3-4). The geology of the area is highly complicated with multiple intrusions, metamorphosed lithologies and extreme deformation. This complex geology tends to limit the existence of continuous aquifers. The amount of groundwater that would have a direct path towards the river through a complicated geology as shown here is negligible. The groundwater level is far below the land surface which also limits the probability of groundwater contributing to the river.

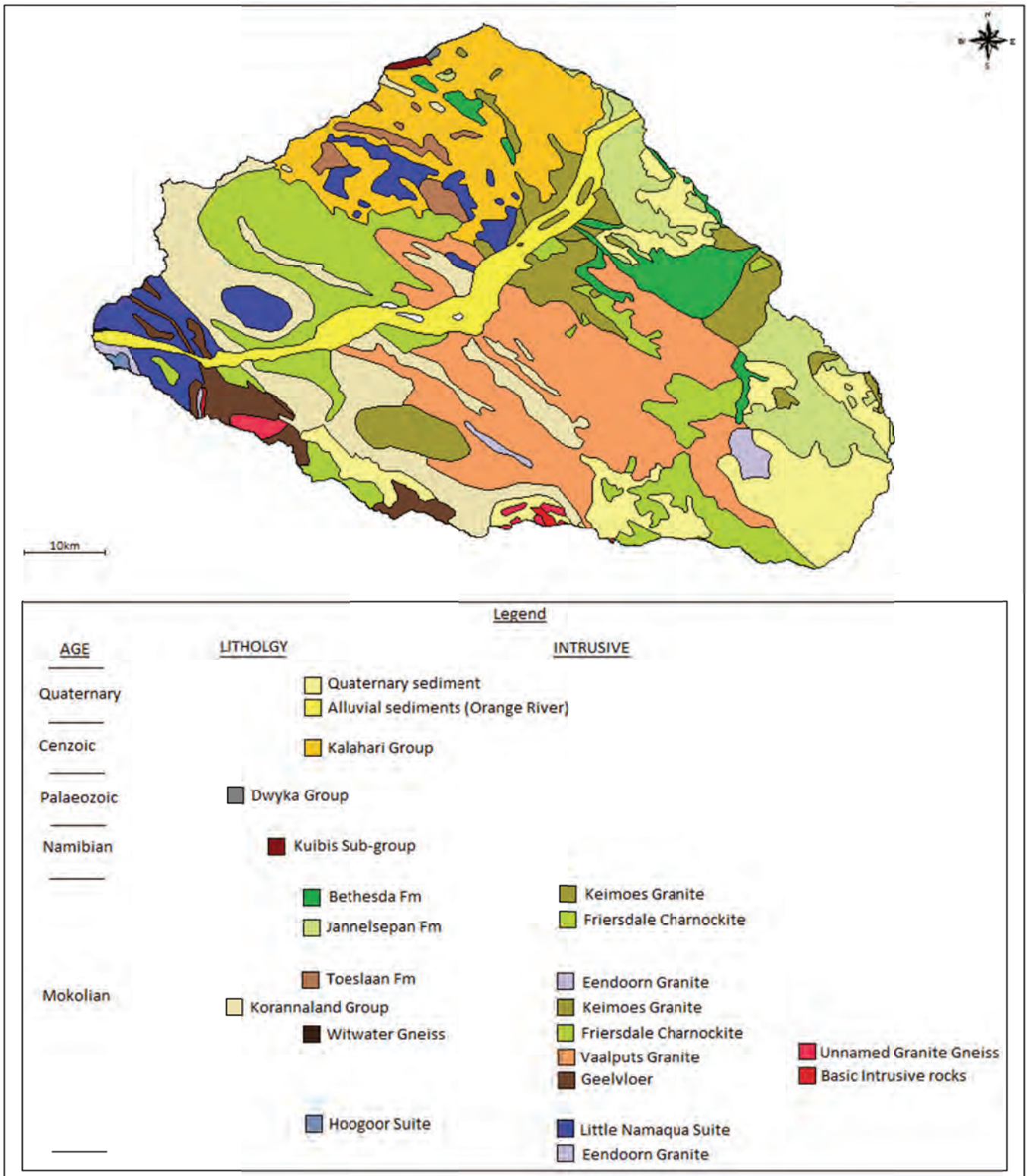


Figure 4-3-2 The outcrop geology of the quaternary catchment D73F (Based on data from GRDM, 2010).

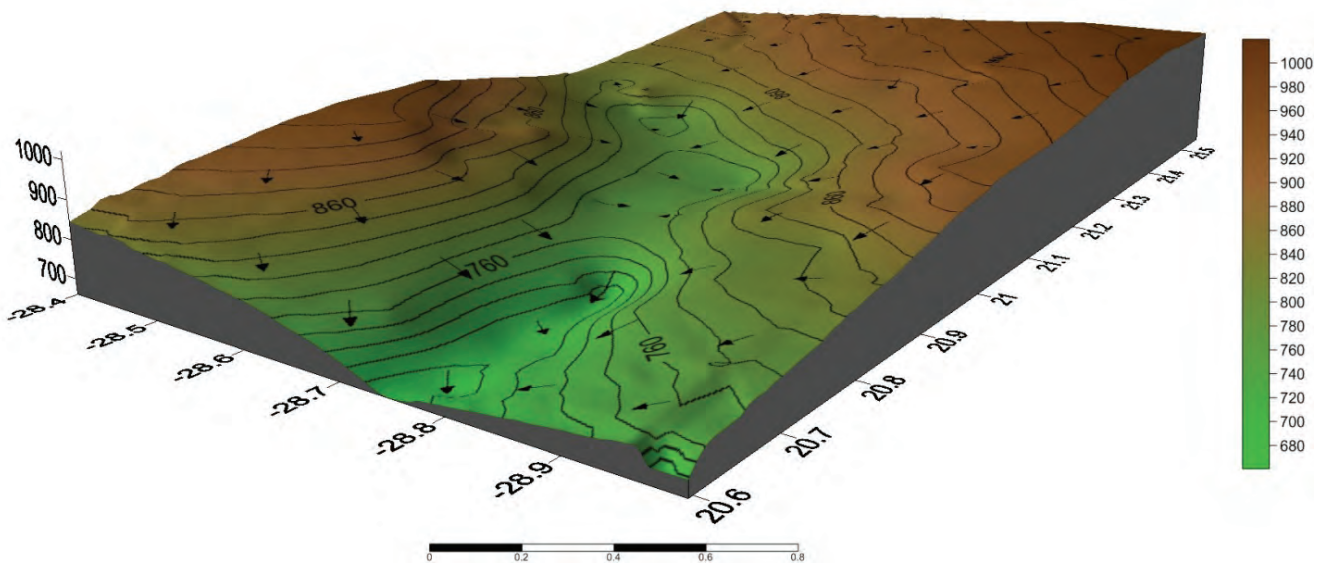


Figure 4-3-3 Generalised topography of the quaternary catchment D73F

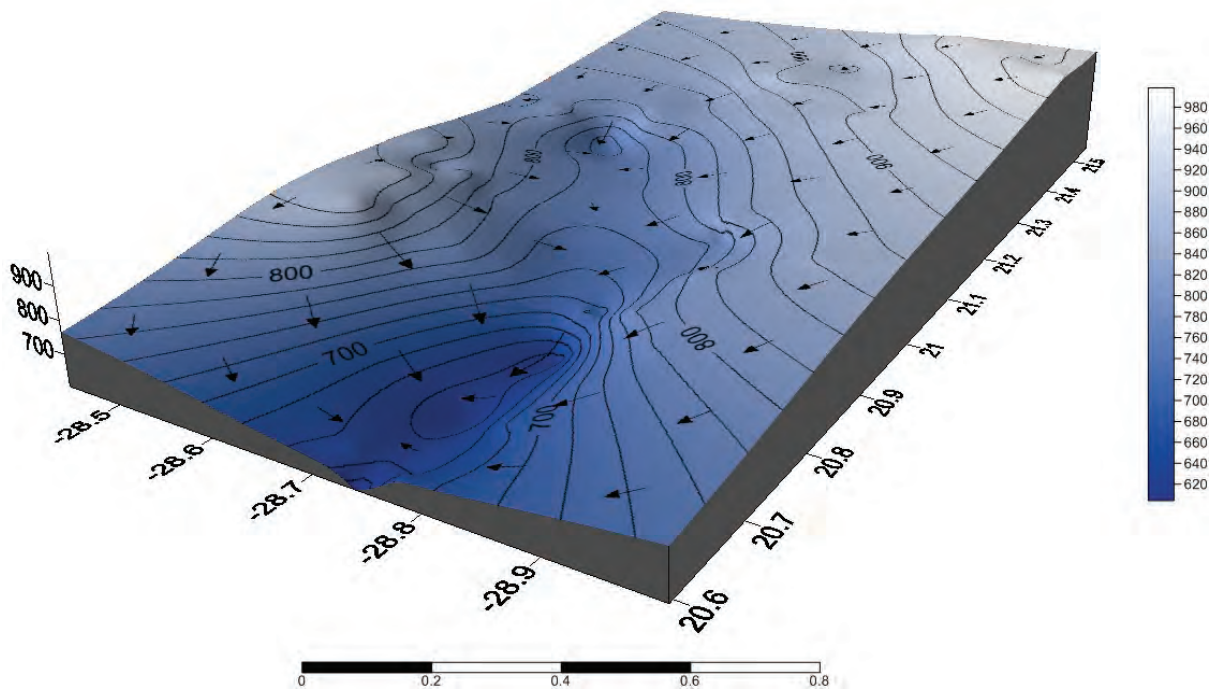


Figure 4-3-4 Generalised groundwater level within the quaternary catchment D73F area.

4.3.4. Mixing Cell Model

The MCM does not directly use physical, geohydrological principles in estimating the groundwater component of baseflow, but estimates the groundwater baseflow by means of a mathematical algorithm. The MCM thus has the potential to allocate a fictitious groundwater baseflow volume to an area that in reality receives no groundwater baseflow. The quaternary catchment D73F falls in the middle of a regionally-defined zero groundwater baseflow zone. The application of the MCM to this catchment will be useful to investigate how the MCM will perform when groundwater sources to a system are defined in the MCM, but it is known that these sources are not contributing to the system.

4.3.5. MCM Conceptualisation and application

The flowstation D7H003 at the downstream boundary of the quaternary catchment D73F would be the best sample point for surface water quality data to define the river mixing cell. Water quality data for this station is only available from 1965 to 1993. Additional data for a MCM model run is thus selected from within this time period as to allow for the use of the chemical data from flowstation D7H003. Additional surface water and groundwater quality data was the most suitable during the year 1989, resulting in the MCM run making use of surface water and groundwater data from this time period.

The section of the Orange River within the quaternary catchment D73F is conceptualised as the river cell D7H003 Cell for the MCM run. The various flows into and out of this river system are conceptualised into a box model which forms the basis of the water balance equation in the MCM (Figure 4-3-5). The MCM requires a minimum of two cells to be defined, thus a fictitious cell (o-D7H003 Cell) is created with exactly the same tracer concentrations as the river cell D7H003 Cell. The D7H003 Cell is defined by a set of tracer concentrations from water quality data sampled at the flowstation D7H003. Groundwater sources are defined by the chemical analysis data from each of the 14 sampled boreholes within the study area over the time period of investigation. The river flow measurements taken at the flowstation D7H003 are used to define the outflow from the flow system. A monthly flow volume is used to define the outflow from the system because the yearly volume of water passing point this point is extremely large. The monthly flow volumes are

extrapolated to yearly flow volumes after computation. All the water quality data used to define the various cells and inflows are shown in Appendix E Table E-1.

The representative EC and chloride (Cl) values for each flow represented in the flow box model are shown in Figure 4-3-6. Figure 4-3-6 shows a trend for the EC value to increase downstream. However, the Cl concentration decreases downstream. The EC trend could indicate that groundwater or some poorer quality water is reaching the stream, while the Cl concentration trend could indicate that no groundwater is contributing to the stream or some lower Cl concentration source of water is reaching the stream. Groundwater was found to be of a poorer quality than surface water, but large ranges of tracer concentrations are found between the various boreholes.

Abstraction and evaporation loss volumes from the section of the Moloko River under investigation as well as direct rainfall volumes are assumed to be negligible for the initial MCM application, but an abstraction volume is assigned to the river cell in an additional model run. Both model runs were applied using different weighting factor scenarios. The first scenario (scenario 1) assigns the maximum weighting factor to all tracers ($\omega = 1$), while scenario 2 assigns a lower weighting factor ($\omega = 0.3$) to tracers showing a high chemical mass balance error (Table 4-3-1).

Table 4-3-1 The weighting factor assigned to each tracer for scenarios 1 and 2.

Weighting Factor Scenario	Tracer														
	Ca	Cl	DMS	EC	F	K	Mg	Na	NH4	NO3_NO2	pH	PO4	Si	SO4	TAL
Scenario 1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Scenario 2	1	1	1	1	1	1	1	0.3	0.3	0.3	1	1	1	1	0.3

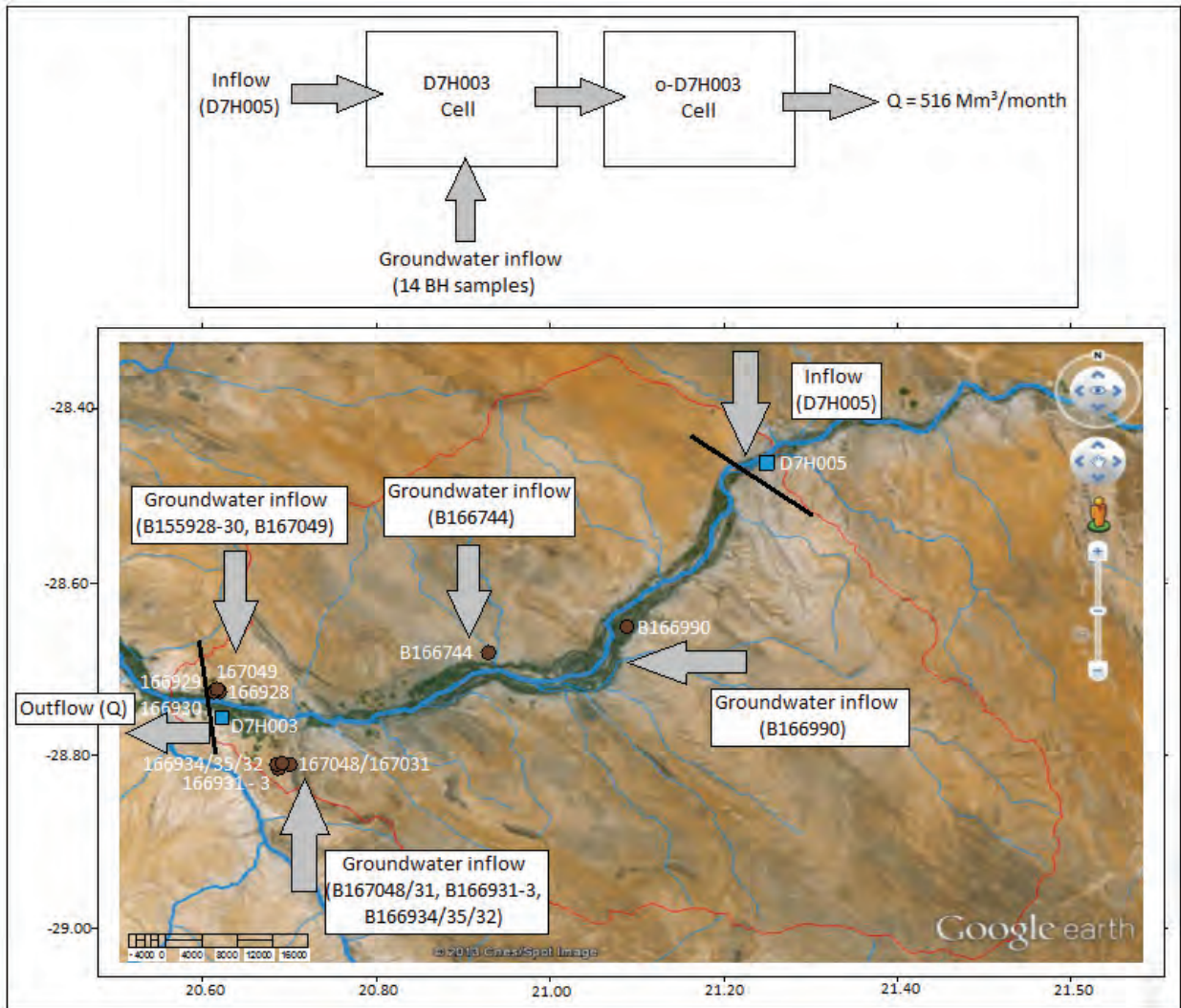


Figure 4-3-5 The box model conceptualisation of the various flows modelled for the quaternary catchment D73F area and a Google Earth® image showing the conceptualisation on a real scale indicating the position of the flowstations (D7H003 and D7H005) and the 14 borehole sample points.

