

Spatio-temporal patterns of agricultural expansion and its effect on watershed degradation: a case from the mountains of Nepal

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Abstract Expansion of agricultural at the cost of forested land is a common cause of watershed degradation in the mountain zones of developing countries. Many studies have been conducted to demonstrate land use changes in such regions. However, current knowledge regarding the changes, driving forces and implications of such change within the context of watershed development is limited. This study analyses changes in spatial patterns of agricultural land use and their consequences for watershed degradation during the 1976–2000 period along an altitude gradient in a watershed in Nepal, by means of remote sensing, GIS and the universal soil loss equation. Estimated soil loss ranged from 589 to 620 t ha⁻¹ y⁻¹, while areas of extreme hazard severity (>100 t ha⁻¹) increased from 9 to 14.5% from 1990 to 2000. Spatial distribution of soil loss in 2000 was characterized by 88% of total soil losses being from upland agricultural areas. The study determined that without considering other forms of land degradation, only water erosion was responsible for erosion of a substantial area in a short timeframe. Areas under upland cultivation are in an extremely vulnerable state, with these areas potentially no longer cultivable within a period of 6 years. As sustainability of the watershed is dependent on forests, continued depletion of forest resources will result in poor economic returns from agriculture for local people, together with loss of ecosystem services. Thus, in order to achieve the goal of watershed development, remaining forest lands must be kept under strict protection.

Keywords GIS · Land use change · Remote sensing · Soil erosion

Introduction

Land degradation is a major global environmental problem (Trimble and Crosson 2000; Vallejo et al. 2006). Water-led soil erosion in particular is a common environmental land degradation issue which can affect the sustainable development and agriculture of developing countries. An area can be prone to soil erosion by water due to specific factors, such as high rainfall intensity, steep slopes and vegetation scarcity (Kefi et al. 2010). According to Pimentel et al. (1995), soil erosion is a major environmental threat to the sustainability and productivity of agriculture due to the damage it causes to arable land. Furthermore, this phenomenon can degrade not only soil productivity but also water quality, as well as causing sedimentation and increasing the probability of floods (Zhou et al. 2008). Several other factors, such as inappropriate agricultural techniques or deforestation, increase this risk. This phenomenon is still continuing and even increasing, especially in developing countries (Bahadur 2009a; Lal 2001; Liu et al. 2010; Wang et al. 2010a; Yue-qing et al. 2009).

Land degradation due to water-induced soil erosion in mountainous regions is mainly due to deforestation—particularly the expansion of agricultural activities in marginal areas. Forestlands have important functions from an ecological perspective and provide services that are essential to maintain the life-support system, including water supply and regulation, and nutrient cycling. Montane forests not only support local residents but also many more residing in lower altitude areas. Sustainable development and management of upland natural resources for the welfare of local populations

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should be the key objective of watershed management and includes the sustainable utilization and conservation of forest resources at community or watershed level as one of its important components (Sharma and Krosschell 1996). However, the expansion of agricultural land at the cost of loss of forestland is a common phenomenon in the mountain zones of developing countries such as Nepal (Bahadur 2009b; Bhattarai et al. 2009; Bhattarai and Conway 2008; Gautam et al. 2004). Indeed, the expansion of agriculture is posited as one of the main dynamics of land cover change globally (Bahadur 2009b; Lambin et al. 1999, 2001). A large body of literature is developing regarding the use of landscape pattern metrics to understand the qualities of a given landscape, both in terms of ecological functions and services and in the detection of human influence in shaping that environment (e.g. Batistela et al. 2000; Forman 1995; Southworth et al. 2002). Similarly, many studies have been conducted to illustrate land use change in the mountains of Nepal. However, knowledge of the changes, driving forces and implications of this change within the context of watershed development is currently limited.

Land use/cover change (LUCC) is an important parameter used in the assessment of global environmental change, with LUCC studies in mountainous regions of Asia making enormous progress in recent years (Bahadur 2009b; Gautam et al. 2004; Rao and Pant 2001; Tekle and Hedlund 2000; Virgo and Subba 1994). Most research has focussed on LUCC itself; very little attention has been given to the relationship between LUCC and its associated eco-environmental effects. The influence of vegetation management on soil erosion has been extensively studied (Bahadur 2009a; De Ona et al. 2009; Klima and Wisniewska-Kielian 2006; Pavanelli and Cavazza 2010; Ripl and Eiseltova 2009; Ruzkova et al. 2008; Shao et al. 2009; Tangyuan et al. 2009), but interactive effects between land use and soil are poorly documented in the literature (Wang et al. 2010b). Tree size and density have been identified as the most important factors affecting slope stability, excluding hydrological factors (Genet et al. 2010). Hubblea et al. (2010) have also argued that riverside vegetation is a significant factor influencing the occurrence and progress of streambed and riverbank erosion. Vegetative restoration may increase the stability of degraded soil through enrichment of soil organic carbon (Yao et al. 2009). In particular, few studies have reported on the interactions between LUCC and soil erosion. In general, scientists believe that human activity is the main reason for triggering soil erosion, but that inappropriate land use activities might also accelerate this process. Zhu and Ren (2000) concluded that while soil erosion is a natural geologic process, severe soil erosion is usually the result of improper land use. Up to now this issue has not been sufficiently described, but there has recently been rapid growth in research regarding the

relationship between LUCC and its associated erosion of the ecological environment.

Different methods and models have been developed to assess and predict soil loss caused by water-induced soil erosion. Widely used (Wishmeier and Smith 1978) to predict the annual average soil loss per hectare in agricultural land due to rill and sheet erosion, the universal soil loss equation (USLE) was revised by Renard et al. (1997) to become the revised universal soil loss equation (RUSLE). The USLE and RUSLE are considered quantitative models because they involve the measurement and quantification of various parameters. The USLE has been widely applied at the small watershed (Dickinson and Collins 1998; Williams and Berndt 1972, 1977) to large catchment scale (Baba and Yusof 2001; Bahadur 2009a; Jain and Kothiyari 2000; Jain et al. 2001). In other studies (Julien and Frenette 1987; Julien and Gonzales del Tanago 1991; Kouli et al. 2008; Onori et al. 2006; Onyando et al. 2005; Wilson and Gallant 1996; Wu et al. 2005), a watershed is subdivided either into cells, regular grids or units. Using the USLE and RUSLE, Renschler et al. (1997) predicted soil erosion and its spatial distribution in a 211-km² catchment at a grid resolution ranging from 200 to 250 m. Bahadur (2009a) mapped the spatial distribution of soil loss in a catchment in northern Thailand using a grid resolution of 100 m. Erosion models involve the use of several landscape factors, including topography, soil data and vegetation coverage, with the latter considered a significant factor in reducing soil erosion (De Asis and Omasa 2007; Zheng 2006; Zhou et al. 2008). Taking this into account, vegetation cover can be detected by remote sensing tools such as satellite images or vegetation indices. Among the large number of satellite images available, Landsat images are commonly employed, especially for land use classification which is helpful in mapping vegetation types (Vrieling 2006). In addition, the employment of multi-temporal satellite images is considered appropriate to assess vegetation cover in different periods of the year (Cyr et al. 1995). In order to identify areas threatened by water-induced soil erosion, and to increase the efficiency of erosion control, many researchers have applied remote sensing and geographic information systems (GIS) in their studies, often combining them with the USLE or RUSLE as qualitative models (De Asis and Omasa 2007; Fistikoglu and Harmancioglu 2002; Yoshino and Ishioka 2005; Yue-ying et al. 2009). GIS-based soil erosion risk assessment models continue to play an important role in soil conservation planning (Bahadur 2009a; Grauso et al. 2008; Kouli et al. 2009; Wang et al. 2010a). Within this conceptual approach, the strengths and weaknesses facing soil conservation planning (Surda et al. 2007) must be known in order to provide a better database for further applications. Because physically based models still require many input parameters, empirical models play an important role in soil conservation studies (Liu et al. 2000).

