ASSESSING LANDSCAPE FRAGMENTATION EFFECTS ON ECOSYSTEM SERVICES IN A SEMI-ARID MOUNTAINOUS ENVIRONMENT: A CASE STUDY ON ABHA WATERSHED, SAUDI ARABIA

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Abstract. The effect of land use land cover (LULC) change is well documented, but the impact of landscape fragmentation on Ecosystem Service Value (ESV) has not been quantitatively explored yet in the study area. The present study designed to evaluate the landscape fragmentation effect of Abha watershed, Saudi Arabia, a new method was proposed by integrating the ESV with the analytical hierarchy process (AHP) and landscape fragmentation model. Results of LULC dynamics showed that urban area has increased significantly by 1648.8 hectares (ha) to 6379.2 ha from 1990 to 2018. While the scrubland has covered a half (15173.6 ha) of the total study area (37001.08 ha) in 1990, but it was significantly reduced to 8907.93 ha in 2018. The calculated ESV of dense vegetation was 0.2 million dollars/year, according to the results of the integrated ESV model, but it was 2 million dollars/year using the initial coefficient in 2018. In 2018, the estimated ESV for water bodies using AHP's integrated ESV model was 7 times lower than the estimated ESV with an initial coefficient That is, a declining trend was discovered after integrating landscape fragmentation for ESV estimation in this study area.
Keywords: *LULC dynamics, analytical hierarchy process, ESV change, fragmentation analysis, spatial mapping*

Introduction

The natural habitat has been hampered and structurally and functionally changed as human interference has increased. This has caused the fragmentation of the landscape, which negatively affects the biodiversity and ecosystem (Sanon et al., 2020; Lhoest, 2020). The landscape fragmentation can be defined as the breaking of a large natural land cover unit into several smaller patches limited by a matrix. The mechanism of fragmentation can take two forms, LULC changes due to natural causes (Flowers et al., 2020; Gerner, 2020; Venier et al., 2020) and anthropic activities (Riechers et al., 2020; Siqueira-Gay et al., 2020). The key anthropogenic processes include urban development, infrastructure development, the intrusion in various cultural environments and so on (Onilude and Vaz, 2020; Kamwi and Mbidzo, 2020; Arroyo-Rodríguez et al., 2020). Many researches have been conducted on the dynamics of LULC over time in a different environment (Motlagh, et al., 2020; Li et al., 2020). The LULC dynamics are highest in the flood plain regions (Thonfeld et al., 2020; Adnan et al., 2020) because of the easy conversion process compare to the mountainous and island regions. Earlier studies show that LULC changes can occur due to severe damage caused by natural and anthropogenic disturbances (Ntihinyurwa and de Vries, 2020; Snep and Clergeau, 2020). The dynamics of LULC cause the landscape to disintegrate, which is described as fragmentation. The fragmented landscape can be turned into other LULC categories quickly. The fragmentation of the landscape will then adversely affect the environment in various ways, such as the deduction of ecological continuity, productivity, functional strength, connectivity, ecological richness etc. (Sanderson, 2020; Jupiter, 2020). Aside from these, landscape fragmentation can impede ecosystem services by interfering with the movement of various matters, species, humans, and energy (Sanderson, 2020). Natural habitats have been increasingly transformed all over the world as a result of human intervention, which has a detrimental impact on natural resources and biodiversity (Patru-Stupariu et al., 2020; Perennes et al., 2020). As a result, it is critical to recognise landscape fragmentation areas that impact ecosystem services. The impact of fragmentation drives biodiversity depletion and shifts in ecological structure (Chase et al., 2020; Herrera et al., 2020; Suárez-Castro et al., 2020). The effect of fragmentation can be different, as it depends on the degree of landscape fragmentation (Rybicki et al., 2020; Yezzi et al., 2020). The fragmentation of the landscape prevented the daily movement of plant and animal species between breeding, feeding habitat, and migration (Rycken et al., 2020). Furthermore, fewer species are assisted by smaller habitat patches, which can only accommodate a smaller number of populations, placing them at risk of extinction (Plaza, 2020). The edge, on the other hand, has a greater impact than ever on the sustainability of native species (Kiene et al., 2020). Lin et al. (2020) investigated the structural change in connectivity for ecological land units using graph theory and the distance of a suitable landscape threshold. Many researchers used an ecological manifestation assessment predictor method to produce a realistic analysis of pattern changes, as well as changes in the vulnerability of ecology and ecosystem resources of respective landscape units (Abad-Segura et al., 2020). The resulting conditions of landscape fragmentation include an increase in patch frequency, edge length, edge-to-area ratio, patch size reduction, and so on. Previous studies reported very clearly that because of the landscape fragmentation, the fragmented units have been exposed to be converted or captured by the anthropogenic activities (Vergara et al., 2020). The diversity of ecology, the quality of habitat, and the security of ecology can be seen to a greater extent in large core areas, while near to fragmented units, such as patch and edge, these have been drastically diminished (Pritchard et al., 2019). For example, if a forest landscape bifurcated by road or railways, which causes the ecological shortage across the buffer distance of railway line or road based on the degree of vehicle frequency, uncertainty and effect of pollution (Kulkarni et al., 2014). Even where forest land is fragmented by highways, the accessibility of networks for certain animals, such as elephants, is hindered. As a result, the accidental demise of many critical species is common (Braulik et al., 2014; Cudowski and Pietryczuk, 2020), exacerbating the conflict between animal and man (Cretois et al., 2019; Mota-Rojas et al., 2020). These combined phenomena have the potential to reduce ecosystem services. Only a few studies have conceptualised the concept of

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fragmentation's effects on ESV (Buckwell et al., 2020). To make it clear about the exposure of natural landscape due to fragmentation to the human inference, the schematic diagram (*Figure 1*) can be substantiated. For the first case (a stretch of landscape continuously), 100 units are considered as the total patch area, while 40 units are edge area. The estimated ratio is 1:2.5 for this. But after the landscape fragmentation, the land has fragmented into four and sixteen patches, the ratio of edge and area would be 1:1.25 and 1:0.625, respectively, which depicts the declining of ESV due to human interference.

