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# Impacts of land use and land cover change on surface runoff, discharge and low flows: Evidence from East Africa

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ABSTRACT

*Region:* East Africa.
*Focus:* A review of catchment studies (n = 37) conducted in East Africa evaluating the impacts of Land Use and Land Cover Changes (LULCC) on discharge, surface runoff, and low flows.
*New hydrological insights:* Forest cover loss is accompanied by increased stream discharges and surface runoff. No significant difference in stream discharge is observed between bamboo and pine plantation catchments, and between cultivated and tea plantation catchments. Trend analyses show that despite forest cover loss, 63% of the watersheds show non-significant changes in annual discharges while 31% show increasing trends. Half of the watersheds show non-significant trends in wet season flows and low flows while 35% reveal decreasing trends in low flows. Modeling studies estimate that forest cover loss increases annual discharges and surface runoff by 16 ± 5.5% and 45 ± 14%, respectively. Peak flows increased by a mean of 10 ± 2.8% while low flows decreased by a mean of 7 ± 5.3%. Increased forest cover decreases annual discharges and surface runoff by 13 ± 1.9% and 25 ± 5%, respectively. Weak correlations between forest cover and runoff ($r = 0.42$, $p < 0.05$), mean discharge ($r = 0.63$, $p < 0.05$) and peak discharge ($r = 0.67$, $p < 0.05$) indicate that forest cover alone is not an accurate predictor of hydrological fluxes in East African catchments. The variability in these results supports the need for long-term field monitoring to better understand catchment responses and to improve the calibration of currently used simulation models.

## 1. Introduction

The sustainable management of the earth's surface including Land Use and Land Cover Changes (LULCC) remains a critical environmental challenge that society must address (Mustard et al., 2004). Besides ecosystem vulnerability, LULCC are major determinants of global environmental change with potential severe impacts on human livelihoods (Olson et al., 2008). Such changes

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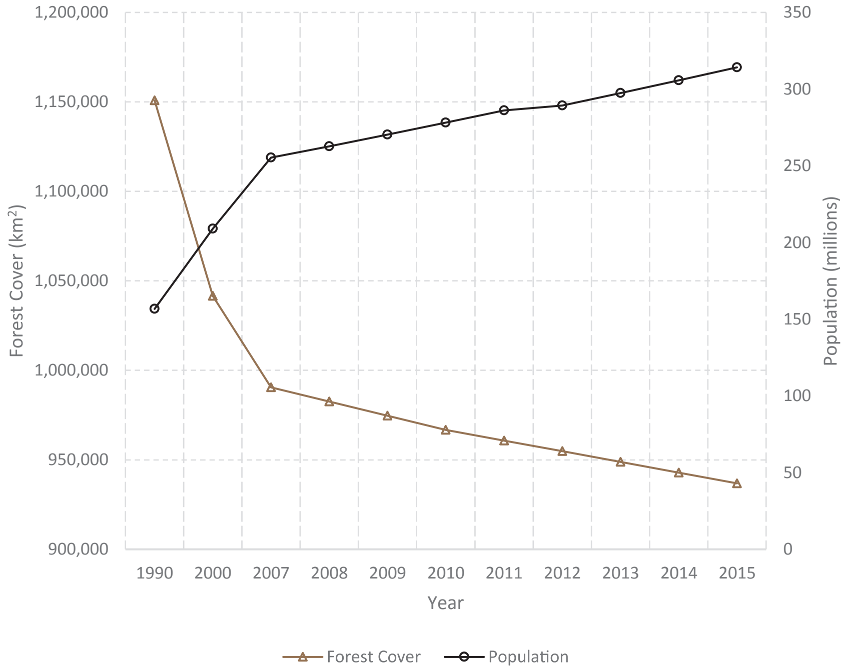


**Fig. 1.** Total forest cover decline and population increase in East Africa between 1990 and 2015.
(Data Source: The World Bank, 2017).

manifest in climatological, hydrological and biodiversity responses. Vitousek et al. (1997) estimated that between 39 and 50% of terrestrial ecosystems have undergone modification due to anthropogenic influence. The main drivers of LULCC are socio-economic development, population expansion, and pressures for land for agriculture (Lambin et al., 2003).

Like the rest of the world, East Africa is not an exception to these dynamics. East Africa depends largely on rain-fed agriculture, which makes rural livelihoods and food security highly vulnerable to shifts in water availability. Land is a critical resource for the livelihood of East Africans and there has been a steady decline in the size of land holdings per household. In order to meet this demand for land, LULCC in this region has resulted in loss of natural forests to human settlements, urban centers, farmlands, and grazing lands (Maitima et al., 2009). Between 1990 and 2015, East Africa forest cover decreased annually by about 1% while human population increased at an average annual rate of 2% (Fig. 1). The main forest types in East Africa that have undergone this decrease include tropical rain forests, tropical dry forests, tropical shrubs, tropical montane forest, and mangrove forests, while there have been concerted efforts to establish plantation forests. LULCC result in a trade-off between provisioning of food and fiber for human consumption, and minimization of negative impacts on other ecosystem services such as water quantity and quality (Mustard et al., 2004). Food production is dependent on water resources and therefore any likely impacts of LULCC on water resources have negative impacts on food production.

Although there has been abundant research on the impacts of LULCC on watershed hydrology, the evidence from the various studies is still contradictory. Malmer et al. (2009) argue that the general notion that "the basics of forest and water relations are well known", does not hold for watersheds with fragmented and dynamic land use patterns such as those observed in the tropical developing world. This means that the variation in catchment characteristics coupled with LULCC increases the uncertainty to find commonalities in observed hydrological signatures attributed to LULCC.

It is commonly argued that forests act both as 'pumps' through enhanced evapotranspiration (ET) rates and as 'sponges' through increased infiltration rates and soil moisture retention (Bruijnzeel, 2004; Arancibia, 2013). Forested watersheds therefore exhibit smaller streamflow rates than watersheds dominated by other managed land uses. Forest cover loss results in changes in albedo, reduction in aerodynamic roughness, reduction of leaf area, and reduction in rooting depth, consequently causing a reduction in ET which subsequently affects streamflow (Costa et al., 2003; Farley et al., 2005). The net effect of forest cover loss is increased water yield (Bosch and Hewlett, 1982). Additionally, a reduction in dry season flow is often cited as a consequence of deforestation (Bruijnzeel, 1988; Arancibia, 2013; Ogden et al., 2013; Liu et al., 2015).

However, despite these general conclusions, which are based on experiments at various spatial scales (e.g. plot, watershed and regional scales), empirical and physically-based (lumped and spatially distributed) modeling, and time series analyses, isolating the impacts of LULCC on water resources in a landscape is problematic because of uncertain interactions of factors driving these effects. Eshleman (2004) stated that these water yield increases associated with forest cover loss depend on a number of factors including the method of forest loss (Beschta, 1998), the extent of forest removal (Bosch and Hewlett, 1982), the rate of plant regeneration impacting ET (Federer and Lash, 1978; Swank et al., 1988), climatic conditions (Chow, 1964; Bosch and Hewlett, 1982; Whitehead and Robinson, 1993), and hydrogeology and catchment physical properties (Likens et al., 1978). The lack of controls in experimental studies, which can be used to attribute observed hydrological changes to causal mechanisms, has also been noted as a confounding factor to these changes (Stonestrom et al., 2009).

The impacts of forest fragmentation and extent of forest loss were demonstrated in a compilation of paired catchment studies in USA where it was observed that timber harvesting could result in a measurable increase in annual water yield in 15% of the catchment area in the Rocky Mountains, while a 50% change is required in the Central Plains of USA (Stednick, 1996). This may mean that there is a threshold in terms of forest fragmentation extent and effects on water yield and flow regimes. Studies in small catchments have indicated that measurable changes in water yield are obtained when there is at least 20% forest cover change (Stednick, 1996).

As a result of the lack of commonalities in observed impacts of LULCC on watershed hydrology, the complex relationships between forests and water continue to be debated (Ellison et al., 2012; Lacombe et al., 2016; Filoso et al., 2017). We focus our study on the East Africa region, which includes most of the Nile basin, an important biodiversity hotspot and the Lake Victoria, the world's largest tropical lake. Even though only approximately 20% of the Lake Victoria water inflows are from rivers draining into the lake (Awange et al., 2008), the ecosystem health of the lake depends on the health and flow dynamics of these rivers. With this context, the main objectives of this study were to review studies that quantify the impacts of LULCC on river discharges, surface runoff, and low flows through a meta-analysis of data for the East Africa region, and to draw over-arching conclusions from these studies and recommendations for future studies.

## 2. Methodology

### 2.1. Study region

This review was based on research data from studies in the East Africa region (Fig. 2), a region that encompasses Kenya, Ethiopia, Somalia, Sudan, Eritrea, Uganda, Rwanda, Burundi, and Tanzania. Most of East Africa is characterized as tropical wet and dry, subtropical dry semi-arid and subtropical dry arid (Köppen and Geiger, 1930). The region has two main climate types depending on rainfall patterns. A tropical type with monomodal summer rainfall pattern is observed to the north of longitude 10° N in Eritrea and south of latitude 6° S, in Tanzania (Le Houérou, 2006). The equatorial climate, experienced in the north of East Africa, is characterized by two rainy seasons irrespective of elevation and aridity class. The highlands in the East Africa region are associated with high rainfalls and (over 2000 mm per year) while some areas, mostly below 1000 m elevation are classified as arid, semiarid, and hyper arid (Le Houérou, 2006).

