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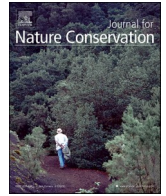
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Vulnerability assessment of natural heritage in the North-Eastern Ecuadorian Amazon using land use cover and nature protection status

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ABSTRACT

This study evaluates the vulnerability of natural heritage in the Northeast Ecuadorian Amazon (NEA), from the perspective of intrinsic vulnerability. Therefore, two proxy indicators of vulnerability were established: (i) the ecological integrity vulnerability index (EIVI), indicating the potential loss of ecological integrity, assessed by protection status (2018); (ii) the biological diversity vulnerability index (BVI), deriving the potential loss of biological diversity, using land cover/land use (2014). Biodiversity values were derived with land cover-related biodiversity data, and (iii) and compared with spatial congruence/correlation between (i) and (ii). The EIVI was directly derived using protection status, which defines vulnerability levels by ranking them according to corresponding levels of anthropic development allowance. A robust relationship between land use classes and species richness was found for both endemic vascular plants in neotropical regions and multi-taxa in tropical forests worldwide, validating the BVI. Furthermore, the moderate spatial correlation between protection status and land cover ($r = 0.52$) may be partially explained by 3,060 km² of unprotected primary forest. Overall, 42 % of the NEA have a high degree of congruence between potential biodiversity and protection status. In conclusion, this study provides insights into nature conservation efforts in areas with sparse biodiversity data using readily available geographic information.

1. Introduction

Vulnerability is the state of susceptibility to harm from exposure to stresses associated with environmental and social change and from the lack of adaptive capacity (Adger, 2006). Ecological vulnerability is further defined as the potential of an ecosystem to regulate its response to a stressor (De Lange et al., 2009; Turner et al., 2003). This potential is determined by ecosystem assets observed at different hierarchical levels, i.e., organisms, populations, communities, and habitats (De Lange et al., 2010). These assets are valuable services or objects that a specific society wants to protect because of the perceived benefits it derives from them and that could be lost (Gleyze, 2002). Vulnerability has two key parameters, the sensitivity to be harmed by a stressor, and the adaptive capacity of the system (Adger, 2006).

Vulnerability in the Northeastern Amazon (hereafter NEA) is driven by multiple stressors: demographic (e.g., population growth and

density), socio-economic (road building, oil industry, agriculture, mining) (Durango-Cordero et al., 2022; Mena, 2008; Province GAD Orellana, 2011), and adaptive capacity may be reduced by shifting or lack of adequate policies regarding land use pressures, resulting in land tenure conflicts over protected areas (Rorato et al., 2022), boundary changes, or private or communal appropriations of untitled land (Holland et al., 2014; Mena et al., 2006a; Sovacool & Scarpaci 2016; Vallejo et al., 2015). Similarly, long-term nature conservation is threatened by stressors such as potential non-renewable extractive projects, such as mining and oil extraction (Bonilla-Bedoya et al., 2014; Durango-Cordero et al., 2018, 2022) and growing concerns about solid waste disposal (Solíz et al., 2020). Private and public land tenure reflects the quality of political and institutional capacity and governance and is a vulnerability stressor of land use dynamics that reduces the adaptive capacity of the system (Holland et al., 2014).

In this study, the NEA is used as the most appropriate test-bed for the

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geographical application of an ecological vulnerability assessment, because of its renowned status as a biodiversity hotspot, (Bass et al., 2010; Myers et al., 2000). To illustrate, at least 210 mammalian, 131 amphibious, 558 bird, and 2,700 vascular plant species were reported (Bass et al., 2010; Finer et al., 2008), including 516 endemic plant species (León-Yáñez et al., 2017). The NEA has been extensively monitored by Ecuadorian (Ecociencia, 2016; INABIO, 2022; PUCE, 2018) or international public services (IUCN, 2016; RAMSAR, 2019), which have contributed to its protection and management through legal and other effective means to achieve its long-term conservation, and the maintenance of associated ecosystem services and cultural values (Dudley, 2008). However, this natural heritage is under threat from a wide range of anthropogenic activities, highlighting the political and institutional difficulty of achieving a territorial development that would take into account the conservation of these ecosystems, both in terms of governance and knowledge capacity building, as identified by the National Research Council (US) Panel on Biodiversity Research Priorities (NRC, 1992).

Given the extensive biodiversity characteristic of NEAs, establishing definitive distinctions and prioritization schemes proves challenging when reliant solely upon incomplete and spatially disparate biodiversity inventories (Lessmann et al., 2016; PUCE, 2018). To accurately evaluate potential biodiversity loss, a comprehensive and homogeneous assessment of the territory is imperative (Mena et al., 2006b). Subsequently, each spatial unit, whether pixel or grid cell, can be assigned a quantitative value to gauge potential loss and relative vulnerability. This valuation should be derived from a composite analysis of conservation efforts within each spatial unit (Marignani et al., 2017).

Land tenure and the protection status have been considered as proxies for assessing biodiversity vulnerability in space and time (Olson et al., 2001). The protected areas are typically associated with higher natural heritage values than unprotected areas (Gray et al., 2016; Shi et al., 2019). However, due to the conflict between socio-economic development and nature conservation (Mena et al., 2006a; Mena et al., 2006b; Sovacool & Scarpaci, 2016), the legal protection of these areas does not guarantee their long-term integrity (Geldmann et al., 2019; Kleemann, 2022; Mestanza-Ramón et al., 2023). The historical alteration of the boundaries of the Cuyabeno Biological Reserve, resulting in increased deforestation, is evidence of this structural gap (Mena et al. 2006a; Messina et al., 2006); the Yasuní-ITT initiative, proposed to keep oil underground but succumbing to national dependence on oil exports, global energy policies, and real limitations on enforcing climate change policy targets (Sovacool & Scarpaci, 2016). These threats are driving land cover change (Mena, 2008; Messina & Walsh, 2005; Van Der Hoek, 2017) and may increase the probability of a hazardous event (Adger, 2006; Turner et al., 2003). Likewise, related development infrastructure and activities are sources of pollutant discharges to the environment, of which oil extraction is the most massive and destructive (Durango-Cordero et al., 2018; Durango-Cordero et al., 2019; MAATE-PRAS, 2016).

Biodiversity value has previously been estimated using land use cover/ land change (LUCC) as a proxy (Bernard & Fenton, 2002; Beukema & Van Noordwijk, 2004; Chaudhary et al., 2015; Haro-Carrión et al., 2009; Harvey et al., 2006, 2016; Lawton et al., 1998; Martínez et al., 2009). In particular, these studies indicated that species diversity generally declined with increasing anthropogenic alteration (Lawton et al., 1998; Martínez et al., 2009). Planning for conservation of biodiversity and other natural values is essential and inherently spatial (Pressey et al., 2007) and may increase the adaptive capacity of a system (Rorato et al., 2022).