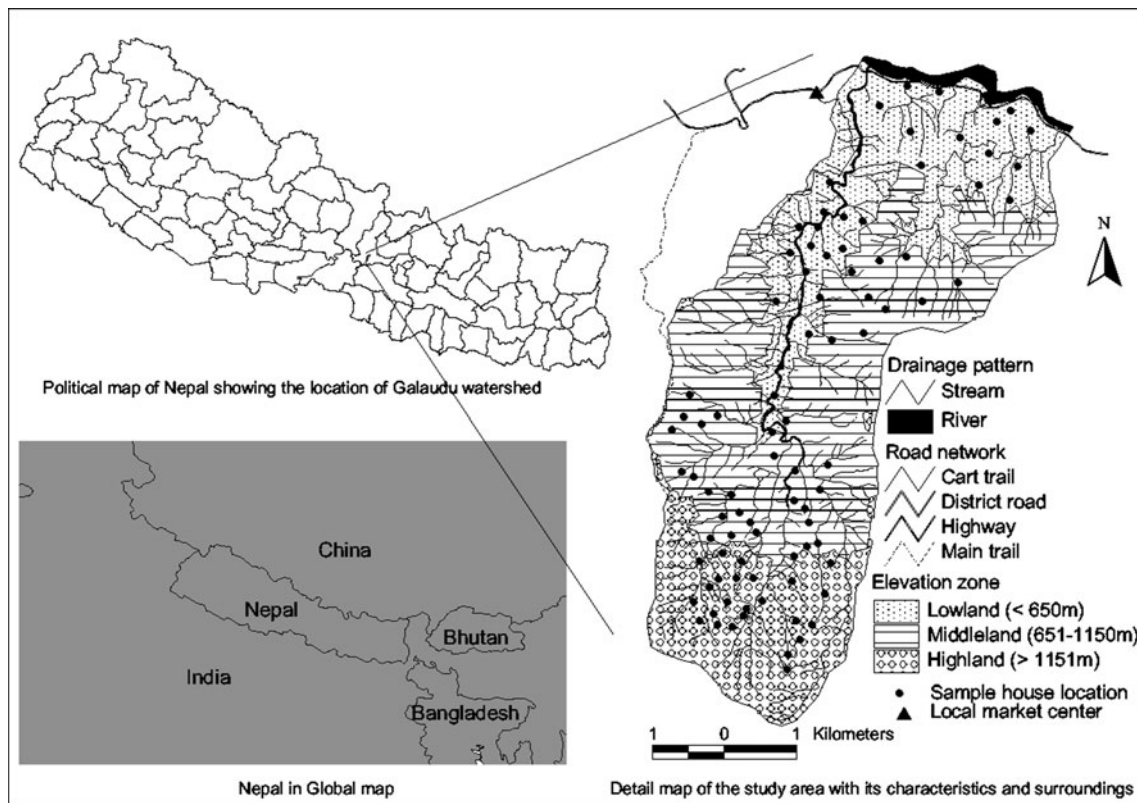


Fig. 1 Location of the study area

In this context, this study analyses changes in spatial patterns of agricultural land use between 1976 and 2000, along an altitude gradient in a Nepalese watershed. The main objectives of this study were to assess and illustrate the expansion of agricultural land at the expense of forest cover in the region and to assess the effect of vegetation on soil erosion. Even a cursory examination of satellite images reveals that this expansion has been discontinuous in both time and space, and thus the objective also includes an assessment of the consequences of this land use change on watershed degradation. Beginning with an examination of the degree to which the patterns of agricultural conversion influence watershed degradation, this paper then explores land use and cover change and the spatial–temporal distribution of soil erosion in the studied watershed, before finally proposing suitable and comprehensive measures for sustainable watershed development.

Methods

Study area

The study area encompasses the mountainous Galaudu/Pokhare Khola sub-watershed (hereafter refer as Galaudu watershed), situated in Dhading district of Nepal (Fig. 1).

The topography of the watershed is mountainous with an average slope exceeding 30%, and shows features which are often found in Asian mountain zones. Most of the watershed is composed of mountainous areas of hill forest and upland cultivation, while the soils of are loam, sandy loam, clay loam, silt loam and sandy clay loam. The area has a sub-tropical climate with a mean annual rainfall of 1,404 mm. The elevations of the highest and lowest points are 1,960 and 217 m above mean sea level, respectively. The watershed can be divided into fertile, relatively flat valleys along the rivers, and surrounding uplands with medium to steep slopes (as shown on Fig. 1). Agricultural lands in the valleys are under intensive management with multiple cropping systems and are mostly irrigated. Paddy, potato, wheat and vegetables are major crops cultivated in the valley. Rain-fed agriculture, with or without outward facing terraces, is practised on the remainder of the agricultural lands, with many of these not suitable for crop production without strong soil and water conservation measures because of their high erodibility and low productivity.

The development of the watershed is not uniform. The lowland valley stretching from Galaudu and Pokhare Khola near to the highway and local market centre is one of the most fertile and economically important areas of watershed. The local economy and employment opportunities

available in these semi-urban areas differ from the more rural areas, with the former directly connected to Kathmandu valley by highway, have alternative sources of energy and an alternative source of income in addition to agriculture. In contrast, residents of the surrounding rural areas are primarily dependent on arable agriculture and livestock raising for their livelihood. This high variability of ecological and economic conditions makes the watershed an appropriate site to study land use dynamics and associated factors.

