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# Groundwater recharge rates and surface runoff response to land use and land cover changes in semi-arid environments

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**Abstract**

The effects of land use and land cover (LULC) on groundwater recharge and surface runoff and how these are affected by LULC changes are of interest for sustainable water resources management. However, there is limited quantitative evidence on how changes to LULC in semi-arid tropical and subtropical regions affect the subsurface components of the hydrologic cycle, particularly groundwater recharge. Effective water resource management in these regions requires conclusive evidence and understanding of the effects of LULC changes on groundwater recharge and surface runoff. We reviewed a total of 27 studies (2 modeling and 25 experimental), which reported on pre- and post land use change groundwater recharge or surface runoff magnitude, and thus allowed to quantify the response of groundwater recharge rates and runoff to LULC.

Comparisons between initial and subsequent LULC indicate that forests have lower groundwater recharge rates and runoff than the other investigated land uses in semi-arid tropical/ subtropical regions. Restoration of bare land induces a decrease in groundwater recharge from 42% of precipitation to between 6 and 12% depending on the final LULC. If forests are cleared for rangelands, groundwater recharge increases by 7.8 ± 12.6%, while conversion to cropland or grassland results in increases of 3.4 ± 2.5 and 4.4 ± 3.3%, respectively.

Rehabilitation of bare land to cropland results in surface runoff reductions of between 5.2 and 7.3%. The conversion of forest vegetation to managed LULC shows an increase in surface runoff from 1 to 14.1% depending on the final LULC. Surface runoff was reduced from 2.5 to 1.1% when grassland is converted to forest vegetation.

While there is general consistency in the results from the selected case studies, we conclude that there are few experimental studies that have been conducted in tropical and subtropical semi-arid regions, despite that many people rely heavily on groundwater for their livelihoods. Therefore, there is an urgent need to increase the body of quantitative evidence given the pressure of growing human population and climate change on water resources in the region.

**Keywords:** Groundwater recharge rate, Infiltration, Surface runoff, Semi-arid, Land use change, Land cover change

## Review

### Introduction

Groundwater is a major source to meet urban, industrial, and particular agricultural water requirements, especially for tropical and sub-tropical semi-arid regions (Siebert et al. 2010). Understanding how groundwater recharge (the water addition from the unsaturated zone (vadose zone) into the saturated zone (phreatic zone)) is affected by land use and land cover (LULC) and respective changes of LULC is a prerequisite for land use planning that ensures sustainable water supply in semi-arid regions. Semi-arid regions, which are defined as having a ratio of precipitation to potential evapotranspiration (ET) ranging between 0.2 and 0.5 (UNESCO, 1979), are increasingly used for cropping due to the increased food demand of a growing human population (Santoni et al. 2010). Therefore, semi-arid regions are experiencing widespread conversion of natural vegetation to agricultural land as well as intensification of

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agricultural practices. In recent years, the savanna vegetation of semi-arid regions has been cleared at a large scale for livestock production or for crops that often depend on, particularly in South America (Santoni et al. 2010). At the process level, the effects of clearing native vegetation on the quantity and quality of water fluxes are influenced by changes in interception and evaporation from vegetation and changes to soil hydro-physical properties such as hydraulic conductivity, bulk density, or water holding capacity (Price et al. 2010).

There are many forces that drive LULC change in semi-arid regions, which involve social and biophysical drivers that are difficult to track and finally result in a complex and evolving system (Geist and Lambin 2004; De Waroux and Lambin 2012). These driving forces have been classified into four groups by De Waroux and Lambin (2012) as neo-Malthusian, climatic, economic, and eco-political. The relevance of any individual driving force for a given situation depends on the geographic and social environments under consideration.

The neo-Malthusian theory suggests that land use change is triggered by overuse of resources mainly attributed by an increase of human population. The overuse of resources includes activities such as overgrazing, fuelwood harvesting, or trade of forest products such as fruits and seeds (De Waroux and Lambin 2012; Odihi 2003; Salehi et al. 2008). For climatic drivers, it is argued that climate change, especially longer and more frequent droughts, is leading to vegetation changes as, e.g., manifested by desertification. On the other hand, climate variability and climate change can also increase regional water resources availability as is for example discussed for the greening of the Sahel (Hickler et al. 2005). However, due to the interwoven effects of climate and LULC change effects, a clear apportionment of the various drivers is often difficult (Kaptué et al. 2015; Rasmussen et al. 2012). With regard to economic drivers, it is postulated that land use change is a result of favorable economic and institutional conditions triggered by urbanization and expansion of agriculture and rangelands. In political ecology, it is argued that land use change is driven by development policies and often, by the maintenance of the globalization and the market forces of capital and multinational enterprises. Land and water grabbing (Rulli et al. 2013), the establishment of unsustainable irrigation schemes where groundwater exploration is greater than recharge rates (Yoshikawa et al. 2014; Rodell et al. 2009), and allocation of land to elites which marginalizes poor populations (De Waroux and Lambin 2012) are all potential drivers of land use change (De Waroux and Lambin 2012).

Independent of the causes of land use change, the expectation is that the changes will alter the hydrologic cycle in the region given that vegetation cover has a profound effect on groundwater recharge rates (Fig. 1). For instance, Gee et al. (1992) reported that areas covered with deep-rooted vegetation such as forests have lower groundwater recharge rates than areas of shallow-rooted vegetation such as annual crops. Both field data (Allison et al. 1990; Scanlon et al. 2007) and modeling results (Keese et al. 2005) show increased groundwater recharge rates when the natural deep-rooted native vegetation (trees and shrubs) is changed to shallower-rooted agricultural crops. The change from native vegetation to managed land use types often results in increases of recharge rates by one or two orders of magnitude (Scanlon et al. 2006). The response of water resources to LULC change is influenced by several factors including the original vegetation to be replaced, the vegetation that is replacing it, whether the change is permanent or temporary, and if the changes are associated with land management practices involving alteration of drainage networks (Scanlon et al. 2007). Evapotranspiration rates in natural forests, bushland, and savanna might be higher than that of other, open vegetation types (Scanlon et al. 2005). However, for actual evapotranspiration rates, this difference depends on the amount of incoming precipitation. Zhang et al. (2001) reviewed more than 250 catchments worldwide. As a rule of thumb, they concluded that evapotranspiration between open and closed vegetation can be separated if incoming annual rainfall is greater than 500 mm. With lower precipitation rates, all vegetation types showed comparable evapotranspiration rates due to water limitations.

If conversion of natural forest to cultivated crops reduces evapotranspiration losses, excess water is available for increasing groundwater recharge and/or streamflow (Scanlon et al. 2005; Scanlon et al. 2006). Further, due to evolutionary adaptation, e.g., osmotic adjustment (Chen and Jiang 2010) or hydraulic redistribution (e.g., Sardans and Penuelas 2014), arid and semi-arid native vegetation is capable of making better use of soil moisture and accessing deeper soil water reserves as compared to agricultural crops (Stonestrom and Harril, 2007; Kesse et al. 2005; Scanlon et al. 2005). Conversion of agricultural land back to "natural" vegetation therefore may result in decreased runoff and decreased in-stream sediment loads due to reduced erosion, all key elements for sustainable water resource management (Scanlon et al. 2007).

Soil hydro-physical properties such as texture, hydraulic conductivity, bulk density, and porosity also impact watershed hydrology. These hydro-physical properties influence how precipitation enters and is retained in the soil. It further determines the rate of transmission and pathways of water to stream networks (Price et al. 2010). Land management practices like tillage alter soil structure as well as soil porosity and pore size distribution. This results in a break in the continuity of macropores in the plough layer. It reduces the infiltration

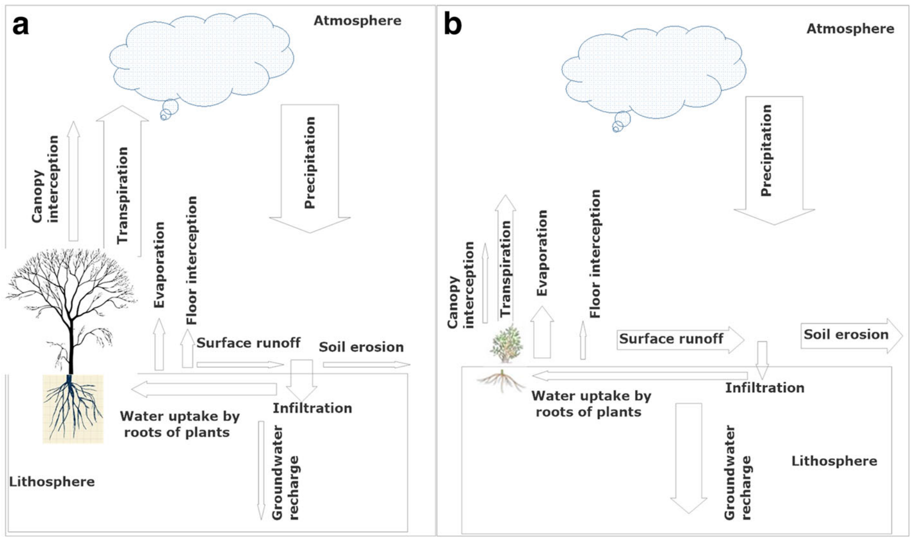


**Fig. 1** Schematic representation of the local water cycle in (**a**) forest and (**b**) managed land uses. *Arrows* represent movement of water and their thickness is proportional to the magnitude of the water flow. Deforestation leads to reduction in evapotranspiration and increase in the surface runoff in to the rivers, groundwater recharge, and soil erosion but reduced infiltration assuming constant precipitation rate

