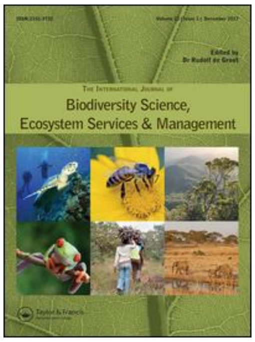

Taylor & Francis
Taylor & Francis Group

International Journal of Biodiversity Science, Ecosystem Services & Management

ISSN: 2151-3732 (Print) 2151-3740 (Online) Journal homepage: http://www.tandfonline.com/loi/tbsm21

Changes in land use and ecosystem services in tropical forest areas: a case study in Andes mountains of Ecuador

Mattias Gaglio , Vassilis George Aschonitis, Marta Maria Mancuso, Juan Pablo Reyes Puig, Francisco Moscoso, Giuseppe Castaldelli & Elisa Anna Fano

To cite this article: Mattias Gaglio , Vassilis George Aschonitis, Marta Maria Mancuso, Juan Pablo Reyes Puig, Francisco Moscoso, Giuseppe Castaldelli & Elisa Anna Fano (2017) Changes in land use and ecosystem services in tropical forest areas: a case study in Andes mountains of Ecuador, International Journal of Biodiversity Science, Ecosystem Services & Management, 13:1, 264-279, DOI: 10.1080/21513732.2017.1345980

To link to this article: http://dx.doi.org/10.1080/21513732.2017.1345980

Published online: 05 Jul 2017.

INTERNATIONAL JOURNAL OF BIODIVERSITY SCIENCE, ECOSYSTEM SERVICES & MANAGEMENT, 2017
VOL. 13, NO. 1, 264–279
https://doi.org/10.1080/21513732.2017.1345980

Taylor & Francis
Taylor & Francis Group

OPEN ACCESS Check for updates

Changes in land use and ecosystem services in tropical forest areas: a case study in Andes mountains of Ecuador

Mattias Gaglio [a], Vassilis George Aschonitis [a], Marta Maria Mancuso [a], Juan Pablo Reyes Puig [b], Francisco Moscoso [c], Giuseppe Castaldelli [a] and Elisa Anna Fano [a]

[a]Department of Life Sciences and Biotechnology, Section of Evolutionary Biology, University of Ferrara, Ferrara, Italy; [b]Fundacion Oscar Efren Reyes-Fundacion EcoMinga, Baños, Ecuador; [c]School of Geography, Pontificia Univerisidad Catolica del Ecuador (PUCE), Quito, Ecuador

ABSTRACT
Tropical Andes are subjected to severe land use/land cover (LULC) changes that significantly alter the capacity of the landscape to provide ecological functions for supporting human well-being. The aim of the study is (a) to investigate the LULC changes in the Ecological Corridor Llaganantes-Sangay (Corredor Ecológico Llanganates-Sangay) (Central Ecuador), a buffer semi-protected area, during the period 2000–2014 and (b) to analyse their possible consequences on ecosystem services (ESs) provision. The analysis was performed using LULC maps of 2000, 2008 and 2014. ESs were analysed using the 'landscape capacity' index, which is based on a multi-criteria assessment framework. The study captured an extremely rapid LULC transition from croplands to pastures during 2008–2014 below the 2000-m altitude, which was followed by a respective rapid socio-economic change of the local society. The landscape index changes were insignificant showing a slight decrease (−1.92%) during 2000–2014. Although the overall coverage of natural ecosystems slightly increased during 2000–2014, it was found that the passive landscape conservation might not be sufficient to maintain ESs provision. This was justified by the different ESs contribution between forest types but also by urbanization, agriculture abandonment and pasture expansion.

ARTICLE HISTORY
Received 26 July 2016
Accepted 19 June 2017

EDITED BY
Neville Crossman

KEYWORDS
Land use change; ecosystem services; landscape capacity index; tropical forests; pastures; agricultural land abandonment; protected area

1. Introduction

The provision of ecosystem services (ESs) is strictly related to land use/land cover (LULC) (Costanza et al. 1997; Metzger et al. 2006). ESs can be strongly affected by changes in LULC patterns, practices, intensity and trade-offs (Fu et al. 2015; Gissi et al. 2016; Gaglio et al. 2017). Despite the fact that LULC changes are ruled by drivers acting at regional or continental extent, the provision of ESs is relevant at different smaller scales (Hein et al. 2006). This scale mismatch results in a process of change that does not pay the proper attention to ecosystem conversions and their consequences. Moreover, ecological structures and functions vary along altitudinal gradients together with the variation of ecosystems and environmental conditions (Coûteaux et al. 2002; Kitayama and Aiba 2002; Moser et al. 2011), introducing an additional dimension to the ESs assessment framework.

The contribution of the majority of ESs to human well-being is not often considered or is underestimated, while humans are prone to exert pressures and changes in LULC with the aim to maximize the provision of one or few ESs, leading to a decline or loss of many others. This phenomenon is widespread around the globe (MA 2005; Ellis et al. 2010), but it is particularly severe in tropical regions of developing countries under the pressure of strong socio-economic changes (Lambin et al. 2003; Curatola Fernandez et al. 2015). An example is the tropical Andes of Ecuador, which are characterized by landscapes with peculiar climatic and topographic conditions where human settlements both affect and depend on natural ecosystems. This region is an extraordinary biodiversity hotspot (Jørgensen et al. 2011; Bendix et al. 2013) that has experienced forest clearance and land degradation since centuries (Valencia et al. 1999; Etter et al. 2008; Bare and Ashton 2016).

For the mitigation of the dramatic deforestation rate of the country (Mosandl et al. 2008), Ecuadorian government promoted incentive-based policies for the conservation of native forests, such as the Socio-Bosque program (Bertzky et al. 2010), as well as the establishment of several protected areas (Keating 2007; Cuenca et al. 2016). The establishment of several protected areas is designated to conserve natural values and processes and can significantly support numerous ESs (Willemen et al. 2013). On the other hand, such conservation activities do not always guarantee the livelihood of local populations, which is mainly supported by food production from croplands, raw materials production from forests and

CONTACT Vassilis George Aschonitis schvls@unife.it
Supplemental data for this article can be accessed at https://doi.org/10.1080/21513732.2017.1345980.

livestock production based on pastures/grasslands (Kovacs et al. 2015). Despite the fact that protected areas seem to be effective for reducing deforestation in Ecuadorian Tropical Andean forests (Cuenca et al. 2016), the outcome of conservation efforts on the capacity of these areas to support human well-being needs to be investigated. Non-natural ecosystems contribute to the provision of ESs (Jose 2009; Porter et al. 2009; Breuste et al. 2013; Rodríguez-Ortega et al. 2014) for which landscapes designed for conservation should also consider these. Overall, the positive correlation between nature conservation and ESs provision is not always observed and should be assessed based on the contribution of ecological functions of both natural and non-natural ecosystems.

Moreover, when analysing the role of environmental protection in maintaining ESs provision, ecosystems variability within the landscape should also be considered. Altitudinal gradients lead to high levels of environmental heterogeneity, which are conditioning factors of LULC transitions in Latin American countries (Redo et al. 2012). In fact, part of environmental heterogeneity seems to be associated to socio-economic and demographic variables (Redo et al. 2012; Aide et al. 2013), which are the main drivers for LULC changes (Sanchez-Cuervo and Aide 2013; Nanni and Grau 2014).

The aim of this study is (a) to examine the temporal LULC changes during a 14-year period (2000–2014) considering the altitudinal dimension in the Ecological Corridor Llaganantes-Sangay (Corredor Ecológico Llanganates-Sangay [CELS], a buffer area between two national parks in the tropical Andes of Central Ecuador), and (b) to assess their consequences on the ESs provision at landscape scale considering both natural and non-natural ecosystems.

2. Material and methods

2.1. Study area

The study area is the Ecological Corridor Llaganantes-Sangay (CELS). It is a transitional area in the Central Ecuador between the Eastern Cordillera of the Andes and the western Amazon forest covering about ~42,850 ha. The study area is a buffer zone between two national parks (the Llaganantes National Park at north and the Sangay National Park at south) (Figure 1(a)) and it is shared between five municipalities (*parroquias*): Rio Verde (8%) and Rio Negro (47%), Cumandà (23%), Mera (19%) and La Shell (3%). The altitude ranges between 960 and 3756 m above the sea level (Figure 1(b)) and the climate belongs to the Af class (Tropical Rainforest) according

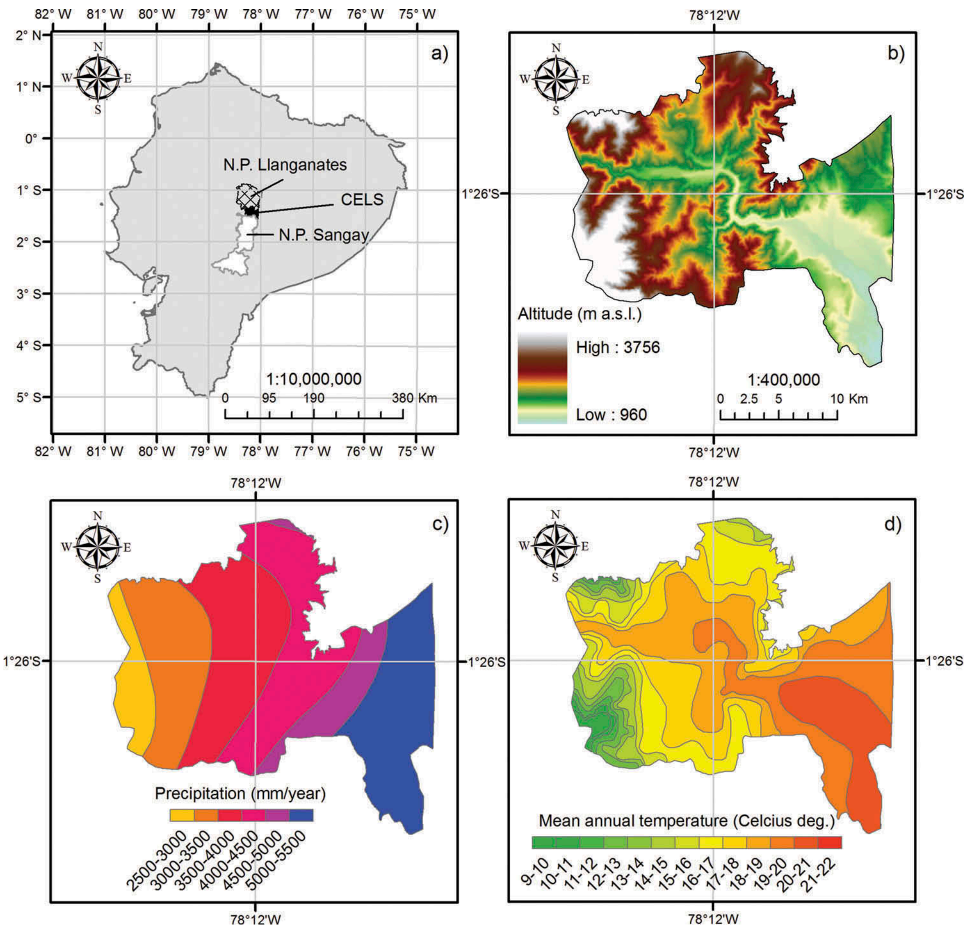

Figure 1. (a) Location of CELS area in Ecuador, (b) altitude, (c) mean annual precipitation and (d) and mean annual temperature.