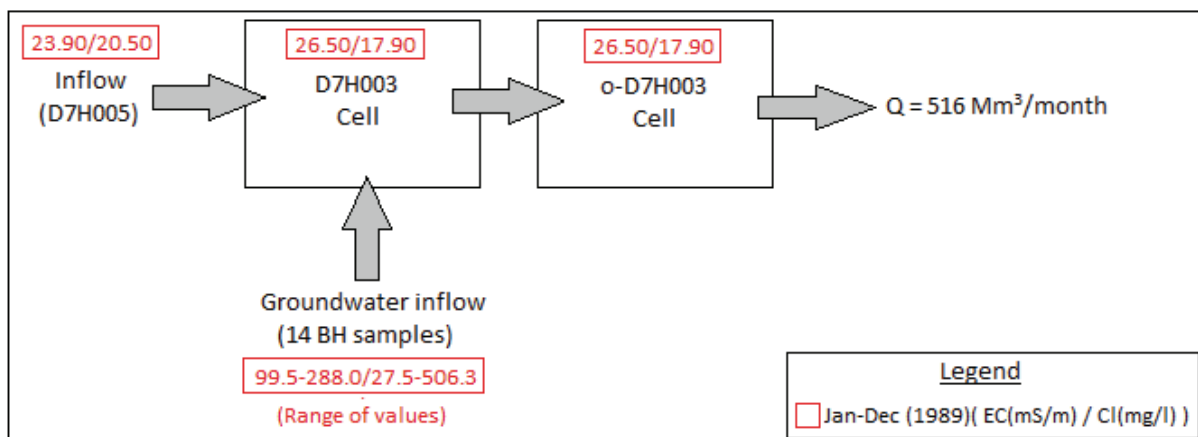


Figure 4-3-6 Representative EC and chloride (Cl) values for the quaternary catchment D73F flow conceptualisation. Concentration trends show an increase in the EC-values, but a decrease in the Cl concentrations moving downstream.

4.3.6. MCM Results

The MCM run, using scenario 1 weighting factors and assigning no abstraction volume to the river cell, estimated the groundwater contribution to baseflow from the groundwater sources defined by the borehole samples B166990 and B166744, located in the middle of the river cell, at 0.00 Mm³/month (Table 4-3-2). The groundwater contribution to baseflow from the groundwater sources located on the northern side of the river and defined by the borehole samples B166928, B166929, B166930 and B167049 were all estimated at 0.00 Mm³/month. The groundwater contribution to baseflow from the groundwater sources located on the southern side of the river and defined by the borehole samples B166931, B166932, B166933, B166934, B166935, B167031 and B167048 were all estimated at 0.00 Mm³/month, while the groundwater source defined by the borehole sample B167032 was estimated at 0.62 Mm³/month, which extrapolated to a yearly flow volume becomes 7.44 Mm³/a. The inflow to the D7H003 Cell, defined with water quality data from the flowstation D7H005, was determined to be 430.35 Mm³/month which extrapolated to a yearly flow volume becomes 5164.20 Mm³/a, while the outflow from the D7H003 Cell to the fictitious o-D7H003 Cell was determined to be 487.32 Mm³/month which extrapolated to a yearly flow volume becomes 5847.84 Mm³/a. The MCM was run for the same setup, using scenario 2 weighting factors, resulting in an estimation of the groundwater contribution to baseflow, from all the defined groundwater sources, at 0.00 Mm³/a (Table 4-3-2). The inflow to the D7H003 cell, defined with water quality data from flowstation D7H005, was determined to be 479.73 Mm³/month or 5756.76 Mm³/a, while the outflow from the D7H003 Cell to the fictitious o-D7H003 Cell was determined to be 506.22 Mm³/month or 6074.64 Mm³/a.

The results from the quaternary catchment D73F MCM application, utilizing scenario 2 weighting factors and assigning no abstraction volume to the river cell, are graphically represented in Figure 4-3-7. The unknown fluxes determined by the model are expressed as percentages of the assigned total outflow. The MCM application determined the inflow from the upstream river source, defined by a water sample taken at the flowstation D7H005, to the D7H003 Cell at 94.7% of the total outflow from the system. The groundwater contribution to streamflow within the D7H003 Cell was estimated at 0.0% from the defined groundwater sources. Flow from the D7H003 Cell to the downstream fictitious o-D7H003 Cell was determined at 98.1% of the total flow.

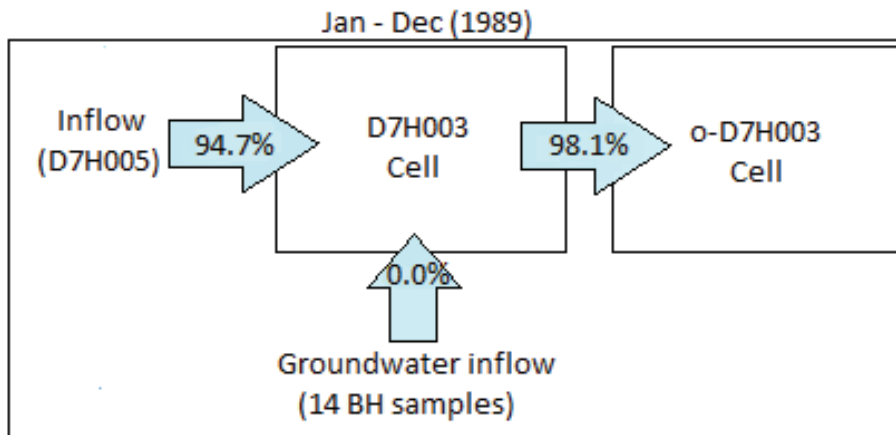


Figure 4-3-7 The unknown fluxes determined by the quaternary catchment D73F MCM application, making use of the scenario 2 weighting factors and assigning no abstraction from the river cell, expressed as a percentage of the total flow volume.

The amount of water abstracted from this section of the Orange River is significant at approximately $226 \text{ Mm}^3/\text{a}$, which lead to a MCM run which assigns an abstraction volume to the D7H003 Cell of $18.8 \text{ Mm}^3/\text{month}$, in order to investigate the significance of omitting this volume initially. The MCM run, using scenario 1 weighting factors and assigning an abstraction volume to the river cell, estimated the groundwater contribution to baseflow from the groundwater sources defined by the borehole samples B166990 and B166744, located in the middle of the river cell, at $0.00 \text{ Mm}^3/\text{month}$ (Table 4-3-2). The groundwater contribution to baseflow from the groundwater sources located on the northern side of the river and defined by the borehole samples B166928, B166929, B166930 and B167049 were estimated at $0.00 \text{ Mm}^3/\text{month}$. The groundwater contribution to baseflow from the groundwater sources located on the southern side of the river and defined by the borehole samples B166931, B166932, B166933, B166934, B166935, B167031 and B167048 were all estimated at $0.00 \text{ Mm}^3/\text{month}$, while the groundwater source defined by the borehole sample B167032 was estimated at $0.64 \text{ Mm}^3/\text{month}$ which extrapolated to a yearly flow volume becomes $7.68 \text{ Mm}^3/\text{a}$. The inflow to the D7H003 Cell, defined with water quality data from flowstation D7H005, was determined to be $446.00 \text{ Mm}^3/\text{month}$ which extrapolated to a yearly flow volume becomes $5352.00 \text{ Mm}^3/\text{a}$, while the outflow from the D7H003 Cell to the fictitious o-D7H003 Cell was determined to be $486.24 \text{ Mm}^3/\text{month}$ which extrapolated to a yearly flow volume becomes $5834.88 \text{ Mm}^3/\text{a}$. The MCM was run for the same setup, using scenario 2 weighting factors, resulting in an estimation of the

groundwater contribution to baseflow, from all the defined groundwater sources, at 0.00 Mm³/a (Table 4-3-2). The inflow to the D7H003 Cell, defined with water quality data from D7H005 flowstation, was determined to be 495.41 Mm³/month or 5944.92 Mm³/a, while the outflow from the D7H003 Cell to the fictitious o-D7H003 Cell was determined to be 504.59 Mm³/month or 6055.08 Mm³/a.

Table 4-3-2 Summary of MCM results for the quaternary D73F area model run for each scenario

Name of inflow	Rate of inflow (Mm ³ /month)			
	No abstraction volume		Abstraction volume	
	Scenario 1	Scenario 2	Scenario 1	Scenario 2
InflowD7H5	430.35	479.73	446.00	495.41
B166744	0.00	0.00	0.00	0.00
B167032	0.62	0.00	0.64	0.00
B167031	0.00	0.00	0.00	0.00
B166928	0.00	0.00	0.00	0.00
B166929	0.00	0.00	0.00	0.00
B166930	0.00	0.00	0.00	0.00
B166931	0.00	0.00	0.00	0.00
B166932	0.00	0.00	0.00	0.00
B166933	0.00	0.00	0.00	0.00
B166934	0.00	0.00	0.00	0.00
B166935	0.00	0.00	0.00	0.00
B166990	0.00	0.00	0.00	0.00
B167048	0.00	0.00	0.00	0.00
B167049	0.00	0.00	0.00	0.00

4.3.7. MCM Discussion and comparison

The water and chemical mass balance errors for each model run and weighting factor scenario are shown in Table 4-3-3. Once again it can be seen that changing the flow volume (abstraction volume from the river cell) has no substantial effect on the water and average chemical mass balance errors. A detailed, tracer-specific table of all the chemical mass balance errors can be seen in Appendix E Table E-2. The assignment of an abstraction volume from the river cell results in an increase of inflow estimates, for both the upstream river inflow and the groundwater baseflow estimated for borehole source B167032, in the MCM run using scenario 1 (Table 4-3-2). A large decrease in both the water and average chemical mass balance errors are seen when changing from scenario 1 to scenario 2 (Table 4-3-3). The groundwater contribution to the river from the one groundwater source defined

by the borehole sample B167032 goes from 7.44 Mm³/a to 0.00 Mm³/a, when changing from scenario 1 to scenario 2, in both model runs. From these results, it can be seen that the unknown inflows estimated by the MCM are more sensitive to the changes in tracer concentrations and weighting factors, than to changes in assigned flow volumes. Considering the fact that no considerable amount of groundwater baseflow could be contributing to the river in this particular area, the use of scenario 2 weighting factors gives the more accurate estimation of the groundwater baseflow.

Table 4-3-3 Water and chemical mass balance percentage errors associated with each model run and scenario for quaternary catchment D73F MCM run

Balance	Percentage Error (%)			
	No abstraction volume		Abstraction volume	
	Scenario 1	Scenario 2	Scenario 1	Scenario 2
Water balance	16.64%	7.21%	16.64%	7.54%
Chemical mass balance	11.14%	2.27%	11.14%	2.63%

The tracers assigned a lower weighting factor in scenario 2, are NO₃_NO₂ (Nitrate/Nitrite), NH₄ (Ammonia), TAL (Total Alkalinity) and Na (Sodium). Extensive agricultural activities take place alongside the Orange River because it is a plentiful source of water for irrigation. The majority of the registered water use within the study area is for irrigation purposes. Nitrogen based pollutants such as nitrate and ammonia are commonly associated with agricultural practices and the fact that these tracers were causing the MCM to initially estimate a positive groundwater inflow to the river might suggest that nitrogen based pollutants are reaching the Orange River by means of runoff from farm areas. No surface water pollution sources to the river cell were assigned in the MCM run due to lack of data, but this omission might have been the reason for the model assigning a groundwater baseflow value when in fact it should have been an additional pollution source from runoff. The particular borehole sample that this supposed inaccurate inflow was assigned to shows no substantial difference in NO₃_NO₂ and NH₄ tracer concentrations when compared to the other borehole samples, but the sodium concentrations are much higher than in all the other boreholes used to define groundwater inflow (Figure 4-3-8). Figure 4-3-8 is an S.A.R diagram indicating the uniqueness of the borehole sample B167032 in terms of Sodium content and EC (Electrical conductivity). The water quality of this borehole indicates that the

Sodium Adsorption Ratio (SAR) is classed between a medium and high threat, while the rest of the samples fall within classes S1 and S2, low and medium threats. The high sodium concentrations of this particular borehole and the assignment of groundwater baseflow to this source alone suggests that a runoff pollution source of sodium was not accounted for in the MCM application and thus could be the reason for the MCM predicting groundwater baseflow when the area is assumed to have none.

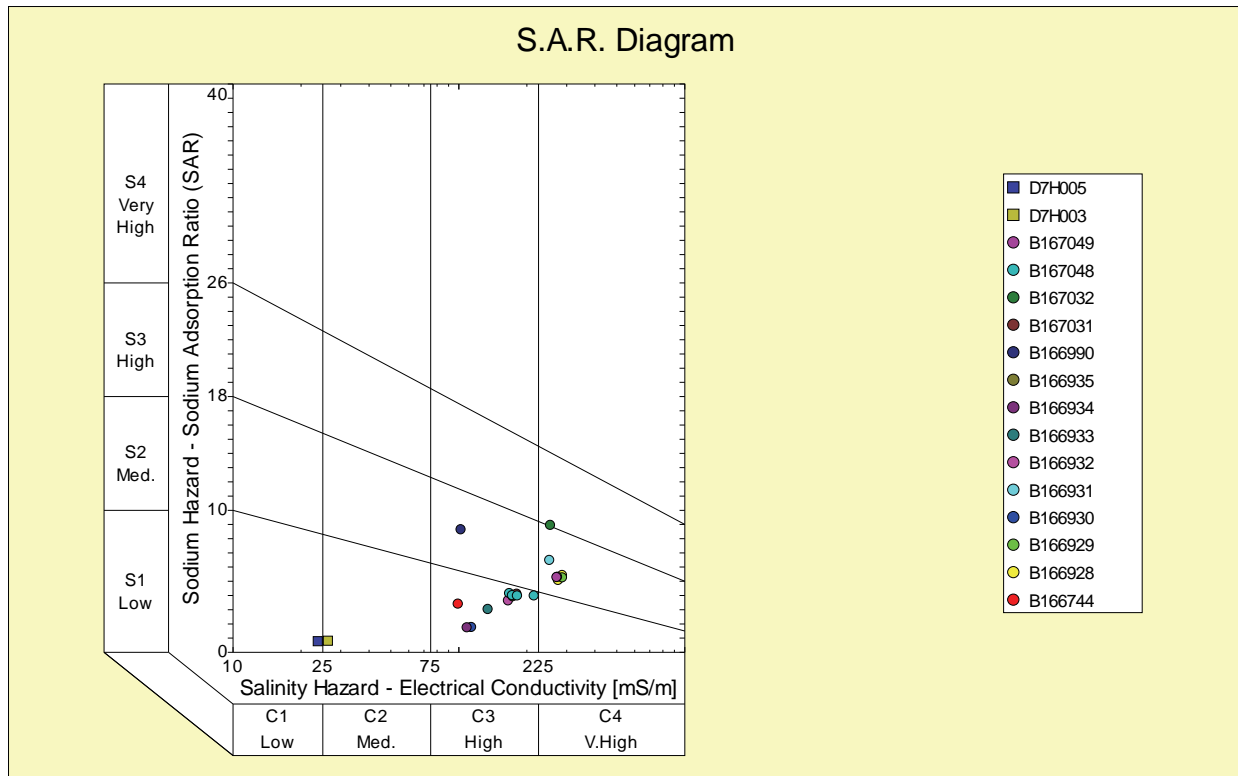


Figure 4-3-8 S.A.R diagram for the 14 borehole samples used for the defining of MCM groundwater sources within the quaternary catchment D73F study area

The MCM results from the model run assigning no abstraction volume to the river cell and making use of scenario 2 are used for comparison purposes. The total groundwater contribution to baseflow within the study area was estimated at 0.00 Mm³/a by the MCM. Groundwater baseflow estimates from the Pitman, Hughes and Sami models for the quaternary catchment D73F as well as an estimate from another chemical method (Tracer method) using the same data as the MCM. The Pitman, Sami and Hughes model groundwater baseflow estimates for the quaternary catchment D73F were all 0.00 Mm³/a (Table 4-3-4). The Tracer method estimate is shown for both the use of only electrical conductivity (EC) as well as an average value from using a number of tracers. The Tracer

method groundwater baseflow estimate for the quaternary catchment D73F, using only EC was 93.49 Mm³/a, while for an average from a number of tracers was 119.18 Mm³/a.

