

Landscape pattern and habitat quality in buffer areas of Mt. Banahaw – San Cristobal protected landscape, Philippines

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Abstract: Buffer zones are crucial for sustaining Protected Areas (PAs) as they serve as frontlines against threats from agricultural expansion, forest fragmentation, urban sprawl, and other human-induced disturbances. Their ability to support PAs as ecological filters and aid for wildlife conservation depends on their suitability as a habitat. Unfortunately, buffer areas are rarely studied. This study assesses the influence of changes in land cover on landscape patterns and habitat quality in 2-kilometer and 5-kilometer buffer areas of the Mts. Banahaw – San Cristobal Protected Landscape. Open-source land-cover data were used to determine changes in landscape pattern using FRAGSTATS 4.2 and habitat quality using the InVEST model. Results showed that forest remains the dominant land cover in the 2km and 5km buffer areas while urban areas grow at the expense of deforestation following a *forest* → *crop* → *urban* pathway. Landscape metrics indicated fragmentation and perforations of forest matrices while urban areas expand and coalesce to larger patches. This led to a steady increase in low-quality but a substantial decline in moderate- and high-quality habitat. Two-way ANOVA revealed significant effects of buffer distance ($F_{1,29994} = 45.77$, $p < 0.001$) on habitat quality. These emphasize the need for sustainable land-use planning to address forest and habitat degradation to maintain ecological integrity and the protective role of the buffer zone.

1. Introduction

The Protected Areas (PAs) are clearly defined geographical areas, recognized, dedicated, and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values (IUCN, 2008). These include areas of land or water set aside to sustain their unique physical and biological significance (eNIPAS Act, 2018). Globally, PAs are considered cornerstones for biodiversity conservation, where ecosystem processes and functions are sustained and serve as refuges for organisms (IUCN, 2008). PAs safeguard habitats supporting more species (Cooke et al., 2023), including key wildlife species (Mi et al., 2023a). It also serves as a refuge for associated organisms such as reptiles and amphibians (Mi et al., 2023b). Aside from these support services, PAs can be designed to support indigenous (Sze et al., 2022) and local culture, tourism, and recreation (Trišić et al., 2023), and others that promote community equity (Adams et al., 2023). Hence, PAs are recognized for their contribution in addressing the impacts of climate change, forest degradation, and species extinction (Rosales et al. 2023).

Buffer zones are transitional and multi-functional areas surrounding PAs' core areas (Chen et al., 2021; Gonçalves-Souza et al., 2021). These areas can improve landscape

permeability and maintain natural vegetation cover that not only support species movement, genetic exchange (Kubacka et al., 2022; Gonçalves-Souza et al., 2021), sustain refuge habitat (Mashapa et al., 2021; Batumike et al., 2022), but also play important productive, regulatory, and cultural ecosystem services. They are designed to promote conservation by reducing risk, serving as social instruments for sustainable livelihoods, and maintaining landscape connectivity. This makes buffer zones critical ecological filters and socio-institutional spaces (Acharya et al., 2025; Batumike et al., 2022). It also serves as a first line of defense against direct threats from encroachment, fire, and illegal extraction while absorbing or diverting pressure away that can lead to fragmentation of the core (Kubacka et al., 2022; Kleemann et al., 2022).

Regulated resource use and livelihood diversification, such as community forestry, agroforestry, and low-impact harvesting, often occur in buffer zones of protected areas. Allowable resource use and sustainable livelihood programs not only foster stewardship but also generate income (Jumiyati, 2023), reducing pressures to utilize resources within the core (Dhakal, 2021; Acharya et al., 2025; Ramadhan et al., 2025). Buffers can host recreation, environmental education, and tourism, contributing to cultural identity and local incomes and promoting community empowerment and conservation awareness (Jumiyati, 2023; Dhakal, 2021). As frontline areas, however, human-wildlife conflicts often occur in the buffer areas as wildlife home ranges overlap with areas used by humans for livestock and crop production (Dhakal, 2021; Lumacad & Swiatkowska, 2025; Chaudhary et al., 2025). Livestock grazing, free-roaming herds, or crop production in buffer areas influence prey availability and land-use mosaics that consequently affect the risk of human-wildlife conflicts.

As the buffer zones reduce external pressures on protected areas (PAs) and support sustainable livelihoods for adjacent communities, they experience land use land cover (LULC) conversion, which weakens their ecological function, including the decline in habitat quality. While the core of PAs retains good forest cover, anthropogenic activities are displaced into the buffer zones (Mas, 2005; Jusys, 2016) where multiple threats overlap and corridors for pressure transmission (Zhang et al., 2022; Chen et al., 2023; Kim & Yoon, 2025). This results in forest fragmentation, reduced habitat connectivity, and increasing ecological vulnerability (Kintz et al., 2006; Oliveira & Barbosa, 2020) within the buffer zones. The effectiveness and sustainability of the buffer zones strongly depend on governance, enforcement, equal benefit-sharing, and community support (Robinson et al., 2013; Andrade & Rhodes, 2012; Dixit et al., 2024). Hence, the degradation of buffer zones can lead to a decline in habitat quality even with formal protection (Wei et al., 2022; Li et al., 2022; Liu et al., 2022; Wei et al., 2024). This means that a higher protection status alone does not guarantee good habitat quality conditions (Bricca et al., 2024).