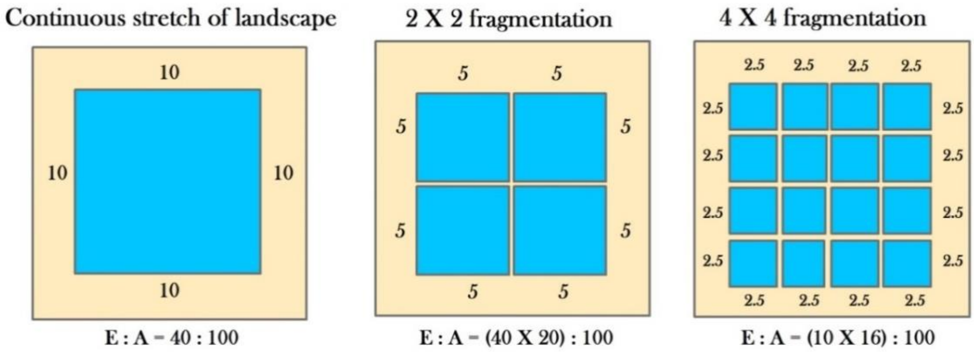

Figure 1. *Schematic diagram showing possible edge effect on fragmented landscape (E= Edge, A= Area)*

Along with other ecosystem services, the landscape fragmentation can hamper the biodiversity in terms of reducing the richness and diversity of the tree species significantly in any regions (Zoderer et al., 2019; Canedoli et al., 2020; Carlucci et al., 2020). Thus, with the declining of biodiversity due to landscape fragmentation has been commonly observed. Also in marine and riverine ecosystem, the fragmentation has largely affected significantly, which includes the aquatic biodiversity and abundance of fish (Kaus et al., 2017; Heino et al., 2020; Bertolini et al., 2020; Sahoo and Swain, 2020; Wang et al., 2020) On the other hand, few services have experienced the restriction of flows of matter or organisms due to fragmentation. Across the natural landscape, the shape, size, arrangement of space and the isolation of patches are affected by fragmentation, which influences the flow of water, organisms, soil and energy negatively or positively (Douglas et al., 2018; Suchara et al., 2019; Rotem et al., 2020). The fragmentation can have negative or positive impacts on the flow of services based on the process and the structure of the landscape (Andriamparany et al., 2020). For example, breakdown of forests due to logging, road construction, the spread of urban land and agriculture can alter the growth and development of many species of plant, which affects negatively the regulation of carbon sequestration and water quality (Mayer et al., 2020). The capacity of the species for travelling across the landscape has decreased due to the effect of the fragmentation for diurnal movement (Benoit et al., 2020; Suraci et al., 2020). For this reason, the risk of species extinction has increased with the fragmented land (Zungu et al., 2019; Zengeya et al., 2019). Apart from it, the ESV of different LULC classes have also decreased due to the landscape fragmentation. Many researchers also integrated the biodiversity with the decreasing ESV for analyzing the effect of landscape fragmentation (Van Bussel et al., 2020). Even, enlarged tree species and diverse flowers can supply various ES with significant amounts (Corcket et al., 2020). Consequently, the ecosystem services have been reduced with the declining of landscape size due to fragmentation effect (Valdés et

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al., 2020). The carbon sequestration and biomass production get affected by the disintegration of the forest or forest fragmentation (Baskent, 2019). This is also applicable in case of water bodies. Big size of the continuous wetlands can supply a larger amount of ecosystem services than a fragmented one, therefore, the wetland with a large number of fragmentations can lead to wetland loss (Gómez-Baggethun et al., 2019). The loss of diversity of animal and plant including water supplies and storage reduction, protection and floodplain loss, groundwater recharge, increasing sedimentation, and soil erosion can happen greatly than many times before due to the degradation as well as fragmentation of wetlands (Zhao et al., 2018; Boardman et al., 2019; Huang et al., 2019; Teixido et al., 2020). Despite the immense values, anthropogenic activities have been affecting the freshwater wetlands and other natural resources, which caused the landscape fragmentation and degradation of such ecosystems (Talbot et al., 2018; Guevara-Ochoa et al., 2020; Han et al., 2020). The conversion of natural resources may be beneficial for the short term in case of the economy, but for the long term, these will be harmful to the environment as well as humans.

To estimate the ESV of different land use land cover classes, several well-known and widely used methods are used, such as the cost-based method, energy analysis model, Integrated Valuation of Environmental Services and Trade-offs (InVEST) model, value/benefit transfer model (VTM), contingent valuation, and so on (Costanza et al., 2014; Yang et al., 2018; Sun et al., 2018; Shi et al., 2020; Chen et al., 2020; Talukdar et al., 2020). Although these approaches have many challenges in estimating ESV on real-world ground. This, too, necessitate extremely sophisticated field data. It is exceedingly difficult to achieve a sophisticated response from locals in countries such as Saudi Arabia, where very few studies on ESV estimation have been performed. As a consequence, people lack adequate understanding of the ESV, which can lead to a great deal of uncertainty and mistake. As a result, we had to focus on Costanza et al. (2014)'s global coefficient value to estimate the ESV of different LULC categories.

Many scholars have researched the effects of land use land cover dynamics on ESV estimation (Shiferaw et al., 2019; Clerici et al., 2019) method. The researchers just looked at the ESV in terms of LULC dynamics. However, no studies have taken into account integrating the dynamics problem with the ESV estimate. As a result, in the current research, we integrated the LULC dynamics in terms of fragmentation for correcting the ESV estimation. We also applied weights (through AHP) to the fragmented units based on the field survey and expert opinion to improve the precision of the ESV estimation. As a result, the aim of this analysis was to calculate the ESV in relation to LULC dynamics over a 30-year period. Another big goal was to investigate the impact of fragmentation on ESV.

