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Quantifying landscape anthropisation patterns: concepts, methods and limits

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Summary

Since human beings began to use and shape the land, their influence on their environment has kept on growing so that little or no ecosystem in the world is now considered as untouched. This induces pressures on ecosystem health and land scarcity. Africa is of particular concern because it still presents broad undisturbed zones and key ecosystem services, despite being submitted to increasing anthropogenic pressures. Landscape ecology appears suitable for the study of such phenomena, thanks to its space-based integrative nature and geographical level of focus. It studies the impact of spatial pattern transformation — especially heterogeneity and its components — on ecological processes and provides powerful analytical tools of landscape anthropisation.

The main objective of this thesis is to organise the concepts and methods, from landscape ecology and related disciplines, into a consistent logic, to pinpoint missing analytical frameworks for response-oriented anthropisation assessment, and to apply them to African cases to explore the spatial patterns of anthropisation. In order to address landscape anthropisation, we assemble diverse disciplines into a logical network (DPSIR). The new theoretical framework is tested on Lubumbashi (DRC). In order to address spatial patterns, we first evaluate the thermodynamic connection of the term entropy in landscape ecology: spatial heterogeneity, unpredictability and scale influence. Then, based on 20 landscapes, we highlight the complex relationship between spatial heterogeneity and landscape anthropisation. We finally use the modelled relationships to test the anthropogenic origin of the spatial pattern of a land cover class in Lubumbashi.

The main results of this research show that several concepts are used to describe different aspects of anthropisation and that its quantification strongly depends on the reference states. Data formats can be combined into a new assessment method ensuring more precision and comparability, but a good field knowledge is required. As for heterogeneity, the existing definitions of landscape entropy follow the logic of thermodynamics or information theory, that are not compatible. Only unpredictability could be properly interpreted in thermodynamic terms if energy transfer measurements were performed at the appropriate level. The anthropogenic effects on heterogeneity completely diverge depending on the amount of already anthropised surface, on the land cover type (natural or anthropogenic), and on the heterogeneity components.

The aforementioned findings could be adapted to include functional aspects and better address the relationship between spatial pattern and ecological processes. Such integration would help designing response actions that can recommend human activities and spatial patterns that could optimise the use of land to ensure ecological functioning while supporting human development.

Keywords: landscape ecology, anthropisation, heterogeneity, DPSIR framework, land use and land cover change, restoration ecology, land scarcity

Résumé

L'influence croissante de l'homme sur son environnement affecte désormais le monde entier, ce qui induit des pressions sur la santé des écosystèmes et raréfie les ressources en terres. L'Afrique présente un intérêt particulier à cet égard car elle contient de vastes zones quasi vierges et fournit des services écosystémiques importants mais est soumise à des pressions anthropiques croissantes. L'écologie du paysage étudie ces phénomènes de façon intégrative au moyen de sa composante spatiale et de son échelle d'intérêt. Elle étudie l'impact des transformations de la structure spatiale — en particulier l'hétérogénéité et ses différentes composantes — sur les processus écologiques et fournit de puissants outils analytiques de l'anthropisation.

L'objectif de cette thèse est d'organiser les concepts et méthodes de différentes disciplines de façon à mettre en évidence leurs forces et faiblesses pour proposer une nouvelle quantification de l'anthropisation, orientée vers la gestion, et de la tester sur des paysages africains pour examiner la structure spatiale de l'anthropisation. Le DPSIR est utilisé pour assembler les différents concepts. La nouvelle méthodologie est testée sur Lubumbashi (RDC). Ensuite, le lien entre la thermodynamique et l'utilisation du terme entropie en écologie du paysage est examiné. Vingt paysages servent alors à mettre en évidence la complexité de l'impact de l'anthropisation sur l'hétérogénéité du paysage. Enfin, cette modélisation sert à mettre en évidence l'origine anthropique de la structure spatiale d'une classe d'occupation du sol à Lubumbashi.

Les résultats principaux de cette recherche sont que pléthore de termes sont utilisés pour représenter différents aspects de l'anthropisation et que sa quantification dépend de la définition d'états de référence. Cependant, la combinaison de différents formats de données peut aboutir à une nouvelle méthodologie plus précise et adaptable, mais cela nécessite une bonne connaissance de terrain. Les définitions de l'entropie dépendent soit de la thermodynamique soit de la théorie de l'information, qui ne sont pas compatibles. Seule l'imprévisibilité pourrait être interprétée thermodynamiquement, si les mesures de transfert d'énergie étaient effectuées à l'échelle appropriée. L'impact humain sur l'hétérogénéité diverge selon la quantité de surface déjà anthropisée, le type de couverture du sol pris en compte ainsi que les composantes de l'hétérogénéité mesurées.

Ces découvertes peuvent être adaptées pour intégrer des aspects fonctionnels de la structure spatiale et mieux cerner le lien entre celle-ci et le fonctionnement écologique, ce qui permettrait de proposer des activités humaines et des structures spatiales qui optimiseraient l'utilisation des ressources en sol pour assurer tant le fonctionnement écologique que le développement humain.

Mots-clés: écologie du paysage, anthropisation, hétérogénéité, cadre d'analyse FPEIR, changement d'occupation et d'utilisation du sol, écologie de la restauration, rareté des terres

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Part I

Introduction

CHAPTER 1

General context

1.1 Anthropisation and heterogeneity: complex interactions, systemic impacts, yet disjointed framework

Since human beings began to use and shape the land, their influence on their environment has kept on growing so that currently, little or no ecosystem in the world is considered as untouched (Sanderson et al., 2002; Ellis and Ramankutty, 2008). For this reason, most landscapes are now referred to as biocultural landscapes: generated by both natural and anthropogenic processes (Bogaert et al., 2014). Human activities have worldwide consequences on landscape structure as well as ecosystem functioning, using techniques that distinguish them from other species (Mackey et al., 1998; Lecomte and Millet, 2005; Mazoyer and Roudart, 2006; Ellis and Ramankutty, 2008; Bogaert et al., 2014). This phenomenon is referred to as anthropisation, anthropogenic effect, as well as many other terms (Vranken et al., in preparation-a). Currently, in the majority of cases, especially in southern countries, it consists of urban sprawl and peri-urbanisation (Andre et al., 2014), deforestation (Bamba et al., 2008; Barima et al., 2011) or agricultural expansion (Bamba et al., 2008; Bogaert et al., in preparation). This induces pressures on ecosystem health and land scarcity in terms of resources exploitation to support human life (Bogaert et al., in preparation).

The ecological impacts of anthropisation, either local or global, are addressed by various disciplines such as botany, population ecology, conservation management, restoration ecology, landscape ecology and geography (Jalas, 1955; O'Neill et al., 1988; Grabherr et al., 1992; Solon, 1995; Peterken, 1996; Lambin and Geist, 2006), while the response to them depends strongly on environmental policy making (Ness et al., 2010). Africa is of particular concern regarding anthropisation because this area still presents broad undisturbed zones and high biodiversity, environmental resources as well as key ecosystem services, even at worldwide scale, despite being submitted to continuously increasing anthropogenic pressures (Groombridge et al., 2002). In order to preserve the capacity of our geosphere to provide adequate resources to support human life (ecosystem services, see for example Fisher et al. (2009)), anthropisation monitoring, as well as environmental impact assessments, therefore appear necessary prior to address responses to such disturbances.

Landscape ecology is suitable for the study of such phenomena and their ecological consequences, thanks to its space-based integrative nature and geographical level of focus. It studies the impact of land transformation on ecological processes that drive the distribution and abundance of organisms and provides powerful analytical tools regarding the study of landscape anthropisation (Burel and Baudry, 2003; Fahrig, 2005; Bogaert et al., 2014). In landscape ecology, such relationships are mainly exploited in the context of pattern heterogeneity measurement, which strongly determines abiotic conditions such as wind and shade (Okin, 2008), the distribution of living organisms (Cale and Hobbs, 1994; Li and Reynolds, 1995; Fahrig, 2005) and biodiversity (Tews et al., 2004).

Although landscape ecology addresses this issue by studying the dynamics of spatial patterns of landscapes and ecosystems within, it rarely focuses on the origin of anthropogenic change, or mixes this up with its ecological impacts (Vranken et al., in preparation-a). In order to deepen our understanding of the causal interactions operating within the context of landscape anthropisation and to integrate it in decision making, the DPSIR (Driver-Pressure-State-Impact) framework (Ness et al., 2010) will be referred to throughout this thesis.

In this thesis, we merge DPSIR and landscape ecology frameworks in order to achieve a comprehensive, logic and action-oriented analysis framework to address the issue of landscape anthropisation and its consequences on ecological processes. As such monitoring activities frame the responses to give to anthropogenic landscape change, these concepts should help addressing the socio-ecological, yet inter-dependent challenge of preserving ecological functioning while supporting human life (Kelble et al., 2013).

1.2 Aims and scopes: Organise and develop concepts and methods to identify spatial patterns

The main objective of this thesis is to organise the concepts and methods associated with landscape anthropisation and its quantification into a consistent logic, pinpoint missing concepts and analytical frameworks for response-oriented anthropisation assessment, and to apply them to African cases in order to explore the spatial patterns of anthropisation. To serve this purpose, two main scientific questions were addressed (Sections 1.2.1 and 1.2.2) , associated with sub-objectives (numbered list) and publications.

1.2.1 What is landscape anthropisation and how can it be quantified?

1. Define landscape anthropisation and associated concepts, evaluate its quantification methods and propose an upgrade (Vranken et al., in preparation-a);

Each discipline addressing the issue of anthropogenic pressure or impact focuses on its own scale, entities and processes. We want to explore these different disciplines to highlight the origins, mechanisms and consequences of landscape anthropisation on ecological processes. Furthermore, responses to such disturbances depend on how quantification of anthropogenic effects is performed, which in turn relies on anthropisation definition. As many definitions exist, many quantification methods exist as well. Therefore, the existing methods lack comparability for a global assessments of the anthropisation level. We want to propose guidelines for a new methodological framework that could be as comprehensive and exportable as possible.

2. Test the application of a new anthropisation metric to an African landscape (André et al., in revision);

As a first step for the application of our proposed guidelines, we want to test them on a landscape undergoing strong anthropisation dynamics and where data acquisition is particularly difficult. Our study case is Lubumbashi, Katanga, Democratic Republic of the Congo.

1.2.2 What is pattern heterogeneity? How is it quantified? In this quantification, how is pattern heterogeneity influenced by anthropisation?

3. Define the various existing landscape entropy interpretations and characterise their links with thermodynamics (Vranken et al., 2015);

Entropy is frequently referred to when measuring landscape spatio-temporal heterogeneity. Some authors state a link between landscape spatio-temporal dynamics and thermodynamics. Such a link would enhance our understanding of what drives landscape dynamics and the consequences of human activities on the functioning of the whole geosphere, with perspectives in the understanding of global warming or sustainable energy production and consumption. However, the current state of the art is far from complete. We want to clarify what landscape entropy refers to and what remains to be done.

4. Characterise the link between anthropisation and spatial heterogeneity based on 20 African landscapes (Vranken et al., in preparation-b);

The relationship between landscape anthropisation and heterogeneity has not been fully addressed yet. Indeed, the current descriptions focus either on the impact of heterogeneity on species distribution or on the anthropogenic impact on species distribution, without explicit connection with heterogeneity. In addition, as tropical Africa is of major concern regarding anthropogenic impacts on its biomass and biodiversity, it represents an interesting case study. We want to show that anthropisation has multiple and complex impacts on pattern heterogeneity and decompose these impacts.

5. Identify anthropogenic disturbances through the study of spatial patterns (Vranken et al., 2013).

Once landscape anthropisation and its links with heterogeneity have been characterised, it should be possible to use these links as a calibration model to identify the natural or anthropogenic origin of definite pattern dynamics. In the present case study (Lubumbashi), pattern heterogeneity of preserved and disturbed zones were compared to highlight the anthropogenic origin (metallurgy) of one specific land cover (bare soils).

1.3 Outline

In order to simplify the references to the different original contributions, short titles will be used instead of their complete titles. Table 1.1 shows the correspondence between article title and their short titles.

Table 1.1 – *Correspondence between complete titles of the original contribution of this dissertation and their associated short titles.*

Short title	Complete title
Habitat loss	Ecological impact of habitat loss on African landscapes and diversity
Anthropisation	Spatially explicit quantification of anthropogenic landscape change. Towards a new methodological framework
Entropy	A review on the use of entropy in landscape ecology: heterogeneity, unpredictability, scale dependence and their links with thermodynamics
Anthropisation Katanga	Quantification of anthropogenic effects in the landscape of Lubumbashi
Heterogeneity - Anthropisation	The importance of studying anthropogenic effects on heterogeneity
Footprint	The spatial footprint of the non-ferrous mining industry in Lubumbashi

The chapters and corresponding articles contained in this dissertation can be organised following three different readings: technical, thematic and logical. The technical reading divides the articles in two groups: conceptual research, on the one hand, and applications to real landscapes, on the other hand. This is the reading presented in the table of contents of this thesis.

The thematic reading organises the articles around the two main focuses of this thesis: anthropisation and heterogeneity, plus a third illustrative focus that presents the approach of landscape ecology and its link with ecology through the study of natural habitat loss (in the introduction). The scientific questions presented in Sections 1.2.1 and 1.2.2 are structured according to this thematic reading. It also structures the original contributions in two levels of importance: the major level (large font chapters in Figure 1.1) represents the main core of this thesis, while the minor level serves as illustrative case studies on more specific aspects of the thematic developed in the major contributions.

As for the logical reading, it corresponds to the DPSIR framework, further detailed in Section 1.5.2, page 15. Those three readings are presented in Fig. 1.1. In order to ease the understanding of the logical flow between each contribution, their position in the technical, thematic and logic readings are reminded at the beginning of each corresponding chapter.

To go on with the structure of this dissertation, the study areas used in this dissertation as well as a brief theoretical introduction are presented before the original contributions

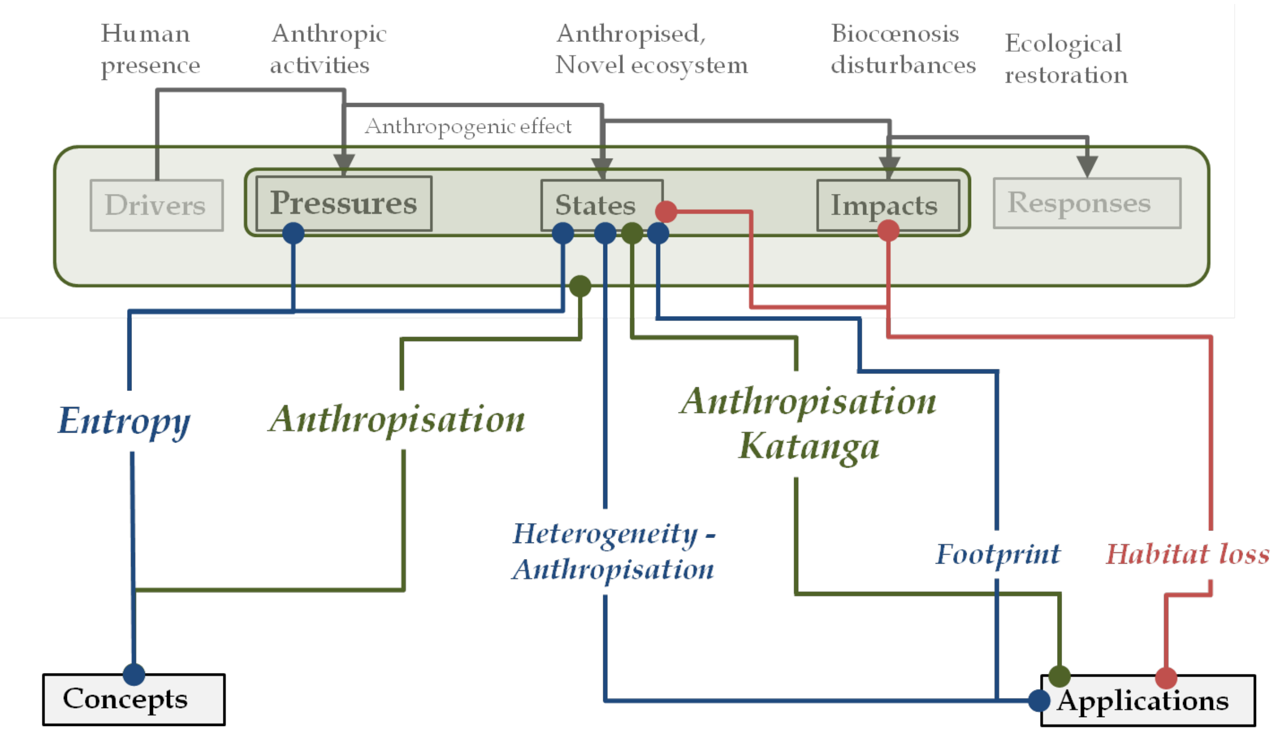

Figure 1.1 – *Thesis mind map: logic organisation of the thesis chapters according to a concepts versus application structure and the DPSIR framework applied to human influence on landscapes.*

(section 1.5) in order to become familiar with the discipline according to which the different studies were performed and the cause-consequence context of each study. Then, each original contribution will be presented as a chapter of this thesis.

First, to illustrate how anthropisation is addressed in landscape ecology, an introductory example is presented : *Habitat loss* (Chapter 2).

The main body of this dissertation is divided in two parts. The first part, *Conceptual research and developments* (Part II), lies with the epistemological study of landscape ecology and presents the conceptual and methodological reviews and developments provided during the past four years of doctoral study. The second part, *Application to Spatial pattern assessment* (Part III), does not focus on concepts and methods, but their application to real landscapes.

In the conceptual part, the first article, *Anthropisation*, Chapter 3, represents the departure point of any further development and application. It aims to assemble diverse disciplines related to landscape ecology and to connect all they can tell about human-driven landscape change into a logical network (DPSIR), the same as used throughout this thesis. It also points out weaknesses of anthropogenic effect assessment and proposes a new theoretical framework to overcome them. This article presents all the steps of the logical framework of this thesis, but focuses more on the pressures and state.

The second conceptual article, *Entropy* (p.53), evaluates the thermodynamic connection of popular topics in landscape ecology referred to with the term entropy: spatial heterogeneity, unpredictability of pattern dynamics and scale dependence on pattern assessment. In the DPSIR framework, this article addresses ecological and human processes in the form of pressures causing the anthropised state, the main focus of landscape ecology.

The first article of the *Application* part of this dissertation, *Anthropisation Katanga*, is a first attempt to apply the new quantification methodology proposed in the first part (*Anthropisation*, Chapter 3) to Lubumbashi, D.R.C. Inside the DPSIR framework, it focuses on landscape state, but also pressures and impacts.

The second article, *Heterogeneity - Anthropisation*, takes place in the state step of the DPSIR framework, that quantifies patterns, both compositional and configurational. Based on the spatial structure of 20 landscapes, it highlights the complex relationship between the different components of spatial heterogeneity and landscape anthropisation.

The last article, *Footprint* focuses again on Lubumbashi. This time, it uses spatial pattern assessment to test the hypothesis that it results from anthropogenic pressures. It addresses the pressures, state and impacts steps of the logical network, but what is actually measured is the state of the landscape: its composition and configuration.

The discussion (part IV) synthesises the main findings of this thesis related to the quantification of landscape anthropisation. It also points out the shortcomings of the methodologies and suggests ways to address these. Finally the conceptual developments open a debate on responses to give to anthropogenic landscapes. This debate articulates around the double challenge of sustainable development: resource self-sufficiency for human development and ecosystem health management.

This thesis contains a large number of technical terms that are not directly defined in the articles because the necessary references for most technical terms are provided. However, to ease the understanding of the content of this thesis for non-specialists, a glossary is provided at the end of this dissertation (p. 133).

The author's contributions that have been published or accepted but are not considered as part of the core of this thesis are presented in the list of publications. The abstract of one of the author's contributions (as a co-author) that is relevant to the topic of this thesis is given in the appendix.

1.4 Team project in Tropical Africa

The present thesis takes place in a team that studies anthropisation cases in various African countries. The team was composed of Ph. D. students in various disciplines such as pedology, geography, botany, zoology, ecology, coming from and studying Tropical African countries such as Democratic Republic of the Congo, Ivory Coast, Benin, Niger, Mali , Madagascar and Burundi. The team was divided in two task groups : 1) Land use and land cover change assessment using remote sensing and Geographic Information Systems (Mama, 2013a; Barima et al., 2010; Bamba et al., 2011) and 2) anthropogenic impact studies on the diversity of plant and animal communities (Iyongo et al., 2009; Diallo et al., 2011; Diouf, 2012). All these theses had the study of anthropogenic disturbances in common, and the studied landscapes are characterised by strong anthropogenic dynamics or high biodiversity that were still relatively untouched at the beginning of the last century. However, as each member of the team initially focused on his own study case, the research lacked genericity and transversal analysis. The role of the present thesis is to merge the various approaches and results here developed into an integrative study to compare the different cases and find unifying anthropisation principles. Such a meta-analysis falls beyond the frame of each individual thesis and has to focus on a more general scale of study. It also requires a higher level of abstraction, not only to create a typology based on the various study cases, but also in order to develop generic concepts and methods to assess anthropisation and its impacts on landscape structure.

Two studies in the present thesis, Vranken et al. (2014) and André et al. (in revision), were based on data acquired directly for the purpose of the study and focus on Lubumbashi, Katanga (D.R.C.). The rest of the research is based on the data previously collected by the rest of the team: the study cases are presented in order to illustrate those concepts and methods. 11 sites from Democratic Republic of the Congo (D.R.C.), Benin and Ivory Coast were used (Fig. 1.2). The *Heterogeneity-Anthropisation* chapter, Vranken et al. (in preparation-b), uses all the 11 sites, dispatched in 20 landscapes, 6 and 9 of which are threefold (3 sites in Benin) and twofold (Lubumbashi and 2 sites in Ivory coast) time series, respectively. All the data used in this third study were acquired for the purpose of previous studies (Bamba et al., 2008; Djibu Kabulu et al., 2008; Munyemba et al., 2008; Bamba et al., 2010b; Barima et al., 2010; 2011; Mama, 2013a; Vranken et al., 2011; 2013). The major pattern dynamics observed in these study sites are deforestation, (sub)urbanisation and savanisation (Vranken et al., in preparation-a;i). The team also generated publications related to each other's theses, which explains that some of the author's publications are not present in this thesis (see list of publications, p. xviii). Considering the large amount of study zones, the time elapsed since data collection, the fact that data quality had already been controlled within the frame of the other members' theses and publications, as well as the conceptual and methodological focus of the present thesis, choice was made not to add new field work to control nor enrich the database.

Indeed, the study zones are only considered here to test the methods: the focus was not the study zones *per se*, unlike the works from which the data were obtained (Barima, 2010; Munyemba, 2010; Bamba et al., 2008; Mama, 2013b). Even in this case, field knowledge is of definite importance, but such importance then relies on the ability to check the likeliness of the results: the study cases are not exploited for their intrinsic value. In this context, indirect field knowledge and data should be able to provide relevant information on anthropisation and heterogeneity states, the two main focuses of this thesis. Considering the concepts and methods applied, such information is provided by the image classifications and metadata.

1.4.1 Katanga, D.R.C.

The Katanga province is a woodland area that is said to contain dry evergreen forest deforested by local populations before the colonial period (Malaisse, 1997). Its geology, characterised by the presence of the *Copper Belt*, strongly influenced anthropogenic landscape changes in the area, where many mining and metal processing sites, then towns developed. Those activities locally generate atmospheric soil deposits and water pollution (Munyemba et al., 2008; Vranken et al., 2013). However, on a larger scale, the major degradations are linked to urban development, partly induced by the economic activity in the area (Vranken et al., 2014).

The city Lubumbashi, in particular, faces massive demographic explosion and urban growth, to such extent that, given the insufficiency of infrastructures and public services like water and energy distribution or roads, slums develop around the town at an unprecedented rate. Consequently, the city is not food self-sufficient and the poorest local populations depend on charcoal production and sale to fulfil their basic needs (Vranken et al., 2014; André et al., in revision). Subsequent deforestation provokes increasing savanisation in the area, facilitated by frequent anthropogenic fires.

Lubumbashi is a reference landscape of study in the *Habitat loss, Anthropisation Katanga, Heterogeneity - Anthropisation* and *Footprint* contributions. In *Heterogeneity-Anthropisation* (Chapter 6), the landscape of Lubumbashi is observed at two dates: 1984 and 2009.