In the Philippines, the risk of buffer zone degradation and decline in habitat quality can be aggravated with high rural population density, widespread dependence on forest resources, and overlapping land tenure systems within protected landscapes. Forest cover that serves primarily as a wildlife habitat is being lost due to accessibility, population pressure, and weak law enforcement (Verburg et al., 2006; Apan et al., 2017; Buhay et al., 2023). Poverty and limited livelihood options in upland and rural areas have continuously exerted pressure on protected areas (Agduma et al., 2023) and their buffer zones, leading to land use conversion for crop production, fuel wood collection, and hunting for subsistence, which in turn results in habitat loss (Llave et al., 2018; Escal & Malabarbas, 2021). These social challenges, coupled with inadequate incentives and limited local empowerment, constrain community participation in conservation initiatives (Tumbaga et al., 2021). Deforestation and forest cover removal lead to fragmentation and isolation of habitats, reducing their connectivity for wildlife mobility (Buhay et al., 2023). These can jeopardize conservation efforts for threatened species, necessitating the identification of safe zones and ecological corridors to sustain viable populations (Tabora et al., 2023). In comparison, the buffer zones have a higher risk and are more threatened with habitat degradation than their PAs. Since the ability to sustain its role as

a sink for pressures and threats to PAs while providing ecosystem services depends largely on maintaining its healthful state, it is imperative that the current status and quality of the buffer zone as habitat are monitored. Unfortunately, the buffer zone of PAs is rarely given focus.

This study aims to assess the land cover changes in Mt. Banahaw – San Cristobal Protected Landscape (MBSCPL) over 30 years and how it influences the landscape pattern and its habitat quality. MBSCPL is one of the most important mountain ecosystems in the region, surrounded by rapidly urbanizing communities from the provinces of Laguna, Quezon, and Batangas. This study hypothesized that these ever-growing pressures had been promoting urban sprawl and forest fragmentation, leading to a decline in habitat quality in the buffer zone of MBSCPL. Open-source land cover data of MBSCPL from different time periods were validated and analyzed for land cover changes. Landcover data were then used to assess changes in landscape patterns, including forest fragmentation and urban sprawl, and changes in habitat quality. The results of this study can benefit decision makers in improving policies regarding buffer zones, their establishment, and sustaining their role in the protection of PAs.

2. Materials and Methods

2.1. Study Site

Mts. Banahaw – San Cristobal Protected Landscape (MBSCPL) covers a land area of 10,900.59 hectares covering portions of 9 municipalities, including 2 cities in the provinces of Laguna and Quezon. It lies between 13055' and 14010' latitude and 121016' and 121035' longitude and is about 120 kilometers southeast of Manila. Its catchment basin is important in providing potable water to more than 1 million people.

At present, there is a vague policy on establishing buffer zones in many protected areas in the Philippines. Hence, many PAs in the country have not properly demarcated their buffer zone. In MBSCPL, a 2km buffer area was acknowledged by different stakeholders. In accordance with Republic Act 9847 of 2009, which established the MBSCPL, multiple 2 km and 5 km buffers were created in GIS for this study. Figure 1 shows the extent and geographic location of MBSCPL.

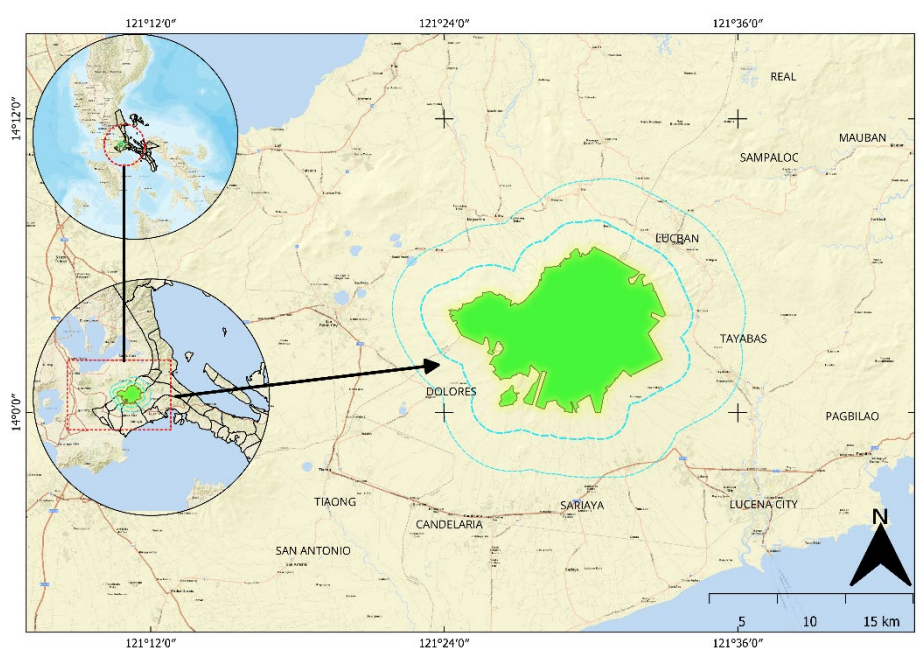


Figure 1. Location of the study.

2.2. Land Cover of MBSCPL

The land cover of the MBSCPL buffer zone in 2000, 2010, and 2020 was adopted from the GlobeLand30 (GLC30) dataset (<https://www.globallandcover.com/>). It was validated from Google Earth images and modified for use in this study. Furthermore, the land cover types were reclassified into 4 major types, namely: 1) cropland, 2) forest, 3) urban, and 4) water bodies. Cropland includes dense and sparse short vegetation cover, which represents annual crops. Forest includes trees greater than 3 meters in height based on GLC30, which represents orchards of fruit trees, coconut-based intercropping, and other perennial multi-canopy layer cropping onsite. These land-cover data were used to model landscape patterns and habitat quality.