rate of the soil, which consequently can result in reduced groundwater recharge and increased surface runoff. Soils under native vegetation often have a lower bulk density and higher saturated hydraulic conductivity, total porosity, macroporosity, and decreased or non-existing overland flow as compared to agricultural soils because of the presence of abundant litter cover, organic inputs, root growth and decay, and abundant burrowing fauna (Lee and Foster 1991; Bens et al. 2007; Zimmermann et al. 2006; Ilstedt et al. 2007; Leblanc et al. 2008). Heavy machinery use or animal draft power may compact the soil and result in an increase in soil bulk density, destruction of macropores, and a decrease in the number of small pores resulting in a decrease in infiltration rates and saturated hydraulic conductivity (Bodhinayake et al. 2002). In most cases, continuous cultivation leads to decreased soil organic matter content, reduced aggregate stability (Peterson et al. 1988), and leaves—at least during part of the year—bare soil exposed to the force of rain drops. The impact of drops on bare soil can break down soil aggregates, leading to the formation of a surface crust, which usually reduces infiltration and hydraulic conductivity (Price et al. 2010). Grass cover or retention of crop residues may help reduce the negative effect of raindrop impact on bare soils, while incorporation of crop residues into the top soil can increase soil organic matter content and thus strengthen soil aggregates, preventing the formation of surface seals and maintaining high infiltration rates and high hydraulic conductivity (Bodhinayake et al. 2002).

This review paper focuses on groundwater recharge because total depletion of groundwater in sub-humid to arid regions has increased from 126 $km^3\ yr^{-1}$ in 1960 to 283 $km^3\ yr^{-1}$ in 2000 as reported by Huang et al. (2013). Further, as noted by Scanlon et al. (2006), previous reviews on recharge studies in semi-arid and arid regions lacked a global perspective. The aim of this study is to summarize current knowledge on LULC change impacts on groundwater recharge and surface runoff in tropical, subtropical, and Mediterranean semi-arid environments, thereby also considering effects of, e.g., soil texture on groundwater recharge. Our hypotheses are that (i) groundwater recharge and surface runoff are higher in managed land use types such as agricultural fields compared to native vegetation such as forests and bushland, (ii) groundwater recharge and surface runoff are similar among managed land use types such as agricultural fields, rangelands, and grasslands under semi-arid conditions as other factors such as (iii) soil texture might override land use type effects.

## Methodology

### *Data compilation*

We compiled datasets on impacts of land use and land cover change on groundwater recharge rates and surface runoff for tropical, sub-tropical, and Mediterranean semi-arid regions from peer-reviewed journals, thereby considering catchment as well as plot studies. We collected data from catchments where the magnitude of groundwater recharge rates and surface runoff were measured or modeled before and after LULC change. In order to identify the publications, major scientific literature databases (Scopus, Google Scholar, and Web of Science) were used. Figure 2 outlines our search strategy. The result from search engines gave a total of 62 published papers.

Potential evapotranspiration (PET) data was not reported for most of the studies and was considered an important parameter for characterizing the sites. Therefore, we obtained the PET values for each study site from a global-PET database (Trabucco and Zomer 2009), which is available as a single annual average data layer for the period from 1950 to 2000. We further divided reported mean long term rainfall by PET values to obtain aridity indices for the individual study sites using Eq. (1) as described by UNESCO (1979):

$$\mathrm{AI} = \frac{P(\mathrm{mm\,year^{-1}})}{\mathrm{PET}\,(\mathrm{mm\,year^{-1}})} \quad (1)$$

where AI is the aridity index, $P$ is the average annual precipitation, and PET is average annual potential evapotranspiration.

Following a first identification of 20 sites from 62 studies, we assigned each site with an AI and we further refined our search by using the following criteria:

- The AI was within a 0.15–0.60 threshold, following an expanded definition from UNESCO (1979) to include sites near the boundaries of the definition of a semi-arid area.
- The studies reported groundwater recharge rate (or surface runoff) both before and after land use change, allowing for paired observations. The publication should explicitly report the absolute quantitative groundwater recharge rate (or surface runoff) before and after land use change.

Information on precipitation or the reported long-term mean precipitation, groundwater recharge rates, surface runoff, runoff coefficients, soil texture, land use, land cover, and vegetation type (e.g., plantation, native bushland, annual crops) were extracted from the retrieved 27 publications (Appendices 1 and 2). Studies with multiple years and locations for groundwater recharge estimates were evaluated as independent data points. The 75 selected cases studies for groundwater recharge and 58 case studies for surface water were distributed in different parts of the world including Africa, North America, Australia, and China (Fig. 3).

### Groundwater recharge rate analyses

We standardized (normalized) annual groundwater recharge rates [mm year$^{-1}$] before and after LULC change with the information on long-term mean annual precipitation [mm year$^{-1}$] reported in the individual studies (Eq. 2):

$$\mathrm{GW\ recharge}\,[\%] = \frac{\mathrm{mean\ annual\ GW}\,[\mathrm{mm\,year^{-1}}]}{\mathrm{mean\ annual\ sum\ of\ precipitation}\,[\mathrm{mm\,year^{-1}}]}\cdot 100 \quad (2)$$

For the groundwater recharge analyses, we considered five major LULC types (both before and after change): forest, cropland, grassland, rangeland, and bare land. Forest vegetation consists of woodland, eucalyptus plantation, and bushland. Cropland includes annual crops and perennial crops. Grassland refers to natural vegetation with no livestock grazing. Rangeland comprises pasture used for livestock grazing. However, due to missing information, we could not further distinguish the intensity of their use, e.g., by stocking rates. Bare lands are an artificial scenario created through clearing of natural vegetation and avoidance of regrowth (e.g., Kesse et al. 2005; Peck and Williamson 1987; Moore et al. 2012). The majority of the studies ($n = 58$) investigated conversion of forest to cropland ($n = 28$), grassland ($n = 12$), and rangeland ($n = 18$). Eleven modeling studies corresponded to restoration of bare land to cropland ($n = 2$), grassland ($n = 4$), and forest ($n = 5$). Additionally, the sites were grouped according to the topsoil soil texture (depth to 0.3 m) into sand, sandy loam, loam, and clay.

We further determined the magnitude of absolute change in groundwater recharge by subtracting the post

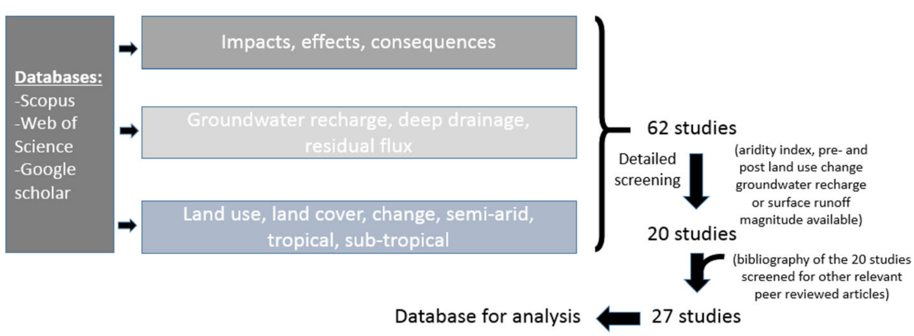


**Fig. 2** Databases, key words, and search strategy used for identification of relevant studies on LULC change on groundwater recharge in semi-arid environments