Data preparation

This study utilises remote sensing and GIS analysis for the mapping of land use land cover change and soil loss assessment. The main data used in the research include a Landsat multispectral scanner (MSS) satellite image from 10th October 1976, as well as Thematic Mapper (TM) data acquired on 4th February 1990 and 13th March 2000 in digital format. In addition, an Indian Remote Sensing satellite image from 7 March 2002 was also used. 1:50,000 scale black-and-white aerial photographs from 1978 were employed as ground reference information for the classification and accuracy estimation of the classified MSS image. 1:25,000 scale topographic maps of published in 1995 by the Survey Department, Government of Nepal (HMG), and digital topographic data with contour intervals of 20 m produced by the same agency were also used. A digital elevation model of the study area was prepared from the digital contour information to obtain slope, flow accumulation and flow direction data in order to estimate soil loss. Rainfall data from Dhading meteorological station, Nepal, was used for calculation of the rainfall erosion factor. Soil samples were collected and hydrometric analysis carried out for the estimation of soil erodibility factors for the different soil types found within the watershed. The ground reference information required for classification and accuracy assessment of satellite images was collected from the field in January–July 2003. Patch level information of forest types, and condition and history of land use provided by both local people and direct observation in the field were collected using a self-designed format.

Digital image processing

Digital image processing was carried out to obtain land use and land cover maps from the RS data. This process involved removing any undesirable image characteristics produced by the sensor, which included calibration of image radiometry, removal of noise and correction of geometric distortions (Schowengerdt 1983). Further processing of the obtained images, which were geometrically corrected to a smaller scale, was performed by registering

the images to the 1:25,000 scale topographic map sheets via the selection of ground control points (GCPs). The root mean square (RMS) error accepted was less than 1 pixel at the first order and the nearest neighbourhood transformation. The study area was clipped with a vector boundary layer.

Classification of remote sensing data was performed by extracting different feature sets using band ratios; normalized differential vegetation index (NDVI) and principal component analysis (PCA) are standard image processing techniques (Matheson and Ringrose 1994). Of the unsupervised classification techniques available, the Iterative Self-organizing Data Analysis Technique (ISODATA) was chosen. This method, which involves repeatedly performing an entire classification and recalculating statistics with minimum user inputs to locate clusters, is also relatively simple and has considerable intuitive appeal. However, the output of this technique can be affected by the choice of initial parameters and their interactions with each other (Vanderzee and Ehrlichlich 1995). Therefore, parameters assigned for each classification scheme were kept the same, including maximum number of clusters (40 clusters) to be formed. The clusters formed were regrouped with Ward's method of hierarchical clustering, which is designed to optimize the minimum variance within clusters (Ward 1963) by calculating means for each variable within each cluster and the squared Euclidean distance to the cluster means for each case (Aldenderfer and Blashfield 1984). Distances are summed for all cases and at each step, with the two clusters merging being those that result in the smallest increase in the overall sum of the squared within-cluster distances. Resulting classes were identified on the basis of knowledge drawn from the field survey, air photos and previous land use map. Results of each classification scheme were compared, leading to the creation of error matrices and analysis of overall classification accuracy. Finally, the classification scheme that gave a superior result during unsupervised classification was selected.

Supervised classification was performed on the selected classification scheme employing Bayesian maximum likelihood classifier (MLC). MLC, a parametric decision rule, is a well-developed method derived from statistical decision theory that has been applied to the problem of classifying image data for several years (Niblack 1985; Settle and Briggs 1987). At first, training signatures for identifiable classes were established by evaluating the field knowledge. After obtaining a suitable indication of satisfactory discrimination between classes during signature evaluation, a final classification was run to produce the land use map. Training areas corresponding to each classification item (hereafter, land use class) were, in the case of the IRS image, chosen from among the training samples collected from the field, and in the case of the MSS and TM

images, were generated from the interpretation of aerial photographs of the study area. Although their dates did not exactly match those of the satellite images, the aerial photographs were used as reference information in these classifications under the assumption that land use in the watershed had not substantially changed between the time of aerial photography and satellite observation. This was considered the best and most feasible option that could be used in this research. In producing land use maps for 1976, 1990 and 2000 and to investigate changes that occurred between these periods, the following four land use land cover (hereafter, land use) classes were considered in image classification: forestland, scrubland, lowland agriculture and upland agriculture. Choice of these classes was guided by (i) the objective of the research, (ii) expected degree of accuracy in image classification and (iii) the easiness of identifying classes on aerial photographs.

Of all land use classes, ‘upland agriculture’ is the most complex, including as it does all other combinations of land uses not included in the other classes. During winter, the uplands in the study area, including most of the Middle Hills, are mostly barren and have spectral values similar to those of barren lands such as non-vegetative hills and riverbeds (Tokola et al. 2001). Moreover, during the time the satellite images were taken (particularly the IRS image), many upland terraces consisted of exposed soil as a result of fresh ploughing by farmers in preparation for the next summer crop. This condition of the cultivated uplands made it impossible to distinguish them from rough roads, new construction sites and other built-up areas. This justifies the combining of settlements, barren lands and built-up areas with upland agricultural lands in this study, which may not be acceptable at any other time of year. Instances of shadow in all images constituted another major problem encountered during image classification. These areas were initially classified as belonging to a separate class, but were later assigned to their real respective classes with the help of ‘ground-truth’ information. Post-classification was performed after the selective combination of classes; classified images were sieved, clumped and filtered before the final output was produced. Sieving removes isolated classified pixels using blob grouping, while clumping helps maintain spatial coherency by removing unclassified black pixels (speckles or holes) in classified images (Richards 1994). Finally, a 3×3 median filter was applied to smooth the classified images. All activities related to image processing were performed in ERDAS Imagine version 8.7 (ERDAS 1997). Classified images were then exported to Arc View GIS Version 3.2 (ESRI 1997) from ERDAS and the rest of the analyses performed in GIS environments. The classified images were first converted to grid and then to shape format in Arc View. The polygon themes generated were exported to Arc Info GIS Version 3.5.1 (ESRI Redlands

USA) and polygons <0.5 ha in size were ‘eliminated’ in Arc Info. This elimination was necessary in order to minimize the effects of classification errors arising from resolution differences between the three satellite images, whilst at the same time without significantly altering the area under each land use class. The resultant polygon themes were used in further analyses.