Table 4-3-4 The groundwater baseflow estimates from the Pitman model, Sami model, Hughes model, tracer method and MCM for the quaternary catchment D73F.

Quat	Pitman Uncalibrated (Mm ³ /a)	Sami Uncalibrated (Mm ³ /a)	Hughes Uncalibrated (Mm ³ /a)	MCM (Mm ³ /a)	Tracer Method (Mm ³ /a)	
					EC	Average
D73F	0.00	0.00	0.00	0.00	93.49	119.18

The Tracer method estimate of groundwater baseflow is extremely unreasonable for this regionally-defined zero groundwater baseflow study area. The large flow volumes associated with the Orange River further highlight the Tracer method's short-coming, in that the method is principally dependent on the flow volumes assigned. The MCM is much less dependent on the flow volume assigned because only a slight difference in the groundwater baseflow estimate is seen when abstraction volumes from the river are included. The MCM is however more sensitive to changes in tracer concentration data and weighting factors assigned, as seen in the drastic change in the groundwater baseflow estimate between scenario 1 and 2 weighting factors. This sensitivity highlights the models shortcomings in terms of undefined sources to the system and non-compliance with the assumption of conservative tracers. However, when tracers showing high chemical mass balance percentage errors are assigned lower weighting factors, the MCM does estimate the groundwater baseflow accurately along with the Pitman, Hughes and Sami models.

5. Overview discussion of MCM results

The application of the Mixing Cell Model to the quantification of groundwater-surface water is discussed in terms of: the groundwater baseflow estimates determined in each of the three study areas and their respective interpretations; the limiting factors and assumptions imposed on the MCM by the scope of this study; and lastly the data restrictions imposed on the application of the MCM in a South African setting.

The first pilot study area is located within a regionally-defined zero groundwater zone according to the Mean Annual contribution of Groundwater to Baseflow Map produced by the Department of Water Affairs (DWA, 2006b). The Pitman, Sami and Hughes models groundwater contribution to streamflow volumes were all zero for the quaternary catchment C52H, wherein the UFS groundwater-surface water interaction test site falls. The MCM estimated the average groundwater contribution to streamflow at $0.52 \text{ Mm}^3/\text{a}$ for the two time periods, while the Tracer method for EC-only estimates the average groundwater contribution at $0.59 \text{ Mm}^3/\text{a}$. The Mean Annual contribution of Groundwater to Baseflow map of South Africa indicates that the Pitman, Sami and Hughes models were correct in estimating the groundwater baseflow at zero, but the general nature of the map often results in inaccuracies on a quaternary catchment scale. Investigations on the UFS test site performed by Gomo (2011) found that groundwater was reaching the river within this area. Gomo (2011) suggested that the groundwater-surface water interaction occurring at this location is not taking place by traditionally-defined groundwater-surface water interfaces, based on the geology of the area and positive indications of groundwater contributions to streamflow. The concurrence between the MCM and Tracer method average groundwater baseflow volumes ($0.52 \text{ Mm}^3/\text{a}$; $0.59 \text{ Mm}^3/\text{a}$) is a good indication of the overall accuracy of these methods. The fact that both the MCM and the Tracer method indicated groundwater to be contributing to streamflow, while traditional methods did not, suggests that methods making use of water quality data could identify groundwater contributing to streamflow by unconventional pathways, where traditional methods could not. However, additional empirical studies and direct measurements are required to confirm this hypothesis.

The middle Modder River was divided into three sections based on the location of active flowstations in the investigated area. Section one is located within the quaternary catchment

C52B. The Pitman model estimated the groundwater baseflow in quaternary catchment C52B at zero. The Sami model has a low, but positive groundwater baseflow estimate of $0.03 \text{ Mm}^3/\text{a}$, while the Hughes model estimated the groundwater baseflow at $5.03 \text{ Mm}^3/\text{a}$. This quaternary catchment falls within a regionally-defined zero groundwater baseflow zone, yet the Sami and Hughes models have found that there is indeed groundwater contributing to the river. The uncalibrated estimates for these two methods (Sami and Hughes) are not in agreement, with a difference of $5 \text{ Mm}^3/\text{a}$. The MCM estimated the average groundwater baseflow at $0.09 \text{ Mm}^3/\text{a}$, while the Tracer method (EC) estimated an average groundwater baseflow of $1.31 \text{ Mm}^3/\text{a}$. The MCM estimate of groundwater baseflow is much lower than both the Tracer method and Hughes model estimates, but is in agreement with the estimate made by the Sami model of $0.03 \text{ Mm}^3/\text{a}$. This lower groundwater baseflow volume could be attributed to the fact that this section is the only section that includes a tributary source to the river mixing cell. The other sections also receive water from tributaries, but these tributaries weren't accessible for water samples to be taken. The inclusion of the tributary source improved the MCM estimate of groundwater baseflow when compared to the Sami estimate, but worsened when compared to the Tracer method. The quaternary catchment C52B still flows over Beaufort Group rocks of the Adelaide sub-group and the alternating layers of sandstone and shale could provide a reasonable supply of groundwater to the river. From these results and the underlying geology it seems that the MCM and Tracer method produce reasonable groundwater baseflow estimates, with the MCM being slightly more suitable due to the inconsistency of the Tracer method between the two time periods. The Tracer method is also variable between the use of only EC and multiple tracers, which makes the MCM better suited for the incorporation of multiple tracer concentration data.

Section two is located within the quaternary catchment C52E. The Pitman and Sami models determined that no groundwater was contributing to streamflow in C52E, while the Hughes model estimated the groundwater baseflow at $2.22 \text{ Mm}^3/\text{a}$. The MCM determined the average groundwater baseflow estimate at $1.30 \text{ Mm}^3/\text{a}$ and the Tracer method determined the average groundwater baseflow estimate at $0.78 \text{ Mm}^3/\text{a}$. Once again the Sami and Pitman models are correct based on the Mean Annual contribution of Groundwater to Baseflow map, but the Hughes, MCM and Tracer methods indicate that groundwater is indeed contributing to the stream in this area. The Modder River in Section two mainly flows over Beaufort Group rocks of

the Adelaide sub-group that assumedly would be able to supply groundwater to the stream, if the two resources were connected. Determining whether the Sami and Pitman models indicating no groundwater baseflow or the Hughes, MCM and Tracer methods indicating groundwater baseflow were correct in this instance would require additional study of the area.

Section three is located within the quaternary catchment C52G. The Pitman and Sami models determined that no groundwater was contributing to streamflow in C52G, while the Hughes model estimated the groundwater baseflow at 5.35 Mm³/a. The average groundwater baseflow estimate for the MCM is 2.37 Mm³/a, and for the Tracer method 2.42 Mm³/a.

From the results of the middle Modder River study area, it can be seen that the Sami and Hughes models are not always in agreement. The Hughes model estimate of the groundwater baseflow is on average 5 Mm³/a higher than the Sami model groundwater baseflow estimate. The MCM and Tracer method groundwater baseflow estimates tend to be in-between the two model estimates. For example, in section three the Sami, MCM, Tracer and Hughes models' groundwater baseflow estimates are 0.00 Mm³/a, 2.37 Mm³/a, 2.42 Mm³/a and 5.35 Mm³/a, respectively. Again, further studies would need to be conducted on these areas in order to validate these findings.

The continuous MCM run for the entire middle Modder River section presented as a demonstration for the models applicability to ungauged catchments provided reasonable percentages of inflow from the defined groundwater sources. Further study would be required to confirm that the MCM would be the most appropriate method to apply as an initial estimate of groundwater baseflow to ungauged river systems.

The second pilot study area in the Limpopo falls within a groundwater zone of 6 000-10 000 m³/km²/a zone according to the Mean Annual contribution of Groundwater to Baseflow map (DWAF, 2006b). Using the area of each quaternary catchment, the groundwater baseflow zone equates to 3.5-6.0 Mm³/a for A42A, 3.1-5.2 Mm³/a for A42B and 4.2-7.0 Mm³/a for A42C. The uncalibrated Pitman, Sami and Hughes models groundwater contribution to streamflow estimates for the quaternary catchment A42A were 17.99 Mm³/a, 8.70 Mm³/a and 15.62 Mm³/a, respectively. The uncalibrated Pitman, Sami and Hughes models groundwater contribution to streamflow estimates for the quaternary catchment A42B were 16.86 Mm³/a, 6.56 Mm³/a and 15.39 Mm³/a, respectively. The uncalibrated Pitman, Sami and Hughes models

groundwater contribution to streamflow estimates for the quaternary catchment A42C were 22.41 Mm³/a, 8.83 Mm³/a and 20.26 Mm³/a, respectively. The groundwater baseflow estimates from the three models for each quaternary catchment are highly variable, with a maximum difference of 11 Mm³/a (between the Sami and Hughes models). The calibration of the Sami model results in an increase in the groundwater baseflow estimate for quaternary catchment A42A (8.70 Mm³/a to 8.99 Mm³/a), for the quaternary catchment A42B (6.56 Mm³/a to 8.50 Mm³/a), and for the quaternary catchment A42C (8.83 Mm³/a to 10.28_(E) Mm³/a). The calibration of the Hughes model results in a large decrease in the groundwater baseflow estimate for the quaternary catchment A42A (15.62 Mm³/a to 11.90 Mm³/a), for the quaternary catchment A42B (15.39 Mm³/a to 11.72 Mm³/a), and for the quaternary catchment A42C (20.26 Mm³/a to 15.43_(E) Mm³/a). From these trends in calibration, the Sami model is an under-estimation of the groundwater baseflow, while the Hughes model is an over-estimation of the groundwater baseflow. The area is extremely data scarce in terms of water quality data and this is reflected in the MCM results as the groundwater baseflow estimate for the considered time period is only 1.21 Mm³/a, with a water balance error of 33% and an average chemical mass balance error of 10%. The MCM groundwater baseflow volume seems to be a substantial under-estimation when compared to the other methods. The Tracer method however does give a reasonable groundwater estimate of 9.83 Mm³/a using EC (similar to the Sami model estimate). The Tracer method is designed to estimate the groundwater baseflow from an entire upstream catchment making use of only a single downstream sample point. In the investigation, the Tracer method produced accurate estimates of the groundwater baseflow in a data-scare area, where only a single river sample point was available.

The third pilot study area (quaternary catchment D73F) falls within a regionally-defined zero groundwater baseflow zone according to the Mean Annual contribution of Groundwater to Baseflow map (DWAF, 2006b). The Pitman, Sami and Hughes models all estimated the groundwater contribution to streamflow as zero, for the quaternary catchment D73F. The MCM estimated the groundwater baseflow at 7.44 Mm³/a when all tracers were assumed conservative, but when the tracers showing high chemical mass balance errors were given a lower weighting factor, the estimated groundwater baseflow becomes 0.00 Mm³/a. The initially positive groundwater baseflow estimated by the MCM could be attributed to the omission of a pollution runoff source to the river which resulted in the runoff being erroneously assigned to a

high sodium borehole sample. The MCM proved more sensitive to the changes in the chemical mass balance than changes in the water balance. The assignment of an abstraction volume from the River Cell had no significant effect on either the groundwater baseflow estimate or the associated balance errors. However, changing the weighting factor scenario had a drastic effect on both the estimated groundwater baseflow and the associated balance errors. The Tracer method proved extremely sensitive to the flow volume assigned. The Tracer method estimated a large groundwater baseflow volume (93.49 Mm³/a) for the study area in response to the large outflow of the Orange River, in spite of the area not being able to supply groundwater baseflow to the river in such large quantities.

The MCM was not operating at its full capacity when applied to the pilot studies due to a number of factors associated with the limited scope of this study. The main limitation was the lack of isotope data. The MCM has the ability to incorporate isotope data into the chemical mass balance equations and the use of such data could have increased the accuracy of the groundwater baseflow estimates. The MCM also has the ability to determine the inflow to the river from both groundwater and interflow, but this capacity was not exploited within this study because water quality data required to define an interflow source is not readily available and difficult to measure. The omission of an interflow source to MCM river cell could also be a source of error in the estimated groundwater baseflow volumes seeing that the method is based on a water balance approach. The method has been investigated on the limiting assumption that the rivers on which the MCM was applied were gaining streams, thus determining positive groundwater flow from the aquifer to the river. The MCM application to the Northern Cape quaternary catchment located in a zero groundwater baseflow zone indicated that the model is able to accurately determine the groundwater baseflow even when there is none. The conceptual understanding of the area however leads to the hypothesis that the river is disconnected from the water table because the water table is far below ground level. The MCM can determine that there is no groundwater baseflow, but would not be able to simulate the flow from the river to the aquifer with the current mixing cell setup. A shift in the main focus of the mixing cells, from the river to the aquifer, is recommended for a losing river system investigation. This shift in focus would however require supplementary groundwater quality data close to the rivers to define alluvial channel aquifer mixing cells. The application of the MCM to connected/disconnected rivers requires further investigation.

The greatest limiting factor, as with most methods of groundwater-surface water interaction quantification in South Africa is data paucity. When the data available has been below perceived optimum levels for a realistic MCM application, the data has been referred to as insufficient. This rough description is used as there is no “required minimum number of data points per area” standard for a MCM application. In light of this short-coming a minimum standard description is attempted based on the findings of this study. It was found that the MCM did not produce optimum results when water quality data from a single river sample were used to define the river system. The MCM was able to produce reasonable estimates of the groundwater baseflow when small tributary inflows to a river segment are not accounted for, but was found to create large balance errors when larger tributary inputs were omitted. Water quality data from borehole samples relatively close to the river are required to define groundwater inflow sources, but terrestrial groundwater sources would be also beneficially. As many borehole samples are possible should be made use of in a MCM application, which can be incorporated as single groundwater sources or can be averaged to represent the general groundwater quality within an area. Considering the MCM setups and results from this study, a rudimentary indication of the minimum data requirements for a small, single-cell MCM application as well as a multi-cell catchment scale application is given. For a single cell MCM application, water quality data from at least two river samples are required to define the river cell and the inflow into that mixing cell; at least one groundwater sample is required to define a groundwater source to the river; if possible tributary inflows should be defined; and lastly an outflow volume from the defined system is required. For a catchment scale MCM application, a river sample should define a river mixing cell at least every 5km, with a river sample every 1km being the optimum situation; at least one borehole sample in relative proximity should define the groundwater source to each river mixing cell; tributary inflows should be defined where possible; and lastly an outflow volume from the defined system is required.