Materials and Methods

Study area

Abha, a semi-arid mountainous watershed of Saudi Arabia, is considered as the study area. It is located in Aseer province of Saudi Arabia. It covers an area of 370 km^2. The geographical location is extended between 18^{o}10'12.39''N and 42^{o}21'41.58''E to 18^{o}23'33.05''N and 42^{o}39'36.09''E. A part of Abha's highland is linked with the Arabian shield in the western part of the kingdom (*Figure 2*). The study area observes heavy rainfall for short period, while the surrounding rural areas have witnessed flash flood in winter. The elevation of this region varies between 1954 meters to 2989 meters above

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mean sea level with undulating topography. The watershed is elongated and dominated by many small wadies, which drain their water into this watershed. The slope of this region ranges between 0^{O} to 52.32^{O}. The landscape of the study area is heterogeneous because of the complexity in terrain. The slope, geological weakness, rain etc. have accelerated the erosional problem in the study area, which affects agricultural productivity, forestland, sedimentation etc. The semi-arid climatic condition is the main climatic feature of this region. The study area observes an average rainfall of 214 mm per year (Since three decades) 1990 - 2018. The northwestern part of this watershed with 3000 m altitude has high richness in flora. Because of the climatic and topographical variation in this region, a diverse plant community has been found (Abbas et al., 2020). In the western part of the study area, a huge amount of acacia trees has been found. Therefore, it can be stated that the study area is rich in natural resources. The socio-economic activities have been formed based on natural resources. For this reason, immediate attention should be paid towards the development and conservation of the natural environment. This is the reason; we have selected Abha watershed as the study area.

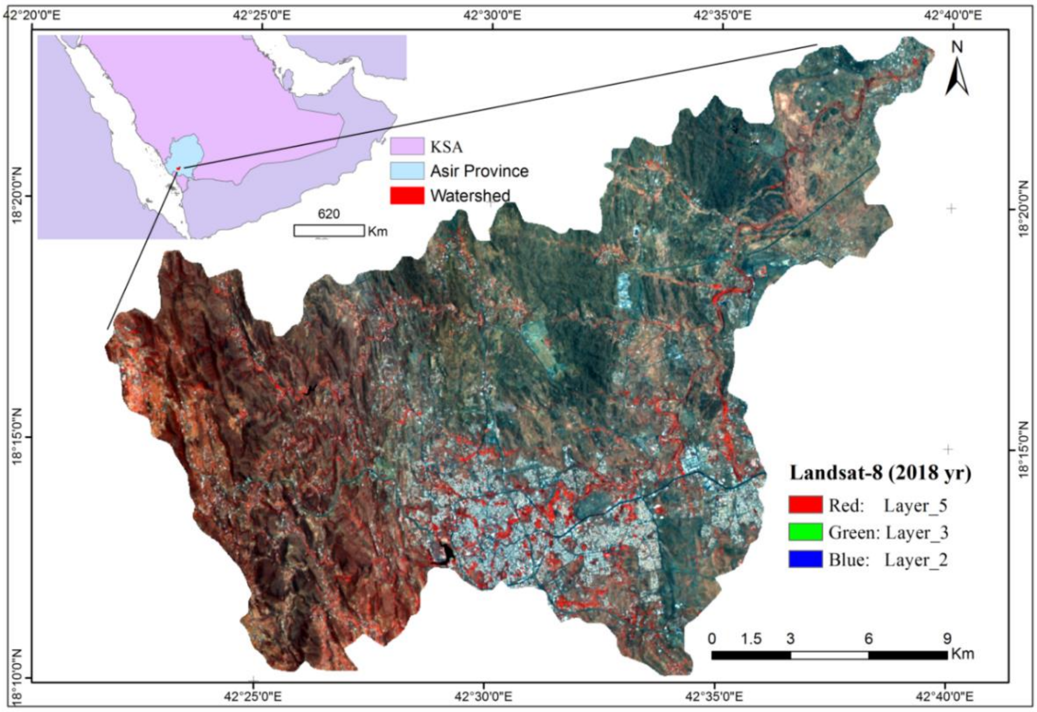

Figure 2. *Location of Abha Watershed*

Methodology

Classification of land use land cover and validation

The LULC change is an identified aspect for analyzing the alteration of global environment and effect on the ecosystem services (Costanza et al., 1997; Clerici et al., 2019; Jiang et al., 2020). In the present study, Landsat 4-5^{TM} and Landsat 8 OLI were utilized for preparing the LULC maps for the year 1990, 2000, and 2018. The required satellite images (Details see *Table 1*) were downloaded from the United State Department of Geological Survey (USGS) (https://earthexplorer.usgs.gov).

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Table 1. *Details of Satellite image*

Satellite data	Path/row	Date and Local Time	Spatial resolution	Number of bands
Landsat TM	167/47	1990-06-02/9.51 am	30	6 (excluding thermal band)
Landsat TM	167/47	2000-05-28/10.07am	30	6 (excluding thermal band)
Landsat 8	167/47	2018-06-15/10.30am	30	9 (excluding thermal band)

The geometrically and radiometrically correction was made on these satellite images using ERDAS software (version 2014). The maximum likelihood classifier (MLC), a supervised image classification technique, was used for LULC classification in ArcGIS 10.5 software (Mazhar and Fadia, 2019). On the theory of probability, the MLC is considered as dependent. In the time of training the data, the statistics of training data for all classes in the area of the band is Gaussian distributed (Alam et al., 2020). We collected 50-70 spectral signatures for each LULC classes, which were used for training the classifier. Based on the collected spectral signatures, eight LULC classes were identified, such as Urban, waterbodies, dense vegetation, sparse vegetation, agricultural cropland, Scrubland, bare soil and exposed rock.

The accuracy evaluation of the LULC map is crucial for the user's confidence. The accuracy measurement quantifies the degree of similarity between the LULC map obtained from satellite images and ground reality (Sánchez-Espinosa et al., 2019). The ground truth samples in this analysis were taken at random from Google Earth real-time data. Statistical methods were used to equate the prepared LULC to the collected ground truth samples. The Kappa coefficient was used to calculate the accuracy of LULC maps. In this analysis, 200 sample sites were chosen at random from Google Earth Pro real-time data, and the considered sites were checked precisely. The Kappa statistics were computed using Stehman's (1996) proposed method. The Kappa coefficient ranges from 0 to 1, with 0 representing the least accuracy in the case of field reality and classified images and > 0.85 representing very high accuracy (Monserud and Leemans, 1992).