to Köppen classification (Peel et al. 2007). The mean annual precipitation and temperature show a very steep transition to higher values towards East with ranges 2500–5500 mm/year and 9–22°C, respectively (Figure 1(c,d)). The strong relief and steep slopes favour the occurrence of highly differentiated habitats with very distinguishable zonation that results in high animal and plant biodiversity (Viteri et al. 2002). Animal biodiversity in CELS accounts for 101 mammals, 242 birds, 49 amphibians and 30 reptiles species. Plant endemism accounts for 195 endemic species in Pastaza watershed, from which 181 have been recorded in the area between Baños and Puyo, with a perspective of increasing the record in the next years (Yánez-Muñoz et al. 2013; Yaguache 2014).

The CELS was established in 2002 with the support of World Wildlife Fund. Nevertheless, this area is not under a true coordinated protection as it happens in the cases of Llaganantes and Sangay National Parks. The EcoMinga and Socio-Bosque foundations established additional conservation areas within the CELS that cover only 8000 ha (19% of total area). The inclusion of additional areas in the future will depend on stakeholder awareness for implementing the development and application of proper incentives. The economy of CELS is mainly based on agricultural activities (mainly orchards and annual crops), tourism and timber production (Yaguache 2014), which support a population of about 13,000 people (INEC 2010). Puyo and Shell are the larger urban systems located at the south-east edge of the territory and they are partly expanded inside CELS with a current population of ~37 thousand people. Puyo was outside CELS territory until 2002 but a clear expansion of city boundaries inside CELS is evident during the last years.

2.2. CELS ecosystems along the altitudinal gradient

Distinct spatial changes in natural ecosystems occurrence and structure appear following the increase of elevation in tropical mountains (Bruijnzeel et al. 2011). Ecosystems and related functions respond to changes in environmental gradients related to altitude, such as the decrease in temperature in higher altitudes. Lower temperatures and consequent weaker microbial activity, nutrient limitations and decrease of primary decomposers limit decomposition rate at increasing altitude (Coûteaux et al. 2002; Wilcke et al. 2002), and therefore promoting soil organic carbon accumulation (Maraun et al. 2008). Above-ground biomass, leaf area index and canopy height decrease with altitude while the restricted nutrient uptake leads to an increase in root production (Kottke et al. 2008; Unger et al. 2013). Although the above general patterns are widely documented, local conditions (e.g. slopes) can affect soil properties and their role on biomass production (Moser et al. 2011). In general, the environmental conditions of CELS promote a high natural ecosystems diversity that follows altitudinal patterns, with consequent variations in ecological functions provided at landscape scale. The forests of these ecological zones are also divided in four main categories based on altitude as follows: foothill forest – FF (<1300 m), lower mountain forest – LMF (1300–2000 m), cloud forest – CF (2000–2900 m) and higher mountain forest – HMF (>2900 m) (Vargas et al. 2000; Muriel 2008). A general description of main CELS ecosystems is provided in Table 1 and Figure 2. Urban centres within CELS are limited to few villages, where inhabitants have a rural lifestyle. More complex urban zones and infrastructures are located at the eastern part of CELS, in the municipality of Shell and Puyo, in proximity of Rio Amazonas airport. Water environments are mainly represented by the river Pastaza and by very few scattered water bodies. The river Pastaza flows from the Andes to Amazonian lowlands, crossing the CELS from west to east. A significant feature of CELS is that LULC changes are regulated by a traditional type of LULC rotation of croplands to pastures rotation and vice versa, which serves the provision of different food products depending on the needs of local population.

2.3. LULC maps and LULC change analysis

LULC change analysis was based on LULC maps of 2000, 2008, 2014. The maps were produced by the Ministry of Environment and the Ministry of Agriculture, Livestock, Aquaculture and Fisheries of Ecuador by using LANDSAT ETM+ for 2000 (MAE 2012), LANDSAT ETM+ and ASTER for 2008 (MAE 2014) and LANDSAT 8 and RapidEye satellite images for 2014 (MAE 2015). Except the last LULC map of 2014, which was developed by supervised classification using data from field surveys (at least 30 positions were monitored for each land-use type) (MAE 2015), the other two LULC maps of previous dates (2000 and 2008) were made using unsupervised classification. Seven LULC types were considered in the LULC change analysis according to the three maps: urban, bare soil, agricultural land, water bodies, páramo, pastures and native forests. The latter was further classified in four classes (FF, LMF, CC, HMF) according to Vargas et al. (2000) and Muriel (2008). Pastures include also a small portion of grasslands–shrublands, which are also used as areas for livestock grazing.

The analysis of LULC changes was performed using LULC transition matrices (TMs). In our study, TMs were developed directly by the LULC changes between 2000, 2008 and 2014 without using

Table 1. Main LULC types and their ecological functions inside CELS region.

LULC type	Description	Typical vegetation species	Main ecological functions
Agricultural land	Mainly orchards, located along water courses. Monocultures with fertilizers and pesticides application	*Solanum quitoense, Solanum betaceum*	Food provision
Pastures	Both cultivated and natural grasslands for feeding livestock. Stabling of animals is not performed while animal grazing is free following a rotation system by moving the animals from one to another area	*Pennisetum clandestinum, Lolium perenne*	Food supply to livestock for meat and milk production
Paramo	Typical ecosystem of Tropical Andes, located above 3400 m a.s.l. Vegetation can reach 50 cm height. Deep A-soil horizon where organic matter accumulation is favoured by the cold and wet climate and low atmospheric pressure (Hofstede et al. 2002; Buytaert et al. 2007). The humic and dark soils have excellent water infiltration and retention capacity (Buytaert et al. 2005, 2007)	*Perennial erbaceous plants* (e.g. Poacee)	Water regulation, medicinal resources
HMF	Trees can reach 10–15 m of height with thick and sometimes gnarled trunks, with adventitious roots occupying up to 70 m^2. Very steep slopes (>15°) affect soil organic carbon content	Clusia spp. in lower part (3200–3330 m). Sclerophyllus in upper part	Erosion prevention
CF	Trees reach a height of 15–25 m. The underwood is very rich and epiphytes and mosses are very abundant. Persistent presence of fog at the vegetation level, which significantly reduces incident solar radiation and evapotranspiration. The frequent contact between canopy and clouds increases water interception (i.e. horizontal rain) and water input to the system (Bendix et al. 2004; Célleri and Feyen 2009)	Melastomataceae, Solanaceae, Myrsinaceae, Aquifoliaceae, Araliaceae, Rubiaceae and several fern families	Water regulation, erosion prevention, biodiversity
LMF	The canopy height can reach 20–35 m tall with sporadic trees of 40 m. Composed by different layers such as canopy, subcanopy, shrub and herbaceous species. Epiphytes are more abundant than in lower altitudes, while lianas decrease in abundance and diversity (Valencia 1995)	Lauraceae, Rubiaceae, Melastomataceae and occasionally Moraceae	Aboveground biomass (carbon storage, charcoal and timber production)
FF	Forest transition between the foothills of Eastern Cordillera and Amazon forest. Substrate mainly composed by volcanic rocks and sediments of recent origins. The canopy height reaches 30 m and subcanopy and undergrowth are very dense (Vargas et al. 2000). The flatter zones near the River Pastaza are characterized by alluvial and terraced sediment deposits newly formed with high percentages of soil organic carbon. It presents extremely high biodiversity (Titira 1999; Yánez-Muñoz et al. 2010; Reyes-Puig et al. 2013)	Saurauia, Hedyosmum, Brunellia, Weinmannia	Aboveground biomass (carbon storage, charcoal and timber production), biodiversity, soil regulation

HMF: Higher mountain forest; CF: cloud mountain forest; LMF: lower mountain forest; FF: foothill forest.

probabilistic approaches in order to show the exact change from one LULC type to another (Wang et al. 2014; Gaglio et al. 2017). TMs compare the extent of LULC types between two time intervals (e.g. t_1 and t_2) providing the area of each LULC type that remained intact and the specific changes to other LULC types during t_1–t_2. The LULC maps of 2000, 2008 and 2014 correspond to three time intervals and for this reason, three TMs were built that correspond to the periods 2000–2008, 2008–2014 and 2000–2014.

Altitudinal patterns of LULC transitions were also investigated, according to four altitudinal zones (960–1300, 1300–2000, 2000–2900 and 2900–3756 m), which were delineated using a 30-m resolution digital elevation model. These zones were based on the altitudinal zonation between the forest classes FF (<1300 m), LMF (1300–2000 m), CF (2000–2900 m) and HMF (>2900 m) (Vargas et al. 2000; Muriel et al. 2008) (Table 1). Since the provision of forest ESs significantly varies along altitudinal zones (Becker et al. 2007; Leuschner et al. 2013), forested areas were further classified in four forest ecosystem classes according to specific altitudinal zones reported by Muriel (2008) and Vargas et al. (2000) for the study area. In this case, the specific altitudinal zones were used not only as a proxy to identify the different forest ecosystems but also to better describe the related services involved in the specific landscape transitions.

The significance of LULC changes was investigated through the comparison of proportion with χ^2 test for P value ≤ 0.01, using StatGraphics Centurion XV (StatPoint Inc.). For each altitudinal range, the comparison was performed between the proportion of each LULC type of the three dates 2000, 2008 and 2014 versus the proportion of the remaining LULC types (e.g. agricultural land vs. non-agricultural land). The null hypothesis was that the extension of the two classes did not change over the three dates. Also, an analysis of means (ANOM) plot with 99% confidence was applied. This procedure was not used to denote strict statistical differences between the years (e.g. as in the case of LSD

Figure 2. CELS ecosystems considered in the analysis: (a) foothill forest (FF), (b) lower mountain forest (LMF), (c) cloud forest (CF), (d) higher mountain forest (HMF), (e) paramo, (f) pastures and (g) agricultural land.

test in ANOVA) but to provide indications about the direction of the significant changes based on the deviation from the grand mean of the ANOM plots. Thus, the three codes a, b and c were used to denote the location of the proportion values from the three dates: above, inside and below the 99% confidence limits of ANOM plots (Fedrigotti et al. 2016).