2.3. Landscape Patterns and Metrics

Landscape patterns describe composition (type and extent or proportion of patches) and configuration (connectivity, distribution, and proximity or contagion of patches within the landscape) of the landscape (Qui and Turner, 2015) based on patches, which refer to the land cover types in this study. Landscape patterns were characterized using metrics derived from Fragstats 4.2. McGarigal (2015) described Fragstats as a spatial pattern analysis program for quantifying the structure (i.e., composition and configuration) of landscapes that can represent any spatial phenomenon. The landscape metrics used included PLAND (percent of the patch), Number of Patches (NP), patch density (PD), largest patch index (LPI), Cohesion (patch cohesion index), Clumpy (Clumpiness index), and AI (Aggregation Index at landscape level). These landscape pattern metrics were used to characterize forest fragmentation and urban sprawl over the buffer areas.

2.4. Habitat Quality Characterization.

The habitat quality in the 2km and 5km buffer areas of MBSCPL was analyzed using the INVEST Habitat Quality module. Habitat Quality denotes the ecosystem's capacity to provide suitable environmental conditions for species success, based on available resources that support survival, reproduction, and population persistence (Hall et al., 1997; National Capital Project, 2024). Intact habitats with structure and function are of high quality and degrade with increasing proximity to intensive human land uses (Foreman, 2003; National Capital Project, 2024). The InVEST Habitat Quality model integrates land cover, threat factors, and spatial influence of disturbances to estimate land-use patterns and how human threats affect habitat condition and biodiversity across a landscape. Based on the studies of Yang (2021), Hu et al. (2022), Wei et al. (2022), and the InVEST model user's guide, the threat factors, including their weights and maximum impact distances in Table 1, were adopted. The habitat types and their sensitivity to threats adopted for this study are presented in Table 2. The habitat quality values were classified as poor (less than 0.3), moderate (0.3 – 0.6), good (0.6 – 0.8), and excellent (greater than 0.8) based on natural breaks following Lei et al. (2022).

Table 1. Weights and maximum influence distance of different threats.

THREATS	MAXIMUM DISTANCE	WEIGHT	DECAY
agriculture	4	0.5	linear
urban	8	1	exponential
Path walk (3m wide)	0.5	0.4	linear
Road _B (6m wide)	1	0.4	linear
Road _M (10m wide)	3	0.5	linear
Road _N (25m wide)	3	0.6	linear

*Road_B – barangay/ village road; Road_M – town/ municipal road; Road_N – national road

Table 2. Sensitivity of habitat to threats

Land cover	NAME	HABITAT	agriculture	urban	Path walk	Road _B	Road _M
1	agriculture	0.6	0.3	0.8	0.2	0.3	0.5
2	urban	0.1	0	0	0	0	0
3	Path walk	0.1	0	0	0	0	0
4	Road _B	0	0	0	0	0	0
5	Road _M	0	0	0	0	0	0
6	forest	1	0.6	1	0.3	0.4	0.5
7	water	0.9	0.65	0.8	0.2	0.3	0.4

*Road_B– barangay/ village road; Road_M– town/ municipal road; Road_N– national road

2.5. Data Analysis

The annual rate of land cover change was measured using the land use degree dynamics given by the equation 1 below. Two-way ANOVA using raster data models was also made in R software to determine the effects of buffer distance and years of observation on habitat quality in 2km and 5km buffer areas.

$$LUDD = \frac{LC_f - LC_i}{LC_i} * \frac{1}{T} * 100 \quad (\text{Eq.1})$$

where: *LUDD* is the land use degree dynamics in % year⁻¹; *LC_i* is the area of a particular land cover in the initial year; *LC_f* is the area of a particular land cover in the final year; *T* is the time interval in years.

3. Results

3.1. Spatial Extent of Land-Covers in the Buffer Zones of MBSCPL

Figure 2 shows that the forest was the major land cover of the buffer areas of MBSCPL in 2000, 2010, and 2020. However, while forest remains the dominant cover type, fragmentation along the protected area boundary and an increase in perforations of forest matrices in the southern and eastern sections from 2000 to 2020 are evident. The spatial extent of cropland was stable within the 2km buffer but continuously declined at the 5km buffer. This decline in cropland areas is due to conversion to urban development. This indicates land-cover changes within the buffer zones, resulting in urban expansion and forest contraction. Likewise, this increases edge effects while reducing landscape connectivity. This urban expansion, mostly near settlement centers, extends road networks toward the buffer zone of MBSCPL, suggesting increasing anthropogenic pressure and a heightened risk of habitat degradation at the park edges.

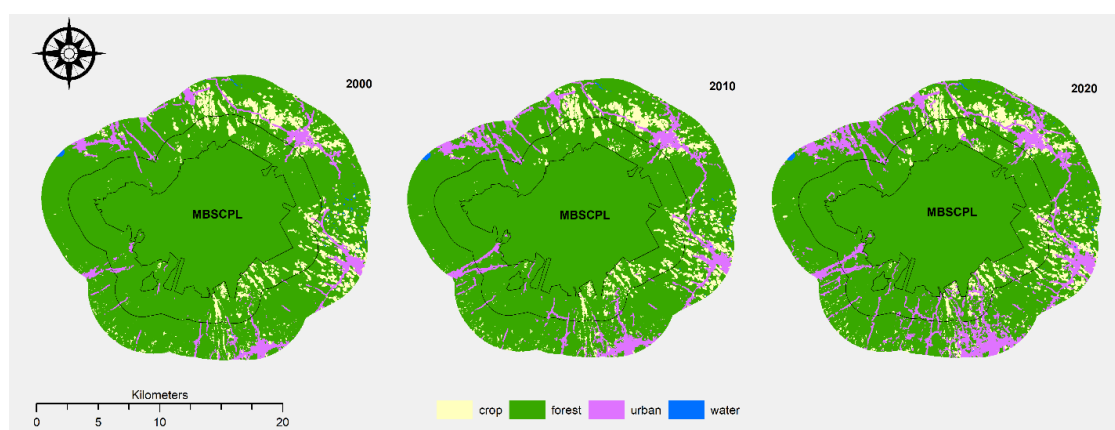


Figure 2. Spatial extent and patterns of major land cover in the buffer areas of MBSCPL.

