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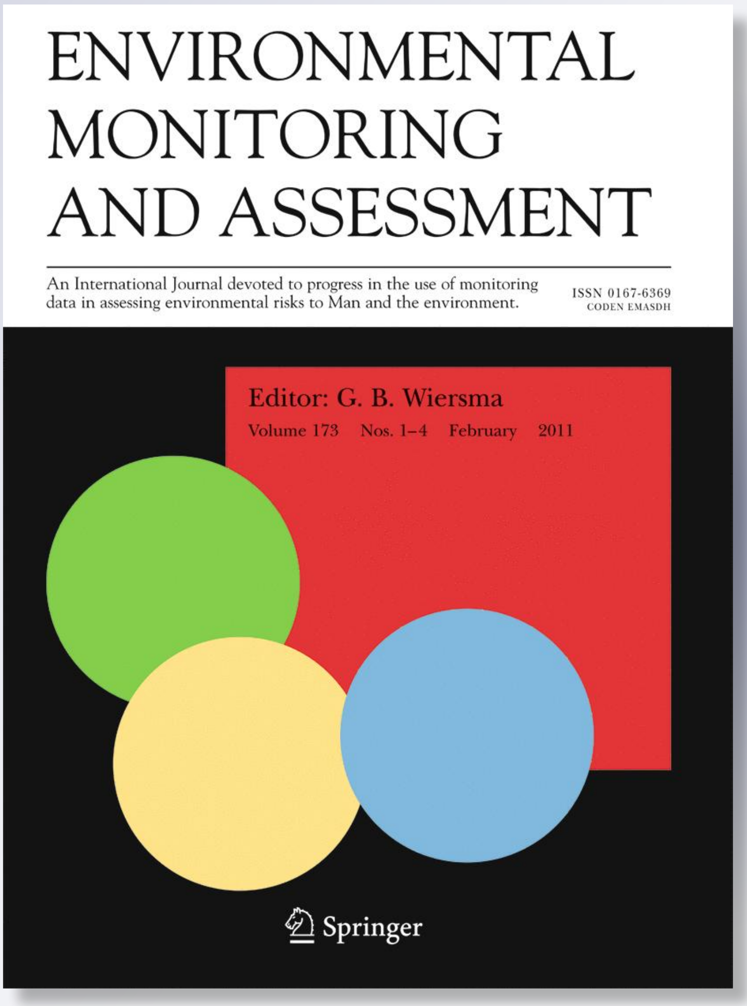

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Effect of land use land cover change on soil erosion potential in an agricultural watershed

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Abstract Universal soil loss equation (USLE) was used in conjunction with a geographic information system to determine the influence of land use and land cover change (LUCC) on soil erosion potential of a reservoir catchment during the period 1989 to 2004. Results showed that the mean soil erosion potential of the watershed was increased slightly from 12.11 t ha^{-1} $year^{-1}$ in the year 1989 to 13.21 t ha^{-1} $year^{-1}$ in the year 2004. Spatial analysis revealed that the disappearance of forest patches from relatively flat areas, increased in wasteland in steep slope, and intensification of cultivation practice in relatively more erosion-prone soil were the main factors contributing toward the increased soil erosion potential of the watershed during the study period. Results indicated that transition of other land use land cover (LUC) categories to cropland was the most detrimental to watershed in terms of soil loss while forest acted as the most effective barrier to soil loss. A p value of 0.5503 obtained for two-tailed paired t test between the mean erosion potential of microwatersheds in 1989 and 2004 also indicated towards a moderate change in soil erosion potential of the watershed over the studied period. This study revealed that the spatial location of LUC parcels with respect to terrain and associated soil properties should be an important consideration in soil erosion assessment process.

Keywords Land use land cover change · Soil erosion · USLE · GIS

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Introduction

Land use land cover (LUC) change associated with climatic and geomorphologic conditions of the area have an accelerating impact on the land degradation. Natural as well as human-induced land use land cover change (LUCC) has significant impacts on regional soil degradation, including soil erosion, soil acidification, nutrient leaching, and organic matter depletion. Since the last century, soil erosion accelerated by human activities has become a serious environmental problem. It has a manifold environmental impact by negatively affecting water supply, reservoir storage capacity, agricultural productivity, and freshwater ecology of the region.

In recent years, many researchers have highlighted the environmental consequences of soil erosion. It includes studies on the long-term effects of soil erosion, such as soil productivity and decreasing crop yield (Verstraeten et al. 2002), and degradation of water quality by carrying nutrients, pesticides, and heavy metal contaminants to surface water bodies (e.g., Boers 1996; De Wit and Behrendt 1999; Verstraeten and Poesen 2002). Sediment delivery also significantly affects channel and floodplain morphology (Asselman and Middelkoop 1995; De Moor and Verstraeten 2008), the ecological functioning of floodplains (Richards et al. 2002), and sedimentation of reservoirs (Verstraeten and Poesen 1999). Two important factors affecting soil erosion and sediment delivery to river channels are changes in land use and climate (Toy et al. 2002; Houben et al. 2006). Given that the climate and LUC are expected to change because of human activities in the future (IPCC 2007), it is important to examine the potential effects of these changes on soil erosion potential at a watershed scale.