Additionally, the annual rate of change for each LULC type was calculated by using the following equation (Puyravaud 2003):

$$\gamma = \left(\frac{1}{(t_2 - t_1)}\right) \times \ln\left(\frac{A_2}{A_1}\right) \quad (1)$$

where r is the annual rate of change of a given LULC, A_1 and A_2 are the area extension of a given ecosystem at the time t_1 and t_2, respectively.

2.4. ESs change assessment

According to the cascade model (Haines-Young and Potschin 2010), the provision of ESs depends on ecological functions that are exploited by humans to support their own well-being. Although the so-called ES delivery chain includes potential ESs stock (capacity), actual supply (flow) and beneficiaries (users demand) (Egarter Vigl et al. 2017), different mapping methods use proxies to assess the ecological function of LULCs (ESs capacity) assuming that they are directly or indirectly exploited by humans. For example, the 'benefit transfer' method is based on the assumption that a given spatial unit provides a set of ecological functions (e.g. Costanza et al. 1997, 2014; de Groot et al. 2012). The method proposed by Balthazar et al. (2015) is an adaptation of benefit transfer approach including the framework proposed by Koschke et al. (2012), where a set of ecological functions is used to assess the ESs provision (Kremen 2005). This method allows combining qualitative and semi-quantitative indicators to obtain a comprehensive index, sensitive to LULC changes, which expresses the overall capacity of a landscape to sustain the human well-being. Thus, the ESs analysis was performed at landscape level and the consequences of LULC change on ESs provided by CELS were assessed through the concept of 'landscape capacity' index (Burkhard et al. 2009; Koschke et al. 2012; Balthazar et al. 2015). This index uses a multi-criteria assessment framework, which is based on important biophysical indicators related to specific ESs for the development of a normalized score that avoids subjectivity due to qualitative expert judgment (Balthazar et al. 2015).

A scoring matrix of 11 indicators was developed for 7 LULC types: 2 non-natural (agricultural land and pastures) and 5 natural (foothill mountain forest, lower mountain forest, cloud mountain forest, higher mountain forest and paramo grassland) (Table 2). Other LULC types observed in CELS such as urban sites, bare soils and water environments were included in the maps but they were not considered in the ESs assessment. The rivers were not included in the ES assessment due to the lack of data for biophysical indicators. A main problem of ES assessment for the rivers of the study area is that the main courses have intermitted flow regulated by upstream dams while the small streams have very small area coverage and high discharge acting as intermediate links for ESs transfer among other land uses. In general, the riverbeds are mainly composed by large stones and when the discharge is low, large stony surfaces appear mainly in the west lowland part.

The 11 ecological indicators (Table 2) were selected according to their importance for the human well-being and data availability (MA 2005). Only peer-reviewed studies, technical reports and documents were considered in order to assign the bio-physical values to each indicator. Field surveys and personal communication from official sources were used to assess the number of touristic sites, the number of plant species used for medicinal resources and livestock supply capacity (see Supplementary material, sources and details about indicators presented in Table 2). When no local studies were present, we considered studies performed at national scale or studies carried out in similar environments (Andean regions).

The calculation of the landscape capacity index according to Balthazar et al. (2015) is performed by the following steps. In order to allow merging of indicators of different nature, the values of each indicator are standardized between 0 (no relevant capacity) and 5 (very high relevant capacity):

$$I_{\text{norm}} = \left(\frac{(I - I_{\min})}{(I_{\max} - I_{\min})}\right) \times 5 \qquad (2)$$

where I_{norm} is the standardized value from 0 to 5, I is the indicator value for a given ecosystem, $I_{\max}$ and $I_{\min}$ are the maximum and minimum values observed for the indicator, respectively. The overall potential of each LULC type is calculated as the sum of the standardized values of each indicator:

Table 2. Indicators used for the estimation of landscape capacity index and their relations to specific ESs in the study area.

Ecosystem services	Agricultural land	Pastures	Paramo	HMF	CF	LMF	FF	Indicators (unit)	References
Food production	1808	97	64	67	67	122	122	Monetary prices (US$2016 ha^{-1} $year^{-1}$)	Guayasamín Guanga (2015), Grimes et al. (1994), Kocian et al. (2011)
Medicinal resources	0	0	14	9	8	4	6	No. of suitable species	Local interview; de la Torre et al. (2008)
Livestock supply (bovines)	0	4	0	0	0	0	0	Cattle density (cattle ha^{-1})	Ministero de Agricultura, Ganaderia, Acuacultura y Pesca (pers. comm.)
Water regulation	646	648	933	741	837	748	748	Discharge (mm $year^{-1}$)	Balthazar et al. (2015), Crespo et al. (2010), Fleischbein et al. (2006)
Erosion prevention	43.0	20.0	100.0	97.6	97.6	98.3	98.2	Vegetation cover (C-Factor) (%)	MAE (2014), Molina et al. (2008), Ochoa-Cueva et al. (2015)
Soil structure	17.4	16.0	20.8	15.8	15.8	16.6	27.6	Organic matter (%)	WWF Ecuador (2014), MAE (2014), Potthast et al. (2010), Hofstede et al. (2002), WWF (2014)
Soil carbon storage	77	80	204	121	160	112	106	Organic matter (Mg C ha^{-1})	Moser et al. (2011), Hall et al. (2012), López-Ulloa et al. (2005)
Above-ground biomass	102.0	41.0	54.1	105.1	105.1	123.1	122.8	Biomass (Mg ha^{-1})	MAE (2014), McGroddy et al. (2015)
Biodiversity (vascular plants)	39	15	2000	2800	3000	2700	2500	Vascular plant richness [no. of species (ha^{-1})]	Jorgensen et al. (2011), Ministero de Agricultura, Ganaderia, Acuacultura y Pesca online database
Scenic quality	2	1	4	5	5	5	5	Relative scale	Burkhard et al. (2009)
Recreation/Education (Turism)	0	0	5	6	10	15	10	No. of touristic sites	Interviews of local stakeholders

HMF: Higher mountain forest; CF: cloud mountain forest; LMF: lower mountain forest; FF: foothill forest.

$$P_i = \sum I_{\text{norm } ij} \quad (3)$$

where P_i is the potential of an i LULC type to provide the considered indicator, and $I_{\text{norm } ij}$ is the standardized indicator value (Equation (2)) of an i LULC type for a j ES. Then, the landscape capacity index is calculated for each LULC type as follows:

$$L_i = A_i \times P_i \quad (4)$$

where L_i is the landscape capacity of i LULC type, A_i the area coverage of the i ecosystem (ha) and P_i (Equation (3)) is the potential of the i LULC type. It has to be noted that all the ESs were equally weighted to calculate the index. Finally, the total landscape capacity index is calculated as follows:

$$L = \sum L_i \quad (5)$$

where L is the total landscape capacity index and L_i the landscape capacity for the i LULC type (Equation (4)). The landscape capacity index is calculated for each of the three dates (2000, 2008, 2014), in order to assess the temporal variation of the ESs provided at landscape scale as consequence of the LULC changes occurred in the CELS.

Finally, the contribution of each LULC type to the total landscape capacity (L) was calculated as follows:

$$R_i = \frac{P_i \times A_i}{L} \quad (6)$$

where R_i is a ranking index, which expresses the contribution of the i LULC type to the total landscape capacity (L). The use of the specific index is based on the simplified version of elasticity coefficient or coefficient of sensitivity provided by Aschonitis et al. (2016) after recalculation of the terms in the original function provided by Kreuter et al. (2001). Aschonitis et al. (2016) found that the initial form of elasticity–sensitivity coefficient could be simplified because the ESs prices are considered always stable without being affected by changes in the demand.

3. Results

3.1. LULC changes

The maps of LULC for 2000, 2008 and 2014 are given in Figure 3. The TMs of LULC changes are given in Table 3 while the absolute, relative and annual rate of LULC changes are given in Table 4. Table 4 also includes the respective changes in the different forest classes (FF, LMF, CF and HMF). The most important changes during the whole period 2000–2014 were related to (a) agricultural land and pasture coverage rotations and (b) urban areas expansion (Tables 3 and 4). During 2000–2008, agricultural land gained 1138.64 ha, mainly from pastures and forest conversion. This trend was completely inverted during 2008–2014 when the 92.65% of the agricultural land

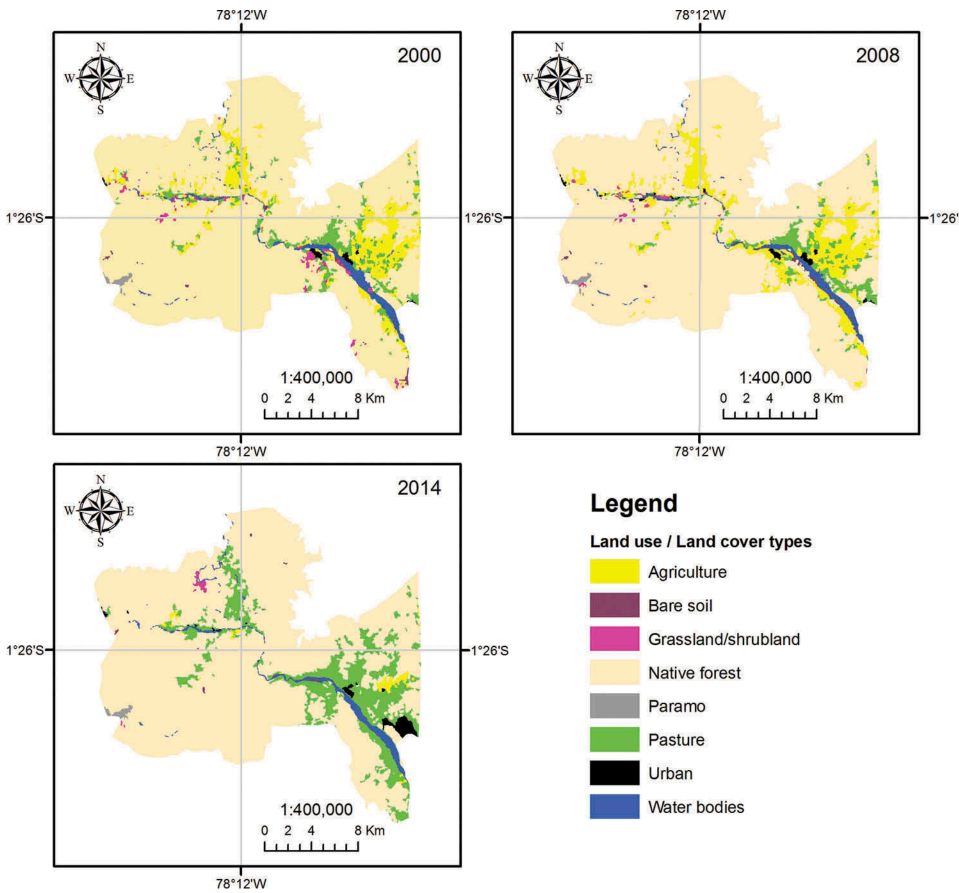

Figure 3. LULC maps of for the CELS region for the years 2000, 2008 and 2014.