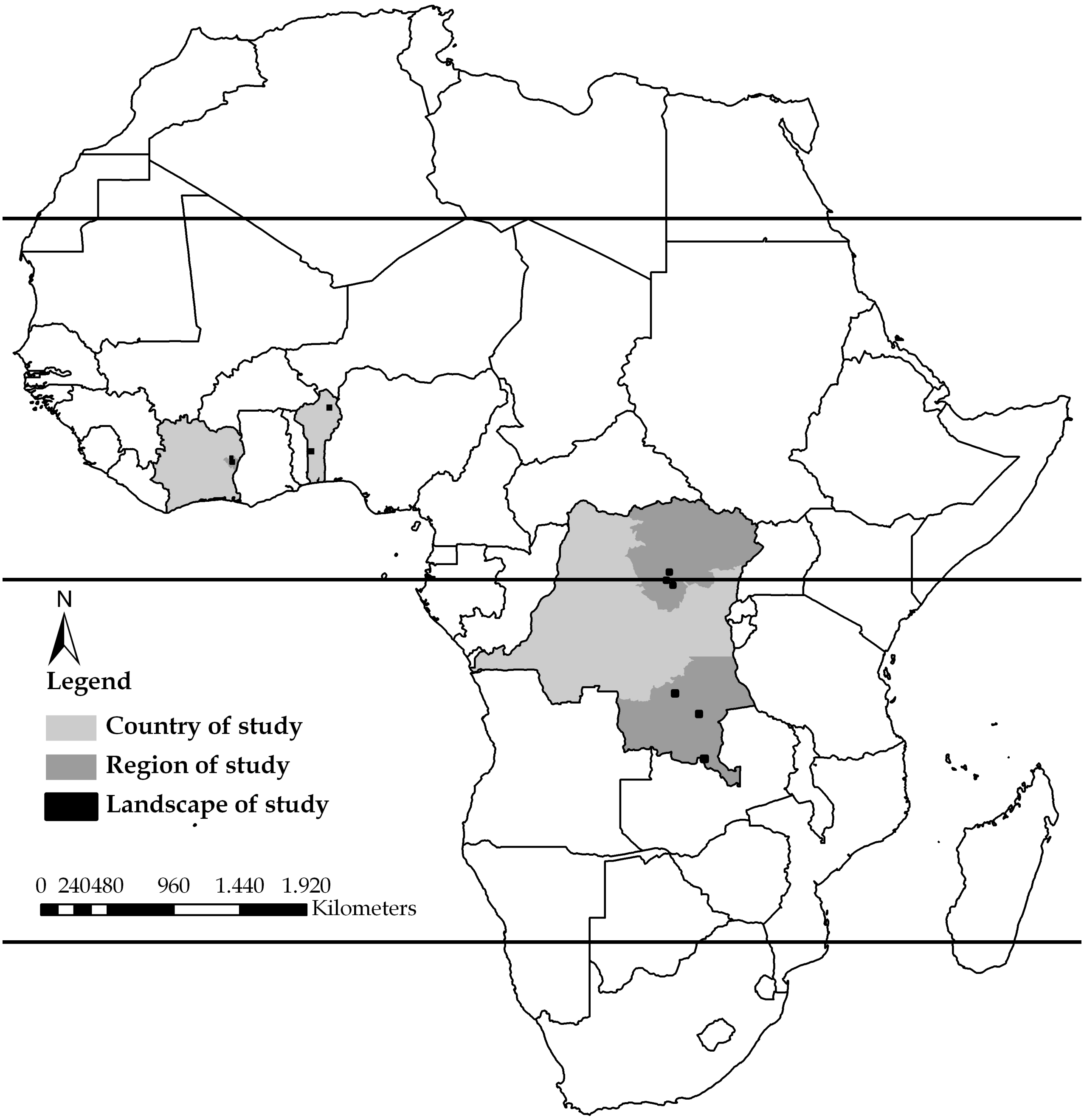

Figure 1.2 – *Location of the 11 sites, in their regions and countries, corresponding to the African landscapes studied in the present thesis. The three horizontal lines represent the equator and the tropics.*

1.4.2 Benin

Benin is a country of West Africa with a broad latitudinal range, from Southward of Niger to the shore on the Gulf of Guinea. Its climate and vegetation follow a gradient from the seashore in the south, with mangroves and rainforest, to woodland and savannah in the North. The main anthropogenic pressures are pastoralism (especially in the northern part); forest exploitation for charcoal production, tree plantation or slash and burn agriculture; and rural outmigration, inducing urban extension, especially in the southern part: in the area of Cotonou, the economic centre of the country (the administrative capital city being a neighbouring city called Porto-Novo)(Mama, 2013b).

In the South, nearly all the mangroves were deforested for charcoal production and replaced with commercial tree and fruit plantations (such as Teak, Acacia, oil palm, coconut palm) around the main urban area of Cotonou, that is also a major space consuming city with rapid extension (Mama, 2013b). The central and northern parts of Benin, that are less urbanised and more characterised by rural dynamics, undergo severe savanisation due to deforestation for agriculture and pastoralism (Heubach et al., 2011; Mama, 2013a). Though forest are submitted to important pressures, they also represent an important income source for the poorest rural populations, with the exploitation of non-timber forest products (Heubach et al., 2011). The situation of forests as well as households is then increasingly critical.

North, center and south of Benin are reference landscapes of study in the *Habitat loss* and *Heterogeneity - Anthropisation* contributions. They are all three observed at three dates: 1972, 1986 and 2006, at different stages of the aforementioned observed dynamics.

1.4.3 Oriental province, D.R.C.

This region in the north-eastern part of the D.R.C. is situated in the rainforest area of the Congo basin, where relatively undisturbed areas can still be found. The main city in the area is Kisangani, along the Congo river. It is surrounded by a halo of croplands, secondary forest and different stages of forest degradation due to local slash and burn agriculture with fallow and charcoal production (Bamba et al., 2010b). Smaller towns in the area, like Ubundu, face the same problem on a smaller extent. The main pressures in this region are logging, extension of the road network and subsequent rural outmigration and agricultural expansion (Zhang et al., 2006a; Bamba et al., 2010b).

Where no town can be found, roads traced for logging favour the implantation of small villages, so dispersed traces of forest degradation can be found due to slash and burn agriculture. A general shortening of the fallow period is observed, along with an increasing extent of forest degradation (Zhang et al., 2006a; Bamba et al., 2010b).

Landscapes in the Oriental province of D.R.C. are studied in the *Habitat loss* and *Heterogeneity - Anthropisation* contributions. In the latter, they are divided in three zones: one centred on Kisangani, one on Ubundu, and one in a nearly undisturbed forested area of the Congo basin. These three landscapes are observed as they were in 2001.

1.4.4 Region of Tanda, Ivory Coast

The department of Tanda is situated in the east of Ivory Coast, in a transition zone between rainforest in the south and savannah in the north (Barima et al., 2010). In the western part of the region, the climate seems to favour ecological succession towards forest (though this increase is almost stabilised by human activities), but everywhere else, climatic conditions as well as economic activities accelerate savanisation (Goetze et al., 2006; Barima, 2010). Indeed, this subhumid tropical region faces precipitation decline, partly due to deforestation (Myers, 1988).

The main deforestation activities in the area are linked with food crops and commercial plantations, mainly industrial (Barima et al., 2010). Climate change even forced the industry to change their crops, from cocoa and coffee to teak and cashew tree (Barima, 2010). The area is essentially rural with a tendency to rural outmigration towards the south, increasing plantation areas and growing economic activity (Barima, 2010).

Two landscapes in this region were observed in the *Heterogeneity - Anthropisation* chapter. One of them is situated in the north-western part of the region and the other one in the south-eastern part. The two landscapes are observed in both 1986 and 2002.

1.5 Theoretical references

1.5.1 Landscape Ecology

Landscape ecology is at the crossroads between ecology and geography (Troll, 1939; 1971). It aims at combining spatial structure, the scope of geography, and ecosystem processes, the scope of ecology (Troll, 1971; Burel and Baudry, 2003). Its approach generally studies landscape spatial patterns following an object-oriented approach (the patch-corridor-matrix model), where each object is an ecosystem. This allows to infer the impact of the spatial structure of those objects on their ecological processes (pattern / process paradigm) (Turner, 1989), which determines in turn the relative abundances and distributions of organisms (Fahrig, 2005).

1.5.1.1 The Landscape level: main focus of Landscape Ecology

Landscape ecology focuses on the hierarchical level of landscape, from where it also approaches the direct higher and lower organisational levels (Green and Sadedin, 2005). The immediately higher level, the system environment or surroundings, is the region; this level represents the outer constraints encountered by the landscape (Fig. 1.3). The immediately lower level, the components or holons (sub-systems) of the landscape, are the ecosystems (Wu and Marceau, 2002; Li et al., 2004). According to complexity theory, the processes regulating the system functioning are specific to each level (Wu and Marceau, 2002; Li et al., 2004; Green and Sadedin, 2005). This explains the importance of choosing the appropriate scale when studying spatial structure in relation to ecological processes.

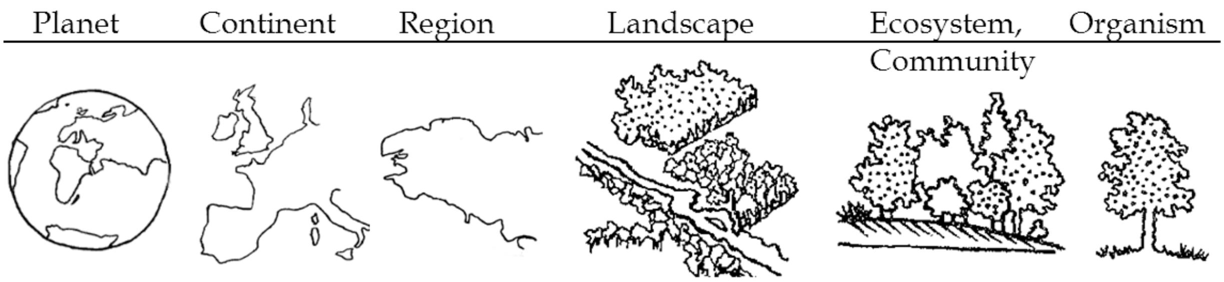

Figure 1.3 – *Main organisational levels studied in Landscape ecology and related disciplines. Landscape Ecology focuses on the landscape, which involves also studying the ecosystems within and the containing region. (Adapted from Burel and Baudry (2003))*

Landscape ecology defines the landscape as a heterogeneous land area composed of a cluster of interacting ecosystems (Forman and Godron, 1986a). This ecology-oriented definition is different from the one in geography and land planning, which distinguishes the physical support (the territory) from the observed image (the landscape), in a human-centred approach (Lynch, 1960; Collot, 1986; Froment, 1987), while the definition from landscape ecology specifies that the landscape exists regardless of perception (Burel and Baudry, 2003). However, the latest landscape ecology research also includes perception of the landscape, but more specifically from the point of view of the species or group of species studied (Cale and Hobbs, 1994; McIntyre and Hobbs, 1999; Tischendorf and Fahrig, 2000). For example, in a field matrix with vegetation remnants, habitat has different compositional and configurational characteristics from the point of view of birds (Cale and Hobbs, 1994) compared to insects (Petit and Burel, 1998): trees, hedgerows, their sizes and proximity to similar habitat have different meanings for these two groups of species. Functional aspects are then integrated to spatial pattern studies, like animal mobility to evaluate habitat connectivity (Tischendorf and Fahrig, 2000; Lindenmayer and Fischer, 2006). Such integration deepens the study of the link between spatial patterns and ecological processes.

1.5.1.2 The patch-corridor-matrix model: an object-oriented approach

The patch-corridor-matrix model is directly linked to the landscape definition in landscape ecology (Fig. 1.4). As the landscape is considered as composed of a multitude of interacting ecosystems, those ecosystems are considered as distinct objects: patches (Forman, 1995). A patch is the elementary unit of a landscape, an ecosystem different from its surroundings (Forman, 1995). The patches are included in the matrix, a larger patch in which the other ones are encompassed and that dominates the landscape. Corridors are defined by their geometric and functional properties as well as their relative location: they are elongated patches that help connecting two habitat patches of similar kind by enhancing species mobility through the matrix. Those corridors can physically connect those patches or act as stepping stones, depending on the mobility of the studied species and the opacity of the matrix (Tischendorf and Fahrig, 2000). However, the connectivity is generally better if the patches are physically connected (Tischendorf and Fahrig, 2000). When there is no matrix, no specific dominance of a certain patch in the landscape, the different patches composing it form a mosaic (Forman, 1995).

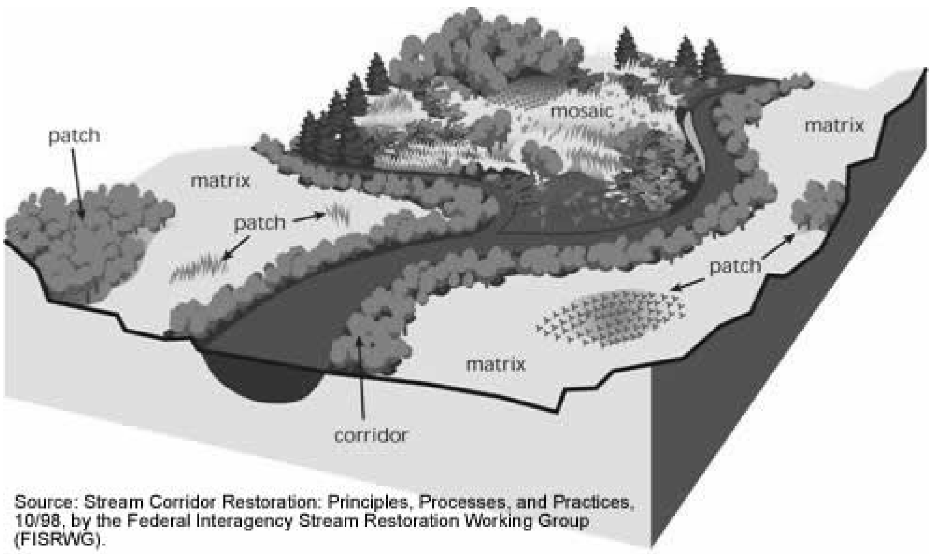

Figure 1.4 – *Patches and corridors in a matrix, mosaic.*

If the patch-corridor matrix model is an object-oriented approach, other data-formatting approaches are also used in landscape ecology. For example, point data analyses measuring biophysical gradients are frequently used to study edge effect. Edge effect is linked to the notion of ecotone in ecology, describing the transition zone, more or less abrupt and wide, between two adjacent ecosystems, such as forest and field. The patch-corridor matrix model helps highlighting the area of the edge zone depending on its width, but also patch shape. If a patch has a complex of elongated shape, its core area will be smaller than for a simple shape of comparable area. This has consequences on species composition, as further detailed in Vranken et al. (2011).

Following the categorical mapping format, patches can be classified according to the ecosystem type, with a thematic resolution (number and type of classes) adapted to the studied species (Gustafson, 1998; Bailey et al., 2007). The number and relative areal abundance of classes in the landscape represent the compositional properties of the landscape spatial pattern. Patch shape, size and spatial arrangement represent the configurational part of the landscape spatial pattern.

1.5.1.3 The Pattern / Process Paradigm: the foundation of landscape ecology

The pattern / process paradigm is the central hypothesis of landscape ecology. It states that landscape spatial patterns or structure (composition and configuration) are connected in causal relationships with the ecosystem processes occurring therein (Turner, 1989). That powerful connection allows to study landscape patterns to infer the underlying ecological functioning that drives the spatial or temporal distribution of organisms of different species (Fahrig and Nuttle, 2005). This has particularly strong implications in restoration ecology and related sciences, when studying the impact of landscape anthropisation, as changing landscape composition as well as configuration, on ecological processes and infer ecosystem health. This is also of great interest in land planning, for example to design networks of ecosystem services in cities or peri-urban areas, like green spaces to mitigate heat islands or nest plant and animal species (Sandström et al., 2006; Larondelle and Haase, 2013; van der Walt et al., 2014). In landscape ecology, the effect of pattern on processes is more often studied than the opposite (Vranken et al., 2015).

1.5.1.4 Heterogeneity in landscape spatio-temporal structure: major concern

Mostly, heterogeneity is what is measured when studying landscape spatial patterns. Pattern heterogeneity can be either compositional or configurational (Turner et al., 2001). Temporal heterogeneity is addressed when studying the dynamics of landscape structure change. Landscape heterogeneity has strong impacts on species distribution, abundances and population dynamics.

In this thesis, attention is focused on heterogeneity for two reasons. First, because of its postulated impacts on ecological processes. Second, because human activities have a strong impact on landscape heterogeneity. If the pattern / process paradigm holds, then the study of the influence of landscape anthropisation on the dynamics of its heterogeneity appears essential for the understanding of the anthropogenic impact on ecological processes (Vranken et al., in preparation-b).

1.5.1.5 Landscape pattern metrics: tools to assess spatial patterns

Based on the pattern / process paradigm, landscape structure, for example spatial heterogeneity, is quantified using pattern metrics. These metrics can also be used to study the dynamics of the landscape structure (Zaccarelli et al., 2013; Vranken et al., 2015).Those are mostly based on the object-oriented patch-corridor-matrix model and are in this case more specific to landscape ecology in itself (Gustafson, 1998; O'neill et al., 1999). Such metrics can be simple measures or composite integration of multiple parameters, generally based on patch or class properties such as their number, area and perimeter. Examples of spatial pattern metrics to quantify spatial heterogeneity are provided in the *Entropy* chapter (Chapter 4). Examples of metrics used to quantify anthropisation are provided in the *Anthropisation* chapter (Chapter 3).

Stretched value analysis is also used in Landscape Ecology (Gustafson, 1998), based on raster data (Wiens et al., 1993; Gustafson, 1998; Vranken et al., in preparation-a). In that case spatial distribution metrics and modelling methodologies are similar to those used in Ecology, such as Clark and Evans's nearest neighbour distance or the MaxEnt spatial distribution modelling method (Clark and Evans, 1954; Harte, 2011). More information about data format and how the data are processed depending on the format are provided in the *Anthropisation* chapter (Chapter 3).

1.5.2 DPSIR: a comprehensive, action-oriented environmental assessment framework

At first, the Driver Pressure State Impact Response framework was used by the European Environmental Agency to classify environmental indicators and analyse environmental problems regarding the relationship between the ecological and human dynamics in a comprehensive and transdisciplinary analysis (Smeets and Weterings, 1999; Ness et al., 2010). This sustainable policy-oriented analysis framework is increasingly used in scientific articles. It integrates social, economic, political and environmental contexts and presents them in the form of a causality chain to highlight which action could be undertaken to tackle them (Smeets and Weterings, 1999).

According to this system analysis view, the Drivers (distant causes, demographic, social and economic development) exert Pressures on the environment, which modifies the State of the environment, such as the provision of adequate conditions for health, resources availability and biodiversity (Smeets and Weterings, 1999). This leads to Impacts on human and ecosystem health that may elicit a societal Response to address those problems (Smeets and Weterings, 1999). This response can be directed towards any step of the causal chain, though addressing distant causes such as economic context is often more difficult to achieve (Vranken et al., in preparation-a).

Fig.1.5 combines the pattern / process paradigm and the DPSIR framework regarding landscape anthropisation, its causes and consequences. The ultimate driver, or the underlying cause of anthropogenic effect on landscapes is the presence of humans, considering their density, but also their lifestyle (production and consumption modes)(Lambin and Geist, 2006). Through activities performed to support human life and lifestyle, they generate environmental pressures, the proximal causes of anthropisation (Vranken et al., in preparation-a). Those pressures influence ecological processes and modify the state of the landscape in its composition as well as its configuration (Sanderson et al., 2002; Lambin and Geist, 2006). This influence is called anthropogenic effect and the landscape in this state is considered as anthropised (Bogaert et al., 2014). States are generally observable through landscape patterns, while pressures act more directly on underlying ecological processes. The anthropised state represents the main focus of landscape ecology regarding human impact on landscapes. That landscape structural change has in turn impacts on ecological processes, disturbing local biocœnosis (Naiman et al., 1988).

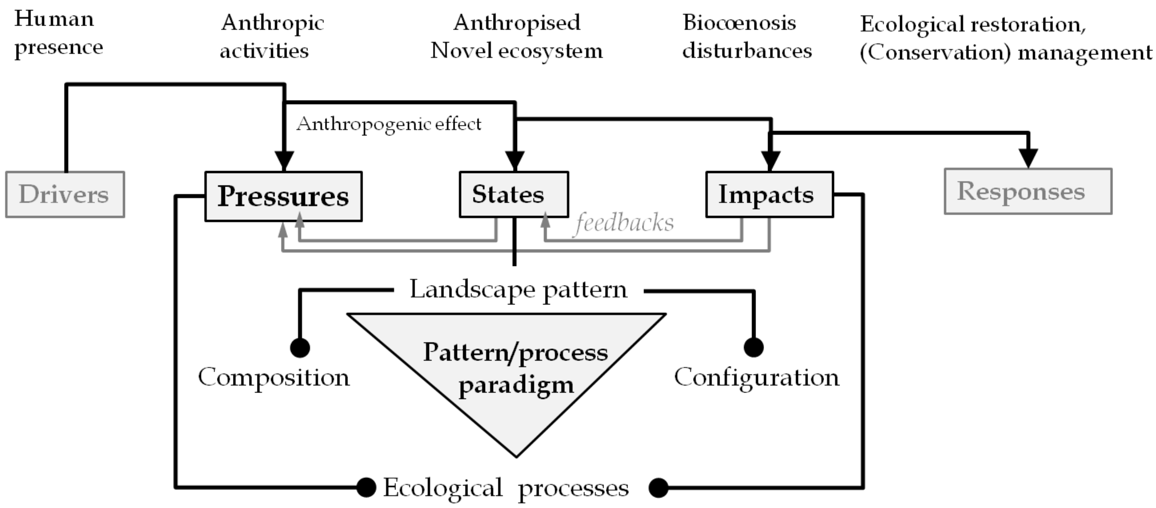

Figure 1.5 – *The issue of human influence on landscapes seen through the DPSIR framework and the pattern / process paradigm. This is further developed in Chapter 3.*

CHAPTER 2

Ecological impact of habitat loss on African landscapes and diversity

Authors: Isabelle Vranken, Jean-Pierre Djibu Kabulu, François Munyemba Kankumbi, Adi Mama, Léon Iyongo Waya Mongo, Issouf Bamba, Mohamed Laghmouch, Jan Bogaert

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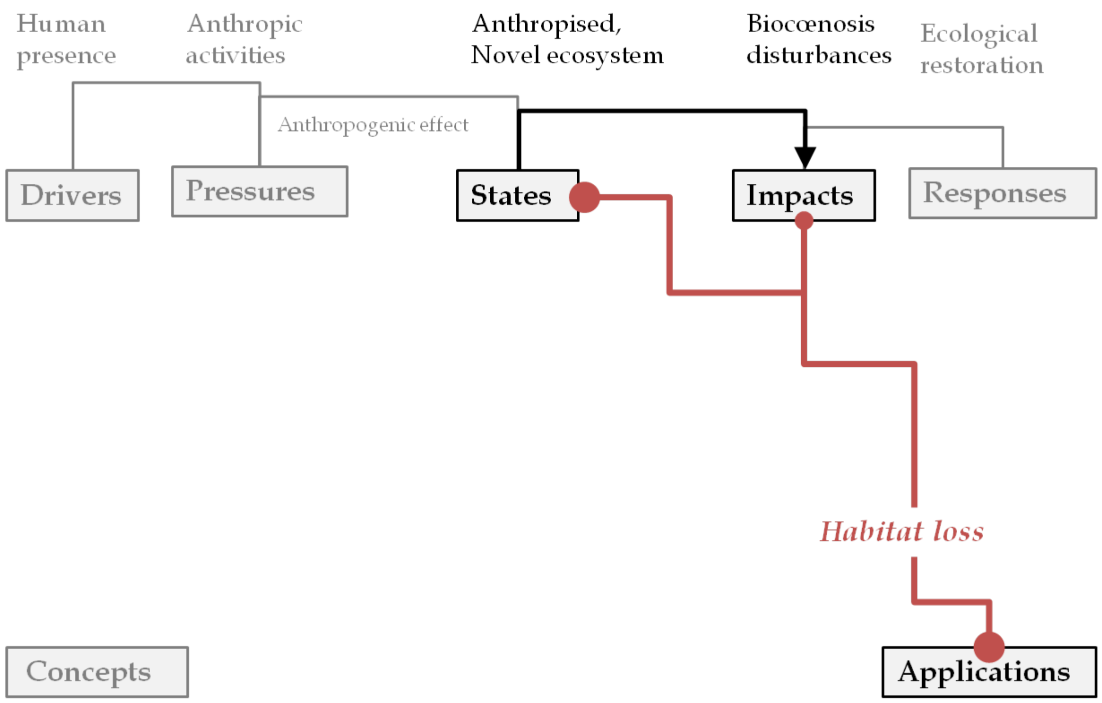

Figure 2.1 – *Location of the habitat loss chapter (red theme) in the thesis mind map. Technical reading: applications, thematic reading: habitat loss (introductive theme), logic reading: states and impacts. This chapter is an illustrative example of how anthropisation can be addressed in landscape ecology and its importance for ecological functioning.*

2.1 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.

2.2 Area change as a key element of landscape transformation

2.2.1 Spatial patterns 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 and Godron, 1986a). 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; Noon and Dale, 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 and 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., 2011). 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 and Mouton, 2009; Bogaert et al., 2011). For both concepts, a series of metrics has been developed. Due to their interrelation (Noon and Dale, 2002), compositional change also implies configuration change and vice versa.

2.2.2 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 and Forman, 1987; Collinge, 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 are 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 2.2).

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., 2011), 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., 2011): 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., 2011).

2.2.3 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

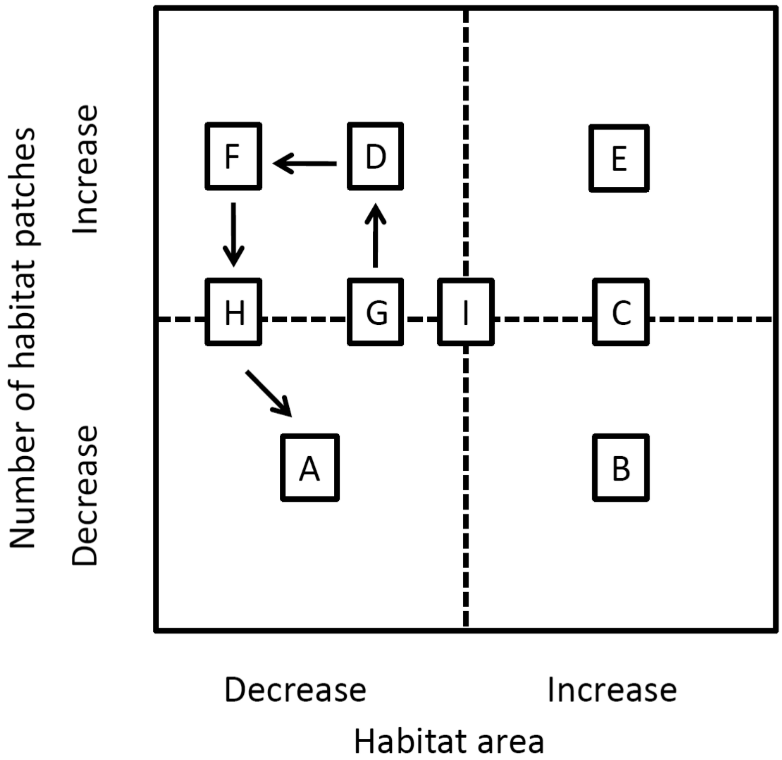

Figure 2.2 – *Landscape transformation processes and their impact on habitat area and on the number of habitat patches. (A) attrition, (B) aggregation, (C) enlargement, (D) dissection, (E) creation, (F) fragmentation, (G) perforation, (H) shrinkage, (I) shift/deformation. To separate shift and deformation, 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).*

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 2.3 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.

