

Groundwater recharge modelling in a large scale basin: an example using the SWAT hydrologic model

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Abstract Groundwater constitutes an essential resource that augments surface water resources in meeting the water supply needs of man and the ecosystem. Most importantly in arid and semi-arid environments where rainfall patterns are erratic, groundwater resources are often the preferred source of water. This causes enormous pressure on the resource leading to diminishing groundwater resources. Land use changes also impact on groundwater resources through alterations in the hydrologic regime. It is imperative therefore to evaluate groundwater recharge dynamics under changing land uses to provide for a better resource planning and allocation. We present in this study, an investigation into groundwater recharge dynamics of the Olifants Basin, a water stressed basin in Southern Africa over the past decade with considerations to land use changes. Three land use change scenarios were developed to simulate the groundwater recharge of the basin within the Soil and Water Assessment Tool (SWAT) environment. The SWAT model was calibrated (1988–2001) and validated (2002–2013) with good model performance statistics; NSE, R^2 , PBIAS, RSR of 0.88, 0.89, –11.49%, 0.34 and 0.67, 0.78, –20.69%, 0.57 respectively for calibration and validation stages. Findings indicate groundwater recharge declined by 10.37 mm (30.3%) and 2.34 mm (9.82%) during the periods 2000–2007 and 2007–2013 respectively. The decline

in groundwater recharge was linked to the changes in urban (9.2%), agriculture (6.1%), rangelands (–16.8%) during the period 2000–2007 and urban (1.3%), agricultural (14%), rangelands (–14.8%) during 2007–2013. The SWAT model reveals its capabilities as a decision support tool (DST) in groundwater recharge assessment for large scale basins.

Keywords Groundwater recharge · Land use · Olifants Basin · SWAT

Introduction

In arid and semi-arid regions of the world, groundwater serves as an essential alternative to surface water resources for water supply purposes. It plays a significant role in meeting the water demands of man and the ecosystem and is perceived as the panacea to the looming water scarcity scare (Robins and Fergusson 2014). This is reflective on its dependency for the supply of 43% of irrigation water, 36% of potable water and 24% of industrial water globally (Doll et al. 2012). At the current rate of abstraction, the sustainability of groundwater resources is questioned on the basis of its overexploitation (Calow and MacDonald 2009) which is further worsened by land use/land cover (LULC) dynamics (GWP 2014) coupled with the on-going climate change phenomenon. Land use /land cover changes (LULCCs) have widely been acknowledged to alter the hydrologic regime with marked repercussions on the quantity of overland flow and indirectly affecting the quantum of groundwater recharge (Nie et al. 2011; Yan et al. 2013; Baker and Miller 2013; Wang et al. 2008). LULCCs are reported to have far reaching implications on the hydrologic cycle compared to the effects of climate change (Vorosmarty et al. 2004). Increasing population is identified as a major driver to LULCCs causing

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a shift in natural vegetation towards more productive uses of land. This has triggered the conversion of the natural cover to arable lands with the focus of expanding the frontiers of dryland and irrigated agriculture in order to meet the ever increasing food demand (Foley et al. 2005; Godfray et al. 2010). The conversion of natural vegetation to agriculture results in the modification of key vegetation parameters that influences recharge (Scalon et al. 2005) and this has the tendency to irreversibly alter aquifer characteristics with replicative effects on groundwater availability (GWP 2014).

Although there exist substantial evidence of LULCC impacts on the hydrologic cycle, most of these studies have focused on the atmospheric component of the hydrologic cycle leaving much to be desired on subsurface components of the hydrologic cycle and more in particular on groundwater resources (Scanlon et al. 2005). In purview of this limitation, the impacts of LULCCs on groundwater resources need to be investigated with particular emphasis on groundwater recharge. Groundwater recharge defined as the portion of rainfall that reaches the saturated zone, either by direct contact in the riparian zone or by downward percolation through the unsaturated zone (Adams et al. 2004) is a vital part of groundwater system that needs to be monitored to provide information of recharge dynamics with oriented focus on long term sustainability strategies for the management of groundwater resources.

