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Chapter 14

ECOLOGICAL IMPACT OF HABITAT LOSS ON AFRICAN LANDSCAPES AND BIODIVERSITY

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ABSTRACT

A main characteristic of human-driven dynamics of landscapes is habitat loss, leading to a degradation and fragmentation of natural land covers. This anthropogenic landscape change is often visible as the development of road systems or as urban growth. As a consequence of the pattern/process paradigm, these dynamics will have profound ecological impacts on biodiversity and ecosystem function, by means of edge effects, connectivity decline, home range reduction, and increased species mortality. In order to evidence the direct link between landscape patterns, their dynamics, and their influence on ecological communities, five case studies are discussed: (1) degradation of forest cover in the Collines department of Central Benin as a consequence of charcoal production, firewood collection and cotton production, (2) edge effects on rodent diversity in the Masako Forest Reserve in the Democratic Republic of the Congo, (3) potential impacts of road development for the mining industry on forest habitat quality in the territory of Kambove in the Democratic Republic of the Congo, (4) simulation of deforestation patterns in the region of Lubumbashi in the Democratic Republic of the Congo and the role of road networks and city proximity herein and (5) deforestation rates in Ubundu and Kisangani (Democratic Republic of the Congo) as a function of population density and proximity to the city limits.

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1. Area Change as a Key Element of Landscape Transformation

1.1.Spatial Pattern of Landscapes

Landscape ecology focuses on three characteristics of the landscape: (1) structure, i.e. the spatial relationships among the distinctive ecosystems or landscape elements present, (2) function, i.e. the interactions among the spatial elements, and (3) change, i.e. the alteration in the structure and function of the ecological mosaic over time (Forman &Godron 1986). The pattern/process paradigm forms a central hypothesis of landscape ecology; it states that there is a direct link between the spatial pattern of the landscapes and the ecological functions characterizing them (Turner 1989; D'Eon 2002; Gustafson & Diaz 2002). A triangular relationship describing the interdependence of the spatial arrangement and geometry of the landscape elements, the types of elements present, and the spatial and ecological processes is widely accepted (Noon & Dale 2002) and justifies the focus of landscape ecology research on spatial pattern analysis.

Landscape ecology is motivated by a need to understand the development and dynamics of pattern in ecological phenomena, the role of disturbance in ecosystems, and characteristic spatial and temporal scales of ecological events (Urban et al. 1987). This focus on landscape dynamics forms the core of this contribution. Landscape dynamics cause the change of two landscape pattern components: composition and configuration (Bogaert et al. in press). Landscape composition refers to the number of patch types in the landscape, and to their proportional area. Landscape configuration refers to the spatial arrangement and geometry of the landscape elements (Estreguil& Mouton 2009; Bogaert et al. in press). For both concepts, a series of metrics has been developed. Due to their interrelation (Noon & Dale 2002), compositional change also implies configuration change and vice versa.

1.2.Landscape Transformation Processes

The most direct way to quantify composition dynamics is to analyze land cover area change. With regard to configuration dynamics, it has been shown that there appears to be a limited number of common spatial configurations that can result from land transformation processes (Franklin & Forman 1987; Collinge& Forman 1998; Bogaert et al. 2004). Initial developments to define a typology of these processes (Forman 1995; Collinge 1998; Jaeger 2000) put the emphasis on transformations characterized by area decrease of the class of interest. In Bogaert et al. (2004), transformations were also included causing an increase of the area of the class of interest, hence covering a wider range of possible types of landscape dynamics. Perforation, dissection, fragmentation, shrinkage and attrition were associated with area decrease; aggregation, creation and enlargement are characterized by area increase; deformation and shift do not cause area change. In the algorithm of Bogaert et al. (2004), for a focal land cover class, traditional pattern geometry (number of patches, area and perimeter) is monitored to identify the dominant process; losses and gains are not addressed separately nor their compensation or cumulative impacts; the dominant spatial process over one short time period was considered exclusive and was not quantified (Estreguil and Mouton 2009). It should be noted that landscape dynamics are often a combination or sequence of different transformations (Bogaert et al. 2004; Figure 1).

The aforementioned typology was successfully applied by Bogaert et al. (2008) and Barima et al. (2009). In Vogt et al. (2007), the typology was erroneously denoted as a landscape-level classification algorithm that identifies ten fragmentation categories, including internal and external fragmentation. Area change remains the most characteristic feature of landscape dynamics; in case of natural land covers, often denoted as habitat loss generally caused by anthropogenic landscape change (also known as “anthropization”) (Bogaert et al. in press), which refers to these changes in which natural land covers, such as forests, wetlands or natural grasslands, lose their dominance and are replaced by anthropogenic types such as agricultural fields, plantations, industrial plants or urban zones. Both groups of land covers are then characterized by opposite dynamics, since the decrease of one land cover type enables another cover type to expand. For example, the expansion of urban zones does imply the loss of other cover types in the peripheral zone. This observation justifies the consideration of dynamics, which are characterized by area increase (Bogaert et al. 2004). Therefore, the landscape transformation processes could be divided into two groups for landscapes experiencing anthropogenic effects (Bogaert et al. in press): perforation, dissection, fragmentation, shrinkage and attrition will be typical for natural land covers; enlargement, aggregation and creation for anthropogenic land covers; deformation is a neutral process that could characterize both and implies no area change. The transformations denoted by “shift” are more probable for anthropogenic types (Bogaert et al. in press).

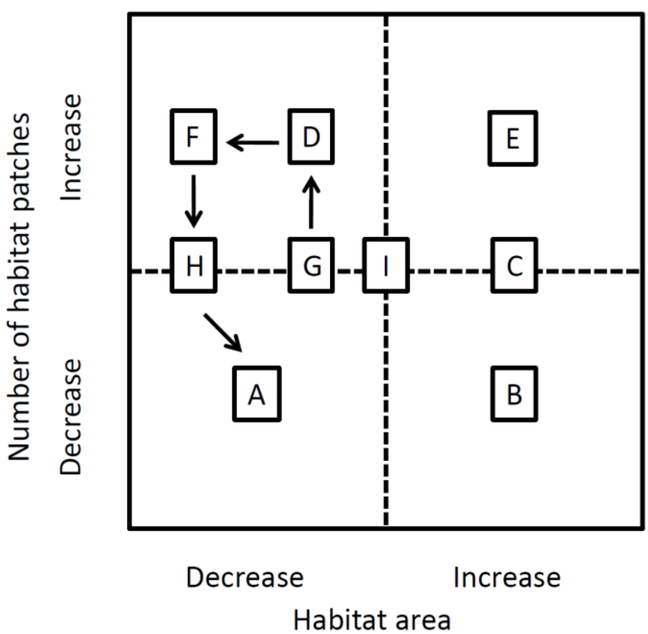

Figure 1. Landscape transformation processes and their impact on habitat area and on the number of habitat patches. (A) attrition, (B) aggregation, (C) enlargement, (D) dissection, (E) creation, (F) fragmentation, (G) perforation, (H) shrinkage, (I) shift/deformation. To separate shift and deformation, consideration of the total patch perimeter is required. The arrows indicate the general transformation sequence of natural land covers subject to degradation (Forman 1995). Transformation processes are defined in Bogaert et al. (2004).