Table 3. LULC transition matrices for (a) 2000–2008, (b) 2008–2014 and (c) (total study period) (values expressed in ha).

(a) Period 2000-2008

LULC type	Agricultural lands	Bare soils	Native forests	Paramo	Pastures*	Urban	Water bodies	Loss	Total 2000
Agricultural lands	**3320.57**[a]	8.36[b]	203.31[b]	0.00[b]	234.91[b]	17.99[b]	13.97[b]	478.54[c]	3799.10[d]
Bare soils	4.93[e]	**49.55**[a]	29.98	0.00	1.92	6.44	9.56	52.83	102.38
Native forests	649.38[e]	4.51	**34124.69**[a]	3.78	397.45	7.47	16.24	1078.83	35203.52
Paramo	0.00[e]	0.00	3.93	**97.47**[a]	7.15	0.00	0.00	11.08	108.55
Pastures*	916.55[e]	2.44	381.67	0.00	**1175.81**[a]	30.98	29.90	1361.55	2537.35
Urban	1.65[e]	0.00	0.30	0.00	0.73	**103.24**[a]	0.55	3.23	106.47
Water bodies	44.67[e]	37.83	21.83	0.00	31.18	7.05	**804.68**[a]	142.56	947.25
Gain	1617.18[f]	53.14	641.02	3.78	673.34	69.94	70.22		
Total 2008	4937.74[g]	102.70	34765.71	101.25	1849.15	173.18	874.90		42804.63

(b) Period 2008-2014

LULC type	Agricultural lands	Bare soils	Native forests	Paramo	Pastures*	Urban	Water bodies	Loss	Total 2008
Agricultural lands	**270.47**	7.09	1494.40	0.00	3021.82	53.55	90.42	4667.27	4937.74
Bare soils	0.00	**1.12**	33.32	0.00	19.78	0.01	48.47	101.57	102.70
Native forests	23.01	18.61	**33502.70**	2.77	1095.14	59.83	63.65	1263.01	34765.71
Paramo	0.00	0.00	3.56	**90.66**	7.02	0.00	0.00	10.59	101.25
Pastures*	65.30	0.33	246.60	13.73	**1342.22**	164.37	16.59	506.93	1849.15
Urban	2.92	0.00	11.31	0.00	73.13	**81.98**	3.84	91.20	173.18
Water bodies	1.11	12.25	70.65	0.00	153.30	1.96	**635.63**	239.27	874.90
Gain	92.34	38.28	1859.86	16.50	4370.20	279.71	222.97		
Total 2014	362.81	39.41	35362.55	107.16	5712.41	361.69	858.60		42804.63

(c) Period 2000-2014

LULC type	Agricultural lands	Bare soils	Native forests	Paramo	Pastures*	Urban	Water bodies	Loss	Total 2000
Agricultural lands	**238.21**	2.89	1145.86	0.00	2314.62	40.67	56.86	3560.90	3799.10
Bare soils	0.00	**0.00**	44.97	0.00	31.72	5.97	19.72	102.38	102.38
Native forests	27.79	19.76	**33383.31**	9.39	1531.29	160.37	71.62	1820.21	35203.52
Paramo	0.00	0.00	3.71	**97.77**	7.08	0.00	0.00	10.78	108.55
Pastures*	89.72	3.47	699.29	0.00	**1594.30**	103.76	46.81	943.05	2537.35
Urban	0.00	0.05	4.74	0.00	53.43	**45.56**	2.70	60.91	106.47
Water bodies	7.10	13.24	80.67	0.00	179.97	5.37	**660.90**	286.35	947.25
Gain	124.61	39.41	1979.24	9.39	4118.11	316.13	197.70		
Total 2014	362.81	39.41	35362.55	107.16	5712.41	361.69	858.60		42804.63

[a]The diagonal bold values show the area coverage of a LULC type, which remained intact during each period.
[b]The values of each row, except the bold ones, show how many hectares of a specific LULC type were converted to another LULC type (e.g. 234.91 ha of agricultural lands were converted to pastures during 2000–2008).
[c]Loss: The total sum of the values of each row, except the bold ones, which provides the total area of a specific LULC which was converted to another LULC types (e.g. 478.54 ha of agricultural lands were converted to other LULC types during 2000–2008).
[d]Total area of a LULC type at the beginning of the study period (e.g. the total coverage of agricultural lands was 3799.10 ha in 2000).
[e]The values of each column, except the bold ones, show how many hectares of a specific LULC type were gained (e.g. agricultural lands gained 916.55 ha after conversion of pastures to the specific LULC during 2000–2008).
[f]Gain: The total sum of the values of each column, except the bold values, which provides the total area which was gained for a specific LULC (e.g. agricultural lands gained a total area of 1617.18 ha during 2000–2008).
[g]Total area of a LULC type at the end of the study period (e.g. the total coverage of agricultural lands was 4937.74 ha in 2008).

Table 4. LULC changes observed in CELS during 2000–2008, 2008–2014 and 2000–2014 (total period).

	Absolute changes (ha)			Relative changes (%)			Annual rate of change (Equation (1)) (%)		
LULC type	2000–2008	2008–2014	2000–2014	2000–2008	2008–2014	2000–2014	2000–2008	2008–2014	2000–2014
Urban	66.71	188.51	255.22	62.66	108.85	239.71	6.08	12.27	8.74
Bare soil	0.32	−63.29	−62.97	0.31	−61.63	−61.51	0.04	−15.96	−6.82
Agricultural land	1138.64	−4574.93	−3436.29	29.97	−92.65	−90.45	3.28	−43.51	−16.78
Water bodies	−72.34	−16.31	−88.65	−7.64	−1.86	−9.36	−0.99	−0.31	−0.70
Paramo	−7.30	5.91	−1.39	−6.73	5.84	−1.28	−0.87	0.95	−0.09
Pastures	−688.21	3863.27	3175.06	−27.12	208.92	125.13	−3.95	18.80	5.80
Forest	−437.81	596.84	159.03	−1.24	1.72	0.45	−0.16	0.28	0.03
HMF	−9.92	2.54	−7.38	−1.11	0.29	−0.83	−0.14	0.05	−0.06
CF	−4.56	71.30	66.75	−0.06	1.01	0.95	−0.01	0.17	0.07
LMF	128.83	551.95	680.78	0.59	2.49	3.09	0.07	0.41	0.22
FF	−552.17	−28.95	−581.12	−10.53	−0.62	−11.09	−1.39	−0.10	−0.84

Absolute changes are expressed in ha, relative and annual changes in percentages. Annual change rates were calculated according to Equation (1). The forest area extension is given by the sum of the four forest ecosystems in which was further classified (see Table 2).
HMF: Higher mountain forest; CF: cloud forest; LMF: lower mountain forest; FF: foothill forest.

of 2008 was lost. After 2008, pastures showed the higher relative gain (208.92%) among all LULC types. During 2000–2008, deforestation occurred with an annual rate of 0.16%, while during 2008–2014, afforestation processes where observed with an annual forest gain of 0.28%. New fragmented urban zones were settled along the Pastaza river during 2000–2008, while the intense urbanization during 2008–2014 was due to the expansion of Mera, Shell and Puyo towns in the south-eastern part of CELS. Urban areas showed the most important relative increase in the total period (239.71%) during 2000–2014 (Tables 3 and 4).

The aforementioned general changes were not evenly distributed along altitudinal ranges (Table S.1 in the Supplementary material). The results of ANOM analysis based on the altitudinal zonation are given in Table 5. Human activities related to LULC typologies, such as agricultural land, pastures and urban areas, are mainly located in 960–1300 and 1300–2000 m zones. Therefore, these two altitudinal zones were mostly affected by LULC changes. Agricultural land significantly expanded during 2000–2008 versus forested areas within the 960–1300-m zone and versus pastures within 1300–2000 m zone (Tables 3–5), while during 2008–2014, an extensive decrease occurred at all altitudinal levels. Loss of agricultural land at 960–1300 m was due to a shift of land-use activity towards pastures, while afforestation phenomena were detected only at 1300–2000 m. In fact, pastures expansion within 960–1300 m zone affected the foothill forest causing further deforestation process during 2008–2014. Significant increase of forested areas was also observed in the 2000–2900-m zone, where cloud forest colonized previously cultivated land, pastures and bare soil. The LULC type of pastures is the only one showing significant changes in the upper altitudinal zone (2900–3756 m) (Table 5) because their coverage in this zone during 2000 was zero (Table S.1). Higher altitudes are less accessible for human activities, which are the main drivers of changes.

During the time span considered, and particularly during the period 2008–2014, LULC changes analysis showed a general migration of human activities to the

Table 5. LULC changes analysis using analysis of means at 99% confidence level.