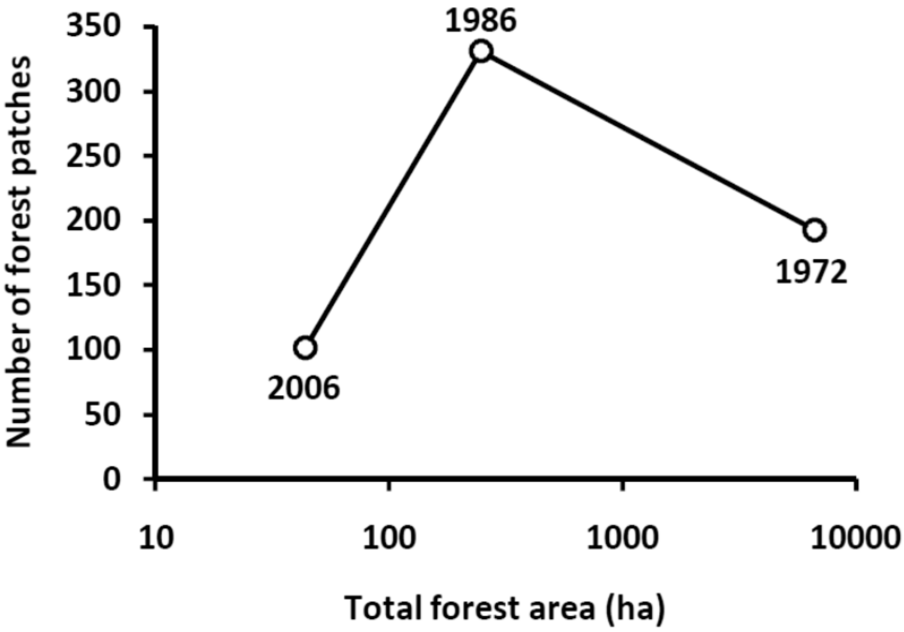

Figure 2.3 – *Forest cover disintegration in the Collines department (Benin) between 1972 and 2006. Land cover data based on Landsat imagery. Due to anthropogenic pressure, open rain forests have been degraded and substituted by a mosaic of agricultural fields, fallow lands and savannahs. Landscape transformations are characterized by fragmentation (between 1972 and 1986) and attrition (between 1986 and 2006).*

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 et al., 2012). Figure 2.4 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.

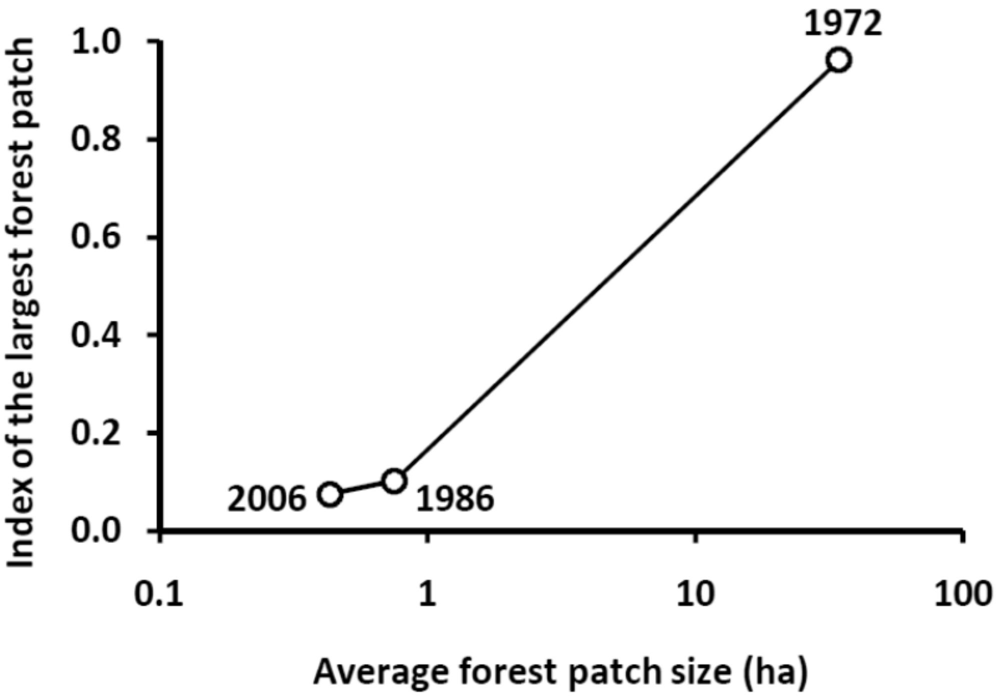

Figure 2.4 – *Forest cover disintegration in the Collines department (Benin) between 1972 and 2006. Land cover data based on Landsat imagery. Due to anthropogenic pressure, open rain forests have been degraded and substituted by a mosaic of agricultural fields, fallow lands and savannahs. Forests patches 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.*

2.2.4 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 and 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., 2011). 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 et al., 1993; Groom and Schumaker, 1993). Edges between different habitats are considered unique as a result of both biotic and abiotic influences (Groom and 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 et al., 1993; Bogaert et al., 2011). 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., 2011). 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 Köppen (Dudu, 1991). MFR is mainly composed of primary forests of *Gilbertiodenron dewevrei* (Caesalpiniaceae), next to secondary forests and fallow lands (Makana, 1986; Kahindo, 1988; Dudu, 1991; Mabay, 1994). Secondary forests are dominated by *Pycnanthus angolensis, Zanthoxe longilletii, Cynometra hankei, Petersianthus macrocarpum, Funtumia elastica* and *Uapaca guineensis*. Fallow vegetations are characterized by associations of *Aframomum laurentii* and *Costus lucanusianus* and by those of *Triumfetta cordifolia* and *Selaginella myosurus*.

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 2.1 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. (2011) 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 ($\chi^2 = 0.27$), nor between the fallow habitat and the forest habitat ($\chi^2 = 1.35$); or between the forest habitat and the edge habitat ($\chi^2 = 0.73$). 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 (*Anomalurus derbianus*) was observed only in the forest habitat. One single species (*Mastomys natalensis*) 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., 2011). The low species numbers in the contingency table could be mentioned as another possible cause of these non-significant results (Causton, 1988).

Table 2.1 – *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*

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 2.5) and between the forest and edge habitat ($R^2 = 0.129; p > 0.05$; Figure 2.6), 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., 2011). 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 *Graphiurus lorraineus* 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 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

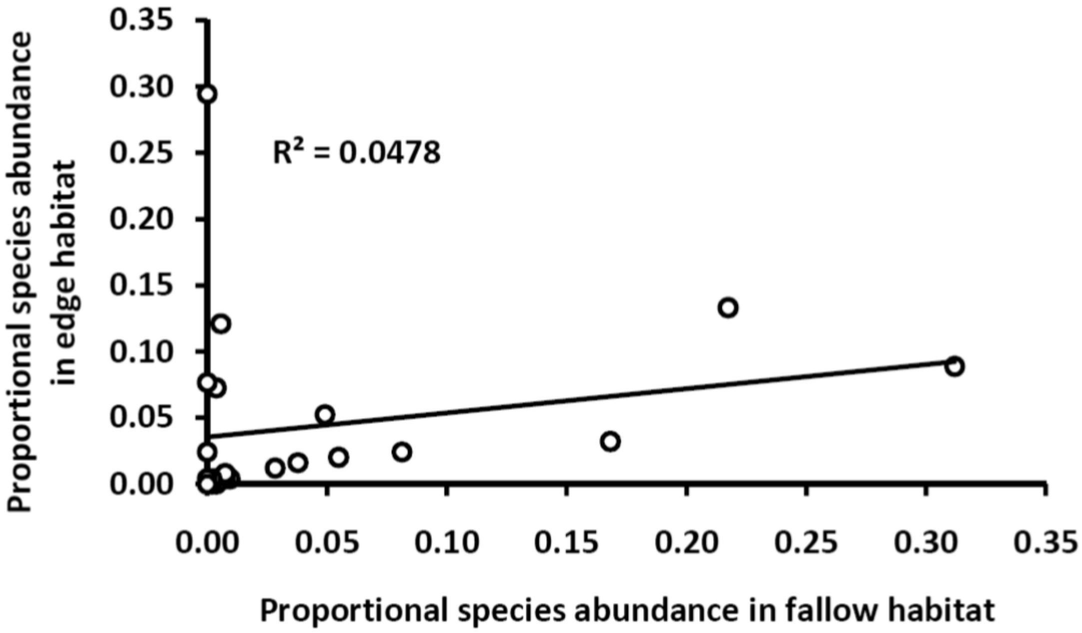

Figure 2.5 – *Comparison of rodent abundances (23 species; 1 species absent in both habitats; 6 species found only in one of both habitats) between the fallow and the edge habitat in the Masako Forest Reserve (Kisangani, Democratic Republic of the Congo). Captures effectuated during seven months between 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.*

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".

2.3 Habitat loss caused by anthropogenic structures: road systems

2.3.1 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; 1998; Carr et al., 2002; Lindenmayer and Fischer, 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 and Fischer, 2006). Road disturbance corridors, acting as filters, affect invertebrates as well

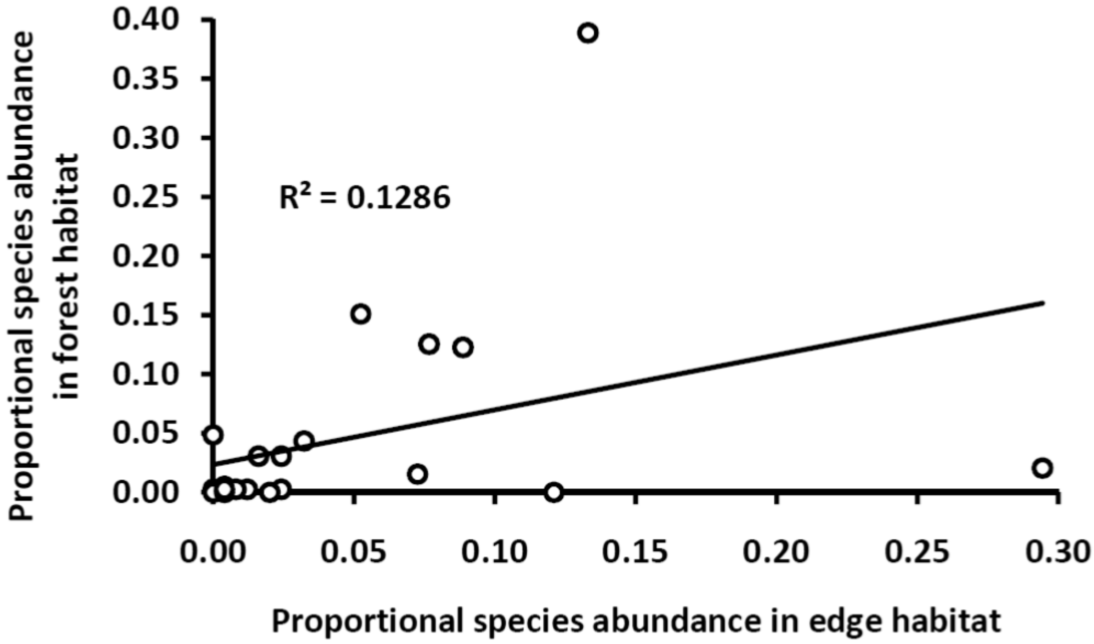

Figure 2.6 – *Comparison of rodent abundances (23 species; 1 species absent in both habitats; 6 species found only in one of both habitats) between the edge and the forest habitat in the Masako Forest Reserve (Kisangani, Democratic Republic of the Congo). Captures effectuated during seven months between 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.*

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 and 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 and Fischer, 2006), as well as on traffic volume and inhospitable road corridor width (Forman, 1995; Carr et al., 2002). Road mortality represents the main sink in road corridors. Where wildlife corridors and roads intersect, road-killed animals are frequently observed (Forman, 1995; Forman and Alexander, 1998; Lindenmayer and Fischer, 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 (Forman, 1995; Forman and Alexander, 1998; Carr et al., 2002).

2.3.2 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 and 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 and 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 km^2 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 and 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 and Fischer, 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:

$$R = \frac{I}{E} \tag{2.1}$$

$I + E = A$, 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 2.7, some examples are given of the different distances of edge influence used in the simulations.

Figure 2.8 shows the effect of the distance of edge influence on R. As a consequence of the large forest cover in the study area (12618 km^2) 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.

This behavior of 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

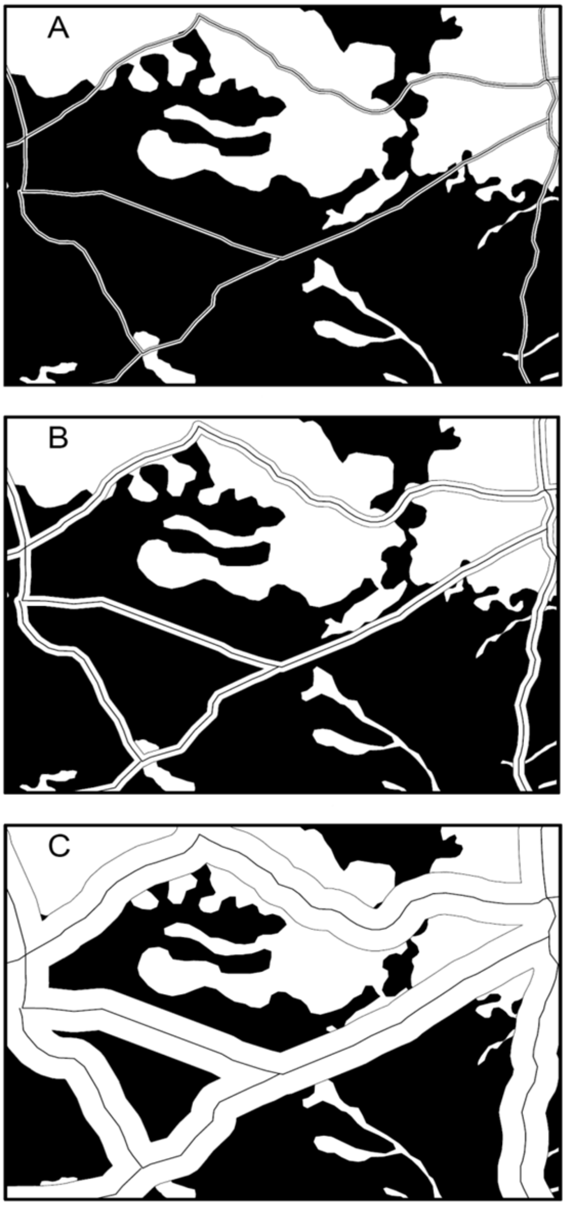

Figure 2.7 – *Potential edge effects as a consequence of road system development in the territory of Kambove (Katanga, Democratic Republic of the Congo). Three distances of edge influence are shown: (A) 100 m, (B) 250 m and (C) 1000 m. As a consequence of the edge effect, forests situated in the edge 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 km^2) of the study area is shown.*

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. (2011) it has been shown that, according to concession maps of the Congolese Mining Cadastre (Kasongo, 2008), 78.5% (91348 km^2) 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.

2.3.3 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

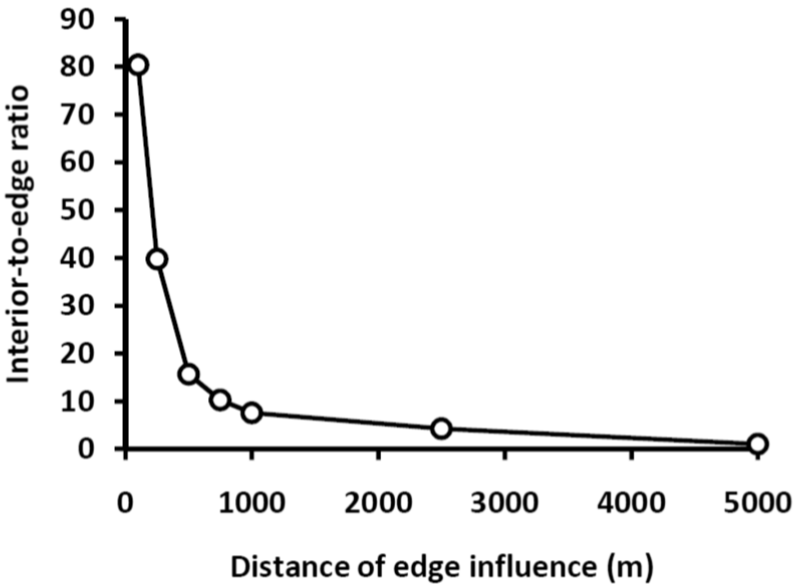

Figure 2.8 – *Assessment of the ecological impact of a road system (2874 km) on forest habitats (territory of Kambove, Katanga, Democratic Republic of the Congo) by means of the interior-to-edge ratio. Only forests situated outside the area defined by the distance of edge influence are considered as 'interior forest habitat'. The low density of the actual road system can explain the high values of the ratio.*

et al., 1997; Forman and Alexander, 1998; August et al., 2002; Lindenmayer and 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 and 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 km^2 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 and 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:

$$X_{t-1} \times M = X_t \tag{2.2}$$

Initially (1984), the Southwestern, Southeastern and Northwestern study areas showed a higher forest presence with respectively 65.3, 54.7 and 47.3 km^2. A lower forest cover of 25.4 km^2 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 2.9 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 and 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 and Wallin, 2002).

2.4 Habitat loss caused by anthropogenic structures: 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 and Lo, 2003; Deng et al., 2009; McGee, 2009; Su et al., 2010).

2.4.1 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

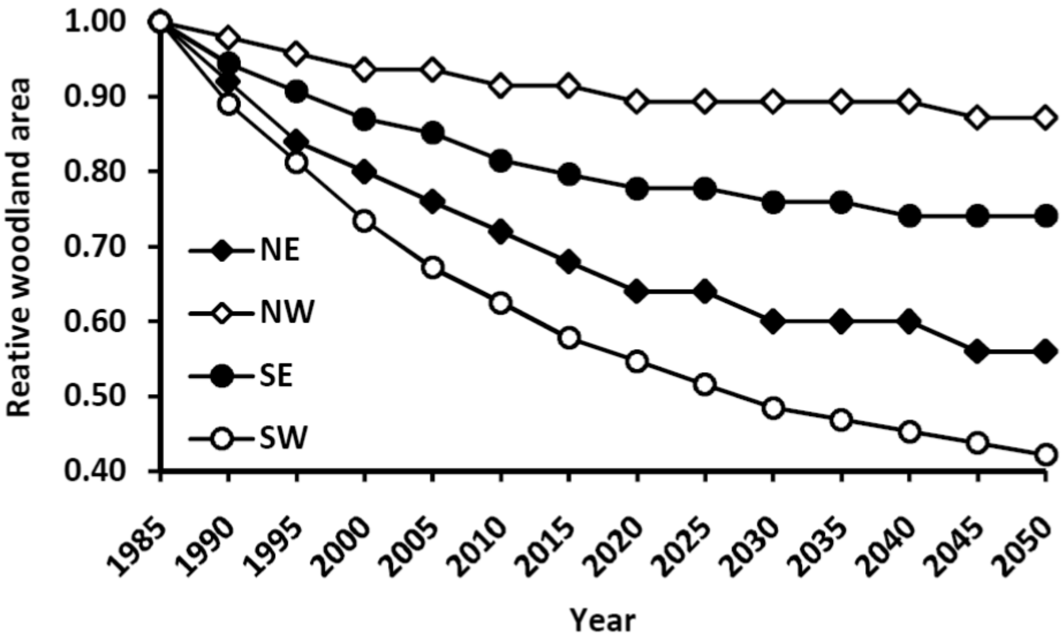

Figure 2.9 – *Deforestation of woodland (miombo) around Lubumbashi (Democratic Republic of the Congo) between 1985 and 2050 based on a first order Markov model. Four zones of about 1700 ha situated Northwest (NW), Northeast (NE), Southwest (SW) and Southeast (SE) are considered. Woodland area is expressed 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.*

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.; 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 and Lo, 2003; Vandermotten et al., 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.; Yang and Lo, 2003; McGee, 2009).

As for developing countries, unprecedented population growth and urban extension are observed (Robinson et al.; Cohen, 2004; 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, 2004; Yang and 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, 2004; 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, 2004; Deng et al., 2009; McGee, 2009).

2.4.2 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 and Fischer, 2006). For example, 90% of the African population depends on firewood for their energy needs (Lindenmayer and Fischer, 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 km^2 (UNEP 2004), and Ubundu (0°21′S, 25°25′E), characterized by about 10 inhabitants per km^2 (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 and 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 2.10).

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

$$D = \frac{a_{1986} - a_{2001}}{a_{1986}} \tag{2.3}$$

With a_j the forest cover in year j . Figure 2.11 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 2.11 also shows the footprint of the cities, which largely exceeds the city limits. These observations

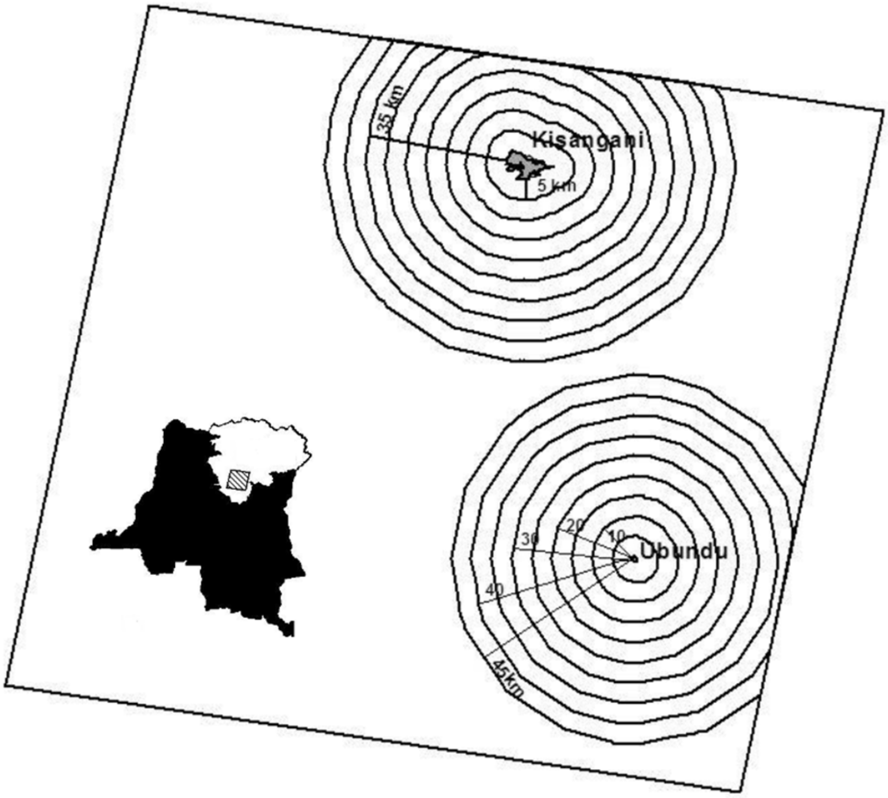

Figure 2.10 – *Situation of the study area (shaded area) containing Kisangani and Ubundu in the Oriental Province (shown in white) of the Democratic Republic of the Congo (small map). Around the limits of both cities, buffer zones with a radius from 5 to 45 km were constructed. In each buffer zone, the deforestation rate between 1986 and 2001 was determined. Land cover was based on Landsat data.*

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., 2010b;a).

2.5 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

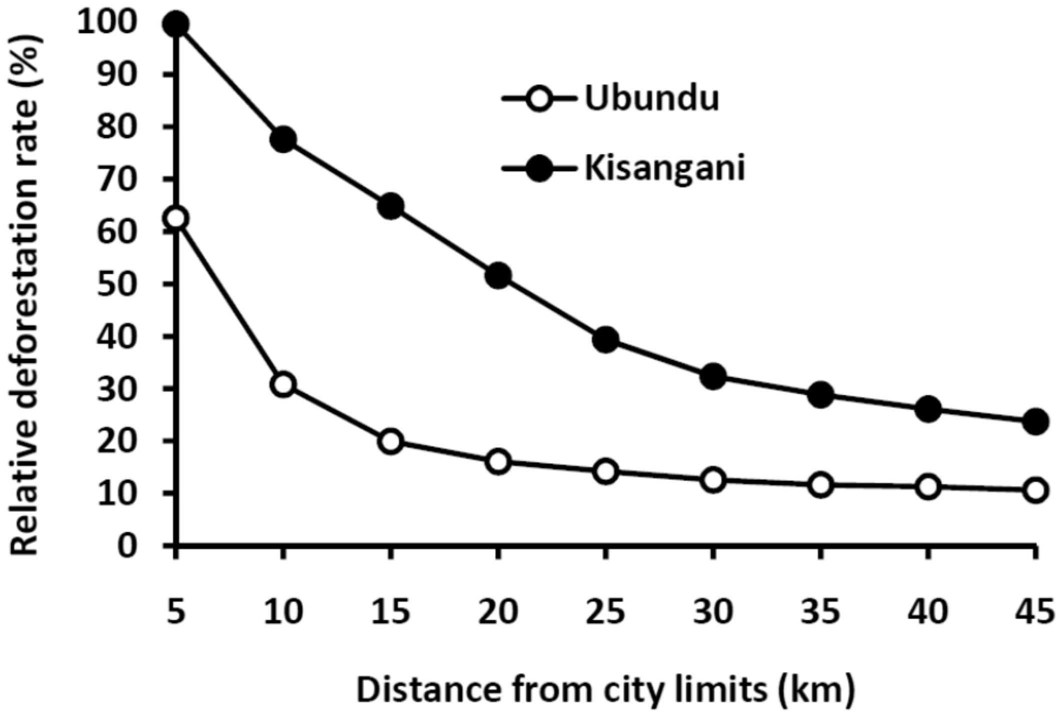

Figure 2.11 – *Deforestation rates between 1986 and 2001 in Kisangani and Ubundu (Oriental province, Democratic Republic of the Congo) as a function of the proximity to the city. Each distance corresponds to the outer limit of a concentric ring or buffer zone. The width of each ring was equal to 5 km. 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.*

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 disappear. 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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Part II

Conceptual research and developments

CHAPTER 3

Spatially explicit quantification of anthropogenic landscape change. Towards a new methodological framework

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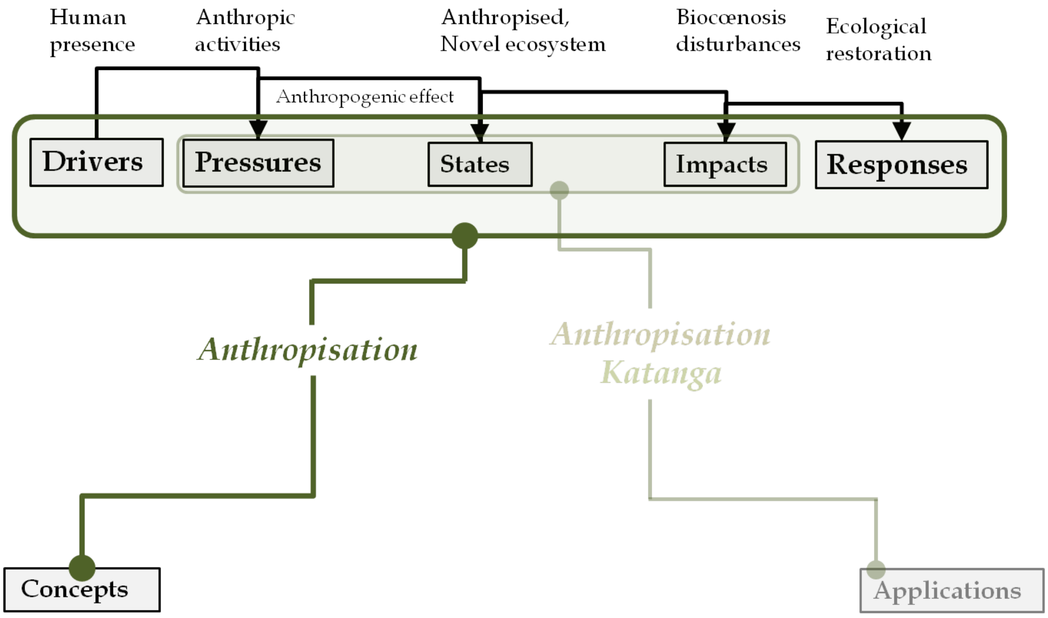

Figure 3.1 – *Location of the anthropisation article (olivine green theme) in the thesis mind map. Technical reading: concepts, thematic reading: anthropisation (question of Section 1.2.1), logical reading: all stages. The hollow green article is thematically related to this one. This chapter represents the departure point of any further development and application. It aims to assemble diverse disciplines related to landscape ecology and connect all they can tell about human-driven landscape change into a logical network (DPSIR). It also points out weaknesses of anthropogenic effect assessment and proposes guidelines to overcome them.*

3.1 Abstract

Anthropogenic disturbances impact the whole biosphere, at least indirectly, and are of major concern in various disciplines and policies. However, most studies are very specific to their thematic, creating their own reference framework without connection with other research on related purposes in other disciplines. This plethora of terms and concepts in turn impedes comprehensive tackling of that issue and complementarity between studies. Here, we aim to give an integrated approach of the concern that could be at the basis of land planning or environmental management practices and policies. To do so, we use the levels of study and paradigms of landscape ecology, a discipline of great integrative power. We combine it with the Drivers-Pressures-State-Impacts-Response (DPSIR) framework to reorganise the causal logic of anthropogenic disturbance processes and patterns addressed at ecosystem or landscape level and to highlight the potential effect of response actions undertaken at each step, in an action-oriented perspective. First, we briefly review the concepts (anthropisation, naturalness, hemeroby, novel ecosystem, climax, etc.) and methods used to assess anthropogenic changes, exploring different branches of ecology and geography. The strengths and weaknesses of the existing approaches are then used to open perspectives on a new analytical framework: we propose a new, action-oriented reference state, and we present general guidelines to quantify landscape anthropisation. Our methodological perspective is to combine object-oriented and gradient analyses and is based on the assessment of ecosystem disturbance, landscape configuration and dynamics, easily acquired data sets and patch dynamics analysis.