Detection of land use changes

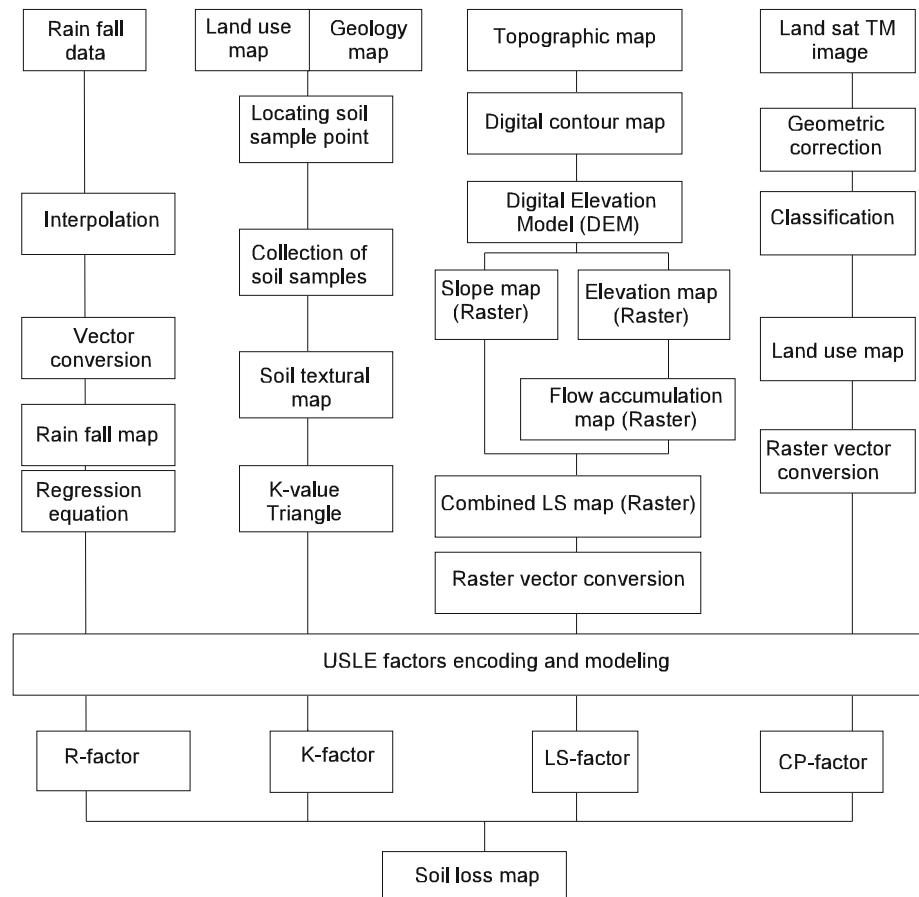
The land use polygon themes for 1976, 1990 and 2000 obtained from the digital classification of satellite data and subsequent GIS analyses using the method described above were overlaid two at a time in Arc View GIS, with the area converted from each of the classes to any of the other classes then computed.

Soil loss assessment

Apart from rainfall and runoff, the rate of soil erosion from an area is also strongly dependent on its soil, vegetation and topographic characteristics. In real situations, these characteristics are found to vary greatly within the various subareas of a watershed. A watershed, therefore, needs to be discretized into smaller homogeneous units before computations of soil loss are made. A grid-based discretization has been found to be the most reasonable procedure in both process-based models and in other simple models (Beven 1996; Kothyari and Jain 1997). For this study, a grid-based discretization procedure was adopted, with a 25-m grid size used since this was considered small enough for a grid cell to encompass a homogeneous area. Soil loss was computed based on USLE in a GIS environment using ERDAS IMAGINE, ARCINFO[®] and ARCVIEW[®] GIS Packages (ERDAS 1997; ESRI 1997; Bahadur 2009a). The entire analytical methodology follows the steps shown in Fig. 2. First, grid cells for rainfall, soil units, combined slope length and steepness, and land use and practice management were prepared. Computed values for R, K, L, S and C, P were encoded into the respective units of each coverage. This coverage was overlaid and the soil loss rate calculated as per the USLE. These were further classified into six major groups to show erosion severity in relation to spatial distribution and aerial extent. The USLE method has been found to produce realistic estimates of soil erosion over small areas (Wishmeier and Smith 1978), and was therefore used to estimate soil erosion within a grid cell. The USLE is expressed as

$$E = RKLSCP$$

where E is the amount of soil erosion ($t\ ha^{-1}\ y^{-1}$); R is the rainfall erosivity factor; K is the soil erodibility factor; LS is the slope steepness and slope length factor; C is the cover

Fig. 2 GIS methodology of estimating soil loss**Table 1** Land uses of Galaudu watershed, Nepal, 1976, 1990 and 2000

Land use classes	1976		1990		2000	
	Area (ha)	%	Area (ha)	%	Area (ha)	%
Upland agriculture	430.1	15.9	414.7	15.3	671.2	24.8
Lowland agriculture	428.1	15.8	809.8	29.9	839.4	31.0
Forest	1597.2	59.1	1475.9	54.8	1189.7	44.2
Scrubland	242.5	9.2	0.0	0.0	0.0	0.0

management factor and P is the supporting practice factor (see Bahadur 2009a for details).

Results and discussion

Agriculture–forest land use dynamics

The areas under the three land use classes during the three study periods are shown in Table 1, while the land use change map for 1990–2000 is presented in Fig. 3. The results show that forests decreased in area while agricultural land increased continuously throughout the study

period. The 1976 extent of scrubland (degraded forest land) was assessed, and by 1990 the condition of these areas had improved sufficiently for them to be reclassified as forest. Lowland agriculture expanded greatly during the first period, whereas upland agriculture increased in area during the later period. Of the major land use groups, around 65 per cent of the area of upland agriculture, 52 per cent of lowland agriculture and 45 per cent of forest remained unchanged from 1976 to 2000. Forest lands therefore shrunk in area by about 55 per cent during the same period (Table 2).

The observed trends of decreasing areas of forest and increasing agricultural land in Galaudu watershed can