Altitude (m a.s.l.)	χ^2 (df = 2)	*P* value	2000	2008	2014	χ^2 (df = 2)	*P* value	2000	2008	2014
		Urban					Bare soil			
2900–3756	–	–	–	–	–	2.63	0.2687	b	b	b
2000–2900	–	–	–	–	–	6.76	0.034	b	b	c
1300–2000	10.97	0.0041	c	a	b	11.34	0.0034	b	b	a
960–1300	178.9	<0.0001	c	c	a	45.91	<0.0001	a	a	c
		Forest					Agricultural land			
2900–3756	0.51	0.7744	b	b	b	2.51	0.2851	b	b	b
2000–2900	54.85	<0.0001	c	c	a	33.53	<0.0001	b	a	c
1300–2000	113.12	<0.0001	c	c	a	2073.69	<0.0001	a	a	c
960–1300	87.52	<0.0001	a	c	c	2132.4	<0.0001	a	a	c
		Water bodies					Paramo			
2900–3756	0	0.9985	b	b	b	0.32	0.8529	b	b	b
2000–2900	4.1	0.1287	b	b	b	–	–	–	–	–
1300–2000	3.65	0.1608	b	b	b	–	–	–	–	–
960–1300	7.48	0.0237	a	b	b	–	–	–	–	–

		Pastures			
Altitude (m a.s.l.)	χ^2 (df = 2)	*P* value	2000	2008	2014
2900–3756	15.03	0.0005	c	a	b
2000–2900	28.48	<0.0001	a	b	c
1300–2000	1415.04	<0.0001	c	c	a
960–1300	1785.98	<0.0001	c	c	a

The three codes a, b and c were used to denote the location of the proportion values from the three dates: above, inside and below the upper and lower 99% confidence limits.

lower altitudes, resulting in the re-naturalization of uplands.

Regarding the water bodies, the only significant change was a decrease within the lower altitudinal belt during 2000–2008 (Table 5) probably caused by the establishment of upstream dams for hydroelectric power generation.

3.2. Changes in ESs

The consequences of LULC changes in ESs provision were assessed through the quantification of a set of indicators to calculate the landscape capacity L index.

Table 6 presents the standardized values (Equation (2)) used for the calculation of the L index. The capacity to support ecological functions in CELS expressed by P_i (Equation (3)) for each LULC type is also given in Table 6. The natural ecosystems show higher P_i in comparison to the anthropic ones. Foothill forests present the larger potential to support human well-being, followed by the other forest types and paramo grassland. Pastures have the lower potential, mainly related to livestock supply, while their potential for other indicators is limited. Agricultural lands present more than double P_i value in comparison to pastures but less than half value if compared with natural ecosystems (Table 6).

The difference between the P_i values of agricultural land and pastures is mainly due to differences in soil-related functions and above-ground biomass production. The intensive grazing activity of cattle causes the decrease of soil coverage and organic matter content with detrimental effects on erosion prevention, soil structure and soil carbon storage. Marked differences in above-ground biomass can easily be identified because of the intensive characteristics of grazing management adopted by breeders, which do not allow the growing of trees and shrubs. Contrary, agricultural land in CELS is characterized by a considerable extension of orchards, which provide a good amount of above-ground biomass. Different values on scenic quality are due to the different scores proposed by Burkhard et al. (2009) for these two ecosystems.

The landscape capacity index for each LULC type L_i (Equation (4)) and the total value L (Equation (5)) for 2000, 2008 and 2014 are given in Table 7. The total landscape capacity L decreased by 0.42% during 2000–2008, by 1.51% during 2008–2014 and by 1.92% during 2000–2014. These L changes were quite small and mainly regulated by the transitions between agricultural lands and pastures, and urban areas expansion. The high and almost constant coverage of natural LULC types during 2000–2014 (84.71% for 2000, 83.50% for 2008 and 84.87% for 2014) was the main reason of small L changes. The % contribution of each ecosystem type R_i (Table 7) showed the importance of lower mountain forests (LMF) with a contribution ranging between 57% and 60% during the period 2000–2014.

From a qualitative point of view, even when the total landscape capacity (L) does not suffer any

Table 6. I_{norm} values (Equation (2)) for the ecosystem functions and total potential P_i of each ecosystem to provide ecological functions (Equation (3)).

Ecosystem service	Agricultural land	Pastures	Paramo	HMF	CF	LMF	FF
Food production	5.00	0.09	0.00	0.01	0.01	0.17	0.17
Medicinal resources	0.00	0.00	5.00	3.21	2.86	1.43	2.14
Livestock supply (bovine)	0.00	5.00	0.00	0.00	0.00	0.00	0.00
Water regulation	0.00	0.03	5.00	1.66	3.33	1.78	1.78
Erosion prevention	1.44	0.00	5.00	4.85	4.85	4.89	4.89
Soil structure	0.67	0.08	2.11	0.00	0.00	0.33	5.00
Soil carbon storage	0.00	0.12	5.00	1.73	3.27	1.38	1.14
Above-ground biomass	3.71	0.00	0.80	3.90	3.90	5.00	4.98
Biodiversity (vascular plants)	0.04	0.00	3.32	4.66	5.00	4.50	4.16
Scenic quality	1.25	0.00	3.75	5.00	5.00	5.00	5.00
Recreation/Education (tourism)	0.00	0.00	1.67	2.00	3.33	5.00	3.33
P_i	12.11	5.33	31.65	27.03	31.55	29.47	32.59

HMF: Higher mountain forest; CF: cloud mountain forest; LMF: lower mountain forest; FF: foothill forest.

Table 7. Landscape capacity index for each ecosystem type L_i (Equation (4)) and its total value L (Equation (5)) for 2000, 2008 and 2014.

	L_i			R_i		
Ecosystem type	2000	2008	2014	2000 (%)	2008 (%)	2014 (%)
Agricultural land	46008.94	59798.44	4393.84	4.07	5.32	0.40
Pastures	13529.41	9859.82	30459.13	1.20	0.88	2.75
Paramo	3435.95	3204.74	3391.89	0.30	0.28	0.31
HMF	24094.21	23826.09	23894.69	2.13	2.12	2.16
CF	222530.54	222386.70	224636.23	19.71	19.78	20.28
LMF	648853.07	652649.79	668916.22	57.46	58.04	60.39
FF	170827.45	152831.79	151888.18	15.13	13.59	13.71
Total L	1129279.58	1124557.37	1107580.18	100.00	100.00	100.00

Also the ranking index R_i is reported (Equation (6)).

significant changes, the LULC transitions determine qualitative changes in ESs provision. For example, the transition from agricultural land to pastures implies the change in provisioning services, with a decrease in crop-derived food and an increase in meat and milk production. Moreover, this transition causes a decrease in erosion prevention and soil structure maintenance, since croplands guarantee good and constant soil coverage compared to pastures subjected to intensive grazing.

No significant total *L* change could be detected also when the loss of forest at lower altitudes is offset by forest gain at higher altitudes. Nonetheless, a qualitative change in the indicators set, and therefore in ES provision capacity, occurs, since different functions are carried out by different forest ecosystems. Forest expansion at upper altitudes (HMF and CF) offers higher protection against soil erosion and better regulation of runoff, while the decrease of forested habitat at lower altitude (LMF and FF) results in loss of biodiversity, carbon storage (i.e. climate change mitigation) and potential for recreational services. The latter is higher for natural LULC types at lower altitudes, whose touristic sites are more accessible if compared with those located at higher and steeper zones.

3.3. Transitions of agricultural land to pasture

One of the most interesting issues of this research study is that LULC changes in CELS were regulated by an extremely high transition between different types of anthropic ecosystems. LULC transitions, where croplands, pastures and secondary vegetation replace each other, are commonly observed in the Andean region (Rodríguez Eraso et al. 2013), as well as in all the tropical part of South America (Wassenaar et al. 2007). The transition from agricultural land to pasture was the most relevant LULC change and mainly occurred during 2008–2014 covering an area of 3004.8 ha, equal to 7.02% of the total area (Figure 4). From the 3004.8-ha, the 69% was already agricultural land, 20% was covered by pastures and 11% was forest during 2000. This indicates that 31% of this area experienced a double conversion (pastures–agricultural–pastures or forest–agricultural–pastures) during 2000–2014. The forest loss during 2000–2008 was mainly observed in the altitudinal zone of 960–1300 m (foothill forest).

4. Discussion

The LULC changes observed in CELS highlight the typical pathway of changes in Ecuadorian Andean mountains. Deforestation typically occurs for wood or charcoal extraction (for 1 or 2 years), then the land parcel is converted to agriculture (2–5 years) and then to pasture (7–10 years), before returning the land to fallow for another 1–5 years (Luoma 2004). Rodríguez Eraso et al. (2013) described general patterns of change for Colombian Andes where

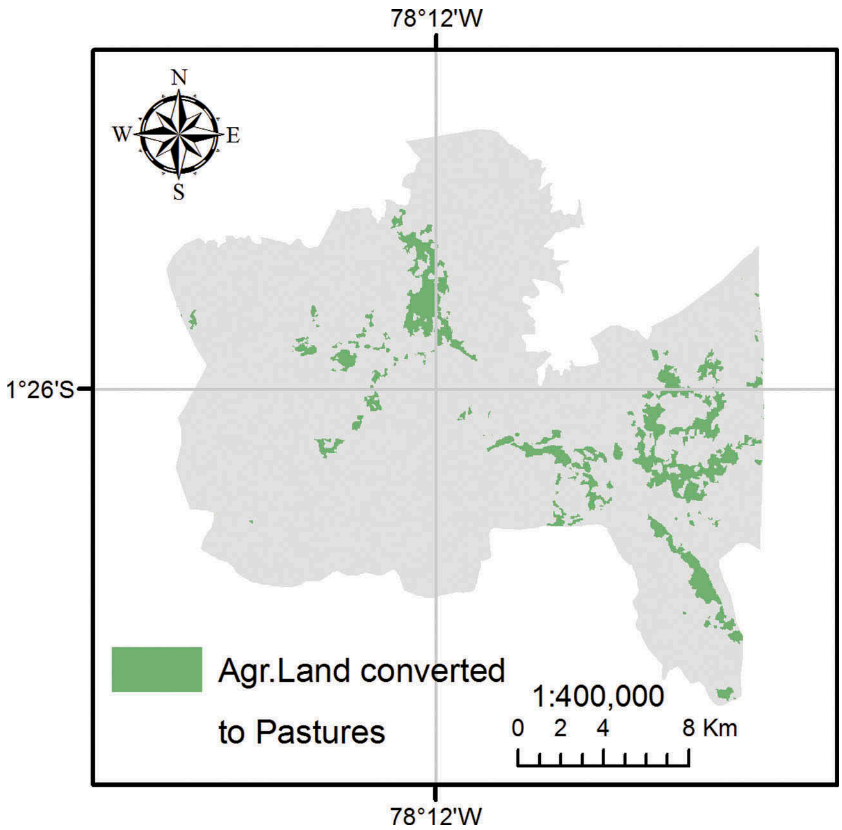

Figure 4. Agricultural land converted to pastures during 2008–2014.