The foregone discussions are not farfetched in the case of South Africa. In South Africa, the reliance on groundwater for agricultural, industrial and household water supply cannot be overemphasized (de Lange et al. 2003; DWA 2011). Perhaps, in many rural parts of South Africa groundwater remains the only reliable source of water supply (Aston 2000). This is particularly the case due to the semi-arid nature of the country predisposing it to erratic rainfall patterns with high inter-annual variations which tend to affect surface water availability. This has caused over dependency on groundwater resources resulting in their overexploitation. In the midst of this quagmire of overexploitation is also the incidence of LULCCs further altering the hydrologic regime and subsequently the recharge process (Lerner and Harris 2009). Awakening to the call for sustainable management of water resources is the need for sustainable strategies to be devised not only for surface water resources but also for the inimitable groundwater resources. A critical approach in ensuring groundwater sustainability in the midst of changing land uses is to understand how LULCCs impact on groundwater recharge in order to provide the requisite knowledge to inform policy direction.

In this paper, we investigate the impacts of LULCC patterns on groundwater recharge through a modelling approach with a distributed hydrologic model. The objective of the study was to investigate the feasibility of using a physically based distributed model to predict the changes that occur in

groundwater recharge as a result of LULCCs and to quantify these changes. The approach is a simplistic way of cost effectively assessing groundwater recharge using readily available sources of information.