The present research aims to assess the vulnerability of natural heritage from the perspective of intrinsic vulnerability, which denotes that a system is vulnerable regardless of the hazard, and as such can be evaluated independently (Mimi & Assi, 2009). In this study, we define two proxy indicators: (i) the vulnerability to ecological integrity loss, which is reflected in the initial decision to protect a natural site, and

promoted by long-term protection measures; (ii) the vulnerability to biodiversity loss, which is derived from potential biodiversity values assigned to land use classes. We also evaluate the spatial congruence/correlation between putative ecological integrity and biodiversity. The main methodologies are ranking, scoring, and spatial indexing for vulnerability assessment, supported by overlay and correlation analysis, to evaluate spatial congruence between ecological integrity and biodiversity at a regional scale (in this study spatial scale ranges from 1:50,000 to 1:250,000) (Egoh et al., 2008, 2011). The provision of such vulnerability estimates enables the formulation of spatial planning elements that facilitate the prioritisation of areas for nature conservation..

2. Materials and methods

2.1. Study area

The study area (Fig. 1A) was restricted to the provinces of Sucumbios and Orellana in the NEA (~144–900 m.a.s.L., Amazonian lowlands), covering an area of 35,051 km² (Fig. 1B). The study area is characterized by a warm climate with an annual temperature range of 20 °C to 30 °C, and an average rainfall of 2,900 mm.yr⁻¹ (Institute of Meteorology and Hydrology-INHAM). The area includes rivers with high flow rates (i.e., Napo, Tiputini, Coca, Payamino, Putumayo, Cuyabeno, and Aguatico). The hydrological regime is irregular (1,000 to 5,000 m³.s⁻¹ average daily discharges), and characterized by flash floods, due to high sensitivity to rainfall events (Laraque et al., 2007).

2.2. Data compilation

A land use/land cover map was produced for Ecuador in 2014, based on combined multispectral satellite imagery: medium spatial resolution Landsat 8 and high spatial resolution RapidEye, and ground-truth verification (MAGAP-SIGTIERRAS, 2015). Different classifications were defined by the Ecuadorian governmental institutions (the Ministry of Environment and Water (MAATE), the Ministry of Agriculture, the Livestock and Fisheries (MAGAP), and the Ecuadorian Spatial Institute (IEE), in addition to the operational definitions provided by the Intergovernmental Panel for Climate Change (IPCC) (IPCC Software, 2013; MAATE & MAGAP, 2015). The spatial information used in this study represents a regional scale between 1:50,000 and 1:250,000, with a 90x 90 m cell size as implemented in the NEA for an agriculture and forestry patches assessment by (Jaderne Houssou et al., 2019). Table 1 summarizes the spatial data compiled for this study.

2.3. Vulnerability assessment of ecological integrity

The network of protected areas established within the NEA served as a parameter for evaluating the degree of ecological integrity, provision of ecosystem services and ecological functions intended to be protected in a given area (Ferraro et al., 2013), hereafter named the Ecological Integrity Vulnerability Indicator (EIVI). This approach is based on two assumptions: (1) the level of protection granted to a given site is a relevant proxy of its ecological value / ecological integrity (both because the initial decision to protect the site was based on its ecological value, and because legal protection positively influences ecological integrity through time); (2) sites with a high ecological value are prone to greater natural heritage loss in the event of anthropogenic alteration (intentional or not, legal or not), hence they are defined here as more vulnerable.

Local land use management plans define 'allowed' levels for some anthropogenic interventions in protected areas according to their protection status (PS). The PS categories are thus a proxy for local levels for the EIVI; the PS categories were defined following the United Nations Educational, Scientific and Cultural Organization (UNESCO, 2005), RAMSAR Convention (1975), and the local management plans of decentralized autonomous governments (GADs) (Table S1). The PS is set

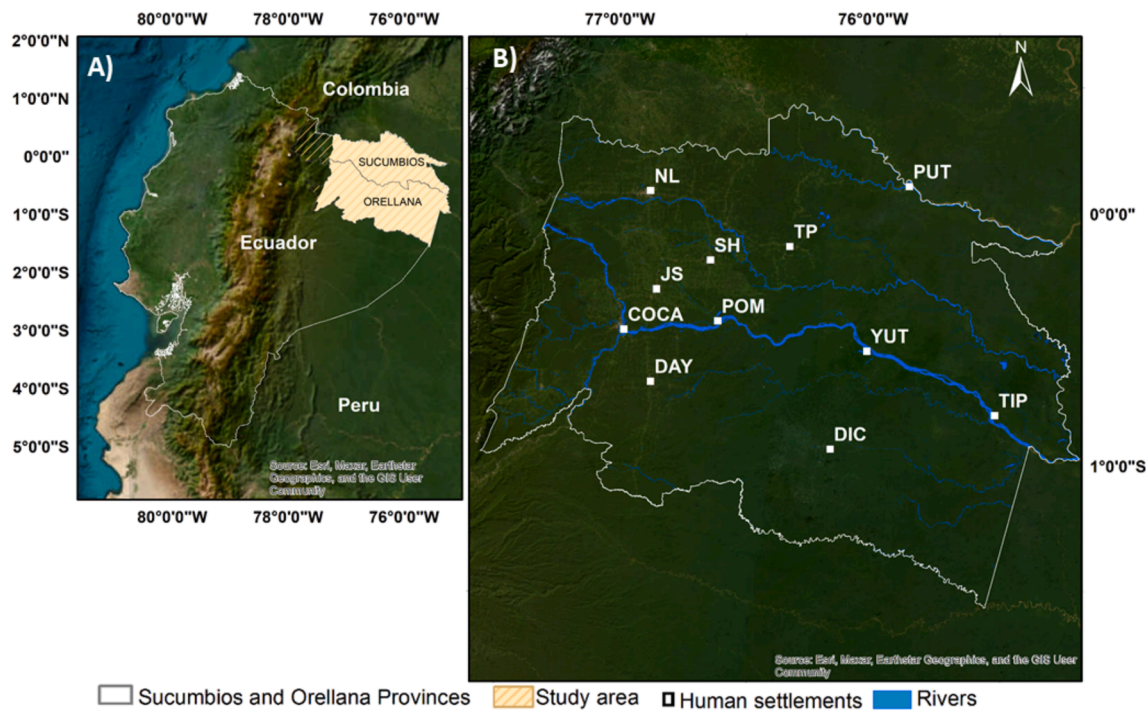


Fig. 1. Location of the study area. (A) The North-eastern Ecuadorian Amazon (NEA) in Sucumbíos and Orellana provinces of Ecuador. (B) The NEA, including major rivers and a land cover mosaic as background (GeoEye, Digital Globe, 2018). Represented human settlements are abbreviated: COCA=El Coca; DAY=Dayuma; DIC=Dícaro; JS=Joya de Los Sachas; NL=Nueva Loja (aka Lago Agrio); POM=Pompeya; PUT=Putumayo; SH=Shushufindi; TP=Tarapoa; TIP=Tiputini; YUT=Yuturi.