abandoned agricultural areas evolve to secondary vegetation, where the latter is converted to pastures. The scarce amount of secondary vegetation observed in all the three LULC maps suggests that the conversion from agricultural land to pastures during 2008–2014 occurred very fast. In general, even when short time intervals were considered for the comparison of LULC, some intermediate stages between LULC changes could not be detected due to the very fast regeneration capacity of CELS ecosystems. Natural regeneration in tropical Andes is influenced by several factors related to the previous land use and management, such as seed availability and dispersion, presence of remnant vegetation, soil structure, light and water availability (Guariguata and Ostertag 2001; Günter et al. 2007; Lozada et al. 2007). In CELS, the natural regeneration in a native forest dominated landscape is fostered by the proximity of natural environment to cropland and pastures, the favourable temperatures and the constant precipitation throughout the year, resulting in up to 2 m of pioneer species growing after only 2 years (Yaguache 2014). Even though transitions in both directions between pastures and croplands are common in tropical landscapes (Wassenaar et al. 2007; Rodríguez Eraso et al. 2013), the massive conversion of agricultural land into pastures during 2008–2014 highlights the important role of this transition to respective changes in socio-economic conditions of the local population. In the case of agricultural land, the cultivation of Naranjilla, the most widespread cultivation in CELS, provides good yields between the second and fourth year but falls markedly after, forcing producers to abandon the plantation for about 10 years (Bajaña and Viteri 2002). Moreover, Naranjilla crops require the application of agro-chemicals in order to cope pests and fungal attacks (Ochoa and Ellis 2005), which affect economic profits. Conversely in the case of pastures, the cattle production offers economic flexibility and lower financial risks (Wassenaar et al. 2007), even if pasture degradation may occur in time as well (Fearnside 1989). Thus, conversion to pastures seems to be more economically sustainable in comparison to agricultural land since the economic contribution of the latter is reduced due to overexploitation and unsustainable practices, which decrease the soil fertility very fast.

Since anthropic environments are focused on the exploitation of one market-oriented ES (e.g. food production), the provision of other functions in the case of cultivated areas and pastures is just a 'side effect' that depends on management practices, which are not considered by farmers and breeders. The protection of mountain natural ecosystems and biodiversity seems to be already effective in CELS, despite the high deforestation rate detected at the country scale in Ecuador (Mosandl et al. 2008). Thus, strategies for supporting farmers towards more sustainable practices are needed, with the aim to avoid agricultural land abandonment and to manage the croplands capacity to support a wider set of ecological functions. These targets are partially discussed in the Landscape Restauration Plan of CELS (Yaguache 2014), which shapes the objective of improving the supply of ecosystem goods and services as well as to strengthen ecosystems resilience and adaptation capacity to climate change. The Plan also suggests the development and implementation of better productive practices and restoration priorities for the 37% of croplands and pastures during the next years, with the goal to increase productivity and to improve hydrological and biodiversity conditions in croplands and pastures. Namely, it suggests the use of mixed permanent crops (e.g. mandarins) with annual ones, the maintenance of orchards multicultures and dispersed trees in pastures (Yaguache 2014). If these measures were applied and extended to all agricultural lands and pastures, both market and non-market ESs provided by CELS would be significantly increased with noticeable benefits for the local population. Mixed crop systems, such as those with fruit trees and annual crops, can improve the ecological functions and sustain farmer's economic profits by reducing the need of chemicals applications. Unlike annual monocultures, the mix of permanent and annual crops provides a continuous vegetation coverage, which is found to exert a fundamental role in preventing soil erosion in Ecuadorian Andes (Molina et al. 2008). Permanent crops lead to an increase on soil organic matter content (Blanco-Canqui 2010), which significantly boosts soil fertility and regulates soil-water dynamics and microbial activities (Lal 2004). The maintenance of tree species diversity in orchards systems could avoid land degradation caused by monocultures of Naranjilla. Diverse and multi-strata orchards can provide additional benefits for biodiversity and biological control (Simon et al. 2009). In general, mixed crops show a better capacity to capture and use biophysical resources (Jahansooz et al. 2007) and to limit disease and pest organism (Perrin 1977; Sapoukhina et al. 2010), leading to a decrease in agrochemicals' requirements.

Regarding natural ecosystems, the analysis showed that foothill forest has the larger potential for ESs provision, while the lower mountain forest exhibits the greater contribution in ESs due to its large coverage. Taking into account these observations, forest management should consider these attributes in order to maintain a high contribution of non-market ESs, which sustain ecological quality. Fuelwood and charcoal are important forest products in Ecuador (Luoma 2004), and for this reason, alternative approaches to mitigate deforestation for such purposes are needed. An alternative approach for obtaining such products could be the use of trees in pastures. This practice was also found to be effective at reducing soil erosion (White and Maldonado 1991), improves biodiversity, and provides shadow and protection to livestock (Luoma 2004).

The uneven LULC changes of CELS determined by altitude are in line with the situation of other mountainous regions of Latin America (e.g. Redo et al. 2012; Nanni and Grau 2014). Nanni and Grau (2014) observed that the interaction between agriculture modernization, human demography and complex topographic gradients of northwestern Argentina has resulted in processes of both forest recovery in uplands and deforestation in lowland areas. Redo et al. (2012) observed that forest transitions in Central America were significantly associated to socio-economic development, but with strong asymmetry in rates and directions of change, which were largely dependent upon the biome where change was occurring. These asymmetric patterns of forest change should be evaluated during the development of strategies for conserving biodiversity and ecosystem services.

Finally, some limitations concerning the method used for ESs assessment should be considered. The benefit transfer approach does not consider the spatial position of ecosystems, simplifying the landscape description based on a simple sum of ecosystems. Moreover, possible bias can be introduced when data from different sources are collected. The landscape index minimizes the effect of such bias but also other effects associated to intra and inter-site variability, by normalizing the indicators (Equation (2)). At the same time, when altitudinal variability is considered, it provides an acceptable approximation of ESs variation.

5. Conclusions

This study provided a description of LULC and ESs changes in the CELS region. Although a 14-year study period may seem a relative short timespan for LULC change analysis, the study captured an extremely rapid LULC transition from croplands to pastures followed by a respective rapid socio-economic change of the local society, suggesting also its high degree of adaptability.

Although the overall coverage of natural ecosystems slightly increased during 2000–2014, confirming the effectiveness of forest protection in Ecuador, it was found that the passive landscape conservation focused on natural ecosystems and biodiversity may not be sufficient to maintain ESs. Urbanization, agriculture abandonment and pasture expansion using unsustainable practices are the main threat to the maintenance of ESs provision in CELS. Governance plans of CELS, such as the Landscape Restoration Plan, should focus more on management practices for croplands and pastures, including also organic cropping and more sustainable alternatives to chemicals applications, with the aim to guarantee both monetary incomes and high environmental standards for CELS population. The role of specific forest types on ESs provision was also highlighted providing significant information about forest conservation based on different altitudinal zones.

The framework applied in this study could support current and future plans of environmental and ESs governance. Moreover, more detailed future field studies are required in order to improve the knowledge of how the different ecosystems of CELS support ecological functions, according to environmental gradients (e.g. altitude). Finally, LULC changes also differentiate the interaction between ecosystems. Thus, the weights assigned to ecosystems functions should be updated considering the knowledge and experience gained by stakeholders and associated institutions participating in management plans.

Acknowledgements

The authors would like to thank the anonymous reviewers for their constructive comments, and Dr. Laura Scalvenzi of UNEP for making possible the cooperation between scientists of Italy and Ecuador.

Disclosure statement

No potential conflict of interest was reported by the authors.

ORCID

Mattias Gaglio http://orcid.org/0000-0002-2027-0732

References

Aide TM, Clark ML, Grau HR, López-Carr D, Levy MA, Redo D, Bonilla-Moheno M, Riner G, Andrade-Nuñez MG, Muñiz M. 2013. Deforestation and reforestation of Latin America and the Caribbean (2001–2010). Biotropica. 45(2):262–271.

Aschonitis VG, Gaglio M, Castaldelli G, Fano EA. 2016. Criticism on elasticity-sensitivity coefficient for assessing the robustness and sensitivity of ecosystem services values. Ecosyst Serv. 20:66–68.

Bajaña F, Viteri J. 2002. Plan Preliminar de Manejo del Corredor Ecológico Llanganates - Sangay [Preliminary management plan for the ecological corridor llanganates - Sangay]. Quito (Ecuador): Fundación Natura, Área de Ecosistemas Naturales y Biodiversidad.

Balthazar V, Vanacker V, Molina A, Lambin EF. 2015. Impacts of forest cover change on ecosystem services in high Andean mountains. Ecol Indic. 48:63–75.

Bare MC, Ashton MS. 2016. Growth of native tree species planted in montane reforestation projects in the Colombian and Ecuadorian Andes differs among site and species. New For. 47:333–355.

Becker A, Körner C, Brun J, Guisan A, Tappeiner U. 2007. Ecological and land use studies along elevational gradients. Mt Res Dev. 27:58–65.