Keywords: anthropogenic effect, novel ecosystem, hemeroby, naturalness, landscape ecology, environmental indicator

3.2 Introduction

Human impacts on the environment have existed for so long that little or no area in the world is now considered as untouched, at least indirectly (Sanderson et al., 2002; Ellis and Ramankutty, 2008). Humans transform the ecosystem patterns and processes, (Sanderson et al., 2002; Burel and Baudry, 2003; Ellis and Ramankutty, 2008; Bogaert et al., 2014). This phenomenon is at the centre of environmental assessment, policy making and land planning issues, addressed by various disciplines such as botany, population ecology, conservation management, restoration ecology, landscape ecology and geography (Jalas, 1955; Naiman et al., 1988; O'Neill et al., 1988; Grabherr et al., 1992; Solon, 1995; Peterken, 1996; Lambin and Geist, 2006). Though some interactions already exist between these disciplines, each focuses on its own issue of interest, analysing such anthropogenic transformations with its own terms and analysis framework (focus scale, hierarchical level and variables of interest) (Green and Sadedin, 2005). The causes and consequences of human activities are therefore generally explored separately.

However, combining the multitude of information arising from the abundant research on anthropogenic transformations in a more transdisciplinary, action-oriented approach could prove a remarkable breakthrough, not only for fundamental research but also regarding sustainable development and policy making. On the other hand, many problems in ecology and resource management are related to landscape use. In that context, the focus scale of landscape ecology may bring the appropriate framework. This discipline relies on the “pattern and process paradigm”, which states that landscape structure (namely its composition and configuration) and ecological processes are interdependent (Turner, 1989). Although centred on the hierarchical level “landscape” (being groups of interacting ecosystems), the knowledge of many other disciplines at higher and lower scales are used in landscape ecology to understand the context and the mechanisms involved (Wiens et al., 1993; Bogaert and André, 2013).

In order to approach a comprehensive and transdisciplinary analysis, the concepts and methods used to represent human impacts on landscapes will also be analysed within the Driver Pressure State Impact Response (DPSIR) framework throughout this review (Ness et al., 2010). This sustainable policy-oriented analysis framework is frequently employed in reports in order to assess environmental problems, but is increasingly used in scientific articles as well. It integrates these problems within a social, economic and environmental context and presents them in the form of a causality chain so as for which action could be undertaken to tackle them (Smeets and Weterings, 1999).

This article has three main objectives: (1) to sort the different concepts from various disciplines related to the anthropogenic landscape changes , (2) to organise the variables and methods regularly used to assess them, (3) to combine the strengths of the existing variables and methods into guidelines for a general methodology, towards more completeness, exportability to various cases and clarity for the assessment of anthropogenic landscape change.

3.3 Current concepts

This section presents the concepts, sheltering disciplines and practices as well as underlying challenges to characterise human impact on the environment. These conceptions strongly influence the way human impacts on landscapes are measured. The most frequently employed terms, their synonyms, the nuances and relationships between each of term are described below and illustrated in Figure 3.3.

3.3.1 Anthropisation, anthropogenic effect

Anthropogenic effect, often employed in its plural form, is a general expression accounting for any kind of influence of human activities on the environment (Solon, 1995; Ellis and Ramankutty, 2008; Barima et al., 2011). In landscape ecology, the phenomenon is studied through the influence of human activities on landscape composition, configuration and dynamics (Bogaert et al., 2011; Vranken et al., in preparation-b). The term *anthropisation* is also used in numerous disciplines related to ecology and covers a wide range of specific features, from water composition (Mancini et al., 2005) to plant communities (Hill et al., 2002). The term *anthropisation* (or *anthropization*) is more frequently used by Latin language speakers than in English (Burel and Baudry, 2003; Machado, 2004; Bogaert et al., 2011; Diallo et al., 2011; Inostroza, 2012; Bogaert et al., 2014). However, those terms are not exactly synonyms, the former being defined as the process of landscape change as a consequence of the latter (Bogaert et al., 2014). They are nevertheless used without distinction by most authors. Several other expressions are employed to represent similar phenomena: anthropogenic disturbance(s), anthropogenic impact(s) / pressures, anthropogenic land-cover / landscape change, human disturbance / footprint, human impact / influence on land surface, etc. (Hannah et al., 1994; Solon, 1995; Menon and Bawa, 1997; Pielke et al., 1999;

Sanderson et al., 2002; Laurance, 2004; Machado, 2004; Lambin and Geist, 2006; Garbarino et al., 2009). The term landuse is also related to anthropisation: the human use of land (Haberl et al., 2001; Brentrup et al., 2002; Ramankutty et al., 2006).

Those expressions show a lack of distinction between cause and consequence. In order to clarify what is actually focused, the causality chain related to this phenomenon, Fig. 1, organises its different elements according to the DPSIR framework. This highlights the issues that can be addressed when providing responses to environmental problems caused by human intervention. The focused causal stages most generally addressed in landscape ecology are there put in perspective with the pattern / process paradigm. The "Driver", underlying cause of anthropogenic effect on landscapes is the presence of humans, considering their density, but also their lifestyle (Lambin and Geist, 2006). Through activities performed to support human needs, people generate environmental "Pressures", that influence ecological processes and modify the "State" of the landscape in its composition as well as its configuration (Sanderson et al., 2002; Lambin and Geist, 2006). This influence is called anthropogenic effect and the landscape in this state is considered as anthropised (Bogaert et al., 2014). "States" are generally observable through landscape patterns, while "Pressures" act more directly on ecological processes. That landscape structural change has in turn "Impacts" on ecological processes, disturbing local biocoenosis (Naiman et al., 1988).

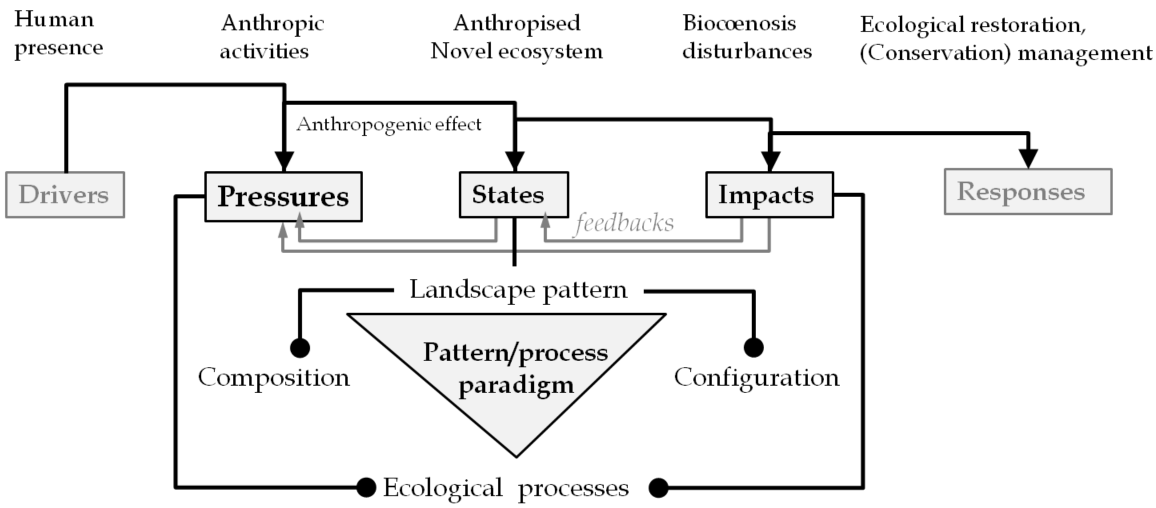

Figure 3.2 – *Combined DPSIR framework and landscape ecology's pattern / process paradigm applied to the issue of human impact on landscapes. Black boxes represent the focus of landscape ecology.*

What is not directly highlighted in the original DPSIR framework itself is that multiple feedbacks may arise (Ness et al., 2010). Processes affected by pattern change can affect landscape pattern in turn and *vice versa* (Forman and Godron, 1986b; Laurance, 2004; Lambin and Geist, 2006; Bogaert et al., 2014). It should be noted that anthropogenic impacts on ecosystems and landscapes are not intrinsically detrimental to ecological processes. For example, species diversity increases in moderately anthropised landscapes or man-made ecosystems have already been observed, like calcareous grasslands (Naiman et al., 1988; Bird et al., 2008; Piqueray et al., 2007; Chakraborty and Li, 2011). Distant drivers, such as delocated production or transformation of goods, also do not fit well to the DPSIR framework (Cumming et al., 2013).

3.3.2 Reference states

The reference states can be used either as a departure point to measure anthropisation intensity or as a goal to achieve while restoring a given site. There is a plethora of reference states that are all difficult to define specifically or achieve in a concrete case.

3.3.2.1 Naturalness

"Naturalness", i.e. the characteristic of what is natural is the state of the system when no human activity has influenced it. It can be considered as the opposite of anthropisation (Machado, 2004; Bogaert et al., 2014). It was first used in botany and ecology (Tüxen and Preising, 1956; Kowarik, 1999; Steinhardt et al., 1999; Machado, 2004; Lecomte and Millet, 2005; Reif and Walentowski, 2008; Rüdisser et al., 2012; Winter, 2012) and is widely used and discussed among the scientific community (Peterken, 1996; Machado, 2004; Lecomte and Millet, 2005; Schnitzler et al., 2008). It can be associated to climax, but the use of this term is obsolete and controversial. Its climatic determinism and the existence of long-term steady conditions for climax settlement are indeed questioned (Henderson et al., 1960; Bournerias, 1982; Hall, 1995; Cook, 1996; Genot, 2006). "Wilderness" is frequently used in Australia and the U.S.A. and wrongly applied as a synonym to naturalness, tough wilderness implies remoteness and large extent (Peterken, 1996, Machado, 2004).

Naturalness is also a measure of the difference between natural and current states called "degree or level of naturalness" (Peterken, 1996; Kowarik, 1999; Machado, 2004; Lecomte and Millet, 2005; Rüdisser et al., 2012; Winter, 2012). However, for this measure, we will prefer the use of the term hemeroby in order to avoid confusion.

Distinguishing what is natural and what is not requires defining what is considered as artificial or detrimental to nature. The historical and theoretical point at which human impacts begin to be taken into account is subject to debate because it addresses the relationship between human and nature (Haila et al., 1997; Machado, 2004; Mascaro et al., 2013). Most authors consider anthropogenic land transformation as artificial (Ellis and Ramankutty, 2008). It can begin with agriculture (Lecomte and Millet, 2005; Mazoyer and Roudart, 2006), or industrialisation (Mackey et al., 1998). Those thresholds were not simultaneous everywhere (Mazoyer and Roudart, 2006). Even once the type of human activity considered as artificial is chosen, it remains practically difficult to identify it in the field due to, for example, ancient human activity such as past slash and burn agriculture dating back to over 300 years in tropical forests (Vleminckx et al., 2014).

"Original naturalness" describes the state before the first anthropogenic land transformations (Peterken, 1996). It is sometimes called "naturalness of the first post-glacial times" or "biological naturalness" in Europe (Gilg, 2005; Lecomte and Millet, 2005; Fuhr and Brun, 2010). In restoration ecology, original naturalness is still mentioned as reference (Bakker and Berendse, 1999; Science and Policy Group, 2004). However, achieving original naturalness is practically not possible, nor meaningful: natural changes (astronomical and tectonic variations) and disturbances have occurred since the first human impacts (Sutherland et al., 2004; Lecomte and Millet, 2005; Harris et al., 2006; IPCC, 2007; Winter et al., 2010). Moreover, knowing in which state the landscape was before the first human influences is practically difficult to achieve in the case of landscapes with a long anthropisation history.

"Virtual naturalness ", also called "present naturalness", would be the current state of the system if human had never had any impact on it (Peterken, 1996; Lecomte and Millet, 2005). This state takes natural evolution into account. If an ecosystem had never been impacted by human activity, even indirectly (which is actually nowhere the case), its virtual and original naturalness would correspond to its current state. As with original naturalness, this reference also depends on the kind of human activities that are considered natural, and this state is also particularly difficult to characterise for landscapes with a long anthropisation history.

"Potential naturalness" or "potential natural vegetation/community" is the state that would develop if human influence disappeared and if the resulting succession were finished instantly (Tüxen and Preising, 1956; Hall, 1995; Peterken, 1996; Reif and Walentowski, 2008). In practice, it represents the self-sustaining ecosystem in the disturbed abiotic condition of the site (Hall, 1995). Such definition is taken as reference for less ambitious restoration projects or conservation management (Lundholm and Richardson, 2010).

"Future naturalness", also referred to as "anthropogenic naturalness", corresponds to the state that the system eventually reaches after human influence ceased and after complete ecological succession (Peterken, 1996; Gilg, 2005; Schnitzler et al., 2008). If the system has been too deeply altered, this state differs from virtual naturalness due to global human impacts such as anthropogenic climate change or species extinction (Peterken, 1996).

Most of the naturalness definitions cannot actually be achieved. Yet, for restoration purposes, an operational definition of naturalness representing an achievable reference state that is more ambitious than potential naturalness could prove useful.

3.3.2.2 Novel ecosystem, analogous

The term "Novel ecosystem" originates from ecology and is mostly employed by restoration ecologists (Mascaro et al., 2013). At a certain intensity of anthropogenic effect on the local landscape as well as its global surroundings (such as climate), some anthropogenic disturbances are so heavy that it seems impossible to go back toward the natural state: tje cjaracteristics of Novel ecosystems differ from one author to another. The most consensual definition mentions (1) that a novel ecosystem has abiotic, biotic and social components that, because of anthropogenic effects, differ from those that prevailed historically in the area and (2) that those historical qualities cannot be recovered due to the crossing of ecological, environmental and social thresholds (Hobbs et al., 2013). The definition and position of this threshold, "barrier" or "point of no-return" are subject to debate (Hallett et al., 2013).

In response to that, a novel ecosystem can be managed to enhance its ecosystem services or health, but it will be different from what it was before anyway (Harris et al., 2006; Hobbs et al., 2006). A typical way to manage a novel ecosystem is to find a natural analogous ecosystem in the same area with similar functional characteristics to use it as a reference. The ecosystem managed in this way is called "artificial analogous" ("analog" or "analogue") (Lundholm and Richardson, 2010). This landscape will have functionally analogous abiotic and biotic features to natural landscapes occurring in the same region (e.g. quarry face can be analogous to cliff) (Lundholm and Richardson, 2010).

3.3.3 Hemeroby

Hemeroby represents the measure of the difference between a reference (natural) state and the anthropised state of a system. It is an integrative measure of the degree of human influence, intended or not (Jalas, 1955; Kowarik, 1990; Hill et al., 2002). This concept originates from botany in Europe and has mostly been applied to plant species (Jalas, 1955; Kowarik, 1990; Steinhardt et al., 1999), forest communities (Steinhardt et al., 1999; Acosta et al., 2003; Rüdisser et al., 2012), ecosystems (Kowarik, 1990) and more recently landscapes, principally agrarian and forested (Machado, 2004; Renetzeder et al., 2010).

This concept is similar to anthropogenic effect, although its quantification methods are different. Naturalness equals zero degree of hemeroby. There exist different definitions of this reference state, though hemeroby rarely specifies which one is taken (Peterken, 1996, Steinhardt et al., 1999, Brentrup et al., 2002, Fanelli et al., 2004, Rüdisser et al., 2012).

3.3.4 Responses to anthropisation

Depending on the stage of anthropogenic impact undergone by the ecosystems of interest, humans can give different types of “Responses” to environmental disturbances. The sooner humans intervene in a response strategy, the less harmful for the virtual naturalness of the ecosystem. Those response actions are, by ascending anthropogenic impact: Avoid, Minimise, Rectify, Compensate, Enhance (Rajvanshi, 2008). Avoiding the potential impact of human activities falls within the frame of biological conservation or preservation (Gutzwiller, 2002; Rajvanshi, 2008). Minimising the spatio-temporal scale of the impact during design and construction of human infrastructures is usually addressed in environmental impact assessment studies (Rajvanshi, 2008). Rectifying after the impact has occurred is applied to hybrid ecosystems — not yet novel but no longer natural —, while compensation is applied to novel ecosystems. Both actions are addressed by restoration ecology (Science and Policy Group, 2004; Rajvanshi, 2008). Applying compensation or enhancement actions to novel ecosystems in order to develop their functional analogy to existing natural ecosystem, even exploit it, is a form of environmental management generally addressed by restoration ecologists or, in the case of agro-ecosystems, agroecologists (Science and Policy Group, 2004; Rajvanshi, 2008; Fischer et al., 2014). Such “Responses” tackle different stages in the causality chain (Smeets and Weterings, 1999; Ness et al., 2010) (Figure 3.2).

Ecological restoration is the process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed (Science and Policy Group, 2004). This supposes the existence of a reference ecosystem that will serve as model (Science and Policy Group, 2004). Management is the process of enhancing an ecosystem towards less disrupted ecological processes and spatial patterns. Conservation, or conservation management, consists in preserving an area from anthropogenic disturbance in order to protect local biodiversity or contribute to its recovery, after restoration or as a management objective (Gutzwiller, 2002; Weddell, 2002; Science and Policy Group, 2004). For example, biodiversity can be artificially enhanced through conservation management applied on novel ecosystems (Lundholm and Richardson, 2010).

3.3.5 Graphic synthesis

A combination of the main concepts developed hereabove is presented in Figure 3.3. Considering at first a landscape completely preserved from human influence under growing human impact (hemeroby scale), the state in which this landscape was before being disturbed is the original naturalness. As time and other local and global natural disturbances go by, anthropogenic influence increases and the landscape is then in “Hybrid state”. If by that time human disturbance ceases, it can recover to what it would be like if no human had ever influenced it, either rapidly through active restoration or more slowly through ecological succession: this state is its virtual naturalness. After the non-return threshold is crossed, the landscape is anthropised and its composing ecosystems are then considered novel due to the changes in biotic composition, but also in local and global abiotic conditions (double arrows).

If the anthropogenic effect on this landscape stops increasing, its hemeroby reaches a plateau, and if the anthropogenic effect ceases or decreases, the hemeroby level decreases as well. However, as the ecosystems are now novel, they cannot recover their previous state: this is represented using a second hemeroby scale, showing that the evolution towards « more naturalness » follows a different path. If no management is applied to the landscape, ecological succession will lead progressively to future naturalness. Either on hybrid or novel ecosystems, potential naturalness can be used as a reference for restoration: the « most » natural ecosystems that can occur in disturbed global and local abiotic conditions. Another reference state could also be used, though inexistent in the literature: restored indigenous naturalness. This state can represent the « most natural » one achievable through restoration or conservative management: it goes further than potential naturalness, as by removing the result of abiotic disturbances (e.g. soil pollution) from the site. Artificial analogous landscapes result from another form of novel ecosystem management. It uses the result of human disturbances as a departure point to build a new landscape.

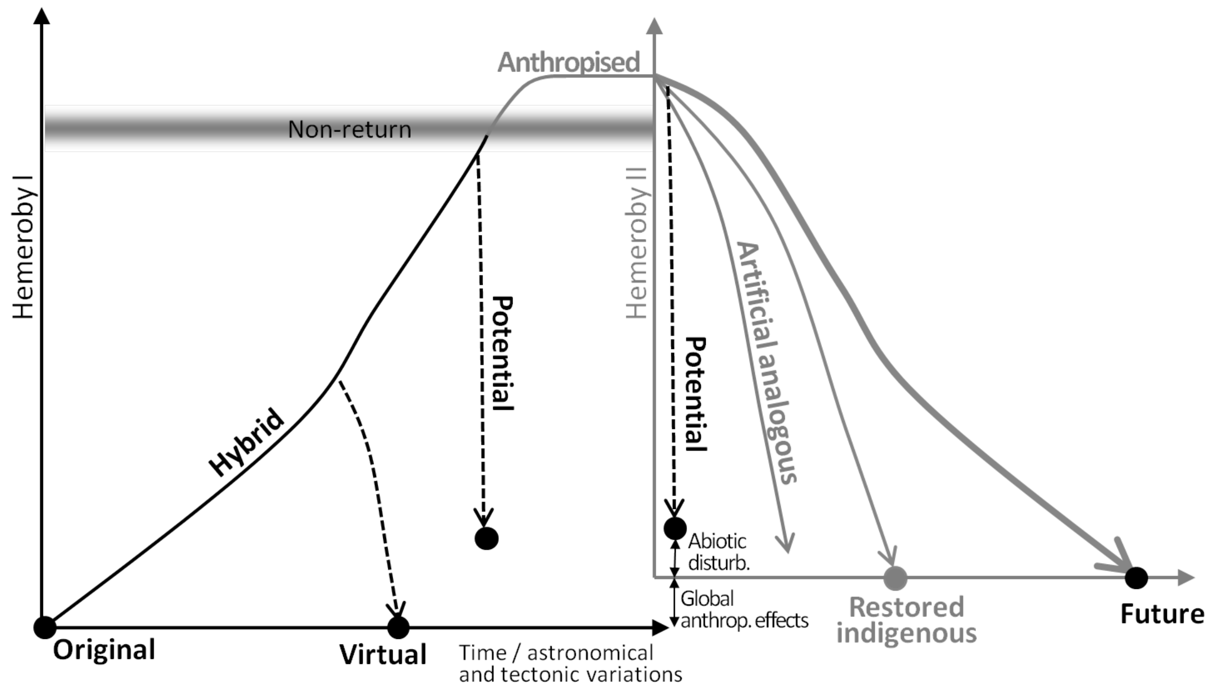

Figure 3.3 – *Graphic synthesis of the concepts related to human impacts on landscapes from ecological, landscape ecological and restoration ecological perspectives. This representation is a theoretical and simplified model of the different anthropisation trajectories that a landscape can follow. Dashed lines represent active restoration, plain upward line represents landscape evolution under increasing human disturbance, plain downward lines represent management, horizontal dashed lines represent conservation and bold line represents ecological succession. Dots represent reference states of naturalness, bold characters non reference states. The wide blurry line labeled « Non-return » represents the novel ecosystem threshold, grey elements being related to novel ecosystems.*

3.4 Current assessment materials

3.4.1 Data format

The three main data types used to represent ecological and landscape variables are stretched values (gradients), categorical maps (classes) and patches (patch dynamics) (Wiens et al., 1993; Gustafson, 1998). Each data model provides its own capabilities, which influences the representation and analysis of the variable distribution, with consequences on pattern identifications (Gustafson, 1998).

3.4.1.1 Stretched values

Stretched values are used to represent the spatial distribution of data under continuous variations (Gustafson, 1998). They are also called point data, but we chose to exclude this term here in order to avoid confusion with data represented as point features in Geographic Information Systems (GIS) softwares. That makes it more thematically precise and suitable to study transitions along ecological gradients like edge effect (Gustafson, 1998; Sutherland et al., 2004). Stretched values are generally represented using raster maps. This format is frequently used in ecology to represent relief, climate or diverse ecological gradients. This format makes less assumption about the nature of spatial structure (Gustafson, 1998).

3.4.1.2 Categorical mapping

Categorical mapping is the simplest and most widely employed thematic cartography method in various disciplines to represent vector or raster data (Boots and Csillag, 2006). In landscape ecology, this method is mainly used to represent landuse and landcover types. This representation involves discontinuous variation of the data, often arbitrarily defined (Gustafson, 1998). The features classified in the same category share the same characteristics (Gustafson, 1998). This is detrimental to thematic resolution: all the local ecosystems (patches) of the same class may not be evenly affected by anthropogenic disturbances (Stromgaard, 1985; van der Werf et al., 2008; André et al., in revision).

3.4.1.3 Patches, patch dynamics approach

Ecosystems can also be studied for their individual properties as habitat patches. This is often referred to as “Patch dynamics” in landscape ecology because this format is used to distinguish the evolutionary trajectories of individual patches (Wiens et al., 1993). Such representation is the expression of the patch-corridor-matrix model in landscape ecology (Forman, 1995). This object-oriented format is comparable with attribute tables of vector layers in GIS softwares: each object has a distinct identifier (Wiens et al., 1993; Forman, 1995). The patch dynamics approach combines advantages of both categorical mapping and stretched values: it distinguishes entities trajectories and highlights individual habitat specificities. Patch dynamics is at the interface between ecology and landscape ecology, between spatial patterns and ecological processes (Wiens et al., 1993).