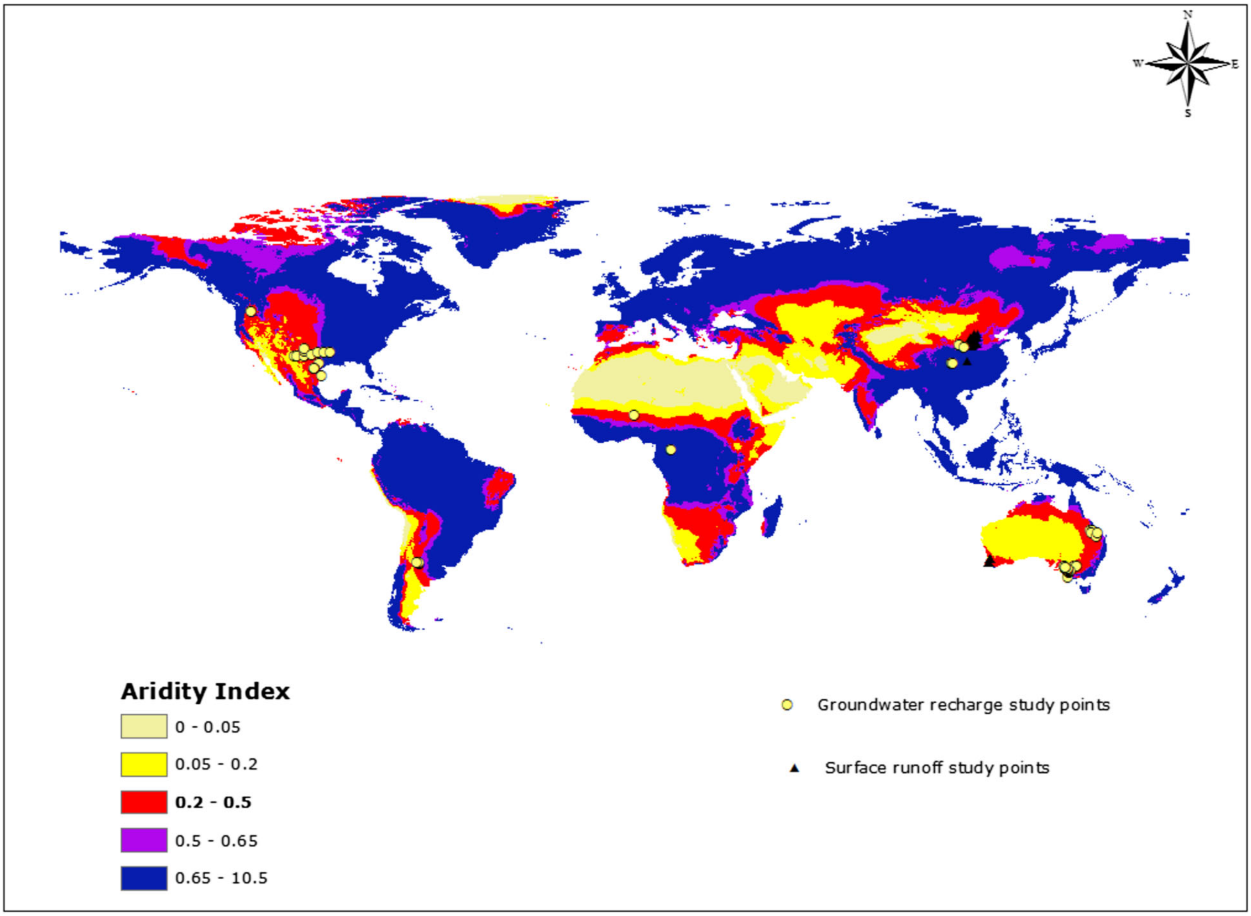


**Fig. 3** Location of published studies on the effects of land use change in semi-arid environments on groundwater recharge and surface runoff used for study. Sites are shown on a global aridity map (FAO 2009)

land use normalized groundwater recharge from the pre-land use normalized groundwater recharge (Eq. 3).

$$\Delta \mathrm{GW}\,[\%] = \mathrm{GW}_{\mathrm{postLULC}}[\%] - \mathrm{GW}_{\mathrm{pre}_{\mathrm{LUCL}}}[\%] \qquad (3)$$

where $\Delta$GW is the absolute change in groundwater recharge, $\mathrm{GW}_{\mathrm{preLULC}}$ is the groundwater recharge before land use change, and $\mathrm{GW}_{\mathrm{postLULC}}$ is the groundwater recharge after land use change.

### Surface runoff analyses

Surface runoff gathered from the studies was standardized (normalized) with the reported long-term mean annual precipitation as follows (Eq. 4):

$$\mathrm{SR}\,[\%] = \frac{\text{mean annual SR}\,[\text{mm year}^{-1}]}{\text{mean annual sum of precipitation}\,[\text{mm year}^{-1}]} \cdot 100 \qquad (4)$$

For the surface runoff analysis, the LULC conversions investigated were rehabilitation of bare land to soil and water conservation measures ($n = 15$), bare land to crops ($n = 4$), crops to soil and water conservation measures ($n = 4$), grassland to forest vegetation ($n = 18$), forest vegetation to rangeland ($n = 2$), forest to cropland ($n = 2$), and forest to bare land ($n = 13$). Here, only data measured in the field was used. We determined the magnitude of absolute change in surface runoff in the same way as for groundwater recharge (Eq. 5).

$$\Delta \mathrm{SR}\,[\%] = \mathrm{SR}_{\mathrm{postLULC}}[\%] - \mathrm{SR}_{\mathrm{preLULC}}[\%] \qquad (5)$$

where $\Delta$SR is the absolute change in surface runoff, $\mathrm{SR}_{\mathrm{postLULC}}$ is the surface runoff after LULC change, and $\mathrm{SR}_{\mathrm{preLULC}}$ is the surface runoff before LULC change.

#### *Statistical analysis*

Groundwater recharge and surface runoff data did not follow normal distributions (Shapiro-Wilk's test), and therefore, non-parametric tests were used. A Wilcoxon-signed ranked test was used for assessing the effect of the different LULC changes in groundwater recharge rates and surface runoff. Kruskal-Wallis tests were used to investigate the effect of soil texture on the groundwater recharge due to LULC change. Linear regression analyses were performed between groundwater recharge and surface runoff and aridity indices of the sites. SPSS v21 software (SPSS 2012) was used for the statistical analyses. The statistical significance level was set to $P \leq 0.05$.

## Results

### *Effects of LULC change on groundwater recharge*

Our results show that groundwater recharge is influenced by LULC change (Fig. 4, Table 1). Restoration of bare land decreased groundwater recharge from 42 to 6–12% of the incoming rainfall depending on final LULC (Table 1). The decrease in groundwater recharge after conversion of bare land was significant only for forests. Conversion of forest to other LULC caused an increase from 0.15 to 3.4–7.8% in groundwater recharge, which was significant for all the considered final LULC.

Soil texture affected the change in groundwater recharge after conversion of forest to other LULC (Fig. 5). The effect of forest loss on groundwater recharge was more pronounced in sand textured soils than in other textural classes.

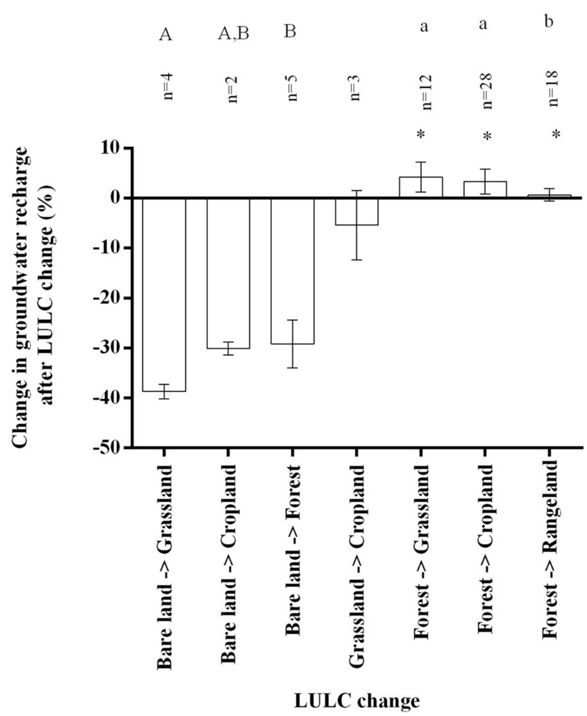


**Fig. 4** Differences in groundwater recharge with change in LULC. Values are mean ± SD. *n* = number of studies. All values with bare land as initial LULC are coming from modeling studies. *significant difference at $p < 0.05$ for a given LULC change. Significant differences among LULC groups are given by *letters*. *Different letters* mean significant differences at $p < 0.05$. *Capital letters* are comparisons from modeling result while *small letters* are comparisons of field measurement

**Table 1** Summary table of Wilcoxon-signed ranked test results on the effects of LULC change on groundwater recharge in semi-arid regions

| GW recharge rate (% of precipitation) | | | | | |
|---|---|---|---|---|---|
| Pre-LULC | Post LULC | Pre-LULC | Post LULC | *p* | *n* |
| Bare land | Cropland | 36 ± 21 | 6.4 ± 0.8 | – | 2 |
| Bare land | Grassland | 50 ± 0.2 | 11.7 ± 1.4 | *0.068* | 4 |
| Bare land | Forest | 38 ± 10 | 8.5 ± 6.8 | **0.043** | 5 |
| Bare land | All | 42 ± 9 | 9.3 ± 4.8 | **0.003** | 11 |
| Grassland | Cropland | 14 ± 12 | 8.6 ± 4.8 | 0.29 | 3 |
| Forest | Cropland | 0.13 ± 0.18 | 3.4 ± 2.5 | **< 0.001** | 28 |
| Forest | Grassland | 0.18 ± 0.54 | 4.4 ± 3.3 | **0.002** | 12 |
| Forest | Rangeland | 0.13 ± 0.14 | 0.64 ± 1.26 | **0.001** | 18 |
| Forest | All | 0.15 ± 0.28 | 3.7 ± 7.1 | **< 0.001** | 59 |

Italicized entries are significant at $P < 0.1$, and bolded text indicates significance at $P < 0.05$

The relationship between the Aridity Index and groundwater recharge (Fig. 6) was weak ($R = 0.22$) but significant ($P = 0.06$), suggesting that in more arid environments, the change in the recharge after LULC change is stronger. Regression results of changes in groundwater recharge varied when each change in LULC was analyzed separately as follows: forest to cropland ($R = 0.1$, $P = 0.42$, $n = 28$), forest to grassland ($R = 0.5$, $P = 0.06$, $n = 11$), forest to rangeland ($R = 0.26$, $P = 0.15$, $n = 18$).