Bendix J, Beck E, Bräuning A, Makeschin F, Mosandl R, Scheu S, Wilcke W, editors. 2013. Ecosystem services, biodiversity and environmental change in a tropical mountain ecosystem of South Ecuador. Berlin-Heidelberg: Springer–Verlag.
Bendix J, Fabian P, Rollenbeck R. 2004. Gradients of fog and rain in a tropical montane cloud forest of Southern Ecuador and its chemical composition. Proceedings 3rd International Conference Fog, Fog Collect Dew; Oct 11–15; Cape Town.
Bertzky M, Ravillous M, Araujo-Navas A, Kapos V, Carrióon D, Chiu M, Dickson B. 2010. Carbon, biodiversity and ecosystem services: exploring co-benefits. Ecuador. Cambridge (UK): UNEP-WCMC.
Blanco-Canqui H. 2010. Energy crops and their implications on soil and environment. Agron J. 102:403–419.
Breuste J, Haase D, Elmqvist T. 2013. Urban landscapes and ecosystem services. In: Wratten S, Sandhu H, Cullen R, Costanza R, editors. Ecosystem Services in Agricultural and Urban Landscapes. Oxford: A John Wiley & Sons; p. 83–104.
Bruijnzeel LA, Mulligan M, Scatena FN. 2011. Hydrometeorology of tropical montane cloud forests: emerging patterns. Hydrol Process. 25:465–498.
Burkhard B, Kroll F, Mueller F, Windhorst W. 2009. Landscapes' capacities to provide ecosystem services - a concept for land-cover based assessments. Landsc Online. 15:1–22.
Buytaert W, Iñiguez V, De Bièvre B. 2007. The effects of afforestation and cultivation on water yield in the Andean páramo. For Ecol Manage. 251:22–30.
Buytaert W, Wyseure G, De Bièvre B, Deckers J. 2005. The effect of land-use changes on the hydrological behaviour of Histic Andosols in south Ecuador. Hydrol Process. 19:3985–3997.
Célleri R, Feyen J. 2009. The hydrology of tropical Andean ecosystems: importance, knowledge status, and perspectives. Mt Res Dev. 29(4):350–355.
Costanza R, D'Arge R, de Groot R, Farber S, Grasso M, Hannon B, Limburg K, Naeem S, O'Neill RV, Paruelo J, et al. 1997. The value of the world's ecosystem services and natural capital. Nature. 387:253–260.
Costanza R, de Groot R, Sutton P, van der Ploeg S, Anderson SJ, Kubiszewski I, Farber S, Turner RK. 2014. Changes in the global value of ecosystem services. Glob Environ Chang. 26:152–158.
Coûteaux MM, Sarmiento L, Bottner P, Acevedo D, Thiéry JM. 2002. Decomposition of standard plant material along an altitudinal transect (65-3968 m) in the tropical Andes. Soil Biol Biochem. 34:69–78.
Crespo P, Celleri R, Buytaert W, Feyen J. 2010. Land use change impacts on the hydrology of wet Andean páramo ecocystems. Proceedings of the Workshop Status and Perspectives of Hydrology in Small Basins; Mar 30 – Apr 2; Germany. Goslar-Hahnenklee: IAHS Publications. p. 336.
Cuenca P, Arriagada R, Echeverría C. 2016. How much deforestation do protected areas avoid in tropical Andean landscapes? Environ Sci Policy. 56:56–66.
Curatola Fernandez GF, Obermeier WA, Gerique A, Lopez Sandoval MF, Lehnert LW, Thies B, Bendix J. 2015. Land cover change in the Andes of Southern Ecuador-Patterns and drivers. Remote Sens. 7:2509–2542.
de Groot R, Brander L, van der Ploeg S, Costanza R, Bernard F, Braat L, Christie M, Crossman N, Ghermandi A, Hein L, et al. 2012. Global estimates of the value of ecosystems and their services in monetary units. Ecosyst Serv. 1:50–61.
de la Torre L, Navarrete H, Macía MJ, Balslev H. 2008. Enciclopedia de las Plantas Útiles del Ecuador [Encyclopedia of the useful plants]. Quito: Herbario QCA & Herbario AAU.
Egarter Vigl L, Depellegrin D, Pereira P, de Groot R, Tappeiner U. 2017. Mapping the ecosystem service delivery chain: capacity, flow, and demand pertaining to aesthetic experiences in mountain landscapes. Sci Total Environ. 574:422–436.
Ellis EC, Goldewijk KK, Siebert S, Lightman D, Ramankutty N. 2010. Anthropogenic transformation of the biomes, 1700 to 2000. Glob Ecol Biogeogr. 19:589–606.
Etter A, McAlpine C, Possingham H. 2008. Historical patterns and drivers of landscape change in Colombia since 1500: a regionalized spatial approach. Ann Assoc Am Geogr. 98:2–23.
Fearnside PM. 1989. A prescription for slowing deforestation in Amazonia. Environment. 31:16–20, 39–40.
Fedrigotti C, Aschonitis V, Fano EA. 2016. Effects of forest expansion and land abandonment on ecosystem services of alpine environments: case study in ledro valley (Italy) for the period 1859-2011. Global Nest J. 18:875–884.
Fleischbein K, Wilcke W, Valarezo C, Zech W, Knoblich K. 2006. Water budgets of three small catchments under montane forest in Ecuador: experimental and modelling approach. Hydrol Process. 20:2491–2507.
Fu B, Zhang L, Xu Z, Zhao Y, Wei Y, Skinner D. 2015. Ecosystem services in changing land use. J Soils Sediments. 15:833–843.
Gaglio M, Aschonitis VG, Gissi E, Castaldelli G, Fano EA. 2017. Land use change effects on ecosystem services of river deltas and coastal wetlands: case study in Volano–Mesola–Goro in Po river delta (Italy). Wetlands Ecology and Management. 25(1):67–86.
Gissi E, Gaglio MReho M. 2016. Sustainable energy potential from biomass through ecosystem services trade-off analysis: the case of the province of rovigo (northern italy). Ecosyst Serv. 18:1–19.
Grimes A, Loomis S, Jahnige P, Burnham M, Onthank K, Alarcon R, Cuenca WP, Martinez CC, Neill D, Balick M, et al. 1994. Valuing the rain forest: the economic value of nontimber forest products in Ecuador. Ambio. 23:405–410.
Guariguata MR, Ostertag R. 2001. Neotropical secondary forest succession: changes in structural and functional characteristics. For Ecol Manage. 148:185–206.
Guayasamín Guanga MA. 2015. Evaluación ex ante del impacto socio-económico del manejo convencional y mejorado del cultivo de naranjilla (Solanum quitoense) en el Ecuador [Ex ante evaluation of socio-economic impact of conventional and improved management of Naranjilla crops (Solanum quitoense) in Ecuador] [PhD thesis]. Quito: UCE. 78 p. http://www.dspace.uce.edu.ec/handle/25000/4551.
Günter S, Weber M, Erreis R, Aguirre N. 2007. Influence of distance to forest edges on natural regeneration of abandoned pastures: a case study in the tropical mountain rain forest of Southern Ecuador. Eur J For Res. 126:67–75.
Haines-Young R, Potschin M. 2010. The links between biodiversity, ecosystem services and human well-being. In: Raffaelli D, Frid C, editors; Ecosystem Ecology: a new synthesis. BES Ecological Reviews Series. Cambridge: Cambridge University Press; p. 110–139.
Hall JM, van Holt T, Daniels AE, Balthazar V, Lambin EF. 2012. Trade-offs between tree cover, carbon storage and floristic

biodiversity in reforesting landscapes. Landsc Ecol. 27:1135–1147.
Hein L, van Koppen K, de Groot RS, van Ierland EC. 2006. Spatial scales, stakeholders and the valuation of ecosystem services. Ecol Econ. 57:209–228.
Hofstede RGM, Groenendijk JP, Coppus R, Fehse JC, Sevink J. 2002. Impact of pine plantations on soils and vegetation in the ecuadorian high Andes. Mt Res Dev. 22:159–167.
[INEC] Instituto Nacional de Estadística y Censos (Secretaría Nacional de Planificación y Desarrollo (SENPLADES)). 2010. VII Censo de Población y VI de Vivienda [VII census of population and VI census of housing]. Quito: Ministerio de Finanzas (GovECU).
Jahansooz MR, Yunusa IAM, Coventry DR, Palmer AR, Eamus D. 2007. Radiation- and water-use associated with growth and yields of wheat and chickpea in sole and mixed crops. Eur J Agron. 26:275–282.
Jørgensen P, Ulloa Ulloa C, León B, León-Yánez S, Beck S, Nee M, Zarucchi JL, Celis M, Bernal R, Gradstein R. 2011. Regional patterns of vascular plant diversity and endemism. In: Herzog S, Martínez R, Jørgensen P, editors. Climate change and biodiversity in the tropical Andes. São José dos Campos: JAI and SCOPE; p. 192–203.
Jose S. 2009. Agroforestry for ecosystem services and environmental benefits: an overview. Agrofor Syst. 76:1–10.
Keating PL. 2007. Fire ecology and conservation in the high tropical Andes: observations from Northern Ecuador. J Lat Am Geogr. 6:43–62.
Kitayama K, Aiba S. 2002. Ecosystem structure and productivity of tropical rain forests along altitudinal gradients with contrasting soil phosphorus pools on Mount Kinabalu, Borneo. J Ecol. 90:37–51.
Kocian M, Batker D, Harrison-Cox J. 2011. An ecological study of Ecuador's intag region: the environmental impacts and potential rewards of mining. Tacoma (WA): Earth Economics; p.108.
Koschke L, Fürst C, Frank S, Makeschin F. 2012. A multicriteria approach for an integrated land-cover-based assessment of ecosystem services provision to support landscape planning. Ecol Indic. 21:54–66.
Kottke I, Beck A, Haug I, Setaro S, Jeske V, Suárez JP, Pazmiño L, Preußing M, Nebel M, Oberwinkler F. 2008. Flora and fungi: composition and function. In: Beck E, Bendix J, Kottke I, Makeschin F, Mosandl R, editors. Gradients in a tropical mountain ecosystem of Ecuador. Heidelberg: Springer Berlin; p. 137–148.
Kovacs E, Kelemen E, Kaloczkai I, Margoczi K, Pataki G, Gebert J, Malovics G, Balazs B, Roboz A, Kovacs KE, et al. 2015. Understanding the links between ecosystem service trade-offs and conflicts in protected areas. Ecosyst Serv. 12:117–127.
Kremen C. 2005. Managing ecosystem services: what do we need to know about their ecology? Ecol Lett. 8:468–479.
Kreuter UP, Harris HG, Matlock MD, Lacey RE. 2001. Change in ecosystem service values in the San Antonio area, Texas. Ecol Econ. 39:333–346.
Lal R. 2004. Soil carbon sequestration to mitigate climate change. Geoderma. 123:1–22.
Lambin EF, Geist HJ, Lepers E. 2003. Dynamics of land-use and land-cover change in tropical regions. Annu Rev Environ Resour. 28:205–241.
Leuschner C, Zach A, Moser G, Homeier J, Graefe S, Hertel D, Wittich B, Soethe N, Iost S, Rödersteinm M, et al. 2013. The carbon balance of tropical mountain forests along an altitudinal transect. In: Bendix J, Beck E, Bräuning A, Makeschin F, Mosandl R, Scheu S, Wilcke W, editors. Ecosystem services, biodiversity and environmental change in a tropical mountain ecosystem of South Ecuador. Berlin-Heidelberg: Springer; p. 117–139.
López-Ulloa M, Veldkamp E, de Koning GHJ. 2005. Soil carbon stabilization in converted tropical pastures and forests depends on soil type. Soil Sci Soc Am J. 69:1110–1117.
Lozada T, de Koning GHJ, Marché R, Klein AM, Tscharntke T. 2007. Tree recovery and seed dispersal by birds: comparing forest, agroforestry and abandoned agroforestry in coastal Ecuador. Perspect Plant Ecol Evol Syst. 8:131–140.
Luoma J. 2004. Silvopastoral practices and the Condor Bioreserve, Ecuador. J Sustain For. 18:277–296.
[MA] Millennium Ecosystem Assessment. 2005. Ecosystems and human well-being: synthesis. Washington, DC: Island Press; p.137.
[MAE] Ministerio del Ambiente de Ecuador. 2012. Línea Base de Deforestación del Ecuador Continental [Baseline deforestation of continetal Ecuador]. Quito: Ministerio del Ambiente de Ecuador.
[MAE] Ministerio del Ambiente de Ecuador. 2014. Evaluación nacional forestal: resultados. Nivel de Referencia de Emisiones Forestales por Deforestación (NREFD) [National forest assessment: results. Forest Reference Level Emissions from Deforestation (NREFD)]. Quito: Ministerio del Ambiente de Ecuador.
[MAE] Ministerio del Ambiente de Ecuador, [MAGAP] Ministerio de Agricultura, Ganadería, Acuacultura y Pesca de Ecuador. 2015. Protocolo metodologico para la elaboracion del Mapa de cobertura y uso de la tierra del Ecuador continental 2013-2014, escala 1:100.000 [Methodological protocol for the preparation of land use and land cover maps of continental Ecuador 2013-2014, scale 1:100.000]. Quito: Ministerio del Ambiente de Ecuador.
Maraun M, Illig J, Sandmann D, Krashevska V, Norton RA, Scheu S. 2008. Fauna: composition and function. In: Beck E, Benedix J, Kottke I, Makeschin F, Mosandl R, editors. Gradients in a tropical mountain ecosystem of Ecuador. Berlin Heidelberg: Springer; p. 181–192.
McGroddy ME, Lerner AM, Burbano DV, Schneider LC, Rudel TK. 2015. Carbon stocks in silvopastoral systems: a study from four communities in southeastern Ecuador. Biotropica. 47:407–415.
Metzger MJ, Rounsevell MDA, Acosta-Michlik L, Leemans R, Schröter D. 2006. The vulnerability of ecosystem services to land use change. Agric Ecosyst Environ. 114:69–85.
Molina A, Govers G, Poesen J, Van Hemelryck H, De Bievre B, Vanacker V. 2008. Environmental factors controlling spatial variation in sediment yield in a central Andean mountain area. Geomorphology. 98:176–186.
Mosandl R, Günter S, Stimm B, Weber M. 2008. Ecuador suffers the highest deforestation rate in South America. In: Beck E, Benedix J, Kottke I, Makeschin F, Mosandl R, editors. Gradients in a tropical mountain ecosystem of Ecuador. Heidelberg: Springer Berlin; p. 37–40.
Moser G, Leuschner C, Hertel D, Graefe S, Soethe N, Iost S. 2011. Elevation effects on the carbon budget of tropical mountain forests (S Ecuador): the role of the belowground compartment. Glob Chang Biol. 17:2211–2226.
Muriel P. 2008. La diversidad de ecosistemas en el Ecuador [The ecosystem diversty of Ecuador]. In: de la Torre L, Navarrete H, Muriel P, Macía M, Balslev H, editors. Enciclopedia de las Plantas Útiles del Ecuador [Encyclopedia of the useful plants]. Quito: Herbario QCA & Herbario AAU; p. 28–38.
Nanni AS, Grau HR. 2014. Agricultural adjustment, population dynamics and forests redistribution in a