### 2.2. Research method and literature search

We searched peer-reviewed articles reporting watershed and catchment river discharges and flow regimes under different land use and land cover (LULC). The search was conducted using the search engines Google scholar, and the databases of ISI Web of Science and Scopus. The key words included were "East Africa"; "land use" "land cover"; "deforestation"; "river discharge"; "water yield"; "flow regimes"; "low flow"; "high flow"; and "surface runoff". We only selected cases from East Africa and then obtained additional publications including theses and white papers using the references in the bibliography section of the retrieved articles.

We identified 56 articles from this search. We selected articles that fulfilled the following criteria:

i) LULC change analyzed, i.e. pre- and post- land use reported and corresponding hydrological information reported;
ii) Quantitative data on methods and results reported.

From this screening, 19 articles were unsuitable for our study and subsequently, 37 articles were identified as suitable to evaluate the hydrological impacts of forest cover losses in East Africa. The selected articles also include four master theses, one doctoral theses, and two scientific reports. The articles were further classified as field experiments (n = 6), modeling studies (n = 20) and trend analyses (n = 12). Two case studies (Gebremicael et al., 2013; Wagesho, 2014) included both simulation modeling and trend analyses. The field experiments were sub divided into plot and catchment scale studies. The plot studies (Hurni et al., 2005; Descheemaeker et al., 2006; Girmay et al., 2009) were undertaken in runoff plots measuring 2 × 15 m, 5 × 2m, and 2 × 10 m, respectively. The locations of selected case studies are shown in Fig. 2 while Table 1 shows the main catchment characteristics of each case study. For each study, we took into account the spatial scale, location, method, and categorized this information in an alluvial diagram (Fig. 3) to better show the distribution of these variables among the studies.

### 2.3. Data analyses

The hydrological variables considered in this review are surface runoff, mean annual discharge, peak discharge and low flows. Due to the differences in spatial scale, data from plot studies and catchment scale studies were analyzed separately because at the catchment scale other interacting factors such as slope could impact on measured hydrological variables. Where data was available, Surface Runoff coefficients were extracted for the different land uses in the plot studies. In order to make comparisons between catchments of different sizes and also varying rainfall, where stream discharges were reported, we calculated the ratios of specific discharge to rainfall (SDR, Eq. (1)).

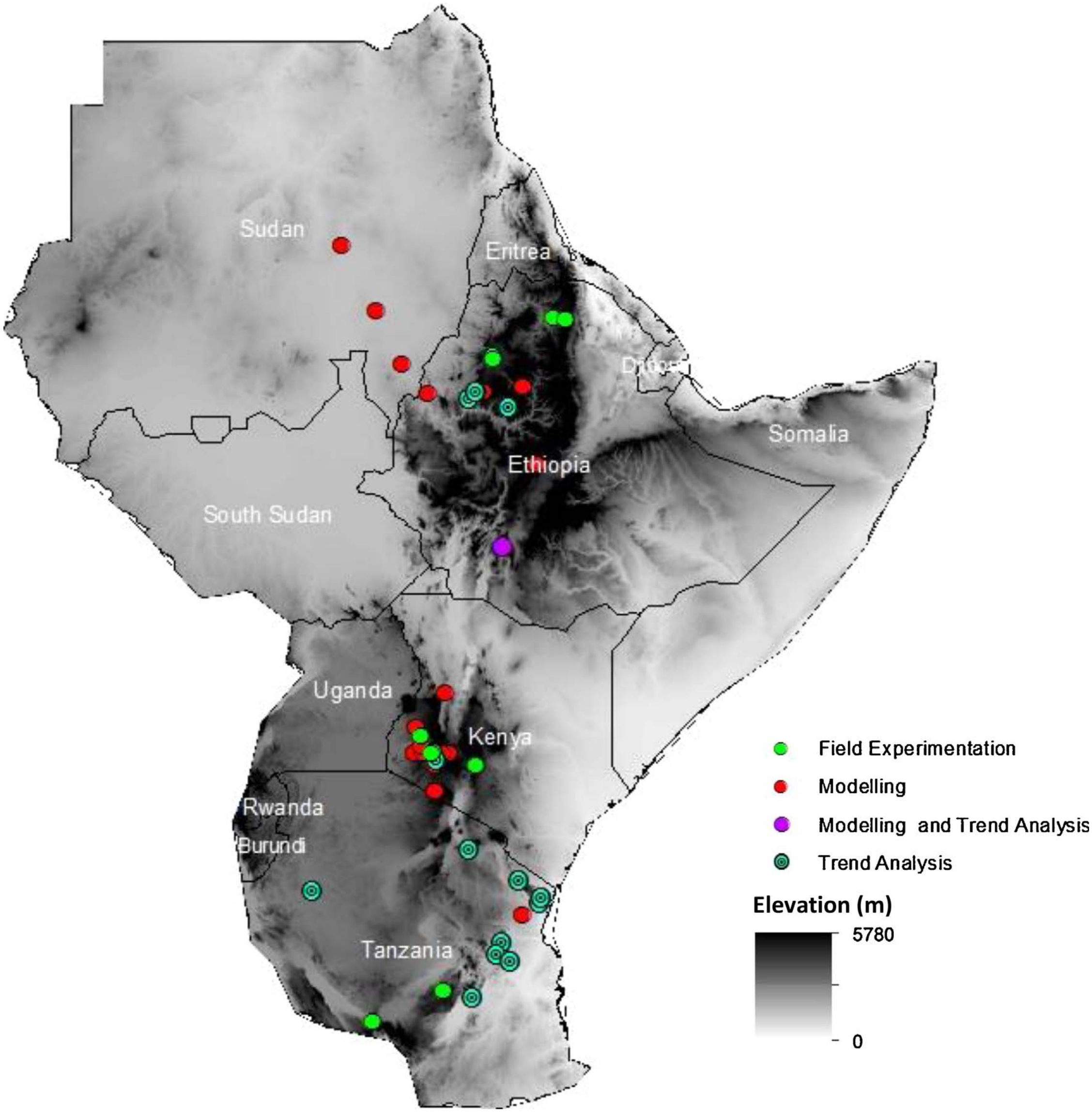


**Fig. 2.** The East Africa region and selected case study sites. Digital Elevation Model derived from 90 m Shuttle Radar Topographic Mission (SRTM).

$$SDR = \frac{Specific\ \ Discharge}{Rainfall} \tag{1}$$

As outlined by Burbank et al. (2012), high SDR indicates that a high fraction of the rainfall becomes stream discharge and less is stored in the soil profile.

We pooled the data from different catchment and plot studies with similar land uses, and the discharge and surface runoff data followed normal distributions (Shapiro-Wilk's test). Subsequently, we used a parametric test (one-way analysis of variance, ANOVA) to test statistical differences between the hydrological variables, i.e. runoff coefficient and discharge, from these catchment studies. Tukey's Honest Significance Difference (HSD) test was used for mean separation when the analysis of variance showed statistically significant differences ($p < 0.05$) between the investigated hydrologic variables. For all hydrological variables, mean values and changes were computed including the Standard Error (SE).

The trend analyses included in this study used the statistical tests – Mann-Kendall and Sen Slope method, to estimate changes in mean annual discharges, peak discharges, and low flows related to LULCC. We collated the data from the selected studies to establish whether common trends in the hydrological variables existed. From the modeling studies, we selected scenarios that represented deforestation or afforestation and calculated the percentage change in the selected hydrological variables.