3.4.2 Variables

Data availability is a critical issue, particularly in the case of developing countries, where accurate data on infrastructure and socio-economic data are scarce. These areas though represent particular interest for land planning and conservation (Groombridge et al., 2002; Sanderson et al., 2002). It should be noted however that the availability of data acquired by field survey may be correlated with human accessibility to landscapes. In the case of anthropogenic effect assessment, this may lead to substantial bias, whereas creating access to it represents the potential future disturbance (Ruiz-Gutiérrez and Zipkin, 2011). In that context, remote sensing is an essential tool.

When applied at landscape level, hemeroby, naturalness and anthropisation are quantified using the same variables. In order to put in perspective which quantification focused on causes and which on consequences, these variables were classified according to the DPSIR framework (Figure 3.4).

Economy and population variables are used to represent human presence as a Driver of anthropisation. Generally, such variables are population density and income, formatted as categories or stretched values (Menon and Bawa, 1997; Haberl et al., 2001; Sanderson et al., 2002). Human activities are regularly quantified using infrastructures and disturbance type (input, output, structural biotope transformation), as Pressures. The data used in this case are linear, punctual or polygon infrastructures like roads, farms, electric power plants, quarries, villages, etc., following a categorical or "patch" formatting (Menon and Bawa, 1997; Sanderson et al., 2002; Garbarino et al., 2009). Though those are used as proxies for disturbance processes, disturbances like the amount of emitted pollutant, field nutrient or the surface of cleared forest can also be measured *per se*, best formatted as stretched values (Brentrup et al., 2002).

State represents the landscape anthropisation phenomenon per se. Anthropised state in itself is most quantified using landuse and landcover data at landscape level, mapped as categories (O'Neill et al., 1988; Pielke et al., 1999; Foley et al., 2005; Bamba et al., 2008; Vranken et al., 2011; Bogaert et al., 2014). At finer scales, biotope descriptive variables are also used (as stretched values), like pollutant concentration, fertilizer, or percentage of soil sealing (Peterseil and Wrbka, 2001; Rüdisser et al., 2012), which can be described as stress factors (*sensu* Grime (1979)). The Impacts of anthropised state on biotic communities are measured using biocenosis parameters, such as species origin, abundance or diversity, formatted as stretched values or categories (Fanelli and De Lillis, 2004; Machado, 2004; Reif and Walentowski, 2008).

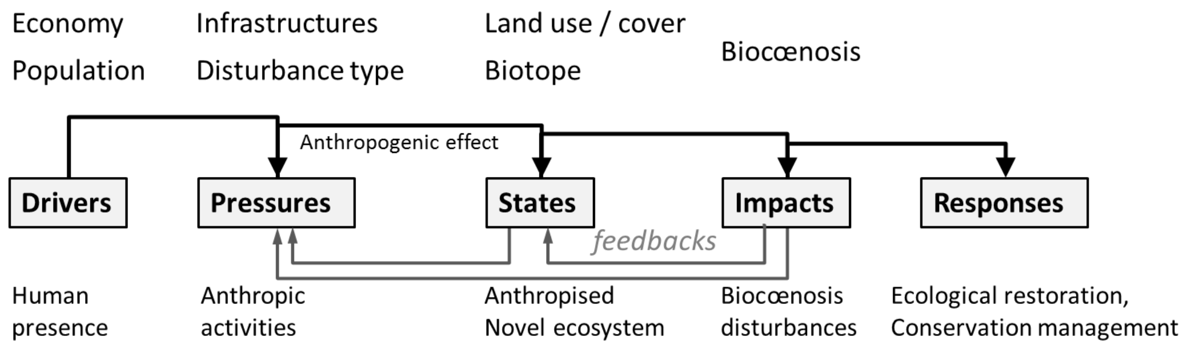

Figure 3.4 – *Thematic types of variables related to the quantification of human impact on landscapes, reported above each corresponding stage of the DPSIR framework. Human impact stages are reminded under the boxes.*

3.5 Current quantification methods

3.5.1 Simple, direct measures

Simple quantitative measures of the variables (OECD, 2008) can represent landscape composition (O'Neill et al., 1988), configuration (Sanderson et al., 2002; Garbarino et al., 2009) or ecological processes (Margalef, 1958; Garbarino et al., 2009). These measures are directly representative for their corresponding phenomenon occurring at a given DPSIR stage, but when interpreting them at other stages or other phenomena occurring at the same stage, they form indirect indicators. For example, pollutant emissions are a measure of anthropogenic disturbance, but also a proxy for level of landscape anthropisation.

These quantitative measures allow powerful comparisons and less arbitrary interpretations, but often give an incomplete perspective of the represented phenomenon (Scientific Commitee on Problems of the Environment, 2012).

3.5.2 Scales

Qualitative ordinal scales is the way hemeroby is represented. It gives categorical representation of anthropisation. The main differences between the existing hemeroby scales are related to: (1) the hierarchical level to which it is applied; (2) the reference state considered; (3) the number of levels of the scale applied; (4) the parameters used.

Each scale is generally designed for a given organisational level: plant community, ecosystem, landscape (Fanelli and De Lillis, 2004; Rüdisser et al., 2012). As for the reference state, some authors use original naturalness, others future naturalness but it is often not specified (Kowarik, 1990; Winter et al., 2010; Chirici et al., 2011). The number of scale levels varies from 4 to 11 and the used parameters to distinguish them are generally based on species or landscape composition (Peterken, 1996; Peterseil and Wrbka, 2001; Brentrup et al., 2002; Machado, 2004; Rüdisser et al., 2012). The criteria defining scale levels and the distance between them are not always specified nor constant.

Scales are arbitrary but allow distinguishing entities and including qualitative data. However, the different components of the system considered can correspond to different levels, for example fauna can be more natural than flora (Lecomte and Millet, 2005), while regressive dynamics are mixed with progressive ones.

3.5.3 Composite indexes

The aforementioned simple measures and scales can be combined in composite indexes. An index integrates complex environmental conditions into one parameter which cannot be measured directly (Rüdisser et al., 2012).

Specific attention should be paid to which information is actually contained in the composite index, how their contribution to the final index was weighed and on which hypotheses it relies if it uses arbitrary classifications. Comparisons would be eased if similar methods could be applied on different landscapes.

3.6 Towards a new assessment framework

3.6.1 The restored indigenous naturalness concept

Due to the lack of an operational natural state we propose a new reference : the maximal naturalness that can be locally achieved, the natural ecosystem present if there had never been any local disturbance, even abiotic. Landscapes with no direct human disturbance would also be in this new type of natural state, being subject to global disturbances but not local ones (Harris et al., 2006). So, any restorative action could be situated along a hemeroby scale, the minimum of which would be this maximal attainable naturalness. We propose to name this new concept “restored indigenous naturalness”(Figure 3.3). Its adaptation to specific cases would however lead to the recurrent question associated with reference states definition: how to know what would be there? In practice, reference states are often used based on functional similarities to current ecosystems and historical data on species composition and biotope, when available (Science and Policy Group, 2004).

3.6.2 Guidelines towards a more integrated methodology

The ideal data should be based on easy data acquisition, formatted to distinguish patch trajectories and connectivity, comparable with other areas, and as direct, objective and complete as possible.

If stretched values are acquired by remote sensing or aerial imagery, spatial as well as temporal location rules can be defined by combining such stretched values with categorical maps. If so, a limited field survey and spatial modelling would allow distinguishing a maximal number of patches and patches originating from different dynamics or encountering different anthropogenic pressures would be isolated, as developed in André et al. (in revision). Therefore, the patch dynamics perspective could be approached by refining categorical mapping through its combination with stretched values

Taking connectivity into account is possible by calculating the “distance from natural habitat” gradient as a stretched values (Rüdisser et al., 2012). Such information represents interesting perspectives for land planning, conservation and restoration by highlighting which areas should be preferably protected or restored. The effect of adding or removing natural patches in the landscape could be simulated in this way. To take functional connectivity into account, Wiens et al. (1993) propose to investigate the relationship between individual behaviour (including human activities, in our case) and patch design on microlandscape with analogous patterns and populations as a larger landscape and to model this relationship to extend it on large scale.

Rüdisser et al. (2012)’s methodology already combined most of the aforementioned advantages. Therefore, a prototypal methodology based on Rüdisser et al. (2012) was developed (Figure 3.5). From the sources mentioned in the first grey rectangle, different variable types related to the quantification of anthropogenic effects on landscape can be acquired (see Figure 3.4). In this way, even if few large scale social or infrastructure inquiries were performed, proxies can be found by relying on other DPSIR stages. This should ease the methodology application to more remote areas or where few data are available, like developing countries (Vranken et al., in preparation-a).

Appropriate georeferenced categorical and stretched (continuous) values can be crossed, at a given period of time or following a time series. In the latter case, post-classification change detection maps may help distinguishing individual habitat patches following or resulting from different (anthropogenic or natural) dynamics. A simple example of post-

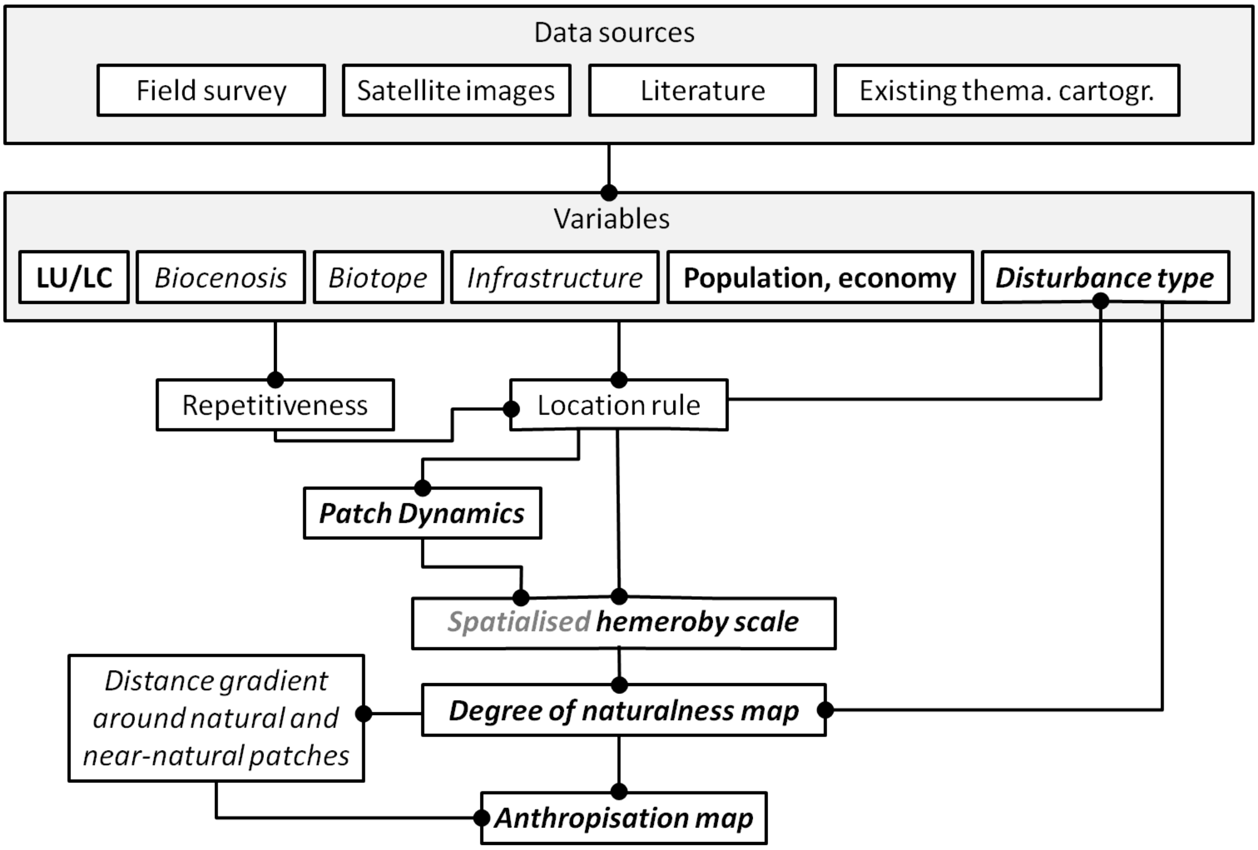

Figure 3.5 – *Proposal of a new methodological framework to spatially quantify the anthropisation. Bold characters represent categorical maps, italic characters represent stretched values, bold italic character represent patch (dynamics) data. LU/LC: Landuse / Landcover*

classification change detection to refine the categorical mapping towards patch dynamics is the way to distinguish an old fallow from a woodland in Katanga. Using remotely sensed images of good classification precision but coarse thematic resolution, distinguishing forest covers and herbaceous vegetation (grassland, savannah or crops) at different periods of time in the same area, it is possible to distinguish the forest patches that were previously herbaceous vegetation (old fallow) from the forest patches that were already forest before. In that case, the temporal resolution should be adapted to the phenomenon studies, here tree growth and slash and burn agriculture.

Infrastructure information can also help identifying the type of disturbance encountered by a given habitat patch, distinguishing it from other patches from the same land cover. Other disturbance information, such as fire (extent, frequency etc.), can be directly acquired from satellite imagery. Sorting this information, a discontinuous hemeroby scaled map can be defined. Compared to classical hemeroby scales, such data will contain additional location information. Different methods to cross continuous data with categorical maps can be applied. As a first example, during the classification process, training sets can be defined on several (categorical or continuous) layers at the same time, such as (stretched) altitude gradient maps with multispectral satellite image of the same area. In that case, the classification will be performed according to the combination of attributes from the aforementioned data layers. As a second example, when the data are standardised, numeric combinations such as multiplication can be performed in order to combine them, such as multiplying one layer by the other, as performed in Rüdisser et al. (2012) or (André et al., in revision).

Along with stretched values representing other anthropogenic disturbances, a hemeroby map, called "degree of naturalness" by Rüdisser et al. (2012) can be designed. This map can represent continuous patch dynamics, being the product of specific patch dynamics information and continuous stretched values. Other continuous data are added in order to include habitat connectivity into anthropogenic effect assessment: distance to the most natural habitats, according to the map of degree of naturalness. Crossing this information with the aforementioned map gives the final output: the anthropisation map (see (André et al., in revision)).

3.7 Conclusion

Anthropisation concepts and variables refer to system states, pressures and ecological impacts without distinction, while its quantification strongly depends on the reference states, which are practically complex to determine for specific sites. Anthropisation is quantified with direct, but partial measures, qualitative scales or composite indexes. Data are formatted as discrete categorical maps or continuously stretched values (gradients).

We propose a new reference state corresponding to the currently most ambitious restoration objective or the currently most preserved area: managed indigenous naturalness. We propose guidelines towards a new, action-oriented and composite methodology that distinguishes individual patch characteristics and dynamics while relying on easily accessed data. The key of this framework is to combine categorical mapping with stretched values analysis and diachronic observation.

CHAPTER 4

A review on the use of entropy in landscape ecology: heterogeneity, unpredictability, scale dependence and their links with thermodynamics

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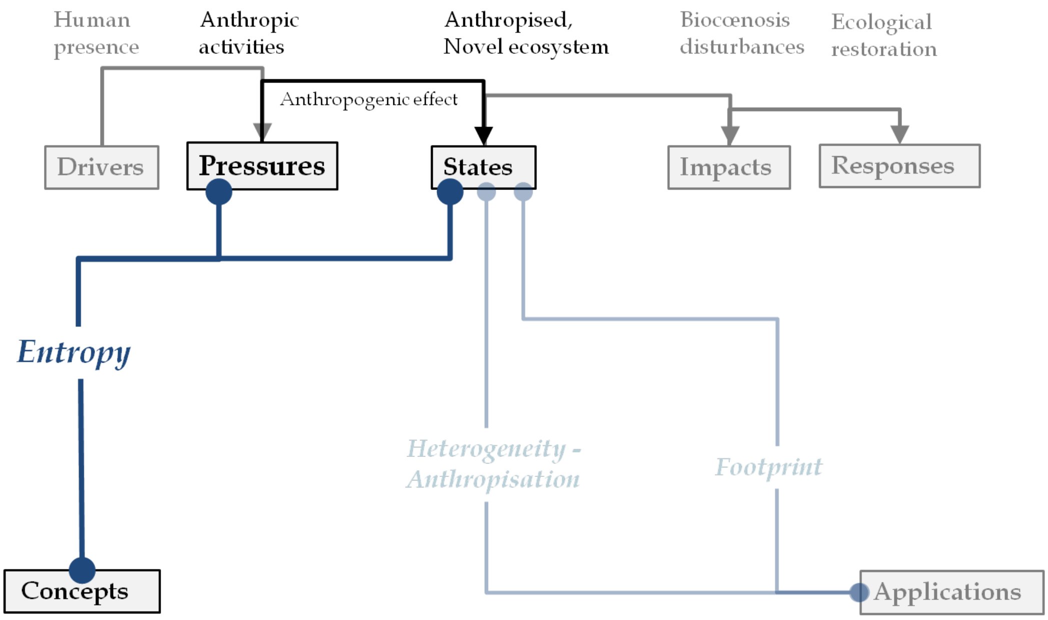

Figure 4.1 – *Location of the entropy article (deep blue theme) in the thesis mind map. Technical reading: concepts, thematic reading: heterogeneity (question of Section 1.2.2), logical reading: pressures and states. The hollow blue articles are thematically related to this one. This chapter evaluates the thermodynamic connection with popular topics in landscape ecology referred to with the term entropy: spatial heterogeneity, unpredictability of pattern dynamics and scale influence on pattern assessment.*

4.1 Abstract

The identification of a universal law that can predict the spatiotemporal structure of any entity at any scale has long been pursued. Thermodynamics have targeted this goal, and the concept of entropy has been widely applied for various disciplines and purposes, including landscape ecology. Within this discipline, however, the uses of the entropy concept and its underlying assumptions are various and are seldom described explicitly. In addition, the link between this concept and thermodynamics is unclear. The aim of this paper is to review the various interpretations and applications of entropy in landscape ecology and to sort them into clearly defined categories. First, a retrospective study of the concept genesis from thermodynamics to landscape ecology was conducted. Then, 50 landscape ecology papers that use or discuss entropy were surveyed and classified by keywords, variables and metrics identified as related to entropy. In particular, the thermodynamic component of entropy in landscape ecology and its various interpretations related to landscape structure and dynamics were considered. From the survey results, three major definitions (i.e., spatial heterogeneity, the unpredictability of pattern dynamics and pattern scale dependence) associated with the entropy concept in landscape ecology were identified. The thermodynamic interpretations of these definitions are based on different theories. The thermodynamic interpretation of spatial heterogeneity is not considered relevant. The thermodynamic interpretation related to scale dependence is also questioned by complexity theory. Only unpredictability can be thermodynamically relevant if appropriate measurements are used to test it.

Keywords: Information theory, Spatial heterogeneity, Pattern dynamics, Scale influence, Complexity, Resilience

4.2 Introduction

The search for a universal law that can apply from physics to social sciences has been challenging scientists since the rise of reductionist theories. In this sense, the laws of thermodynamics are expected to explain any process at any scale (Li et al., 2004). Therefore, the term entropy is now used in a variety of disciplines. However, in Landscape ecology, numerous interpretations and uses are associated with entropy: as a pattern or processes descriptor, with or without reference to thermodynamics.

In addition, the current thermodynamic interpretations of landscape entropy can be questioned by complexity theory (Li, 2000a; Wu and Marceau, 2002), while the meaning of entropy is often not discussed, or even mentioned, when used in landscape ecology (Bolliger et al., 2005). The interpretations of entropy in landscape ecology (landscape entropy) can even be contradictory because entropy can be associated with chaos or the opposite depending on the interpretation. This ambiguity causes confusion in using and interpreting metrics related to entropy.

Therefore, this review aims to distinguish amongst the various applications of landscape entropy and analyse their consistency. We address four questions: (1) What are the links between landscape entropy and the origins of the concept? (2) How can we quantify landscape entropy? (3) What are the relevant interpretations of entropy? (4) Can thermodynamics predict landscape spatio-temporal structure?

To answer these questions, we first explore the origin of the entropy concept in thermodynamics, explain how a parallel concept developed in information theory, and discuss the link between these two origins. We then describe how these two concepts were adopted in ecology and, through a bibliographic survey, examine how various interpretations evolved in landscape ecology. Finally, we explore the various metrics used to quantify entropy. The discussion questions the validity of the thermodynamic interpretation of landscape entropy according to complexity theory and briefly examines the relevance and limitations of the metrics. We conclude with recommendations for a transparent use of the term.

4.2.1 Origins: thermodynamics

The notion of entropy originates from classical thermodynamics: it was developed by Clausius in 1850 as a system state function (Figure 4.2). Entropy was originally used to quantify the degree of irreversibility of a thermodynamic transformation in an isolated system. Indeed, according to the second law of thermodynamics, a system spontaneously evolves towards the thermodynamic equilibrium, that corresponds to its maximal entropy level; hence, the entropy of an isolated system increases with every transformation it undergoes (Benson, 1996; Benatti, 2003; Harte, 2011). As entropy was not measurable *per se* in the mid-nineteenth century, it was defined by its variation during a theoretically reversible transformation within a closed system, as in equation 4.1:

$$D_s = \frac{D_Q}{T} \tag{4.1}$$

where D_S is the entropy variation (Joules per Kelvin), D_Q is the heat transfer between the system and its surroundings, and T is the equilibrium temperature (Harte, 2011).

In approximately 1875, Boltzmann formulated a probabilistic interpretation of the second law of thermodynamics using atomic theory (Benson, 1996; Harte, 2011). He introduced the macrostate and microstate concepts. The former concept describes the general state of a system at a macroscopic level, characterised by state functions (e.g., temperature, pressure, volume). The latter concept takes the configuration of each system element (position and movement of each particle) into account. Boltzmann demonstrated that one given macrostate could correspond to numerous different microstates and stated that, according to the second law of thermodynamics, a system spontaneously evolves to the most probable macrostate, i.e., the state that would result from the largest number of different microstates: the state of maximum entropy (Depondt, 2002; Harte, 2011). Following this approach, entropy was defined as follows:

$$S = k_B \log W \tag{4.2}$$

where S is the system entropy (J/K), k_B is the Boltzmann constant (1.38062 J/K), and W is the number of different microstates corresponding to a given macrostate.

The most probable macrostate has the most homogeneous (i.e., undifferentiated or uniform) configuration (Benson, 1996; Harte, 2011). Boltzmann justified the correspondence with Clausius' theory stating that an isolated system spontaneously loses its structure and becomes a homogeneous mixture of all its molecules (Forman and Godron, 1986b; Benson, 1996; Harte, 2011). This definition explains why entropy is associated with disorder, in contrast to a differentiated structure in which the various elements would be sorted into separate locations instead of being evenly distributed.

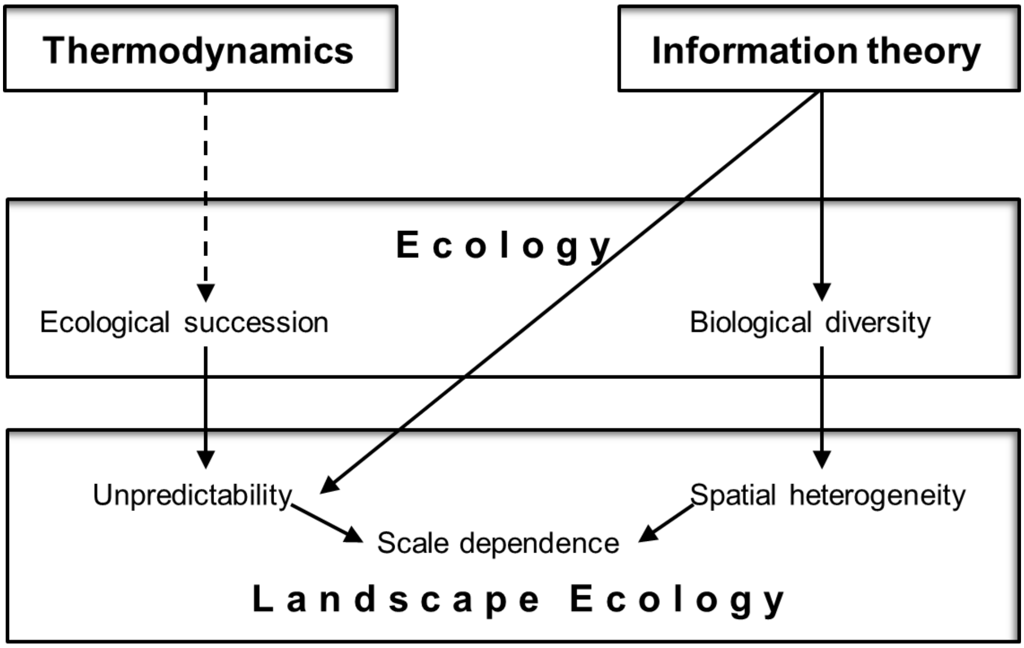

Figure 4.2 – *Summary of the development of the entropy concept and relevant relationships from its foundations to landscape ecology. The arrows indicate the sense of the evolution. No arrow is drawn from thermodynamics to information theory since our analysis concluded that there was no relationship. The dashed arrow shows that the thermodynamic interpretation of ecological succession needs to be further investigated. It will influence the relevance of the thermodynamic interpretation of unpredictability and scale influence*

4.2.2 Parallel development in information theory

An alternative use of the term entropy was developed in 1948 by Claude Shannon for information theory (Figure 4.2). Shannon studied the way information contained in messages such as telegrams was degraded during transmission (Shannon and Weaver, 1948; Harte, 2011). In this context, entropy (H) is defined as follows:

$$H = -\sum_{i=1}^{n} p_i \log p_i \tag{4.3}$$

where n is the number of elementary message components (i) and p_i is the probability of the occurrence of each form this component can assume. H varies from 0 to $\log n$ (Shannon and Weaver, 1948).

Here, information is a function of the ratio between the number of possible contents before and after the information is received (Margalef, 1958). Entropy represents the missing information, i.e., the amount of information that could be gained by receiving a supplementary message component (Shannon and Weaver, 1948; Margalef, 1958; Harte, 2011). Entropy production represents information loss during signal transmission (Moran et al., 2010); this metric is used to determine the degree of redundancy that is required in the message in order to preserve its (unaltered) meaning upon reception (Depondt, 2002). The more elaborate the emitted signal and the less information contained in the received signal, the higher the entropy production (Moran et al., 2010). Indeed, if the emitted signal is elaborate (n is high and the various p_i are small), a single message component i provides only a small amount of the total message content. Negentropy, the inverse of entropy, is the information contained in the received message, i.e., the degree of organisation (Margalef, 1958; Harte, 2011).