The MCM has been run for a maximum of two time periods for each study area, based on the available data. The water quality sample points and data records within most quaternary catchments are irregular and insufficient, which does not allow for a MCM to be run on a regular monthly or yearly basis. A continuous application of the MCM over an extended time period, which could then be averaged, would give a better estimate of the groundwater contribution to streamflow.

6. Overview discussion of Groundwater data

There is a theme of data paucity in South Africa (Jonck and Meyer, 2002; DWA, 2009; NGS, 2010; Pietersen *et al.*, 2011; Knüppe, 2011; Pietersen *et al.*, 2012). Data paucity poses a problem to groundwater resource assessments because groundwater data forms the building blocks of any method used to determine how much groundwater South Africa has and if these building blocks are insufficient or not available, the quantification of groundwater becomes inaccurate and the selection of a particular method becomes meaningless.

Knüppe (2011) and Pietersen *et al.* (2012) attribute the incomplete groundwater datasets in South Africa to a lack of governance and implementation as well as the general under appreciation of groundwater resources. Knüppe (2011) suggested that a change in mindset regarding groundwater and its significance would be the starting point for both improving the groundwater datasets in South Africa and the implementation of an adaptive management strategy.

However, there are a number of authors who have highlighted these shortcomings, pointed out where failures were found and recommendations for improvement. Yet, an improvement in the groundwater information has not been seen. The question is have the recommendations been implemented and the resulting changes are delayed or slow and therefore not noticeable, or have the numerous recommendations to improve data collection and accuracy been under prioritized and not implemented? A further challenge is to have a centralized access point to all groundwater information. The National Groundwater Achieve was meant to service as a central means of access, but not all data is filtered through this system, where the GRIP database is separate for example.

In order for groundwater to be used to its full potential and simultaneously responsibly managed, a good understanding of the groundwater systems in our country is needed. Understanding groundwater systems is complicated due to the fact that groundwater is an unseen resource, but good quality, continuous groundwater data will go a long way to improving this understanding and consequentially the management of groundwater too.

7. CONCLUSIONS

The findings of the Mixing Cell Model applications are:

- The traditionally used Pitman, Hughes and Sami models were found to give dissimilar groundwater baseflow volumes for most of the study areas. The Sami and Hughes models give better groundwater baseflow estimates than the well-known, hydrology-based Pitman model. The Sami model was found to under-estimate groundwater baseflow volumes, while the Hughes model over-estimated groundwater baseflow volumes, based on changes resulting from calibration. Based on the results of this study, the Sami model appears to produce more reasonable and stable groundwater baseflow estimates than the Hughes model.

- The Mixing Cell Model (MCM) application to groundwater-surface water interaction quantification has limitations in terms of its use in South Africa. The MCM requires both surface water and groundwater quality data for a successful model run. In data scarce areas of South Africa the water quality data required for a MCM might not always be available which will limit the use of this method. The use of insufficient water quality data in a MCM application has proven to result in severe under-estimation of groundwater baseflow. The chemical hydrograph separation method (Tracer method) however, proved to give reasonable groundwater baseflow volumes under these circumstances, but is highly dependent on the assigned outflow volume.

- The Tracer method and the MCM both require an outflow volume from the system to be defined in order to volumetrically determine the groundwater baseflow. The requirement of a volumetrically-defined outflow could prove problematic in ungauged catchments. The MCM can determine inflows to a system relatively as percentages of an assumed 100% outflow, in the absence of a known outflow volume, which could serve as a means of initially determining the groundwater baseflow in ungauged catchments.

- Both the water quality data based methods indicated groundwater baseflow to a system where the Pitman, Sami and Hughes models had indicated zero groundwater baseflow. The area was previously found to have untraditional groundwater-surface water

connections, implying that the water quality data methods might detect groundwater baseflow that traditionally defined methods could not.

- The MCM is better suited for the incorporation of multiple tracer datasets than the Tracer method because the latter showed considerable inconsistencies. The incorporation of multiple tracer concentration data allows for the maximum amount of data to be used, ensuring the best possible groundwater baseflow estimate.
- The MCM was found to be more sensitive to the tracer concentrations assigned than the flows defined in the model, indicating that the assumption of conservative tracers is the limiting assumption. The assignment of lower weighting factors to non-conservative tracers does however allow the MCM to overcome this limitation to some degree.
- The MCM application to groundwater-surface water interaction quantification has advantages in physical-parameter data scarce and complex geological areas of South Africa because the model only requires water quality and flow data.
- The MCM is not limited to a specific scale of investigation as with the Sami model. The MCM can be applied to quaternary catchment scale, multiple catchments and local, site-specific investigation scales. The MCM does however perform better on smaller scales because the amount of data required for large scale applications is extensive and tends to result in greater errors.
- The algorithm-based MCM accurately determined the groundwater baseflow for a quaternary catchment in a zero groundwater baseflow zone, proving that the model can produce accurate results in spite of not directly using physical geohydrological principles.
- The MCM is able to report the associated water and chemical mass balance error of each groundwater baseflow volume determined which is advantageous in terms of expressing the level of confidence in the estimates.
- When there is sufficient data for a MCM run, the MCM groundwater baseflow volume tends to be in-between the Sami and Hughes model estimates. Considering that the

Sami model was found to under estimate the groundwater baseflow and the Hughes model to over-estimate, the MCM is a good indication of the amount of groundwater contributing to a river.

In light of the findings of this study which found both advantages and disadvantages in using the Mixing Cell Model (MCM), the possible applications and scenarios where the use of the MCM would be beneficial are described. The MCM should be used to quantify the groundwater contribution to streamflow where:

- ✓ sufficient surface water and groundwater quality data is available or can be acquired,
- ✓ traditional methods of quantification are only able to produce low confidence estimates of groundwater baseflow (the MCM can be used to validate these estimates),
- ✓ traditional methods are not able to be utilised due to lack of physical-parameter data or complex geology, and/or
- ✓ a multiple-method approach is applied (the MCM can serve as an additional method incorporating chemical data).

The natural environment can hardly be described by linear, homogeneous expressions or with the assumption of conservative behaviour of solutes, especially in the geochemically very active hyporheic zone separating surface and groundwater. While the model allows for a partial compensation of such violations, the infringement of assuming conservative tracers is still likely to result in errors. We thus have to accept a certain level of inaccuracy in the quantification of natural phenomena such as the groundwater contribution to baseflow using the mixing cell or other methods. Similarly, data paucity and ungauged catchments limit the applicability of baseflow estimation models based on empirical data in large parts of the country. The number and maintenance of flowstations as valuable empirical data collection points should therefore be increased. The development of groundwater-surface water interaction test sites throughout the country is furthermore a major step towards improving the quality and quantity of data, with likely subsequent improvements in the different quantification tools once they can be tested against detailed data. Groundwater

contribution to streamflow has proven to be a challenging property to determine, however the benefits of quantifying this flux greatly outweigh the difficulties.

As a final conclusion, this report and the proposed method should be a step in a continual process aimed at improving the quantification of groundwater baseflow. The significant role of this estimation has the potential to greatly influence our environment and how the country's water resources are managed and utilized. The quantification of surface water-groundwater interaction is important, but it is also important to keep the bigger picture in mind. Surface water-groundwater interaction forms only a small part of a larger groundwater resource assessment that is required to make management decisions to assist with both the protection and responsible use of South Africa's water resources.

8. RECOMMENDATIONS

Recommendations have been made based on the findings of this study in terms of both surface water-groundwater interaction quantification, application of the Mixing Cell Model, and groundwater resource assessments.

The Mixing Cell Model (MCM) is recommended as an additional tool used to quantify the groundwater baseflow volume in South Africa, and the following scenarios are suggested:

Validation:

The MCM uses a completely different dataset from the currently applied Sami and Hughes models for groundwater baseflow quantification. The MCM thus could serve as a means of confidently validating the groundwater baseflow volumes estimated by the traditional models.

Low confidence groundwater baseflow areas:

The GRA2 dataset which made use of the Sami model to quantify groundwater baseflow is known to have areas where the groundwater baseflow volumes have a low confidence. The MCM could be applied to these areas to increase the confidence of the estimated groundwater baseflow volume.

Physical-parameter data scarce and complex geology areas:

The MCM is able to estimate the groundwater baseflow without geohydrological parameters such as hydraulic conductivity (K) and the various parameters required for the Sami and Hughes models. Complex geology tends to limit the ability to determine such aquifer parameters which might limit the applicability of the traditional methods. The MCM could be applied to such areas to increase the confidence in the groundwater baseflow volume.

Ungauged quaternary catchments:

The MCM is able to determine the relative contribution of groundwater to a river system when there is no volumetrically known outflow from a flow system. The MCM could be used as an initial estimate of the groundwater baseflow in ungauged catchments.

Site-specific scale:

The MCM is not limited to a specific scale, allowing for both multiple catchment estimations and local scale applications. In highly impacted or high water use areas, the MCM can be applied on a small scale to more accurately determine the groundwater baseflow volume at these specific locations.

Multiple method approach:

The MCM could serve as a valuable additional method to a multiple-method approach of quantifying groundwater baseflow because the method incorporates chemical data (a different dataset to traditionally methods). The multiple method approach for the quantification of groundwater-surface water interaction has been recommended by numerous authors.

The following additional recommendations are made based on the pilot testing of the MCM and quantification of surface water-groundwater interaction:

✓ Tracer method substitute:

When there is insufficient data for a MCM application, but a single river sample point at the outflow point of a catchment with water quality and flow data is available, the use of the Tracer method is recommended in place of the MCM. The Tracer method would allow for the incorporation of some chemical data when the MCM cannot be used.

✓ Improvement of water quality data and flowstation networks:

The above applications of the MCM are made based on the assumption that there is sufficient groundwater and surface water quality data for an accurate MCM run. It is therefore recommended that the water quality data monitoring system within South Africa be both maintained and expanded. The optimum situation for each quaternary catchment would be a water quality monitoring site at the inflow point, outflow and on each significant tributary to the main river system. Furthermore, in order to volumetrically determine the groundwater baseflow using the MCM, an outflow volume from the river system is required. It is thus also recommended that

the flowstation network be maintained and expanded. The optimum situation would be an active flowstation at the base of each quaternary catchment.

✓ Multiple method approach:

The quantification of groundwater baseflow is a difficult component of the hydrological system to define because it is unseen. Recharge is a similarly difficult component to quantify and this has resulted in it becoming common practice to use an average recharge volume from a number of methods. It is recommended that a similar approach be applied to the quantification of the groundwater contribution to streamflow, in that an average volume from methods such as the Sami model, Hughes model and Mixing Cell Model (MCM) are used instead of a groundwater baseflow volume from one individual method.

✓ Further investigation of the Mixing Cell Model (MCM):

The MCM requires further investigation in terms of a comprehensive sensitivity and statistical analysis, which was out of the scope of this study. A rudimentary indication of the minimum data requirements for a MCM application was given within the context of this study, but also requires further investigation and definition.

✓ Comprehensive groundwater-surface water interaction test sites:

In order to ultimately conclude which methods are able to accurately determine the groundwater baseflow, a perfectly-defined or theoretical test site should be developed to compare the various methods. The comparison would be conclusive as the actually groundwater baseflow would be known and all necessary data for each method would be available. The test site would be able to determine which method is best at optimum conditions (all necessary data is available) as well as which methods work best in data scarce environments as data could be systematically be omitted.

There is a large amount of assessment that is still required on the MCM, but it has the potential to greatly improve the existing estimates of the groundwater contribution to streamflow if the method is made use of correctly, implemented in the correct situations

and ultimately used as an additional method to a multiple-method approach for the quantification of groundwater-surface water interaction.

The following additional recommendations are made based on the review of the Groundwater Resource Assessment Phase II (GRA2):

- Review and improvement of methodologies applied for the GRA2 project for sections:
 - Quantification,
 - Planning Potential Map,
 - Recharge and Groundwater/Surface Water interaction,
 - Aquifer Classification, and
 - Groundwater Use.

- Further research is needed into methodologies for estimating groundwater volumes in South Africa, including the following:
 - Specific yield and specific storage in South African aquifers and the typical extent of the active zone or zone of fluctuation
 - Groundwater recharge, especially episodic and non-linear recharge and the effect of thresholds
 - Groundwater / surface-water interaction

9. POLICY RECOMMENDATIONS

Recommendations on groundwater information collection and accuracy in South Africa have been incorporated as policy recommendations.

The recommendations made by Pietersen *et al.* (2011) are re-stated here and once again recommended for implementation to achieve the goal of better data collection and improved sharing of groundwater information:

- Finalisation and expansion of the National Groundwater Archive (NGA),
- Registration of drillers and capturing of driller's groundwater data in public databases such as the NGA,
- Support water services institutions in the development and implementation of asset registers on groundwater infrastructure and monitoring of groundwater use and groundwater quality,
- Engage with relevant authorities to maintain hydrological and environmental monitoring programmes necessary for groundwater management; such as the rainfall monitoring systems maintained by the South African Weather Service,
- Develop and implement an integrated groundwater information system to support water services provision at municipal level. Improve the compatibility (and/or integration) of existing groundwater databases / information systems maintained by different institutions (including water quality databases and municipal groundwater asset registers), and
- Re-assess the funding required by DWA head and regional offices for groundwater monitoring, data capture as well as the operation and maintenance of groundwater infrastructure.

Better primary data is needed in the following areas

- Existing groundwater use in South Africa (actual quantities)
- Groundwater quality variations, both natural and anthropogenic
- Groundwater levels, including relationship to geology and topography
- Rainfall (both volumes and intensity) and rainwater quality
- River / stream flows

A promising field of research may be one that looks at finding proxy methodologies (e.g. derived from remote sensing) for estimating spatial hydraulic properties and processes. Primary data collection is very expensive, and if relevant hydraulic data can be better linked to remotely sensed phenomena then this might provide a way of collecting large amounts of data with lower cost. An example is the recent application of the GRACE satellite data to estimating groundwater volumes across large areas of the United States of America. Another example would be using crop areas or electricity consumption to estimate volumes of groundwater pumped. Proxies for hydraulic parameters could be calibrated by using small sub-sets of “real” data then rolled out at much larger spatial scales. With increasing demands for water coupled with a decline in real hydraulic data in South Africa this approach may need to be considered seriously in future.

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Appendix A

A step-by-step guide to the use of MCMsf

The MCMsf (Mixing Cell Model – steady flow) is accompanied with three example datasets which are used for tutorial purposes. The examples include a single-cell model, a four-cell model and a twelve-cell model. The four-cell model dataset is made use of and applied as an example flow system to present a step-by-step guide to a MCMsf model run. The conceptual box model representation of the simulated flows for the four-cell example model is shown in Figure A-1. From the flow diagram (Figure A-1) it can be seen that water from Cell 1 flows into Cell 2 and Cell 3, Cell 2 flows to Cell 3 and Cell 4, and Cell 3 flows to Cell 4. The outflow from the last cell, Cell 4 is defined within the model at $100 \text{ m}^3/\text{year}$. Fourteen inflow sources are defined for this flow system. Sources (Flow 1-4) are assigned to Cell 1, sources (Flow 5-9) are assigned to Cell 2, sources (Flow 10-12) are assigned to Cell 3 and sources (Flow 13-14) are assigned to Cell 4. The abstraction rates are assigned at $0.5 \text{ m}^3/\text{year}$, $3.5 \text{ m}^3/\text{year}$, $15.5 \text{ m}^3/\text{year}$ and $5.0 \text{ m}^3/\text{year}$ for the Cells 1-4, respectively.