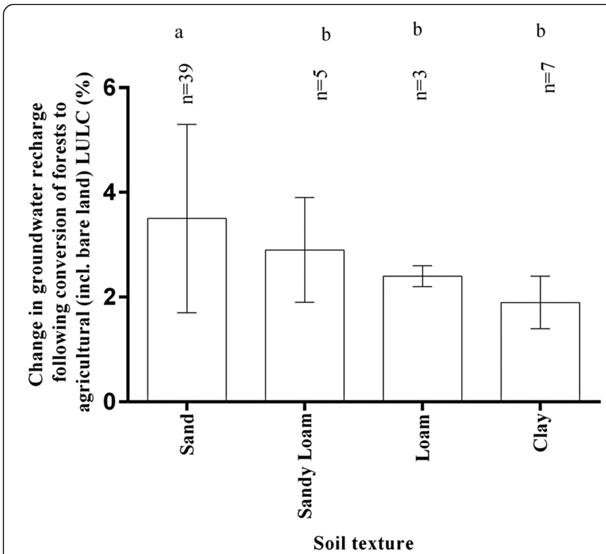


**Fig. 5** Effect of soil texture on the difference of groundwater recharge following conversion of forest to another LULC class in arid and semi-arid lands. Given are mean values ± SD. *n* = number of studies. *Different letters* mean significant difference at $p < 0.05$ for different soil textural classes

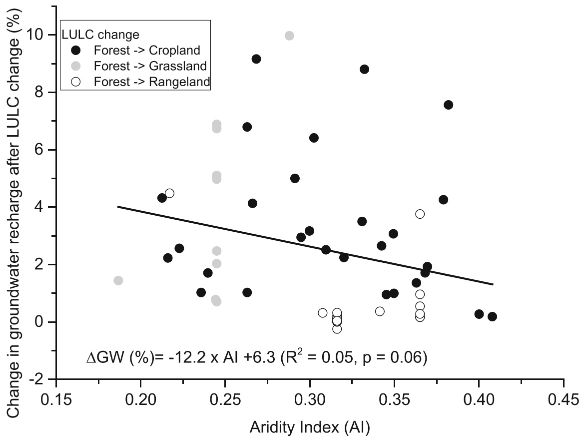


**Fig 6** Relationship between the Aridity Index and changes in groundwater recharge after conversion of forest to other agricultural land uses

### *Effects of LULC change on surface runoff*

Rehabilitation of bare land resulted in reduced surface runoff from 7.3% of incoming precipitation lost as runoff to 5.2% in croplands and 6.4% when soil and water conservation measures were implemented (Fig. 7). The conversion of forest vegetation to managed LULC showed an increase in surface runoff from 1 to 4.2–14.1% depending on final LULC (Fig. 7, Table. 2). Surface runoff was reduced from 2.5 to 1.1% when grassland was converted to forest vegetation. The linear relationship between surface runoff and the Aridity Index was weak but significant ($R = 0.37$, $P = 0.01$) (Fig. 8). Regression results of change in surface runoff varied when each change in LULC was analyzed separately as follows: grassland to forest ($R = -0.16$, $P = 0.5$, $n = 18$), forest to bare land ($R = 0.6$, $P = 0.01$, $n = 13$), and bare land to bare land + SWC ($R = 0.24$, $P = 0.189$, $n = 15$).

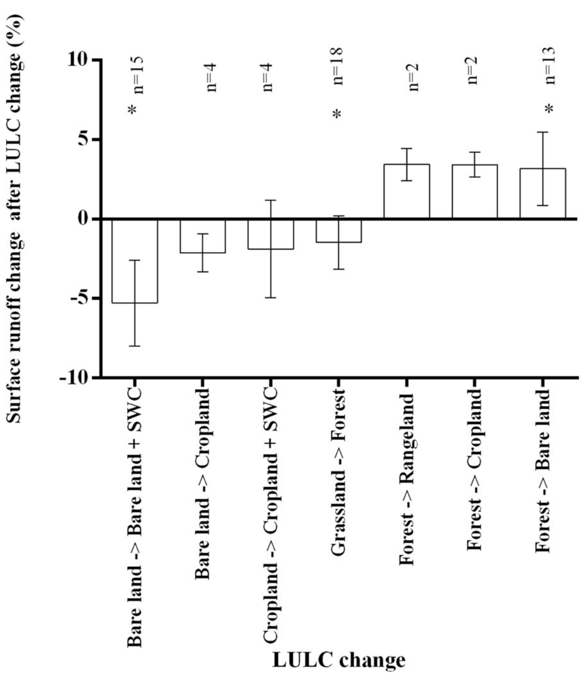


**Fig. 7** Differences in surface runoff as a result of LULC. Given are mean values ± SD. *n* = number of studies. *SWC* = soil and water conservation measures such as terracing or building of small dams within the catchment. *significant difference at $p < 0.05$ for a given LULC change

## Discussion

This study shows that groundwater recharge varies across different types of LULC. Moreover, groundwater recharge in arid and semi-arid lands is influenced by the soil texture of the topsoil to 0.3 m depth. Despite high heterogeneity associated with differences in estimation methods, study periods, and locations, our results consistently indicate that the conversion of forest land/native vegetation to managed LULC systems leads to increased groundwater recharge. The influence of LULC on groundwater recharge is mostly driven by the vegetation rooting system, interception capacity of the canopy, and transpiration rates (Taniguchi 1997, Wang et al. 2004). Lower groundwater recharge rates are recorded for deep-rooted native vegetation types compared with shallow-rooted vegetation such as annual crops or bare lands. In forests, which are characterized by deep roots, rain passes through the forest soil to subsoil layers. As discussed by Burch et al. (1987) and Taniguchi (1997), the soil water flow under natural vegetation is enhanced by macropores created by worms and other soil fauna and by the growth and decay of tree roots. However, forest systems have higher evapotranspiration rates than managed land use types (Scanlon et al. 2005); therefore, any gains in recharge are often offset by evapotranspiration losses (see also Fig. 1).

Forest systems also influence water loss by interception (Schofield and Ruprecht 1989; Williamson et al. 1987; Taniguchi 1997). High interception under forest systems therefore further limits water availability for recharging groundwater. In contrast, the managed LULC such as annual crops and grassland interception losses are lower (Williamson et al. 1987), which also contributes to the finding of higher groundwater recharge rates across the evaluated studies.

Native vegetation is adapted to soil water limitations and is highly effective in extracting soil moisture, e.g., due to a spatially extensive and/or deep rooting system (Stonestrom and Harril 2007; Keese et al. 2005; Scanlon et al. 2005). Managed LULC such as crops and grasses are less efficient in extracting soil moisture (Keese et al. 2005).

**Table 2** Summary table of Wilcoxon-signed ranked test results on the effects of LULC change on surface runoff

| Surface runoff (% of precipitation) | | | | | |
|---|---|---|---|---|---|
| Pre-LULC | Post LULC | Pre-LULC | Post LULC | *p* | *n* |
| Bare land | Cropland | 7.3 ± 7.3 | 14.9 ± 5.2 | *0.068* | 4 |
| Bare land | Bare land + soil and water conservation | 11.7 ± 6.3 | 6.4 ± 4 | **0.001** | 15 |
| Grass land | Forest | 2.5 ± 2.9 | 1.1 ± 1.4 | **<0.001** | 18 |
| Cropland | Cropland + soil and water conservation | 5.2 ± 6.8 | 3.3 ± 3.8 | 0.109 | 4 |
| Forest | Cropland/rangeland | 10.7 ± 7.6 | 14.1 ± 7.5 | 0.68 | 4 |
| | Cropland | 10.7 ± 9.3 | 14.1 ± 8.5 | | |
| | Rangeland | 10.7 ± 9.3 | 14.1 ± 10.3 | | |
| Forest | Bare land | 1.0 ± 0.17 | 4.2 ± 2.2 | **0.001** | 13 |

Italicized entries are significant at $P < 0.1$, and bolded text indicates significance at $P < 0.05$

Therefore, conversion of forests to managed LULC results in increased groundwater recharge. Generally, the conversion of native vegetation to managed land use types leads to increased groundwater recharge. However, water quality may decrease too, specifically in cases of semi-arid environments where salts previously stored in the soil get mobilized due to rise in groundwater level (Scanlon et al. 2007; Leaney et al. 2003). Moreover, agricultural land uses are mostly associated with intensive use of fertilizers to enhance crop growth, with part of it being leached to the groundwater. For example, Scanlon et al. (2008) reported a significant increase in nitrate concentrations from 2–10 kg $NO_3$-N $ha^{-1}m^{-2}$ under a natural ecosystem to 28–580 kg $NO_3$-N $ha^{-1}m^{-2}$ for agricultural land uses when the land use was changed to rain fed (non-irrigated) agricultural LULC in the southern High Plains, USA.

Our study indicates that comparison of changes from forest/native vegetation to rangelands in semi-arid regions tended to have a lower groundwater recharge than a change to grasslands (Fig. 4). This observation is most likely due to compaction of rangeland top soils by grazing animals. Radford et al. (2009), Betteridge et al. (1999), and Broersma et al. (1999) have reported increased soil compaction with high grazing intensity caused by the mechanical stress exerted on soil by grazing animals (Zhou et al. 2010). Both Rezkowska et al. (2011a) as well as Krümmelbein et al. (2009) reported that grazing of semi-arid steppe soils also reduced soil water pore volume and saturated soil hydraulic conductivity. The reduced soil hydraulic conductivity was attributed to destruction of macropores (Reszkowska et al. 2011b).