Table 3 summarizes the landcover distribution in buffer areas of MBSCPL. Results show marked temporal changes from 2000 to 2020, particularly in forest and urban areas, for both 2km and 5km buffer areas. Within the 2km buffer, forest cover declined 367 ha (4.038%) from 9,086.77 ha in 2000 to 8,719.66 ha in 2020, while the urban areas increased more than fourfold, from 102.15 ha to 459.58 ha. Crop areas remained relatively stable with slight fluctuations at around 930 ha. Similarly, a 4.008% reduction in forest cover was observed in the 5km buffer, while the urban area almost doubled, from 1,744.88 ha to 3,450.09 ha (an increase of 97.726%) between 2000 and 2020. Crop areas also showed a continuous decline from 26,818.69 ha to 25,267.10 ha over the study period, which can be attributed to the gradual conversion to built-up land.

Table 3. Landcover distribution from 2km and 5km Buffer areas of MBSCPL in the years 2000, 2010, and 2020 by area (hectares)

Land cover type	Buffer						LUDD (% year ⁻¹)	
	2km			5km			2000-2020	
	2000	2010	2020	2000	2010	2020	2km	5km
crop	934.48	926.49	928.43	26,818.69	26,298.35	25,267.10	(0.032)	(0.289)
forest	9,086.77	8,968.06	8,719.66	3,630.81	3,495.94	3,485.27	(0.202)	(0.200)
urban	102.15	221.31	459.58	1,744.88	2,417.49	3,450.09	17.495	4.886
water	0.26	0.32	1.21	50.64	33.23	42.55	18.269	(0.799)

*LUDD – Land Use Dynamic Degree

3.2. Patterns of Landcover Change

The chord diagram in Figure 3 illustrates the changes and transitions of land cover in both buffer zones. Urban areas in the 2km buffer exhibited the most pronounced gain, which accumulated 349.89% over two decades from 116.64% urban cover expansion in 2000-2010, followed by 107.66% from 2010-2020. The LUDD_{2km} was also considered very high at 17.495% per year, indicating very rapid urban expansion. In the 5km buffer, urban areas nearly doubled (97.73% gain) between 2000 and 2020. LUDD_{5km} was also moderate to high at 4.886% per year, indicating an escalated development pressure toward the protected area boundary. With respect to landcover change directions, both the 2km and 5km buffers reveal a landscape trajectory characterized by *forest* → *crop* → *urban* conversions. The forest remains the primary source of land conversion, followed by substantial flows toward crops and urban classes. This indicates that agricultural expansion and urbanization are the drivers of forest loss in the buffer zones of MBSCPL. A secondary pathway involves the transition of crop areas into urban land. Crop areas decline over time, while the urban land shows the largest increase, specifically in the inner buffer.

3.3. Landscape Patterns and Configuration

Figure 4 shows the changes in spatial structure and configuration of the landscape within 2km and 5km buffer areas of MBSCPL. At the class-level landscape scale, forest remained the dominant land cover, but pronounced land-use changes between 2000, 2010, and 2020 were observed, coupled with changes in spatial structure and configuration. The proportional area (PLAND) of forest declined steadily from 89.76% in 2000 to 88.65% in 2010 and further to 86.26% in 2020. Urban land, on the other hand, showed a remarkable increase in PLAND from 1.01% in 2000 to 2.19% in 2010 and reaching 4.55% by 2020. The number of patches (NP) and patch density (PD) of the forest remained relatively stable over the years, suggesting limited fragmentation at the class level despite declining area. In comparison, the NP of urban areas increased sharply from 55 in 2000 to 87 in 2010 and 221 in 2020, with a corresponding rise in PD from 0.46

to 1.87. Similarly, the largest patch index (LPI) for forest decreased substantially from 89.33% in 2000 to 66.95% in 2010 and 64.71% in 2020, while the LPI of urban areas increased from 0.23% to 1.13%. This indicates a contraction of the forest core area and the emergence of larger, more dominant urban patches, respectively. Aggregation index (AI), COHESION, and CLUMPY consistently indicate high aggregation across all land-cover classes, indicating a well-connected, aggregated landscape.

These results suggest a gradual loss or shrinking of contiguous forest patches, which may reduce core habitat availability and increase edge effects, indicating fragmentation. In addition, the relatively stable NP and PD values for the forest indicate that forest loss results from patch shrinkage rather than extensive fragmentation. This pattern is commonly associated with encroachment and incremental land conversion.

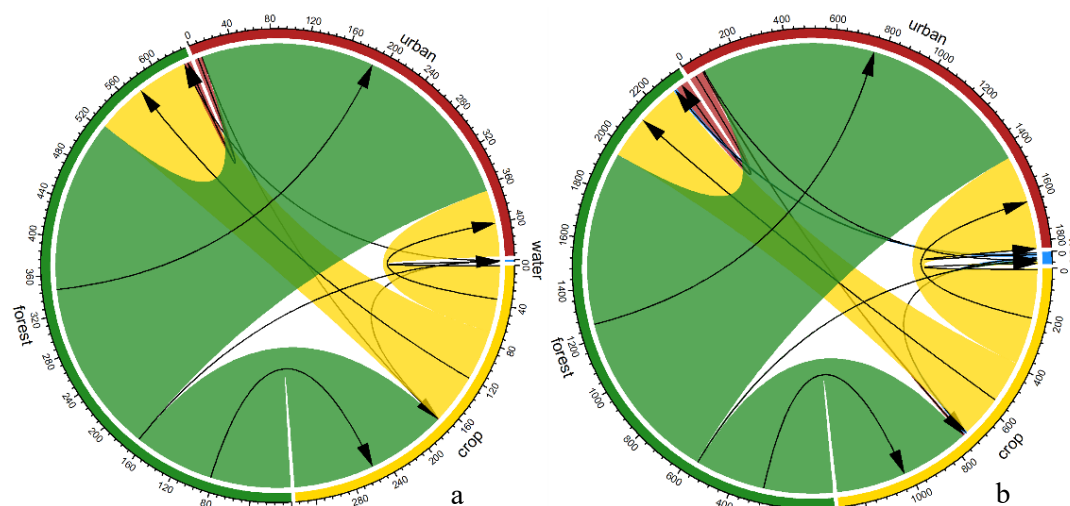


Figure 3. Transition in Landcover of MBSCPL for 2000 (a) and 2020 (b).