1.3.An Example of Forest Degradation in Central Benin

Fragmentation is undoubtedly the transformation type with the largest impact on landscape pattern and function. Forest fragmentation is therefore considered one of the most important conservation issues of recent times (D'Eon 2002) and has been identified as the most important factor contributing to the decline and loss of species diversity worldwide (Noss and Cooperrider 1994). The relative importance of habitat loss versus fragmentation with respect to impacts on biota is not yet entirely clear. In many cases, both factors are correlated (August et al. 2002). However, there is increasing evidence (August et al. 2002) that habitat abundance rather than patchiness is the dominant landscape feature controlling biotic integrity. This will have far-reaching implications for the integration of human activity with natural habitats (August et al. 2002).Forest area decrease is the main parameter to describe forest fragmentation (Gascon et al. 2003); an increase of the number of patches is also observed (Bogaert et al. 2004). Although the same observation can be made for the dissection process (Forman 1995; Bogaert et al. 2004), both processes should be distinguished because of the lower transformed area for this latter process.

Habitat area loss and a change of the number of patches are the main characteristics of pattern change in the case of habitat degradation. Deforestation patterns in Central Benin, for a study area situated in the Collines department (8°45'N, 2°39E), exemplify this type of pattern dynamics. Vegetation in the Collines department was initially dominated by open rain forests, woodlands and savannahs; due to anthropogenic pressure for charcoal production, firewood collection or cotton production, the original forest class was intensively disintegrated between 1972 and 2006. Mosaics of agricultural fields, fallow lands and savannahs became the dominant land cover type. Landscape dynamics were studied by means of three Landsat images (resolution 30 m) of November 1972 (MSS, paths 206-207, rows 54-55), January 1986 (TM, path 192, row 54) and December 2006 (ETM+, path 192, row 54). By means of a supervised classification, the forest class was identified; consequently its spatial pattern was studied by means of the evolution of the sizes and numbers of the patches between 1972 and 2006.

Figure 2 shows the changes of the total forested area and the number of forest patches from 1972 to 2006. Total forest area decreases throughout the period, with a decrease from 6626 ha to 248 ha between 1972 and 1986, and further on to 44 ha in 2006. The number of patches increased initially from 193 (1972) to 331 (1986) which indicated that the forest underwent fragmentation (Bogaert et al. 2004); consequently, the number of patches decreased to 102 in 2006, indicating that a part of the remaining patches were subjected to attrition.

It is appealing to analyze the patch size distribution for the time period considered. This can be done by means of two characteristic values of the distribution: the average patch area and the area of the largest patch. This latter value is preferentially expressed relative to the total forest area (largest patch index, McGarigal& Marks 1995). Figure 3 shows that both metrics decrease between 1972 and 2006, with a very strong decrease between 1972 and 1986, where the largest patch index drops from 0.96 to 0.10; the average patch size is characterized by a similar evolution and went from 34.3 ha to 0.8 ha. The second period, from 1986 to 2006, is characterized by similar changes but at a slower rate: the largest patch index decreases from 0.10 to 0.08, while the average patch size drops from 0.8 ha to 0.4 ha. Both figures signal an important degradation of the forest cover in the study area, characterized by a first phase of fragmentation, subdividing the large forest patches in many small ones and converting a large area of forest in other cover types. In a second phase, a decrease in number and in size of the remaining forests is observed, leading to attrition according to Bogaert et al. (2004). At the end of these transformations, forest area has almost disappeared in the landscape: only 0.7% of the initial forest remains and the largest patch in 2006 represents less than 0.1% of the largest patch initially observed.

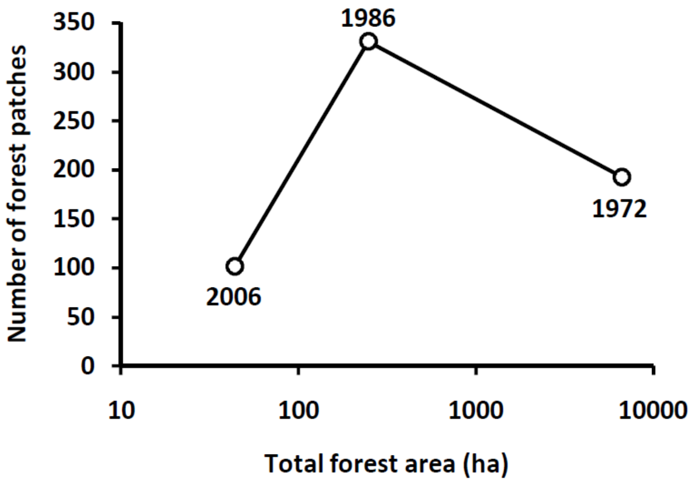

Figure 2. Forest cover disintegration in the Collines department (Benin) between 1972 and 2006. Land cover data based on Landsat imagery. Due to anthropogenic pressure, open rain forests have been degraded and substituted by a mosaic of agricultural fields, fallow lands and savannahs. Landscape transformations are characterized by fragmentation (between 1972 and 1986) and attrition (between 1986 and 2006).

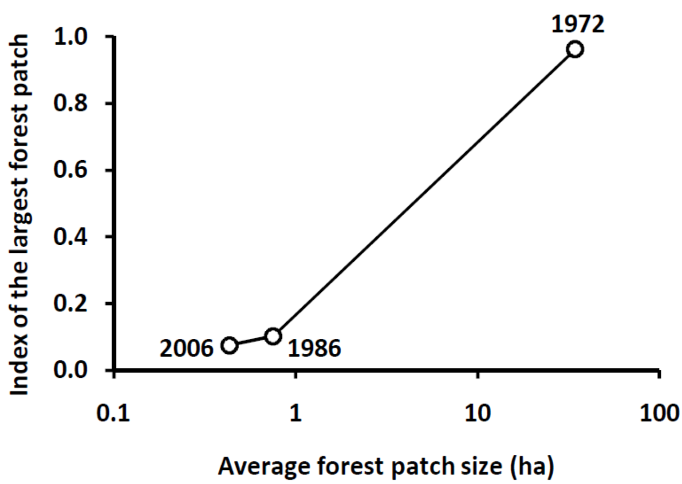

Figure 3. Forest cover disintegration in the Collines department (Benin) between 1972 and 2006. Land cover data based on Landsat imagery. Due to anthropogenic pressure, open rain forests have been degraded and substituted by a mosaic of agricultural fields, fallow lands and savannahs. Forests patches have decreased in size, especially between 1972 and 1986, when the forests underwent fragmentation, as evidenced by the evolution of the largest patch index. After 1986, the remaining forest patches were subject to attrition.