A functional connection between thermodynamics and information theory can be drawn. Indeed, for an ideal gas, considering i as the number of groups of molecules with the same properties, there is a larger number of possible spatial arrangements of i (the number of microstates) when the various p_i are smaller and more numerous; the Boltzmann equation (4.2) is directly proportional to the Shannon equation (4.3). Therefore, information entropy (entropy as applied in information theory) is a measure of our confusion regarding the state of the system (Shannon and Weaver, 1948; Benatti, 2003; Harte, 2011). Stonier (1996) even asserted that energy and information were interconvertible. Shannon himself noticed this similarity in his research (Shannon and Weaver, 1948), even though his theory was not derived from Boltzmann's theory (Benatti, 2003).

However, the hypothesis of a physical correspondence between thermodynamic and information entropies is questionable for three reasons. First, thermodynamic entropy (S) depends upon the various microstates of a system at the molecular level, which is more strongly related to the law of large numbers (Sanov, 1958). In information theory, the number of possible message components varies across an entirely different range (Depondt, 2002; Benatti, 2003; Maroney, 2009). Second, despite the use of the same formalism, information entropy and thermodynamics are based on clearly divergent theoretical assumptions: according to Boltzmann, entropy corresponds to a homogeneous structure, while Shannon's entropy corresponds to elaborate (heterogeneous) signals (Ricotta, 2000; Depondt, 2002; Harte, 2011). Third, considering classical thermodynamics, if transformation irreversibility, which is fundamental to entropy variation, can correspond to the irreversible degradation of the received signal, a redundancy in the message content that allows for the reconstruction of the message meaning does not make sense for thermodynamic entropy (Depondt 2002). In conclusion, there is no confirmation that any thermodynamic interpretation of information theory is relevant. Information entropy is, therefore, merely a formal parallelism to thermodynamic entropy (Renyi, 1961).

4.2.3 Thermodynamics and information theory applied to ecology

Ecologists rapidly applied information entropy to assess biological diversity with the Shannon diversity index (MacArthur, 1955; Margalef, 1958; Ulanowicz, 2004). Later, the development of the analysis of landscape heterogeneity was based on those metrics (Romme, 1982). Diversity is understood as the interaction between the number of species and their relative abundances. Diversity represents the probability that two individuals sampled at random will not belong to the same species (Pielou, 1975). The majority of information entropy-related metrics are derived from the Shannon index (3), where, p_i represents the relative abundance of individuals of species i in an ecosystem containing n species. Before Stonier (1996), Margalef (1958) proposed that information theory as applied to ecology and evolution could have a thermodynamic meaning. However, ecological thermodynamic interpretations do not refer to information theory (Wurtz and Annila, 2010; Chakraborty and Li, 2011). Pielou (1975) even highlighted the absence of an ecological meaning of information theory. In contrast, authors currently referring to information theory do not generally refer to thermodynamics (Ulanowicz, 2004), with a few exceptions, such as Ricotta (2000).

As for the thermodynamic heritage of entropy in ecology (Figure 4.2), it is mainly used to describe the evolution of a food web through ecological succession (Wurtz and Annila, 2010). Ecosystems, like living organisms, can be described as dissipative structures, i.e., open systems that consume available energy. These systems are non-equilibrium (Li et al., 2004) : the possible equilibrium state achieved by the ecosystem is not the thermodynamic equilibrium but, rather, a situation of stability ("metastability", "dynamic equilibrium", "homeostasis" or "stationary state") in the ecosystem structure (Yarrow and Salthe, 2008; Parrott, 2010; Chakraborty and Li, 2011; Ingegnoli, 2011). This energy originates more or less directly from the sun (external source) and is transformed by organisms at various trophic levels following non-linear dynamics. This process is associated with an entropy increase outside the system and an entropy decrease inside the system. This phenomenon is called self-organisation, an emergent property in complex systems (Li, 2000a; Li et al., 2004; Green and Sadedin, 2005; Parrott, 2010; Chakraborty and Li, 2011). Various studies have examined the roles of the aforementioned processes in determining biodiversity and resilience, which can be related to entropy in terms of species diversity (Parrott, 2010; Chakraborty and Li, 2011), but these studies do not explicitly establish such a link. During the ecological succession process, ecological communities are highly effective at minimising the entropy increase through the food web (Wurtz and Annila, 2010; Hartonen and Annila, 2012). Note that this type of entropy use in ecology, though described using thermodynamic equations, does not provide any quantitative measurements of entropy production and energy fluxes (Maldague, 2004), most likely because such measurements would require considerable infrastructure (Ulanowicz, 2004). It has even been stated that classical thermodynamics cannot predict the evolution of ecological systems because the latter are non-equilibrium systems that follow non-linear dynamics (Li, 2000a; Li et al., 2004; Ulanowicz, 2004).

4.3 Methods

This review consists of a quantitative survey on the uses and interpretations of entropy in landscape ecology based on a selection of representative papers. A bibliographic search was based on journal articles, conference proceedings and books published or in press in 2012 according to a joint database search in ScienceDirect, Scopus and Google Books. To select articles applying or discussing entropy concepts in landscape ecology, research filters were applied on the term "entropy" in the full text and "landscape ecology" in the full text or "landscape" in the source title. As Google Books did not provide these search tools, only books with "landscape ecology" in their subject and "entropy" as well as "land" or "landscape ecology" in the full text were selected. This search resulted in 297 publications: 215 papers from Science- Direct, 60 from Scopus (11 in common with ScienceDirect), and 37 books from Google Books. Papers citing entropy only in the references and book reviews were excluded. Fewer than 200 papers remained after this filtration. Fifty of these were selected as the most representative papers, i.e., the journal articles published in the highest impact factor journals and the most cited books or conference proceedings according to Google Scholar. This selection was performed with the goal of encompassing the widest possible range of metrics and interpretations of entropy in landscape ecology.

Each selected document was analysed regarding the interpretation, use and metrics of landscape entropy. The results were listed, and similarities were grouped for further description and comparison. The representativeness of each quantification method was studied, and the documents were classified according to the interpretation of entropy and its links with thermodynamics.

4.4 Results

Amongst the various discussions, metrics and uses contained in the 50 selected papers, three interpretations of landscape entropy could be distinguished: (1) spatial pattern heterogeneity, (2) unpredictability of pattern dynamics and (3) scale dependence of spatial and temporal patterns (Figure 4.2, Table 4.1). The interpretations mentioning a thermodynamic relationship generally describe processes in a qualitative way, whereas the non-thermodynamic interpretations are quantitative and describe patterns. As the same quantification methods can be used in various ways, these methods are presented in a separate subsection.

4.4.1 Entropy in space: heterogeneity

The use of entropy concepts to quantify landscape heterogeneity was reported in half of the selected references (Table 4.1), although a link with thermodynamics was rarely discussed. In these papers, entropy represents the intricacy of the landscape pattern, either compositionally (numerous land covers present in even proportions) or configurationally (numerous patches of tortuous forms) (Fahrig and Nuttle, 2005). This use of entropy was inherited from information theory. Within this interpretation, the majority of authors consider the link between entropy and thermodynamics as simply a formal parallelism, associating entropy with disorder, applied at the landscape level (Li and Reynolds, 1993; Joshi et al., 2006; Leibocivi, 2009). Zhang et al. (2006b) add that interactions within the landscape and with its surroundings cannot strictly be evaluated from a thermodynamic point of view, though a few authors provide thermodynamic interpretations (Table 4.1).

There are two opposing thermodynamic interpretations of heterogeneity (Table 4.1). In the case of a direct correlation, higher heterogeneity would mean higher entropy (Bogaert et al., 2005). This interpretation is based on information theory applied at the landscape level: entropy here means a signal as well as landscape heterogeneity. For the authors considering an inverse correlation, higher entropy corresponds to higher homogeneity. This homogeneity is understood as an undifferentiated structure covering the entire landscape (the "macrostate"), following Boltzmann's theory (see Section 4.2.1) applied at the landscape level (Forman and Godron, 1986c; Benson, 1996; Harte, 2011). Spatial heterogeneity is used indirectly to assess species distributions (Cale and Hobbs, 1994; Farina, 2000a; Johnson et al., 2001; Cushman and McGarigal, 2003; Tews et al., 2004; Fahrig et al., 2011), to assess the effects of disturbances such as urban sprawl (Sudhira et al., 2004; Rahman et al., 2011) or habitat loss through fragmentation (Wilkinson, 1999; Tews et al., 2004; Fahrig et al., 2011).

4.4.2 Entropy in time: unpredictability

The concept of entropy is also used to describe the instability of landscape evolution. Two approaches are employed for this application. The first approach applies a thermodynamic interpretation of unpredictability (Table 4.1) and aims to describe landscape evolution in energetic terms. Here, as with ecosystems in ecology, landscapes are considered as dissipative structures composed of living organisms that consume energy from the sun directly or indirectly to increase their inner structure (and thus decrease their inner entropy) while the entropy of their surroundings increases (McHarg, 1981; Naveh, 1982; Li, 2000a; Leuven and Poudevigne, 2002; Zhang et al., 2006b; Gobattoni et al., 2011). In the (theoretical) absence of a disturbance, landscapes tend to evolve towards a condition of metastability over time, and the dissipative processes only maintain the inner structure of the landscape

Table 4.1 – *Survey on 50 journal articles and reference works within the scope of landscape ecology (see references section for detailed list) according to entropy type (columns) and explicit mention of the link with thermodynamics (rows). The figures represent the number of articles fitting each category. The total exceeds 50 because two references (Li, 2000a; Bolliger et al., 2005) refer to spatial heterogeneity and unpredictability, while Johnson and Patil (2007) refers to spatial heterogeneity and scale dependence.*

	Spatial heterogeneity	Unpredictability	Scale dependence
No Thermodynamic relationship	25	5	5
Thermodynamic relationship	3	13	1

system. In this case, the inner entropy decrease compensates for the entropy increase in its surroundings (Naveh, 1987; Li, 2000a; Ingegnoli, 2011). Several authors have associated any instability ("transition phase") and change with an increase in outer entropy production (Dorney and Hoffman, 1979; McHarg, 1981; Lee, 1982; Corona, 1993; Wilkin, 1996; Newman, 1999).

In contrast, the second approach consists of estimating unpredictability by applying information entropy metrics to landscape patterns. This approach is essentially quantitative and does not refer to thermodynamics, instead employing an analogy between signal transmission and landscape structure evolution. From this perspective, entropy is referred to as unpredictability because the irregularity of landscape change is measured using entropy metrics. This view depicts the evolution of spatial patterns, or biophysical gradients, such as those in meteorological data or the Normalised Difference Vegetation Index (NDVI). The data are generally obtained from remote-sensing image time series. Such time series can now be chosen according to the desired temporal scale of observation (Zaccarelli et al., 2013). The data are analysed using information theory-derived metrics and interpreted in relation to disturbance and stability (Mander and Jongman, 1998; Martín et al., 2006; Zaccarelli et al., 2013; Zurlini et al., 2013). Some of the authors following this approach have criticised the thermodynamic approach to assessing unpredictability, stating that a change in the energy, matter and information fluxes between a landscape system and its surroundings does not necessarily imply unpredictability because the system may return to its previous metastable state (see "Thermodynamics and information theory applied to ecology", Section 4.2.3) after such a disturbance (Zaccarelli et al., 2013).

According to both approaches, however, a minor disturbance temporarily interrupts the stationary state. When there is a high level of landscape resistance, the landscape spatial structure is not modified and the landscape processes progressively return to their previous state. Alternatively, when there is a high level of landscape resilience, the landscape spatial structure can be modified but progressively returns to the same stationary state as before the disturbance (Pimm, 1984; Ingegnoli, 2011). Unpredictability arises when a disturbance is sufficiently severe to disrupt the within-landscape processes and patterns to a degree that the system resilience cannot overcome the disturbance. This state is described as transitive and is linked with the terms "phase transition", "bifurcation", "perturbing transitivity", "critical threshold", "severe outside disturbance" or "instability" (Naveh, 1987; Li, 2000a; Li et al., 2004; Ulanowicz, 2004; Zhang et al., 2006b; Chakraborty and Li, 2011; Ingegnoli, 2011). At this stage, it is not possible to predict the new metastable state into which the landscape will evolve, either thermodynamically or structurally. The level of unpredictability can be used to assess landscape resilience under various types of pressures, including those caused by humans (Naveh, 1987; Zaccarelli et al., 2013; Zurlini et al., 2013).

4.4.3 Entropy over space and time: pattern scale dependence

The use of entropy concepts to study the effect of scale on spatial and temporal patterns is the least frequent usage (Table 4.1) (O'Neill et al., 1989; Riitters et al., 1995; Johnson et al., 2001). This usage emerged in the literature shortly after the linkage of entropy to heterogeneity and unpredictability. Typically, this approach examines irregularities in pattern measurements across a gradient of scales by employing disorder metrics derived from information theory (Johnson et al., 1999). Decreasing the spatial resolution can obscure ecologically relevant contrasts along ecological gradients such as rainfall distribution or species abundance, since this can influence the shape and size of habitat patches and merge or even erase patches when their sizes are smaller than the pixel or when they cross multiple pixels (Turner, 1989). This scale dependency may have an important influence on the identification of patterns and, therefore, on inferences of underlying ecological processes (Cale and Hobbs, 1994).

Only one paper studying landscape entropy as pattern scale dependence mentioned thermodynamics (O'Neill et al., 1989). In this context, scale dependence is discussed in terms of hierarchy and complexity theory (Wu and Marceau, 2002; Li et al., 2004; Green and Sadedin, 2005). It should be noted however that Cushman et al. (2010) highlighted the difference between considerations of scale and hierarchical levels: scale refers to a continuous property measured in common units, whereas hierarchical level refers to a discrete property with various entities studied at each level. Hierarchy theory states that the existence of emergent properties that arise from non-linear interactions of the components of a system with each other and with external constraints prevents the prediction of system behaviour when only considering the properties of its components (Wu and Marceau, 2002; Li et al., 2004; Green and Sadedin, 2005). Hence, the inner and outer constraints applied on the studied system need to be described. In landscape ecology, the level of focus is the landscape. The immediately higher level, the system environment or surroundings, is the region; this level represents the outer constraints encountered by the landscape. The immediately lower level, the components or holons (sub-systems) of the landscape, are the ecosystems (Wu and Marceau, 2002; Ingegnoli, 2011). According to O'Neill et al. (1989), entropy is considered in terms of the laws of thermodynamics applied on living systems at various levels, recognising that living systems spontaneously tend towards minimal entropy production.

We stress here that the study of scale dependence in landscape ecology extends beyond the sole usage of the term entropy within this framework. The majority of the research studying this issue is conducted within the framework of complexity theory but does not refer to thermodynamics. This research highlights the influences of scale and hierarchical levels of observation on the explanatory power of observed patterns and processes (Levin, 1992; Wu and Marceau, 2002; Green and Sadedin, 2005; Yarrow and Salthe, 2008; Cushman et al., 2010).

4.4.4 Quantification methods

The present section provides a short description of the indexes that are referred to as "entropy indexes" in landscape ecology. As previously explained, these indexes are not connected with thermodynamics. Only the fundamental quantification methods (Figure

4.3) are presented, beginning with those inherited from ecology. Note that the majority of these metrics are included in the Fragstats landscape pattern analysis software (McGarigal et al., 2012).

4.4.4.1 Metrics inherited from ecology

The oldest and still most widely used metrics are the Shannon index and its analogous forms (Antrop, 1998; Antrop and Van Eetvelde, 2000; Palang et al., 2000; Antrop, 2004). When evaluating compositional or configurational heterogeneity, p_i in Table 4.2, equation (4), represents the areal proportion of either the land cover or patch i (Yeh and Li, 1999; Carranza et al., 2007). When evaluating unpredictability, p_i can also represent the proportion of an ecological factor broken down in classes, such as *NDVI*, precipitation or the distance from a town. The latter is calculated over time rather than as a spatial series. The Simpson and Shannon diversity indexes are analogous. The Simpson diversity index (Table 4.2, equation (5)) is used in statistics and ecology. This index considers relative abundances, as does the Shannon index, but it is computed using the arithmetic mean rather than the geometric mean and is normalised (Pielou, 1975). The Brillouin index (Table 4.2, equation (6)), used in physics and ecology, is used to evaluate the diversity of a fully censused area, whereas the Simpson and Shannon indexes are better suited for samples (Pielou, 1975; Orloci, 1991; Bogaert et al., 2005).

Those metrics were later grouped into a generic form: the Renyi generalised entropy index (Table 4.2, equation (7)), (Renyi, 1961; Pielou, 1975). By this definition, $0 < \alpha < \infty$, and according to its value, the Renyi index may correspond to one of the above-cited indexes. The value of H_1 equals the Shannon index, while H_2 is a logarithmic version of the Simpson index (Pielou, 1975). The Brillouin index can also be approached using this formula (Orloci, 1991). Evenness (Table 4.2, equation (8)) is a component of diversity that considers only the relative abundances of the measured elements (Pielou, 1975; Forman and Godron, 1986b; Johnson et al., 1999; Martín et al., 2006; Proulx and Fahrig, 2010).

Conditional entropy, also associated with the Shannon and analogous indexes, (Table 4.2, equations (4) to (8)), is used to measure the unpredictability or scale dependence of the spatial heterogeneity of a landscape. When the degree of entropy is known to be partially attributed to a known random variable, conditional entropy is the entropy conditioned by the other random variables (Shannon and Weaver, 1948; Legendre and Legendre, 2012).

This concept is based on conditional probability studies (Shannon and Weaver, 1948; Jost, 2006; Martín et al., 2006). The p_i variables in equation 4.3 are then spatial pattern indexes themselves that are applied to an entire landscape at varying times or resolutions i (Pablo et al., 1988; Patil et al., 2000; Martín et al., 2006). A particular use of conditional entropy and the Shannon index for the measurement of unpredictability has also been proposed: normalised spectral entropy. It integrates the frequency at which a certain pattern or gradient can be recovered in a time series and its Fourier power spectrum (Johnson et al., 1999; Zaccarelli et al., 2013; Zurlini et al., 2013).

Another application of conditional entropy was adapted from statistics to ecology and landscape ecology to measure the a, b and c diversities (Table 4.2, equation (9)). These indices evaluate the mean habitat diversity at the landscape level (a) and the differences between distinct thematic layers (b) representing the landscape (e.g., land cover, human activities, soil types); $\alpha + \beta = \gamma$ (Pablo et al., 1988; Ernoult et al., 2003). These metrics are calculated using Shannon derivatives (Shannon and Weaver, 1948; Whittaker, 1960; Jost, 2006; Wurtz and Annila, 2010).

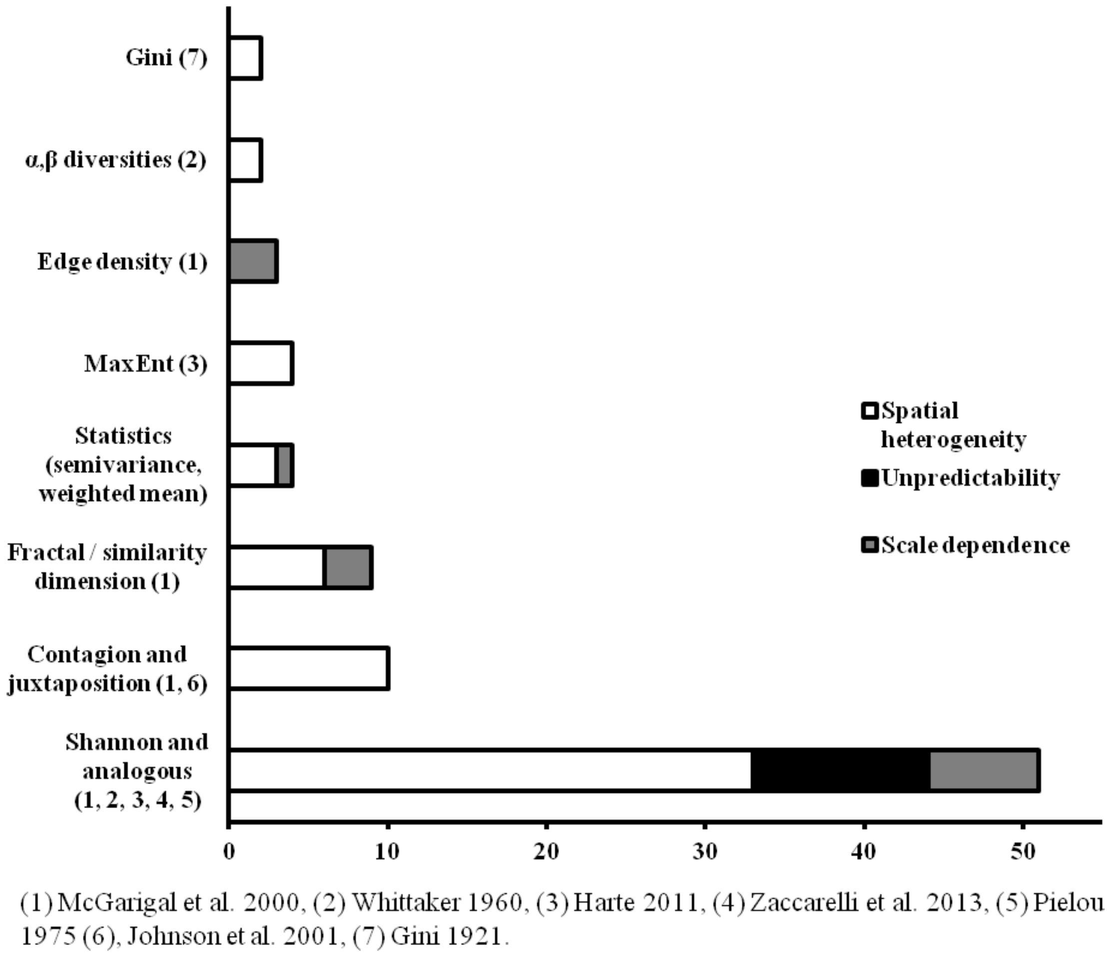

Figure 4.3 – *Number of citations and use of the most cited (at least twice) entropy quantification methods in landscape ecology, classified by entropy type. According to a survey of 50 journal articles, conference proceedings and reference books within the scope of landscape ecology (see references section for detailed list). Statistics are not referenced as they can represent any kind of general statistics. Shannon and analogous include Shannon, Simpson, diversity, evenness and conditional entropy.*

As a particular case, the MaxEnt (Maximal Entropy) method uses information regarding entropy without measuring it for its own purpose. Inherited from ecology, the versions adapted for landscape ecology assess geographic distributions of species or habitats, based on a sample, by finding the most uniform distribution subject to the applied constraints (i.e., the measured distribution parameters) using Lagrange multipliers on Shannon indices (Phillips et al., 2006; Powell et al., 2010; Harte, 2011).

4.4.4.2 Metrics developed in landscape ecology and non-ecological disciplines

With regard to landscape entropy indexes not inherited from ecology, contagion indexes are the most widely employed (Table 4.2; Figure 4.3). Contagion indexes measure configurational heterogeneity or, more precisely, the spatial distribution and intermixing of patch types (Riiters et al., 1996). These indexes are also applied to landscapes at various scales and compared (Johnson et al., 1999; Gaucherel, 2007) but are not indexes of scale dependence *per se* (Benson and Mackenzie, 1995). Contagion represents the relative importance of adjacencies between pixels of different patch types in a landscape, hence the

Table 4.2 – *Summary of the main landscape quantification methods referring to entropy, their computation and main characteristics. The grey zone comprises metrics not inherited from ecology.*

Name	Formula	Note	Equation number
Shannon (diversity)	$H_a = -\sum_{i=1}^{n} p_i \log p_i$	Conditional entropy: related to Renyi's formula	(4)
Simpson (diversity)	$H_b = 1 - \sum_{i=1}^{n} p_i^2$	Conditional entropy: related to Renyi's formula	(5)
Brillouin (diversity)	$H_c = \log_2 \frac{f.!}{f_1!f_2!\ldots f_n!}$	$f.! = (f1! + f2! + \ldots + fn)$ conditional entropy: related to Renyi's formula	(6)
Renyi	$H_\alpha = \frac{1}{1-\alpha} \ln \sum_{i=1}^{n} p_i^\alpha$	Generalised entropy formula (diversity): conditional entropy	(7)
Evenness	$E = \frac{H}{H_{\max}}$	Conditional entropy: related to Renyi's formula	(8)
α, β, g diversity	$\alpha + \beta = \gamma$	computed with Shannon derivatives	(9)
MaxEnt	Methodology: see Harte (2011)	Sample-based habitat distribution assessment	
Contagion, juxtaposition	*Large amount of metrics, see McGarigal et al. (2012)*	*Contagion for raster. juxtaposition for feature maps*	
Fractal, similarity dimension	$\log P = \log k + \frac{D}{2} \log A$	*D found using a linear regression between log A and log P (see*	*(10)*
Edge density	*See McGarigal et al. (2012)*	*Configurational heterogeneity*	
Gini	*See Gini (1921)*	*"Unevenness" index*	

level of aggregation of the patch types. The most common contagion index is the Shannon contagion index (McGarigal et al., 2012). Juxtaposition represents the relative importance of the common edge lengths between patches and also uses an adaptation of the Shannon index (Johnson and Patil, 2007; McGarigal et al., 2012).

Fractal dimension is a widely used measure of patch shape tortuosity that is reported to relate to entropy (Figure 4.3) (Kenkel and Walker, 1996). Whereas a patch edge has one topological dimension and a surface has two, the fractal dimension of its edges varies from 1 (straight line) to 2. The fractal dimension approaches 2 if the shape tortuosity is sufficiently important that the edge can fill a surface (Mandelbrot, 1983; Kenkel and Walker, 1996). The fractal dimension of a single patch cannot be properly calculated, for multi-scalar information is then not available (Krummel et al., 1987). Therefore, a linear regression of equation (9) (Table 4.2) is used to calculate the fractal dimension of a population of patches, which should preferably be of similar shapes (Krummel et al., 1987), where, k is an unknown constant, D is the fractal dimension, P is the patch perimeter, and A is the patch area. In our survey, the most recent use of the fractal dimension was reported in 2007, compared to 2013 for the Shannon index and the MaxEnt method. A variant, the similarity dimension, evaluates scale dependence (Patil et al., 2000). This index is also frequently used to assess fragmentation, but the majority of interpretations do not explicitly mention entropy, and the various calculations are still debated (Krummel et al., 1987; Turner, 1989; Li, 2000b; Halley et al., 2004).