Materials and methods

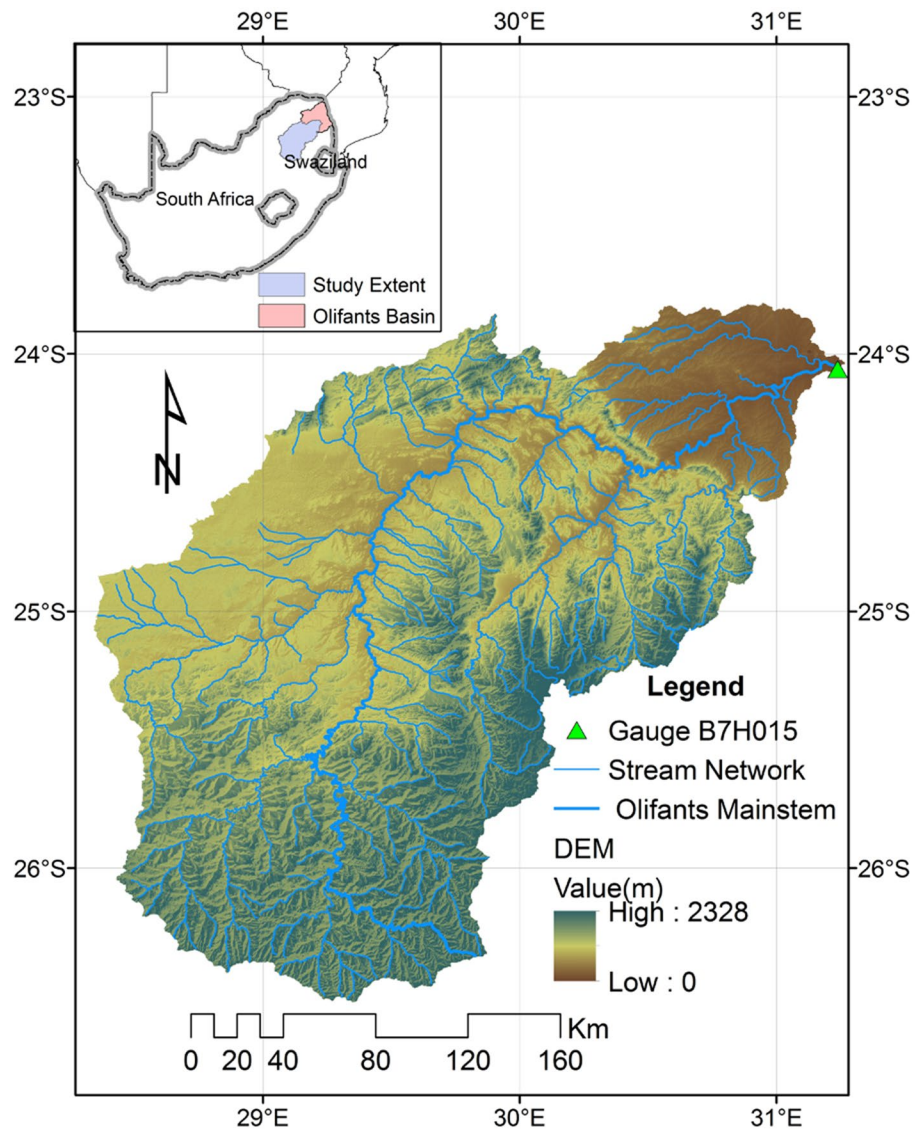
Description of study area and extent

The Olifants River Basin is located in the northeastern part of South Africa with a total drainage surface area of 74,000 km² (Fig. 1). With a main stem of 770 km, the Olifants River flows from Trichardt to the east of Johannesburg in the province of Gauteng and then flows in northeasterly direction through the provinces of Mpumalanga and Limpopo crossing the Mozambique border where it finally empties into the Massingir dam. Geographically, the basin lies on longitudes 28.3°E–31.9°E and latitudes 22.6°S–26.5°S. For the purposes of this study, the Olifants Basin is herein referred to as the area extending from the upper Olifants to the location of gauge B7H015 (Fig. 1). The selection of the study area extent was solely informed by data availability on existing gauge stations that were required to calibrate and validate the model. The Olifants River is drained by some major tributaries; on the right bank are Klein Olifant, Steelpoort and Blyde rivers with Wilge, Moses, Elands, Ga-Selati and Letaba on the left bank. Generally, the elevation of the basin ranges from 0 to 2328 m above mean sea level (masl). Erratic rainfall characterizes the basin and occurs in the months of October to April with noticeable variations in both space and time (Gyamfi et al. 2016a; McCartney and Arranz 2007). Mean annual precipitation (MAP) is estimated to be 664 mm with rainfall peaks in January (Gyamfi et al. 2016a). Temperatures range from 18–34 °C in summer and 5–26 °C in winter. Chromic vertisols, orthic acrisols, cambic arenosols, chromic luvisols and rhodic ferralsols (FAO 2005) are the main soil types in the basin. The population of the basin is estimated to be slightly over 5 million with a greater proportion being rural populace (DWA 2002; STATS SA 2011).

Hydrological setting and groundwater occurrence

The basin is characterized by four types of aquifers namely; weathered rock aquifer, fractured (structural) aquifer, dolomitic (karst) and the alluvial aquifers. Groundwater in the basin is mostly exploited from the dolomitic and weathered aquifer systems (Aston 2000; DWA 2011). The weathered aquifer has depth ranges of 5–12 m (Hodgson and Krantz 1998). Groundwater yields from the weathered aquifer are low with approximately 1 l s⁻¹. Groundwater in fractured aquifers normally occurs in crevices. Fractured aquifers are

Fig. 1 Location and extent of study area showing gauge station



encountered some few meters from the earth surface to a depth of about 30 m (Hodgson and Krantz 1998). At depths below 30 m, the crevices tend to close up due to the exertion of weight from the overlying formations. Yields in fractured aquifers are highly variable with high initial yields but tend to decline as a result of continuous abstraction.

Dolomitic aquifers in the Olifants Basin are mainly located in the western foothills of Drakensberg Mountains, Delmas and Marble Hall with yields ranging between 5 and 40 l s⁻¹ (Aston 2000). Dolomitic aquifers have the highest yields. Similar to dolomitic aquifers, alluvial aquifers have high yields and are located along watercourses with historic floodplains (Aston 2000). However for management purposes the Olifants Basin has been classified into three aquifer regions (Parsons and Conrad 1998) to include major, minor and poor regions (Fig. 2). The major aquifer regions are associated with high yielding aquifer systems with good

water quality whiles the minor aquifer regions are noted for moderately yielding aquifer systems. The poor regions have aquifers with low to negligible yielding aquifers.

Modelling approach

Model selection

The assessment of LULCC impacts on groundwater recharge was carried out within the Soil and Water Assessment Tool (SWAT) environment. SWAT was developed through a joint effort by United States Department of Agriculture–Agricultural Research Services (USDA–ARS) and Agricultural Experiment Station in Temple, Texas. As a physically based distributed model, SWAT demonstrate capabilities for a continuous and long-term simulation of complex watershed