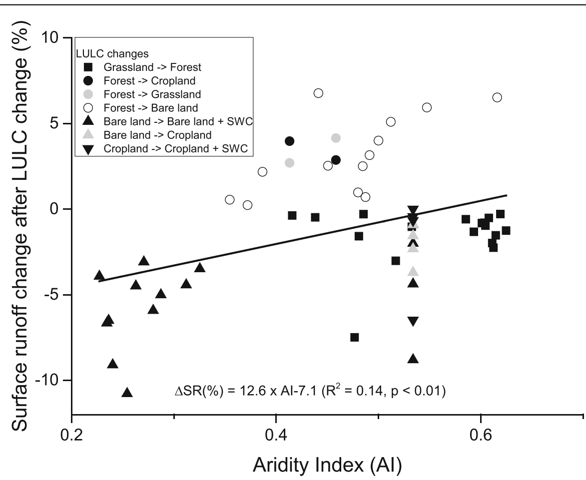


**Fig 8** Relationship between the Aridity Index and changes in surface runoff after conversion of forest to other agricultural land uses

Kim and Jackson (2012) and Turner and Lambert (2014) reported higher groundwater recharge under grassland than under rangeland in humid climates. Kim and Jackson (2012) reported relative differences in groundwater recharge when grassland was replaced with woodland and cropland to be −70 and −250%, respectively, in arid areas while the conversion of grassland to woodland and cropland was −20 and −60% in humid climates. Generally, it can be stated that humid areas with high groundwater recharge rates show large absolute differences in recharge in response to LULC. Whereas in drier, arid, and semi-arid climates with rather low groundwater recharges, absolute changes in recharge rates in response to LULC changes are small.

Irrespective of land use type, our literature review also showed that variability in groundwater recharge in semi-arid environments depends on the texture of the top soil (Fig. 5). The influence of soil texture on the effect of LULC change on groundwater recharge can be attributed to differences in hydraulic conductivities and water infiltration rates of different soil textures (Gee et al. 1992; Mamedov et al. 2001). Sandy soils have faster water infiltration rates and, in this context, the presence of a closed canopy is highly significant for varying the groundwater recharge, compared with clayey soils with lower infiltration rates. As observed by Santoni et al.

(2010) and Wang et al. 2009, an increase in sand content results in deeper wetting fronts in unsaturated condition under similar climatic condition. To exhaustively utilize the precipitation inputs by the natural vegetation, different root traits are needed, i.e., a deeper and denser root system is required as compared to soils with other texture characteristics. This highlights the relationship between natural vegetation composition, soil texture, infiltration, and groundwater recharge.

LULC change not only affects groundwater recharge in semi-arid regions but also surface runoff. All studies evaluated show that the lowest surface runoff can be found for forest systems, which can be explained by several factors such as (a) lower and less intensive rainfall reaching the ground due to canopy effects and interception losses, (b) enhanced infiltration capacity due to plant roots and biopores, and (c) higher topsoil water retention due to organic litter accumulation on soils and higher organic matter concentrations in the top mineral soils (e.g., Saiz et al. 2015; Burch et al. 1987; Taniguchi 1997). Conversely, managed land use types have higher surface runoff due to lower soil infiltration rates and capacities and low evapotranspiration that leads to a higher amount of effective rainfall, i.e., the rainfall available for surface runoff. Global data from 185 catchment studies compiled by Sun and Li (2005) also showed that forest clearing increased water yield while tree planting decreased water yields. High runoff recorded in bare land can also be the result of surface sealing and crusting resulting from the impact force of rain drops which reduces infiltration rates and capacity (Price et al. 2010). In semi-arid regions of West Africa, on a scale of 2 $km^2$, Favreau et al. (2009) reported a threefold increase in surface runoff even though rainfall had decreased by approximately 23% from 1970 to 1980. This increased runoff has been associated with conversion of natural savanna to millet crops that have expanded sixfold since 1950 in the area, i.e., stimulation of surface runoff is caused by reduced ET of the millet crop as compared to natural savanna. This interpretation confirms the observation of Santoni et al. (2010) who found that increase in water supply under natural vegetation cannot trigger onset of deep drainage because natural vegetation effectively consumes additional water supply.

Soil conservation measures also reduce surface runoff (Fig. 7) as those measures increase infiltration rates and increase the main residence time of water on the soil surface by reducing slopes or in upstream areas by terracing or dams. Such trends were reported by Krois and Schulte (2013) who investigated the impact of soil and water conservation techniques on the hydrology of a watershed in the northern Andes and reported that earth work structures (terraces and bund structures) and afforestation significantly affected the flow volume, overland flow generation, and high flows. The earth work structures led to a reduction of surface runoff by 12–28% and afforestation led to a reduction in surface runoff by 9–11%.

The magnitude of specific surface runoff generation mechanisms as a result of LULC change vary considerably across study sites. Here, it has been found that the permanent groundwater discharge area is an important factor determining the magnitude of the streamflow response after LULC change (Ruprecht and Stoneman 1993; Ruprecht and Schofield 1989).

Even though the data for semi-arid environments do not necessarily show it, the change in runoff due to land use change can be understood as a three-stage event (Ruprecht and Schofield 1989). The first stage is an initial abrupt jump in runoff in the first year of clearance which is attributed to immediate decrease in interception losses; the second stage is a linear increase in runoff attributed to the expansion of groundwater recharge and the final stage is the gradual return to the pre-disturbance value attributed to attainment of new groundwater recharge-discharge equilibrium.

Land use also affects the minimum amount of rainfall required to initiate surface runoff. The relationship between Aridity Index and surface runoff (Fig. 8) indicates that the impact of LULC change on surface runoff is more pronounced in areas of high aridity. Long-term analysis of runoff in semi-arid catchment of Lemon, a study region located in southwest Western Australia (Ruprecht and Schofield 1989), showed that the minimum rainfall to initiate runoff had shifted fundamentally due to land use change. Prior to forest clearing, approximately 700 mm of annual rainfall was required for commencement of streamflow while only 100 mm of annual rainfall was required for commencement of streamflow in post forest clearance. The regression curves of rainfall before and after deforestation were similar but shifted to accommodate the increased runoff. Similar trends were also documented by the same authors in 1991 for the Don catchment in southwest Western Australia, where a minimum of 739 mm of annual rainfall was required to initiate streamflow in the pre-clearing period while during the post clearing period, a minimum of 532 mm of annual rainfall was required. As observed by Hallema et al. (2016), the propagation of generated runoff depends on the ability of hillslope to retain and release water, which largely controls the hydrologic response over the course of a given rainfall event. Dunjó et al. (2004) observed that high rainfall intensity generated more runoff than similar amounts of precipitation at low intensity. It has further been noted by Allison et al. (1994) that high rainfall variability causes high variability in recharge. Scanlon et al. (2006) have reported up to three times greater groundwater recharge during periods of El Niño relative to periods dominated by La Niña.

### Study limitations and recommendations

To the best of our knowledge, our review is the first global analysis that synthesizes quantitative information on groundwater recharge and surface runoff in semi-arid landscapes experiencing different LULC changes. From our analyses, we found clear indicators that LULC significantly alters hydrological fluxes.

Our study only focused on the relationship between LULC change and groundwater recharge and surface runoff in semi-arid landscapes, neglecting additional interactions due to other landscape properties such as slopes, parent material, or underlying geological structures, which have been found to have significant impacts on observed responses in a number of studies, most of which carried out in humid environments (Senthilkumar et al. 2015).

There were four main limitations to this synthesis. Firstly, the evapotranspiration data was missing from most of the reviewed studies. Evapotranspiration plays a significant role in the water balance of semi-arid tropical regions as indicated by an Aridity Index of 0.6 in this study, compared to humid regions where the index is mostly well above 1. Our analysis suggests that a threshold between the response changes in evapotranspiration due to LULC change and associated groundwater recharge responses exist. Secondly, there was no case study that quantified both groundwater recharge and surface runoff responses to LUC change in the same landscape. Thirdly, most of the studies precluded detailed assessments of individual and interactive effects of other stressors (e.g., hydro-electric power production and groundwater abstraction) besides LULC. As outlined by Przeslawski et al. (2015), quantifying only one stress factor obviously oversimplify complex systems. This calls for multifactorial studies to better understand interactive effects of LULC and other landscape characteristics.

Finally, different methods used for obtaining groundwater recharge rates also affect the results of our analysis. Allison et al. (1994) noted that chemical and isotopic methods yield more reliable results when used to estimate groundwater recharge than physical methods that can have an error of as much as an order of magnitude or higher in semi-arid areas where groundwater recharge is low. The merit of tracers is that they take all hydrological processes into account that interact to influence water movement in the unsaturated zone (Allison et al. 1994).

The influence of reforestation on the reduction on runoff is dependent on the percentage of the catchment under reforestation (Bruijnzeel 2004). However, a conclusive threshold on the area of the catchment which must be cleared in semi-arid environments before substantial change in runoff is realized has not yet been determined. Simulation results by Li et al. (2007) on Lake Chad and Niger in West Africa indicated that at least 50% of the catchment needed to be cleared to realize an impact on water yield and river discharge. This is greater than what was found in humid environments where the thresholds ranged from 13% (Sun and Li 2005) to 20% (Bosch and Hewlett 1982; Stednick 1996).

Overall, the number of available case studies from semi-arid regions worldwide is limited. Given the spatial extent of semi-arid regions in Africa specifically, our study shows that the topic is largely understudied. Most of the cases studies evaluated here are from Australia, South America, and North America, and extrapolating results from those studies to other continents such as Africa needs to be done with caution due to environmental and geological variability. Based on our analyses and in view of the rapid growing demand for irrigation water and the water demand for industrial and domestic use in Africa by a dramatically growing population (Odongo 2013), it is evident that there is a need for increased investment in groundwater monitoring in sub-Saharan Africa. Long-term field studies should be initiated to cover this data gap.

## Conclusions

We reviewed published research on the impacts of LULC change on groundwater recharge and surface runoff in semi-arid areas. From our study, the following conclusions can be drawn.

i) Conversion of forests to managed land use types resulted in increased groundwater recharge and surface runoff.
ii) Besides LULC change, our analysis also confirmed the significant effect of topsoil texture on groundwater recharge, with highest recharges being found for coarse-textured soils, thereby confirming observations by Gee et al. (1992).