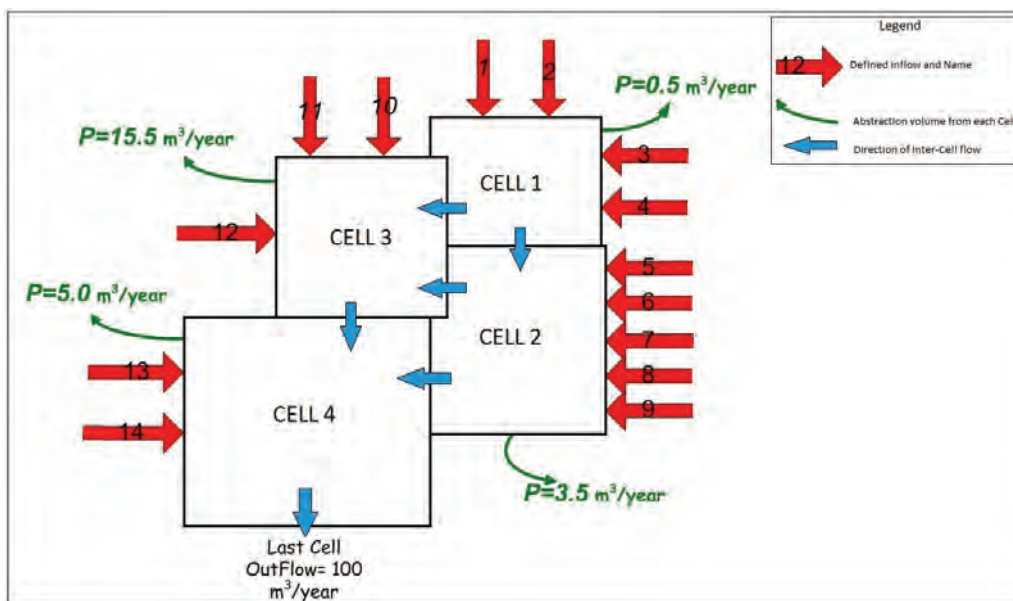


Figure A-1 Conceptualise box model flow diagram for the MCMsf four-cell example (Modified from Adar (2012)).

The MCMsf follows a specifically structured procedure to define the mixing cell model and create an input matrix for the FORTRAN© solver code based on the Wolf Algorithm. The

structure of the process followed within the MCMsf programme is shown in Figure A-2. This guide to using the MCMsf programme will thus also follow these steps.

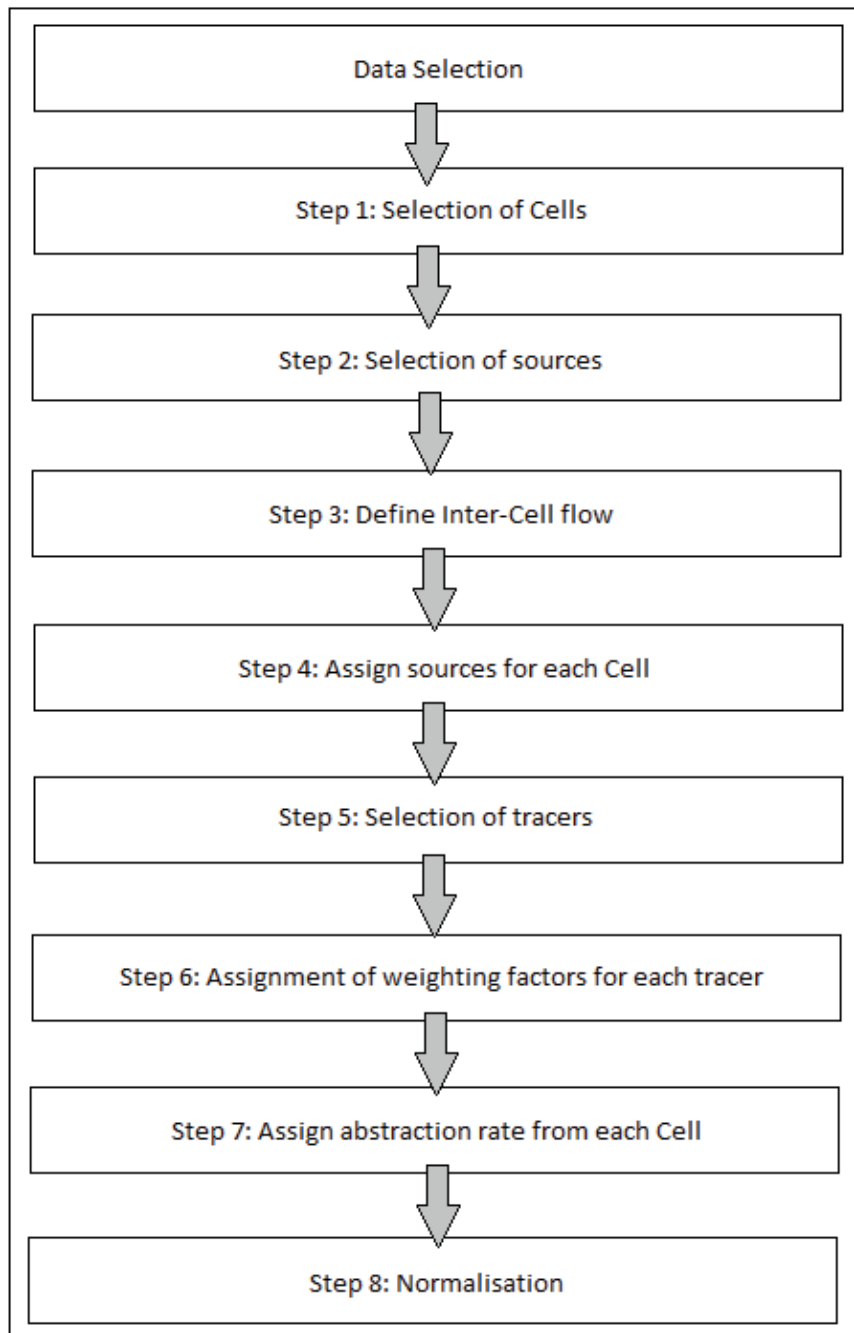


Figure A-2 The structured procedure followed in the MCMsf programme to define a mixing cell model (Modified from Adar (2012)).

Data Selection

The dataset from the MCMsf included four-cell example is made use of and shown in Table A-1. The water quality data forms the input into the programme is stored in an Excel format and requires a specific order of column labels to be read by the MCMsf programme (Table

A-1). The data is first imported into the Microsoft® Access database and then is shown within the MCMsf programme under *List of Table Data* shown on the user-friendly interface as the MCMsf programme is started (Figure A-3). In the case of the example four-cell model, the data is automatically available for selection within the MCMsf programme. The “tbl_data_4_14” is selected from the *List of data Tables*.

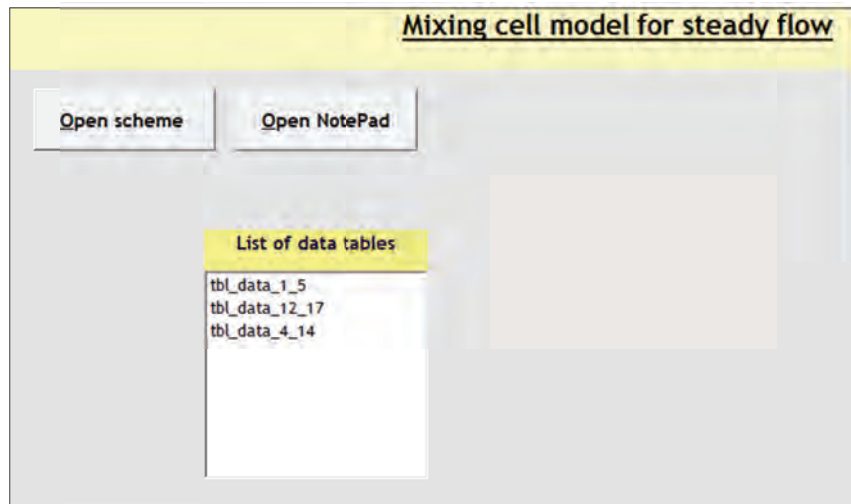


Figure A-3 MCMsf programme interface for the Data Selection step

Selection of Cells

The selection of Cells MCMsf interface is shown in Figure A-4. This interface allows the user to select the water quality analysis datasets which are used to define the mixing cell model Cells. For the four-cell example model, the datasets for Cell 1-Cell 4 are selected from the *List of Cells* displayed on the interface (Figure A-4).

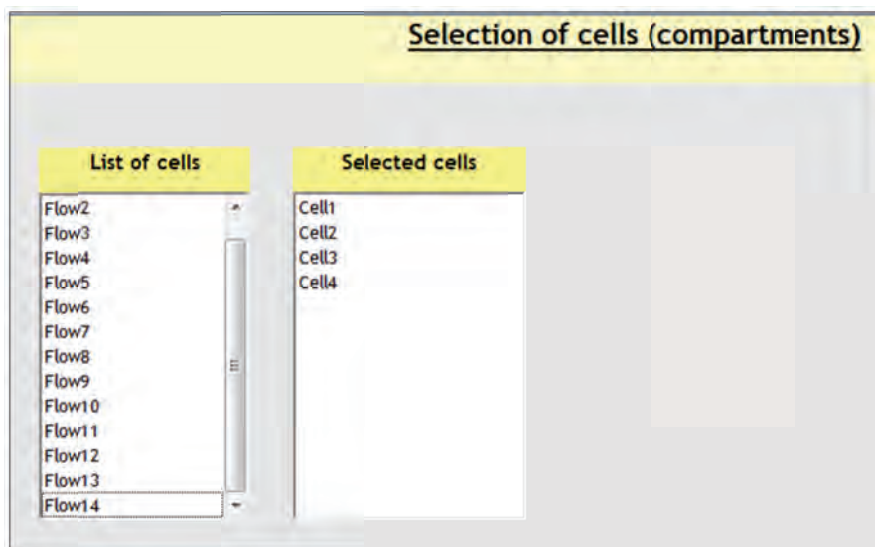


Figure A-4 MCMsf interface for Step 1 (Selection of Cells).

Table A-1 Water quality dataset for the MCMsf four-cell example model.

Location	Sample	Longitude	Latitude	GWL	TDS	Mg	Ca	Na	K	HCO3	Cl	NO3	SO4	F	Li	Si	D	O-18
Flow1					353.80	0.74	1.97	0.97	0.06	3.08	0.21	0.08	0.42	0.023	0.0026	18.01	-73.80	-9.73
Flow2					187.10	0.36	0.81	0.39	0.04	0.89	0.09	0.00	0.80	0.001	0.0003	18.18	-66.30	-8.56
Flow3					146.00	0.53	1.13	0.51	0.05	1.40	0.08	0.04	0.66	0.006	0.0012	17.53	-60.60	-8.55
Flow4					332.30	0.62	2.31	0.69	0.17	2.97	0.15	0.03	0.38	0.025	0.0013	16.83	-58.70	-7.41
Flow5					304.50	0.10	0.24	2.01	0.05	2.52	0.26	0.02	0.05	0.036	0.0022	14.00	-67.20	-8.70
Flow6					370.30	0.77	1.67	1.70	0.07	3.04	0.69	0.22	0.14	0.008	0.0032	8.80	-63.10	-6.52
Flow7					454.00	1.28	2.52	0.98	0.03	3.71	0.42	0.06	0.64	0.012	0.0016	16.35	-76.30	10.18
Flow8					242.00	0.39	1.51	0.33	0.06	1.20	0.09	0.04	0.63	0.030	0.0020	17.70	-67.10	-8.98
Flow9					332.30	0.62	2.31	0.69	0.17	2.97	0.15	0.03	0.38	0.025	0.0013	16.83	-58.70	-7.41
Flow10					356.00	1.72	3.18	0.73	0.03	3.64	0.14	0.05	1.93	0.015	0.0016	14.80	-67.00	-8.43
Flow11					147.80	0.84	0.41	0.33	0.03	0.07	0.07	0.01	0.98	0.120	0.0020	7.80	-68.90	-9.36
Flow12					362.00	0.66	2.06	0.87	0.05	2.93	0.14	0.08	0.51	0.022	0.0023	17.10	-69.30	-9.66
Flow13					242.00	0.39	1.51	0.33	0.06	1.20	0.09	0.04	0.63	0.030	0.0020	17.70	-67.10	-8.98
Flow14					332.30	0.62	2.31	0.69	0.17	2.97	0.15	0.03	0.38	0.025	0.0013	16.83	-58.70	-7.41
Cell1					321.91	0.68	1.79	0.87	0.06	2.71	0.18	0.07	0.48	0.019	0.0022	17.98	-71.82	-9.46
Cell2					323.27	0.65	1.71	0.92	0.06	2.62	0.21	0.06	0.46	0.021	0.0021	17.23	-70.55	-9.24
Cell3					303.85	0.98	1.94	0.74	0.05	2.50	0.15	0.05	0.96	0.037	0.0020	15.13	-69.63	-9.16
Cell4					317.59	0.77	1.91	0.80	0.08	2.64	0.18	0.05	0.64	0.028	0.0019	16.35	-67.94	-8.86

Selection of Sources

The selection of sources MCMsf interface is shown in Figure A-5. This interface allows the user to select the water quality analysis datasets which are used to define the inflows to the MCM Cells. For the four-cell example model, the datasets for Flow 1-14 are selected from the *List of Sources* displayed on the interface (Figure A-5).

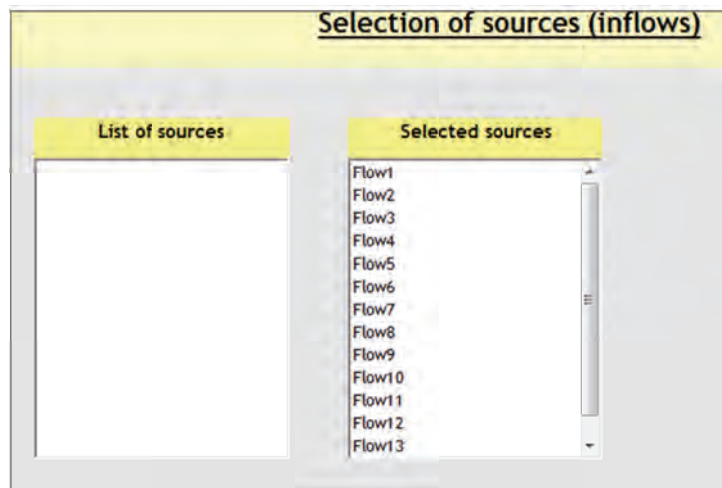


Figure A-5 MCMsf interface for Step 2 (Selection of Sources).

Definition of Inter-Cell flow

The inter-cell flow is assigned by first selecting the Discharging Cell from the *Discharging cell* list on the MCMsf interface and then selecting a Receiving Cell from the *Receiving cell* list (Figure A-6). For the four-cell example model Cell 3 is selected as the discharging cell and Cell 2 and Cell 3 are then selected as receiving cells. Similarly, flow from Cell 2 to Cell 3 and Cell 4 as well as flow from Cell 3 to Cell 4 are assigned.