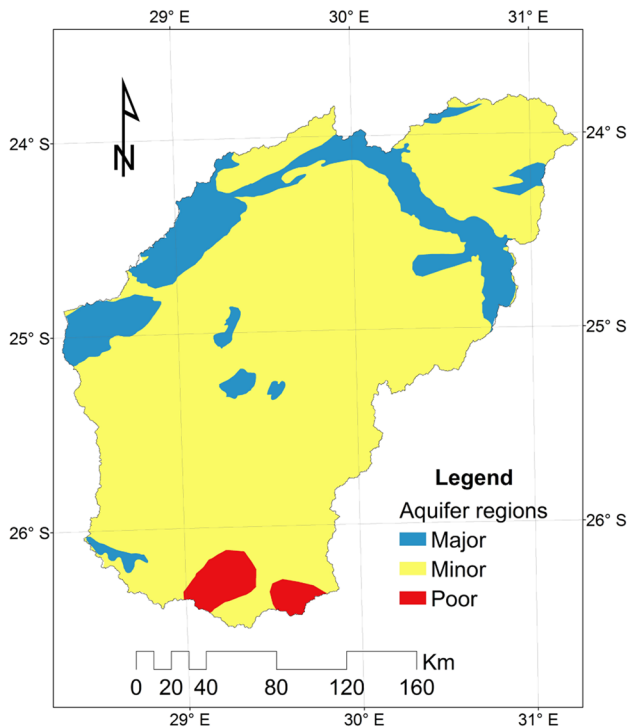


Fig. 2 Aquifer regions in the Olifants Basin

processes such as sources of agricultural pollutants, impacts of land uses on water resources and sediment generation patterns (Arnold and Fohrer 2005; Arnold et al. 1998). Due to the model's versatility, it has been employed by many in diverse areas of land and water resources studies (Yesuf et al. 2015; Cai et al. 2012; Githui et al. 2009; Ghaffari et al. 2010). A comparison of SWAT with other hydrologic models revealed a higher success rate in SWAT (Borah and Bera 2003; Van Liew and Garbrecht 2003; Srinivasan et al. 2005). SWAT operates at a functional unit referred to as the hydrologic response unit (HRUs). HRUs are homogenous combination of areas for land use, soil characteristics and management practices. Watershed processes in SWAT are firstly simulated and aggregated at the HRU level and further transmitted to respective subbasins. The model simulates the major components of the hydrologic cycle (surface runoff, evapotranspiration, percolation, lateral flow, return flow, transmission losses and ponds) base on the water balance equation represented as (Arnold et al. 1998);

$$SW_t = SW_o + \sum_{i=1}^t (R_{day} - Q_{surf} - E_a - W_{seep} - Q_{gw}), \quad (1)$$

where SW_t is final soil water content (mm), SW_o is initial soil water content in day i (mm), t is time in days, R_{day} is amount of precipitation in day i (mm), Q_{surf} is amount of surface runoff in day i (mm), E_a is amount of evapotranspiration in day i (mm), W_{seep} is amount of water entering the

vadose zone from the soil profile in day i (mm) and Q_{gw} is amount of return flow in day i (mm).

Surface runoff and evapotranspiration estimation

Surface runoff (Q_{surf}) which refers to overland flow of excess water after infiltration and depression storages are fulfilled was estimated using a modification of the SCS-CN method (SCS 1972). The SCS-CN method is a function of antecedent moisture conditions, infiltration, soil type, land cover and other basin characteristics such as topography. The SCS-CN method as used in this study is defined as (SCS 1972);

$$Q_{surf} = \begin{cases} (R_{day} - 0.2S)^2 \\ (R_{day} + 0.8S) \end{cases}, \quad R_{day} > 0.2S, \quad (2)$$

where Q_{surf} is rainfall excess (mm), R_{day} is the rainfall depth for the day (mm), S is the retention parameter (mm).

The retention parameter S is influenced by the changes that occur in land uses, soil water content and slopes and as result varies spatially across a watershed. The retention parameter was estimated as;

$$S = 25.4 \left(\frac{1000}{CN} - 10 \right), \quad (3)$$

where S is retention parameter (mm) and CN is the curve number. CN is a function of soil permeability, antecedent soil conditions and land use. CN can be read from tables available in the literature by combining soil type and land use of a particular watershed.