**Table 1**
Catchment characteristics for the selected case studies.

| | Site Name | Spatial Scale | Country | Coordinates | Study Period | Method[a] | Model | Area (km$^2$) | Climate Zone[b] | Annual Rainfall (mm) | Reference |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 1 | Nyando | Catchment | Kenya | 0.416, 34.833 | 1973–2008 | M | NRCS-CN | 3550 | Af | 1200 | Kundu and Olang (2011) |
| 2 | Nzoia | Catchment | Kenya | 1.500, 35.750 | 1973–2001 | M | SWAT | 12,709 | Aw | 1076–2230 | Githui et al. (2009) |
| 3 | Blue Nile | Catchment | Ethiopia/Sudan | 11.150, 38.180 | 1976–1979 | M | SWAT | 309,875 | Aw | 350–1400 | Sead et al. (2010) |
| 4 | Nyangores | Catchment | Kenya | −0.750, 35.504 | 1996–2008 | M | SWAT | 695 | Aw | 1800 | Mango et al. (2011a) |
| 5 | Amala | Catchment | Kenya | −0.750, 35.754 | 2000–2006 | M | SWAT | 692 | Aw | 1800 | Mango et al. (2011b) |
| 6 | Njoro | Catchment | Kenya | −0.416, 35.833 | 1986–2003 | M | SWAT | 272 | Aw | 1200 | Baldyga et al. (2004) |
| 7 | Molo | Catchment | Kenya | −0.400, 35.666 | 1980–2000 | M | SWAT | 528 | Aw | 1100–2700 | Kirui and Mutua (2014) |
| 8 | Hare | Catchment | Ethiopia | 6.050, 37.616 | 1967–2004 | M | SWAT | 182 | Aw | 750–1300 | Mengistu (2009) |
| 9 | Beles | Catchment | Ethiopia | 10.933, 35.200 | 1985–2005 | M | SWAT | 13,959 | Aw | 1250 | Surur (2010) |
| 10 | Mara | Catchment | Kenya/Tanzania | −1.583, 35.416 | 1973–2000 | M | GeoSFM | 13,750 | Aw | 1400 | Mati et al. (2008) |
| 11 | Sondu | Catchment | Kenya | −0.383, 34.766 | 1961–2010 | M | SWAT | 3500 | Af | 1700 | Rwigi (2014) |
| 12 | Wami | Catchment | Tanzania | −6.500, 37.500 | 1987–2000 | M | SWAT | 43,000 | Aw | 550–1000 | Nobert and Jeremiah (2012) |
| 13 | Upper Blue Nile | Catchment | Ethiopia | 12.000, 37.250 | 1973–2000 | M and TA | SWAT | 176,000 | Aw | 1000–2000 | Gebremicael et al. (2013) |
| 14 | Bilate | Catchment | Ethiopia | 6.632, 37.985 | 1973–2000 | M and TA | SWAT | 5330 | Aw | 1250 | Wagesho (2014) |
| | Hare | Catchment | Ethiopia | 6.159, 37.543 | 1980–2000 | M and TA | SWAT | 167 | Aw | 1250 | |
| 15 | Ngerengere | Catchment | Tanzania | −5.500, 38.167 | 1995–2005 | M | SWAT | 2780 | Aw | 800–1500 | Natkhin et al. (2015) |
| 16 | Njoro | Catchment | Kenya | −0.417, 35.033 | 1986–2003 | M | SWAT | 272 | Aw | 1200 | Baker and Miller (2013) |
| 17 | Mara | Catchment | Kenya/Tanzania | −0.789, 35.445 | 1974–1982 | M | SWAT | 13,750 | Aw | 1400 | Mwangi et al. (2016a) |
| 18 | Upper Gilgel Abbay | Catchment | Ethiopia | 11.000, 36.917 | 1973–2001 | M | HBV | 1656 | Aw | 1550 | Kebede (2009) |
| 19 | Nyando | Catchment | Kenya | −0.184, 34.974 | 1973–2000 | M | HEC-HMS | 3550 | Af | 1300 | Olang and Fürst (2011) |
| 20 | Melka Kuntrie | Catchment | Ethiopia | 8.700, 38.600 | 1986–2003 | M | HBV | 4456 | Aw | 1216 | Getahun and Van Lanen (2015) |
| 21 | Ketar | Catchment | Ethiopia | 8.000, 39.000 | 1985–1995 | M | PRMS | 3220 | Aw | 1100 | Legesse et al. (2003) |
| 22 | Nyangores | Catchment | Kenya | −0.665, 35.462 | 1965–2007 | TA | | 695 | Aw | 1370 | Mwangi et al. (2016b) |
| 23 | Malagarasi | Catchment | Tanzania | 4.800, 31.600 | 1975–2002 | TA | | 80,933 | Bsh | 900 | Kashaigili and Majaliwa (2013) |
| 24 | Weru Weru-Kiladeda | Catchment | Tanzania | −3.500, 36.500 | 1990–2009 | TA | | 164 | Aw/BWh | 1000 | Chiwa (2008) |
| 25 | Chemoga | Catchment | Ethiopia | 10.450, 37.733 | 1960–1999 | TA | | 364 | BWh | 1300 | Bewket and Sterk (2005) |
| 26 | Upper Gilgel Abbay | Catchment | Ethiopia | 10.933, 36.733 | 1973–2005 | TA | | 1656 | Aw | 1550 | Rientjes et al. (2011) |
| 27 | Sigi | Catchment | Tanzania | −5.013, 38.799 | 1957–2004 | TA | | 1436 | Aw | 1800 | Forestry and Beekeeping Division (2005) |
| | Luengera | Catchment | Tanzania | −5.133, 38.725 | 1953–2004 | TA | | 1008 | Aw | 1285 | |
| | Mkomazi | Catchment | Tanzania | −4.463, 38.061 | 1952–2004 | TA | | 4418 | Aw | 790 | |
| | Wami | Catchment | Tanzania | −6.423, 37.533 | 1953–2002 | TA | | 106,250 | Aw | 785 | |
| | Ruvu | Catchment | Tanzania | −6.763, 37.368 | 1952–2004 | TA | | 31,226 | Aw | 1800 | |
| | Kilombero | Catchment | Tanzania | −8.150, 36.633 | 1954–1990 | TA | | 98,853 | Aw | 1300 | |
| 28 | Sigi | Catchment | Tanzania | −5.010, 38.800 | 1957–1989 | TA | | 1436 | Aw | 1800 | Yanda and Munishi (2007) |
| | Ruvu | Catchment | Tanzania | −7.020, 37.800 | 1959–2005 | TA | | 31,226 | Aw | 1800 | |
| 29 | Upper Blue Nile | Catchment | Ethiopia | 12.000, 37.250 | 1963–2004 | TA | | 176,000 | Aw | 1200 | Tesemma et al. (2010) |
| 30 | Abbay | Catchment | Ethiopia | 10.716, 36.500 | 1960–2004 | TA | | 311,000 | Aw | 950 | Gebrehiwot et al. (2013) |
| 31 | Gilgel Abbay | Catchment | Ethiopia | 10.933, 36.733 | 1980–2005 | TA | | 1656 | Aw | 1550 | Enku et al. (2014) |
| 32 | Kericho, | Catchment | Kenya | −0.366, 35.350 | 1958–1964 | FE | | 7 | Af | 1800 | Edwards and Blackie (1981) |
| | Kimakia | Catchment | Kenya | −0.766, 36.733 | 1958–1964 | FE | | 5 | Af | 2300 | |
| | Mbeya | Catchment | Tanzania | −8.833, 33.466 | 1958–1964 | FE | | 0.2 | Aw | 1000 | |
| 33 | Tigray Highlands | Plot | Ethiopia | 13.316, 39.166 | 2003–2004 | FE | | 5 m*2 m | BSh | 700 | Descheemaeker et al. (2006) |
| 34 | Kapchorwa | Catchment | Kenya | 0.166, 35.000 | 2006–2008 | FE | | 0.5 | Af | 2000 | Recha et al. (2012) |
| 35 | Maileba Gum Selassa | Plot | Ethiopia | 13.233, 39.533 | 2006–2007 | FE | | 2 m*20 m | BSh | 588 | Girmay et al. (2009) |
| | | | | | | FE | | | BSh | 452 | |
| 36 | Upper Blue Nile | Plot | Ethiopia | 12.000, 37.250 | 1981–1987 | FE | | 2 m*15 m | Aw | 1200 | Hurni et al. (2005) |
| 37 | Mgera | Catchment | Tanzania | −8.169, 35.424 | 1993–1996 | FE | | 5 | Aw | 1257 | Lorup and Hansen (1997) |
| | Muhu | Catchment | Tanzania | −8.169, 35.424 | 1993–1996 | FE | | 5 | Aw | 1128 | |

(continued on next page)

**Table 1** (*continued*)

| Site Name | Spatial Scale | Country | Coordinates | Study Period | Method[a] | Model | Area ($km^2$) | Climate Zone[b] | Annual Rainfall (mm) | Reference |
|---|---|---|---|---|---|---|---|---|---|---|
| Gendavaki | Catchment | Tanzania | −8.169, 35.424 | 1993–1996 | FE | | 5 | Aw | 1231 | |

[a] M – Simulation Modeling, TA – Trend Analyses, FE – Field Experimentation.
[b] Koppen Geiger classes- Tropical wet and dry (Aw); Tropical wet (Af); Subtropical dry semi-arid (BSh); Subtropical dry arid (BWh).