Method for analyzing the LULC dynamics

The change detection technique was used in this research to analyse the dynamics of LULC maps for the years 1990, 2000, and 2018. (Kalinicheva et al., 2020; Fahad et al., 2020; Mishra et al., 2020). Change detection techniques are divided into two categories: pre-classification and post-classification approaches (Haque and Basak, 2017). The post-classification technique was used in this research to determine LULC changes from 1990 to 2018.

Methods for computing the ESV

Simulated market approach (Caparrós et al., 2020), benefits transfer approach (Msofe et al., 2020; Custodio et al., 2020), the surrogate (proxy) market approach (Phoomirat et al., 2020) are widely applied approaches for estimating the ESV. The benefit transfer approach is considered as one of the standard method, used for estimating service values (Poudel et al., 2020). Costanza et al. (1997) used a simple benefit transfer approaches for estimating global ESV and provided global coefficients for different types of biomes, which have been widely used to calculate the ESV. Researchers have used LULC classes

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as a proxy of biomes. Then, the area of different LULC classes was computed and integrated with the respective coefficients using equation 1. In this way, the ESV from different LULC classes was estimated.

$$ESV = A \times VC \quad \text{(Eq.1)}$$

where 'ESV' denotes the value of ecosystem services for various LULCs, 'A' denotes the region of each LULC type, and 'VC' denotes the coefficient value of each LULC type. *Table 2* shows the coefficient value of six biomes or LULC categories. The maps for ESV mapping were generated by assigning the computed ESV values to the relevant LULC classes.

Table 2. *Land use land cover and ecosystem service values as per Costanza et al. (1997)*

Land use types	Equivalent biome	ESV coefficient (USD/ha/yr)
Urban	Urban	6661
Water bodies	Lakes and river	12512
Dense vegetation	Forest	3800
Sparse vegetation	Grassland	4166
Agricultural cropland	Cropland	5567
Scrubland	Grassland	4166
Bare soil	Barren land	0
Exposed rock	Rock	0

Modelling landscape fragmentation

Landscape fragmentation is defined as the division of a natural landscape or a broad unit of the landscape into multiple landscapes or units. Landscape fragmentation has an effect on both environmental changes and biodiversity. Until now, very few methods for modelling landscape fragmentation at a spatial scale have been created. The landscape fragmentation tool in ArcGIS 10.5 software was used to model the landscape fragmentation in this analysis. At the spatial scale, the tool distinguishes six types of landscape fragmentation, such as patch, edge, perforated, small core (<250 acres), medium core (250-500 acres), and large core (>500 acres). *Table 3* provided the overview of the above fragmentation categories. In the present study, we prepared fragmentation for LULC maps of 1990, 2000, and 2020.

Table 3. *Reference landscape unit definition*

Landscape	Definition
Patch	Relatively discontinuous areas (spatial domain) or periods (temporal domain) or environmental condition which is relatively homogeneous is represented by patches.
Edge	An edge represents an area where the rapid changes of observed value are found or where the change rate is very high.
Perforated	The edge habitat generated by a small area of non-forest habitat which is enclosed by core habitat is the perforated section.
Small core	The internal area of any landscape which covered <250 acres refers to an as small core.
Medium core	The internal area of any landscape which covered 205 – 500 acres.
Large core	The core area is defined as the internal area of patches after the elimination of edge buffer which is specified by a user

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Computation of ESV by proposing fragmentation integrated method

Previous studies have shown that a number of key ecosystem services have declined due to Landscape fragments, such as carbon sequestration, soil formation, seed dispersal and pollination; and nutrient cycling (Leal Filho et al., 2020; Loewen, 2020; Wang and Dai, 2020). In order to formulate ecosystem service coefficient Costanza et al. (1997) did not consider the impact of landscape fragmentation. Although the ESV produced from a broad and compact landscape unit cannot be identically produced with the ESV produced from a landscape fragmentation unit. From the compact and fragmented landscape, different ESVs should be created. However, no such different coefficient for estimating the ESV has been discovered. If we apply the same coefficient to all broken units of the landscape, the qualitative deterioration of the landscape would not be expressed in calculating the ESV. As a result, to capture the effect of fragmentation on ESV estimation, we used the AHP methodology (Saaty, 2004; Mehdipour et al., 2019), which took the comparison pair matrix into account for six hierarchic units of landscape fragmentation. These fragmented units were weighted using AHP based on expert opinion and local people's perception. Consistency ratio findings showed <3% that was found satisfactory and should be continued for further study. Using equations 2–7 was calculated for the ESV of the patch, edge, perforated, small core, medium core and large core. The patch's ESV was calculated by multiplying the assigned patch weight of one landscape unit by the unit's respective areal coverage and CV (*Eq.2*). Similarly, ESV was measured for various hierarchic landscape units, but when assigning weights to edge, perforated, small core, medium core, and large core, the total weight of the corresponding lower hierarchic landscape unit was taken into account (*Eqs.2 to 7*).