subtropical watershed of NW Argentina. Reg Environ Change. 14(4):1641–1649.

Ochoa J, Ellis M. 2005. Seed transmission of Fusarium oxysporum f. sp. quitoense appears an important cause of naranjilla collapse in Ecuador. In: Jarvis D, Sevilla-Panizo R, Chávez-Servia JL, Hodgkin T, editors. Seed systems and crop genetic diversity on-farm. Proceedings of IPGRI Workshop; Sep 16–20; Pucallpa. Rome: IPGRI. p. 122–125..

Ochoa-Cueva P, Fries A, Montesinos P, Rodríguez-Díaz JA, Boll J. 2015. Spatial estimation of soil erosion risk by land-cover change in the Andes of Southern Ecuador. L Degrad Dev. 26:565–573.

Peel MC, Finlayson BL, McMahon TA. 2007. Updated world map of the Koppen-Geiger climate classification. Hydrol Earth Syst Sci. 11:1633–1644.

Perrin R. 1977. Pest management in multiple cropping systems. AgroEcosystems. 3:93–118.

Porter J, Costanza R, Sandhu H, Sigsgaard L, Wratten S. 2009. The value of producing food, energy, and ecosystem services within an agro-ecosystem. Ambio. 38:186–193.

Potthast K, Hamer U, Makeschin F. 2010. Impact of litter quality on mineralization processes in managed and abandoned pasture soils in Southern Ecuador. Soil Biol Biochem. 42:56–64.

Puyravaud J-P. 2003. Standardizing the calculation of the annual rate of deforestation. For Ecol Manage. 177:593–596.

Redo DJ, Grau HR, Aide TM, Clark ML. 2012. Asymmetric forest transition driven by the interaction of socioeconomic development and environmental heterogeneity in Central America. PNAS. 109(23):8839–8844.

Reyes-Puig MM, Reyes-Puig JP, Yánez-Muñoz MH. 2013. Ranas terrestres del genero Pristimantis (Anura: Craugastoridae) de la Reserva Ecologica Zunag, Tungurahua, Ecuador: Lista anotada y descripcion de una especie nueva [Terrestrial frogs of Genus Pristimantis (Anura: Craugastoridae) Ecological Reserve Zunag, Tungurahua, Ecuador: annotated list and description of a new species]. Av en Ciencias e Ing. 5:2.

Rodríguez Eraso N, Armenteras-Pascual D, Alumbreros JR. 2013. Land use and land cover change in the Colombian Andes: dynamics and future scenarios. J Land Use Sci. 8:154–174.

Rodríguez-Ortega T, Oteros-Rozas E, Ripoll-Bosch R, Tichit M, Martín-López B, Bernués A. 2014. Applying the ecosystem services framework to pasture-based livestock farming systems in Europe. Animal. 8:1361–1372.

Sanchez-Cuervo AM, Aide TM. 2013. Identifying hotspots of deforestation and reforestation in Colombia (2001–2010): implications for protected areas. Ecosphere. 4(11):1–21.

Sapoukhina N, Tyutyunov Y, Sache I, Arditi R. 2010. Spatially mixed crops to control the stratified dispersal of airborne fungal diseases. Ecol Modell. 221:2793–2800.

Simon S, Bouvier JC, Debras JF, Sauphanor B. 2009. Biodiversity and pest management in orchard systems. Sustain Agric. 2:693–709.

Titira D. 1999. Mamíferos del Ecuador [Mammals of Ecuador]. Vol. 2. Museo de Zoología, Centro de Biodiversidad y Ambiente, Pontificia Universidad Católica del Ecuador.Quito : EcoCiencia/SIMBIOE

Unger M, Homeier J, Leuschner C. 2013. Relationships among leaf area index, below-canopy light availability and tree diversity along a transect from tropical lowland to montane forests in NE Ecuador. Trop Ecol. 54:33–45.

Valencia R. 1995. Composition and structure of an Andean forest fragment in eastern Ecuador. In: Churchill SP, Balslev H, Forero E, Luteyn JL, editors. Biodiversity and conservation of neotropical montane forests. Proc. symposium. New York: Botanical Garden; p. 239–249.

Valencia R, Cerón C, Palacios W, Sierra R. 1999. Las formaciones naturales de la Amazonía del Ecuador [The natural formation of Ecuadorian Amazon]. In: Sierra R, editor. Propuesta Prelimar de un Sistema de Clasificacion de Vegetation para el Ecuador Continental [Preliminary proposal of a Vegetation Classification System for Continental Ecuador]. Quito (Ecuador): Proyecto INEFAN/GEF-BIRF y EcoCiencia; p. 111–119.

Vargas H, Neill DA, Asanza M, Freire-Fierro A, Narvaez E. 2000. Vegetación y flora del Parque Nacional Llanganates [Vegetation and flora of Llaganantes National Park]. In: Vazquez MA, Larrea M, Suarez L, editors. Biodiversidad en el Parque Nacional Llanganates [Biodiversity in Llaganantes National Park]. Quito (Ecuador): EcoCiencia; p. 13–66.

Viteri X, editor. 2002. Corredor ecológico entre los Parques Nacionales Llanganates y Sangay: un informe de los estudios biológicos y sociales. [Ecological corredor between Llaganantes and Sangay National Parks: a report on biological and social studies]. Quito (Ecuador): Fundación Natura y Fondo Mundial para la Naturaleza.

Wang W, Guo H, Chuai X, Dai C, Lai L, Zhang M. 2014. The impact of land use change on the temporospatial variations of ecosystems services value in China and an optimized land use solution. Environ Sci Policy. 44:62–72.

Wassenaar T, Gerber P, Verburg PH, Rosales M, Ibrahim M, Steinfeld H. 2007. Projecting land use changes in the Neotropics: the geography of pasture expansion into forest. Glob Environ Chang. 17:86–104.

White S, Maldonado F. 1991. The use and conservation of natural resources in the Andes of Southern Ecuador. Mt Res Dev. 11:37–55.

Wilcke W, Yasin S, Abramowski U, Valarezo C, Zech W. 2002. Nutrient storage and turnover in organic layers under tropical montane rain forest in Ecuador. Eur J Soil Sci. 53:15–27.

Willemen L, Drakou EG, Dunbar MB, Mayaux P, Egoh BN. 2013. Safeguarding ecosystem services and livelihoods: understanding the impact of conservation strategies on benefit flows to society. Ecosyst Serv. 4:95–103.

WWF. 2014. Report of chemical soil properties for 13 selected sites inside CELS. Survey financed by WWF, Raw data report. p. 13.

Yaguache R. 2014. Plan de restauración a nivel de Paisaje del Corredor Ecológico Llanganates Sangay [Restoration landscape plan of ecological corredor llaganantes-Sangay]. Quito: WWF.

Yánez-Muñoz MH, Cisneros-Heredia DF, Reyes-Puig JP. 2010. Una nueva especie de rana terrestre Pristimantis (Anura: Terrarana: Strabomantidae) de la cuenca alta del Rio Pastaza, Ecuador [A new species of terrestrial frog Pristimantis (Anura: Terrarana: Strabomantidae) of upper basin of Pastaza river, Ecuador]. Av en Ciencias e Ing. 2:B28-B32.

Yánez-Muñoz MH, Reyes-Puig JP, Morales Mite MA. 2013. Herpetofauna en areas prioritarias para la conservación: El sistema de Reservas Jocotoco y Ecominga [Herpetofauna in protected areas for conservation: the system of Reserves Jocotoco and Ecominga]. Quito: Museo Ecuatoriano de Ciencias Naturales-Jocotoco-Ecominga; p. 392.