Table 1
GIS database used in this study.

GIS data	Scale	Source
Land use	1: 100,000	MAATE & MAGAP (2015)
National System of Protected Areas	1: 250,000	MAATE & MAGAP (2015)
National System of Protected Areas	1: 50,000	MAATE & MAGAP (2015)
Forest Patrimony	1: 100,000	MAATE (2019)
Intangible Zone*	1: 250,000	MAATE (2012)MAATE (2012)
Reserve of Biosphere	1: 250,000	

* It is a protected area for use by non-contacted Indigenous populations that could be exceptionally exploitable if natural resources are considered of public interest (Finer et al., 2008; Vallejo et al., 2015).

to range from 1 (low) to 5 (high) (Table 2). A vulnerability score is assigned to each site independently, with the value of the highest PS retained where two or more PS overlap. To illustrate, if pixels were equally associated with a biosphere reserve and a national park, the value for the national park was retained.

2.4. Assessing biodiversity value from land use types

To assess vulnerability to potential biodiversity loss we developed an index, hereafter called the Biodiversity Vulnerability Index (BVI). Indexing for vulnerability assessment requires the identification of relevant parameters (Durango-Cordero et al., 2022). In this case, land use classes (LUs) were used as parameters. They were reclassified into 5 classes (to standardize with the number of classes set for the EIVI). Except for the infrastructure and urban classes, which were regrouped into human settlements, the other classes were kept as originally designated (Table 3).

We estimate the BVI for each LU along a biodiversity value (BDv). A higher BDv indicates greater biodiversity per unit of surface area within a given LU. The method for assigning BDv to LUs is based on vascular plants species richness of (Marignani et al., 2017; Schmidt, 2008).

Derived BDv from neotropical LUs are obtained from a combined meta-analysis and expert surveys previously conducted by Henzen (2008).

The assumed hypothesis is that the BDv of a LU is positively proportional to the risk of biodiversity loss under anthropogenic change. Therefore, higher vulnerability values were attributed to the more biodiverse LUs. Each LU received a biodiversity value from 0 (low) to 5 (high), calculated using the following expression (Henzen, 2008):

$$BDv = (S/SH) \times Ss, \tag{1}$$

where:

BDv = the biodiversity value (0–5) of a sample, determined for each land use category.

Ss = the number of species observed per hectare for a given land use category.

Sh = the highest species number per hectare among the different categories of land use.

To ensure standardisation and comparability of measurements from the two vulnerability indices, e.g., EIVI and BVI, the biodiversity values obtained were transferred into ArcGIS® system and rescaled from a continuous 0.01 to 5.00 scale (=BVI) to a qualitative scale (index scores), from 1 to 5 to match the EIVI map using equal intervals (20 % each class).

2.5. Validity of the relationship between biodiversity and land use

The representativeness of vascular (woody) plant species as a proxy for total species diversity in the neotropics was tested by comparing the BDv obtained from vascular plant species richness in the neotropics (using the meta-analysis and expert opinion described in Section 2.4) with multi-taxa species richness (n = 21, Table S2) in similar land uses in pan-tropical regions, as there were insufficient studies at this level in the neotropics. A total of 101 plots in different LUs were selected from a set of 53 publications (Table S2), and a biodiversity value was calculated for each of them, following the procedure described in Section 2.4. A non-

Table 2
Protection statuses of natural sites within the NEA with their corresponding level of anthropic development allowance and their given vulnerability value.

Protection statuses (PSs)	Allowed activities	Ecological Integrity Vulnerability Indicator
National System of Protected Areas (SNAP), Intangibleareas	None to limited human presence, maintaining of ecological and hydrological functioning processes, integrity, biodiversity and relative uniqueness (IUCN, 2016). Intangible Zone (ZITT) are exceptionally exploitable if natural resources are considered of public interest (Finer et al., 2008; Vallejo et al., 2015).	5
Protected forests	Private, communal or public-owned forests, with recreational, touristic, and scientific research purposes which have been recognized by local decentralized governments (GAD) but not the SNAP.	4
Biosphere reserves, Forest patrimony	<i>Forest patrimony</i> is a restricted land category with limited extractive activities such as farming or timber extraction, it cannot be private property or open sold to the market (Mena, Barbieri, et al., 2006). They serve as buffer areas to larger reserves. <i>Biosphere reserves</i> are sustainable areas where human activities are allowed under sustainable practices. (UNESCO, 2017).	3
Communal Socio-Bosque	<i>The Socio Bosque Program</i> (PSB) was created to provide economic incentives to forested lands for at least 20 years (http://sociobosque.ambiente.gob.ec/node/174).	2
Zones without protecting status	All other land use areas dedicated to extensive agriculture, cattle ranching, urbanization, extractive activities, etc.	1

parametric Kruskal-Wallis test with χ^2 approximation and a Wilcoxon-Mann-Whitney test with post-hoc Bonferroni corrections, were used to compare species' richness between LUs, as high heteroscedasticity precluded the use of analysis of variance (ANOVA) (Zar, 1974). Pairwise comparison was then performed using Spearman's rank correlation (Spearman, 1907) to determine the relationship between the ranks given to each LU according to the diversity of either vascular plants in the neotropical zone or other taxa in the entire pantropical zone. All analyses were conducted with R software (R Development Core Team, 2008).

2.6. Standardisation and combination of vulnerability indices

Two spatial analyses were performed by combining the two standardized vulnerability indices, using an overlay approach:

Spatial addition: considering the EIVI and BVI as independent attributes of the same system, the spatial addition of the biodiversity and ecological integrity indices at each pixel results in a more integrated vulnerability score, namely the overall vulnerability assessment developed for this study. The spatial operation of adding vulnerability levels

Table 3
Definition of land use classes in the NEA.