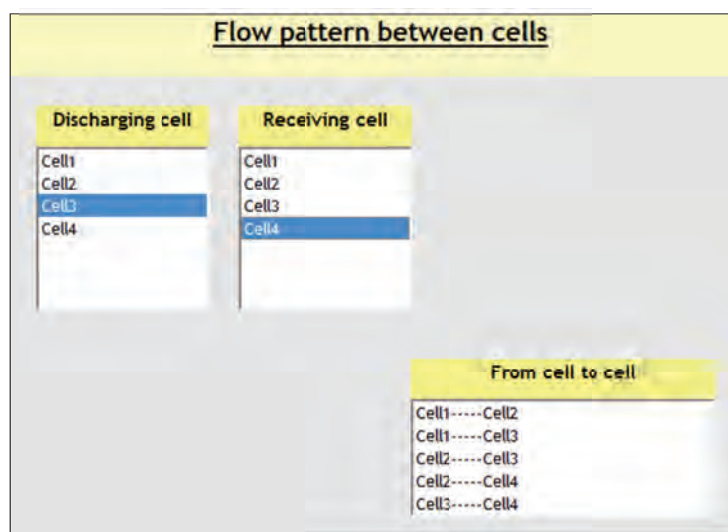


Figure A-6 MCMsf interface for Step 3 (Defining Inter-Cell flows).

Assignment of Sources to Cells

The MCMsf programme then requires the user to specify which of the chemically defined sources are have been identified as contributing to which Cells. For the MCMsf four-cell example, the defined sources (Flow 1-4) are assigned to Cell 1 by first selecting Cell 1 in the *Cells* list and then selecting each of the sources (Flow 1-4) in the *Sources* list. Similarly, sources (Flow 5-9) are assigned to Cell 2, sources (Flow 10-12) are assigned to Cell 3 and sources (Flow 13-14) are assigned to Cell 4 (Figure A-7).

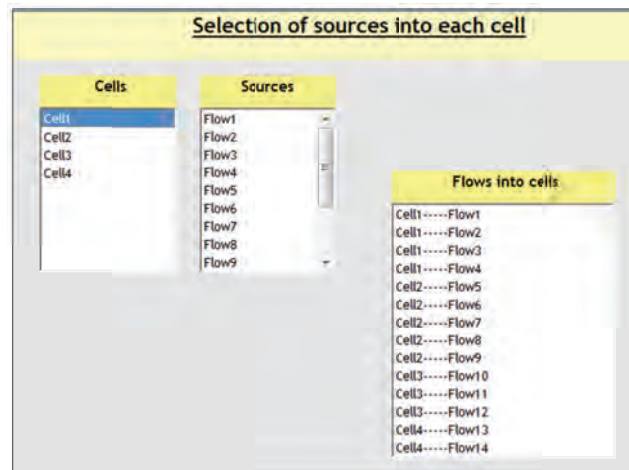


Figure A-7 MCMsf interface for Step 4 (Assign sources for each Cell).

Selection of Tracers

A list of tracers from the dataset loaded into the MCMsf is now shown in order to allow the user to selection which tracers are to be used in the chemical mass balance equation. For the four-cell example, all the available tracers are selected for use in the MCMsf programme, namely TDS, Mg, Ca, Na, K, HCO₃, Cl, NO₃, SO₄, F, Li, Si, D (Deuterium) and O-18 (Figure A-8).

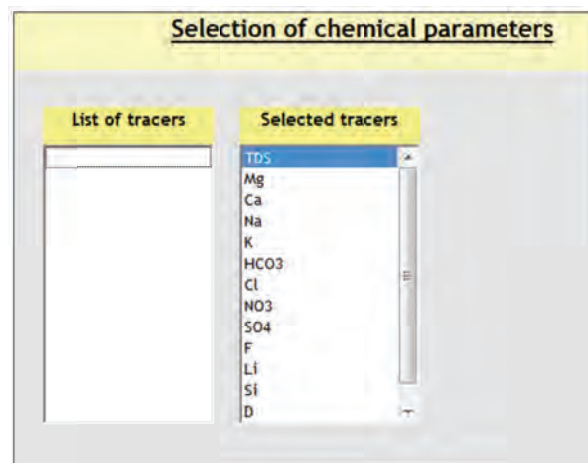


Figure A-8 MCMsf interface for Step 5 (Selection of Tracers).

Assignment of weighting factors for each tracer

The MCMsf programme allows weighting factors to be assigned to individual tracers in order to compensate for their lack of total conservation of that tracer. For the MCMsf four cell example all tracers are given a weighting factor of 1 which is the default setting (Figure A-9). In order to change the weighting factor the user simply selects the tracers to be changed and assigns a new weighting factor in a pop-up screen.

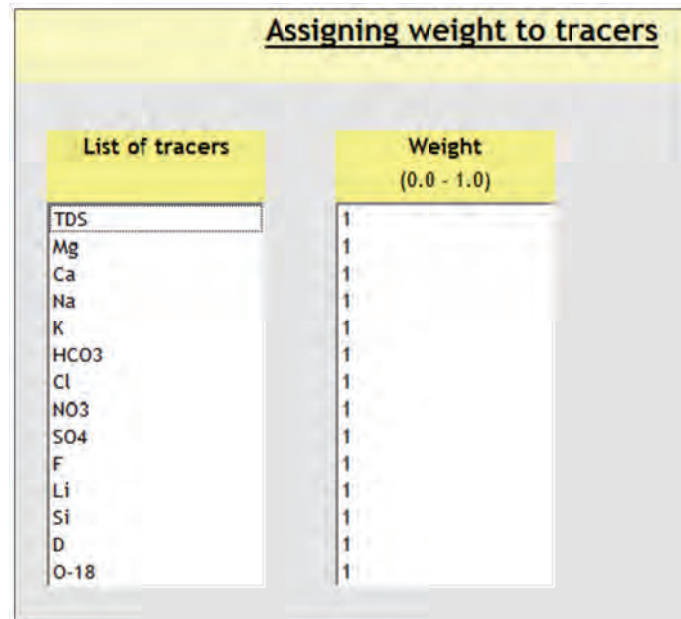


Figure A-9 MCMsf interface for Step 6 (Assignment of weighting factors for each tracer).

Assignment of abstraction rates for each Cell

The MCMsf programme allows the abstraction volume from each cell to be defined. The abstraction volume in the four-cell example for Cell 1 is $0.5 \text{ m}^3/\text{year}$. To assign this volume in the MCMsf programme Cell 1 is first selected from the *Cell name* list and then the abstraction volume can be inserted into a pop-up screen. Once inserted, the abstraction volume assigned will be shown in the *Outflow (volume/time)* list (Figure A-10). Similarly, the abstraction volumes of Cell 2, Cell 3 and Cell 4 of $3.5 \text{ m}^3/\text{year}$, $15.5 \text{ m}^3/\text{year}$ and $5.0 \text{ m}^3/\text{year}$, respectively are assigned. The total outflow from the last cell is also required to be defined by the MCMsf programme which can be done in the *Outflow from the last cell* insertion block in the bottom right-hand corner of the interface screen. In the case of the four-cell example the outflow from the last cell is set to $100 \text{ m}^3/\text{year}$.

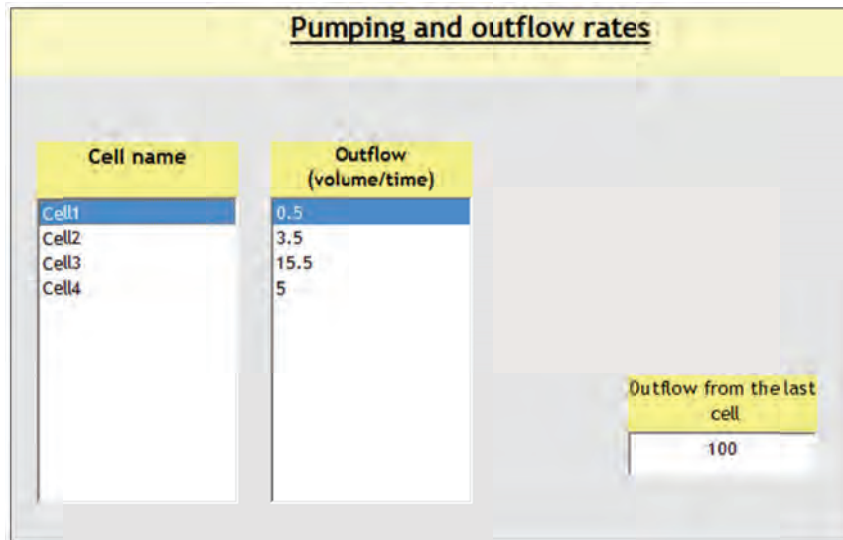


Figure A-10 MCMsf interface for Step 7 (Assign abstraction rate from each cell).

Normalisation

The MCMsf programme allows the user to select a set of tracer concentrations which are suitable to use a reference for normalisation. The MCMsf programme automatically indicates which datasets are suitable and shown under the *List of Cells* list. For the MCMsf four-cell example Cell 4 is used as a reference for normalisation and simply assigned by selecting Cell 4 from the list (Figure A-11). The MCMsf programme also allows for the option of selecting no dataset for normalisation by clicking on the button in the bottom right-hand corner of the screen indicating *Without Normalisation* (Figure A-11).

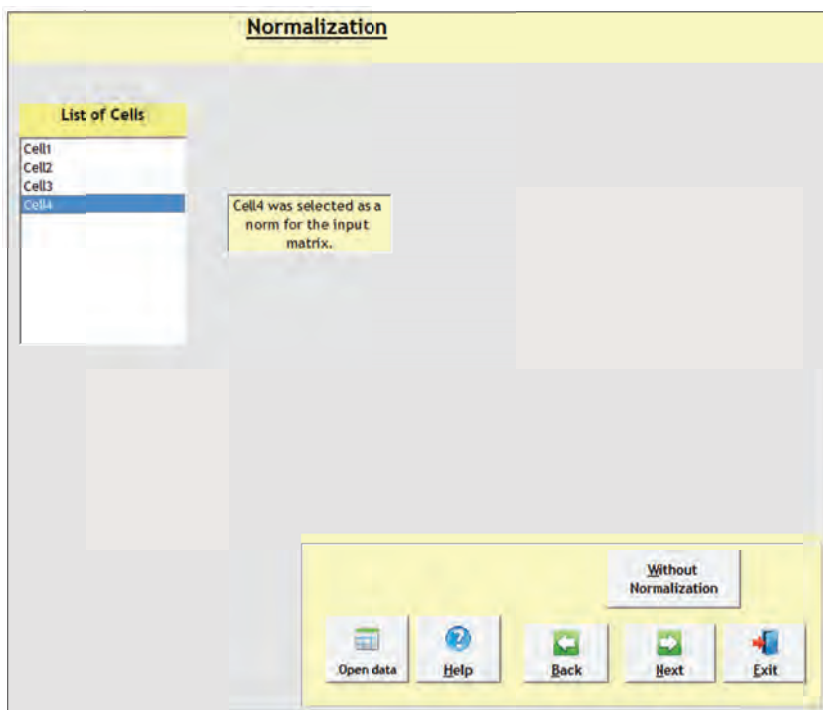


Figure A-11 MCMsf interface for Step 8 (Normalisation).

Input Matrix and FORTRAN© solver execution

The completion of Step 8 (Normalisation) is the final step in defining the Mixing Cell Model. One completion of this step, an additional interface will appear which then initiates the creation of the input matrix for the Wolf algorithm, save the input file and then allow the input file to be run in the Wolf Algorithm FORTRAN solver (Figure A-12). The input matrix has to be created first by selecting the *Build Input Matrix* button, secondly the input matrix has to be saved by selecting the *Save Input File* button and lastly the input matrix is used for a Wolf algorithm run by selecting the *Run Wolf Algorithm* button.

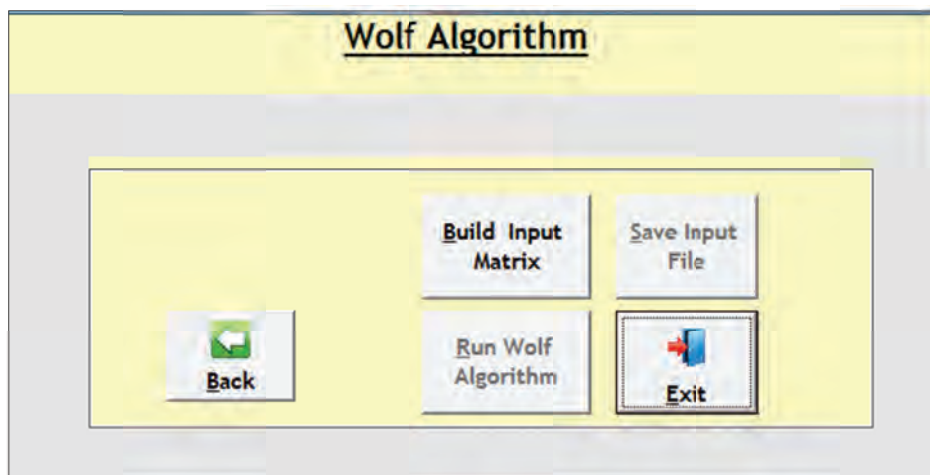


Figure A-12 MCMsf interface for the creation and application of the input matrix.

The FORTRAN solver has a basic interface shown in Figure A-13. The solver requires initially the input file name under which the created input matrix was saved. The user is then asked to assign a name to the output file, whether a printout of concentration data and the maximum number of iterations allowed for the algorithm. Once the number of iterations has been entered the solver will be executed and an output file created within the MCMsf Program File.

```

*****
Mixing Cell Model MCM (Double Precision )
*****
*** Type the name of Input File (data file)
name.txt
*** Type the name of Output File (results)
outname.txt

Do you want a printout of all concentration data
*** TYPE Yes = 1, No = 0

1
1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
16
17
18
Input number of iterations for WOLF
100_

```

Figure A-13 The FORTRAN© solver interface

MCMsf Output

The resultant output file from the MCMsf four-cell example model run is:

```

*****
** Estimation of Flow into and between Cells and Transmissivities **
**           Program MCMsf-Wolf Algorithm           **
*****

```

Data_4_14

The date is: 03:20:2005

```
*****
```

*** Input File In414.txt

*** Output File Output414.

```
*****
```

There are 14 potential inputs.
There are 14 tracers to be considered.
There are 4 cells in this model.
There are 5 flows between the cells.

N = 14 NN = 18 NOC = 4 NOP = 14

Number of inflows to each cell:

```
Num.cell 1 2 3 4
Inflows 4 5 3 2
```

Number of internal flows from and into each cell:

```
Num.cell 1 2 3 4
Interfl. 2 3 3 2
```

The internal flows are:

From cell 1 1 2 2 3
 To cell 2 3 3 4 4
 The outflow out of the last cell is [m³/year]:
 QOUT = 100.00

The rate of output (pumpage) and/or evapotranspiration from each cell is [m³/time]:

- PM(1) = .50
- PM(2) = 3.50
- PM(3) = 15.50
- PM(4) = 5.00
- Constant for Flows = 5.000
- PM(1) = .10
- PM(2) = .70
- PM(3) = 3.10
- PM(4) = 1.00

The weighting parameters are:

1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000 1.000
 1.000 1.000

There are 60 mass balance equations.
 Number of iterations: 28

The rest of the variables are equal to zero.