The variability in the magnitude of responses to LULC changes indicates a need for site-specific studies to understand the influences of land cover changes in semi-arid environments, specifically in Africa. It is important that such studies also quantify the combined influences of multiple stressors, e.g., LULC, climate, as well as landscape and soil properties.

## Appendix

**Table 3** Data for groundwater recharge rate study

| Author | Soil texture | Aridity index | Location of study area | Period of study (years) | Latitude | Longitude | Method used | Precipitation (mm/year) | Potential evaporation (mm/year)[a] | Type of natural vegetation | Type of vegetation after change |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Huang et al. (2013) | | 0.497 | Hequan, Guyan, China | 130 | 35.9 | 106.25 | Muttiple tracers (chloride mass balance, stable isotopes, tritium and water chemistry) | 450 | 906 | Grassland | Winter wheat |
| Huang and Pang (2010) | | 0.481 | Guyuan, China | 100 | 30.1 | 104 | Chloride-mass balance | 500 | 1040 | Grassland | Winter wheat |
| Huang and Pang (2010) | | 0.501 | Xifeng Loess Plain, China | 7 | 30.1 | 104.5 | Chloride-mass balance | 523 | 1044 | Winter wheat | Apple |
| Allison et al. (1990) | Sand | 0.213 | Western Murray Basin, Australia | | −34.3 | 139.6 | Groundwater model | 300 | 1245 | Mallee | Cropland |
| Allison et al. (1990) | Sandy loam | 0.262 | Western Murray Basin, Australia | | −35.1 | 140.3 | Groundwater model | 370 | 1346 | Mallee | Cropland |
| Allison and Hughes (1972) | | 0.631 | Western Murray Basin, Australia | | −37.8 | 140.8 | Tritium method | 686 | 1132 | Mallee | Grassland |
| Allison and Hughes (1983) | | 0.236 | South Australia | | −35.1 | 142.1 | Chloride method | 335 | 1379 | Mallee | Wheat cropland |
| Scanlon et al. (2005) | | 0.331 | High plains (HP1) in South High Plains, the USA | | 32.9 | −102.1 | Matric potential, environmental tracers and water table fluctuations and trends in groundwater solutes | 500 | 1670 | Creosote bush and saltbush | Alfalfa |
| Scanlon et al. (2005) | | 0.291 | High plains (HP 2) in South High Plains, the USA | | 32.9 | −102.1 | Matric potential, environmental tracers and water table fluctuations and trends in groundwater solutes | 440 | 1670 | Creosote bush and saltbush | Alfalfa |
| Scanlon et al. (2005) | | 0.302 | High plains (HP 3) in South High Plains, the USA | | 32.9 | −102.1 | Matric potential, environmental tracers and water table fluctuations and trends in groundwater solutes | 457 | 1670 | Creosote bush and saltbush | Alfalfa |
| Santoni et al. 2010 | Sandy loam | 0.350 | Central Argentina | > 30 years | −33.6 | −65.8 | Residual moisture flux (RMF) and chloride front displacement (CFD) | 518 | 1371 | Dry forest | Crops (dryland agriculture) |
| Santoni et al. (2010) | Sandy loam | 0.368 | Central Argentina | > 30 years | −33.5 | −65.8 | Residual moisture flux (RMF) and chloride front displacement (CFD) | 542 | 1371 | Dry forest | Crops (dryland agriculture) |
| Santoni et al. (2010) | Sandy loam | 0.370 | Central Argentina | > 30 years | −33.4 | −65.9 | Residual moisture flux (RMF) and chloride front displacement (CFD) | 538 | 1383 | Dry forest | Crops (dryland agriculture) |
| Santoni et al. (2010) | Sandy loam | 0.295 | Central Argentina | > 30 years | −33.4 | −66.5 | Residual moisture flux (RMF) and chloride front displacement (CFD) | 447 | 1383 | Dry forest | Crops (dryland agriculture) |
| Leduc et al. (2001) | | 0.266 | Southwest Niger | | 13.6 | 2.6 | | 557 | | Natural bush | Crops |
| Lebel et al. (2009) | | | Niamey (Southwest Niger) | 1990–2009 | 2.5 | 14.7 | | 520 | | Grassland | Millet |

**Table 3** Data for groundwater recharge rate study *(Continued)*

| | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Allison et al. (1983) | Sand | 0.263 | | | −35.11 | 140 | Cores of material in the saturated zone were taken from depths of from 9 to 15 m beneath native and cropped land. Analyses made of water content and chloride, stable isotopes, and tritium. | 335 | 1379 | Eucalyptus scrub (mallee) | Cropped wheat-fallow |
| Allison et al. (1985) | | 0.217 | Murray Basin, South Australia | | | | Chloride mass balance and tritium dating | 300 | 1374 | Native mallee vegetation | Poorly developed pasture cleared in the 1930s |
| Holmes and Colville (1970a, 1970b) | | 0.288 | Karstic region of southern Australia | | | | | 632 | | Plantation forest of Monterey pine | Grassland |
| Gee et al. (1992) | Sand | 0.153 | 200 East | 1971–1985 change 1988–1989 | 46.6 | −119.4 | Lysimeter | 168 | 1083 | Tumbleweed | Bare |
| Cook et al. (1994) | Sand | 0.187 | Borrika (BUF 18) | 60 | −35.1 | 140.1 | Chloride method | 340 | 1800 | Eucalyptus | Grassland |
| Cook et al. (1994) | Sand | 0.245 | Borrika (BUF 15) | | −35.1 | 140.1 | Chloride method | 340 | 1800 | Eucalyptus | Grassland |
| Cook et al. (1994) | Sand | 0.245 | Borrika (BEM 35) | | −34.3 | 139.6 | Chloride method | 340 | 1800 | Eucalyptus | Grassland |
| Cook et al. (1994) | Sand | 0.245 | Borrika (BEM 28) | | −35.1 | 140.1 | Chloride method | 340 | 1800 | Eucalyptus | Grassland |
| Cook et al. (1994) | Sand | 0.245 | Borrika (BRC 02) | | −35.1 | 140.1 | Chloride method | 340 | 1800 | Eucalyptus | Grassland |
| Cook et al. (1994) | Sand | 0.245 | Borrika (TATIARA) | | −35.1 | 140.1 | Chloride method | 340 | 1800 | Eucalyptus | Grassland |
| Cook et al. (1994) | Sand | 0.245 | Borrika (BINNUM) | | −35.1 | 140.1 | Chloride method | 340 | 1800 | Eucalyptus | Grassland |
| Cook et al. (1994) | Sand | 0.245 | Borrika (JOANNA) | | −35.1 | 140.1 | Chloride method | 340 | 1800 | Eucalyptus | Grassland |
| Cook et al. (1989) | Loam | 0.216 | | | −34.6 | 142.8 | Chloride method and electromagnetic techniques | 312 | 1800 | Eucalyptus | Cropland |
| Cook et al. (1989) | Loam | 0.223 | | | −34.6 | 143.6 | Chloride method and electromagnetic techniques | 322 | 1800 | Eucalyptus | Cropland |
| Cook et al. (1989) | Sand | 0.244 | | | −35.1 | 140.1 | Chloride method and electromagnetic techniques | 340 | 1800 | Eucalyptus | Grassland |
| Cook et al. (1989) | Sand | 0.245 | | | −35.1 | 140.1 | Chloride method and electromagnetic techniques | 340 | 1800 | Eucalyptus | Grassland |
| Radford et al. (2009) | Clay | 0.363 | Central Queensland, northern Australia (Baralaba) | | −24.3 | 149.8 | Chloride method | 632 | 1548 | Woodland | Cropland |

**Table 3** Data for groundwater recharge rate study *(Continued)*

| | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Radford et al. (2009) | Clay | 0.345 | Central Queensland, northern Australia (Capella) | | −23.1 | 148.1 | Chloride method | 597 | 1638 | Woodland | Cropland |
| Radford et al. (2009) | Clay | 0.345 | Central Queensland, northern Australia (Dysart) | | −22.9 | 148.9 | Chloride method | 580 | 1621 | Woodland | Cropland |
| Radford et al. (2009) | Clay | 0.342 | Central Queensland, northern Australia (Gindie) | | −23.9 | 148.4 | Chloride method | 600 | 1596 | Woodland | Cropland |
| Radford et al. (2009) | Clay | 0.379 | Central Queensland, northern Australia (Jambin) | | −24.3 | 150.4 | Chloride method | 639 | 1548 | Woodland | Cropland |
| Radford et al. (2009) | Clay | 0.408 | Central Queensland, northern Australia (Theodore) | | −24.8 | 150.1 | Chloride method | 700 | 1502 | Woodland | Cropland |
| Radford et al. (2009) | Clay | 0.400 | Central Queensland, northern Australia (Wowan) | | −23.9 | 150.3 | Chloride method | 659 | 1600 | Woodland | Cropland |
| Huang and Gallichand (2006) | Loam | 0.273 | | | 35.3 | 107.8 | Simulation model Simultaneous heat and water transfer (SHAW) | 545 | 810 | Winter wheat | Orchard |
| Kesse et al. (2005) | Sand | 0.162 | El Paso | 30 | 32.5 | −105.3 | Simulation | 224 | 2087 | Bare | Shrubs/brush |
| Keese et al. (2005) | Sand | 0.276 | CPA | 30 | −32.4 | −104.9 | Simulation | 380 | 2169 | Bare | Shrubs/brush |
| Kesse et al. (2005) | Sand | 0.247 | Midland | 30 | −32.5 | −102.5 | 1-D simulation using UNSAT-H | 380 | 2169 | Bare | Shrubs/brush |
| Kesse et al. (2005) | Sand | 0.329 | Lubbock | 30 | 34 | −102.5 | 1-D simulation using UNSAT-H | 474 | 2034 | Bare | Crops |
| Kesse et al. (2005) | Sand | 0.363 | Carson | 30 | 35 | −102.5 | 1-D simulation using UNSAT-H | 497 | 2096 | Bare | Crops |
| Kesse et al. (2005) | Sand | 0.411 | Fisher/Jones | 30 | 33 | −100 | 1-D simulation using UNSAT-H | 620 | 2132 | Bare | Trees |
| Kesse et al. (2005) | Sand | 0.491 | Starr | 30 | 26 | −97 | 1-D simulation using UNSAT-H | 671 | 1788 | Bare | Trees |
| Kesse et al. (2005) | Sand | 0.540 | Bastrop | 30 | 30 | −97.5 | 1-D simulation using UNSAT-H | 810 | 1732 | Bare | Grasses |
| Kesse et al. (2005) | Sand | 0.590 | Parker | 30 | 33.5 | −98 | 1-D simulation using UNSAT-H | 855 | 1819 | Bare | Trees and grasses |
| Kesse et al. (2005) | Sand | 0.625 | Hopkins/Rains | 30 | 33.8 | −96 | 1-D simulation using UNSAT-H | 855 | 1819 | Bare | trees and grasses |