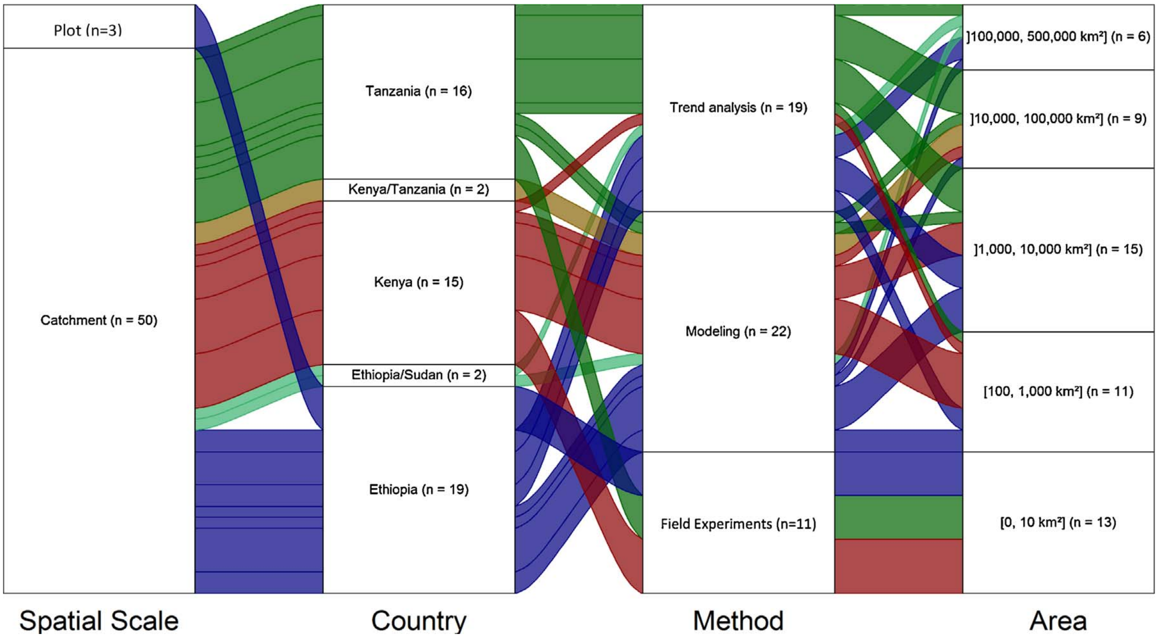


**Fig. 3.** Alluvial diagram showing selected attributes distribution for all case studies.

## 3. Results

### 3.1. Distribution and spatial scale of studies

Fig. 3 is an alluvial diagram that shows the distribution of all the selected studies including spatial scale. There is a strong focus on studies on catchment scale, which are distributed mostly in Ethiopia, Kenya, and Tanzania. While the majority of the trend analyses were conducted in catchments over 1000 km$^2$, the modeling applications were in catchments between 100 and 10,000 km$^2$, and the field experiments in plots and catchments up to 10 km$^2$. In terms of spatial scale and location, the field experiments were the most well distributed, being applied from plot and catchment scales in three different countries. The modeling applications were mostly concentrated in Ethiopia and Kenya, and half of the trend analyses studies were done in Tanzania. Most of the cases studies (n = 28) are in the tropical wet-dry climate (Köppen and Geiger, 1930), which is characterized by distinct wet and dry seasons, and most of the precipitation occurs in the summer season. Six of the case studies fall in the tropical wet climates with mean annual precipitation greater than 1000 mm (mostly in the highlands of Ethiopia and Kenya), while the remainder lie in the arid and semi-arid zones. Results for each of the methods, catchment studies, trend analyses and simulation modeling, are presented in the following sections.

### 3.2. Catchment and plot experiments

Compared to other regions of the world, e.g. South America, relatively few long term field studies have been undertaken in East Africa, specifically analyzing the impacts of LULCC on watershed hydrology. Fig. 4 shows runoff coefficients from plot studies in this region. Four main land uses were identified in the plot studies, namely cultivated lands on which different crops were grown, plots with eucalyptus trees (plantation), plots in grazing lands used as pastures for livestock, and plots in natural forests. The results show significantly lower runoff coefficients for forested plots compared to plots under cultivation and grazing lands. Mean runoff coefficient and SE in natural forest plots were 0.39 ± 0.17% while in the cultivated areas they were 22 ± 3.2%. There is no statistically significant difference in runoff coefficients from eucalyptus plantation plots and grazing plots. The runoff coefficient in the eucalyptus plantation forest were significantly lower (mean 9.6 ± 2.1%) than in both cultivated and grazing plots but higher than for natural forest plots.

In the catchment scale studies, the main land uses that were compared included pine tree (*Pinus patula* sp.) plantation, bamboo forest (*Arundinaria alpine* sp.) with scattered forest tree species, tea plantation, cultivated catchment with annual and perennial crops and forest catchment covered with native broad leafed montane forest. SDR values for five land uses in catchments in Kenya and Tanzania are shown in Fig. 5. Across the datasets from the five catchments, there is no statistically significant difference in SDR between pine plantation and bamboo catchments. A significant difference was observed in SDR between tea plantation and pine plantation catchments. The mean SDR in the natural forest catchments was the lowest (0.28 ± 0.02) and differed significantly from all other catchments. However, it is interesting to note that the mean SDR for cultivated catchments was significantly lower than for both bamboo and pine plantation catchments. There is a significant decrease in SDR with plantation age in the pine plantation ($r^2 = 0.38$, $p < 0.01$) while in the tea plantation the decrease in SDR with age is non-significant (Fig. 7).

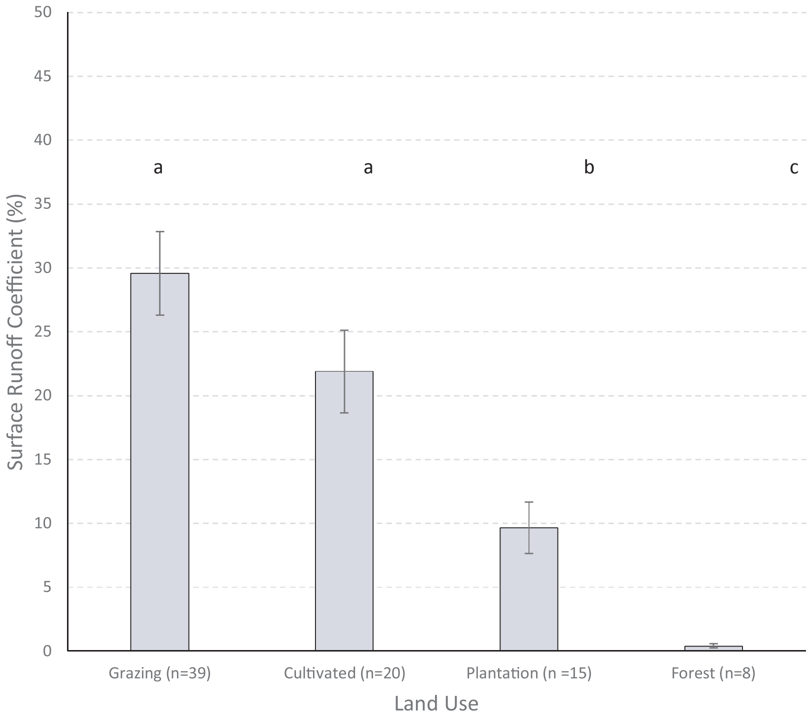


**Fig. 4.** Runoff coefficients derived from data for plot studies.
(Data sources: Hurni et al., 2005; Descheemaeker et al., 2006; Girmay et al., 2009)

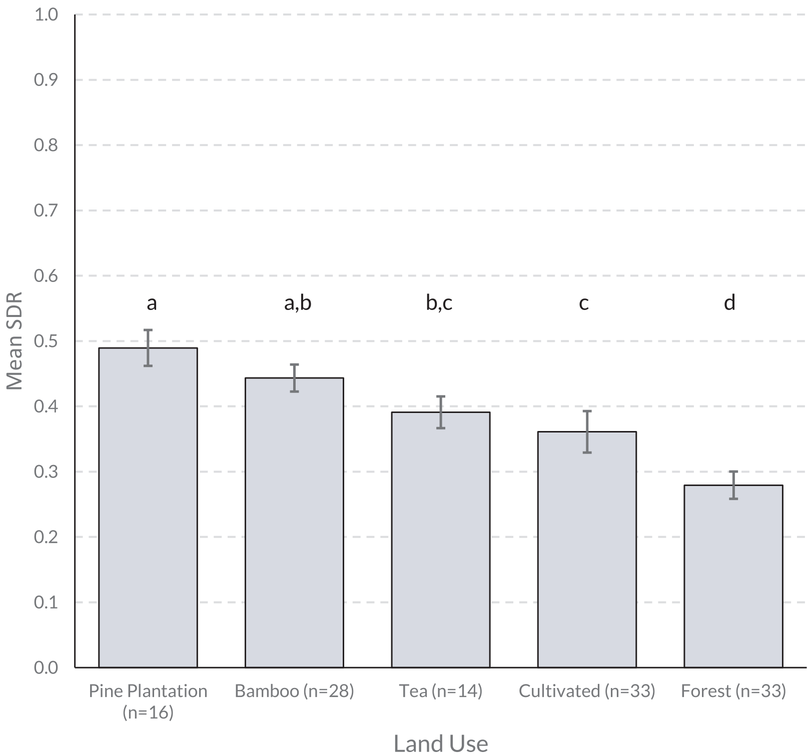


**Fig. 5.** Mean SDR from catchment scale studies in Kenya and Tanzania.
(Data Source: Edwards and Blackie, 1981; Lorup and Hansen, 1997; Recha et al., 2012).