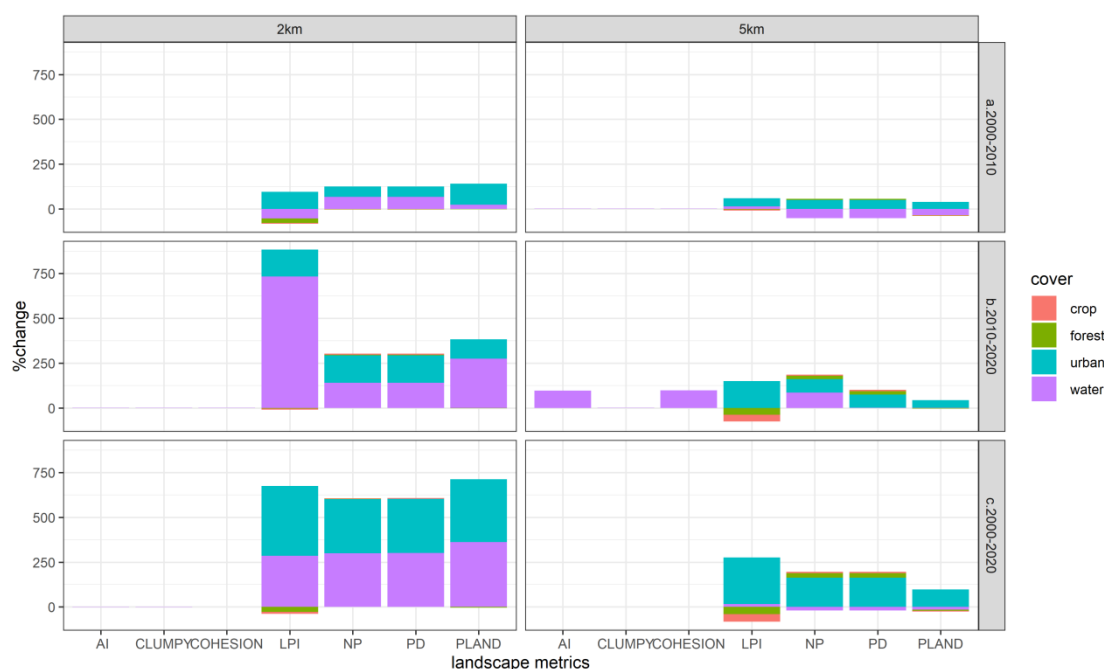


Figure 4. Changes in landscape metrics in the 2km and 5km buffer areas of MBSCPL.

3.4. Habitat Quality

The habitat quality maps in Figure 5 reveal a decline in ecological condition in the 2km and 5km buffer zones in MBSCPL between 2000 and 2020. In the 2km buffer area, the moderate to good habitat patches associated with contiguous forest cover were

noticeable in 2000 but became increasingly fragmented in 2010 and 2020. By 2020, the poor habitat classes dominated the 2km buffer areas with only minimal moderate-quality and virtually no high-quality (>0.6) habitat remaining. Similarly, the 5km buffer areas exhibited a decline in habitat quality. The low-quality habitat already dominated in 2000, with scattered occurrences of moderate-quality habitat. These moderate-quality habitats contracted and became spatially isolated, until low-quality habitats occupied almost the entirety of the buffer landscape by 2020.