**Table 3** Data for groundwater recharge rate study *(Continued)*

| | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Kesse et al. (2005) | Sand | 0.623 | Upshur/Gregg | 30 | 33.5 | −94 | 1-D simulation using UNSAT-H | 855 | 1819 | Bare | Trees and grasses |
| Moore et al. (2012) | Sand | 0.365 | T1 | 5 | 28.45 | −99.18 | Chloride method | 604 | 1664 | Woodland | Rangeland |
| Moore et al. (2012) | Sand | 0.365 | T2 | 5 | 28.45 | −99.18 | Chloride method | 604 | 1664 | Woodland | Rangeland |
| Moore et al. (2012) | Sand | 0.365 | T3 | 5 | 28.45 | −99.18 | Chloride method | 604 | 1664 | Woodland | Rangeland |
| Moore et al. (2012) | Sand | 0.365 | T1 | 15 | 28.45 | −99.18 | Chloride method | 604 | 1664 | Woodland | Rangeland |
| Moore et al. (2012) | Sand | 0.365 | T2 | 15 | 28.45 | −99.18 | Chloride method | 604 | 1664 | Woodland | Rangeland |
| Moore et al. (2012) | Sand | 0.365 | T3 | 15 | 28.45 | −99.18 | Chloride method | 604 | 1664 | Woodland | Rangeland |
| Moore et al. (2012) | Sand | 0.365 | Z1 | 30 | 28.43 | −99.23 | Chloride method | 526 | 1664 | Woodland | Rangeland |
| Moore et al. (2012) | Sand | 0.365 | Z2 | 30 | 28.43 | −99.23 | Chloride method | 526 | 1664 | Woodland | Rangeland |
| Moore et al. (2012) | Sand | 0.365 | Z3 | 30 | 28.43 | −99.23 | Chloride method | 526 | 1664 | Woodland | Rangeland |
| Moore et al. (2012) | Sand | 0.365 | Z4 | 20 | 28.43 | −99.23 | Chloride method | 526 | 1664 | Woodland | Rangeland |
| Moore et al. (2012) | Sand | 0.365 | Z5 | 20 | 28.43 | −99.23 | Chloride method | 526 | 1664 | Woodland | Rangeland |
| Moore et al. (2012) | Sand | 0.365 | Z6 | 25 | 28.43 | −99.23 | Chloride method | 526 | 1664 | Woodland | Rangeland |
| Moore et al. (2012) | Sand | 0.365 | Z7 | 30 | 28.43 | −99.23 | Chloride method | 526 | 1664 | Woodland | Rangeland |
| Moore et al. (2012) | Sand | 0.365 | Z8 | 30 | 28.43 | −99.23 | Chloride method | 526 | 1664 | Woodland | Rangeland |
| Moore et al. (2012) | Sand | 0.365 | Z9 | 25 | 28.43 | −99.23 | Chloride method | 526 | 1664 | Woodland | Rangeland |
| Leaney et al. (2003) | | 0.380 | U1 | | 28.43 | −99.23 | Chloride method | 500 | 1664 | Malle eucalyptus | Crop |
| Leaney et al. (2003) | | 0.340 | U2 | | −36.5 | 141.3 | Chloride method | 450 | 1664 | Malle eucalyptus | Rangeland |
| Leaney et al. (2003) | | 0.307 | U3 | | −36.4 | 141.4 | Chloride method | 415 | 1664 | Malle eucalyptus | Rangeland |
| Leaney et al. (2003) | | 0.320 | U4 | | −36.2 | 141.8 | Chloride method | 430 | 1664 | Malle eucalyptus | Crop |

**Table 3** Data for groundwater recharge rate study *(Continued)*

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| Leaney et al. (2003) | 0.332 | U5 | −36.1 | 141.1 | Chloride method | 450 | 1664 | Malle eucalyptus | Crop |
| Leaney et al. (2003) | 0.310 | U6 | −36 | 140.9 | Chloride method | 430 | 1664 | Malle eucalyptus | Crop |
| Leaney et al. (2003) | 0.300 | U7 | −35.5 | 140.8 | Chloride method | 120 | 1664 | Malle eucalyptus | Crop |
| Leaney et al. (2003) | 0.268 | U8 | −35.4 | 140.8 | Chloride method | 380 | 1664 | Malle eucalyptus | Crop |
| Leaney et al. (2003) | 0.240 | U9 | −35 | 140.2 | Chloride method | 340 | 1664 | Malle eucalyptus | Crop |
| | | | −34.9 | 140.1 | Chloride method | | | | |

[a]Obtained from global-PET database (Trabucco and Zomer 2009)

**Table 4** Data for runoff studies

| Author | Year of study | Location of study area | Area coverage ($km^2$) | Period of study (years) | Latitude | Longitude | Method used | Precipitation (mm/year) | Potential evaporation (mm/year) | Type of natural vegetation | Type of vegetation after change | Runoff under natural condition (mm/year) | Runoff under changed condition (mm/year) | Aridity index |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Huang et al. (2003) | 1956 | Loess Plateau, China | 1.15 | 0 | 37.55 | 110.27 | Field measurement | 544 | 890 | Grassland | Afforestation | 29.3 | 18.5 | 0.611 |
| Huang et al. (2003) | 1956–1957 | Loess Plateau, China | 1.15 | 1 | 37.55 | 110.27 | Field measurement | 537 | 890 | Grassland | Afforestation | 14 | 9.7 | 0.603 |
| Huang et al. (2003) | 1956–1959 | Loess Plateau, China | 1.15 | 3 | 37.55 | 110.27 | Field measurement | 551 | 890 | Grassland | Afforestation | 3.1 | 1.5 | 0.619 |
| Huang et al. (2003) | 1956–1962 | Loess Plateau, China | 1.15 | 6 | 37.55 | 110.27 | Field measurement | 521 | 890 | Grassland | Afforestation | 5.5 | 2.4 | 0.586 |
| Huang et al. (2003) | 1956–1963 | Loess Plateau, China | 1.15 | 7 | 37.55 | 110.27 | Field measurement | 474 | 890 | Grassland | Afforestation | 7.5 | 2.6 | 0.533 |
| Huang et al. (2003) | 1956–1965 | Loess Plateau, China | 1.15 | 9 | 37.55 | 110.27 | Field measurement | 370 | 890 | Grassland | Afforestation | 1.9 | 0.5 | 0.415 |
| Huang et al. (2003) | 1956–1967 | Loess Plateau, China | 1.15 | 11 | 37.55 | 110.27 | Field measurement | 541 | 890 | Grassland | Afforestation | 3.8 | 1 | 0.608 |
| Huang et al. (2003) | 1956–1969 | Loess Plateau, China | 1.15 | 13 | 37.55 | 110.27 | Field measurement | 428 | 890 | Grassland | Afforestation | 9.8 | 3 | 0.481 |
| Huang et al. (2003) | 1956–1970 | Loess Plateau, China | 1.15 | 14 | 37.55 | 110.27 | Field measurement | 538 | 890 | Grassland | Afforestation | 7.8 | 2.6 | 0.600 |
| Huang et al. (2003) | 1956–1971 | Loess Plateau, China | 1.15 | 15 | 37.55 | 110.27 | Field measurement | 460 | 890 | Grassland | Afforestation | 18.7 | 4.8 | 0.516 |
| Huang et al. (2003) | 1956–1972 | Loess Plateau, China | 1.15 | 16 | 37.55 | 110.27 | Field measurement | 432 | 890 | Grassland | Afforestation | 1.4 | 0.1 | 0.485 |
| Huang et al. (2003) | 1956–1974 | Loess Plateau, China | 1.15 | 18 | 37.55 | 110.27 | Field measurement | 545 | 890 | Grassland | Afforestation | 18.6 | 6.3 | 0.610 |
| Huang et al. (2003) | 1956–1976 | Loess Plateau, China | 1.15 | 20 | 37.55 | 110.27 | Field measurement | 535 | 890 | Grassland | Afforestation | 5.5 | 1.1 | 0.601 |
| Huang et al. (2003) | 1956–1977 | Loess Plateau, China | 1.15 | 21 | 37.55 | 110.27 | Field measurement | 547 | 890 | Grassland | Afforestation | 14 | 5.6 | 0.614 |
| Huang et al. (2003) | 1956–1979 | Loess Plateau, China | 1.15 | 23 | 37.55 | 110.27 | Field measurement | 390 | 890 | Grassland | Afforestation | 2.4 | 0.5 | 0.438 |
| Huang et al. (2003) | 1956–1980 | Loess Plateau, China | 1.15 | 24 | 37.55 | 110.27 | Field measurement | 528 | 890 | Grassland | Afforestation | 12.4 | 5.4 | 0.593 |
| Huang et al. (2003) | 1956–1980 | Loess Plateau, China | 1.15 | 24 | 37.55 | 110.27 | Field measurement | 556 | 890 | Grassland | Afforestation | 12 | 5 | 0.624 |
| Guo et al. (2014) | | | | | | | | 525.9 | 1103.4 | Grassland | Forest | 68 | 28.6 | 0.476 |
| Thorburn et al. (1991) | | Brigalow, north-eastern Australia | | | −23.3791 | 150.51 | | 699 | 1525 | Forest (Bigalow) | Cropland | 121 | 141 | 0.458 |