Evapotranspiration which refers to water losses through evaporation and transpiration were accounted for using the Penman–Monteith method given as;

$$ET = \frac{0.408\Delta(R_n - G_o) + \gamma \frac{900}{T+273} u_2 (e_s - e_a)}{\Delta + \gamma(1 + 0.34u_2)}, \quad (4)$$

where ET is the reference evapotranspiration (mm d^{-1}), Δ is the slope of the saturation vapour pressure temperature curve ($\text{kPa } ^\circ\text{C}^{-1}$), R_n is the net radiation ($\text{MJ m}^{-2} \text{d}^{-1}$), G_o is the soil heat flux density ($\text{MJ m}^{-2} \text{d}^{-1}$), e_s is the saturation vapour pressure (kPa), e_a is the actual vapour pressure (kPa), γ is the psychrometric constant ($\text{kPa } ^\circ\text{C}^{-1}$), $e_s - e_a$ is saturation vapour pressure deficit (kPa), u_2 is wind speed (ms^{-1}), mean daily temperature ($^\circ\text{C}$).

Groundwater recharge estimation

Groundwater resources are replenished through the downward movement of water by percolation and further through

the vadose zone to recharge aquifers. The amount of recharge that occurs is dependent on the hydraulic properties of existing geologic formations in the vadose zone and the water table (Neitsch et al. 2009). In estimating the recharge, the exponential decay function proposed by Venetis (1969) was used. The exponential function is formulated as;

$$W_{rchrg,i} = \left\{ W_{seep} \cdot \left(1 - \exp \left[\frac{-1}{\delta_{gw}} \right] \right) + W_{rchrg,i-1} \cdot \exp \left[\frac{-1}{\delta_{gw}} \right] \right\}, \tag{5}$$

where $W_{rchrg,i}$ is the amount of recharge entering the aquifers on day i (mm), δ_{gw} is the delay time or drainage time of the overlying geologic formations (days), W_{seep} is the total amount of water exiting the bottom of the soil profile on day i (mm) and $W_{rchrg,i-1}$ is the amount of recharge entering the aquifers on day $i-1$ (mm).

Input datasets and sources

Required data for the model setup were digital elevation model (DEM), digital soil data, digital land use maps and

climatic datasets (Fig. 3). The DEM was acquired from the global land cover facility database (GLCF) and is of spatial resolution $90\text{ m} \times 90\text{ m}$ (3 arc sec). The DEM was used for basin discretization and extraction of geomorphologic characteristics such as width, depth, length of streams and slopes. Slopes discretization for the study area followed FAO classification scheme (FAO 2003) to include; level to gently undulating (< 8%), rolling to hilly (8–30%) and steeply dissected to mountainous (> 30%). Soil data and information on related soil properties were obtained from FAO soil map (FAO 2005). This data was augmented with information from field sampled soils. The extracted FAO soil data for the study area indicates five major soil types namely; chromic luvisols (Lc) (38.81%), cambic arenosols (Qc) (33.03%), chromic vertisols (Vc) (21.21%), orthic acrisols (Ao) (5.77%) and rhodic ferralsols (Fr) (1.18%).

LULC information for three epochs (2000, 2007 and 2013) was extracted from Landsat 7 ETM+ imagery through a supervised classification in Erdas Imagine 2014. The images of spatial resolution 30 m were downloaded from the USGS database (<https://glovis.usgs.gov/>) for Path/Row; 168/077,