1.4.Fragmentation and the Edge Effect: Evidences from the Masako Forest Reserve (Democratic Republic of the Congo)

The direct ecological impact of landscape fragmentation is evidenced by the edge effect (Bogaert et al. 2008). Edge effects are observed when two different land cover types are adjacent and when the edge contrast is sufficiently high (Forman 1995, Farina 2000a); vegetation structure (e.g. height or density) can be used as a proxy for this contrast (Estreguil& Mouton 2009). The contact between contrasting land covers is often a consequence of the substitution of natural land covers such as forests by anthropogenic types such as agricultural fields (Bogaert et al. in press). The peripheral contact zones of both patches involved are altered with regard to their microclimates, which can be assessed by means of variables such as wind velocity, air temperature, relative humidity, soil water content, and light intensity (Chen 1991, Groom &Schumaker 1993). Edges between different habitats are considered unique as a result of both biotic and abiotic influences (Groom &Schumaker 1993). The impact of land cover change will be underestimated if only the area, which is converted to an anthropogenic land cover type is taken into account (Chen 1991; Bogaert et al. in press). As a consequence of the direct relation between ecological conditions and biodiversity, the edge zone will be characterized by a distinct fauna and flora, when compared to the centrally situated parts of the adjacent patches, generally denoted as interior habitats (Bogaert et al. in press). For forest habitats, it has been shown that edges develop distinct environmental gradients that in turn lead to the development of unique edge communities dominated by a suite of species adapted to edge conditions, such as shade intolerant species (Estreguil and Mouton 2009).

The impact of edge formation on faunal diversity was tested for rodent populations (Rodentia) in the Masako Forest Reserve (MFR; 0°36'N, 25°13'E) in the Democratic Republic of the Congo. MFR, with an area equal to 2105 ha, is situated at about 15 km from Kisangani and is characterized by an equatorial continental climate type denoted as Af according to Koppen (Dudu 1991). MFR is mainly composed of primary forests of *Gilbertiodenrondewevrei* (Caesalpiniaceae), next to secondary forests and fallow lands (Makana 1986; Kahindo 1988; Dudu 1991; Mabay 1994). Secondary forests are dominated by *Pycnanthusangolensis*, *Zanthoxelongilletii*, *Cynometrahankei*, *Petersianthusmacrocarpum*, *Funtumiaelastica* and *Uapacaguineensis*. Fallow vegetations are characterized by associations of *Aframomumlaurentii* and *Costuslucanusianus* and by those of *Triumfettacordifolia* and *Selaginellamyosurus*.

To explore the differences in rodent diversity between the secondary forest habitat, the fallow habitat and the edge habitat situated in between the former two types, rodents were captured in the three habitats by means of a grid covering 1 ha and composed of the following types of rat traps: 50 Lucifer traps, 50 Sherman traps and 20 traditional traps composed of a branch and a halter. Animals were captured during three periods (November 2008 till January 2009; May and June 2009; April and May 2010). Species have been identified based upon morphological characteristics and phylogenetic sequence analyses of the mitochondrial DNA (Terryn et al. 2007) at the Royal Belgian Institute of Natural Sciences. In order to show that the edge zone was characterized by a different rodent diversity, the association between the habitats was tested by means of a χ^2 test using species presence/absence data in a 2×2 contingency table (Causton 1988). If habitats were different, their species composition was also expected to be different, and a negative association was expected. Since species abundance data were also available, the correlation between the abundances for the different habitats was also explored. In case of different habitat characteristics, the abundances between the habitats were also expected to be different, leading to a non-significant relationship. For both analyses, habitats were compared two by two. To exclude population size effects, abundances were expressed as a function of the population size for each habitat.

Table 1. Species captured in the Masako Forest Reserve (Kisangani, Democratic Republic of the Congo) during seven months between November 2008 and May 2010. A grid containing 120 traps covering 1 ha was placed in three habitats: secondary forest, fallow land, and the edge zone separating them.

Family	Species
Muridae	*Praomys jacksoni*
	Hybomys univittatus
	Deomys ferrugineus
	Lophuromys dudui
	Hylomyscus stella
	Stochomys longicaudatus
	Hylomyscus aeta
	Malacomys longipes
	Oenomys hypoxanthus
	Lemniscomys striatus
	Nannomys minutoides
	Praomys misonnei
	Thamnomys rutilans
	Hylomyscus parvus
	Mastomys natalensis
Gliridae	*Graphiurus lorraineus*
	Graphiurus surdus
Sciuridae	*Funisciurus anerythrus*
	Funisciurus pyrropus
	Paraxerus boehmi
Cricetidae	*Cricetomys emini*
Thryonomyidae	*Thryonomys swinderianus*
Anomaluridae	*Anomalurus derbianus*

Table 1 shows the species list of the rodents captured (all habitats pooled): 6 families and 23 species were identified. The fallow habitat was characterized by 18 species (529 individuals), the forest habitat by 18 species (391 individuals), and the edge habitat by 20 species (248 individuals). This higher species number, despite a smaller number of individuals, is not unusual, since edges are often denoted as biological cornucopias, characterized by high species richness and density (Forman 1995). It should be noted that this observation does not confirm the analysis cited in Barima et al. (in press) regarding the same study area, but using data collected in a shorter time frame.

The association test showed no significant negative associations between the fallow habitat and the edge habitat (χ^2=0.27), nor between the fallow habitat and the forest habitat (χ^2=1.35); or between the forest habitat and the edge habitat (χ^2=0.73). This suggests a certain similarity between the three habitats considered, resulting in many common species between the habitats (fallow-edge: 16 species; fallow-forest: 13 species; forest-edge: 16 species). Only one species (*Anomalurusderbianus*) was observed only in the forest habitat. One single species (*Mastomysnatalensis*) was observed only in the fallow habitat. Not a single species was found only in the edge habitat, which confirms the often-observed commonness of edge species (Forman 1995). Species mobility between the adjacent habitats could be invoked as another reason for this faunal similarity (Duplantier 1989; Dudu 1991; Duplantier et al. 1997; Fagan et al. 2003; Strayer et al. 2003; Bogaert et al. in press). The low species numbers in the contingency table could be mentioned as another possible cause of these non-significant results (Causton 1988).

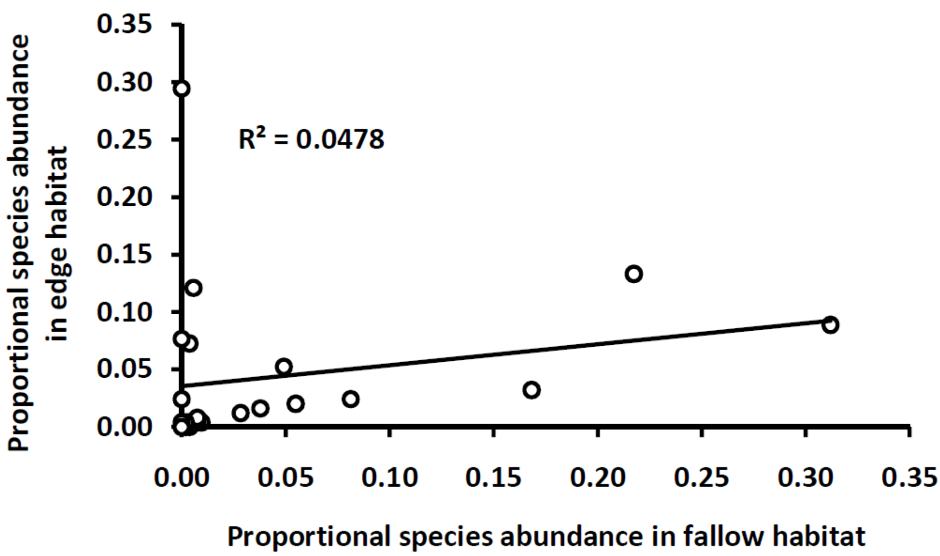

Figure 4. Comparison of rodent abundances (23 species; 1 species absent in both habitats; 6 species found only in one of both habitats) between the fallow and the edge habitat in the Masako Forest Reserve (Kisangani, Democratic Republic of the Congo). Captures effectuated during seven months between November 2008 and May 2010 using grids containing 120 traps and covering 1 ha. The non-significant linear relationship suggests differences in habitat characteristics between the fallow and edge habitat. Abundances were expressed as a function of the population size for each habitat.