**Table 4** Data for runoff studies *(Continued)*

| | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Thorburn et al. (1991) | Brigalow, north-eastern Australia | −23.3791 | 150.51 | | 699 | 1525 | Forest (Bigalow) | Pasture | 121 | 150 | 0.458 |
| Thorburn et al. (1991) | Brigalow, north-eastern Australia | −23.3791 | 150.51 | | 630 | 1525 | Forest (Bigalow) | Cropland | 26 | 51 | 0.413 |
| Thorburn et al. (1991) | Brigalow, north-eastern Australia | −23.3791 | 150.51 | | 630 | 1525 | Forest (Bigalow) | Pasture | 26 | 43 | 0.413 |
| Ruprecht and Schofield (1991a) | Southwest Western Australia | −33.3 | 115.73 | Field measurement | 727 | 1500 | Forest | Bare (53% clearance) | 7.2 | 25.4 | 0.485 |
| Ruprecht and Schofield (1991a) | Southwest Western Australia | −33.3 | 115.73 | Field measurement | 731 | 1500 | Forest | Bare (53% clearance) | 7.2 | 12.3 | 0.487 |
| Ruprecht and Schofield (1991a) | Southwest Western Australia | −33.3 | 115.73 | Field measurement | 768 | 1500 | Forest | Bare (53% clearance) | 7.2 | 46.3 | 0.512 |
| Ruprecht and Schofield (1991a) | Southwest Western Australia | −33.3 | 115.73 | Field measurement | 532 | 1500 | Forest | Bare (53% clearance) | 7.2 | 10.1 | 0.355 |
| Ruprecht and Schofield (1991a) | Southwest Western Australia | −33.3 | 115.73 | Field measurement | 821 | 1500 | Forest | Bare (53% clearance) | 7.2 | 55.9 | 0.547 |
| Ruprecht and Schofield (1991a) | Southwest Western Australia | −33.3 | 115.73 | Field measurement | 676 | 1500 | Forest | Bare (53% clearance) | 7.2 | 24.3 | 0.451 |
| Ruprecht and Schofield (1991a) | Southwest Western Australia | −33.3 | 115.73 | Field measurement | 737 | 1500 | Forest | Bare (53% clearance) | 7.2 | 30.4 | 0.491 |
| Ruprecht and Schofield (1991a) | Southwest Western Australia | −33.3 | 115.73 | Field measurement | 580 | 1500 | Forest | Bare (53% clearance) | 7.2 | 19.8 | 0.386 |
| Ruprecht and Schofield (1991a) | Southwest Western Australia | −33.3 | 115.73 | Field measurement | 558 | 1500 | Forest | Bare (53% clearance) | 7.2 | 8.5 | 0.372 |
| Ruprecht and Schofield (1991a) | Southwest Western Australia | −33.3 | 115.73 | Field measurement | 924 | 1500 | Forest | Bare (53% clearance) | 7.2 | 67.4 | 0.616 |
| Ruprecht and Schofield (1991a) | Southwest Western Australia | −33.3 | 115.73 | Field measurement | 662 | 1500 | Forest | Bare (53% clearance) | 7.2 | 52 | 0.441 |
| Ruprecht and Schofield (1991a) | Southwest Western Australia | −33.3 | 115.73 | Field measurement | 750 | 1500 | Forest (eucalyptus) | Bare (53% clearance) | 8 | 38 | 0.500 |

**Table 4** Data for runoff studies *(Continued)*

| | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Ruprecht and Schofield (1991b) | | Southwest Western Australia | | −33.3 | 115.73 | Field measurement | 720 | 1500 | Forest (eucalyptus) | Bare (38% clearance) | 6 | 13 | 0.480 |
| Zhang et al. (2009) | | Huangfu | 3211 | 39 | 111 | Field measurement | 393.6 | 1678 | Bare | Soil erosion conservation measures | 57.8 | 31.6 | 0.235 |
| | | Gushan | 1304 | 39.5 | 111 | Field measurement | 433.1 | 1803 | Bare | Soil erosion conservation measures | 83 | 43.6 | 0.240 |
| | | Kuye | 9289 | 38.5 | 110 | Field measurement | 399.9 | 1693 | Bare | Soil erosion conservation measures | 84.6 | 58.6 | 0.236 |
| | | Jialu | 1279 | 38 | 110.5 | Field measurement | 407.4 | 1602 | Bare | Soil erosion conservation measures | 82.6 | 38.7 | 0.2543 |
| | | Wuding | 30,111 | 37 | 109 | Field measurement | 397.3 | 1749 | Bare | Soil erosion conservation measures | 50.4 | 34.8 | 0.228 |
| | | Shiwang | 2327 | 35.5 | 110.5 | Field measurement | 536.6 | 1649 | Bare | Soil erosion conservation measures | 42.2 | 23.5 | 0.325 |
| | | Xinshui | 4069 | 36 | 111 | Field measurement | 524.3 | 1680 | Bare | Soil erosion conservation measures | 46.3 | 23.1 | 0.312 |
| | | Sanchuan | 4123 | 36.5 | 112 | Field measurement | 462.8 | 1654 | Bare | Soil erosion conservation measures | 71.1 | 43.7 | 0.279 |
| | | Weifen | 1548 | 38.5 | 112 | Field measurement | 491.6 | 1711 | Bare | Soil erosion conservation measures | 53.3 | 28.7 | 0.287 |
| | | Zhujia | 2956 | 39.5 | 112 | Field measurement | 450.4 | 1664 | Bare | Soil erosion conservation measures | 19.8 | 5.9 | 0.270 |
| | | CSHC | 129,654 | 31 | 109 | Field measurement | 440.7 | 1677 | Bare | Soil erosion conservation measures | 49.6 | 29.8 | 0.262 |
| Freebairn and Boughton (1985) | 1978–1981 | Toowoomba | | −31.9 | 116.05 | Field measurement | 750 | 1405 | Bare | Soil erosion conservation measures | 7.5 | 0.7 | 0.533 |
| | 1978–1981 | Toowoomba | | −31.9 | 116.05 | Field measurement | 750 | 1405 | Bare | Soil erosion conservation measures | 19.6 | 4.6 | 0.533 |
| | 1978–1981 | Toowoomba | | −31.9 | 116.05 | Field measurement | 750 | 1405 | Bare | Soil erosion conservation measures | 62.7 | 29.9 | 0.533 |

**Table 4** Data for runoff studies *(Continued)*

| | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 1978–1981 | Toowoomba | −31.9 | 116.05 | Field measurement | 750 | 1405 | Bare | Soil erosion conservation measures | 128.9 | 62.9 | 0.533 |
| | 1978–1981 | Toowoomba | −31.9 | 116.05 | Field measurement | 750 | 1405 | Bare | Cropland | 7.5 | 0.7 | 0.533 |
| | 1978–1981 | Toowoomba | −31.9 | 116.05 | Field measurement | 750 | 1405 | Bare | Cropland | 19.6 | 7.9 | 0.533 |
| | 1978–1981 | Toowoomba | −31.9 | 116.05 | Field measurement | 750 | 1405 | Bare | Cropland | 62.7 | 34.9 | 0.533 |
| | 1978–1981 | Toowoomba | −31.9 | 116.05 | Field measurement | 750 | 1405 | Bare | Cropland | 128.9 | 111.5 | 0.533 |
| | 1978–1981 | Toowoomba | −31.9 | 116.05 | Field measurement | 750 | 1405 | Cropland | Soil erosion conservation measures | 0.7 | 0.7 | 0.533 |
| | 1978–1981 | Toowoomba | −31.9 | 116.05 | Field measurement | 750 | 1405 | Cropland | Soil erosion conservation measures | 7.9 | 4.6 | 0.533 |
| | 1978–1981 | Toowoomba | −31.9 | 116.05 | Field measurement | 750 | 1405 | Cropland | Soil erosion conservation measures | 34.9 | 29.9 | 0.533 |
| | 1978–1981 | Toowoomba | −31.9 | 116.05 | Field measurement | 750 | 1405 | Cropland | Soil erosion conservation measures | 111.5 | 62.9 | 0.533 |

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OSO, KBB, AZG, and MCR designed the research; all authors contributed to the literature research and analysis and to draft the manuscript. All authors read and approved the final manuscript.

**Competing interests**
The authors declare that they have no competing interests.

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