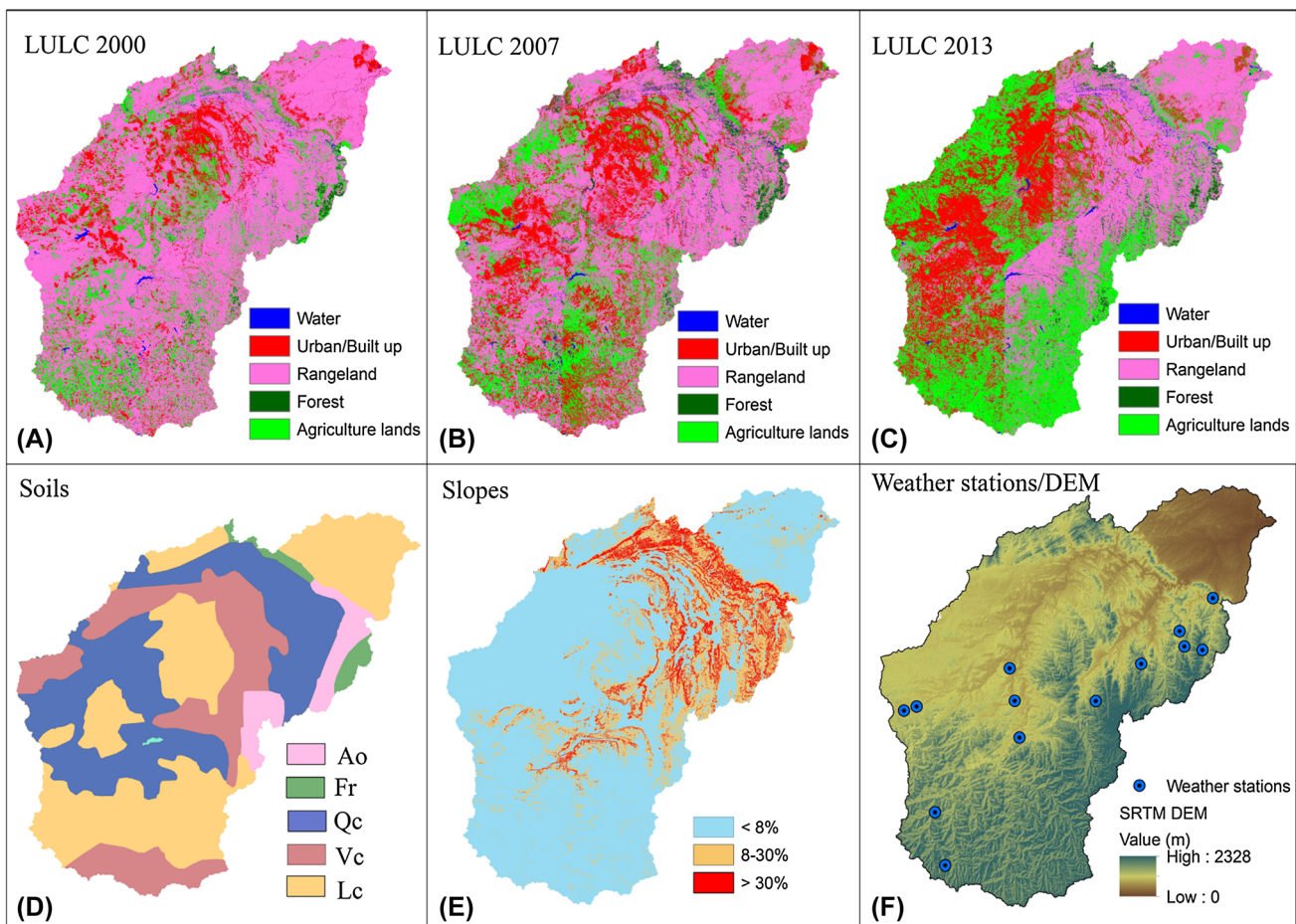


Fig. 3 Spatial model input parameters

169/077, 169/078, 170/077 and 170/078. Adopting the Anderson classification scheme (Anderson et al. 1976), five –level 1 land use classes were extracted from the images. Historical climatological data (1980–2013) consisting of maximum and minimum temperatures, daily rainfall and wind speed at 13 weather stations were sourced from the South African Weather Service (SAWS). Supplementary data from the climate forecast system reanalysis (CFSR) database augmented ground measured climatological data.

Calibration and validation analysis

Historical monthly discharge data for gauge station B7H015 was used for calibration (01/01/1988–01/12/2001) and validation (01/01/2002–01/12/2013). The model was warmed up for 8 years prior to 1988 to equilibrate the model for simulations. Gyamfi et al. (2016b) in their earlier work identified sensitive parameters to streamflow and these parameters are adopted in this study. Four objective functions were used to evaluate the performance of the model (Santhi et al. 2001; Moriasi et al. 2007; Oeurng et al. 2011).

- Coefficient of determination (R^2): R^2 is calculated as follows;

$$R^2 = \left[\frac{\sum_{i=1}^n (O_i - S_i)(S_i - \bar{S})}{\left(\sum_{i=1}^n (O_i - \bar{O})^2 \right)^{0.5} \left(\sum_{i=1}^n (S_i - \bar{S})^2 \right)^{0.5}} \right]^2. \quad (6)$$

- Nash–Sutcliffe (NSE): NSE is formulated as;

$$NSE = 1 - \frac{\sum_{i=1}^n (O_i - S_i)^2}{\sum_{i=1}^n (O_i - \bar{O})^2} \quad (7)$$