Correlation analysis of the species abundance data enabled distinguishing of the habitats. A non-significant linear correlation was found between the fallow and the edge habitat (R^2=0.048; $p>0.05$; Figure 4) and between the forest and edge habitat (R^2=0.129; $p>0.05$; Figure 5), which underlines the unique character of the edge biotope, as already concluded by

Iyongo et al. (2009). This observation also underlines the importance of considering the different components of diversity, i.e. richness and abundance distribution, when comparing species assemblages (Magurran 2004; Barima et al. in press). A high significant linear correlation (R^2=0.396; p<0.01; data not shown) was observed when the abundances of the forest and the fallow habitats were compared, which was not expected due to the lower number of species in common, when compared to the edge habitat. This correspondence between the abundances of the forest and fallow habitats, as well as the non correspondence of the abundances of the edge habitat with its neighboring habitats, was likely caused by the most abundant species in the edge habitat (*Hylomyscusaeta*; 73 individuals) which was only captured eight times in the forest habitat and which was absent in the fallow habitat. This similarity between the forest and fallow habitats was already mentioned by Iyongo et al. 2009. The preference of species for the edge habitat was already reported for *Graphiuruslorraineus* in Iyongo et al. (2009) and confirms the distinct character of the edge biotope.

For an optimal biodiversity in a landscape, an intermediate frequency of edge and interior habitats is suggested (Naiman et al. 1988 in Farina 2000b). In Vogt et al. (2007) an alternative spatial interpretation of forest conditions is proposed. “Core forest” refers to forested zones situated relatively far away from forest-non forest boundaries; “patch forest” comprises coherent forest regions that are too small to contain core forest; “perforated forest” defines the boundaries between core forest and relatively small perforations; “edge forest” includes interior boundaries with relatively large perforations as well as the exterior boundaries of core forest regions. Core forest can be considered here as a synonym of “interior habitat”.

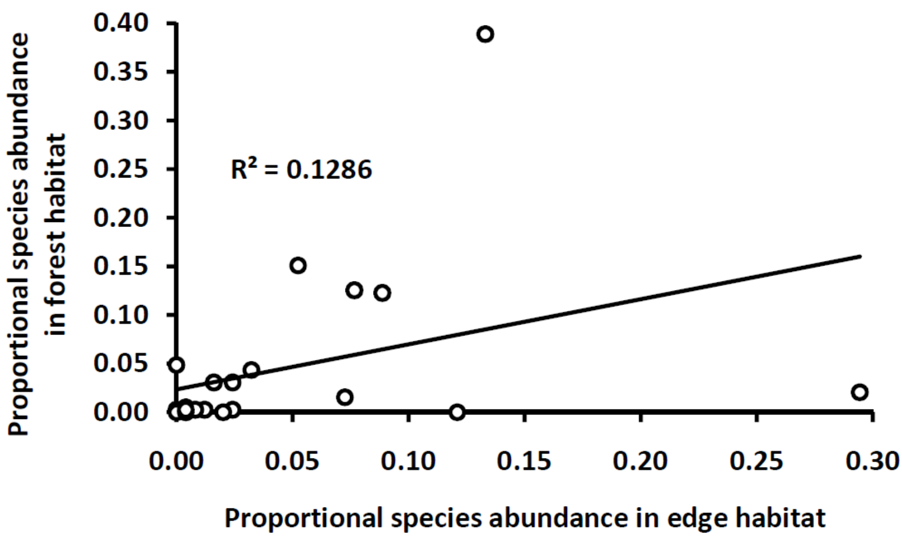

Figure 5. Comparison of rodent abundances (23 species; 1 species absent in both habitats; 6 species found only in one of both habitats) between the edge and the forest habitat in the Masako Forest Reserve (Kisangani, Democratic Republic of the Congo). Captures effectuated during seven months between November 2008 and May 2010 using grids containing 120 traps and covering 1 ha. The non-significant linear relationship suggests differences in habitat characteristics between the edge and the forest habitat. Abundances were expressed as a function of the population size for each habitat.

2. HABITAT LOSS CAUSED BY ANTHROPOGENIC STRUCTURES: ROAD SYSTEMS

2.1. Impacts of Roads on Fauna

Forman (1995) identifies six major causes of land transformation: deforestation, suburbanization, corridor construction, desertification, agricultural intensification and reforestation. A road corridor refers to a road as the surface for vehicle movement, plus any associated usually vegetated parallel strips (Forman 1995). Carr et al. (2002) distinguish road effects on landscape composition and connectivity. The former effects are habitat loss, introduction and quality changes (Forman 1995; Carr et al. 2002; Collinge 1996). The latter ones refer to reduced animal movement across roads, also known as the barrier-effect (Forman 1995; Forman 1998; Carr et al. 2002; Lindenmayer& Fisher 2006; Eigenbrod et al. 2008). It is generally difficult to separate habitat loss from habitat subdivision effects, although necessary to adapt landscape management (Forman 1995; Lindenmayer& Fisher 2006). Road disturbance corridors, acting as filters, affect invertebrates as well as large mammals (Forman 1995; Carr et al. 2002). Roads separating home ranges may produce subpopulations that are genetically different and more subjected to local extinction (Carr et al. 2002; Forman 1995; Forman & Alexander 1998). The barrier-effect is species-specific and depends on the scale and on the animal's movement mode (e.g. flying versus crawling) (Lindenmayer& Fisher 2006), as well as on traffic volume and inhospitable road corridor width (Forman 1995; Carr et al. 2002; Eigenbrod et al.2008). Road mortality represents the main sink in road corridors. Where wildlife corridors and roads intersect, road-killed animals are frequently observed (Forman 1995; Forman & Alexander 1998; Lindenmayer& Fisher 2006).

Though designed as conduits for human populations, big mammal predators may use unpaved, narrow roads at night (Forman 1995; Carr et al. 2002). Some generalist, disturbance-adapted species use roads for their dispersal, some of which are invasive, facilitating the spread of disturbance and disease (Carr et al. 2002; Forman 1995; Forman & Alexander 1998).

2.2. Road Systems Causing Edge Effects: Estimating the Potential Impact of Road Network Development in Kambove (Democratic Republic of the Congo)

Road creation crossing wooded areas increases the amount of forest edge through habitat dissection, resulting in a loss of habitat for forest interior species (August et al. 2002), which generally leads to reduction or extinction of local large-range populations, often already rare and endangered (Forman 1995; Carr et al. 2002; Collinge 1996). Wide (>90 m) natural, forested or shrubby roadside strips, as observed in Australia (Forman 1995, Forman & Alexander 1998) can contain a small mammal diversity similar to the matrix, while narrow (<10m) roadside strips can contain more exotic species. Overall, the number of species in road corridors is usually high, though mainly composed of edge species (Forman 1995). Dust from roads affects the matrix mostly in dry climates, and where the chemistry of the road materials differs from that of the matrix. Vegetation changes due to this dust are reported to extend 10-20 m from the road (Forman 1995). The denser a road network, the higher its ecological effects (Forman, 1998). Additional edge habitat exposed to road effects (noise, runoff of chemicals, particulate matter, etc.) affects local plant and animal communities (Forman 1995; Carr et al. 2002; Marmor&Randlane 2007).