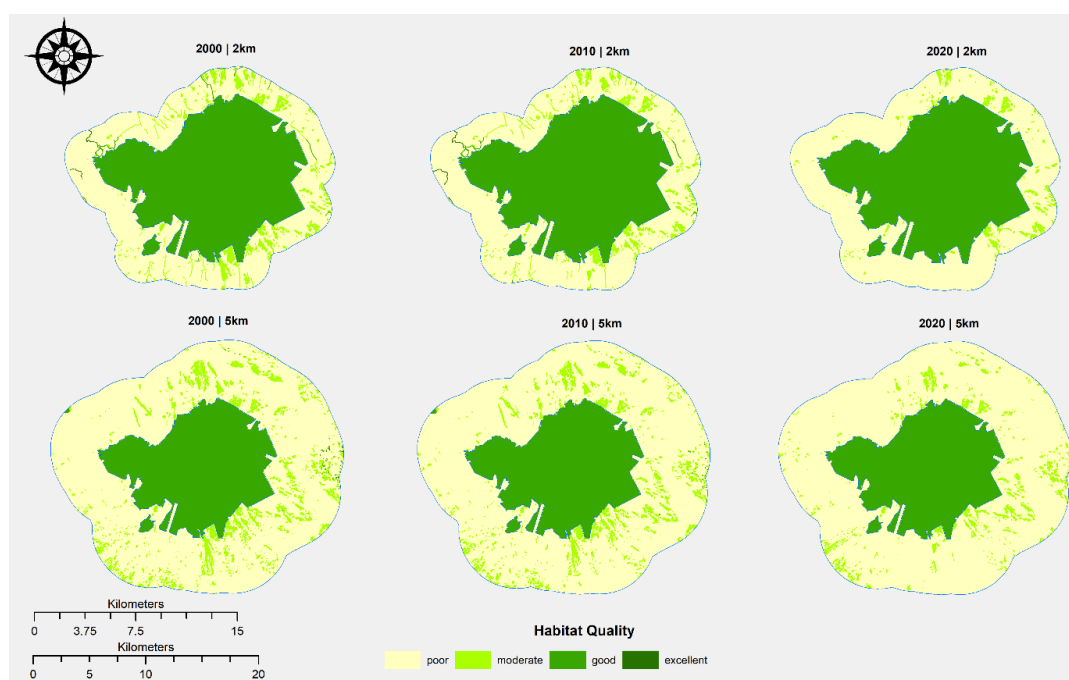


Figure 5. Map showing the habitat quality in buffer areas of MBSCPL.

Furthermore, the areas with low-quality habitat increased steadily for both buffer areas, as shown in Table 4. The low-quality habitat increased from 9,394.25 ha in 2000 to 9,753.34 ha in 2020, and from 28,320.93 ha in 2000 to 29,210.28 ha in 2020 for 2km and 5km buffer areas, respectively. This was an overall increase of more than 3% in the spatial extent of low-quality habitat from 2000 to 2020 for both buffer areas.

In contrast, moderate- and high-quality habitats experienced a substantial decline over the study period. Moderate-quality habitats (0.3–0.6) declined from 701.09 ha to 355.55 ha in 2km buffer areas, while the 5km buffer areas lost almost half of their areal extent over two decades. These were sharp declines in moderate-quality habitats by 49.29% and 43.77% in 2km and 5km buffer areas, respectively (see Table 5). The higher-quality classes (0.6–0.8 and >0.8) were only minimally to completely disappear by 2020 in both buffer areas. This habitat degradation along the buffer areas of the protected area indicates strong edge effects and anthropogenic influence. In the 2km buffer, these intensifying pressures close to MBSCPL can increase habitat isolation and reduce landscape connectivity between the protected core and surrounding ecosystems.

Two-way ANOVA revealed significant effects of buffer distance ($F_{1,29994} = 45.77$, $p < 0.001$) and year ($F_{2,29994} = 107.53$, $p < 0.001$) on habitat quality in 2km and 5km buffer areas. Specifically, the 5km buffer had significantly ($p < 0.001$) more areas with higher habitat quality values than the 2km buffer area, and the habitat quality in 2020 was also significantly lower ($p < 0.001$) than in 2000 and 2010. Likewise, a significant interaction between buffer and year ($F_{2,29994} = 4.00$, $p = 0.018$) was found, indicating that temporal changes differed between buffer zones. At any given buffer distance, habitat quality was significantly higher in 2000, which further declined in 2010 and 2020. These findings suggest that spatial proximity to protected areas, as reflected by buffer areas, influences landscape conditions and habitat quality over time.

Table 4. Spatial distribution of buffer areas to different habitat quality classes.

HABITAT QUALITY	BUFFER					
	2km			5 km		
	2000	2010	2020	2000	2010	2020
Poor	9,394.25	9,499.32	9,753.34	28,320.93	28,493.65	29,210.28
Moderate	701.09	594.94	355.55	2,016.28	1,840.73	1,133.84
Good	22.13	14.73	0.03	21.23	16.19	
Excellent	6.24	7.23		3.40	1.01	

**Poor (less than 0.3); moderate (0.3 – 0.6); Good (0.6-0.8); Excellent (greater than 0.8)*

Table 5. % Changes in Habitat Quality in 2km and 5km buffer areas of MBSCPL.

% Changes in Habitat Quality	BUFFER					
	2km			5km		
	2000-2010	2010-2020	2000-2020	2000-2010	2010-2020	2000-2020
Poor	1.12	2.67	3.82	0.61	2.52	3.14
Moderate	(15.14)	(40.24)	(49.29)	(8.71)	(38.40)	(43.77)
Good	(33.44)	(99.80)	(99.87)	(23.76)	(100.00)	(100.00)
Excellent	15.81	(100.00)	(100.00)	(70.20)	(100.00)	(100.00)

**Poor (less than 0.3); Moderate (0.3 – 0.6); Good (0.6-0.8); Excellent (greater than 0.8)*

Figure 6 also shows that the changes (losses and gains) in habitat clusters converge on the southern and southeastern peripheries of the protected core of MBSCPL, overlapping the rapidly urbanizing areas of Lucena City. Habitat losses in the 2km buffer areas concentrate along the immediate boundary of the protected area, forming localized clusters but appear fragmented and spatially confined. Habitat losses in the 5km buffers, however, appear more continuous and widespread towards the southern and eastern portions. These suggest that habitat losses in the 2km buffer areas are edge-related pressures operating near the park boundary, while losses in the 5km buffer areas indicate broader landscape-level transformation beyond the immediate boundary.

Habitat gains in both buffers exhibit limited, dispersed, and less spatially cohesive patterns than habitat losses. This suggests that natural regeneration or restoration processes are occurring, but at smaller spatial scales relative to loss and are insufficient to offset the scale of degradation. In addition, while the core of MBSCPL remains intact, the surrounding landscape is undergoing progressive modification, which may increase fragmentation, reduce connectivity, and intensify edge effects. The increasing dominance of low-quality habitats also increases vulnerability of the buffer areas and may imply the need for effective restoration and stricter land-use regulations to prevent further ecological decline.