Soil erosion estimation at a regional scale is influenced by the complexity of the soil erosion process and the availability of data describing the soil erosion factors. In the last decade, regional and national level assessments of soil erosion were carried out using different approaches, ranging from indicator or factor-based approaches to process-based models (Gobin et al. 2004). However, the universal soil loss equation (USLE) and its modifications (e.g., RUSLE, Renard et al. 1997) are still widely used because of its simplicity and a greater availability of input parameters. One way to investigate the effect of LUC on soil erosion involves using historic or temporal satellite images to analyze the LUCC in relation to change in soil erosion potential of the watershed or sediment discharges at watershed outlet (Jordan et al. 2005). In recent years, a number of studies have been carried out to estimate the potential effects of LUCC on soil erosion at different spatio-temporal scales. These include the scale of small watersheds (Favis-Mortlock and Boardman 1995; Pruski and Nearing 2002; Dunjo et al. 2004; Van Rompaey et al. 2005; Jordan et al. 2005; Nearing et al. 2005; Cebecauer and Hofierka 2008) and global scale (Yang et al. 2003; Ito 2007). Similarly, effects of LUCC have been studied at temporal scale of few years (Neil Munro et al. 2008; Siyuan et al. 2007) to number of decades (Martha et al. 2008; Szilassi et al. 2008; Piccarreta et al. 2006). All these studies identified a strong influence of land use changes on soil erosion and sediment transport rates.

The changing socioeconomic environment of the Maithon reservoir catchment has introduced significant regional transformation of the land use. Moreover, inappropriate land management practices made the catchment vulnerable to soil erosion, which has negatively impacted the watershed health and the livelihood of local inhabitants. In the context of environmental and economic concern of land use dynamics, this paper studied the impacts of land use and land cover change on soil erosion potential in the Maithon reservoir catchment using remote sensing and geographic information system (GIS) techniques.

Methodology

Study area

The study site of this work is the Maithon reservoir catchment, which is situated in Jharkhand state, India. It lies between 85.41° and 86.90° E longitude and 23.75° and 24.56° N latitude and covers an area of approximately 5,553 km^2. The elevation ranges from 120 to 1,360 m above mean sea level. A high variability in the ecological and economic conditions makes the watershed an appropriate site to study the implication of land use dynamics on watershed health. The land of the watershed is also highly susceptible to soil erosion by water mainly because of erosion-prone soil and intensive agricultural and mining activities.

Land use land cover (LUC) dynamics

To determine the LUCC in the study area, Landsat TM and IRS P6 LISS III satellite images of 1989 and 2004, respectively, were used. Out of the available seven bands in Landsat TM, only four bands (bands 2 to 5), which are spectrally and radiometrically identical to that of the IRS P6 LISS III image, were chosen for classification purposes. To ensure an accurate change analysis, preprocessing of images from the two different satellites were carried out with due care. Preprocessing methods such as geometric correction, radiometric corrections, and Lambertian correction were applied to minimize the depiction of changes due to misregistration and environmental and sensor differences between the two images. Image of both dates were georeferenced with respect to each other by collecting high-quality ground control points, and the total RMSE error of georeferencing was kept below 0.2 pixels.

To obtain the thematic LUC map from satellite images, hybrid classification method, which is a combination of ISODATA clustering and maximum likelihood classifier, was used. Five typologies of LUC, such as water, forest, cropland, wasteland, and settlement, were mapped. Land use and land cover map of year 2004 generated using LISS III data was resampled from its original 23.5-m resolution to a 30-m resolution for better comparison. The geocoding, classification, and resampling of satellite images were carried out using the Erdas Imagine 8.5 software. Land use change was examined through cross-tabulation of independently classified images of 1989 and 2004 using the ArcGIS Desktop software. The changes in LUC were not only characterized by their extent but also by their patch characteristics, such as patch density and mean patch size, and their association with terrain and soil properties.

Soil loss estimation

The soil erosion risk in this study was estimated for 1989 and 2004 using the universal soil loss equation (USLE), which can be described using following equation:

$$A = R \times K \times LS \times C \times P \qquad (1)$$

where A is amount of soil erosion calculated in tons per hectare per year, R is rainfall factor (in megajoules millimeter per hectare per hour per year), K is soil erodibility factor (ton hectare hour hectare^{-1} megajoule^{-1} millimeter^{-1}), L is slope length factor, S is slope steepness factor, C is cover and management factor, and P is erosion control practice factor. The rainfall erosivity (R factor) was calculated using the formula suggested by Wischmeier (1959), which is given below:

$$R = \sum_{i=1}^{12} 1.735 \times 10^{\left(1.5 \log_{10}\left(p_i^2/p\right) - 0.08188\right)} \qquad (2)$$

where R is rainfall erosivity factor (in megajoules millimeter per hectare per hour per year), P_i is monthly rainfall in millimeters, and P is annual rainfall in millimeters. Ten years (1993–2002) daily rainfall data collected from three rain gauge stations were used to calculate the mean R value of each station. Finally, the mean R value was interpolated using the Thiessen polygon method to obtain spatially distributed R value. Soil erodibility (K factor) was estimated using the following equation given by Foster et al. (1991):

$$K = 2.8 \times 10^{-7} \times M^{1.14} \times (12 - a) + 4.3 \times 10^{-3} \times (b - 2) + 3.3 \times (c - 3) \qquad (3)$$