In order to illustrate the potential impact of road networks on natural habitats, road infrastructure development for the mining industry in a forested area in Katanga (Democratic Republic of the Congo) has been studied. The territory of Kambove (10°52'S, 26°38'E) with an area of 24164 km² was chosen as the study area. Three maps have been used: (1) a map showing the road system of 1990 (road width equal to 12 m) provided by the *Office National des Routes Congolaises* (scale 1:3,000,000), (2) a map showing the road system of 2008 (road width equal to 8 m) provided by the *Office National des Routes Congolaises* (scale 1:3,000,000), and (3) a land cover map provided by the Royal Museum for Central Africa (scale 1:2,500,000) (Laghmouch& Hardy 2008). The land cover map was simplified into a binary map, to separate the forest cover from all other cover types. Road systems of 1990 and 2008 were combined in one single map; consequently the road and forest maps were overlaid and the scales were adjusted. The potential edge effect caused by the road network was simulated for the following distances of edge influence: 100 m, 250 m, 500 m, 750 m, 1000 m, 2500 m and 5000 m. This wide range of distances corresponds to edge penetration distances observed in situ and/or applied in simulation studies (Laurance 1991; Murcia 1995; Lindenmayer& Fisher 2006). Forest interior habitat I) was defined as the forest habitat situated outside the zone of edge influence. For every distance of edge influence, the corresponding interior-to-edge ratio (R) was calculated:

$$R = \frac{I}{E}$$

$I + E = A$ with A equal to the total forest area. If no edge penetration distance is considered, $E = 0$ and R cannot be calculated; for $I = E$, the distance of edge influence will equal the 'interior-to-edge breakpoint distance' (Bogaert et al. 2001); for very large distances of edge influence, $I \approx 0$, $E \approx A$ and $R \approx 0$. In Figure 6, some examples are given of the different distances of edge influence used in the simulations.

Figure 7 shows the effect of the distance of edge influence on R. As a consequence of the large forest cover in the study area (12618 km²) and the relative simple road system (2874 km), R remains relatively high ($R > 10$) up to a distance of edge influence of 750 m. It should be noted that a sharp decrease of R is observed from $R \approx 80$ for a distance of edge influence of 100 m to $R \approx 10$ for a distance of 750 m. Later on, for larger edge widths, R decreases rather slowly.

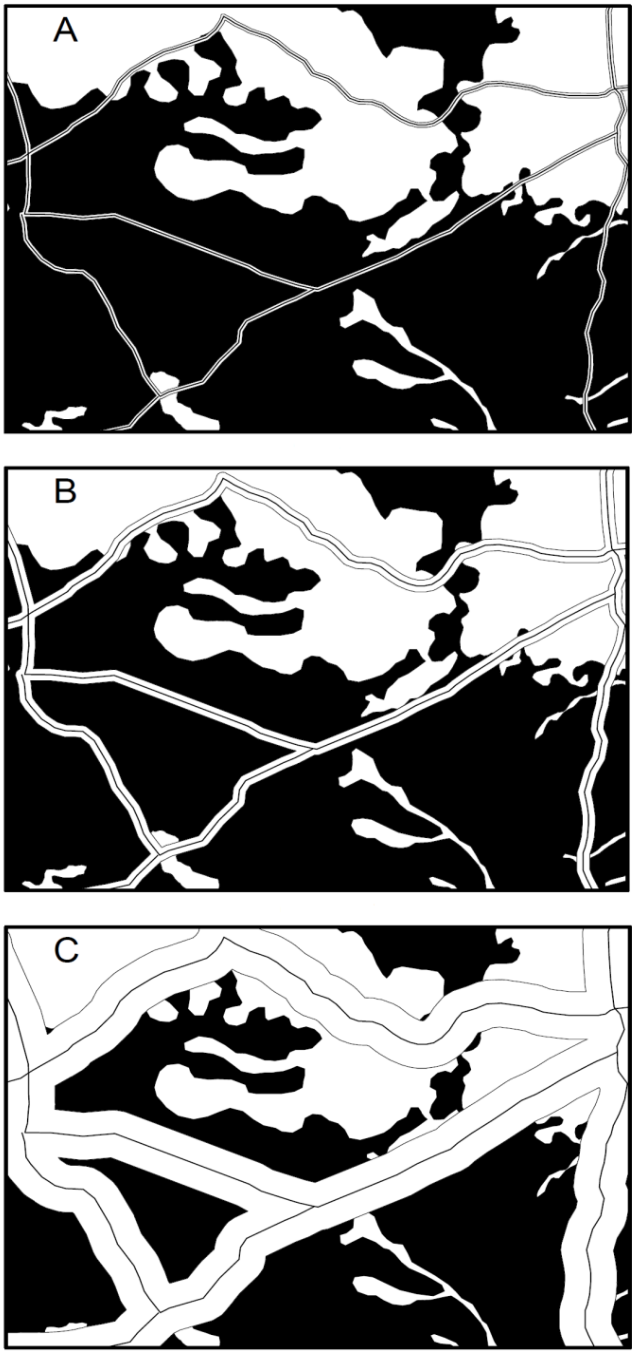

Figure 6. Potential edge effects as a consequence of road system development in the territory of Kambove (Katanga, Democratic Republic of the Congo). Three distances of edge influence are shown: (A) 100 m, (B) 250 m and (C) 1000 m. As a consequence of the edge effect, forests situated in the edge zone will be altered with regard to their microclimate and diversity. The disturbed forest vegetations situated inside the edge zones are given the same color as the 'non forest' class to emphasize the difference with the non disturbed forest patches forming the 'forest interior' (shown in black). A subzone (380 km²) of the study area is shown.

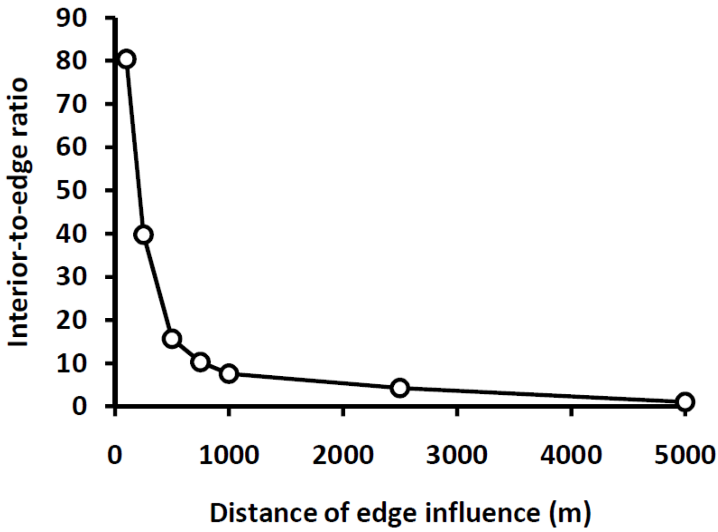

Figure 7. Assessment of the ecological impact of a road system (2874 km) on forest habitats (territory of Kambove, Katanga, Democratic Republic of the Congo) by means of the interior-to-edge ratio. Only forests situated outside the area defined by the distance of edge influence are considered as 'interior forest habitat'. The low density of the actual road system can explain the high values of the ratio.