Land use class	Definition (IPCC Software, 2013; MAATE & MAGAP, 2014)
Primary forest	Plant community characterized by the dominance of trees of different native species, varied sizes and ages, with one or more strata.
Secondary forest	Regenerate native forests, which have been cleared by natural or man-made causes, such as agriculture or ranching. They display a major difference in forest structure and/or species composition concerning primary forests. Secondary vegetation is generally unstable and represents intermediate successional stages.
Scrubland	Areas with a substantial component of native non-arboreal woody species. It includes areas degraded in transition to a dense canopy cover.
Grassland	Native herbaceous vegetation with spontaneous growth, used to sporadic cattle ranging, wildlife, and conservation, which do not need human managing.
Agricultural mosaic	Group of cultivated species that are mixed altogether and cannot be individualized; and exceptionally may be associated with natural vegetation.
Perennial crops	Croplands with a vegetative cycle of more than 3 years that may be harvested several times a year.
Semi-perennial crops	Croplands with a vegetative cycle between 1 and 3 years.
Pasture	Herbaceous vegetation dominated by <i>Poaceae</i> and leguminous introduced species, used with cattle ends, which for their establishment and conservation need human managing.
Annual crops	Croplands with seasonal vegetative cycle that may be harvested several times a year.
Human settlements	Built environment encompassed by urbanized areas, artificially made with high population density and related infrastructure.

per pixel thus yields values that range from 2 to 10. It is possible to add up as many integrate indicators as a researcher needs or as information becomes available.

Spatial difference: the difference between the EIVI and the BVI illustrates whether land use categories with high biodiversity values are (or are not) located in regions deemed to sustain their biodiversity in the long-term (Ferraro et al., 2013). Values can range from -4 to 4. Total spatial congruence between the two metrics yields zero values. Negative values indicate situations where a high protection status is not associated with high biodiversity, while positive values indicate situations where high biodiversity is not well protected. Per unit decrease or increase reflects the intensity of the spatial incongruence.

3. Results

3.1. Vulnerability map obtained from protection status

The Ecological Integrity Vulnerability Index (EIVI) was derived from the protection status layers, which were merged into a single spatial layer (Fig. 2). In a descending order, the vulnerability classes (Fig. 3) represented variable areas: 41 % (14,390 km²) high, 6.6 % (2,312 km²) medium-high, 21.4 % (7,495.8 km²) medium, 0.3 % (96 km²) medium-low and 30.7 % (10,757.2 km²) low. The geographic location and size of the two largest protected areas, namely the Yasuni National Park and the Cuyabeno Wildlife Reserve, are of particular relevance in this context. The highest concentration of human population is observed in the western and central zones, where no protected area has been designated, with the exception of the Sumaco Biosphere Reserve. Table S3 illustrates the surface area of the various protection statuses, with the corresponding relative percentages in the NEA (35,051 km²).

3.2. Vulnerability map obtained from biodiversity

Neotropical vascular plants were used as a surrogate for the

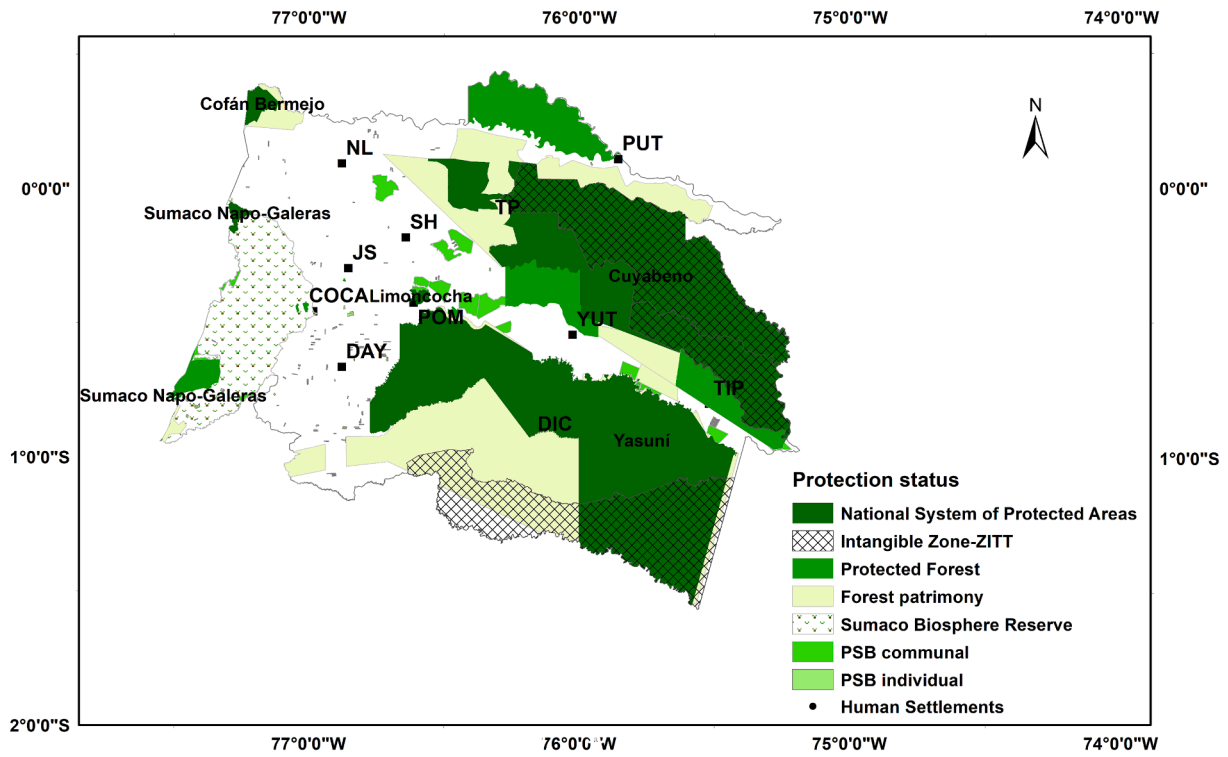


Fig. 2. Distribution of protected areas in the NEA. White coloured areas indicate no protection status.