Fig. 3 Land use change in 1990–2000 periods in Galaudu Watershed Nepal

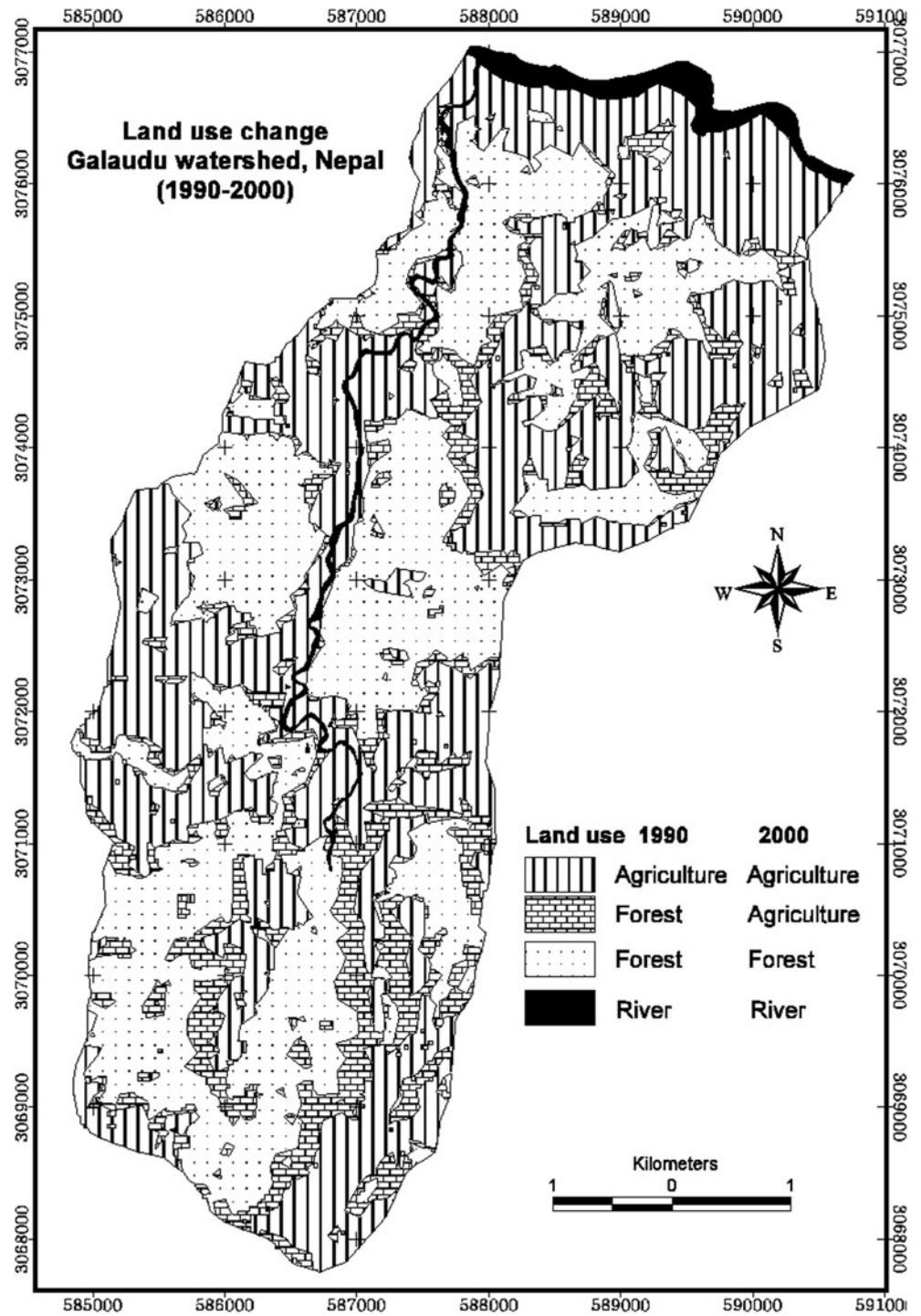


Table 2 Percent change in land use during the three periods, Galaudu watershed, Nepal

Land use classes	1976–1990	1990–2000	1976–2000
Upland agriculture	−3.71	+38.21	+35.92
Lowland agriculture	+47.13	+3.52	+48.99
Forestland (forest + scrub)	−25.15	−24.05	−55.26

potentially be explained by the following three main reasons. First, a substantial proportion of agricultural land in the study area is located in more steeply sloping areas where slope stability and soil erosion is of critical concern (ICIMOD 1993). These steep agricultural fields suffer from rapid soil erosion and nutrient depletion, which forces farmers to find new land to meet their growing food needs. Forest areas may therefore have been converted by farmers for use in their agricultural activities. Other studies of mountainous regions of Southeast Asia have found that many households practice shifting cultivation (Bahadur 2009a). There is also evidence from the hills of Thailand (Fox et al. 1995) and Honduras (Kammerbauer and Ardon 1999) that declining soil productivity and increased weed competition have led farmers to search for new land for the fulfilment of their basic needs.

Second, most settlements in the upland area are located on poorer quality land (less productive, outward-facing sloping terraces, no irrigation facilities) compared with lower elevation areas; because of this a higher level of human–forest interaction may be expected in these areas, thereby resulting in more pronounced forest loss compared to in lower elevation areas.

Third, most of the community forestry activities expected to have a positive influence on the balance of forest cover are concentrated at lower altitudes. Less forest loss in lower elevation zones suggests that forest conservation by local communities and concerned agencies has played an important role in bringing a positive outcome to the balance of forestry land use in the watershed. The same process was not possible at higher elevations (highland) both because of the inability of community-based forest management programs to cover those areas and the virtual non-existence of forest monitoring by the forestry department. This effectively led to a state of open access to high altitude forests.

The existing model of community forestry was unable to bring high elevation forests under proper management. This was probably because of difficulties in identifying users and use patterns due to the presence of a continuous and extensive forest accessed by a widespread population.

Although there was a net loss in forest area, a substantial proportion of degraded forests (scrubland) in the lowlands were reclassified as forest due to improvements in condition, especially in the first study period (1976–1990). This improvement may have been due to the success of community forestry programs in lower elevation areas, as well as the proximity of roads and market centres and subsequent increase in accessibility.

The continuous loss of forest area over time, despite the presence of different forestry development programs, represents a challenge to the efforts of forest conservation and development, the forestry department and the donor agency. Combined investment from multiple actors at

various levels is therefore an important condition for a successful outcome from collective action at the local level (Ostrom 1990).

The expansion of lowland at the expense of upland agriculture during the first study period indicates increased agricultural intensification and diversification. Conversations with local farmers revealed that there was indeed a big shift in the use pattern of lowlands during this period, with a shift towards winter cropping of mainly wheat and potato on irrigated land. More recently, potato cultivation for commercial purposes has gained momentum in the lowlands, mainly due to improved access to local markets and higher profitability compared with wheat and other cereal crops.