where K is soil erodibility factor (ton hectare hour hectare^{-1} megajoule^{-1} millimeter^{-1}), M is particle size parameter [silt (%) + very fine sand (%)] × [(100 − clay (%)], a is organic matter content (%), b is the soil structure code, and c is soil permeability class. Soil texture and organic carbon content used to derive the K factor were obtained from a standard soil series map of the area obtained from All India Soil and Land Use Survey (AISLUS) and 179 points soil sample data collected from Soil Conservation Department of Damodar Valley Corporation (DVC). The L (slope length) and S (slope steepness) factors were calculated using topographic information using formula proposed by McCool et al. (1987):

$$L = \left(\lambda/22.13\right)^m \qquad (4)$$

where L is slope length factor, λ is field slope length (in meter), m is a dimensionless exponent that depends on slope steepness, being 0.5 for slopes exceeding 5%, 0.4 for 4% slopes, and 0.3 for slopes <3%. The percent slope was determined from digital elevation model (DEM) (Fig. 2), while a grid size of 200 m was used as field slope length (l). The S factor for slope longer than 4 m can be derived as follows:

$$S = 10.8 \sin\theta + 0.03 \text{ for slope} < 9\%, \qquad (5)$$

$$S = 16.8 \sin\theta - 0.50 \text{ for slope} \geq 9\% \qquad (6)$$

Table 1 The values of USLE crop management and practice factor used

Crop management factor		Practice factor	
Class	*C* factor	Slope (%)	*P* factor
Water body	1.000	0–2	0.5
Settlement	0.002	2–12	0.6
Crop	0.320	12–16	0.7
Forest	0.004	16–20	0.8
Wasteland	0.100	20–25	0.9

where S is slope steepness factor and θ is slope angle (in degrees). The processed shuttle radar topography mission (SRTM) DEM of 90-m resolution available publicly from CGIAR website (Jarvis et al. 2006; http://srtm.csi.cgiar.org) was downloaded and LS factor was derived using the above equations in ArcGIS. The crop management factor (C) used in this study was based on the LUC maps. Each of the LUC categories were assigned with a value for the C factor based on the available literature on the study site (Table 1). We assumed there was no soil erosion control practice implemented in the watershed during the study period. Since for the same crop or land use practice the P factor depends on the slope, we have used a variable P factor based on slope (Table 1).

To evaluate the effects of LUCC on the soil erosion potential in the watershed, the USLE model was run for 1989 and 2004 separately. During each model run, all parameters remained the same except values of the C factor, which was changed according to the LUC of the respective year. However, its effect on the erosion model output was twofold. The first effect results from overall compositional change of LUC categories, which has direct bearing on spatial distribution of the C factor. The second effect arises from redistribution of individual patches of LUC categories, which would alter the association of particular LUC category with terrain slope and soil properties. This is of significance because a particular LUC category will have a different erosion potential for a different slope and soil. Thus in the present study, special emphases were given for the different associations of LUC with terrain and soil to explore and reveal the underlying mechanism for changed erosion pattern of the study site.

Results and discussion

Land use land cover dynamics

The various image preprocessing techniques along with hybrid classification techniques enabled us to

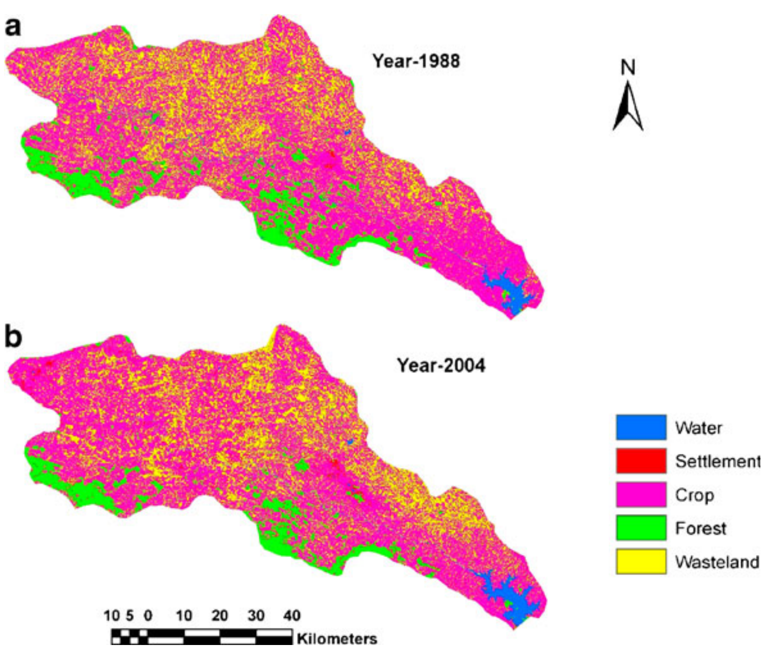

Fig. 1 Land use land covers map of 1989 (**a**) and 2004 (**b**)

obtain an overall classification accuracy of 88.6% and 81.6% for the LUC maps of 2004 and 1989, respectively (Fig. 1).

Figure 1 shows that the studied watershed is predominantly an agricultural watershed. The highly connected agricultural land and wasteland were the two most abundant land use type, while settlement occupied a small portion of the entire watershed area. Examination of the satellite-derived thematic maps and field survey revealed a varying degree of changes in the composition of LUC categories in the study areas over a span of 15 years. The total extent or composition of individual LUC class and their gain/loss is presented in Table 2. However, it is important to report and characterize the patch characteristics of individual class along with their aerial extent for a better insight into land use dynamics. For this purpose, we used patch density (number of patch per 100-ha area) and mean patch size (in hectares) for further characterizing the land use dynamics in the region (Table 2).