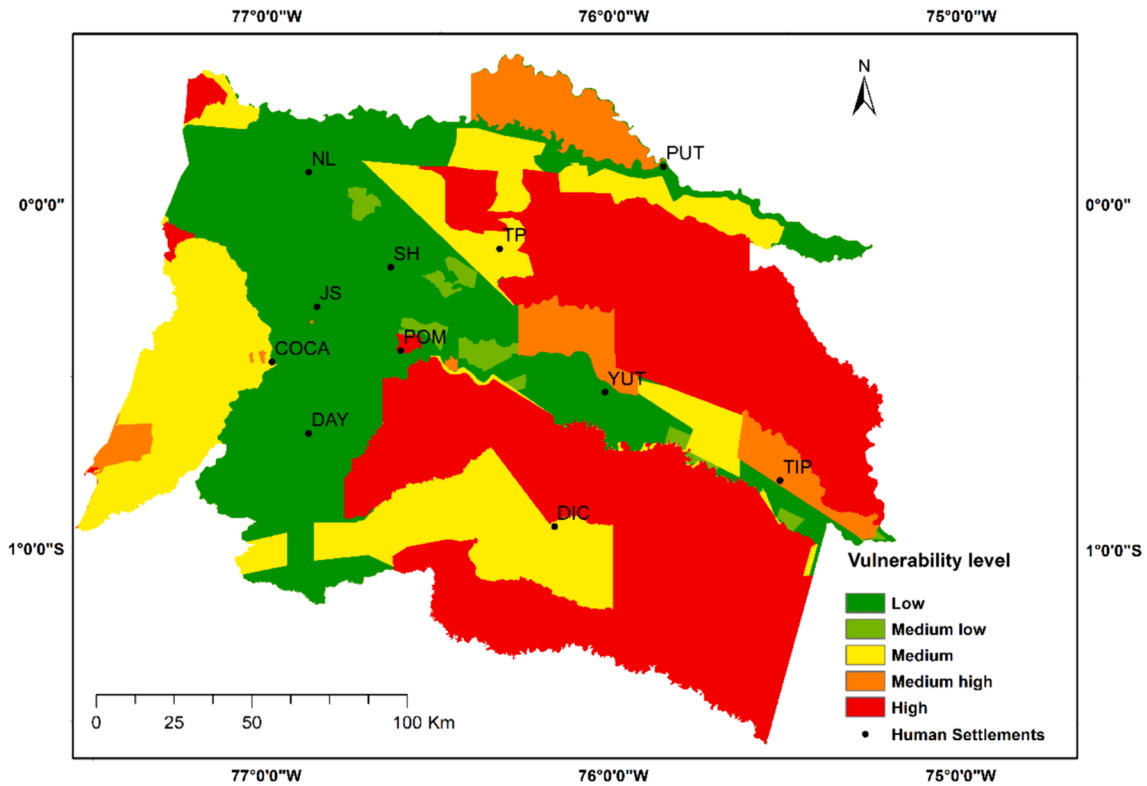


Fig. 3. Ecological Integrity Vulnerability Index (EIVI).

Biodiversity Vulnerability Index (BVI). Pairwise Kruskal–Wallis comparisons indicate that this surrogate may be a potentially useful tool for providing values based on LUs given the differences observed between the groups ($\chi^2 = 31.3$, $df = 3$, $P < 0.0001$). Primary forest is significantly different from other land uses ($P = 0.0008$) according to the Wilcoxon–

Mann–Whitney test. Score ranks for remaining land use categories were positively correlated with neotropical and pantropical vascular plant data. However, despite the same hierarchy across the different LUs, this correlation was only marginally significant ($P = 0.0833$) due to the reduced number of categories ($\chi^2 = 1.00$, $n = 4$). Table 4 compares

Table 4
Biodiversity Vulnerability Index calculated from species richness in taxa in other tropical regions and compared to this study.

Biodiversity Value	Primary forests	Secondary and logged forests	Perennial cultures	Pastures
Vascular plants in neotropics	5	3.2	2.26	1.3
Multiple-taxa average (n = number of studies)	4.73* (n = 40)	4.14 (n = 27)	3.32 (n = 26)	2.3 (n = 8)

* Significance at 0.01%.

mean BDv derived from multi-taxa species richness and neotropical vascular plant richness in the different LUs. Fig. 4 illustrates the area extent of each land-use. Table 5 presents biodiversity values (BDv) per neo-tropical LU category. Fig. 5 indicates the BVI in the study area.

3.3. Relationships between vulnerability indices

There is a moderate positive correlation ($r = 0.52$) between the EIVI and BVI maps (Fig. 6). An area of 3,217 km² of unprotected primary forest reduces this correlation, as indicated by the red patches of value 4 in Fig. 6. Regardless of the metric used, the largest yellow areas correspond to areas with equivalent levels of vulnerability. Overall, 42 % of the NEA have a high degree of congruence between potential biodiversity and the state of protection (Table 6). The land use analysis indicates that in the south-western zone of the Sumaco Biosphere Reserve (where only sustainable socio-economic activities are allowed), and in the eastern zones, of the Yasuní and Cuyabeno Reserves (where human activity is restricted), less than 1 % of annual crops and pastures with the lowest biodiversity are assigned the highest level of protection (value – 4 in blue in Fig. 6).

The natural heritage results from the addition of EIVI and BIV spatial indices. The areas where the ZITT, the Yasuní National Park and the Cuyabeno Reserve overlap with primary forest exhibit the highest overall ecological integrity values. The lowest values are observed in areas where human settlements and annual crops are present, and which are not protected. The medium vulnerability category encompasses a range of BDv and SP combinations, including primary forest areas that overlap with unprotected zones and secondary forest regions that intersect with socio-bosque community territories. These diverse configurations are illustrated in Fig. 7.

4. Discussion

4.1. Relevance of protection status used in vulnerability assessment

This vulnerability scoring system allows natural heritage vulnerability to be mapped using two spatial indices of biodiversity and ecological integrity at a homogenised regional scale. Previous studies indicate that an acceptable level of long-term conservation of natural heritage has been achieved when protection status and land use change are compared using remote sensing techniques (Bonilla-Bedoya et al., 2014; Mena et al. 2006a; Messina et al., 2006). The multi-taxa analysis performed in this study further supports these findings. In fact, vascular plant species can be detected by satellite data for the corresponding land-use related biodiversity classes (Salovaara et al., 2005). Moreover, the assessment of the potential loss of natural heritage under plausible future anthropogenic impacts on conservation areas is a relevant assessment of vulnerability (Adger, 2006), since the driving causes of environmental degradation can persist despite protective measures at a given time (Holland et al. 2014; Mena et al., 2006b; Mjachina, 2014). For instance, Geldmann et al. (2019) implemented an index of temporal human pressure for over 15 years showcasing that population density