The unknown inflows are:

	Name of inflow	Rate of inflow	Perc. of tot. inflow	Perc. of cell inflow	flux
Cell 1					
1	Flow1	= 8.95	36.0%	79.7%	44.74
2	Flow2	= 1.48	6.0%	13.2%	7.42
3	Flow3	= .50	2.0%	4.5%	2.51
4	Flow4	= .30	1.2%	2.6%	1.48
Cell 2					
5	Flow5	= .80	3.2%	22.5%	3.98
6	Flow6	= .30	1.2%	8.4%	1.49
7	Flow7	= .83	3.3%	23.4%	4.14
8	Flow8	= 1.20	4.8%	33.8%	5.99
9	Flow9	= .42	1.7%	11.9%	2.11
Cell 3					
10	Flow10	= 3.00	12.1%	49.2%	15.01
11	Flow11	= 2.01	8.1%	32.9%	10.03
12	Flow12	= 1.09	4.4%	17.9%	5.45
Cell 4					
13	Flow13	= .00	.0%	.0%	.00
14	Flow14	= 4.01	16.1%	100.0%	20.05

Estimated fluxes for unknown inflows

The intermediate flows are:

From cell	To cell	Rate of flow	Real number
1	2	7.096	35.48
1	3	4.034	20.17
2	3	.981	4.91
2	4	8.973	44.86
3	4	8.012	40.06

} Inter-Cell flows

Total: 124.41 100.00%
QOUT + PPP = 124.50

} Water balance error

Absolute diff.: .09
Percentage diff.: .07%

**** Ion balance over the entire basin ***

Id	SUMIN	SUMOUT	Abs. error:	Perc. error:
TDS	24.770	24.780	-.01	.0%
Mg	25.649	25.636	.01	.1%
Ca	24.864	24.861	.00	.0%
Na	24.714	24.743	-.03	-.1%
K	23.534	23.517	.02	.1%
HCO3	24.723	24.731	-.01	.0%
Cl	24.587	24.581	.01	.0%
NO3	24.937	25.015	-.08	-.3%
SO4	26.264	26.265	.00	.0%
F	25.634	25.690	-.06	-.2%
Li	25.372	25.153	.22	.9%
Si	24.689	24.716	-.03	-.1%
D	24.987	25.010	-.02	-.1%
O-18	25.018	25.042	-.02	-.1%

} Individual tracer chemical mass balance errors

Total salt transport:
 Observed output (LB/DAY): 1887.5
 Estimated input (LB/DAY): 1887.6

} Overall chemical mass balance error

End of MCMsf Model.

From the output file of the four-cell example model the inflow from each of the defined sources into their respective Cells were calculated as well as the flows rates for defined inter-cell flows. The water balance percentage error is reported along with the chemical mass balance percentage errors for each individual tracer. An overall chemical mass balance error is also reported as total salt transport with observed versus estimated.

Appendix B

Table B-1 Model run data for UFS surface water-groundwater interaction test site application (24/01/2011). All values expressed in mg/l, except EC expressed in mS/m and pH expressed in pH units.

Name	pH	EC	Ca	Mg	Na	K	TAL	F	Cl	NO3	SO4
R 1	7.31	30.00	16.230	7.260	26.990	6.520	99.00	0.170	24.530	0.400	16.150
R 2	7.26	29.90	21.960	8.880	23.470	6.630	99.70	0.160	22.160	0.460	15.080
River Cell	7.29	29.95	19.095	8.070	25.230	6.575	99.35	0.165	23.345	0.430	15.615
o-River Cell	7.29	29.95	19.095	8.070	25.230	6.575	99.35	0.165	23.345	0.430	15.615
C5H039	8.11	23.33	18.182	6.975	15.267	4.795	79.35	0.299	15.737	0.489	10.991
BH1	7.45	96.90	39.320	40.650	113.100	5.100	411.00	0.560	57.000	0.370	25.650
BH2	7.36	90.10	39.420	40.090	93.290	8.230	381.00	0.490	54.000	0.820	21.870
BH3	7.57	109.00	36.310	50.060	157.340	5.590	589.00	0.670	70.000	0.180	31.270
BH4	7.48	89.10	39.950	43.300	100.120	5.420	381.00	0.510	53.000	0.370	21.320
BH5	7.46	105.00	35.540	56.600	123.660	5.580	462.00	0.620	61.000	0.000	26.310
BH6	7.45	99.30	41.660	48.910	114.490	5.580	426.00	0.550	60.000	0.230	25.760
BH7	7.43	100.00	34.350	49.500	111.730	5.700	430.00	0.570	60.000	0.080	26.050
BH8	7.68	105.00	35.650	47.350	124.830	5.600	436.00	0.550	69.000	0.250	32.020
BH9	7.55	104.00	37.550	45.180	121.540	5.310	424.00	0.570	70.000	0.320	34.050
BH10	7.52	93.40	37.290	42.370	103.350	5.260	394.00	0.540	57.000	0.340	24.820
BH11	7.61	94.90	34.620	41.430	108.990	5.500	398.00	0.530	59.000	0.330	26.410
BH12	7.63	93.60	36.320	41.590	106.680	5.400	395.00	0.490	57.000	0.310	24.870
BH13	7.63	94.50	35.470	42.210	108.560	5.410	396.00	0.510	59.000	0.370	26.560
BH14	7.46	88.30	40.940	42.280	86.500	4.750	380.00	0.410	51.440	0.510	20.560
BH15	7.49	86.50	40.450	41.530	84.140	4.850	373.00	0.360	48.840	0.510	20.020
SP	7.57	111.00	44.020	58.280	119.830	5.850	485.00	0.650	64.000	0.070	29.160

Table B-2 Model run data for UFS surface water-groundwater interaction test site application (05/08/2011). All values expressed in mg/l, except EC expressed in mS/m and pH expressed in pH units.

Name	pH	EC	Ca	Mg	Na	K	MAIc	F	Cl	SO4	Si
River 1	7.41	21.80	14.595	4.255	18.209	5.280	76.00	0.113	14.989	10.236	5.253
River 2	7.22	22.30	14.637	4.168	19.134	5.168	75.00	0.079	16.293	10.742	5.184
River Cell	7.32	22.05	14.616	4.211	18.672	5.224	75.50	0.096	15.641	10.489	5.219
o-River Cell	7.32	22.05	14.616	4.211	18.672	5.224	75.50	0.096	15.641	10.489	5.219
C5H039	7.99	22.91	23.559	7.557	16.787	4.431	84.15	0.418	15.594	11.632	6.431
BH1	7.53	107.00	44.471	48.253	122.343	5.557	443.00	0.550	86.176	37.235	20.435
BH2	7.72	84.80	34.481	41.926	89.049	5.052	376.00	0.516	54.000	21.319	21.721
BH3	7.50	108.00	36.251	51.339	132.011	5.615	477.00	0.375	71.772	28.159	23.648
BH4	7.46	87.70	36.664	41.400	94.401	5.313	392.00	0.554	56.000	22.060	21.791
BH5	7.25	168.00	53.622	107.197	172.241	7.310	613.00	0.512	206.021	82.338	23.225
BH6	7.30	102.00	45.229	53.007	114.453	6.102	449.00	0.401	70.726	28.009	23.684
BH7	7.31	109.00	43.493	63.526	123.158	6.590	470.00	0.246	84.179	31.530	23.932
BH8	7.54	124.00	39.770	59.230	143.891	6.038	509.00	0.555	97.549	53.305	23.031
BH9	7.36	165.00	56.389	82.750	209.587	6.612	539.00	0.734	204.496	105.043	20.254
BH10	7.33	91.90	41.908	47.327	108.947	6.111	407.00	0.618	59.000	25.806	22.192
BH11	7.37	93.70	35.749	42.731	108.666	5.684	410.00	0.501	63.000	25.578	21.483
BH12	7.51	93.60	36.871	44.146	106.863	5.610	408.00	0.456	65.000	25.819	21.756
BH13	7.36	92.50	35.968	41.626	103.871	5.527	411.00	0.428	59.000	24.291	21.079
BH14	7.40	86.60	45.459	45.852	86.420	4.997	388.00	0.351	53.000	20.639	21.641
BH15	7.44	84.80	41.044	41.705	82.906	5.008	383.00	0.328	51.028	22.154	21.274
Seep 1	7.52	135.00	54.446	74.174	139.166	5.737	533.00	0.265	106.088	74.832	21.846

Appendix C

Table C-1 Chemical tracer data for Section 1 (October 2012). All values expressed in mg/l, except EC expressed in mS/m and pH expressed in pH units.

Location	pH	EC	Ca	Mg	Na	K	MAIK	F	Cl	Br	SO4	TDS	Al	Fe	Mn	Si	Ba	Cu	Se	Zn
MR02	7.61	37.9	35.337	14.884	21.905	5.602	180	0.193	13.042	0.133	9.385	280.15	0.011	0.017	0.009	4.603	0.036	0.006	0.011	0.019
MR03	7.22	53.9	35.099	14.108	50.918	11.423	164	0.233	50.162	0.171	20.717	376.70	0.071	0.044	0.011	4.568	0.038	0.006	0.011	0.015
MR06-A	7.51	54.4	33.572	13.696	52.132	10.549	171	0.244	51.140	0.082	21.187	373.35	0.061	0.040	0.013	4.769	0.042	0.007	0.009	0.021
MR02 BH	7.48	50.3	49.713	22.522	27.097	0.930	245	0.215	9.673	0.017	20.290	377.73	0.009	0.018	0.009	12.621	0.007	0.003	0.013	0.048
MR03BH	7.94	46.2	24.210	2.838	71.667	0.538	203	1.597	9.515	0.075	28.584	342.24	0.010	0.012	0.053	7.984	0.003	0.004	0.010	0.072

Table C-2 Chemical tracer data for Section 1 (January 2013). All values expressed in mg/l, except EC expressed in mS/m and pH expressed in pH units.

Location	pH	EC	Ca	Mg	Na	K	MAIK	Cl	SO4	TDS	Si	B	Al	Ba	Cu	Fe	Mn	Zn
MR03	7.42	31.9	23.283	10.156	22.393	5.799	118	21.022	12.639	223	5.524	0.126	0.101	0.063	0.019	0.084	0.018	0.021
MR06ave	7.64	30.2	19.290	8.681	24.329	5.575	105	20.784	13.539	205	5.273	0.120	0.104	0.052	0.017	0.088	0.022	0.018
MR02	7.43	36.1	29.683	13.150	19.658	4.963	163	12.952	9.468	254	5.284	0.142	0.060	0.069	0.012	0.065	0.016	0.019
MR02BH	7.42	50.9	47.263	21.585	23.878	0.871	243	9.788	20.590	371	13.018	0.133	0.012	0.014	0.012	0.026	0.014	0.111
MR03BH	7.68	49.1	32.714	9.738	60.622	0.779	223	8.640	25.801	364	8.856	0.512	0.019	0.007	0.012	0.025	0.066	0.610
MR06B	7.73	47.2	30.701	14.095	41.556	6.259	160	43.657	18.251	316	5.051	0.120	0.102	0.094	0.013	0.090	0.025	0.018
MR06CBH	8.11	76.1	46.022	19.921	94.119	6.228	349	22.453	23.791	573	9.371	0.173	0.016	0.067	0.010	0.013	0.011	0.018
MR07BH	8.05	73.7	61.678	19.884	66.545	1.735	244	59.300	61.300	518	6.842	0.130	0.017	0.067	0.012	0.098	0.062	0.148

Table C-3 Chemical tracer data for Section 2 (October 2012). All values expressed in mg/l, except EC expressed in mS/m and pH expressed in pH units.

Location	pH	EC	Ca	Mg	Na	K	MAlk	F	Cl	Br	SO4	TDS	Al	Fe	Mn	Si	Ba	Cu	Se	Zn
MR11	8.09	50.4	32.17	17.39	42.009	8.40	190	0.209	39.37	0.202	17.70	350	0.010	0.014	0.016	0.952	0.049	0.004	0.010	0.014
MR12	7.85	69.0	41.87	24.69	68.489	9.59	225	0.220	62.10	0.267	37.12	479	0.008	0.017	0.010	3.272	0.034	0.004	0.011	0.018
MR13	7.56	70.9	43.74	21.24	67.844	11.4	212	0.204	63.90	0.204	43.07	507	0.047	0.035	0.012	5.378	0.027	0.006	0.012	0.017
MR12BH	8.27	280	101.5	169.1	317.811	1.17	678	0.244	508.28	1.470	226.80	2054	0.008	0.011	0.011	20.672	0.033	0.006	0.020	0.015

Table C-4 Chemical tracer data for Section 2 (January 2013). All values expressed in mg/l, except EC expressed in mS/m and pH expressed in pH units.

Location	pH	EC	Ca	Mg	Na	K	MAlk	F	Cl	SO4	TDS	Si	B	Al	Ba	Cu	Fe	Mn	Zn
MR11ave	7.56	20.4	15.953	6.546	14.418	4.896	88	0.274	9.345	7.322	148	5.701	0.083	0.128	0.048	0.013	0.185	0.016	0.018
MR12	7.57	24.8	20.029	9.539	16.141	4.830	104	0.228	12.197	11.608	180	7.014	0.076	0.104	0.052	0.011	0.121	0.016	0.016
MR12BH	7.83	301.0	60.765	138.893	375.110	0.997	629	0.099	535.527	235.319	2032	21.444	0.174	0.007	0.034	0.010	0.013	0.016	0.012
MR13	7.97	41.8	28.743	13.177	33.050	6.472	139	0.165	33.134	23.350	282	4.987	0.090	0.104	0.060	0.016	0.108	0.019	0.020

Table C-5 Chemical tracer data for Section 3 (October 2012). All values expressed in mg/l, except EC expressed in mS/m and pH expressed in pH units.

Location	pH	EC	Ca	Mg	Na	K	MAlk	F	Cl	Si
MR18-20	8.31	67.67	40.09	22.65	65.38	9.87	208.33	0.29	69.50	0.14
C5H039	7.81	64.6	48.124	22.033	56.291	8.359	174	0.445	54.817	0.607
MR21ave	8.17	102.6	44.519	49.704	120.431	6.347	404	0.488	67.680	18.707

Table C-6 Chemical tracer data for Section 3 (January 2013). All values expressed in mg/l, except EC expressed in mS/m and pH expressed in pH units.

Location	pH	EC	Ca	Mg	Na	K	MAIk	F	Cl	SO4	TDS	Si	B	Al	Ba	Cu	Fe	Mn	Zn
MR15BH	8.46	412	81.976	0.667	692.3	9.712	17	5.640	1297	1.621	2104	8.416	2.159	0.045	0.216	0.040	0.033	0.025	0.039
MR16-17	7.63	24.5	17.384	7.101	18.79	4.902	91	0.239	18.08	10.94	168	4.886	0.096	0.125	0.050	0.013	0.158	0.017	0.019
MR18-20	8.24	38.9	22.468	12.77	34.34	7.241	128	0.243	35.18	21.01	260	0.774	0.085	0.112	0.070	0.014	0.138	0.021	0.020
MR21BHave	7.86	106	45.956	46.90	115.1	5.913	409	0.264	64.58	41.52	756	19.095	0.181	0.020	0.149	0.009	0.044	0.024	0.019

Table C-7 Water Quality data for the continuous, gauged model run for the middle Modder River (MR01-MR20). All values expressed in mg/l, except EC expressed in mS/m and pH expressed in pH units.