Table 2 indicates that cropland and wasteland remain the two dominant land use classes during the study period and together occupy more than 80% of the area. The wasteland described in this study actually includes degraded forest, bare soil, river sand, rock outcrop, abandoned mines, and heavily eroded land. The cropland was reduced to 310,676.67 ha (55.92%) in 2004 compared to 321,858.99 ha (57.93%) in 1989. This reduction is approximately 3.47% compared to the original area in 1989. There is not only a decrease in aerial extent croplands but they also became more fragmented during the study period. The patch density for croplands was increased by 17.59%, while the mean patch size for croplands was decreased was by 17.75%. In contrast, wastelands gained almost 3.59% during the same period. The possible reason for reduction in cropland might be degradation of cropland due to soil erosion and nutrient depletion because of the lack of required soil conservation measures. The patch density for wasteland category was reduced by approximately 25%, while the mean patch size was increased from 4.25 ha in 1989 to 5.87 ha in 2004. The increase in mean patch size is possibly due to transition of patches of other LUC categories in between the wasteland patches to wasteland to form a contiguous patch. Forest, the third most prominent land cover in the watershed, was mostly found in the hilly southern part of the watershed, with few forest patches being distributed in relatively flat central region of the watershed. Similar to phenomena elsewhere in the world, the study site has also witnessed an encroachment of forestland for agricultural and infrastructure development and for mining activities, which resulted in the decline of the total forest area. The forest area shrank to 10.56% in 2004 from 12.55% in 1989, i.e. reduction of almost 18.28% since 1989. The mean patch size and patch density of forestland were also decreased by 2.47% and 16.01%, respectively, which is attributed to complete disappearance of small and medium-size isolated forest patches during the study period. Analyzing the LUC maps of the two years, it was further evident that the disappearance of forest patches took place mostly in relatively flat parts of the watershed, while the forest on hilly portions remained intact.

Human settlements including other built-up land occupy a relatively small portion (<3%) of the watershed. The town of Giridih is the only urban area and is the largest contiguous area under settlement category, while other settlements were mainly small, dispersed village communities. However, prevailing human-dominated activities in the study area led to a large increase in

Table 2 Area percentage, patch density, and mean patch size for different classes

	Area percentage			Patch density (per 100 ha)			Mean size		
	1989	2004	Percent change	1989	2004	Percent change	1989	2004	Percent change
Water	1.6	2.7	65.2	0.3	1.4	425.9	6.0	1.9	−68.4
Settlement	1.0	2.9	207.4	2.8	6.6	139.9	0.3	0.5	44.1
Crop	57.9	55.9	−3.5	2.2	2.5	17.6	26.8	22.1	−17.8
Forest	12.6	10.6	−15.9	3.1	2.6	−16.1	4.1	4.0	−2.5
Wasteland	26.9	27.9	3.6	6.3	4.8	−25.0	4.3	5.9	38.1

settlement cover (including built-up land) during the last 15 years. The area under the settlement category underwent a drastic increase from 0.95% in the 1989 to 2.9% in 2004, which is approximately a 300% increase in aerial extent for settlement category in a span of 15 years. The patch density and mean patch size for settlement category were observed to have increased by 139.86% and 44%, respectively, in the same span of time. The increased patch density and mean patch size of the settlement class is largely because of the construction of newly built-up areas for infrastructure development and sprawling of existing settlement due to increased population. The sizable extent of water category was due to presence of a large reservoir of a multipurpose dam at the outlet of the watershed. Besides the reservoir, the watershed has been characterized by a number of scattered ponds and water in the river itself. A phenomenal increment in the aerial extent of water category was observed from 1989 (1.64%) to 2004 (2.71%). Annual variation in rainfall and surface water storage was the reason for increased aerial extent of water in 2004. In the case of water, patch density increased drastically from 0.27 (1989) to 1.47 (2004), while the mean patch size decreased by 68.37%. As described earlier, the variation is mainly due to change in surface water spread area or formation of small temporary ponds caused by annual rainfall variation.

Estimation of soil erosion potential

The LUC dynamics at patch level indicates that the watershed is under intensive management and other local human activities. Its subsequent effect on soil erosion potential of the watershed was evaluated through average annual soil erosion potential of the watershed using the USLE model. The spatial distributions of various USLE factors and the resultant soil erosion map for both years are presented in Fig. 2. The reservoir part of the watershed, which does not contribute to the net soil erosion and rather acts as a sink for eroded soil, was excluded from further soil erosion analysis.