To compute the effect of landscape fragmentation on ESV, first LULC specific ESV was estimated using the coefficient mentioned in *Equation 1* similar to patch and edge etc. Finally, the total ESV from the fragmented units was estimated using *Equation 8*. The ecosystem service values of LULC in total were calculated by *Eq.9*. Then, the difference between these two ESV has been considered as the effect of fragmentation on ESV (*Eq.10*).

$$ESV_p = W_p \times CV_i \quad \text{(Eq.2)}$$

$$ESV_e = ((W_p + W_e) \times CV_i) \quad \text{(Eq.3)}$$

$$ESV_{pr} = ((W_p + W_e + W_{pr}) \times CV_i) \quad \text{(Eq.4)}$$

$$ESV_{sc} = ((W_p + W_e + W_{pr} + W_{sc}) \times CV_i) \quad \text{(Eq.5)}$$

$$ESV_{mc} = ((W_p + W_e + W_{pr} + W_{sc} + W_{mc}) \times CV_i) \quad \text{(Eq.6)}$$

$$ESV_{lc} = ((W_p + W_e + W_{pr} + W_{sc} + W_{mc} + W_{lc}) \times CV_i) \quad \text{(Eq.7)}$$

$$ESV_{flti} = \left(ESV_p \times A_p\right) + \left(ESV_e \times A_e\right) + \left(ESV_{pr} \times A_{pr}\right) + \left(ESV_{sc} \times A_{sc}\right) + \left(ESV_{mc} \times A_{mc}\right) + \left(ESV_{lc} \times A_{lc}\right) \quad \text{(Eq.8)}$$

$$ESV_{ti} = (CV_i \times A_i) \quad \text{(Eq.9)}$$

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$$ESV_{fe} = (ESV_{ti} - ESV_{flit}) \quad \text{(Eq.10)}$$

where, the ecosystem service values of patch, edge, perforated, small core, medium core and large core are represented by ESV_p, ESV_e, ESV_{pr}, ESV_{sc}, ESV_{mc}, ESV_{lc}, respectively. CV_i indicates the Coefficient value of ith LULC types. The weight values based on AHP are indicated by W_p, W_e, W_{pr}, W_{sc}, W_{mc}, W_{lc}, respectively. ESV_{flti}, ESV_{ti}, ESV_{fe} refer to the total value of ecosystem service of fragmented landscape, ecosystem service values of LULC in total and effect of fragmentation on ecosystem service respectively. A_i is the areal coverage of i^{th} wetland type and the A_P, A_e, A_{pr}, A_{sc}, A_{mc}, A_{lc} are the areal coverage of patch, edge, perforated, small core, medium core and large core, respectively.

Results

LULC mapping and validation

For the years 1990, 2000, and 2018, six LULC classes were identified, such as urban area, water bodies, dense forest, sparse forest, agricultural cropland, and scrubland (*Figure 3*). *Table 3* displays the area of six LULC classes that were computed. Around 1990 and 2018, the urban area grew from 1648.8 hectares to 6379.2 hectares. Vegetation covers the western portion of the study area. Dense forest coverage rose from 219.15 hectares in 1990 to 780.57 hectares in 2018 (*Table 4*). The sparse vegetation, on the other hand, covered 10250 ha in 1990, but has shrunk to 9963.72 ha in 2018 (*Table 4*). In 1990, the cropland area was 1644.75 ha, but by 2018 it had shrunk to 604.8 ha. Scrubland occupied half of the study area in 1990, but it was reduced to 24.07 percent in 2018. The study area has relatively few water bodies, and even those that do exist have been converted into other land uses over time. The region of water sources in 1990 was 17.28 ha, but it has now shrunk to 15.48 ha. Over the study period, the metropolitan region experienced the greatest transformation, going from 4.46 percent to 17.24 percent.

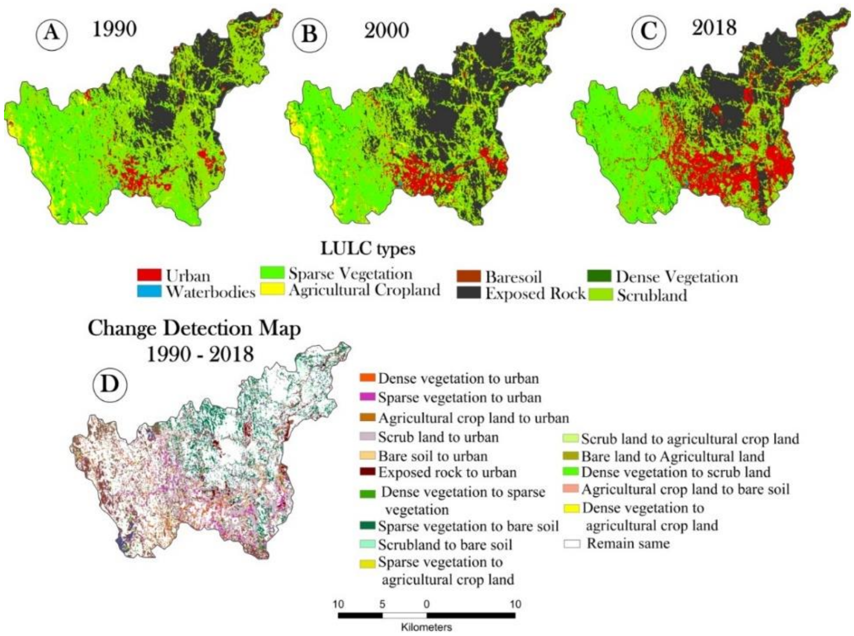

Figure 3. *Year wise change of LULC (1990 - 2018) (A,B,C) and change detection map (1990-2018)(D) of Abha watershed*

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Table 4. *Areal coverage of different LULC type from 1990 to 2018*

Land use types	1990		2000		2018	
	Area (ha.)	**Area in %**	**Area (ha.)**	**Area in %**	**Area (ha.)**	**Area in %**
Urban	1648.8	4.46	2211.93	5.98	6379.2	17.24
Water bodies	17.28	0.05	18.36	0.05	15.48	0.04
Dense vegetation	219.15	0.59	435.69	1.18	780.57	2.11
Sparse vegetation	10250.8	27.70	8058.33	21.78	9963.72	26.93
Agri. Cropland	1644.75	4.45	1666.71	4.50	604.8	1.63
Scrubland	15173.6	41.01	11928.1	32.24	8907.93	24.07
Bare soil	100.35	0.27	291.96	0.79	237.87	0.64
Exposed rock	7946.37	21.48	12390	33.49	10111.5	27.33
Total	37001.08	100	37001.08	100	37001.08	100

The LULC maps were validated using the Kappa coefficient. We calculated the kappa coefficient values for 1990 and 2018 using 250 Google Earth reference points and 60 ground control points. The kappa coefficient for all of the LULC maps is greater than 0.84, suggesting very strong conformity between classified maps and ground reality (K=>0.84).