Several other metrics are less frequently used for studies of landscape entropy (Figure 4.3). There are simple indexes such as the Largest Patch Index (LPI), and edge density (Johnson and Patil, 2007; McGarigal et al., 2012) and statistics such as semivariance (Ernoult et al., 2003). A number of metrics have been adapted from other disciplines, e.g., the Gini Coefficient from social statistics (Gini, 1921). This latter metric is used and interpreted similarly to Shannon-based indexes (Jaeger, 2000; Kilgore et al., 2013).

4.5 Discussion and conclusion

The theoretical research on the evolution of the entropy concept from its origins to landscape ecology (Figure 4.2) has revealed that the various interpretations of this term are not consistent. The ways the term entropy is used in thermodynamics and in information theory do have functional similarities, but these concepts represent different realities; the term is used more as a formal analogy than as a physical correspondence.

4.5.1 Spatial heterogeneity: contested thermodynamic correspondence

Reconsidering the links between thermodynamics and spatial heterogeneity, the existence of two opposing interpretations must be considered. The majority of authors implicitly assume an analogy to (particle) disorder that works well: a higher degree of landscape entropy reflects greater spatial heterogeneity. This analogy can be observed at various levels, such as in terms of species or habitat diversity, but the authors employing this interpretation do not necessarily confer thermodynamic properties to the systems they study: greater spatial heterogeneity does not necessarily indicate greater (or less) thermodynamic entropy.

However, the most detailed link between spatial heterogeneity and entropy is the inverse correlation proposed by Forman and Godron (1986b) based on conceptual arguments that connect the Boltzmann equation to landscape patterns and dynamics (Forman and Godron, 1986b; Forman, 1995; Depondt, 2002; Harte, 2011). Notably, this link is also present in the different forms that spatial heterogeneity metrics can assume: high variability in the metrics values is generally measured for landscapes with the intermediate levels of class dominance and aggregation (Neel et al., 2004).

Forman's conceptual framework of the production of heterogeneity through dissipative structures assumes that pattern properties that exist at the particle level are the same at the levels of the living organism, habitat and landscape (Forman and Godron, 1986b). However, this assumption can be questioned based on the complexity and hierarchy theories described above: the processes, patterns and entities at play are not the same at any hierarchical level (Baas, 2002; Wu and David, 2002; Green and Sadedin, 2005). Employing the same logic as when evaluating the link between information theory and thermodynamics, this amalgam between the molecular and landscape levels appears inappropriate (O'Neill et al., 1989; Maroney, 2009). Indeed, the possible patch arrangements in a landscape are far less numerous than the number of possible microstates. Moreover, at various organisational levels, the time frames at which processes occur differ considerably. Moreover, neither irreversibility nor signal redundancy for reconstruction after transmission are possible when considering a strict correspondence between landscape spatial structure and (thermodynamic) entropy. Indeed, when landscape structure changes, for example, because of a disturbance, the landscape can, in certain instances, return to its previous structure as a result of resilience and ecological succession (see Section 4.4.2).

In addition, no (thermodynamic) entropy quantification methods have been proposed. Measurements of energy fluxes at the landscape level, which requires an enormous recording infrastructure, have been reported in rare cases, such as in Ryszkowski and Kędziora (1987), but, to date, no study has provided a sufficient level of integrative results to evaluate the link between spatial heterogeneity at the landscape scale and entropy. Therefore, any statement specifying a link, direct or inverse, between spatial heterogeneity and thermodynamic entropy should be treated with caution. The majority of the authors that use the term entropy when they mean spatial heterogeneity do not even mention a thermodynamic interpretation of entropy. Hence, in this context, the use of the term entropy may simply be language abuse.

4.5.2 Unpredictability: incomplete thermodynamic framework

Thermodynamic descriptions of landscape evolution in terms of unpredictability are more frequent than those in terms of spatial heterogeneity (Table 4.1). However, to date, none of these studies have been able to predict landscape stability or instability based on the production of entropy and energy exchanges (Li, 2000a; Ingegnoli, 2011). Such attempts are unlikely to succeed because, similar to ecosystems, landscapes are complex systems that exist in states that are far from equilibrium and exhibit non-linear dynamics (see Section 4.4.3); hence, landscape evolution cannot be described using classical thermodynamics (Li, 2000a; 2002; Li et al., 2004; Ulanowicz, 2004). In this case, unpredictability cannot merely be associated with an increase or decrease in entropy, whether within or outside of the landscape system. Even for a stable landscape, the production of entropy in the surroundings is higher than the entropy decrease within the landscape because of the irreversible transformations caused by the organisms; therefore, the exchanges described by Ingegnoli (2011) are irrelevant (Benson, 1996). Li (2000a) has reported that there have been numerous misinterpretations of landscape thermodynamics by (landscape) ecologists. Evolutionary processes and unpredictability have been described to an extent, but variations in entropy are not described in the case of phase transition or compared between old and new (meta)stable states.

Moreover, as the energy exchanges within the system and with its environment are not measured, knowledge of the manner in which thermodynamics are related to landscape dynamics requires further deepening. This shortcoming might be overcome by measuring the spatio-temporal variations of albedo in the infra-red channels of passive remote sensing images in order to test the aforementioned theories.

4.5.3 Scale, hierarchy and complexity: how thermodynamics fails to predict landscape spatio-temporal dynamics

The majority of the mentions of entropy as a measure of scale dependence did not refer to thermodynamics. Therefore, the use of the term entropy can be viewed as an abuse of language that has arisen from the use of metrics first used as entropy metrics in information theory. With regard to the relevance of a thermodynamic interpretation of the influence of scale, the measurement of energy exchanges at the level of matter or living organisms to infer thermodynamic behaviour at the landscape level appears inappropriate and insufficient because of the complexity of landscapes. Thermodynamic laws apply at every organisational level, but complex interactions within and amongst various levels imply that structures and processes are not self-similar across levels (Wu and Marceau, 2002). These discrepancies have two consequences. First, the measurement of such exchanges appears, in practice, unfeasible because of the number of interactions. Second, even if such a computation could be performed, a given set of departure conditions could generate various spatiotemporal structures (Green and Sadedin, 2005).

Currently, predicting the spatiotemporal structure of a landscape is better accomplished by studying the processes and interactions at the immediately lower (ecosystems) and higher (region) organisational levels to study the departure conditions and constraints applied to landscapes. Non-deterministic behaviours are also observed at those levels, especially because of the lack of predictability of human influences on landscape structure.

Therefore, simulation models are performed to evaluate trajectory scenarios (Green and Sadedin, 2005; Ingegnoli, 2011). Even if self-similar structures exist across multiple scales in nature, resulting from self-organised criticality and often displaying fractal patterns or distributions (e.g., in coastal geomorphology or in body size through a food web), these cases are particular (Baas, 2002; Wu and David, 2002; Green and Sadedin, 2005; Parrott, 2010).

Notably, perceptions of processes are strongly influenced by the observation scale, whether spatial or temporal. What seems unpredictable at a given spatiotemporal scale, e.g., the variation in albedo across a landscape during a year, can be predictable at a larger scale, e.g., when seasonal variations appear with more regularity across multiple years (Zurlini et al., 2013).

4.5.4 Metrics, terminology and insufficiencies

Such a contrast between the application of a concept's meaning and the use of its metrics is very unexpected. However, though thermodynamic issues are rarely addressed, numerous landscape entropy metrics have been proposed. Most of these metrics remain marginal (Figure 4.3): only three metrics appear to be commonly used. The Shannon index is clearly the most persistent and polyvalent. Its use and formula have evolved, but the interpretation still relies on the same basis (Renyi, 1961; Phipps, 1981; Ricotta, 2000; Johnson et al., 2001; Zaccarelli et al., 2013). This level of stability allows for comparisons amongst studies and may explain the success of this family of metrics. As a consequence of its wide use, the Shannon index has also been misused regarding its interpretation and its purpose (Pielou, 1975; Bogaert et al., 2005).

Contagion and juxtaposition indexes are the second most used landscape entropy metrics, though employed five times less frequently than the Shannon index and its analogous indexes (Figure 4.3). These indexes are mainly used to specifically address configurational heterogeneity (Ricotta et al., 2003; McGarigal et al., 2012).

With regard to the fractal dimension metric, it appears that its use has recently decreased, most likely because of the practical difficulty and lack of specificity of its calculation methods and the lack of relevance of its interpretation in terms of landscape entropy (Xu et al., 1993; Li, 2000b; Halley et al., 2004). The fractal dimension is, therefore, not recommended for assessing landscape entropy. Note that some authors identified fractal dynamics in interactions amongst system components and power-law scaling of frequency distribution features at various levels and linked it to thermodynamic processes in dissipative structures (Li, 2002). However, this interpretation is not related to the spatial structure. It is important to note that the aforementioned metrics do not include every existing heterogeneity, unpredictability and scale dependence metrics, but only those that were associated with the term entropy, and that these metrics use the term entropy for a non-thermodynamic representation.

As the term entropy is used in various ways with the same metrics and often without an explicit interpretation framework, we recommend using the term entropy with more accuracy and explicitness by employing the following three expressions. "Spatial heterogeneity" is proposed to describe the intricateness of the spatial pattern. "Unpredictability" should describe the irregularity in the pattern of change over time. "Scale dependence" should assess the effect of the spatial resolution on the observed patterns. The use of the expression "spatial heterogeneity" is already widespread in landscape ecology in the same sense. "Scale dependence" is used slightly less frequently but is often described by similar expressions: scale influence, influence of scale, or scale impact. The term "unpredictability" is still rarely used and no specific and unambiguous term is yet preferentially used to describe it, instead being referred to by the terms entropy, stability, persistence, or using sentences not suitable for a keyword search.

This quantitative survey highlights insufficiencies of the sampling methodology. Only searching for the mention of the term entropy did not allow for an understanding of its meaning, which required complementary research. Nevertheless, a lack of justification for the use and the interpretations of entropy in landscape ecology was revealed, as if the link between entropy and landscape dynamics was a well-established fact. This review demonstrates that this is not the case.

In addition, note that a change in entropy results from a process. In ecology, ecosystem process studies rarely explain pattern formations in landscapes (Levin, 1992; Cushman et al., 2010; Chakraborty and Li, 2011), while landscape ecology more often focuses on inferring the impact of spatial patterns on ecological processes than the opposite (Turner, 1989; Baudry, 1991). Therefore, there is still a gap to fill in the pattern/process paradigm.

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Part III

Application to spatial pattern assessment

CHAPTER 5

Quantification of anthropogenic effects in the landscape of Lubumbashi

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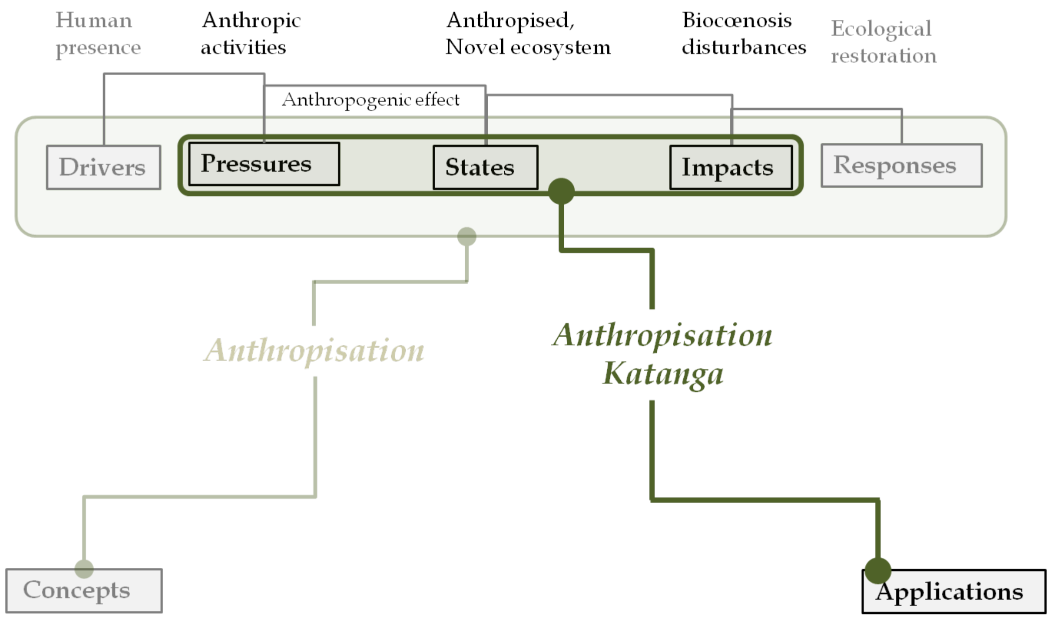

Figure 5.1 – *Location of the anthropisation katanga chapter (olivine green theme) in the thesis mind map. The hollow green article is thematically related to this one. Technical reading: application; thematic reading: anthropisation (question of Section 1.2.1); logical reading: pressures, states, impacts. This chapter is a first attempt to apply the new quantification methodology proposed in the Anthropisation chapter to Lubumbashi, D.R.C.*

5.1 Abstract

In this contribution, the Rüdisser et al. (2012) «Distance to Nature» methodology has been chosen to assess the effect of local human activities on the landscape surrounding the city of Lubumbashi (DRC). That methodology takes landscape hemeroby into account as well as structural connectivity by computing the distance to natural habitats. As that methodology had never been applied to an African city before, some adjustments (fitting of the local land uses types into the hemeroby levels designed in Austria) and amendments (suppression of the final classification simplification) are proposed. Moreover, an analysis of the decadal hemeroby dynamics is suggested. Initially, two classified Landsat images from 2002 and 2013 were used as input data. Finally, the transferability of the method is evaluated. Results suggest that the Distance to Nature methodology is transferable but requires good field knowledge to define reference habitats and identify them in the classified images. There was a dramatic decrease of the «natural» and «near-natural» levels in the study area during the studied period (2002-2013). In addition, 46% of the land underwent anthropisation dynamics, mostly around cities and following a ribbon development.

Keywords: anthropic influence – Geographic information system – land cover mapping – landscape – land cover change – tropical Africa

5.2 Résumé

Dans cette étude, la méthodologie « Distance to Nature » de Rüdisser et al. (2012) a été choisie pour évaluer l'effet des activités humaines sur le paysage entourant la ville de Lubumbashi (RDC). Cette méthodologie prend en compte l'hémérobie du paysage ainsi que la connectivité structurelle via le calcul de la distance aux habitats naturels. La méthodologie n'ayant encore jamais été appliquée au contexte d'une ville africaine, certains ajustements (mise en correspondance des utilisations du sol locales avec les niveaux d'hémérobie mis au point en Autriche) et amendements (suppression de la simplification finale de la classification) ont été proposés. De plus, une analyse de la dynamique décennale d'hémérobie est suggérée. Dans un premier temps, deux images Landsat classifiées datées de 2002 et de 2013 ont été utilisées comme données d'entrée. Enfin, la transférabilité de la méthode ainsi que la pertinence de la prise en compte des fréquences de feux ont été évalués. Les résultats suggèrent que la méthodologie « Distance to Nature » est transférable mais nécessite une bonne connaissance de terrain pour définir les habitats de référence et les identifier dans la classification des images. Durant cette période (2002-2013) et sur la zone d'étude, les niveaux « naturel » et « proche de naturel » ont considérablement diminué. De plus, 46% du territoire ont vu leur anthropisation augmenter, principalement autour des villes et suivant un développement en ruban.

Mots-clés: influence anthropique – Système d'information géographique – cartographie de l'occupation du sol – paysage – changement de couvert végétal – Afrique tropicale

5.3 Introduction

In this chapter, the issue of anthropisation in Katanga will focus on the city of Lubumbashi in its neighbourhood and will be analysed using landscape ecological paradigms. In landscape ecology, landscapes are considered as groups of interacting ecosystems. Landscape ecology is the science studying landscape spatial and temporal patterns to infer their impact on ecological processes (Turner, 1989).

Within this context, anthropisation is studied as for the impact of human activities on landscape composition, configuration and dynamics. They can be assessed in three ways. First, using simple indexes and proxies of anthropogenic activity like land use area, road and population density. Secondly, using a qualitative hemeroby scale, which will assess how much local naturalness has been disturbed or replaced, based on general criteria of disturbing processes and resulting land cover. Thirdly, anthropogenic effects on landscapes can also be represented by composite indicators of any of the aforementioned tools (Vranken et al., in preparation). Among the existing methods, the Rüdisser et al. (2012) «distance to nature» (D_2N) index and methodological framework appear most suitable for a landscape ecological analysis in our study zone. It will be further referred to as D_2N methodology.

The region of Lubumbashi is representative for landscape anthropisation in Katanga: situated in the Copper belt, this formerly rural zone has been built with new towns, quarries and plants since the beginning of the 20th century for non-ferrous metal exploitation and processing. This area is therefore subject to different anthropogenic effects, mostly reported as deforestation and (sub)urbanisation patterns (Vranken et al., in preparation-a;i).

In this chapter, the D_2N methodology will be applied for the first time to the context of an African city. The objectives of the study are to assess the effect of local activities on landscape spatiotemporal structure in Lubumbashi and its hinterland and the transferability of the method to a different context.

First, we describe the study zone and its main natural and human components. We go on summarizing the details of satellite image acquisition, treatment and classification we used to obtain a land cover map of the area. Then, we detail the adjustment we brought (calculation of the decadal anthropisation dynamic) to the D_2N method, its application and the meaning of its outputs. The results and discussion section highlights the anthropogenic and natural patterns for each of the two years (2002 and 2013), their decadal dynamics and comments the transferability of the method. Anthropisation (mostly (sub)urbanisation and deforestation) is expected to increase within this period. Some recommendations for future application of the methodology in the same context are then detailed in the perspective and conclusion section.

5.4 Material and methods

5.4.1 Study zone

Lubumbashi is the capital city of Katanga, a province in the Southern part of the Democratic Republic of the Congo (DRC). The site consists of a plateau that has been eroded into a wide valley by the Lubumbashi River and its tributaries (Chapelier, 1957). The altitudes of the inner-city, on the plateau, vary between 1200 and 1250 m (Sys and Schmitz, 1959). Lubumbashi is located in the Katangese copper belt and well known for its copper and cobalt veins (Chapelier, 1957).

This new town developed for mining development under Belgian colonialism since the beginning of the 20th century. It has been a place of intensive mining in the colonial period and afterwards (Banza et al., 2009). The population of Lubumbashi (1,200,000 inhabitants in 2006, near 2,020,000 estimated in 2015) is characterized by rapid growth and food dependency on imports linked to heavy metal deposits and absence of a tradition of farming (Chapelier, 1957; G8, 2009; Khonde and Rémon, 2006).

The colonial part of the city consists of a densely built-up zone provided with aged infrastructures. Massive rural flight in Lubumbashi leaded to unplanned urbanization and suburbanisation (Khonde and Rémon, 2006) Industrial infrastructure is also present in the former suburbs, now fully included into the urban fabric, and recent industries installed just outside the urban belt (Vranken et al., 2013). Near the smelters, debris are piled into huge slag heaps, tall and wide enough to be identified on Landsat images.

The local climate is characterised by a wet season (from November to April) and a dry season for the rest of the year and corresponds to the Cw Köppen category (Vranken et al., 2014). Currently, the vegetation cover is continuous only during the wet season (Adam, 2010). Islets of dry evergreen forest called *Muhulu*, present in the area (Malaisse, 1997), could indicate that the actual climax was Muhulu and that *Miombo* (called Woodland in this paper), the woodland forest strata that has dominated the area at least since the first observations during the colonial period, would be a pyro-climax resulting from former slash and burn agriculture (Noti et al., 2003; Schmitz, 1962; Sys and Schmitz, 1959; White, 1986). Diverse forms of savannah, from wooded savannah to grassland, as well as bare soils are now progressively replacing the woodland (Malaisse, 1997). Bare soils result mostly from mining activities and heavy metal eolian deposits (Vranken et al., 2013). If forest, more or less degraded, still covers about 50 % of the area, savannah and cultivated areas, generally resulting from forest clearance by fire, represent now the second largest land cover in the area (about 30 %). These diverse savannahs therefore result from anthropogenic degradation and do not present the same biophysical characteristics (including floristic composition) as natural savannahs (Parr et al., 2014). In tropical Africa, most fires are of anthropogenic origin (van der Werf et al., 2008). In our study area, where fire practices seem to have significant importance on the land use, this is corroborated by Malaisse (1997) and an existing correlation between fire starts and proximity to the city and roads ($R^2 = 0.78$, data not shown) as well as surrounding villages ($R^2 = 0.77$, data not shown). Moreover, the absence of correlation between fire starts and proximity to industrial sites also suggests that those fires are mostly operated by villagers for agriculture and charcoal production, which is a regular practice in the area (Stromgaard, 1985; Vranken et al., 2011). Specific metallophyte herbaceous flora is also present in contaminated soils but also natural highly metalliferous soils (mainly Copper and Cobalt), generally found on hills among the forest, called 'copper hills' (Leteinturier et al., 1999; Brooks et al., 1992; Malaisse et al., 1994). Specific features called *dembos*, natural grasslands temporarily flooded in valleys around water streams, are frequent in the area (Sys and Schmitz, 1959; White, 1986). Permanent wetlands are found as well round the riverbanks and depressions on impermeable ground, some of which are cultivated (Sys and Schmitz, 1959). Nearly all the lakes are of anthropogenic origin: reservoirs built during the colonial period.

5.4.2 Choice of the analytical framework

Vranken et al. (in preparation-a) expose, analyse and criticise in details the different concepts and quantifications related to anthropogenic effects assessment. According to this contribution, we will refer to anthropogenic effects assessment and associated terms using the term anthropisation, that represents human-driven landscape changes (Vranken et al., in preparation-a). In this contribution, the D_2N methodology was chosen for several reasons: first, it is designed to be used at landscape level ; then, it combines stretched values, patch and categorical analyses (Gustafson, 1998; Wiens et al., 1993), while the resulting values exist in both continuous and discontinuous variations, combining the advantages of the two output types. Moreover, it takes processes (different types, intensities and frequencies of human pressures on ecosystems) into account and integrates the presence of secondary habitats. Structural connectivity between natural habitats, which is important to ecological processes, is also included through the «distance to natural habitat» (D_n) component of the index. It can be evaluated even when few other data than land cover are available. Finally, it has been designed as an index easily understandable, facilitating the interpretation, comparison and communication of the results. We though implemented some adaptations to the methodology according to (Vranken et al., in preparation-a) and the specificities of our study zone and data availability.

5.4.3 Adaptation of the D_2N methodology to local landscape classes

5.4.3.1 Data acquisition

We used Landsat ETM+ and OLI multispectral images, from 2002.07.07 and 2013.07.13, with a spatial resolution of 30m (USGS, 2014). They were pan-sharpened with the corresponding panchromatic images to obtain a resolution of 15m using the ENVI 5.0 software. The first study site consists of the intersection of the areas covered by the Landsat images from 2002 and 2013 (about 23,400 km^2).

In order to obtain a minimal surface of burned areas (that were very abundant in our study zone), we applied a filter for the spectral signature of the burned areas on a set of Landsat calibrated images shot at different moments of the same year (05.04, 07.07, 08.08 and 10.11 for 2002, 06.27, 07.13 and 08.30 for 2013). We then recomposed a multidate image for each year from the filtered original images before performing a multiresolution segmentation using all the spectral bands of both images. Afterwards, we performed a supervised object-oriented classification based on spectral values and a shuttle radar topography mission (SRTM) image with a 90m resolution (Trimble Germany GmbH, 2013). The training sets for this classification were defined by direct field surveys regularly conducted between January 2012 and April 2014 and the freeware licenced version of Google Earth© imagery (from 2002 and 2012) for the remote areas (Giglio, 2013). Both segmentation and classification operations were performed using the eCognition© software. After a first land cover based classification (13 classes), we refined the results to display more information on the land cover and specific patches. As it was not possible to distinguish which wetlands were cultivated according to their sole spectral signature, a proximity rule was used : wetland segments touching anthropogenic ones (burned areas, continuous and discontinuous built-up areas, crops, pastures, young fallow and slag heap) were assumed to be potentially cultivated at least sporadically and therefore considered as anthropised. These latest were called « Anthropised wetlands » while the uncultivated ones were called « Wetlands ». As for the reservoirs, first identified as « Water », another kind of proximity rule was used in order to isolate them: water segments sharing 70% or more of their edges with other water segments were assigned as «Reservoirs», while the others were labeled «Streams». This classification refinement is particularly relevant for the quantification of the anthropogenic impact. That method allowed to distinguish 15 classes: « Natural grasslands », « Wetlands », « Streams », « Woodland », « Wooded savannah and old fallow», « Savannah and bushland «, « Savannah-crops mosaic », « Reservoir », « Crops, pastures, grasslands and young fallow », « Anthropised wetlands », « Recurrent burned areas », « Bare soil », « Slag heap », « Discontinuous built-up » and « Continuous built-up ». The Landsat classified images were then exported in raster format with 25m pixels, as in Rüdisser et al. (2012) for further treatment in ArcGIS©.

The latter were called « Anthropised wetlands » while the uncultivated ones were called « Wetlands ». As for the reservoirs, first identified as « Water », another kind of proximity rule was used in order to isolate them: water segments sharing 70% or more of their edges with other water segments were assigned as «Reservoirs», while the others were labeled « Streams ». This classification refinement is particularly relevant for the quantification of the anthropogenic impact. That method allowed to distinguish 15 classes: « Dembo », « Wetlands », « Streams », « Miombo », « Wooded savannah and old fallow», « Savannah and bushland «, « Savannah-crops mosaic », « Reservoir », « Crops, pastures, grasslands and young fallow », « Anthropised wetlands », « Recurrent burned areas », « Bare soil », « Slag heap », « Discontinuous built-up » and « Continuous built-up ». The Landsat classified images were then exported in raster format with 25m pixels, as in Rüdisser et al. (2012) for further treatment in ArcGIS©.

5.4.3.2 Data analysis

In order to apply the D_2N methodology to our dataset to obtain the D_2N index values, we proceeded in three steps. First we built hemeroby scales and maps, called « degree of naturalness » (N_d) by Rüdisser et al. (2012). We preferred using the term hemeroby here because, according to Vranken et al. (in preparation-a) hemeroby corresponds to a scale directly correlated with anthropisation, to the contrary to naturalness, and the purpose of Rüdisser et al. (2012) was an anthropisation-oriented index.

The land use types provided by Rüdisser et al. (2012) were only suitable for Austria and similar landscapes, but their qualitative hemeroby scale provided process information on type, intensity and impacts of human activities for each hemeroby level. Based on this description and specific site knowledge on local ecosystems and activities, we were able to fit the existing land use types of Lubumbashi into the D_2N-related hemeroby scale. The sorting of Katanga ecosystem into hemeroby classes was set according to a joint reflexion of an expert group on the ecosystems of Katanga.