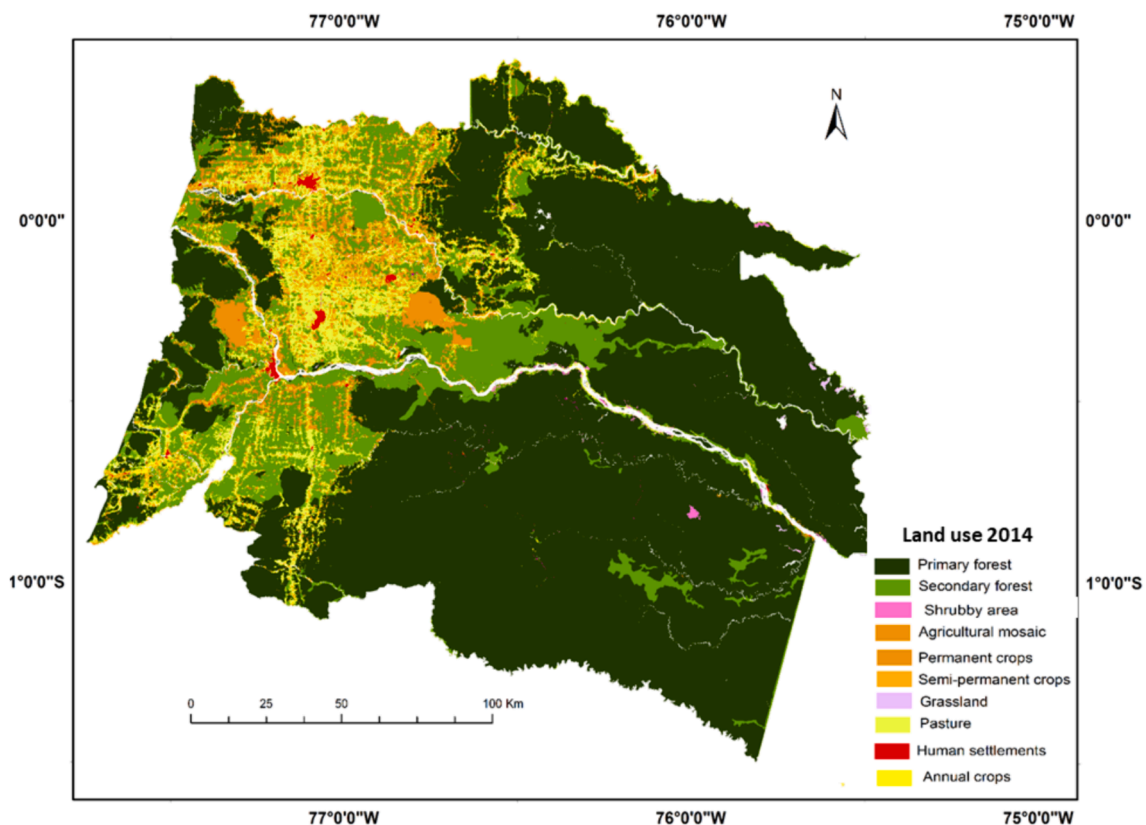


Fig. 4. Classification of land use classes according to MAGAP-MAATE –IEE (2014). Colour hues selected from combination of the CORINE and the Inventaire National Forestier (INF) land cover systems (IGN, 2018) to ease distinction between shrubby lands, grasslands and annual crops.

Table 5

Land use classes with their relative surface in the NEA and biodiversity values, ranked along their vulnerability levels.

	Biodiversity value (Calculated)	Relative surface (%)	Vulnerability level	Biodiversity Vulnerability Index (BVI)
Primary forest	5.00	70.07	High	5
Secondary forest	3.20	12.40	Medium High	4
Shrubby areas	2.53	0.12	Medium	3
Agricultural mosaic	2.40	3.02	Medium	3
Permanent crops	2.25	3.40	Medium	3
Semi-permanent crops	2.25	0.05	Medium	3
Grassland	2.08	0.08	Medium	3
Pasture	1.30	8.16	Medium low	2
Human settlements	1.13	0.43	Medium low	2
Annual crops	0.89	0.25	Low	1

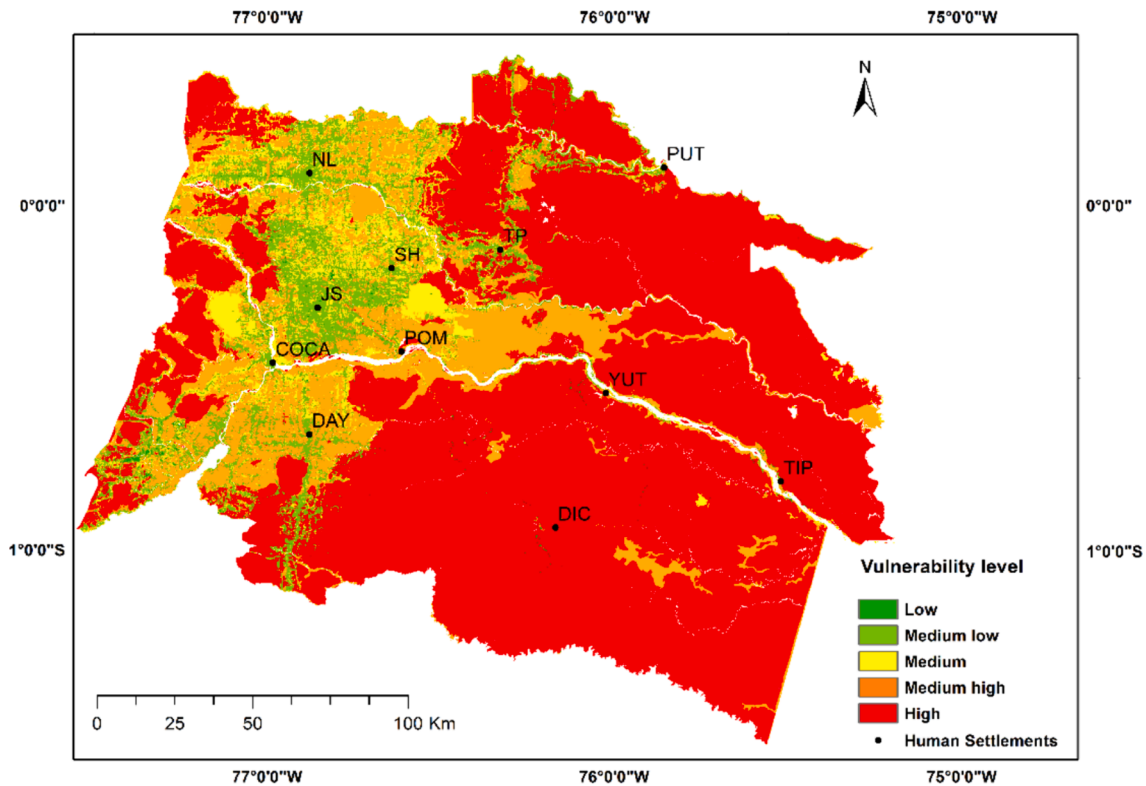


Fig. 5. Biodiversity Vulnerability Index (BVI) according to biodiversity values derived from land use in 2014. The surface extent of vulnerability classes from the highest to the lowest are: 70.1 % high (14 390 km²), 12.4 % medium high, 6.7 % medium, 8.2 % medium low and 0.3 % lowest.

and night lights within protected areas in the neotropics increased regardless of their protection status. Conservationist concerns rise because of increasing possibility of failure for conservation in these protected areas. Some studies refer to them as “paper parks” to describe protected areas prone to situations on the legal status that undermine the actual conservation of nature within their designated jurisdiction (Bonham et al., 2008; Di Minin & Toivonen, 2015). Meanwhile, a possible perspective for this work would be to define the status according to the historical depth of the protection status (the longer a high protection was given to an area, the higher the protection score).