- RMSE—observations standard deviation ratio (RSR): RSR is calculated as;

$$RSR = \frac{RMSE}{STD_{obs}} = \frac{\sqrt{\sum_{i=1}^n (O_i - S_i)^2}}{\sqrt{\sum_{i=1}^n (O_i - \bar{O})^2}}. \quad (8)$$

- Percent Bias (PBIAS): PBIAS is calculated as shown;

$$PBIAS = \frac{\sum_{i=1}^n (O_i - S_i)}{\sum_{i=1}^n O_i} \times 100\%, \quad (9)$$

where O_i is observed variable, S_i is simulated variable, \bar{O} is mean of observed variable, \bar{S} is mean of simulated variable, n is number of observations under consideration, $RMSE$ is root mean square error, STD_{obs} is standard deviation of observed variable.

Model application and statistical analysis

The impacts of LULCCs on groundwater recharge of the Olifants Basin was assessed using the “fix-changing” method (Wang et al. 2009; Ghaffari et al. 2010; Tang et al. 2011; Yan et al. 2013; Lin et al. 2015). In this method, except varying land use data for the different time slice (2000, 2007 and 2013), all other spatial input parameters remain unchanged. Results from model simulations were then compared and analyzed. Findings were presented based on changes in annual recharge and corresponding changes in water balance ratios (WBRs). All analysis was done in SPSS 20.0.

Results and discussion

Land use change detection

Changes observed in LULC are shown in Fig. 4 for the period 2000–2013. All land use classes had undergone some degree of change. However, dominant changes occurred in urban areas, agricultural lands and rangelands. A continuous increment in urban and agricultural land covers was observed for all the three epochs. Between 2000 and 2007, urban land uses expanded from 13.2 to 22.4%. A gradual expansion in urban land uses was further observed in 2013 increasing from 22.4% in 2007 to 23.7%. Again, agricultural areas expanded from 15.2 to 21.3% for the period 2000 and 2007 respectively. Agricultural lands further increased from 21.3% in 2007 to 35.3% in 2013. Contrary to the continuous expansion in agriculture and urban areas, a continual decline was observed in rangeland, from 69.2 to 52.4% for 2000 to 2007. Rangelands had decreased from 52.4 to 37.6% by the end of 2013. The annual rate of change for forest, urban, agriculture and rangeland for 2000–2007 were 10.1%, 9.9%, 5.8% and –3.5% respectively. Similarly during 2007–2013, the annual rates of change were –2.8%, 0.9%, 10.9% and 4.7% for forest, urban, agriculture and rangelands respectively.