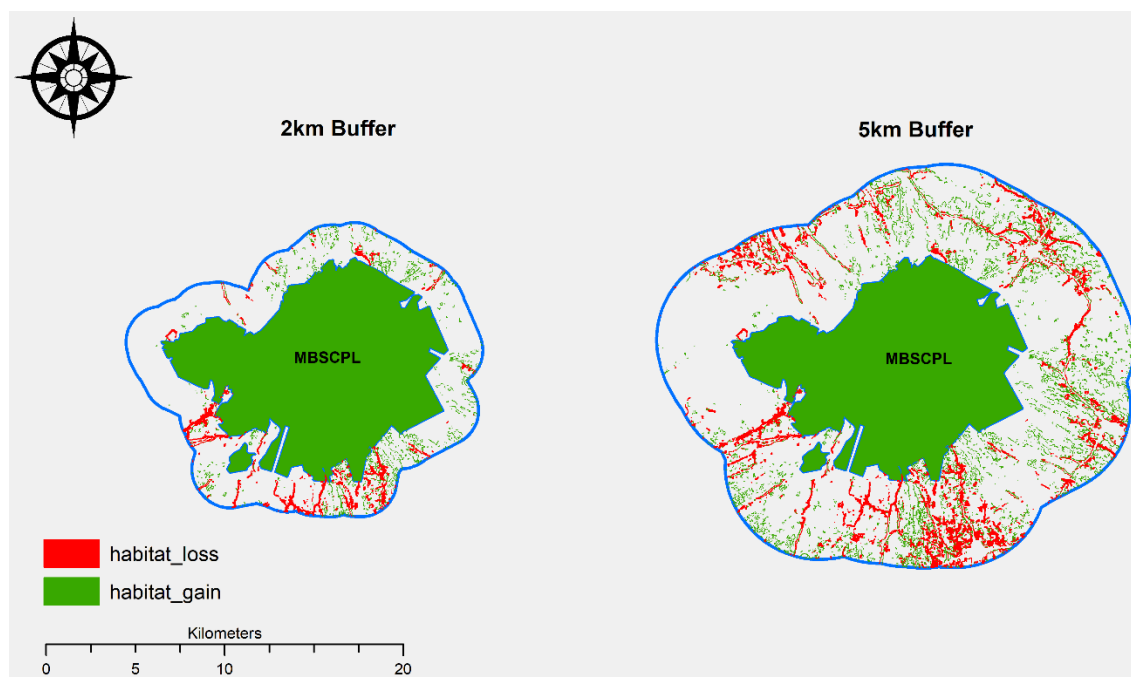


Figure 6. The distribution of habitat losses and gains in buffer areas of MBSCPL.

4. Discussion

Landcover Change, Forest Fragmentation, and Urban Sprawl in the Buffer Areas

The landcover change assessment results showed a consistent expansion of urban areas and a reduction in forest cover within the 2km and 5km buffer areas of MBSCPL. Since cropland remains relatively stable, the notable increase in built-up areas for both buffers may have resulted from the conversion of forest to settlement areas, perhaps to address the demands of the growing population. The landcover change mechanism demonstrated a dominant forest → crop → urban transition pathway. The forest acts as the principal donor class, and urban land as the primary recipient, resulting in forest fragmentation and urban sprawl. The conversion pattern initiates with forest clearing for agriculture, which is progressively followed by urban development and accelerating landscape fragmentation.

The role of urban sprawl and agricultural encroachment on forest losses in the buffer zones resulting in forest fragmentation was also emphasized in the study of De Oliveira et al. (2020), Oliveira & Barbosa (2020), Kleemann et al. (2022), Kija et al. (2020), and Wang et al. (2021). While agricultural intensification, livestock expansion, and livelihood diversification can primarily drive landcover change (Kintz et al., 2006; Ramadhan et al., 2025; Dhakal, 2021), cropland remains almost the same over the study period, indicating it does not significantly contribute to forest loss, but rather to urban sprawl. Urban sprawl may introduce permanent infrastructure, road networks, and settlements reduces forest patch size and accelerating fragmentation despite the protection of the core protected areas (De Oliveira et al., 2020; Oliveira & Barbosa, 2020). Agricultural expansion may fluctuate with market demands and livelihood options. The forest in many parts of these buffer areas of MBSCPL includes orchards and tree farms planted to fruit trees, with some in combination with coconut. These landcover types are less regulated (particularly beyond the 2km buffer of MBSCPL), and since the protected area was only institutionalized in the early 2000s, legal rights over these areas are recognized. This may also explain why forest loss often concentrates near protected area edges, with buffer zones in the Philippines (Apan et al., 2017; Verburg et al., 2006; Buhay et al., 2023). This shift of landcover change from protected areas to its surrounding zone was also considered edge effect and spillover (Jusys, 2016; Li et al., 2024).

In addition, landcover changes in the buffer zones can be influenced by governance issues such as weak law enforcement, inadequate buffer delineation and demarcation, and limited stakeholder coordination (Robinson et al., 2013; Zhang et al., 2021; Al Mamun & Kim, 2020). Also, despite the assertion of buffer zones as regulated multiple-use areas under the National policy under the ENIPAS Act (DENR DAO 2019-05), challenges in the policy implementation remain (Mallari et al., 2013; Agduma et al., 2023). Socio-economic factors, including the community's perceptions towards conservation responsibilities (Fang et al., 2021; Dixit et al., 2024), can also strongly influence compliance and land-use decisions. These not only effectively deter illegal encroachment and poaching in buffer zones and protected area cores, but also the adoption of unsustainable land management practices. For instance, community engagement, participatory management, and promoting conservation value have demonstrated potential to moderate land conversion in the Philippines (Escal & Malabarbas, 2021; De Alban et al., 2015; Tumbaga et al., 2021). When government mechanisms are limited, urban sprawl accelerates (Shen et al., 2022; Gonçalves-Souza et al., 2021). Furthermore, land-use decisions in buffer areas often prioritize short-term economic growth over long-term ecological integrity in developing countries (Mallari et al., 2013). This preference may be associated with poverty and the need for income and livelihood options for upland rural communities in the buffer zones.