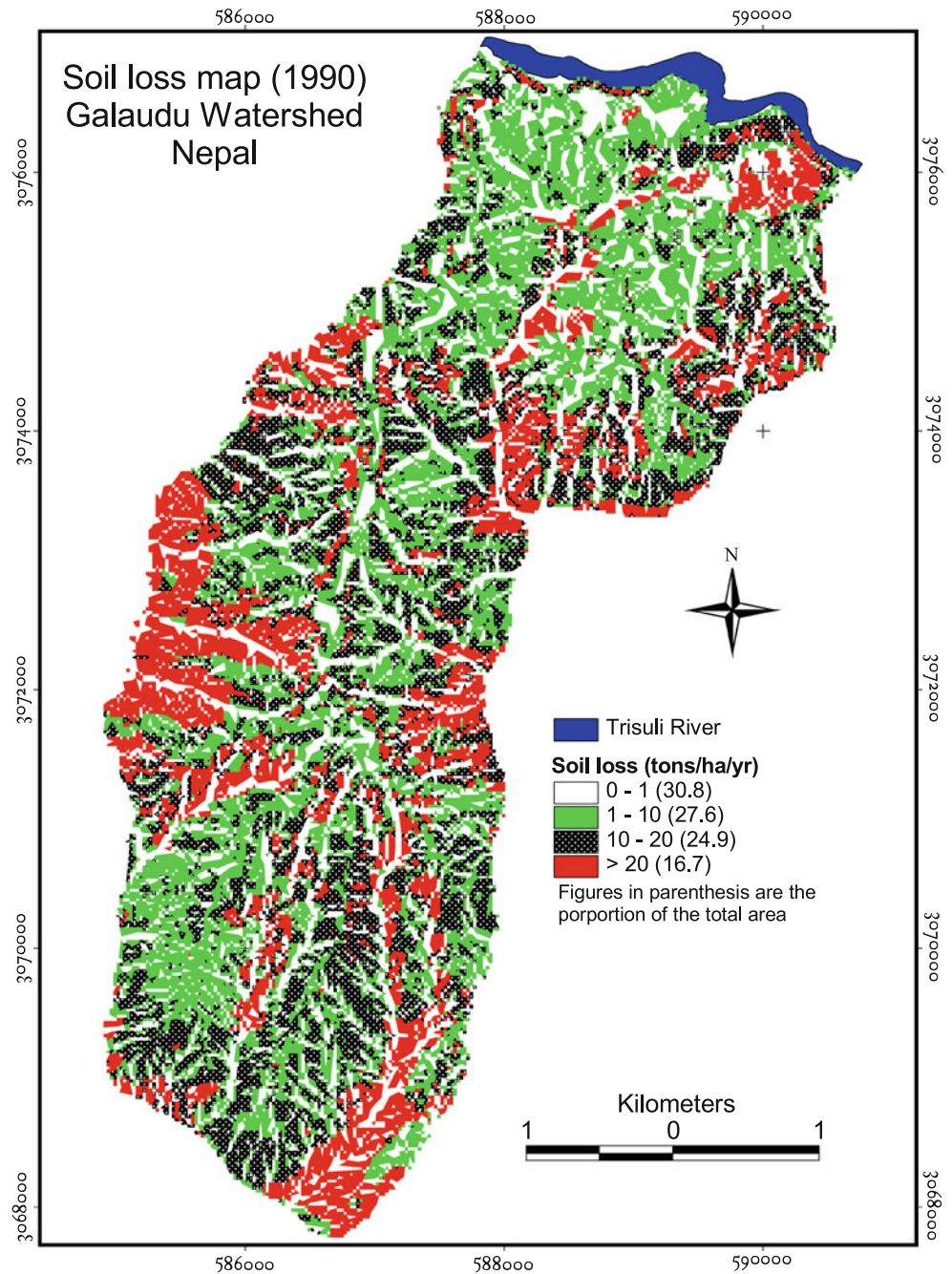
Effect of land use change on soil loss

Two methods of estimating erosion rates can be carried out using the USLE, with the fundamental difference being the respective factors taken into consideration. The first, ‘potential erosion’, is computed based on only four factors; R, K, L and S, while the second, ‘actual erosion’, uses these four as well as the factors C and P. Potential erosion implies that even under natural conditions erosion occurs, as the factors considered are difficult to change due to human interference. But in reality, most areas are heavily subjected to human interference, whether it be for cultivation, deforestation or any other form of land use. In such cases, factor C or ‘vegetation cover’ plays an important role in the actual amount of soil loss or rate of erosion. Similarly, the types of conservation measures undertaken (mechanical or biological) can further determine the extent of actual erosion. Hence, the estimation of actual erosion provides a better real world picture of erosion rates, with all the factors R, K, L, S, C and P taken into consideration.

Potential soil loss rates were observed to be as low as 0 to a maximum of more than $800 \text{ t ha}^{-1} \text{ y}^{-1}$. Analysis showed that majority of areas have a soil erosion rate of more than $800 \text{ t ha}^{-1} \text{ y}^{-1}$, followed by areas experiencing $400\text{--}800$, $0\text{--}50$, and $50\text{--}400 \text{ t ha}^{-1} \text{ y}^{-1}$.

Taking into consideration factors C and P, actual erosion rates ranged from 0 to $589 \text{ t ha}^{-1} \text{ y}^{-1}$ for the year 1990 and from 0 to $619 \text{ t ha}^{-1} \text{ y}^{-1}$ for the year 2000, as presented in Fig. 4. The rate of soil loss ranged from as low as 0 to a maximum of more than $100 \text{ t ha}^{-1} \text{ y}^{-1}$. Erosion rates were regrouped into six classes (Table 3). Most (30.8%) of the watershed can be said to experience a very slight erosion hazard, less than or equal to 1 t ha^{-1} soil loss annually in both study periods. The second highest erosion hazard class (27.6% in 1990 and 23.4% in 2000) in terms of aerial coverage is the slight hazard severity class, with $1\text{--}10 \text{ t ha}^{-1} \text{ y}^{-1}$, followed by moderate (24.9% in 1990 and 21.3% in 2000), extremely severe (9.0% in 1990 and

Fig. 4 a Spatial distribution of estimated soil loss in Galaudu Watershed, Nepal in 1990; **b** Spatial distribution of estimated soil loss in Galaudu Watershed, Nepal in 2000