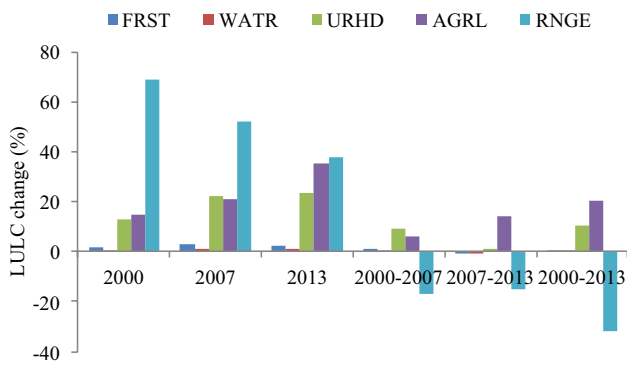


Fig. 4 Land use and land cover change for 2000–2013

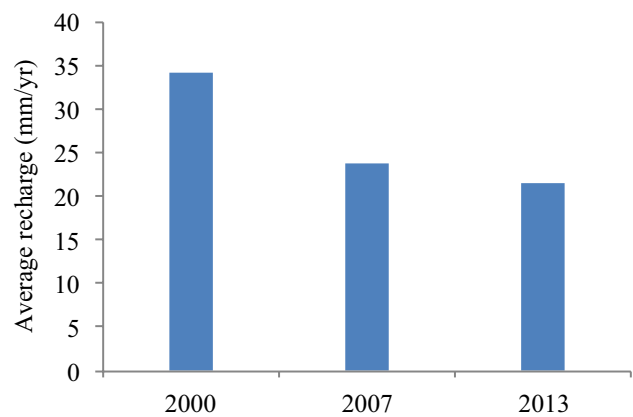


Fig. 6 Trend in groundwater recharge for 2000–2013

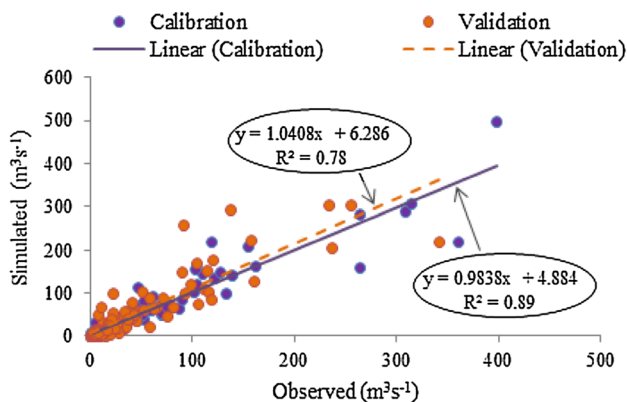


Fig. 5 Monthly simulated and observed discharge for calibration and validation periods

Table 1 Model performance evaluation statistics

Model stage	Objective function			
	NSE	R ²	PBIAS (%)	RSR
Calibration (1988–2001)	0.88	0.89	−11.49	0.34
Validation (2002–2013)	0.67	0.78	−20.69	0.57

Calibration and validation of model

Figure 5 compares observed and simulated streamflow for the calibration period (01/01/1988–01/12/2001) and the validation period (01/01/2002–01/12/2013). There is a match fit between the observed and simulated streamflow with NSE and R² values for both calibration and validation period greater than 0.6 (Table 1). The model overestimated observed streamflow by 11.49 and 20.69% for calibration and validation periods respectively. Those notwithstanding the PBIAS values are within acceptable limits (Moriassi et al. 2007).

Impact of LULCC on groundwater recharge

Noticeably, groundwater declined continuously during 2000–2013 (Fig. 6). From 2000 to 2007, the annual groundwater recharge decreased by 10.37 mm (30.3%) and the reduction was associated with LULCCs in urban (9.2%), agriculture (6.1%) and rangelands (−16.8%) for the same period. A further decline in groundwater recharge of 2.34 mm (9.8%) was observed during 2007–2013 with concomitant changes in urban (1.3%), agriculture (14%) and rangelands (−14.8%). The observation made in the Olifants Basin with respect to groundwater recharge concurs with findings in other jurisdictions (Tripathi et al. 2005; Ghaffari et al. 2010; Baker and Miller 2013). The declining trend seen in the average groundwater recharge is attributed to increases in impervious areas due to urban and agriculture expansion which causes less soil infiltration. Arguably, the over reliant on groundwater for irrigation, industrial, animal husbandry and household water uses could also account for the decline in groundwater recharge (de Lange et al. 2003; DWA 2011) resulting in the rate of abstraction exceeding that of recharge. The decline in groundwater resources may even worsen in semi-arid environments and particularly in the study region where groundwater is often the most preferred source of water due to its readily reliability and sustainability (Calow et al. 1997; Calow and MacDonald 2009). A 3–5% of basin-wide mean annual precipitation (Table 2) goes into groundwater recharge. This proportion in time past was estimated to be 3–6% (DWA 1991).

Conclusion

The effects of LULCCs on groundwater recharge were investigated in this study using a physically based distributed

Table 2 Simulated ratios of basin hydrology based on three LULC scenarios for the hydrologic year (October–September)

LULC scenario	Water balance ratios					
	B/TF	SR/TF	SF/P	PC/P	DR/P	ET/P
2000	0.26	0.74	0.07	0.1	0.05	0.78
2007	0.25	0.75	0.09	0.05	0.04	0.82
2013	0.24	0.76	0.09	0.08	0.03	0.80

SF/P streamflow/precipitation, *PC/P* percolation/precipitation, *DR/P* deep recharge/Precipitation, *ET/P* evapotranspiration/precipitation, *B/TF* baseflow/total flow, *SR/TF* surface runoff/total flow

hydrologic model. Results indicate that groundwater resources in the Olifants Basin are declining as a result of the continuous decline in recharge and also due to over-exploitation issues. The declines in recharge were associated with the changes in major land uses within the Olifants Basin. The feasibility of using the SWAT distributed model with readily available data has proven worthwhile in the investigation of groundwater resources in terms of its recharge rate. It is recommended that further groundwater investigations should couple hydrologic models with field monitored groundwater data to optimize the application of such models.

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