**Table 2**
Trend analyses studies and statistical significance of observed trends from selected East Africa case studies.

| | Country | Watershed | Area ($km^2$) | Land use and land cover change | Study period | Rainfall | Annual Discharge | Dry Season Flows | Wet Season Flows | Reference |
|---|---|---|---|---|---|---|---|---|---|---|
| 1 | Kenya | Nyangores | 690 | 50% forest cover loss (1976–2006) | 1965–2007 | * | + | * | * | Mwangi et al. (2016a,b) |
| 2 | Tanzania | Sigi | 1436 | 39% forest cover loss (1955–1995) | 1957–1990 | ns | ns | ns | ns | Yanda and Munishi (2007) |
| | | Ruvu | 31,226 | 12% forest cover loss (1970–2000) | 1952–2005 | ns | ns | ns | ns | |
| 3 | Tanzania | Weru-Weru | 164 | 12% forest cover loss (1990–2008) | 1990–2008 | − | + | − | − | Chiwa (2008) |
| 4 | Tanzania | Sigi | 1436 | Forest cover decreased from 85% to 52% (1955–1995) | 1955–1990 | ns | ns | ns | ns | Forestry and Beekeeping Division (2005) |
| | | Luengera | 1008 | At least 39% deforestation (1950–1980) | 1968–1990 | ns | ns | ns | ns | |
| | | Mkomazi | 3341 | At least 39% deforestation (1950–1984) | 1962–1984 | ns | ns | ns | ns | |
| 5 | Tanzania | Malagarasi | 80,933 | 0.4% Forest cover increase (1984–2001) | 1975–2002 | ns | ns | − | * | Kashaigili and Majaliwa (2013) |
| 6 | Ethiopia | Gilgel Abbay | 1300 | 34% forest cover loss (1973–2001) | 1973–2001 | − | − | ns | − | Rientjes et al. (2011) |
| 7 | Ethiopia | Chemoga | 1300 | 19% forest cover increase (1957–1982) | 1960–1999 | − | * | − | ns | Bewket and Sterk (2005) |
| 8 | Ethiopia | Bilate | 5330 | 68% forest cover loss (1976–2000) | 1971–2005 | ns | ns | + | + | Wagesho (2014) |
| | | Hare | 167 | 39% forest cover loss (1970–2000) | 1970–2007 | ns | ns | + | ns | |
| 9 | Ethiopia | Upper Blue Nile | 175,300 | Cropland increased from 27% to 48% of catchment (1970–2005) | 1970–2005 | ns | + | − | + | Gebremicael et al. (2013) |
| 10 | Ethiopia | Gilgel Abay | 1656 | Forest cover area decrease from 51% in 1973 to 17% of the catchment in 2001 | 1980–2005 | | * | − | | Enku et al. (2014) |
| 11 | Ethiopia | Koga | 260 | Forest cover decrease from 16% of catchment in 1957 to 2% in 1982 | 1957–1986 | ns | ns | ns | ns | Gebrehiwot et al. (2013) |
| 12 | Ethiopia | Upper Blue Nile | 175,300 | | | | | | | Tesemma et al. (2010) |
| | | Bahir Dar | | No data | 1964–2005 | ns | + | ns | + | |
| | | Kessie | | No data | 1964–2005 | ns | + | + | + | |
| | | El Diem | | No data | 1964–2005 | ns | ns | − | + | |

+/−: significant increasing/decreasing; ns: non-significant; *: no information.

### 3.3. Trend analyses

Table 2 summarizes the results from 12 studies in East Africa in which trend analyses were performed to estimate the effects of LULCC on mean annual discharge and flow regimes in 18 catchments. Results from five catchments show statistically significant ($p < 0.05$) increasing trends in mean annual discharge with forest cover loss. However, for the three catchments in the Upper Blue Nile watershed (Tesemma et al., 2010), there is relatively limited information about how forest cover has changed over the years, even though trend analyses showed increasing trends in mean annual discharge. The watershed is dominated by small holder agriculture with about 1% forest cover mainly in the uplands. One catchment (Gilgel Abbay) showed decreasing trends while ten catchments exhibited non-significant trends in mean annual discharge. For dry season flow, six catchments showed significantly decreasing trends while three catchments exhibited significantly increasing trends. Eight catchments showed non-significant trends in dry season flows. Increasing trends in wet season flows were observed in five catchments while eight catchments exhibited non-significant wet season flow trends. Except for three catchments (Gilgey Abbay, Chemoga and Weru-Weru), which show decreasing trends, there is no significant trend in annual rainfalls. As summarized in Table 2, the catchments showed varied level of natural forest loss for the periods in which the trend analyses were applied.

### 3.4. Modeling studies

The use of hydrological models has been the main method for estimating impacts of LULCC on catchment hydrology in East Africa. Most of these modeling studies used the Soil Water Assessment Tool – SWAT (Arnold and Fohrer, 2005), while the other studies were conducted using the Hydrologic Modeling System – HEC-HMS (U.S. Army Corps of Engineers), Curve Number method (USDA Soil Conservation Service), the *Hydrologiska Byrans Vattenbalansavdelning* (HBV) model (Bergström, 1976) and Geospatial Stream Flow Model-GeoSFM model (Artan et al., 2008).

From the modeling case studies, mean annual discharge, water yield, surface runoff, peak discharges, and low flows/base flow data were extracted, for different land cover scenarios, where the information was available. For each of these studies, the model was calibrated and validated using measured hydrological data from the study catchments with the measured data sets divided into two: half the time series data used for calibration and the other half for model validation. Alternative LULC scenarios were developed from current LULC that were used as baseline. Each of these alternative scenarios, mostly assuming deforestation without climate changes, were used as input to the models and mean annual discharge, surface runoff, peak discharges, and low flows were simulated for each watershed. For the selected LULC scenarios and simulated water balance components we summarize the change in surface runoff and annual stream discharges (Fig. 7) and low flows and peak discharges (Fig. 8). In the results, we refer to percentage changes in surface runoff, peak discharges, low flows and mean annual discharge in comparison with baseline/initial values. The forest cover change represented in Figs. 7 and 8 include both forest losses, negative change, and forest gain (positive change).

The summarized results (Fig. 7) show that forest cover loss leads to increases in surface runoff (4–90%). Results from case studies with forest cover increase scenarios show that surface runoff decreased with forest cover gain. However the magnitude of surface runoff increase and decrease varied among the watersheds. Similar results were also observed for mean annual discharge with the exception of Mango et al. (2011a) and Nobert and Jeremiah (2012), where increases in annual discharge where obtained.

With the exception of one scenario in the study by Sead et al. (2010), results show that forest cover loss leads to increases in peak discharges while afforestation results in a reduction in peak discharge (Fig. 8). Deforestation resulted in decreased low flows (up to 46%) with the exception of the study by Githui et al. (2009) in which an increase in low flows was observed following forest cover loss. Forest cover change had an inverse relationship with mean annual discharge ($r = 0.63$, $p < 0.001$), peak discharge ($r = 0.67$, $p < 0.05$) and surface runoff ($r = 0.42$, $p < 0.05$) while low flows did not exhibit a significant relationship with forest cover change (Fig. 8).

## 4. Discussion

### 4.1. Emerging patterns and comparison with results from other regions

Our analyses of the literature show that there are emerging patterns on the effects of forest cover loss on hydrological responses. Results from this review demonstrate that forest cover loss results in stream flow increases (mean 16 $\pm$ 5.5% from modeling studies in East Africa). Field experiments also show higher discharge in cultivated catchments in comparison to forest catchments (Fig. 5). These results agree with observations in other tropical environments including Tomasella et al. (2009) and Hayhoe et al. (2011); the latter observed higher water yields from catchments where natural forest was replaced by croplands in Brazil. Similar results after conversion of natural forests to soya fields in Brazil were attributed primarily to decreased ET (Dias et al., 2015). In comparison to forest landscapes, vegetation in cultivated and disturbed natural ecosystems generally have lower leaf area indices and shallow root depths causing reduced evapotranspiration (Foley et al., 2003). Jackson et al. (2005), in a global meta-analysis of 504 catchments, found that stream flow was reduced by 50%, while Farley et al. (2005) observed water yield reductions ranging from 30 to 60% in moderate and high rainfall areas, as a result of forest cover increase. The flow reductions are also attributed to the increased buffering capacity (increasing amount of water retained in the watershed) as forest cover increases as well as increased ET rates. From the catchment studies in Kenya and Tanzania, the general conclusion that forest cover loss results in increased discharge is not prominently evident. Notably, the discharge from pine plantation catchment was higher than from cultivated catchments.

The results of this review of stream discharge trend analyses indicate a lack of significant trends in stream flow regimes due to

LULCC. These results do not support the common finding that deforestation results in increased streamflow as concluded in some studies (e.g. Brown et al., 2005; Farley et al., 2005). In studies done in tropical watersheds in the Amazon basin, significant trends in the measured hydrological parameters due to LULCC, were detected only in low order streams (Rodriguez et al., 2010). Increased peak discharges with forest losses were observed in these low order streams and the results agreed with those obtained in paired catchment studies in which there are minimum confounding factors influencing trends. Large watersheds may have a relatively high buffering capacity that masks forest cover effects on hydrological regimes, because of a large number of interacting factors such as soil type and slope (Bi et al., 2014).

Surface runoff increases were also observed with forest cover loss in field experimentation with significantly higher runoff in cultivated plots than in forest plots (Fig. 4). This is also reflected in the positive change in surface runoff with deforestation scenarios (Fig. 7) in the modeling studies. These results agree with those by Nóbrega et al. (2017) who compared catchments with contrasting land use (pasture vs. dry forest) in the Mato Grosso state, Brazil. Similarly, Guzha et al. (2015) also observed higher runoff in a pasture catchment compared to natural forest catchment in field experiments in the Southern Amazon region. Simulation modeling in Suiá-Miçu River basin, Brazil, estimated that deforestation in this basin resulted in a 6% increase in surface runoff (Maeda et al., 2009). Similar results, increases in surface runoff due to forest cover loss, have also been observed by Valentin et al. (2008), in 27 catchments in South East Asia.