Analysis of LULC dynamics

The urban area has improved dramatically from 5.98 percent to 17.24 percent, according to the findings of the change detection. Sparse vegetation, on the other hand, declined from 27.70 percent to 26.93 percent over the study period (*Figure 3D*). Scrubland also substantially reduced from 41.01% to 27.07%. Agricultural land, in addition to other natural resources, also declined spatially from 4.50 to 1.63 percent. The field of research includes smaller parts of waterbodies, but during the study time it was transformed. The rate has dropped from 0.05% to 0.04% (*Figure 3D*). The results showed that vegetation cover in terms of dense vegetation, sparse vegetation, and sparse land deteriorated over time as a result of urbanisation or human activity (*Figure 3D*).

Ecosystem service values (ESV) and its change

Total ESV maps for 1990, 2000, and 2018 were generated at a spatial scale using the coefficient of Costanza et al. (2014) (*Figure 4A-C*). The results revealed that the ESV of water bodies, sparse forest, agricultural land, and scrubland decreased, while the ESV of the urban area increased significantly. We calculated ESV for all LULC classes in 1990, 2000, and 2018. Then, during the periods from 1990-2000 and 2000-2018, we measured changes to the ESV for all LULC classes as defined in *Table 5*. Results showed that there was a declining trend of -3, 0.01, 0.8, - 0.1 and -43 million USD/y, respectively, in the period 1990–2000 for urban area, water bodies, dense vegetation, agricultural cropland and scrubland (*Table 5*). While there has been an increasing trend in the ESV in sparse vegetation of $9 million per year. However, since bare soil and exposed rocks did not have any ESV, these LULC classes did not see any changes. The ESV of water bodies, agricultural cropland, and scrubland increased by 0.03, 5, and 12 million USD/y, respectively, in the second phase (2000 to 2018) (*Table 3*). The ESV of the urban area, dense vegetation, and sparse vegetation, on the other hand, have demonstrated negative trends of -27 million USD/y, -1 million USD/y, and -7 million USD/y, respectively. ESV from urban areas was found to have risen in both phases.

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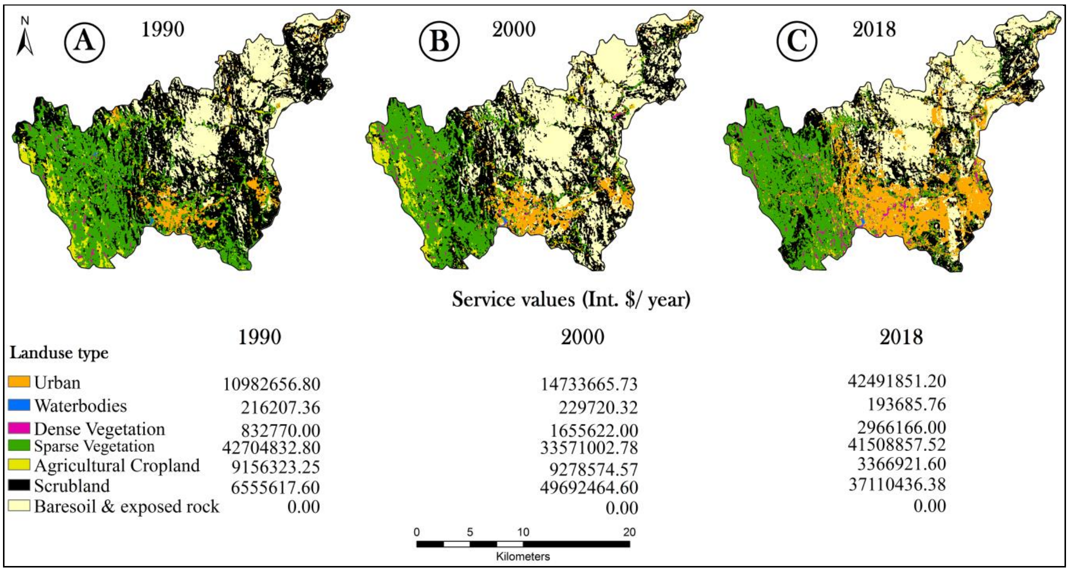

Figure 4. *Estimated total ESV since 1990-2018 by using the CV of Costanza (2014)*

Table 5. *Total ESV Change from 1990 to 2018 obtained from the CV of Costanza et al.*

Land use types	ESV Change (1990 – 2000) ESV (USD /yr)	ESV Change (2000 – 2018) ESV (USD /yr)
Urban	-3751008.93	-27758185.47
Water body	-13512.96	36034.56
Dense vegetation	-822852	-1310544
Sparse vegetation	9133830.02	-7937854.74
Agricultural cropland	-122251.32	5911652.97
Scrubland	-43371142.84	12582028.22
Bare soil	0	0
Exposed rock	0	0

Analysis of the effect of landscape fragmentation on ESV

Figure 5 shows the fragmentation state of the landscape of the years considered and its associated ESVs. *Table 6* presented the measured ESV of various fragmented categories such as patch, edge, perforated, small, medium, and large core of six LULC categories. During the study period, the area under edge was higher in the urban area, followed by patch, small core (<250 acres), and perforated (*Figure 5A,B,C*), indicating that the new urban area has been expanded around the main city and into natural resources, such as vegetation cover. During the study period, the region underneath water bodies was fragmented. The Edge of water bodies had the highest amount of area. In 1991 the area was 9.81 hectares and in 2000 and 2018 it steadily fell to 11.7 ha and 10.98 ha (*Table 6*). Dense vegetation had the highest area, followed by an edge between 1990 and 2018 (*Figure 5A,B,C*), indicating a considerable fragmentation of the dense forest over the years. During periods of study, the large core and medium core of sparse vegetation found that the sparse vegetation remained exactly the same throughout the study period. It was not substantially fragmented. In the case of agricultural cropland, the edge and

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patch had the greatest area as compared to another fragmented group in 1990 (patch 570.33 ha, edge 694.26 ha), 2000 (patch 599.13 ha, edge 656.37 ha), and 2018 (patch 545.31 ha, edge 50.31 ha) (*Table 6*). In the case of scrubland, the patch and edge occupied the most area and demonstrated a growing pattern over time (patch area: 1095.75 ha in 1990, 1256.13 ha in 2000, 1743.3 ha in 2018).