Location	pH	EC	Ca	Mg	Na	K	MAIk	Cl	SO4	TDS	Si	B	Al	Ba	Cu	Fe	Mn	Zn
Rust Dam	7.92	37.9	22.011	8.689	16.530	5.486	122	8.872	10.518	195	2.049	0.158	0.145	0.067	0.013	0.200	0.018	0.018
MR01	7.58	30.2	23.293	9.661	18.470	5.667	137	8.804	8.806	212	1.122	0.151	0.128	0.062	0.013	0.111	0.019	0.018
MR02	7.43	36.1	29.683	13.150	19.658	4.963	163	12.952	9.468	254	5.284	0.142	0.060	0.069	0.012	0.065	0.016	0.019
MR02BH	7.42	50.9	47.263	21.585	23.878	0.871	243	9.788	20.590	371	13.018	0.133	0.012	0.014	0.012	0.026	0.014	0.111
MR03	7.42	31.9	23.283	10.156	22.393	5.799	118	21.022	12.639	223	5.524	0.126	0.101	0.063	0.019	0.084	0.018	0.021
MR03BH	7.68	49.1	32.714	9.738	60.622	0.779	223	8.640	25.801	364	8.856	0.512	0.019	0.007	0.012	0.025	0.066	0.610
MR06ave	7.64	30.2	19.290	8.681	24.329	5.575	105	20.784	13.539	205	5.273	0.120	0.104	0.052	0.017	0.088	0.022	0.018
MR06B	7.73	47.2	30.701	14.095	41.556	6.259	160	43.657	18.251	316	5.051	0.120	0.102	0.094	0.013	0.090	0.025	0.018
MR06CBH	8.11	76.1	46.022	19.921	94.119	6.228	349	22.453	23.791	573	9.371	0.173	0.016	0.067	0.010	0.013	0.011	0.018
MR07Bridge	7.69	34.3	22.179	9.765	29.240	5.431	121	25.425	15.655	235	5.297	0.106	0.116	0.055	0.016	0.091	0.074	0.023
MR07BH	8.05	73.7	61.678	19.884	66.545	1.735	244	59.300	61.300	518	6.842	0.130	0.017	0.067	0.012	0.098	0.062	0.148
MR08	7.57	24.3	17.078	7.514	16.836	5.070	99	11.811	8.662	167	5.366	0.092	0.151	0.049	0.012	0.191	0.019	0.020
MR10	7.44	20.7	16.378	7.023	15.656	5.115	91	9.841	7.800	154	5.550	0.087	0.111	0.048	0.013	0.151	0.019	0.016
Koranaspruit	7.82	21.9	18.255	7.741	14.828	3.558	104	5.540	6.072	160	6.178	0.078	0.204	0.054	0.013	0.291	0.023	0.017
MR11ave	7.56	20.4	15.953	6.546	14.418	4.896	88	9.345	7.322	148	5.701	0.083	0.128	0.048	0.013	0.185	0.016	0.018
MR12	7.57	24.8	20.029	9.539	16.141	4.830	104	12.197	11.608	180	7.014	0.076	0.104	0.052	0.011	0.121	0.016	0.016

MR12BH	7.83	301.0	60.765	138.893	375.110	0.997	629	535.527	235.319	2032	21.444	0.174	0.007	0.034	0.010	0.013	0.016	0.012
MR13	7.97	41.8	28.743	13.177	33.050	6.472	139	33.134	23.350	282	4.987	0.090	0.104	0.060	0.016	0.108	0.019	0.020
MR14	7.96	40.7	28.775	13.020	30.280	5.883	145	30.207	21.648	277	6.208	0.084	0.156	0.083	0.014	0.210	0.022	0.018
MR14BH	7.95	87.7	33.746	25.709	121.136	4.913	316	53.245	36.364	647	17.687	0.443	0.018	0.097	0.015	0.014	0.012	0.026
MR15	7.90	34.3	22.787	10.085	27.134	5.454	113	27.572	16.324	224	5.317	0.087	0.153	0.068	0.014	0.263	0.031	0.022
MR15BH	8.46	412.0	81.976	0.667	692.357	9.712	17	1297.000	1.621	2104	8.416	2.159	0.045	0.216	0.040	0.033	0.025	0.039
MR16-17	7.63	24.5	17.384	7.101	18.793	4.902	91	18.089	10.945	168	4.886	0.096	0.125	0.050	0.013	0.158	0.017	0.019
MR18-20	8.24	38.9	22.468	12.779	34.342	7.241	128	35.185	21.016	260	0.774	0.085	0.112	0.070	0.014	0.138	0.021	0.020
MR21BH1	7.85	108.0	48.055	47.889	112.505	5.706	414	66.224	42.319	768	19.688	0.183	0.024	0.154	0.008	0.048	0.019	0.019
MR21BH2	7.81	112.0	49.228	51.111	121.393	6.125	433	65.898	48.213	801	19.656	0.188	0.020	0.186	0.009	0.020	0.019	0.022
MR21BH3	7.82	103.0	43.626	45.336	113.723	5.869	389	65.658	39.084	734	19.684	0.180	0.024	0.128	0.009	0.024	0.020	0.019
MR21BH4	7.96	101.0	43.539	43.797	105.653	5.778	386	64.824	36.386	712	16.889	0.168	0.018	0.130	0.010	0.113	0.048	0.018
MR21BH5	7.84	106.0	45.332	46.381	122.339	6.086	421	60.323	41.611	765	19.557	0.187	0.015	0.146	0.009	0.016	0.015	0.018
MR21BHave	7.86	106.0	45.956	46.903	115.122	5.913	409	64.585	41.523	756	19.095	0.181	0.020	0.149	0.009	0.044	0.024	0.019
MR21BHP	8.39	179.0	37.749	63.382	235.315	20.374	506	285.182	8.917	1159	3.451	0.307	0.021	0.098	0.010	0.030	0.016	0.012
MR Weir	7.60	27.8	20.171	8.446	21.443	5.657	101	20.248	13.163	192	4.057	0.082	0.152	0.053	0.013	0.281	0.022	0.017
Renoster Weir	8.28	50.8	32.119	13.570	44.529	8.722	150	48.694	31.898	340	3.437	0.095	0.041	0.052	0.014	0.034	0.018	0.017

Table C-8 Chemical mass balance percentage error for each tracer in each section model run.

Chemical Mass Balance Errors (%)													
Tracer	Section 1			Section 2						Section 3			
	Jan '13			Oct '12			Jan '13			Oct '12			
	Scenario 1	Scenario 2	Scenario 2	Scenario 1	Scenario 2	Scenario 2	Scenario 1	Scenario 2	Scenario 2	Scenario 1	Scenario 2	Scenario 1	Scenario 2
pH	-20.10%	-22.80%		-5.80%	4.40%		-16.90%	-2.40%		-4.20%	-2.80%	-78.10%	-48.20%
EC	9.70%	3.40%		-11.20%	-9.80%		-15.50%	-8.10%		-4.30%	-3.00%	-64.60%	-46.40%
Ca	27.40%	22.00%		-22.50%	-18.10%		-41.70%	-33.20%		-24.70%	-23.60%	-75.90%	-51.50%
Mg	26.40%	21.20%		30.00%	26.00%		6.10%	13.70%		-4.30%	-3.20%	-88.00%	-69.90%
Na	-3.70%	-13.00%		-12.50%	-13.40%		6.20%	12.90%		7.10%	8.40%	-46.60%	-33.10%
K	-21.30%	-24.80%		-40.50%	-32.50%		-37.60%	-26.00%		5.30%	7.00%	-82.70%	-61.00%
MAIk	25.20%	21.10%		-2.50%	2.20%		-20.40%	-10.70%		10.60%	12.00%	-84.50%	-61.40%
F				-8.80%	0.60%		38.50%	63.90%		-39.30%	-38.50%	-31.10%	-4.00%
Cl	-0.30%	-13.70%		13.80%	7.60%		23.70%	28.10%		14.20%	15.90%	-12.90%	-4.00%
B	-9.40%	-10.30%										-23.10%	8.50%
Br				37.50%	36.60%								
SO4	-19.10%	-28.00%		-24.40%	-28.50%		-11.70%	-6.80%				-88.70%	-71.80%
TDS	10.70%	4.80%		-12.10%	-11.10%		-12.60%	-4.50%				-69.40%	-50.10%
Al	-41.00%	-44.90%		-81.30%	-79.20%		0.90%	19.90%				-75.20%	-39.00%
Fe	-31.50%	-35.60%		-66.40%	-62.50%		39.70%	65.80%				-75.00%	-37.80%
Mn	-23.80%	-29.50%		19.40%	33.40%		-25.70%	-12.70%				-80.70%	-55.30%
Si	-20.10%	-22.30%		-54.60%	-59.00%		19.60%	37.20%		-2.30%	-8.90%	57.80%	260.80%
Ba	24.00%	17.20%		52.70%	70.70%		-32.50%	-20.40%				-78.20%	-55.50%
Cu	-41.60%	-43.70%		-37.80%	-31.80%		-29.90%	-17.30%				-73.90%	-43.70%
Zn	-3.20%	-3.80%		-25.60%	-17.70%		-20.30%	-6.00%				-75.20%	-44.10%
Se				-15.10%	-8.10%								
Average	-6.2%	-11.3%		-13.4%	-9.5%		-7.2%	5.2%		-4.2%	-3.7%	-60.3%	-26.7%

Appendix D

Table D-1 MCM input data for the A42 area in 2005. All values expressed in mg/l, except EC expressed in mS/m and pH expressed in pH units.

Location	Ca	Cl	EC	F	K	Mg	Na	NO3	pH	PO4	SO4	TAL
A4H2Cell	5.717	6.412	7.95	0.123	1.001	2.565	4.987	0.123	7.41	0.013	2.000	22.81
o-A4H2 Cell	5.717	6.412	7.95	0.123	1.001	2.565	4.987	0.123	7.41	0.013	2.000	22.81
A4H002c	6.923	13.770	10.57	0.050	0.940	3.773	7.409	0.040	7.74	0.018	2.000	20.43
B89783	17.569	16.267	40.50	0.438	1.041	6.327	60.595	2.739	8.12	0.012	9.788	154.34
B89784	21.489	5.632	26.00	0.142	1.336	12.297	9.740	0.212	8.47	0.014	2.000	110.96

Table D-2 The individual tracer chemical mass balance percentage error for the MCM run for A42 area.

Tracer	Mass balance error (%)					
	Outflow (Jan '05)		Outflow (Sum 2005)		Outflow (Sum 2006)	
	Scenario 1	Scenario 2	Scenario 1	Scenario 2	Scenario 1	Scenario 2
Ca	-6.40%	-7.30%	-6.40%	-7.30%	-6.40%	-7.30%
Cl	24.20%	40.60%	24.20%	40.60%	24.20%	40.60%
EC	2.40%	3.80%	2.40%	3.80%	2.40%	3.80%
F	-62.40%	-62.70%	-62.40%	-62.70%	-62.40%	-62.70%
K	-40.30%	-36.00%	-40.30%	-36.00%	-40.30%	-36.00%
Mg	11.70%	9.90%	11.70%	9.90%	11.70%	9.90%
Na	21.70%	27.80%	21.70%	27.80%	21.70%	27.80%
NO3	-13.20%	-17.90%	-13.20%	-17.90%	-13.20%	-17.90%
pH	-35.70%	-30.00%	-35.70%	-30.00%	-35.70%	-30.00%
PO4	-18.80%	-9.80%	-18.80%	-9.80%	-18.80%	-9.80%
SO4	-28.70%	-23.30%	-28.70%	-23.30%	-28.70%	-23.30%
TAL	-6.40%	-14.00%	-6.40%	-14.00%	-6.40%	-14.00%
Average	-12.7%	-9.9%	-12.7%	-9.9%	-12.7%	-9.9%

Appendix E

Table E-1 Water quality data used in the MCM application for the quaternary catchment D73F. All values expressed in mg/l, except EC expressed in mS/m and pH expressed in pH units.

Location	Ca	Cl	DMS	EC	F	K	Mg	Na	NH4	NO3_NO2	pH	PO4	Si	SO4	TAL
CellD7H3	22.500	17.900	195	26.5	0.180	1.800	9.000	17.000	0.080	0.300	7.85	0.024	8.040	16.200	89.3
InflowD7H5	21.500	20.500	177	23.9	0.250	1.910	8.600	16.000	0.060	0.580	8.00	0.023	7.780	17.900	72.0
B166744	66.800	77.100	795	99.5	3.740	1.050	25.600	128.000	0.020	0.020	7.73	0.015	23.370	121.700	304.2
167032	103.100	409.100	1854	255.0	6.600	16.000	40.400	421.700	0.040	0.020	7.55	0.016	15.240	313.500	445.9
B167031	109.100	280.000	1279	181.0	4.910	1.150	52.700	207.600	0.040	2.390	7.68	0.011	19.430	218.300	323.2
B166928	192.600	506.300	2013	282.2	4.410	4.630	77.700	339.550	0.040	6.090	7.70	0.028	17.760	410.150	369.8
B166929	199.800	500.000	2031	288.0	4.310	10.760	80.200	346.100	0.450	5.600	7.60	0.033	16.190	397.300	383.0
B166930	91.800	182.700	733	114.0	3.170	3.800	37.700	78.300	0.640	4.060	7.10	0.016	11.590	137.900	146.7
B166931	143.200	407.400	1874	252.8	5.080	3.810	57.700	362.000	0.080	4.960	7.80	0.031	16.690	414.400	375.8
B166932	109.800	161.300	1216	166.0	5.970	2.030	46.200	178.800	0.360	4.960	7.70	0.046	17.970	187.600	411.6
B166933	97.600	122.000	1020	135.0	6.280	4.290	39.900	139.300	0.070	3.380	7.60	0.041	22.930	106.900	400.4
B166934	96.200	33.200	892	109.0	6.410	3.690	47.700	82.300	0.070	1.550	7.40	0.073	27.630	202.600	338.6
B166935	118.100	192.400	1247	174.0	5.550	1.700	43.300	194.900	0.060	2.340	7.70	0.037	21.430	198.600	395.6
B166990	25.900	27.500	827	102.4	12.040	10.600	7.200	192.300	0.020	11.840	7.87	0.063	30.240	57.400	362.2
B167048	132.975	281.525	1387	184.8	4.745	1.220	53.250	215.350	0.088	2.405	7.85	0.027	17.985	221.725	381.7
B167049	183.050	503.300	1960	273.0	4.315	5.785	75.500	335.000	0.085	7.090	7.90	0.025	17.625	381.500	360.4

Table E-2 The individual tracer chemical mass balance percentage error for the MCM run for quaternary catchment D73F.

Tracer	Chemical mass balance error (%)					
	No abstraction volume			Abstraction volume		
	Scenario 1	Scenario 2	Scenario 1	Scenario 2	Scenario 1	Scenario 2
Ca	-19.90%	-11.30%	-19.90%	-11.60%	-19.90%	-11.60%
Cl	-1.90%	6.30%	-1.90%	5.90%	-1.90%	5.90%
DMS	-23.30%	-15.80%	-23.30%	-16.10%	-23.30%	-16.10%
EC	-23.80%	-16.30%	-23.80%	-16.60%	-23.80%	-16.60%
F	20.00%	28.90%	20.00%	28.40%	20.00%	28.40%
K	-10.60%	-1.50%	-10.60%	-1.90%	-10.60%	-1.90%
Mg	-19.90%	-11.30%	-19.90%	-11.60%	-19.90%	-11.60%
Na	-18.70%	-12.70%	-18.70%	-13.00%	-18.70%	-13.00%
NH4	-37.50%	-30.40%	-37.50%	-30.70%	-37.50%	-30.70%
NO3_NO2	60.90%	79.40%	60.90%	78.80%	60.90%	78.80%
pH	-15.10%	-5.40%	-15.10%	-5.80%	-15.10%	-5.80%
PO4	-20.10%	-11.10%	-20.10%	-11.40%	-20.10%	-11.40%
Si	-19.20%	-10.20%	-19.20%	-10.50%	-19.20%	-10.50%
SO4	-5.70%	2.50%	-5.70%	2.20%	-5.70%	2.20%
TAL	-32.30%	-25.20%	-32.30%	-25.50%	-32.30%	-25.50%
Average	-11.1%	-2.3%	-11.1%	-2.6%	-11.1%	-2.6%