4.2. Biodiversity and land use

The usefulness of the BVI depends on the ability to rank the land use classes according to their biodiversity value, which in turn depends on the ability to measure the actual biodiversity. First, previous studies have supported the use of tree species as a proxy for species richness (Estrada & Meritt 1993; Kessler et al., 2016). This was also consistent with a multi-taxa study. Species richness decreased proportionally with the degree of habitat modification. Schulze et al. (2017) supported by

the validity procedure (multi-taxa analysis), at least at the regional scale. These studies have shown species richness to be of higher conservation value in primary forest than in secondary forests or agricultural mosaics; at least for geographically restricted species (Dumbrell & Hill, 2005) and specialist species that can only thrive in narrow and restricted ecological niches or exhibit a limited diet (Faria et al., 2006). The marginally significant designation of vulnerability values to medium vulnerability classes can be explained because biodiversity richness is associated with trees in secondary forest or agricultural mosaics. Species might use these zones as transient corridors (Eggleton et al., 2016; Gardner et al., 2007; Medellín & Equihua, 1998). Conversely, some studies have also shown that species richness is not necessarily higher for primary forests compared to secondary ones (Faria et al., 2007; Hawes et al., 2009; Martínez et al., 2009), yet, proximity to primary forests probably plays an important role (Hawes et al., 2009). Agricultural mosaics are higher in biodiversity than annual crops probably because of mixed crops and woody plants on borders (Holland & Fahrig 2000).

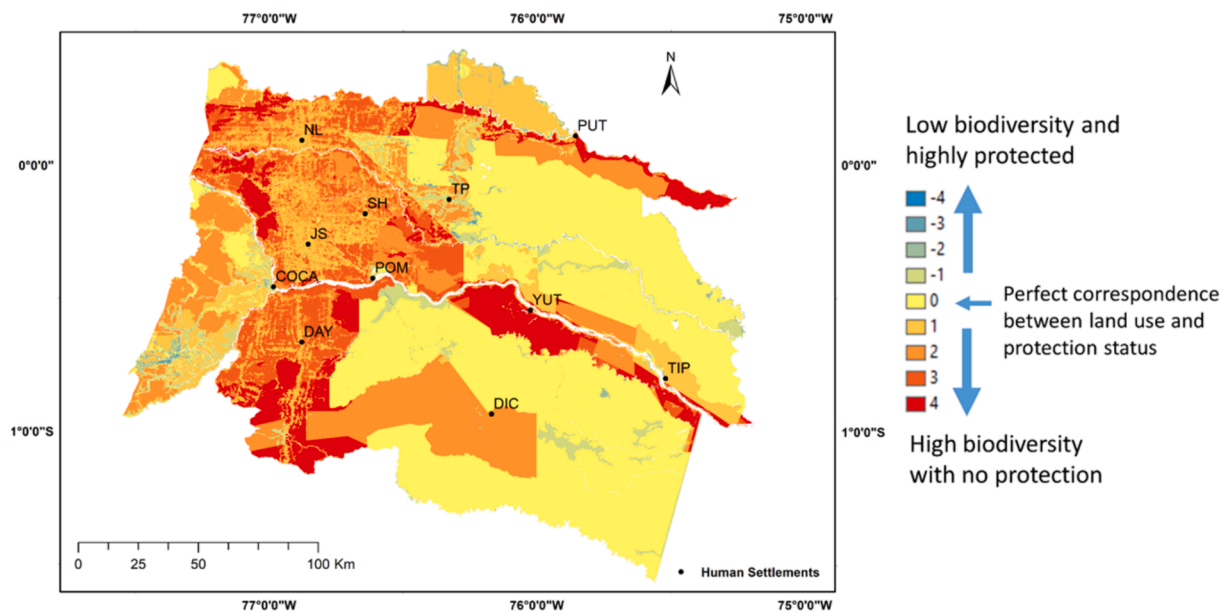


Fig. 6. Difference between potential ecological integrity and biodiversity indices. Positive values indicate high vulnerability according to potential biodiversity but with a low protection status. Negative values indicate low vulnerability according to potential biodiversity, but with a high protection status. Null values indicate an agreement between potential biodiversity and protection status.

Table 6

Spatial congruence analysis of overlapped zones of Protection status and Land Use.

Overlapped zones (ecological integrity and land use)	Congruence value (difference)	Area (km ²)	Contribution (%)
No protection on PF	-4	3,217.3	8.8
No protection on SF	-3	3,430.7	9.4
SB, FP or BR on PF or SF	-2	7,658.7	21.7
BR, FP on AgM	-1	4,5023.5	13.9
YNP, ZITT or CBR on PF	0	14,690.7	42.0
SB on AgM	1	1,284.3	3.8
SB or ProFor on Pasture	2	242	0.5
ProFor on AC	3	58	0.1
YNP, CBR or ZITT on AC	4	4.8	0.02
Total overlapped zones	-	35,051	100

AC=Annual crop; AgM=Agricultural mosaic; BR=Biosphere Reserve; CBR=Cuyabeno Biological Reserve; FP=Forest Patrimony; PF=Primary Forest; ProFor = Protected Forest; SF=Secondary Forest; SB=Socio-Bosque; YNP=Yasuni National Park; ZITT=Intangible Zone.

4.3. Implications for spatial planning and conservation policies

Land use management and planning should integrate both EIVI and BVI for natural heritage vulnerability. In other words, ecosystem services provided by biodiversity can be managed through long-term ecological integrity (Huq et al., 2019). Similarly, the spatial difference operation displays zones where protection status levels and biodiversity value scores are incongruent, e.g., low-scored annual crops that are highly protected, or highly biodiverse primary forest that is not protected. In summary, this analysis indicates areas for prioritizing conservation efforts to highly vulnerable areas. The conservation priority showcases primary forest as the main target, yet careful planning in old secondary forest and agricultural mosaics is also important for conservation (Schulze et al., 2017), because fragmented forests deprive fauna and flora genetic pools (Foster et al., 2013; Silbert, 2002). Specifically, in the NEA, connectivity management should target reconnection of the forested areas with highest EIVI and BVI score values.