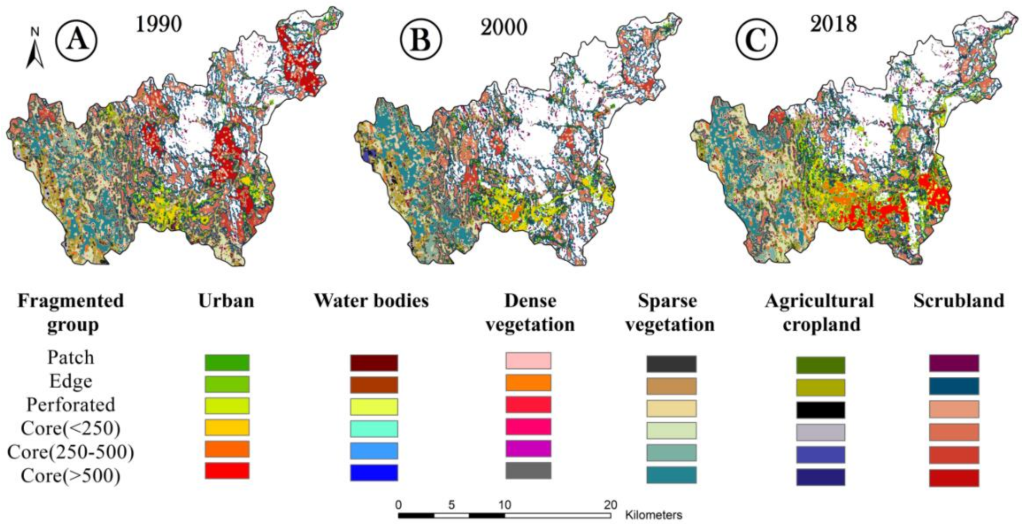

Figure 5. *Different fragmented landscape categories year wise*

The ESV of six LULC classes was calculated using the fragmentation integrated approach in 1990, 2000, and 2018 (*Table 7*). By taking fragmentation into account, the ESV of urban areas decreased during the study period (*Table 7*). In 1990, 2000, and 2018, the total ESV of urban areas was 9 million USD/year, 11 million USD/year, and 28 million USD/year, respectively (*Table 7*). As a result, the study area observed an increasing trend in projected ESV for urban areas. Since the spatial extension of water bodies was negligible, the ESV would be insignificant as well. Despite having a small spatial coverage, the ESV of water bodies showed a declining trend. This is a cause of concern about environmental changes. Over the study period, the ESV of dense vegetation changed by 0.7 million USD/year, 1 million USD/year, and 2 million USD/year, while the ESV of sparse vegetation changed by 27 million USD/year, 18 million USD/year, and 3 million USD/year (*Table 7*). For the years 1990, 2000, and 2018, the ESV adjustment after fragmentation for agricultural cropland was 0.7 million USD/year, 7 million USD/year, and 3 million USD/year, respectively (*Table 7*).

Scrubland's ESV adjustment was 12 million USD/year in 1990, 40 million USD/year in 2000, and 31 million USD/year in 2018 (*Table 7*). According to the results of the study, the ESV of all LULC classes has decreased after integrating the fragmentation effect (*Table 7*). In 1990, the largest change in ESV was seen in sparse vegetation (27 million USD/year), followed by urban areas (9 million USD/year) and agricultural cropland (7 million USD/year). Scrubland (40 million USD/year) and urban areas (31 million USD/year) witnessed the largest ESV transition between 2000 and 2018 (*Table 7*).

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Table 6. *Area of landscape fragmentation and estimated ESV in USD/year*

1990

Fragmented group	Urban		Water bodies		Dense vegetation		Sparse vegetation		Agricultural cropland		Scrubland	
	Area in ha.	ESV	Area in ha.	ESV	Area in ha.	ESV	Area in ha.	ESV	Area in ha.	ESV	Area in ha.	ESV
Patch	427.59	122471.6	5.94	3195.815	162.36	26529.62	782.91	140248.9	570.33	136526.2	1095.75	196290.5
Edge	774.45	557130	9.81	13256.21	52.29	21459.82	4054.59	1824274	694.26	417414.1	6617.61	2977448
Perforated	110.43	154470.6	0	0	0	0	1845.36	1614432	60.84	71126.22	2154.96	1885288
Core(<250 acres)	336.33	831149.1	1.53	7102.187	4.5	6344.1	1159.83	1792617	319.32	659509.8	2601.27	4020486
Core (<250-500acres)	0	0	0	0	0	0	193.86	500724.9	0	0	788.49	2036607
Core (>500acres)	0	0	0	0	0	0	2214.27	9224649	0	0	1915.47	7979848
Total	1648.8		17.28		219.15		10250.82		1644.75		15173.55	

2000

Fragmented group	Urban		Water bodies		Dense vegetation		Sparse vegetation		Agricultural cropland		Scrubland	
	Area in ha.	ESV	Area in ha.	ESV	Area in ha.	ESV	Area in ha.	ESV	Area in ha.	ESV	Area in ha.	ESV
Patch	485.19	138969.6	3.51	1888.436	350.55	57279.87	996.12	178442.9	599.13	143420.3	1256.13	225020.6
Edge	964.44	693806.6	11.7	15810.16	76.86	31543.34	2532.33	1139366	656.37	394633.3	6126.12	2756313
Perforated	165.24	231139.4	0	0	0	0	1175.94	1028783	94.95	111003.2	1567.8	1371606
Core(<250 acres)	445.68	1101378	3.15	14622.15	8.28	11673.14	531.18	820984.4	208.26	430131.2	2385.09	3686362
Core (<250-500acres)	151.38	625172.2	0	0	0	0	184.68	477013.7	108	372766.3	592.92	1531465
Core (>500acres)	0	0	0	0	0	0	2638.08	10990241	0	0	0	0
Total	2211.93		18.36		435.69		8058.33		1666.71		11928.06	