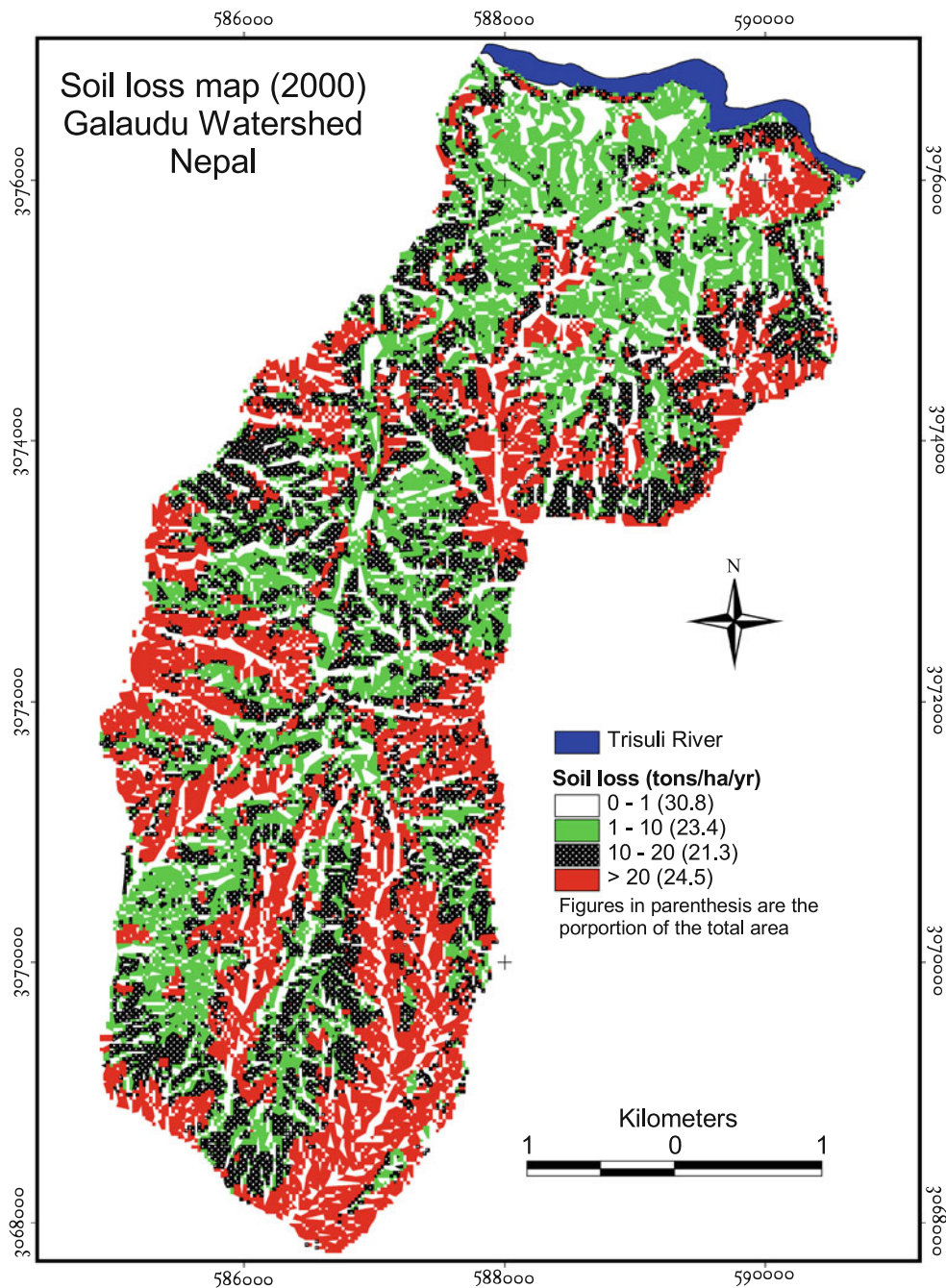
14.5% in 2000), severe (7.6% in 1990 and 9.7% in 2000) and very severe (0.1% in 1990 and 0.2% in 2000), with soil erosion rates ranging from 10 to 20, more than 100, 20 to 50 and 50 to 100 t ha⁻¹ y⁻¹, respectively (Table 3). Various researchers have used different soil erosion rate class ranges in the categorization of hazard severity, with these classes depending on erosion in the specific locality under study. In the Galaudu watershed, areas experiencing soil erosion of more than 100 t ha⁻¹ annually were classified as facing an extremely severe hazard severity. These areas accounted for 9.0% of the total watershed in 1990, increasing to 14.5 per cent in 2000. Severe hazard classes

collectively comprised about 8% of the total area in 1990 and about 10% in 2000.

Once soil erosion rating classes have been established, the next step is to understand the relationship between land use types and soil erosion hazard. This information is extremely valuable as it can be used to formulate plans focusing conservation measures on areas particularly at risk, minimizing not only on-site effects but also that of sediment transport downstream.

In this study, the spatial distribution of soil loss in areas of different land use for the year 2000 was examined. Analysis of the results showed that an absolute majority of total soil

Fig. 4 continued



loss is associated with upland land use types, particularly upland agriculture in areas of steep slope which contributes about 88%. Average rates of soil loss for the different land use types are presented in Table 4. The average soil loss from forested areas was $10.09 \text{ t ha}^{-1} \text{ y}^{-1}$, from lowland agricultural areas $24.28 \text{ t ha}^{-1} \text{ y}^{-1}$ and from upland agriculture areas the average was $412.62 \text{ t ha}^{-1} \text{ y}^{-1}$. Onchan (1993) published similar findings of $0.02\text{--}0.2 \text{ t ha}^{-1} \text{ y}^{-1}$ soil loss from forest and $10\text{--}100$ from cultivated land. Similarly, Suddhapreda et al. (1988, cited in Shrestha and

Giri 1995) reported soil loss of $4.5\text{--}132 \text{ t ha}^{-1} \text{ y}^{-1}$ for field crops and $2\text{--}8 \text{ t ha}^{-1} \text{ y}^{-1}$ for forestland in the mountainous areas of northern Thailand, a region environmentally similar to the Galaidu watershed.

An attempt was made to estimate the period of potential soil productivity if the present rate of erosion continues, with the aim of establishing the need for long-term planning and soil conservation. In the past, many countries have only put in place short-term plans, which were not able to properly address such problems and may result in

Table 3 Soil erosion classes, rating and percentage of area coverage for 1990 and 2000 for the watershed studied

Class	Soil loss rating (t ha ⁻¹ y ⁻¹)	Hazard severity	Area (%) on the year 1990	Area (%) on the year 2000
1	0–1	Very slight	30.8	30.8
2	1–10	Slight	27.6	23.4
3	10–20	Moderate	24.9	21.3
4	20–50	Severe	7.6	9.7
5	50–100	Very severe	0.1	0.2
6	>100	Extremely severe	9.0	14.5
Total			100.0	100.0

Table 4 Average rate of soil loss for different land use types

Land use types	Total area		Average rate of soil loss (t ha ⁻¹ y ⁻¹)	Total soil loss (t y ⁻¹)	Contribution to total loss %	Potential soil productivity (years)
	(ha)	%				
Forest	1180.2	45.0	10.09	11910.2	4.17	247
Upland agriculture	614.0	23.4	412.62	253354.5	88.77	6
Lowland agriculture	828.1	31.5	24.28	20112.6	7.04	102
Total	2622.4	100	108.9	285377.4	100	

unrecoverable long-term soil loss. The mean rate quoted by Hurni (1988 in Shrestha and Giri 1995; Shrestha 1999); “a loss of 1 cm top soil is equivalent to 125 tonnes of annual soil loss per ha” was employed when calculating potential soil productivity for different land uses in Galaudu watershed.

Since the top 20 cm of soil is regarded as the most crucial for vegetation (particularly for field crops), the values calculated represent the ‘years’ within which this layer will be completely weathered away. Soil under lowland agriculture and forest seem to be in fairly satisfactory states, requiring 102 and 247 years for complete erosion, respectively. For paddy fields, there is a chance of erosion occurring during the off-season rather than during the cropping season. In contrast, forestland surface soil appears to be safe from disturbance, with in most cases computed erosion essentially due to erosivity and erodibility factors. The most vulnerable soils are those in upland agricultural areas, with the results suggesting that if present erosion rates continue, all upper layers will be eroded within the next 6 years and the soil will no longer useful for crop production. These findings demonstrate the need for immediate and appropriate soil conservation measures to be undertaken.