As shown in Fig. 8, deforestation resulted in decreased low flows (mean reduction of 7 ± 5.3%). This conforms to the conclusion by Bruijnzeel (2004), noting that deforestation and cropland expansion leads to changes in soil hydro-physical conditions in tropical forests, which subsequently results in reduced low flows due to reduced infiltration and, groundwater recharge. Bewket and Sterk (2005), also attributed the decreasing trend in the mean dry season low flows to the expansion of eucalyptus plantations, given the high water use of eucalyptus species (up to 7.00 mm/day, Albaugh et al., 2013), which reduces the water stored in the soil profile and ultimately reduces base flow (Bosch and Hewlett, 1982; Scott and Lesch, 1997). The modeling study by Githui et al. (2009) estimated increases in low flows with forest cover loss. Mohns (n.d.) asserts that when there is no soil degradation, such as changes in infiltration properties, deforestation could lead to increased dry season flows due to lower plant water use. Studies including Eisenbies et al. (2007) and Webb et al. (2007) also found similar results with increased dry season flows due to deforestation and attributed this increase to a reduction in ET. According to Liu et al. (2015), changes in low flows involve both vegetation and soil changes, and therefore the degree of soil disturbances after deforestation finally determines the responses of low flows.

The variability in increases in surface runoff (45 ± 14%) and annual discharge (16 ± 5.5%), in Fig. 7, peak discharge (10 ± 2.8%), and reduction in low flows (7 ± 5.3%), Fig. 8, from our case studies, agrees with the results of Fritsch (1990) in a tropical catchment in South America (French Guiana), in which they observed increases in peak flows after deforestation ranging from 17 to 166%. Stednick (1996) noted that changes are less pronounced when less than 20% of the basin area has been deforested.

Regression analyses of the collated data from modelling studies (Fig. 9), showing weak associations between LULCC and the four variables (surface runoff, mean annual discharge, peak discharge and low flows) agree with results by Stednick (1996) and MacDonald and Stednick (2003) in conifer forest watersheds in USA. Our results support the view by van Noordwijk et al. (2017) that the percentage of forest cover in a watershed is probably not a good metric on its own to infer conclusions on water related ecosystem services provided by a watershed.

### *4.2. Factors influencing LULCC impacts*

Results from this study, for all three methods, show relatively wide variability in magnitudes of hydrological changes (cf. Figs. 7 and 8). Interactions between diverse site-specific factors (e.g. catchment slope, soil infiltrability and vegetation age influence) mean that standardized watershed responses will hardly be obtained. Soil depth plays an important role in influencing how watershed discharge responds to LULCC (FAO, 2008). The insignificant increase in discharge in Mbeya under cultivation was attributed to the stability of the volcanic ash derived soils in the wet season which impacts runoff generation processes. The clay soils in the Kericho catchment (Edwards and Blackie, 1981) coupled with the very low hydraulic conductivity of *ca* 0.05 cm $h^{-1}$ (Obieiro, 1996) may have contributed to the reduced infiltration rates and enhanced surface runoff and storm discharge (cf. Fig. 5) and, thus there are no significant differences in discharge between bamboo, pine and tea catchments. The deep and porous soils at Kimakia, Kenya, with the very high available water capacities of up to 765 mm in the top 3 m soil layer (Edwards and Blackie, 1981), coupled with the canopy cover establishment may also have contributed to the significant reduction in discharge over time.

Another factor emerging from this study is the influence of precipitation events. The high SDR for the tea catchment have been attributed partly to the high rainfall intensities. Frequent high intensity rain storms in already saturated soils result in high surface runoff rates and subsequently higher discharges irrespective of full plant canopy cover (FAO, 2008). The importance of rainfall event characteristics was also noted by Niehoff et al. (2002), who concluded that LULCC impacts under small-scale convective storm events with high precipitation intensities are more pronounced than for long-lasting advective rainstorms. The high discharge in Kericho catchment indicates that at high rainfall intensities, the impacts of forest cover in reducing stream flows are masked. Romero et al. (2016) also concluded that the effect of forest cover on peak discharges becomes insignificant with increase in size of hydrological event. The Kericho experiments were performed in periods with generally very high rainfall, averaging 2000 mm/year. In a study in Northern China, Wang et al. (2011) observed that runoff was influenced by altitude, annual precipitation, forest cover and potential ET, and concluded that geographic differences could buffer the true role of forests in runoff generation in a catchment. Forests tended to be located at higher altitudes with steeper slopes, higher precipitation, lower ET and, therefore, higher surface runoff volumes.

According to the National Research Council (2008), the spatial arrangement of deforestation or locations of deforestation zones within a watershed (forest fragmentation) also affects the watershed responses. This is reflected in the modeling study by Mango et al.

(2011a,b), in the Mara basin, which showed that 100% forest cover loss (248 $km^2$) in the uplands of the Nyangores watershed had minimal impacts on annual water yield (+0.3% increase). Mwangi et al. (2016a) also highlighted the influence of spatial location and extent of LULCC (forest fragmentation) on magnitude of change in hydrology. This has also been observed in other regions. Ziegler et al. (2001) and Salemi et al. (2013) observed that forest fragmentation reduces flow attenuation resulting in increased dominance of overland flow paths in tropical catchments in Thailand and Brazil, respectively, leading to higher surface runoff. Our review also showed relatively weak correlations between LULCC and changes in the hydrological components. According to Merz and Blöschl (2009), the poor correlation of forest cover loss and flow regimes, especially peak discharge and low flows, can be attributed to spatial scale because at larger scales some of the catchment responses are likely to average out over the catchment area. The weak relationship could also be a result of unaccounted water abstractions, which are often associated with urbanization. Beck et al. (2013) and Le Tellier et al. (2009) found no significant relationship between low flows, runoff and cloud forest cover, which agree with the lack of significant correlation found for low flows in the modeling studies included in our analyses. Gaál et al. (2012) found similar results and attributed this observation to the influence of other confounding catchment variables such as watershed topography and geology, which influences the subsurface water fluxes as catchments respond non-linearly to changes in land use.

Watershed area (spatial extent) also impacts the observed changes in hydrologic variables due to LULCC. Large watersheds exhibit a relatively high buffering capacity that masks the effects of LULCC (Bi et al., 2014). In our analyses, most of the watersheds that show no significant trends in the measured hydrological variables have catchment areas greater than 1000 $km^2$. This may support the conclusion stated by Rodriguez et al. (2010) that the influence of LUCC on river hydrology diminishes with catchment scale, compared to the influence of the interactions between areas of different LULCC and other factors (e.g. population density and irrigation practice). Bloschl et al. (2007) also note that at the large scale, LULCC impacts are difficult to verify because of long time lags between cause and effect, and the influence of other confounding factors including the connectivity of stream networks and flow paths in catchments which may influence how LULCC translates into hydrological responses. Lack of a clear pattern of significant trends in stream flow regimes with LULCC can be attributed to the influence of spatial scale as in most of the analyses in this review, the trends are quantified on discharge data measured only at the catchment outlet. There are no trend analyses at the sub catchment scales, likely due to missing observational data. Hall et al. (2014) suggest that instead of analyzing a single record, an analysis of an ensemble of records in a particular watershed can increase the power of detection by reducing sampling uncertainty.

Our analyses also shows that even after deforestation, with vegetation regeneration, stream flows may decrease (Eucalyptus plantation, Fig. 6) or show no significant change (tea plantation, Fig. 6). The significant reduction in discharge with time in the pine tree plantation at Kimakia can be attributed to the pine tree canopy development over the years, i.e. increase in leaf area and rooting depth, leading to higher ET rates as noted in other studies such as Simic et al. (2014) and Engel et al. (2002). Results from paired watershed studies in tropical montane cloud forests (TMCF) in Latin America and the Caribbean show that the effects of deforestation on hydrological fluxes are most pronounced immediately after clearing and decline in subsequent years (Bruijnzeel, 2001; Ogden et al., 2013). As outlined by Bruijnzeel (1990), similar results were obtained in the sub-humid part of Fiji (Kammer and Raj, 1979) and in South Africa (Bosch, 1979; Van Lill et al., 1980) and they attribute these reductions in flow to increased ET loss from the vegetation. However, from our review, there was no significant reduction trend in observed discharge over time in the tea plantation. The non-significant trend in the tea plantation can be attributed to soil processes. Filoso et al. (2017) highlighted that processes such as infiltration may not recover in such plantations due to factors such as lack of undergrowth vegetation, access road construction, and the inherent vulnerability of tropical soils to compaction and structural changes.

### 4.3. Limitations of the studies and recommendations

The case studies reviewed in this study considered three main methodologies, i.e. field experimebts, trend analyses and, simulation modeling with nearly half of the studies being model applications utilizing the HBV, SWAT, HEC-HMS, GeoSFM models and the CN method. The HBV, SWAT, GeoSFM and HEC-HMS are semi-distributed models, which divide the landscape into sub-catchments or hydrological response units (in SWAT) while the CN method is an empirical model for estimating direct runoff from a landscape. The CN method used by Kundu and Olang (2011) uses a lumped empirical model to quantify total runoff with relatively limited consideration of catchment physiographical factors. Even though they are data intensive, the semi distributed models capture the spatial distribution of the input variables (e.g. land use, soil, and elevation) and therefore improved simulation results in comparison to the lumped CN method.

As outlined by Golmohammadi et al. (2014), models' performances are very site specific and because no model is superior under all conditions, a complete understanding of comparative model performance requires applications under different hydrologic conditions and watershed scales. The clarity and reliability of modeling outputs can be improved by the use of model ensembles instead of the use of single models (Dwarakish and Ganasri, 2015). We therefore recommend inter comparison of models at local scale as simulating efficiency of the models varies depending on uncertainty introduced by calibration strategy, model input and structure and parameterization, among other factors.