This behavior of R can be explained by the fact that many forests are actually still situated far away from the roads; a R of about 80 (for a distance of 100 m) indicates that there is 80 times more interior habitat than edge habitat, hence about 99% of the forest area is situated farther away than 100 m from the road network. For a distance of 750 m, the R value suggests that still about 91% of the forests are situated at more than 750 m away from the roads. At a distance of 5000 m, equality between I and E is observed, indicating that even at this large distance, 50% of the forests will not be disturbed by the roads. However, this observation should be interpreted with caution. In Barima et al. (in press) it has been shown that, according to concession maps of the Congolese Mining Cadastre (Kasongo, 2008), 78.5% (91348 km²) of the forest cover of the Katanga province is situated inside recognized mining concessions. When all these concessions are activated, the density of the road system will increase significantly, and the interior-to-edge ratio will consequently decrease strongly, since every forest patch is expected to be situated very close to one or several roads; in this case interior habitats will undoubtedly become scarce.

2.3.Accessibility by Roads as a Trigger of Landscape Dynamics Exemplified by Deforestation Patterns in Lubumbashi (Democratic Republic of the Congo)

Road and railway corridors, even in well-managed forests, provide access to previously remote regions through landscape dissection, encouraging human expansion and associated disturbances like logging, hunting and agricultural practices (Forman 1995; Pedlowski et al. 1997; Forman & Alexander 1998; August et al. 2002; Lindenmayer& Fischer 2006). Hereby, the significance of road access to remote areas emphasizes that roads are of central ecological importance in most landscapes (Forman 1995). According to the corridor model of Forman (1995), land transformation commonly results from construction of a new corridor such as a road, rail line or irrigation canal, which opens up an area in a linear fashion, dissecting the initial land type at the outset. Spread then proceeds outward from the corridor on both opposite sides. Branch lines generally follow forming a dendritic pattern, which is a typical deforestation fishbone pattern (Forman 1995; Pedlowski et al. 1997; Batistella et al. 2003; Frohn&Hao 2006).

An example of deforestation patterns as a consequence of a road system is presented for Lubumbashi (Democratic Republic of the Congo). In a study area of 1445 km² containing Lubumbashi and its surroundings, defined by the geographic coordinates 27°17’17”E, 27°38’57”E, 11°29’35”S, 11°49’31”S, four peripheral subzones of about 1700 ha each were studied, situated Northeast (NE), Northwest (NW), Southeast (SE) and Southwest (SW) of the city. Land cover dynamics were analyzed by means of two Landsat TM images of August 1984 and June 2009 (resolution 30 m; path 173; row 68). Three land cover classes were considered. The forest class was dominated by the *miombo* woodland forest type; the savannah class contained different types of savannah vegetation and also those vegetations forming a part of the agricultural system (fallow lands, fields); a third class was denoted as “other” and contained land covers such as bare soil, water bodies or constructions.

For each subzone, a transition matrix was composed (Bamba et al. 2008; Barima et al. 2009). Consequently, a first order Markov model was applied to simulate future forest cover change between 1985 and 2050; the annual probabilities for the Markov model were derived from the transition matrix using the Urban &Wallin (2002) method. The composition of the landscape at time t, denoted as the vector x_t, is determined by means of the composition of the landscape at time $t-1$, denoted as x_{t-1}, and the probability matrix M containing the annual transition probabilities between the classes:

$$x_{t-1} \times M = x_t \ .$$

Initially (1984), the Southwestern, Southeastern and Northwestern study areas showed a higher forest presence with respectively 65.3, 54.7 and 47.3 km². A lower forest cover of 25.4 km² characterized the Northeastern zone. In 2009, forest cover had decreased in all zones, but at a different rate; a high decrease was noted in the Southwestern (56%) and Northeastern zones (47%); an intermediate decrease was noted in the Southeastern zone (29%) while a low decrease was observed in the Northwestern study zone (13%). Figure 8 shows the results of the simulation analysis by means of the Markov model. *Miombo* area change is expressed as a fraction of the largest value observed for each subzone. Due to the overall decrease in forest area, the highest value is observed at the beginning of the period for every subzone. The Northeastern and Southwestern study zones are clearly marked by a sharper forest area decline, of about 40% and 60%respectively; on the contrary the Northwestern zone is expected to lose less than 10% of its forest area when compared to the 1985 level; the Southeastern zone is expected to lose about 20% of its forests. This stronger regression rate on the Northeast-Southwest axis is not a coincidence; both zones are crossed by important roads connecting Lubumbashi with Likasi and Kasenga to the Northeast and to Kasumbalesa and Kipushi to the Southwest. A star- or tentacle-shaped pattern change, determined by the presence of cities and the roads connecting them (Merlin 1991), is observed (Bruneau& Pain

1990). The Northeastern zone is also characterized by anthropogenic pressure caused by the presence of the Kinsevere mining site and the military camp of Kimbeimbe. It should be noted that these observations should be verified in the future, since the first order Markov simulation technique is limited by its stationarity precondition and by the fact that spatial dependencies are not considered (Urban &Wallin 2002).

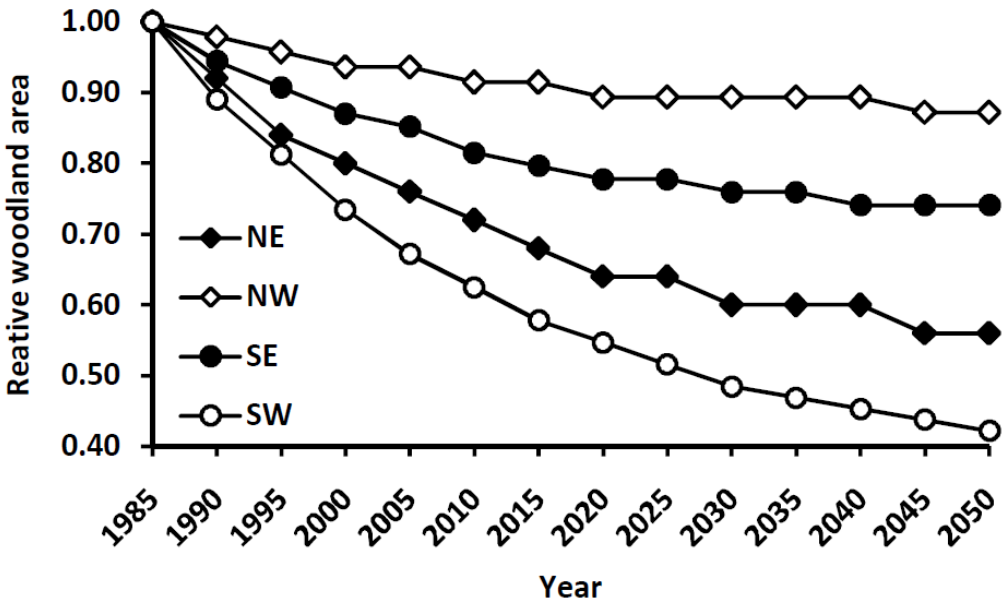

Figure 8. Deforestation of woodland (miombo) around Lubumbashi (Democratic Republic of the Congo) between 1985 and 2050 based on a first order Markov model. Four zones of about 1700 ha situated Northwest (NW), Northeast (NE), Southwest (SW) and Southeast (SE) are considered. Woodland area is expressed as a fraction of the largest value observed for each subzone. Land cover data are based on Landsat images from 1984 and 2009. The higher deforestation rates observed Northeast and Southwest of Lubumbashi are caused by the presence of important roads to neighboring cities.