4.4. Caveats of the study and steps forward

Future studies on vulnerability assessment should include: the integration of species diversity, species evenness or assemblage composition (Martínez et al., 2009). Relying only on species richness to evaluate biodiversity may skew the rankings in favour of forests as the marginally significance of our statistical analysis indicates. This current method may rank forests over natural native open areas, even though open areas can be essential for hosting rare, endemic, and key species. In addition, species presence/absence using big datasets from the Global Biodiversity Information Facility could be implemented, in which case, auto-correlation analysis, uncertainty and sampling effort should be considered. Furthermore, futures studies may integrate the mapping of biodiversity variability according to local perceptions (Kelemen et al., 2013), or through pixel neighbourhood impacts (Vimal et al., 2012). Neighbourhood impacts in a pixel can be evaluated using weighted averages of border effects, fragmentation or proximity to other land uses that influence biodiversity variability (Brinck et al., 2017). To enhance biodiversity vulnerability indexing, basic research on some taxa in the neotropics, and specifically in the NEA is still needed, e.g., pollinators and insects may need more sampling. Spatial overlay methods have indicated contrasting results i.e., none or low positive correlations have been reported between biodiversity and ecosystem services (Egoh et al. 2009). The integration of other attributes, e.g., habitats or ecosystems in vulnerability assessment may improve future studies. A time-series analysis of shifts in protected site boundaries and land use may assess gain and losses in regard to long-term vulnerability monitoring. Future scenarios modeling approaches may consider landscape configurations (López et al., 2020; Navarro Rau et al., 2023). In an applied geographical context the EIVI and BVI evaluated in a time series may account for protected sites land ownership and boundary shifts due to institutional lock-ins that may, in effect, reduce the scope for increasing the range of adaptation options available (Adger, 2003). In addition, biodiversity conservation priorities may be assessed using ecological knowledge or social and political values (Egoh et al., 2011), because they have been indirectly linked to human wellbeing (Jaderne Houssou et al., 2019; Maes et al., 2016).

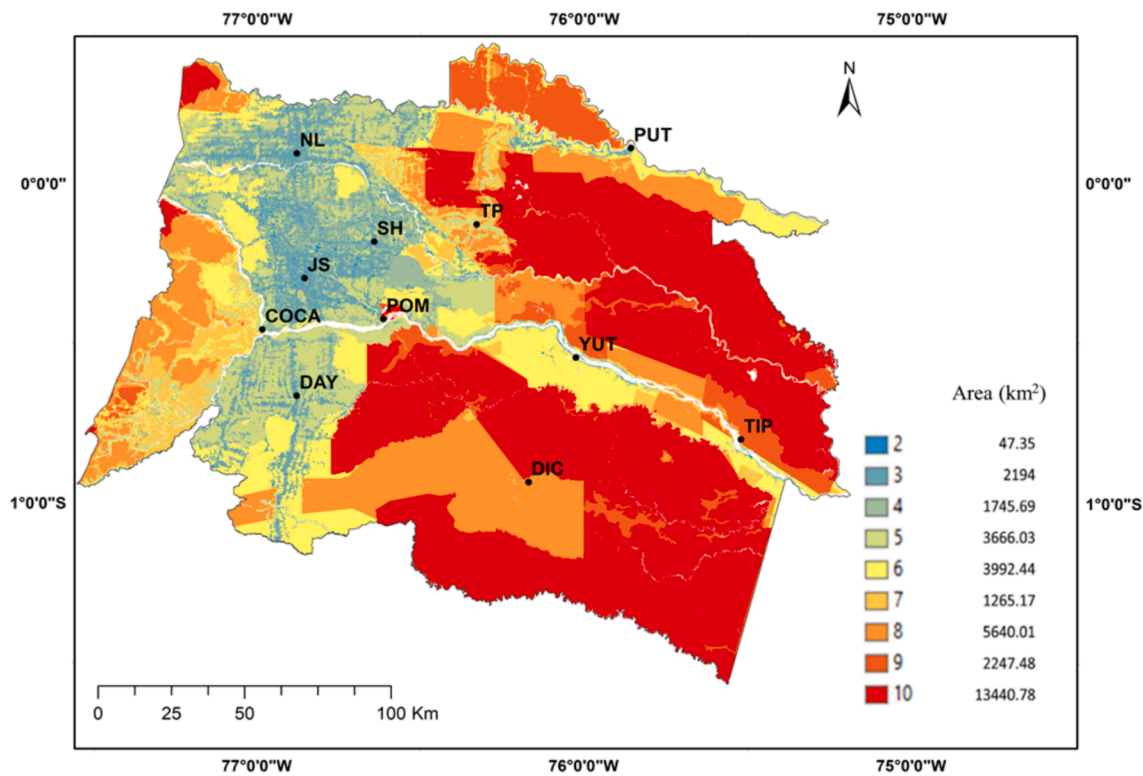


Fig. 7. Addition of single vulnerability indices (EIVI and BIV) for the overall vulnerability of natural heritage in the NEA to any given hazard.

5. Conclusions

The present research integrates homogeneous regional spatial data to rapidly develop two readable vulnerability indices for assessing the vulnerability of natural heritage, e.g., EIVI and BIV; and to identify areas with insufficient protection, mainly those with higher loss potential in the face of unplanned human interventions. The EIVI used protection status as a proxy for ecological integrity. The reliability of the BVI was confirmed by integrating additional biodiversity information. In addition, a time series analysis of land use/cover change and protection status could help to describe the cause-and-effect relationships between temporal boundaries and land use change in order to predict and strengthen the hypotheses established in this study. Other complementary parameters could be included in the vulnerability assessment, e.g., habitats or ecosystems, or related to social and political values. As a step forward, key hazards such as solid waste disposal, mining and oil extraction, and population growth/agricultural expansion could be combined for risk assessment. Finally, this study provides insights for prioritising conservation efforts in areas with sparse biodiversity data using readily available geographic information.

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CRediT authorship contribution statement

J. Durango-Cordero: Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Software, Resources, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization. **M. Saqalli:** Writing – review & editing, Validation, Supervision, Conceptualization.

S.F. Bonilla: Writing – review & editing. **A. Elger:** Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Methodology, Funding acquisition, Data curation, Conceptualization.

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: [Durango-Cordero declares that he currently works for the General Secretariat of Comunidad Andina (SGCAN). This academic work does not represent the SGCAN perspective, which is based on the personal analysis and assessment of the author.].

Data availability

Data will be made available on request.

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Appendix A. Supplementary material

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