In most of the reviewed studies in East Africa, model calibration and validation is based on short-term measured data sets, compromising the utility of the model as a prediction tool. A further caveat of these model applications is that flow components such as surface runoff, interflow and base flow are most often not validated, as measured data for such a validation are completely missing. It is common knowledge in hydrologic modelling that satisfactory modeling results can be achieved with relative unrealistic representation of the hydrological flux pathways in the model. In most SWAT applications, the SCS curve number method is used to estimate surface runoff, which is an empirical method and it is often modified by using calibration methods, in order to correctly simulate the observed discharge. This supports the need for intercomparison of models to enhance credibility of model outputs.

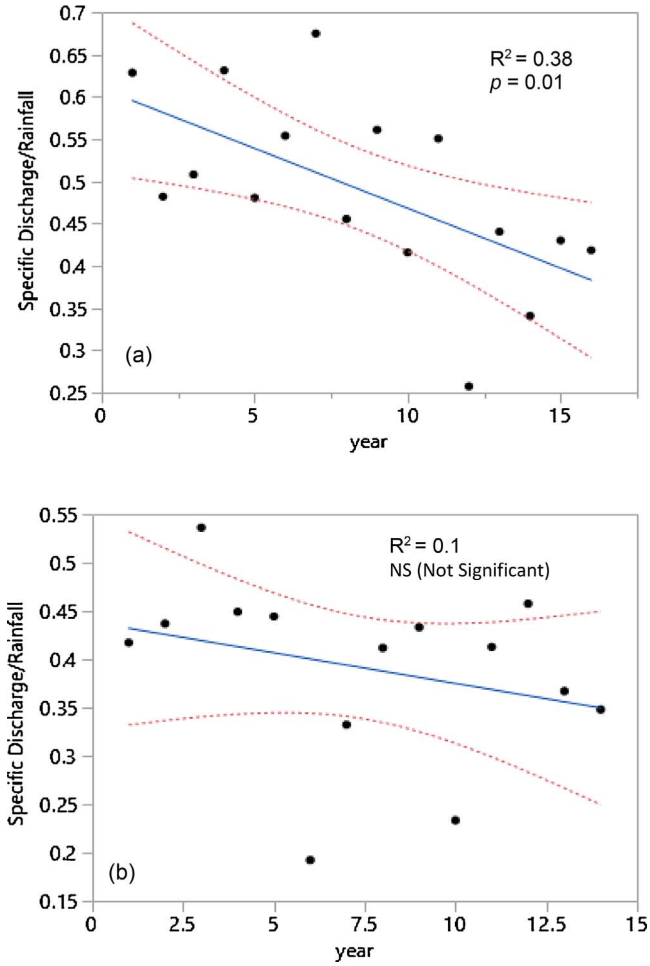


**Fig. 6.** Variation of SDR with time for (a) eucalyptus plantation and (b) tea plantation. Regression equations: (a) Y = 0.6099–0.0142X, (b) Y = 0.438–20.006X. (Data Source: Edwards and Blackie, 1981).

Most often, SWAT is calibrated to a given land use map situation, and then use a new land use scenario and the same calibrated parameters. In most cases this is not feasible, as calibration parameters change as land use changes. Therefore the simulations are basically examining an effect of changes in HRU composition, but not the effect of the LULCC itself. Studies such as Kebede (2009) and Mango et al. (2011a,b) do not report parameter sensitivity and uncertainty analyses and the simulations result in very limited changes in discharge and surface runoff for substantial decreases (up to 100%) in forest cover. Therefore, as revealed in studies e.g., Eckhardt et al. (2003) and Breuer et al. (2006), to improve our understanding of the significance of simulated changes in hydrological parameters due to LULCC, intensive parameter sensitivity analyses need to be carried out to obtain improved model parameterization. Furthermore, as highlighted by van Noordwijk et al. (2017), key credibility questions for models are consistency of numerical results and how sensitive are these results to bias and random error in data sources. Parameter sensitivity analyses can assist to improve credibility of obtained simulation results.

Hydrological impacts are not restricted to the influences of LULCC. Observed signatures are usually a result of the interaction of a wide range of factors including landscape characteristics as well as climate variability and change. Relatively few studies (e.g. Natkhin et al., 2015; Rwigi, 2014) considered the interaction of LULCC and climate. Therefore, further studies should incorporate interactions between LULCC and climate variability into the scenarios simulated in order to achieve reliable assessments of hydrological impacts. Most of the studies in the East Africa region do not evaluate the influence of rainfall event characteristics and therefore undertaking more studies incorporating assessments for different rainfall characteristics will shed light on the associate impacts on surface runoff generation.

Even though deforestation in East Africa is associated with expansion of agriculture and human settlements (Maitima et al., 2009), in the case studies reviewed in this study, there was no information on water extraction for purposes such as irrigation and domestic uses. Such anthropogenic activities influence measured flows at catchment outlets, thus disturbing the signals of deforestation manifested in stream flows. This could explain, partly, the variability in the quantified changes in stream discharges as influenced by forest cover change in East Africa. Beck et al. (2013), in studies in twelve mesoscale humid tropical catchments in Puerto Rico, noted that anthropogenic activities could mask the effects of forest cover change on stream flows due to alterations in stream flow due to

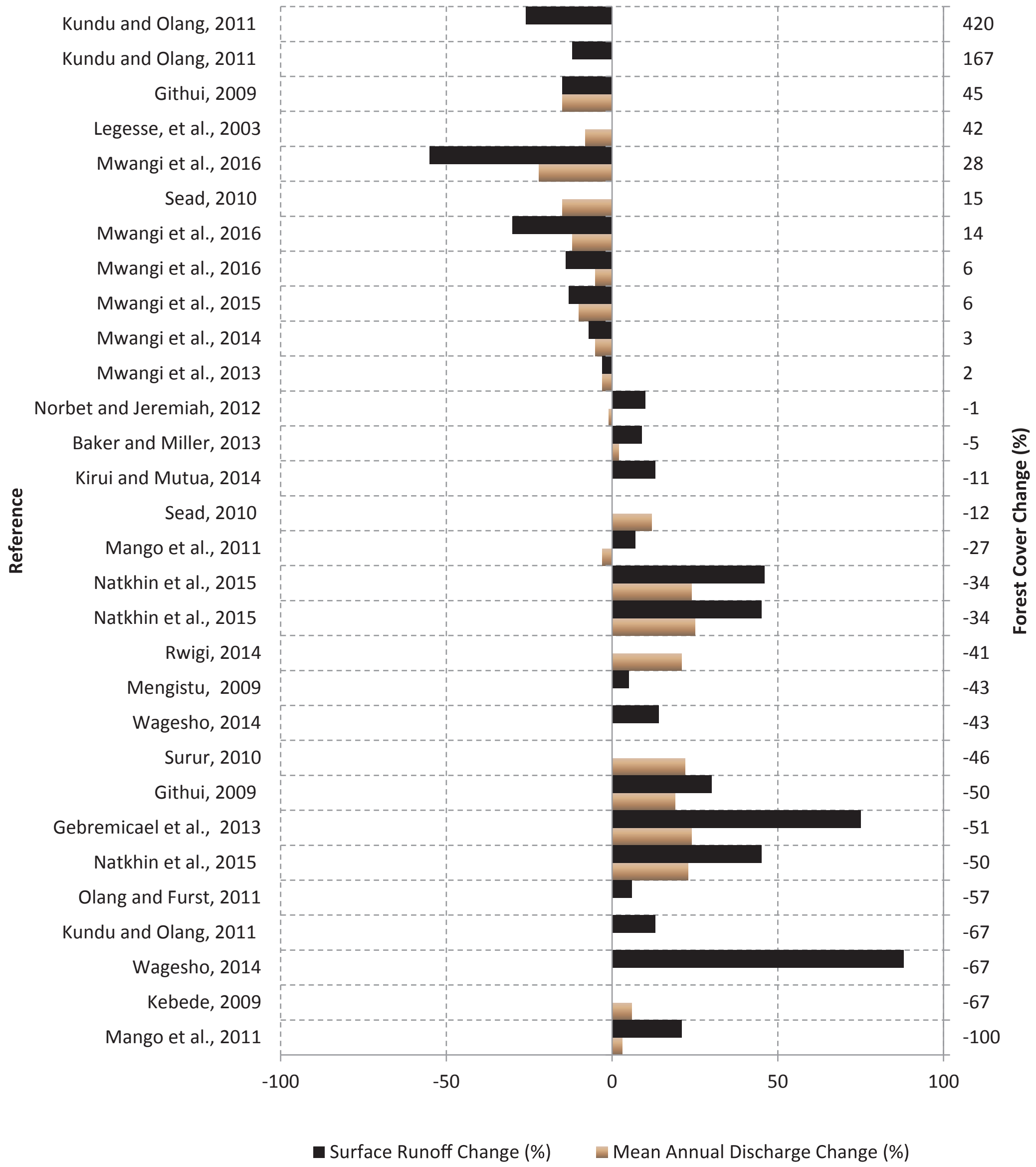


**Fig. 7.** Percentage change in Mean Annual Discharge and Surface Runoff following simulated forest cover change for watersheds in East Africa (lack of a bar means no data available).

extractions for agricultural uses. As there is a multiplicity of factors that affect the observed impacts of LULCC on watershed hydrology, additional studies must also consider the use of Principal Component Analyses (PCA) to investigate which of the important variables play a greater role in the observed changes in hydrological fluxes.