The average *R* factor (1993–2002) of three rain gauge stations (Santrabad, Rajdhanwar, and Karso) was found to be 3,320, 4,940, and 4,090 MJ mm ha^{-1} h^{-1} $year^{-1}$, respectively, and their spatial interpolation through the Thiessen polygon method are shown in the Fig. 2a. The values of the *K* factor range between 0.18 and 0.25 and their spatial distribution is presented in Fig. 2b. The value of the LS factor calculated using SRTM is found to vary from a minimum of 0.008 (flat terrain) to a maximum of 11.1 (high-elevation area), particularly in hillslope region. The spatial distributions of the LS factor and the *P* factor are shown in Fig. 2c and d, respectively. While all the four USLE parameters temporally remain constant, the value of crop management (*C* factor) changes both spatially as well as temporally. The spatio-temporal distributions of the *C* factor are presented in Fig. 2e and f. The values of the *C* factor ranges from 0.002 to 1 during both years, but their spatial distributions are different because of changes in the LUC. Once all the parameters were obtained, they were put into the USLE model to obtain the erosion potential of the watershed, and the results are shown in Fig. 2g and h. The figures demonstrated that the soil erosion potential of the watershed varied spatially to a great extent in both years. The erosion potential varied between 0.005 and 170 t ha^{-1} $year^{-1}$ in 2004, while in 1989, it ranged from 0.008 to 150 t ha^{-1} $year^{-1}$. The figures also show that the areas with high soil erosion potential were mostly located along the foothill regions in the southern part and some portions of the central watershed during both years.

Soil erosion potential and LUC

In general, for given climatic conditions (*R* factor), the intensity of soil erosion from a particular LUC depends on the local slope of the terrain and associated soil properties. Slopes within the watershed vary from 0% to 91%, with a mean slope of 3.59%. To explore the effect of slope on soil erosion from a particular LUC, the slope map of the catchment was categorized into three distinct slope categories, i.e., steep slope (>5%), which occurred in the southern boundary of the watershed; moderate slope (2–5%), in the central part; and relatively flat area (0–2%), in the northern region of the watershed. The temporal

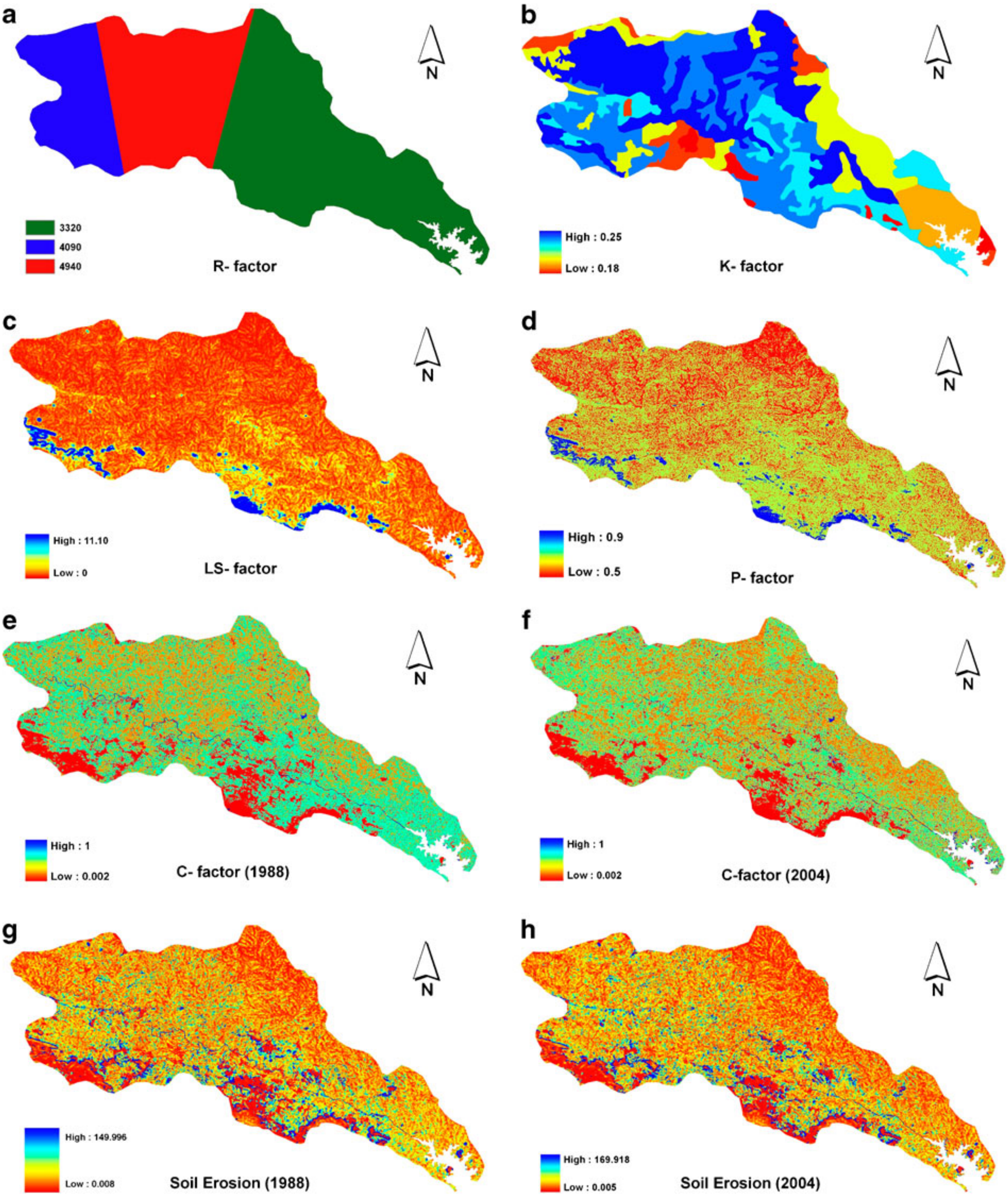

Fig. 2 Spatial distribution of different USLE factors and soil erosion; **a** *R* factor, **b** *K* factor, **c** LS factor, **d** *P* factor, **e** *C* factor for 1989, **f** *C* factor for 2004, **g** soil erosion for 1989, **h** soil erosion for 2004

distribution pattern of LUC with respect to slope steepness is presented in Fig. 3.