Soil erosion as a function of land use and topography

It has been established that topography and land cover are the two most important factors affecting soil erosion, with variation in soil loss due to the soil erodibility factor having much less of an influence. Similarly, even though the rainfall factor has an important effect on the overall rate of soil loss, it does not affect the spatial variation in erosion

rate, since it has been assumed to be constant for the study area in question.

In terms of land use, upland agriculture has the highest rate of soil erosion, followed by lowland agricultural land and forested areas. Average rates of annual soil loss for different types of land use and their contribution to total soil loss are listed in Table 4. As can be seen from this table, the average soil loss from forest and its contribution to total soil loss across the watershed is much less when compared with that from agricultural land. Another major variable affecting soil erosion in the study area is topography. For any given land cover, areas of steep slope have much higher rates of erosion compared to flat areas. Table 5 shows the total soil loss and average rate of soil loss for various classes of slope. It can be seen that the average rate of soil loss and the contribution to total soil loss from steeper slopes are tremendously high compared with that from gentle slopes. This clearly demonstrates the pressing need for improved management of steep slopes.

Soil loss along altitudinal gradients

As seen from Fig. 4, the spatial distribution of soil loss varied throughout the watershed, with greater loss occurring from areas of higher elevation (highland) compared with lowland areas. The results of this study, obtained by overlaying a polygon theme of elevation gradients in three zones with a polygon theme of soil loss, showed that areas of higher elevation have a higher rate of soil loss compared with lower elevations. Indeed, soil loss rates from the former more than twice as high when compared with that from lowland areas (64.4 vs. 27.5 t ha⁻¹ y⁻¹, respectively;

Table 5 Soil loss values for various ranges of slope

Slope range	Total area (ha)	Total soil loss (t ha ⁻¹ y ⁻¹)	Average rate of soil loss (t ha ⁻¹ y ⁻¹)	Contribution to gross soil loss (%)
0–5	116.8	1,326.8	11.35	0.46
5–10	112.3	3,567.6	31.74	1.25
10–20	564.1	45,413.5	80.49	15.92
20–30	1057.5	130,854.7	123.73	45.90
>30	764.8	103,923.3	135.88	36.45
Total	2615.5	285,086.1	108.98	100.00

Table 6 Variation of soil erosion with altitude

Zone	Area		Total soil loss (t y ⁻¹)	Average soil loss (t ha ⁻¹ y ⁻¹)	Contribution (%)
	ha	%			
Lowland (<650 m)	835.8	31.0	22,970.5	27.5	18.3
Middle land (651–1150)	1,293.5	47.9	65,532.0	50.7	52.3
Highland (>1150)	571.1	21.1	36,776.3	64.4	29.4
Total	2,700.4	100.0	125,278.8	46.4	100.0

Table 7 Variation of soil erosion with aspect

Aspect	Total soil loss (t y ⁻¹)	Area (ha)	Average rate of soil loss (t ha ⁻¹ y ⁻¹)	Contribution to gross soil loss (%)
Flat	116.32	78.53	2.92	0.04
North	49,405.5	479.82	107.4	17.31
Northeast	70,267.3	523.26	136.7	24.62
East	56,698.0	402.54	144.3	19.86
Southeast	36,109.5	252.60	147.9	12.65
South	9,587.5	98.83	105.5	3.35
Southwest	10,359.9	161.87	67.9	3.63
West	16,133.6	288.28	58.2	5.65
Northwest	36,686.7	461.54	81.7	12.85
Total	285,364.8	2620.31	108.9	100.00

Table 6). The observed contribution to total soil loss as a proportion of land area was also higher from highland areas (29%) compared with other zones. There are two possible explanations for the higher amount of soil loss occurring in higher elevation areas. First, most highland areas are characterized by the presence of steeper slopes, with most upland cultivation practised on slopes above even 35%. Second, the conversion of forest to agricultural land in these areas may also be responsible for an increase in the rate of soil loss with respect to that from lowland regions.

Soil erosion rates and hill slope aspect

As described in the previous section, the different land use patterns on different hill slopes have a significant impact on soil erosion. Results obtained from analysis of the relationship between soil erosion rates and slope aspect are presented in Table 7. From this table it is clear that with an average value of 108.9 t ha⁻¹ y⁻¹, south-facing slopes

experience the highest average rate of soil erosion, greater than that faced by north-facing slopes. The lower values of soil loss reported for west-facing slopes is, however, likely due to the effect of topography, as this group contains a relatively higher proportion of gentler slopes.

Conclusion

This study has demonstrated that even without considering other forms of land degradation, water erosion alone may cause a substantial volume of soil to be removed within a short timeframe. Soil erosion is present in Galaudu watershed in areas of all land use types, both small and large in extent. Upland cultivation regions are in an extremely vulnerable state, as it has been shown that if current levels of erosion persist, these areas will become unproductive within 6 years—not a long time from an agricultural perspective. This is the equivalent, taking an average soil N

content of 0.22%, of nearly 220 kgs of N/ha being removed annually. Merely adding higher levels of nutrients is not the solution to a loss of crop productivity due to soil erosion, since even a heavy application of N would also be washed away. In any case, most of what is applied is not available to plants due to various factors such as leaching, soil nutrient imbalance, etc. Lal (1989) observed no difference in corn yield with and without the addition of fertilizer to the top 20 cm of eroded soil. Such a situation illustrates the urgency required for prescribing proper conservation measures to protect soil quality. This represents an incomparably better solution than attempting to restore soil quality by direct nutrient replenishment, which increases not only production costs but also off-site environmental damage.

Soil erosion is location specific, with its technical characteristics and economic impact varying widely both within and between locations. The findings presented here are not solutions, but demonstrate the essentiality and intensity of the required conservation planning approaches. Since the unit of land on which it occurs is ‘the farm’ and the decision entity is ‘the farmers’, planning objectives for soil conservation should target a ‘Bottom-up’ approach, integrating public participation and proper conservation technology with the knowledge of soil formation processes and economics of production, along with institutional development within the framework of sound policy legislation. In addition, as the sustainability of the watershed is dependent on the forests, continued depletion of forest resources will result in poor economic returns from agriculture for local people, together with loss of ecosystem services. Thus, in order to achieve the goal of watershed development, remaining forest lands should be kept under strict protection. In order to maintain such a management scheme, policies should be established which support technologies enhancing agricultural productivity, crop diversity and efficient resource recycling within agro-ecosystems through soil and water conservation activities, as well as a community forestry program and effective forest monitoring across the watershed.

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