A further recommendation is that more field experimentation and hydrological data collection, for longer temporal scales, in a variety of landscapes should be undertaken to support the refinement of models and reduce the uncertainties in their performance. "Space-for-time" substitution field studies (Pickett, 1989), at different spatial scales, could also be prioritized and these should take into consideration other catchment hydro-physical properties and also examine the impacts of vegetation regeneration. Due to the prominent use of groundwater, for both domestic and use and agriculture, in East Africa (Pavelic et al., 2012) and given the large complexities in hydrogeology (impacts groundwater) in this region, it is recommended that field experimentation and further studies need to include assessment of impacts of LULCC on groundwater fluxes, as base flows and dry season flows are normally generated from groundwater stores. The lack of statistical significance in some of the comparisons for field experimentation (Figs. 4 and 5) in this review can be attributed to small sample sizes e.g. only 14 and 16 measurement points for the tea and pine plant plantation catchments. This supports the need for long-term studies in a variety of landscapes, as most of the direct and indirect hydrological effects of LULCC are not detected in the short-term analyses.

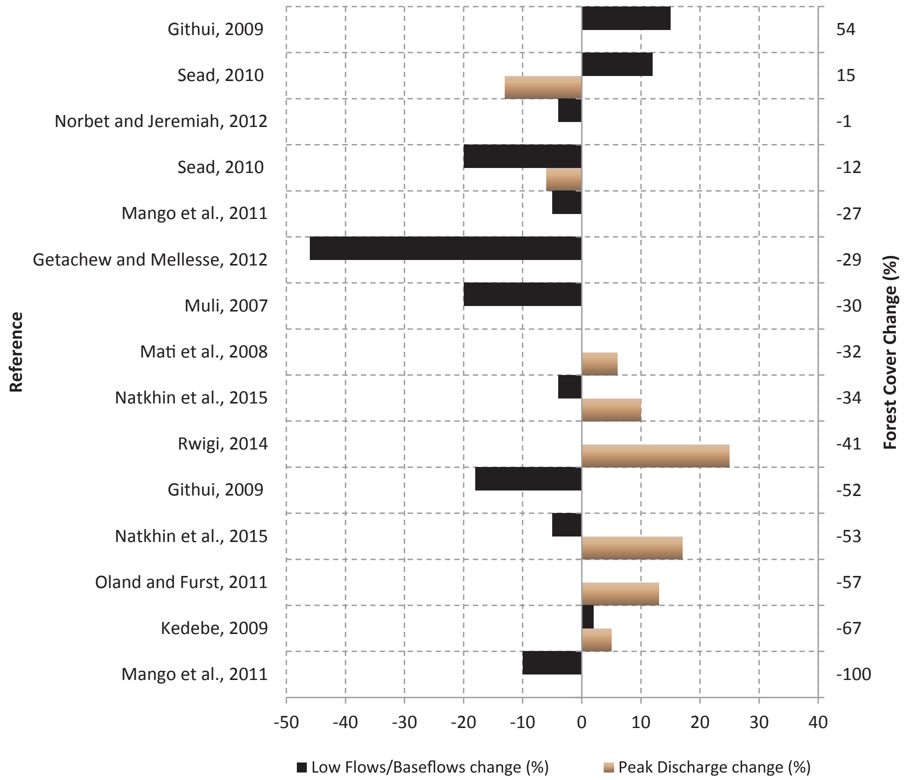


**Fig. 8.** Percentage change in Low Flows and Peak Discharges following simulated forest cover change for selected simulation modeling case studies in East Africa. (lack of a bar means no data available).

## 5. Conclusions

In order to understand the current state of knowledge on forest cover loss and LULCC impacts on catchment hydrology in the East Africa region, we compiled relevant available studies including catchment studies, simulation modeling studies and trend analyses. Although there is lack of information for making uniform quantitative generalizations about the impacts of forest cover loss, it is evident from the presented studies that LULCC impacts hydrological fluxes, specifically discharge, and surface runoff, despite the wide variability in results. Our analyses indicates that while LULCC impacts discharge, surface runoff and low flows, other catchment variables also play significant roles.

From the field studies and hydrologic modeling case studies reviewed in this study, deforestation results in increased surface runoff, mean annual and peak river discharges. While generally this review shows a decrease in low flows with forest cover loss, some studies indicate increased dry season flows. Despite varying scales of deforestation, ranging from 3% to 68% forest cover loss, trend analyses of measured data did not consistently show significant negative or positive trends in discharge and flow regimes. This lack of common trend indicates the need to take into account spatial scale effects on the hydrologic variables.

There is heterogeneity in the magnitudes of the changes in hydrological fluxes with LULCC. This heterogeneity presents a challenge to water resources managers and therefore point to the need for increased local scale research to understand better the effects of land cover transitions on hydrological fluxes. Understanding future LULCC and mitigating their impacts on water resources in specific watersheds will require integration of results from regional scale analyses and those from local scale studies.

This review is based on a relatively limited body of applicable research undertaken in the East Africa region and to better understand the impacts of LULCC on hydrological fluxes in East Africa requires holistic studies that place emphasis on both time and spatial scales and consider ecological scope and climate variability. Therefore, an overall conclusion is that there is still need for more field studies, at longer time scales and variable spatial scales, and integrate these with modeling studies that takes full consideration of parameter uncertainty. For increased clarity, model inter-comparison studies are also recommended to verify their predictive power in relation to spatio-temporal changes in hydrological fluxes, using long term field data for improved model calibration and validation.

## Conflicts of interest

None.

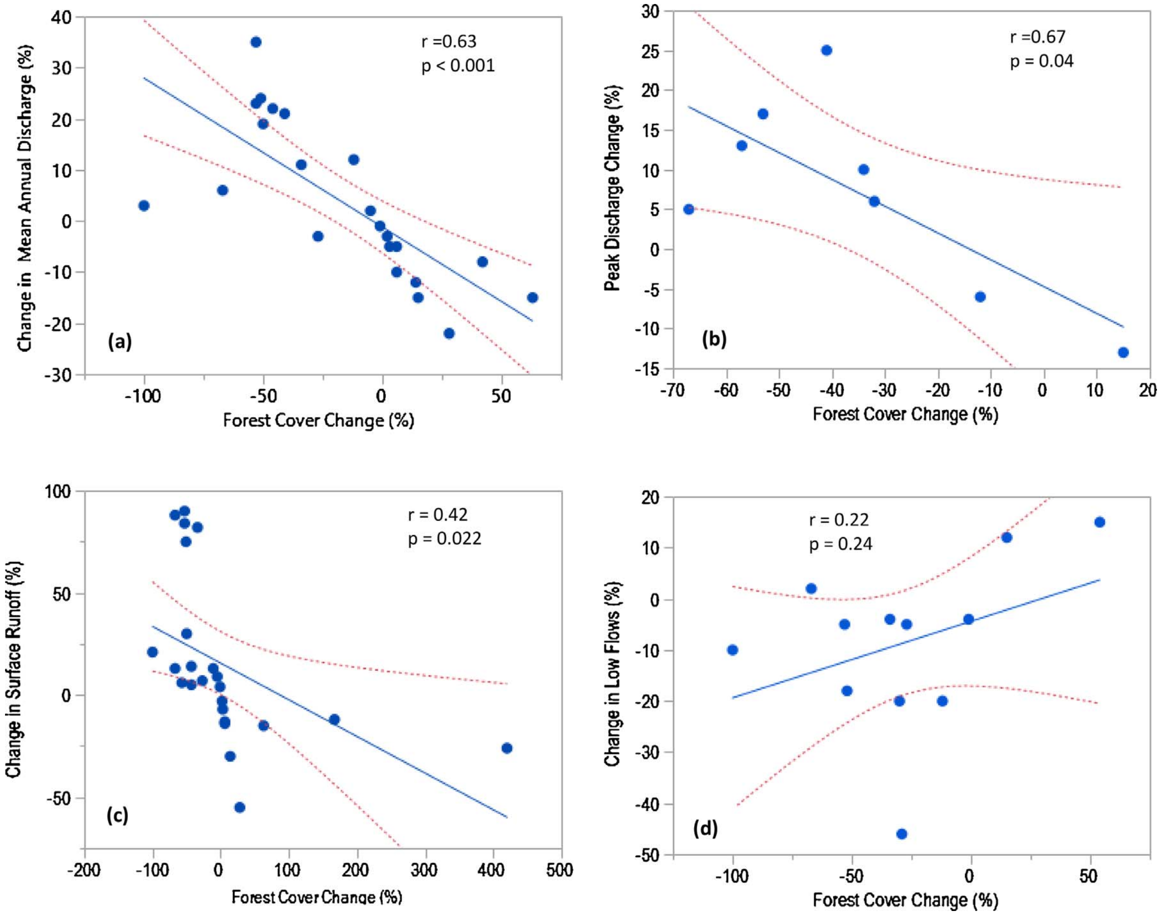


**Fig. 9.** Correlation coefficients (r) and significance (p) of the linear regressions of forest cover change and simulated changes in **(a)** Mean Annual Discharge **(b)** Peak Discharge, **(c)** Surface Runoff, and **(d)** Low Flows. Red lines indicate the 95% confidence interval for the regression line.

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