Figure 3 reveals that the increased human settlement and other built-up land during the study period took place in moderately slope (2–5%) and flat (0–2%) areas of the watershed. The percentage of settlement under steep slope (>5%) of the watershed decreased. Although, the distribution of croplands did not show much variation in the same duration of study, but a trend similar to settlement category was observed for it. Figure 3 shows that more than 40% of the forest area occurred on the steep slope (>5%) during both periods. However, the percentage distributions of forest area in steep slope as well as moderate slope (2–5%) increased in 2004 owing to afforestation activities in the head water zone, while in the flat region, it decreased owing to conversion of forestland to agricultural and built-up land (come under settlement category). A trend similar to that in the forest area was also observed for wasteland category. Similarly, to explore the change in spatial association of different LUCs with soil, the soil erodibility (K factor) map of the watershed was divided into three categories: low (0.18–0.203), medium (0.203–0.226), and high (0.226–0.25); results are presented in Fig. 4. The distribution of settlements and croplands followed a similar trend during the study period. Figure 4 shows that the association of both the LUC categories with high- and medium-erodibility soil was higher in 2004 compared to that in 1989, which is of very serious

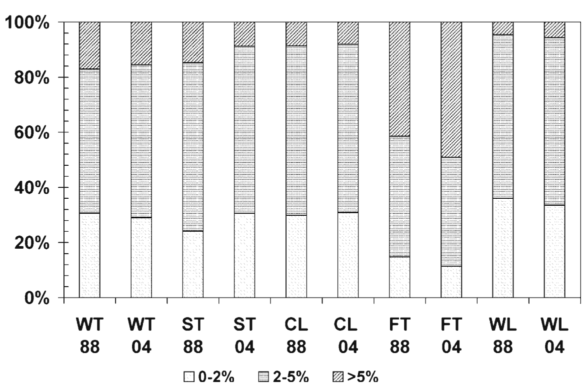

Fig. 3 LUC distribution relative to slope category. *WT* water, *ST* settlement, *CL* cropland, *FT* forest, *WL* wasteland. A suffix of 88 and 04 in LUC name indicates the years 1989 and 2004, respectively

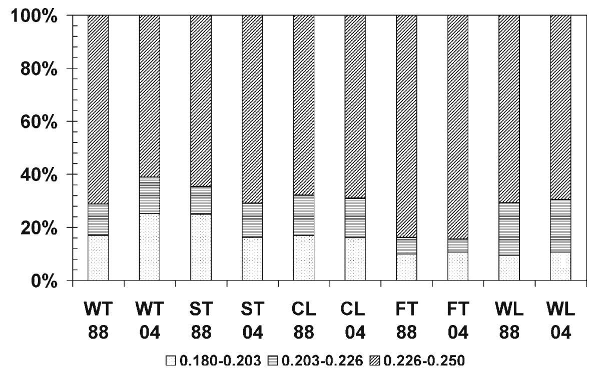

Fig. 4 LUC distribution relative to soil erodibility category. *WT* water, *ST* settlement, *CL* cropland, *FT* forest, *WL* wasteland. A suffix of 88 and 04 in LUC name indicates the years 1989 and 2004, respectively

concern in terms of the soil erosion potential of the watershed. The distribution of forest patches also increased in high- and low-erodibility soil owing to proper management practices taken by the soil conservation and forest department. However, its association with medium-erodibility soil was decreased over the same period. In the case of wasteland, its patches were more distributed in soil of low and medium erodibility in 2004 compared to 1989, indicating best management practices implemented in the watershed.

The changed association of different LUC categories with slope and soil category ultimately affected the net soil erosion from the different LUC categories as well as the watershed as a whole. For this purpose, we calculated the mean soil erosion potential of different land use category. Figure 5 shows the mean soil erosion potential and the mean slope of different land use categories as well as the whole watershed for 1989 and 2004. The pattern of mean soil erosion potential in 1989 was in the order of settlement > crop > forest > wasteland, but in 2004, it was changed to forest > crop > wasteland > settlement. The simultaneous analysis of Figs. 4 and 5 revealed the prevailing mechanisms behind the changed erosion potential of different LUC categories and the watershed as whole. The mean soil erosion potential for settlement and wasteland category decreased, while the same increased for croplands and forestlands. The increased mean soil erosion potential of forestlands was due to the