3. Habitat Loss Caused by Anthropogenic Structures: Expansion of Urban Areas

Human population growth is a fundamental driver of land conversion by increasing the need to produce or extract more food, fuel, and fibers, and to develop infrastructure to support homes and commerce (Pedlowski et al. 1997;Robinson et al. 2005; Su et al. 2010). Currently, the majority of people live in urban environments, most of them in developing countries. There, urbanization is broader and faster than what happened earlier in developed countries (Yang & Lo 2003; Deng et al. 2009; McGee 2009; Su et al. 2010).

3.1.Urban Growth Trends and Their Impacts on Landscape Pattern and Ecology

Most common at the landscape scale is the little-planned spread of suburbs. Such patterns are highly non-random, and tend to mainly reflect geomorphic and transportation templates (Forman 1995). A few common mosaic sequences can be identified: concentric rings spreading outward from an adjacent city, growth along an exurban transportation corridor, and spread from satellite towns, plus infilling (Forman 1995; Greene 1997). Forman (1995) designed models for simulating the spread of urban habitat. The main models are nuclear ones: bubble growth around one or more centers or nuclei. The nuclei model is described as growth from a few spots within the landscape, as for many settlement patterns or non-native species invasions, producing new areas expanding centrifugally towards one another (Forman, 1995). The nucleus model, considering one single spot, can be considered as a zoom to one single case (Forman, 1995).

Urban growth also implies conversion of peripheral farmland into urban areas, which forces agriculture to be developed further away, in more remote and undisturbed areas (Greene 1997). Urban sprawl affects natural habitat through total area decrease of the matrix by perforation (complementary process to urban enlargement) and patch suppression, but also through fragmentation and decreasing connectivity between adjacent habitat patches as well (Bogaert et al. 2004; Deng et al.2009; Su et al. 2010).The results of this fragmentation are an altered habitat composition, edge effects, disrupted hydrological systems, and modified interactions among patches (Su et al. 2010).

Moderate to negative urban population growth with spatial sprawl and density decrease occur in developed countries, which is called suburbanization (Forman 1995; Robinson et al. 2000; August et al. 2002; McGee 2009). This phenomenon began after the Second World War, when massive economy boosting required increase in production and consumption, leading to car production, wages increase and eased access to propriety. These conditions favoured family settlement in individual detached houses with gardens in the suburbs (Yang & Lo 2003; Vandermotten&Marissal 2004; McGee 2009). These tendencies are particularly developed in the United States, where farmland has been progressively converted into urban land use, with satellite towns (edge cities) around the primary kernel (Greene 1997; Robinson et al. 2000; Yang & Lo 2003; McGee 2009).

As for developing countries, unprecedented population growth and urban extension are observed (Robinson et al. 2000; Cohen 2003; Sudhira et al. 2004; McGee 2009). Urban growth is here linked to colonialism and globalization. Since the end of the Second World War, Occidental power began an economic development campaign with the idea that globalization is the only way to modernize the state, which required labour concentrations (Cohen 2003; Yang & Lo 2003; McGee 2009). Urban-oriented development strategies and restructuring of agriculture decreased the proportion of agricultural employment, increased rural-urban income disparities and accentuated migration from rural areas to towns, which could not absorb such a demand in labour force (Cohen 2003; McGee 2009). Poverty and fast growth prevented public authorities from providing proper infrastructure and housing to the new inhabitants, who were forced to live in slums. Emerging countries like China or Brazil present intermediate dynamics, with similar drivers as occidental nations during their industrial revolution, where rural exodus met labour force demand, though combined with much higher population growth (Cohen 2003; Deng et al. 2009; McGee 2009).

3.2.Urban Growth, Demographic Pressure and Deforestation in Kisangani and Ubundu (Democratic Republic of the Congo)

In many developing countries, severe deterioration of vegetation and the physical environment occurs as a result of firewood collection for domestic and industrial use, particularly in savannah woodlands, shrubs, and, increasingly, tropical forests (Lindenmayer& Fisher 2006). For example, 90% of the African population depends on firewood for their energy needs (Lindenmayer& Fisher 2006), which they find in surrounding forests.

The impact of urbanization on deforestation rates is illustrated for two cities situated in the Oriental Province of the Democratic Republic of the Congo: Kisangani (0°31'N, 25°11'E), the province's capital and characterized by about 360 inhabitants per km² (UNEP 2004), and Ubundu (0°21'S, 25°25'E), characterized by about 10 inhabitants per km² (UNEP 2004). By means of two Landsat TM images (resolution 30 m, path 176, row 60) from February 1986 and March 2001, forest cover was determined for both dates through unsupervised classification. This classification was visually validated by means of the land cover map produced by the Royal Museum for Central Africa (Laghmouch& Hardy 2008). Two land cover types were considered for further analysis: forest and non-forest. Concentric buffer zones were constructed around the 1986 city limits of Kisangani and Ubundu with radius of 5, 10, 15, 20, 25, 30, 35, 40, and 45 km (Figure 9).

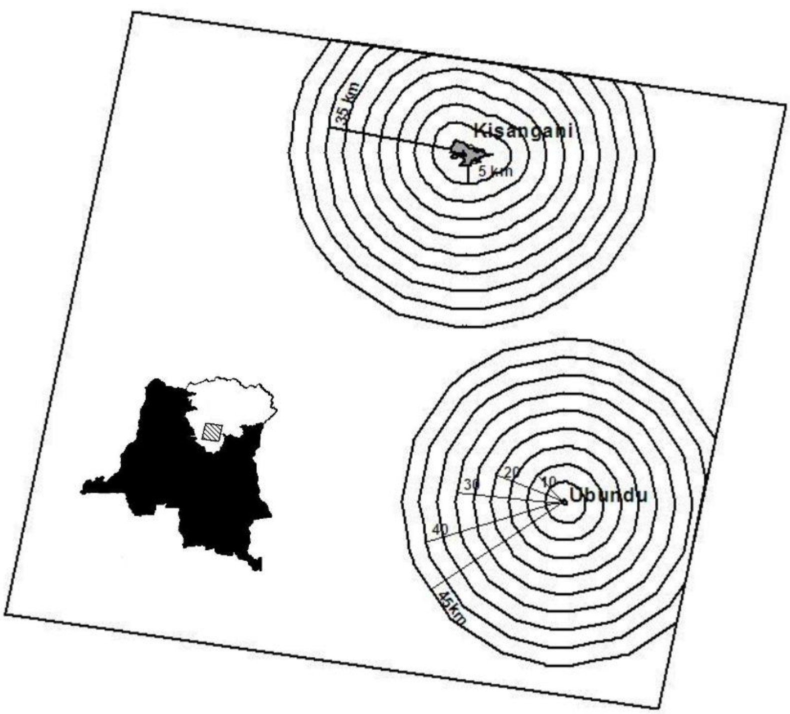

Figure 9. Situation of the study area (shaded area) containing Kisangani and Ubundu in the Oriental Province (shown in white) of the Democratic Republic of the Congo (small map). Around the limits of both cities, buffer zones with a radius from 5 to 45 km were constructed. In each buffer zone, the deforestation rate between 1986 and 2001 was determined. Land cover was based on Landsat data.