Landscape Patterns Change and Habitat Quality Degradation

From a largely forest-dominated buffer landscape in 2000, two decades of urban sprawl have created a more fragmented mosaic, particularly in buffers near MBSCPL, a landscape transition towards urban expansion at the expense of forest and croplands. As buffer landscape transitions along *forest* → *crop* → *urban* pathway, it has rippling consequences on the landscape ecosystem and its services. The decrease in values of PLAND and LPI while NP and PD remain the same for forest patches indicates forest fragmentation due to loss and shrinkage in forest patches. When fragmented, habitat quality and connectivity in forests landscape declines (Kubacka et al., 2022; Mas, 2005), but the edge effects and the isolation increase. In contrast, urban land exhibited intensification with sharp increases in NP, PD, LPI, and cohesion between 2000 and 2020. This reflects early-stage urban sprawl characterized by dispersed development and suggests subsequent coalescence of urban patches into larger, more continuous built-up areas over time. This transition from fragmentation to consolidation is typical of maturing urban landscapes and may have significant implications for hydrology, microclimate, and ecosystem services.

The forest fragmentation and urban sprawl threaten the ecological integrity while increasing landscape ecological risk (Kija et al., 2020; Wang et al., 2021) and weakening the buffering function against climate stressors (Xu et al., 2022). Forest fragmentation alters landscape configuration by reducing average patch size, increasing edge-to-core ratios (De Oliveira et al., 2020; Oliveira & Barbosa, 2020), and isolating remnant forest patches within agricultural or built-up matrices. Urban sprawl amplifies habitat quality decline and disturbance by introducing permanent infrastructure, transportation corridors, and human settlements. Expansion of built-up area fragments continuous natural habitat while increasing noise, pollution, and human presence. Built-up areas reconfigure space and alter microclimatic conditions, and increase susceptibility to invasive species (Kubacka et al., 2022). These alter the suitability of the buffer areas as habitat.

Likewise, forest fragmentation and urban sprawl put biodiversity and ecosystems at risk and further degrade the habitat. While edge effect may increase species diversity, particularly generalist species, interior-dependent and less mobile species become more vulnerable as the landscape becomes edge-dominated (Cooke et al., 2023; Mi et al., 2023). In the tropics, forest fragmentation and the proliferation of transport corridors, combined with urban encroachment, increase human–wildlife interactions and ecological instability (Kija et al., 2020). As habitat degrades in buffer areas, human–wildlife conflicts

intensify (Branco et al., 2020; Lumacad & Swiatkowska, 2025; Chaudhary et al., 2025) as wildlife searches for food across the landscape, thereby increasing the probability of human-interaction. This, in addition to the vulnerability of species such as insects and herpetofauna to land-cover modifications (Chowdhury et al., 2023; Mi et al., 2023), can endanger these species and marginalize the area for their habitation.

The consistent high values of Aggregation index (AI), COHESION, and CLUMPY indicate high aggregation across all land-cover classes and a well-connected landscape, although evidence shows the gradual decline in forest connectivity metrics. This decline in forest connectivity can affect species movement and gene flow. The increasing connectivity of urban areas, on the other hand, emphasizes the growing structural dominance of built-up land while aggravating habitat isolation for forest-dependent species.

Implications on Protected Area Management

Sustainable management of buffer areas. The results of the study imply that buffer areas need to be properly managed as an additional layer of protection for protected areas. Proper demarcation of both the protected area core and its buffer areas can benefit the sustainable management of protected areas, as it guides local communities and law enforcement. In addition, the determination of an appropriate size (distance from the protected area core) of buffer is important in the sustainability of the buffer zones. The size of the buffer zone can determine how many threats and pressures can be absorbed while remaining functional as habitat for flora and fauna. While results show that the immediate buffer (2km) still has better-quality habitat, policy restrictions prove insufficient to prevent habitat loss. The establishment of buffer zones also often conflicts with the rights of the landowners. In the study, urban sprawl still encroaches on restricted buffer zones due to the legal rights of the landowners, particularly those with land titles before the enactment of the law establishing the protected area.

Landcover change-led Habitat Degradation. The results demonstrate urbanization as a key driver of landscape change during the study period, with progressive urban expansion at the expense of forest integrity. These emphasize the need for appropriate and sustainable land-use planning strategies that limit further forest degradation, maintain landscape connectivity, and mitigate the ecological impacts of continued urban growth. It also implies that incorporating ecological connectivity into urban planning to support biodiversity, habitat quality, and overall ecosystem functioning amid accelerating land-cover change. The coalescence of urban patches into larger, more continuous built-up areas can further degrade habitat quality by increasing the isolation of forest-dependent species and impairing hydrology, microclimate, and ecosystem services.

5. Conclusions

This study assesses the influence of changing land cover on landscape pattern and habitat quality in 2km and 5km buffer areas of the Mts. Banahaw – San Cristobal Protected Landscape for over two decades. This study concludes that land cover change in the buffer areas results in forest fragmentation and urban sprawl, leading to degradation in the quality of habitat for flora and fauna. While forest remains the dominant land cover in the 2km and 5km buffer areas, fragmentation and perforations of large forest patches are noticeable, and urban areas expand and coalesce to larger patches. This is indicative of growing edge effects and reduced landscape connectivity. Urban areas grow at the expense of forest losses, while agriculture remains stable. Forest fragmentation and urban sprawl alter the landscape, isolating forest-dependent species and enhancing human-wildlife contact. These results indicate increased risk, reduced ecological integrity, weakened buffer function, and reduced habitat quality.

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