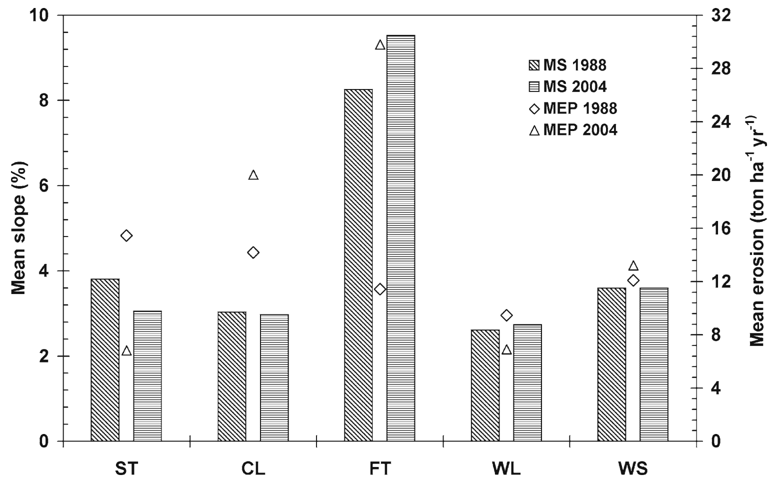

Fig. 5 Mean slope and erosion intensity for different classes and whole watershed. A suffix of 88 and 04 in LUC name indicates 1989 and 2004, respectively. *ST* settlement, *CL* cropland, *FT* forest, *MS* mean slope, *MEP* mean erosion potential, *WL* wasteland, *WS* watershed

loss of forest patches in flat areas of the watershed but not in steep mountainous slopes. The mean slope of croplands almost remained unchanged during the study period. Thus, the increase in the soil erosion potential of croplands was solely due to its more occurrences in high erodible soil in 2004 compared to 1989. The reduction in the mean erosion potential of settlement is due to the decease in mean slope as well as its lesser occurrence in high erodible soil in 2004 compared to 1989. In spite of the increase in the mean slope of wastelands, a decrease in the mean erosion potential was observed for the wasteland category during the same period. This may be attributed to its comparatively less association with high-erodibility soil in 2004. However, the mean erosion potential for the whole watershed increased slightly from 12.11 21 t ha^{-1} year^{-1} in 1989 to 13.21 21 t ha^{-1} year^{-1} in 2004, which may be due to subdued effects of different land transitions. The main reason for this slight increase in the soil erosion potential of the watershed over the studied period could be attributed to the reduction of forest area, increase in wastelands in higher slopes, and increased cultivation practice in more erosion-prone soil. However, it is important to remember that even this small change in the erosion potential could have significantly affected the reservoir sedimentation process. It was estimated that approximately 7.336 million tons of sediments might have potentially transported into the Maithon reservoir in 2004 compared to 6.725 million tons in 1989. It means that the LUC dynamics in the watershed has resulted in approximately 9.1% more sedimentation of the reservoir, which would have definitely affected the reservoir life negatively.

Changes in soil erosion potential at watershed scale

The magnitude of change in the soil erosion potential and their spatial distribution was depicted in Fig. 6a. The figure shows that changes were strongest in mountainous and submountainous regions with high topographic potential for soil erosion during the studied period. This is attributed to the degradation of forestland predominantly in foothill regions due to increasing human activities. The relatively flat areas of the central watershed also had some patches where the erosion potential had undergone a great degree of changes. This can be attributed to phenomena such as increased agricultural practices in high erosion-prone soil and to the disappearance of forest patches from that area. Land use changes such as intensification of agricultural activity in vulnerable areas and deforestation were the most dangerous changes, as these make the land more prone to soil erosion. The change mapped further reveals that the soil erosion potential of 913.95 km^2 (16.73%) and 926.71 km^2 (16.96%) of the total watershed area decreased and increased, respectively, while the remaining 66.3% of the watershed witnessed no change.

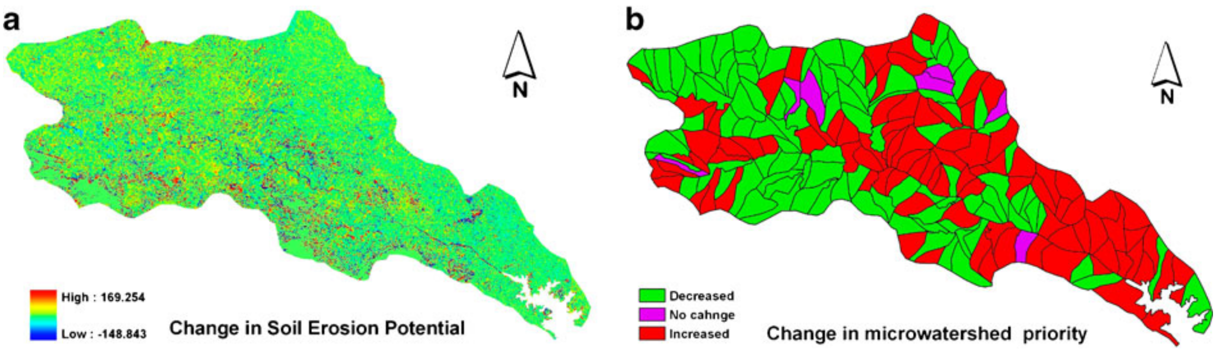

Fig. 6 Spatial distribution of change in **a** soil erosion potential and **b** microwatershed priority

The superimposition of maps such as those showing the change in erosion potential and LUC transition further enabled us to reveal the cause–effect relationship between LUC dynamics and subsequent changes in the soil erosion potential of the watershed. The results of the superimposition of the land use transitions map and the erosion change map are summarized in Table 3.