In each buffer zone, the deforestation rate D was calculated as a percentage of the initial forest area of 1986, i.e.:

$$D = \frac{a_{1986} - a_{2001}}{a_{1986}} \times 100 \, ,$$

With a_j the forest cover in year j. Figure 10 shows that the distance to the city can be used as a proxy for anthropogenic pressure; for both cities, decreasing deforestation rates are observed with increasing distance, as could be expected. Both tendencies are highly statistically significant (logarithmic curve fit; Kisangani: R^2=0.990; Ubundu: R^2=0.857; for both cities $p<0.01$). Deforestation rates are clearly higher for Kisangani, likely due to the higher population density, which forces people to look for forest resources or land at longer distances. Moreover, the high deforestation rates at short distances suggest an expansion of the Kisangani city area of about 5 to 10 km in the period considered. Figure 10 also shows the footprint of the cities, which largely exceeds the city limits. These observations confirm the nucleus model of Forman (1995) and earlier observations of the direct link between anthropogenic pressure, urbanization, road density, and deforestation patterns in the Kisangani region (Bamba et al. 2010a,b).

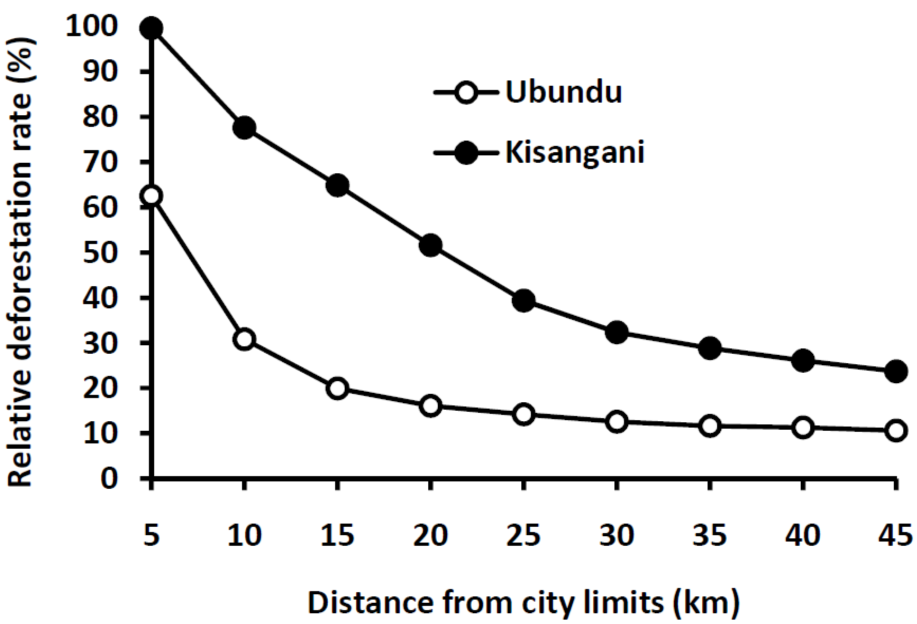

Figure 10. Deforestation rates between 1986 and 2001 in Kisangani and Ubundu (Oriental province, Democratic Republic of the Congo) as a function of the proximity to the city. Each distance corresponds to the outer limit of a concentric ring or buffer zone. The width of each ring was equal to 5 km. The deforested area was expressed as a percentage of the forested area in 1986 to calculate the rate. The higher population density of Kisangani results in higher deforestation rates. The expansion of Kisangani is shown by the high deforestation rates at short distances from the city.

4. CONCLUSIONS

Landscape transformation driven by anthropogenic action leads to habitat loss and to a change of the number of habitat patches. Road system development and urban expansion are the main elements of anthropogenic pattern change in landscapes. These dynamics have profound ecological impacts on biodiversity and ecosystem function, by means of edge effects, connectivity decline, home range reduction, and increased species mortality. This chapter provides some theoretical background and shows field data in order to evidence the direct link between landscape patterns, their dynamics, and their influence on the ecological communities in the landscape.

In the Collines department of Central Benin, deforestation patterns were quantified between 1972 and 2006, based on the analysis of Landsat data. A strong fragmentation of the former open rain forest was observed between 1972 and 1986, followed by attrition of some of the remaining patches between 1986 and 2006. A landscape initially dominated by forest vegetation, woodlands and savannahs is nowadays characterized by mosaics of agricultural fields, fallow lands and savannahs, as a consequence of charcoal production, firewood collection, and cotton production.

The link between edge creation and biodiversity patterns was shown for rodent diversity in the Masako Forest Reserve (Kisangani, Democratic Republic of the Congo). Although many species were found in common between the three habitat types considered (fallow land, edge, secondary forest), which lead to similar presence/absence records, a significant difference was found between the edge habitat and its neighboring sites when species abundances were taken into account. This case study emphasized the uniqueness of the edge habitat and the importance of both components of the diversity concept: richness and abundance distribution.

The impact of road systems on forest habitats was illustrated for the territory of Kambove (Katanga, Democratic Republic of the Congo) by means of the interior-to-edge ratio. Using a range of distances of edge influence, the potential impact of the road network on the forest cover was assessed. Although actually only marginally disturbed, forest cover in Katanga remains threatened as a consequence of the presence of mining concessions in forested areas throughout the province.

In the surroundings of Lubumbashi (Democratic Republic of the Congo), deforestation patterns were compared between four zones situated peripherally to the city. Based on Landsat data from 1984 and 2009, a first order Markov model was applied to simulate forest cover change between 1985 and 2050. A direct relationship was found between the presence of roads, the proximity of neighboring cities, the presence of anthropogenic sites, and the rate of deforestation.

An analysis of deforestation rates between 1986 and 2001 in Kisangani and Ubundu (Democratic Republic of the Congo) based on Landsat data indicated that forests situated closer to the city are more prone to dissapear. Rates are also found higher for Kisangani than for Ubundu, which was explained by the higher population density of the former city, forcing people to look further beyond the city limits for forest resources and land. It was shown that the footprints of the cities largely exceed their limits.

The aforementioned case studies illustrate the increasing anthropogenic pressure on landscapes and their natural resources. Accessibility created through road systems or expressed as the distance to a city, seems to be the trigger for landscape transformation, development and, regretfully, degradation. A combination of high population density and high landscape accessibility leads inevitably to anthropogenic landscape transformation in which natural habitats are lost and in which the remaining habitats are reduced in size and are scattered throughout the landscape, often functioning as isolated systems. The ecological consequences of these dynamics are well known: edge effects, loss of biodiversity, and loss of ecosystem functions. These academic concepts should now be converted in straightforward guidelines to be incorporated in landscape management plans in order to prevent further degradation of the world's precious ecosystems.

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