2018

Fragmented group	Urban		Water bodies		Dense vegetation		Sparse vegetation		Agricultural cropland		Scrubland	
	Area in ha.	ESV	Area in ha.	ESV	Area in ha.	ESV	Area in ha.	ESV	Area in ha.	ESV	Area in ha.	ESV
Patch	1239.03	354886.7	1.98	1065.272	512.82	83794.79	1305.09	233791.2	545.31	130536.9	1743.3	312291.3
Edge	2476.53	1781586	10.98	14837.23	250.38	102756	2476.08	1114058	50.31	30248.18	4421.7	1989447
Perforated	590.4	825857.4	0	0	1.44	1149.12	2336.49	2044102	5.49	6418.194	871.83	762729.2
Core(<250 acres)	612.45	1513505	2.52	11697.72	15.93	22458.11	978.75	1512742	3.69	7621.167	1744.74	2696646
Core (<250-500acres)	268.65	1109476	0	0	0	0	264.24	682510.8	0	0	126.36	326377.8
Core (>500acres)	1192.14	7940845	0	0	0	0	2603.07	10844390	0	0	0	0
Total	6379.2		15.48		780.57		9963.72		604.8		8907.93	

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Table 7. *Computed ESV gap after fragmentation in different LULC units*

Land use types	1990	2000	2018
	ESV change (USD /year)	ESV change (USD /year)	ESV change (USD /year)
Urban	9317435.5	11943199.93	28965695.1
Water body	192653.148	197399.574	166085.538
Dense vegetation	778436.46	1555125.65	2756007.98
Sparse vegetation	27607887	18936171.78	16431594
Agricultural crop land	7871746.93	7826620.27	3192097.159
Scrubland	12774645.74	40121698	31022945.08
Bare soil	0	0	0
Exposed rock	0	0	0

Discussion

It is evident from the research that the study region has experienced massive LULC dynamics over the last 30 years. Land use change has been noted in all LULC classes. Other land use types, such as sparse vegetation, scrubland, and dense vegetation, have been reclaimed by the built-up area as a result of the rapid and incremental urbanisation process. As a result of this type of urbanisation trend, environmental changes have worsened, affecting the landscape as well as biodiversity (Dong and Xu, 2019). Agricultural land, on the other hand, showed a declining trend over time. These results are primarily in contrast to those of other researchers (Han and Song, 2019). The reason behind the decline in agriculture is that the study area is a hilly region, so a limited amount of flat or slope-based areas for conventional agriculture are available. Since the year 2000, urbanisation has resulted in the conversion of certain agricultural lands. Rapid and unscientific urbanisation of mountainous regions is a cause for concern, since the mountain environment is regarded as one of the most natural locations on the world (Han and Song, 2019). While the conversion of vegetative land to other commercial land has been noted in other mountainous regions as well (Corton et al., 2020; Shi et al., 2020). Previous research has found that one of the causes of the declination of the vegetative area and other natural resources is the growth of the settlement area. The study area therefore noted an enormous shift in land use that resulted in a fragmentation of the landscape, particularly vegetation cover and other natural resources. This landscape fragmentation would have an effect on the ESV, which in turn has an impact on the environment as well as the livelihood status of the local people.

Based on the interpretation of ESV changes, the study area found a declining trend in natural resources such as vegetation cover and water bodies over time. The ESV in urban areas has been rising. This has occurred as natural land covers, such as dense vegetation, sparse vegetation, agricultural lands, and water bodies, have been transformed into anthropogenic land uses, such as settlement areas and barren land. Not only in this region, but also in other countries, such as India, China, Bangladesh, etc., this condition has been noticed. Because of the high demand, unscientific resource discovery and mismanagement lead ESV to decline over time. The total ESV of 2.43 billion USD in northeast China was damaged in just 35 years because the grasslands were produced for minimum benefit (Wang et al., 2015). Similarly, ESV from natural land cover has

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decreased in the present study area. With the advent of commercial operations barren land and exposed rock have declined over time (Mallick et al., 2014).

The present study, on the other hand, stated that the ESV was estimated using the global coefficient given by Costanza et al. (2014) and newly developed fragmentation corrected weights. The coefficient ESV and the fragmentation corrected ESV were found to be quite altered. The integration of fragmentation and weightage by AHP has improved the accuracy of ESV estimation.

Conclusion

This analysis calculated the cumulative ESV of various land use and land cover units over time, as well as the impact of fragmentation on ESV. The LULC change was discovered. The LULC transformation affects the ESV of this region. Research has however been done on the mountainous areas, but changes to ESV and LULC would also influence the plain climate. In addition, this analysis also modelled the fragmentation effect on ESV. The declining trend for ESV was observed to be diminished by the increased fragmentation of natural land use classes. The effect of fragmentation is measured by calculating the distance between ESVs without and with fragmentation. In dealing with this problem, the calculation of ESV gain or loss and its spatial distribution is novel in this work.

The large core areas were considered to be in ecologically good shape in order to provide the ESV. The theoretical weight was assigned for determining the effect of fragmentation based on this definition and expert opinion. It did better when calculating the ESV. Although the current research has certain drawbacks, such as the use of coarse resolution satellite images, the use of conventional MLC classifiers, and a smaller field survey. One challenge for this research is obtaining cloud-free images, without which we cannot accurately evaluate the area, and this is one of the study's main limitations. Many times, these images provided inaccurate results due to spectral mixing in the zone of hill shadow. Obtaining a specific result for the truth of the field was another obstacle to overcome for this work. However, in the future, ESV estimation can be enhanced by addressing the issues listed above, such as high-resolution satellite images, detailed field surveys, and the use of machine learning algorithms, deep learning algorithms for LULC classification, and sophisticated econometric models.

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Conflicts of Interests. The authors declare no conflict of interests.

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