Table 3 demonstrates that most of the areas that experienced a high increase in erosion potential corresponded to the area where forestlands were converted to croplands (change in average erosion potential was 22.38 t ha^{-1} year^{-1}). The second most detrimental land transition in terms of the change in erosion potential was settlement to croplands, with a mean change in erosion potential of 19.1 t ha^{-1} year^{-1}. Other land transitions that contributed to increased erosion potential watershed were in the order of wasteland to cropland > forest to settlement > settlement to wasteland > forest to wasteland. The positive land transitions that balanced the impact of negative transitions were in the order of wasteland to forest > cropland to wasteland > cropland to settlement > cropland to forestland. From these results, it can be inferred that any land transition to cropland would be detrimental, as it was the major source of sediment. On the other hand, forest was the most effective barrier to soil loss. Since cropland and wasteland together cover approximately 80% of the geographical area of the watershed, best agricultural practices, tillage operation, and development of vegetative cover in wastelands would be more effective in reducing the soil erosion potential of the watershed. From a management perspective, the erosion potential maps of both years were subsequently classified into four erosion potential categories such as low, medium, high, and very high as suggested by Singh et al. (1992). The results of the difference of two maps are presented in Table 4.

Table 4 shows a clear pattern of changes characterized by gradual shifting of low erosion potential category to next higher erosion potential category. However, there is a net decrease in the total under the low, moderate, and high categories by −2.94%, 0.61%, and 0.54%, respectively. At the same time, the area under very high category is increased by 4.1%, which is a major issue of concern. However, this is only a cumulative effect of changes for the entire watershed that may not facilitate the full picture of spatial changes in soil erosion risk at practical management unit and their prioritization so that future soil conservation works can be implemented. Therefore, comparison of soil erosion risk maps was carried out at a microwatershed (a basic management unit for implementing the soil conservation practices)

Table 3 Different land use transition categories and corresponding change in mean erosion potential

1989	2004			
	Settlement	Cropland	Forest	Wasteland
Settlement	6.08	19.10	5.20	7.66
Cropland	−12.41	−0.75	−28.36	−9.69
Forest	8.15	22.38	1.54	6.96
Wasteland	1.40	9.47	−3.91	0.66

Table 4 Different soil erosion potential categories and their area statistics

Erosion intensity ($t\ ha^{-1} year^{-1}$)	Erosion intensity category	1989		2004		Percent change
		Area (km^2)	Percent area	Area (km^2)	Percent area	
0–5	Low	1,751.22	32.06	1,590.45	29.12	−2.94
5–10	Moderate	1,205.49	22.07	1,172.16	21.46	−0.61
10–20	High	1,596.00	29.22	1,566.37	28.67	−0.54
>20	Very high	909.89	16.66	1,133.61	20.75	4.10

level to provide the watershed manager and policy maker better information on the change on spatial extent and intensity of soil erosion. For this purpose, the microwatershed theme was laid over the soil erosion potential map of both years, and a mean erosion potential was obtained for each microwatershed using the *summarize zone* option available in the ArcGIS software. Microwatersheds were placed in a priority list in descending order of their mean soil erosion potential, i.e., microwatershed with the highest mean soil erosion potential was given the first priority and vice versa. The change in soil conservation priority is shown in Fig. 6b. The map reveals that out of the total 205 microwatersheds, only seven microwatersheds in the priority list remained unchanged over the study period. During the same period, 104 and 94 microwatersheds were moved down and moved up in the priority list, respectively. The R^2 value for the mean erosion potential and priority rank of microwatersheds between the two years were found to be 0.88 and 0.96, respectively. A p value of 0.55 was obtained with two-tailed paired t test conducted for equal mean using mean erosion potential of the microwatersheds for the two years. The intermediate p value is an indication that there can be a substantial chance of getting larger variations in the mean erosion potentials of different microwatersheds during the period examined in the present study.

Conclusion

The study presented the potential of remote sensing and GIS for rapid assessment of LUCC and its consequences on regional soil erosion potential in an agricultural watershed between 1989 and 2004. Our study showed that the local human activities were the main cause of the land use changes during the last 15 years leading to changes in the soil erosion potential with mixed responses, i.e., both positive and negative. Some parts of the watershed experienced an increased protective function of land cover, mostly in steep terrain, while other areas, predominantly relatively flat areas affected by deforestation, increased infrastructure and associated land degradation, which have inherently increased erosion risks. However, the overall effect of land cover change over the period 1989–2004 has affected the watershed slightly negatively by increasing soil erosion risks. The mean erosion potential for the whole watershed was increased slightly from 12.11 $t\ ha^{-1}\ year^{-1}$ in 1989 to 13.21 $t\ ha^{-1}\ year^{-1}$ in 2004. The main reason for this slight increase in the soil erosion potential of the watershed over the study period is attributed to the reduction of forest, increase in wasteland in higher slopes, and increased cultivation practice in more erosion-prone soil. Our results also showed that the transition of other LUC categories to cropland was most detrimental, while the forest was the most effective barrier to soil loss. Since cropland and wasteland are the two dominant LUC categories, implementation of best agricultural practices, tillage operation, and development of vegetative cover in wasteland would be suggested for reducing soil erosion potential of the watershed. Although there was little difference in mean erosion potential of the watershed, a high p value (0.55) obtained in two-tailed t test is pointing toward the possibility of getting larger variations of the mean erosion potential of different microwatersheds. Results also indicated that it is not only the composition but also the spatial position of LUC parcels with respect to terrain and associated soil properties are also equally important in soil erosion assessment processes and should be considered as a key factor for proper implementation of soil conservation practices in the study site.

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