As a first step, we assigned one of the seven hemeroby levels to each ecosystem (land use or cover) existing in Lubumbashi. As a second step, we sorted each of the 15 classes from our classification into the seven hemeroby levels. The decision tree used by the analyst in the field to discriminate the land cover classes, based on Trochain (1957); Letouzey (1982); Bellefontaine et al. (1997), is shown in Appendix B. As all classes did not allow distinguishing the ecosystems, sometimes grouped into the same class, the latter was allocated to the level of hemeroby corresponding to the dominant ecosystem (Table 5.1). Dry evergreen forest, copper hills, wetlands and natural grasslands were assigned to the first level, « Natural », but the classification identified only the latter two. The amount of anthropised grassland was so negligible that they were discriminated from natural grassland. Woodland and Streams were assigned to level 2, « Near-Natural » in both cases. Indeed, water streams are in most cases of natural origin but present eutrophication. Regenerating forest, wooded savannah and old fallow were put in the third level (« Semi-natural ») because those vegetation types are most of the time not found in the area before human direct intervention (Parr et al., 2014). The classification did not highlight the first. The ecosystems young fallow, savannah, bushland, pasture and grassland were assigned to level 4, « Altered », corresponding to the definition given by Rüdisser et al. (2012). In the class sorting, young fallow, pasture and grassland had to be assigned to the level 5 (« Cultural »), being grouped with crops. Crops, along with anthropised wetlands and reservoirs were as well put in this level. As the savannahs in the area mostly correspond to early stages of ecological succession to fires, the recurrent burned areas that could not be eliminated were assigned level 5 too. We put one of the classes (« Savannah – Crops mosaic ») in between hemeroby levels 4 and 5 because the crops land use was assigned to level 5 while savannah was assigned to level 4. Level 6, « artificial with natural elements», corresponded to discontinuously built areas and bare soils. Finally, the seventh level, « Artificial », with soil sealing over 30 %, was assigned to continuous built areas and slag heap. After performing those operations, we normalized the class values along a scale from 0 to 1, following the D_2N methodology and built the ${}_Nd$ map.

Secondly, we built a map of distance to natural habitat (D_n): this corresponds to the Euclidean distance (in meters) from each pixel of the images to the nearest natural or near-natural habitat (levels 1 and 2). Following the D_2N methodology, the cut off value for the case study was set to 1000 m considering foraging and dispersal ranges of many animal and plant species, so the distances superior to 1000 m were all set to 1000 m. In order to increase the effect of closer anthropogenic features, we took the square root of the resulting distances. Then, we normalized the results in order to obtain dimensionless values ranging from 0 to 1.

Thirdly, we multiplied the N_d maps by the D_n maps and normalized the result from 0 to 1 in order to enlarge the variation range of the results and re-scale it in accordance with the other normalised indexes. This gave the D_2N index map also called D_2N maps. The Rüdisser et al. (2012) methodology also reclassifies the results in four levels, but in our case, the choice was made to keep the continuous variation.

Hemeroby level	Description		Examples of land use types found in Austria	Ecosystems (land use / land cover) in the area of Lubumbashi	Matching image classes	
	Type of anthropogenic influence	Ecosystem patterns and processes			Matching hemeroby class	Class composition
1. Natural	No or only minimal anthropogenic influence (e.g. global pollution)		Bare rock, sparsely vegetated areas, glaciers and perpetual snow, inland marshes, peatbogs, natural forests	Shallow soil woodland, Dry evergreen forest, Wetlands, Natural grassland, Copper hills	1	Wetland, Natural grassland
2. Near-natural	Anthropogenic influences	Structure and type of ecosystem is basically the same as naturally expected at the side but some characteristics (e.g. plant species composition) are altered	Natural grasslands (above timberline), moors and heathland, water bodies, sustainably managed forests	Deep soil woodland, Water	2	Woodland, Streams
3. Semi-natural	Anthropogenic activities	The naturally occuring ecosystem is no longer present but has been transformed into a new ecosystem type because of anthropic activity	Alpine meadows substituting forest pastures, fallow land	Regenerating forest, Wooded savannah, Old fallow	3	Wooded savannah and old fallow
4. Altered	Regularly disturbing anthropogenic activities (e.g. drainage, regular passing over, intense fertilisation)	Changed ecosystem type, edaphon regularly disturbed	Vineyard, intensively used grasslands, plantation of energy forests	Young fallow, Savannah, Bushland, Grassland, Pastures	4	Savannah and bushland
					4.5	Savannah / crops mosaic
5. Cultural	Intense and regular impacts	Destruction of the natural occuring edaphon. Natural occuring floristic elements are reduced to a minimum (< 25% coverage)	Arable land, green urban areas, sport and leisure facilities	Anthropised wetlands, Crops, Reservoirs, Anthropised grasslands	5	Anthropised wetlands; Crops, pastures, grassland and young fallow; Recurrent burned areas; Reservoirs
6. Artificial with natural elements	Intensive and irreversible changes of terrain and landscape structure; soil sealing up to 30%	Natural elements only in the form of secondary biotopes	Rural settlements, mineral extraction sites, dump sites, airports	Discontinuous built, Bare soil	6	Discontinuous built, Bare soil
7. Artificial	soil sealing over 30%	Artificial systems or structures	Continuous urban fabric, industrial or commercial units, road and rail networks	Continuous built, Slag heap	7	Continuous built, Slag heap

Table 5.1 – *Rüdisser's hemeroby scale (2012) (« Degree of naturalness ») and its adaptation to the region of Lubumbashi (DRC). The four first columns are extracted from Rüdisser et al. (2012). The fifth column shows ecosystem correspondence to the seven hemeroby levels, while the two last columns (dashed lines) show the level of each land use and land cover classes of the classified Landsat images.*

5.4.4 Anthropisation dynamics analysis

In addition to the D_2N methodology, we highlighted the dynamics of anthropogenic influence in Lubumbashi between 2002 and 2013 by subtracting 2002 to 2013 D_2N values, constructing a post-classification change detection map. That step aims at evaluating whether anthropisation increases during the period.

5.5 Results and discussion

5.5.1 Adaptation of the D_2N methodology to local landscape classes

As for the image classification precision, the obtained Kappa coefficients were low (0.397 for 2002 and 0.364 for 2013, see confusion matrices in Appendix B) (Congalton, 1991). This may be due to different factors: first, seasonal variation is strong in the area, especially concerning fire dynamics, and may have led to misclassifications (Congedo and Munafò, 2012). Second, bare soils have a very similar spectral signature to built-up areas (Congedo and Munafò, 2012). Thirdly, the very fast urban dynamics in the area may lead to land use change-driven differences between field survey and image capture given the time elapsed between the two operations. Fourthly, spatial structure in Africa is loose, compared to Northern countries: the land cover patches are less clearly bordered (Vranken et al., 2013), probably due to differences in land planning practices. This may lead to confusion between adjacent land covers. Fifthly, due to the medium spatial resolution of the images, pixel may contain different ecosystems (this phenomenon is known as "mixel" problem) but must be attributed to a single class, therefore not representative of the whole pixel (Pham and Yamaguchi, 2011). Finally, ecosystems, as often responding to regressive or progressive processes, are seldom "pure" but are transition states from one ecosystem to another. The analyst and the classification software may settle the threshold between borderline land covers differently, leading to virtual misclassifications.

It should however be noted that the accuracy of the classifications is inversely proportional to the thematic resolution (number of classes): if we had dropped our number of classes down to 8, the Kappa would have risen to 0.59. Note that Congedo et al. (2012) obtained a Kappa value of 0.57 on a similar area using Landsat images, but their classification contained only 5 classes, while we have 13 of them. Here, we chose to privilege thematic resolution in order to obtain relevant classification for the construction of our hemeroby scale. This however does not question the validity of the D_2N methodology, which was applied as a post-treatment on the classified images.

The consequences of the misclassifications depend on: (1) the confusion between classes of distinct hemeroby levels, (2) the definition of the reference states (natural and near-natural levels) and other hemeroby levels, (3) the correct classification (user precision) of these reference states. Errors in the two latter points have multiplicative effects on the results, given that the D_2N methodology is based on both hemeroby levels and distance to natural and near-natural levels, and given that the dynamics map depends on the two D_2N maps. In the case of this research, the most problematic misclassification is the wrong classification of wooded savannah (level 3) as woodland (level 2) (see Appendix 2 and 3), that wrongly naturalises the landscape.

5.5.2 Data analysis

The application results of the D_2N methodology to Lubumbashi in 2002 (Figure 1a) and 2013 (Figure 5.2b) show the anthropisation levels in the area. The extent of this first study area includes more than one urban zone (4% of total area in 2002, 11% in 2013), dispersed in a natural or near-natural matrix (about 60% of the total extent, against 75% in 2002). Connectivity between natural habitats seems not only hampered by urban and cultivated areas, but also influences those areas themselves, making them benefit from the proximity of natural and near-natural areas via a decrease of their D_2N value. Therefore, the inclusion of the notion of connectivity in the analysis naturalises the landscape. Dark spots, representing the highest levels, correspond to the main urbanised zones in the area, the largest of which is Lubumbashi (center-right), followed by Likasi, along the Tshangalele Reservoir, Northwest of Lubumbashi, Kipushi, the closest dark zone Southwest to the city.

To build their N_d map, Rüdisser et al. (2012) used a large amount of data (forest hemeroby, CORINE classification, roads, etc.), some of which were available as continuously stretched values, but those were integrated in the form of a discontinuous and qualitative hemeroby scale, displayed as a categorical map. In our first approach, we followed the same guidelines, except that less reliable data were available in our study zone.

Patches of the same land cover may follow different dynamics regarding anthropisation level. For example, one savannah area may result from a regressive series that can be linked with fire disturbance, while the other may result from a progressive series, i.e. ecological succession, linked with the stop of previous disturbances. In the present study, such distinction between progressive and regressive series could not be achieved yet. This is partly due to coarse data spatial resolution and lack of classification precision (map categories include here different land covers, sometimes displaying different hemeroby levels).

The choice of the ecosystems corresponding to the reference states and their identification on the classified image is of particular importance. In the case of Katanga, woodland is said to be a *pyro-climax* on deep soils (dry evergreen forest being the natural vegetation in this case) but natural vegetation on shallow soils (Schmitz, 1962; Lawton, 1978; White, 1986), which was the choice we made (see Table 5.1). However, this is to some extent controversial among scientists (Mahy, personal communication). In Table 5.1, level 1 is considered as virtual naturalness, considering pre-colonial human interventions on landscape structure as anthropogenic, distinct from nature (Lecomte and Millet, 2005; Peterken, 1996; Vranken et al., in preparation-a).

The choice not to apply the the original four-level D_2N scale is justified by two facts. First, continuous variations of the D_2N values appears more precise: simplifying them in only four levels, while there were seven levels in the original scale, represents information loss. Secondly, the loose and less clearly bordered spatiotemporal structure of African (Vranken et al., 2013) appears best represented using continuous transitions between anthropisation levels. Attention should be paid to the fact that the results, showing dominant natural or near-natural classes, are due to the very large extent of the study zone (23,400 km^2). It may give the wrong impression that Lubumbashi causes little anthropisation in the area.

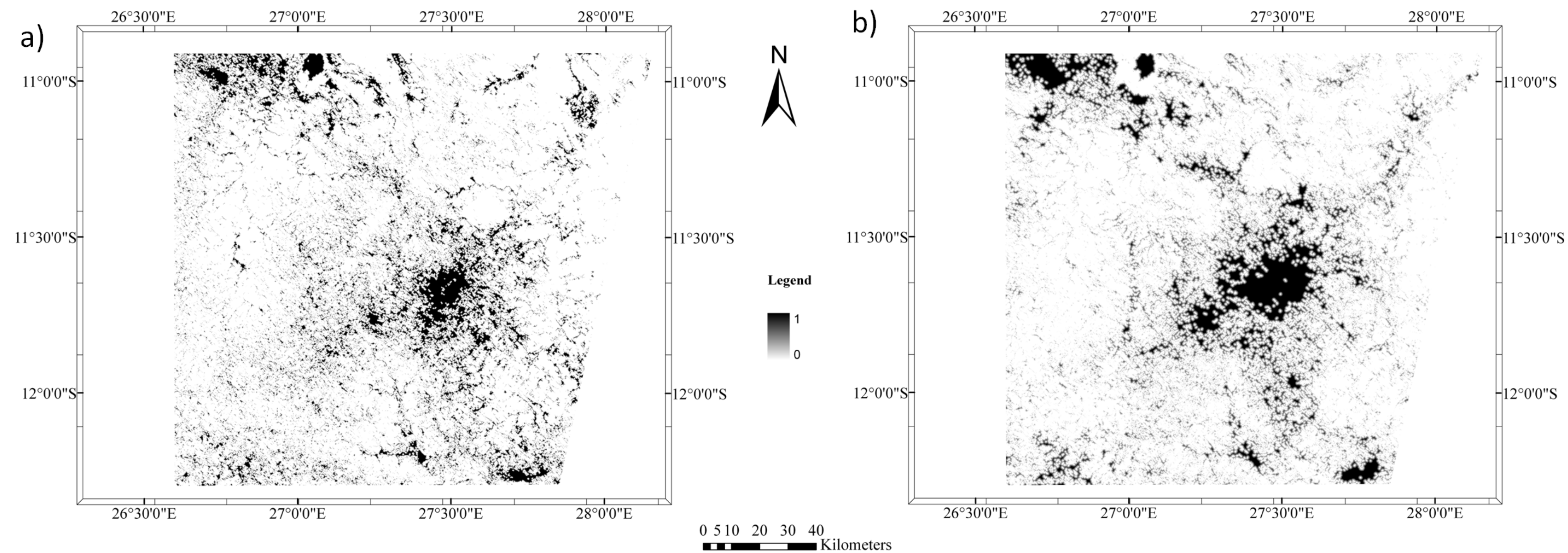

Figure 5.2 – *Application of the Rüdisser et al. (2012) « Distance to nature » methodology to the landscape of Lubumbashi (DRC) in 2002 (a) and 2013 (b). An index value of 0 means the lowest distance to nature; a value of 1 means the highest distance to nature.*

5.5.3 Anthropisation dynamics between 2002 and 2013

In 2013, the suburban zones of Lubumbashi and Kipushi almost merge , while they were still distinct 12 years earlier. Though natural and near-natural areas still dominate the area, they are now strongly fragmented. These preliminary observations, along with the overall darker colour of the 2013 image, tend to state that urbanisation has indeed increased.

The area change for each anthropisation level between 2002 and 2013 (Figure 5.3c) shows that dominant dynamics are dramatic decrease (about 11 % of total extent) in the natural and near-natural levels and substantial increase in intermediate levels of anthropisation, probably following the aforementioned (sub)urbanisation dynamics.

The relative increases and decreases in anthropisation levels, quantified in Figure 5.3b, show that about 46% of the land underwent anthropisation level change in 12 years. Moderate increase dominated the anthropisation dynamics (24.5% of total extent increased by 0.001 to 0.3 in D_2N value). The zones encountering anthropisation increase cover 32% of the total area, while the total anthropisation decrease only represents 15%, which confirms an overall anthropisation increase. The anthropisation increase is concentrated around cities and spreads in ribbon development (Figure 5.3a) (Dumont and Bossé, 2006; Ewing, 1994).

It should also be noted that the highest gains in naturalness are mostly dispersed near the suburban belts. This phenomenon may be linked to young fallow developing into forest, misclassification or set-aside (G8, 2009). The observed ribbon development is similar to the urbanisation patterns in U.S. and European cities (Ewing, 1994; Grosjean, 2010).

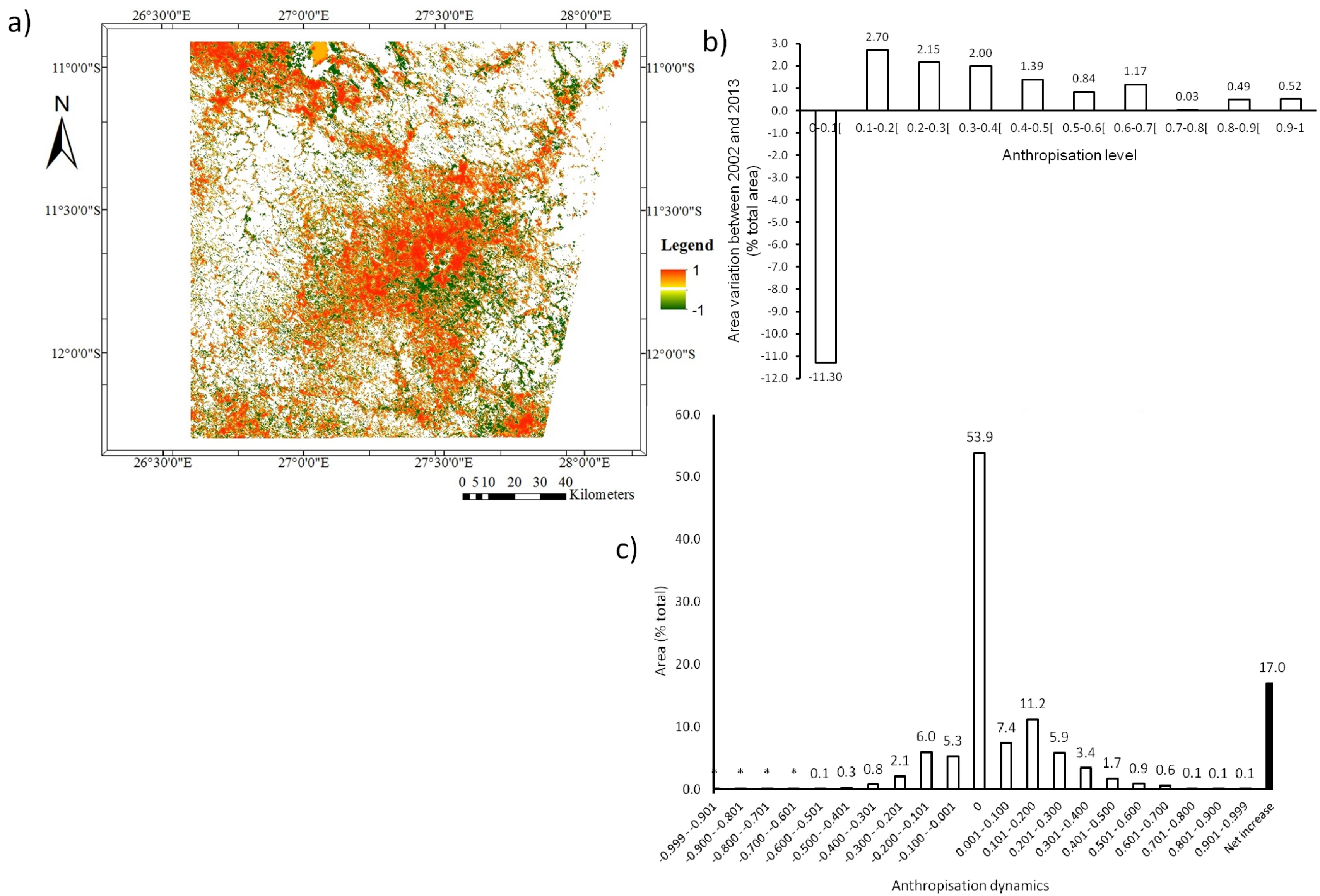

Figure 5.3 – *Dynamics of anthropisation levels between 2002 and 2013 in Lubumbashi (DRC). The numbers from -1 to 1 represent the number of levels respectively gained or lost during this period; 0 represents no anthropisation level change. The map (a) shows the localisation of the dynamics, while the graph (b) shows the percentage of the total area concerned by each change type (* represents values* $< 0.1\%$*). The bar diagram (c) shows the percentage of the total area increase of each anthropisation level between 2002 and 2013, ranging from 0 (least anthropised level) to 1 (most anthropised level).*

5.6 Conclusion and perspectives

This first attempt to apply an hemeroby-based anthropogenic effect quantification to a region in Tropical Africa appears promising for comparison with other countries. It should be applied on other southern cities in order to compare and assess the relevance of the results. It could also be compared with other methodologies applied on the same area but the lack of information currently available at this scale is a limiting factor for such implementation. This case study still lacks relevant information specific to local data, dynamics and practices.

The functional part of connectivity must be taken into account when evaluating the configurational aspects of landscape anthropisation (Tischendorf and Fahrig, 2000). For example, a large compact patch tends to shelter more interior and rare species, while having less positive impact on structural connectivity than various corridor-shaped patches (Turner, 1989). The methodology should be amended in order to take that functional factor into account and could even be species-oriented.

Concerning the distinction between progressive and regressive series, the distinction should be more feasible using more data, with better spatial and temporal resolution or producing smaller objects via segmentation. This would also allow building the same post-classification change detection maps as in this contribution as well as Nd transition matrices to study specific patch dynamics. Reliable thematic maps such as up-to-date road net, activities and infrastructure could also be added, as in Rüdisser et al (2012). In this case, specific attention should be paid to weighing respective influences of each human activity or disturbance data, in order to avoid redundancy and biases.

Considering the methodology application, the mutual influences of anthropised and natural landscape, highlighted by the introduction of a distance gradient , open interesting perspectives for land planning, conservative management and restoration. Indeed, the effect of adding or removing natural patches in the landscape depending on the distance to existing natural habitats and on the surrounding land use types could be simulated using D_2N maps. This could help to highlight which areas should be preferably protected and where it would be most useful to restore degraded ecosystems. It should however be noted that this methodology does not distinguish the natural habitat richness or conservation interest linked to specific composition, which is necessary to every conservative management and should be complementarily examined (Séleck et al., 2013).

List of abbreviations

D_2N : Distance to nature (index, map, methodology)
D_n: Distance to natural habitat
DRC: Democratic Republic of the Congo
Ff: fire frequencies
N_d: Degree of naturalness
SRTM: Shuttle Radar Topography Mission

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CHAPTER 6

The importance of studying anthropogenic effects on landscape heterogeneity

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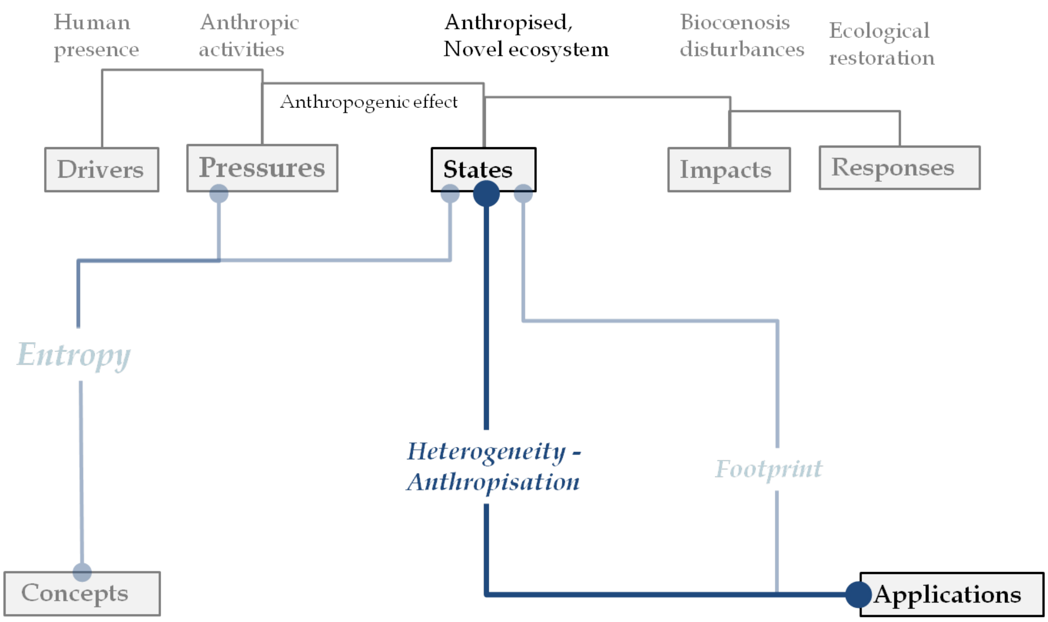

Figure 6.1 – *Location of the heterogeneity - anthropisation article (deep blue theme) in the thesis mind map. Technical reading: application, thematic reading: heterogeneity (question of Section 1.2.2), logical reading: states. The hollow blue articles are thematically related to this one. Based on the spatial structure of 20 landscapes, it highlights the complex relationship between the different components of spatial heterogeneity and landscape anthropisation.*

6.1 Abstract

Based on the pattern / process paradigm, landscape ecology provides an integrative framework for the assessment of anthropogenic landscape change. However, though numerous studies of anthropogenic effect or spatial heterogeneity on species distribution already exist, the link between anthropogenic effect and spatial heterogeneity itself has incompletely been addressed. This link would help to understand the interaction between ecological vs. cultural processes and their resulting spatial patterns. This paper therefore aims to test various indicators representing the different components of spatial heterogeneity and anthropogenic effect on 20 landscapes and to model each heterogeneity - anthropogenic effect relationship. The results show (1) opposite anthropogenic effect on pattern heterogeneity depending on the amount of already anthropised surface (2) opposite tendencies of the natural vs. anthropogenic heterogeneity dynamics under anthropogenic landscape change and (3) opposite tendencies of the various heterogeneity components under anthropogenic landscape change. These differentiated dynamics justify the need for such studies in further work on the impact of anthropogenic effects or heterogeneity on species distribution, with valuable predictive outcomes for land planning, restoration and conservation.

Keywords: Spatial heterogeneity, Anthropisation, Anthropogenic effect, Pattern / process paradigm, Landscape ecology, Tropical Africa

6.2 Introduction

In landscape ecology, the link between spatial pattern and ecological functioning (pattern / process paradigm) is especially exploited for studying pattern heterogeneity, which is of capital importance for the distribution of living organisms (Turner, 1989; Cale and Hobbs, 1994; Li and Reynolds, 1995; Fahrig, 2005). Heterogeneity is a characteristic of landscape structure, described by its composition (types of habitat patches present and their relative proportions) and configuration (patch spatial arrangement and geometry, patch size distribution) (Li and Reynolds, 1995).

Since humans have settled on earth, their influence on ecosystems has kept on growing (Bogaert et al 2011). This phenomenon is called *anthropisation*, defined as the process of landscape change as a consequence of anthropogenic effects (Bogaert et al., 2011; Vranken et al., in preparation-a), but it is also referred to with many other terms (Vranken et al., in preparation-a). According to the pattern/process paradigm, human intervention on pattern should influence ecological processes within the affected landscape ecosystems (Turner, 1989). It is therefore useful to study anthropogenic effects on landscape structure to infer consequences for conservation or restoration ecology .

The issue has already been addressed in the past, though incompletely (Krummel et al., 1987; Turner and Bratton, 1987; O'Neill et al., 1988; Mladenoff et al., 1993; Kaufmann et al., 2000). The search for the anthropogenic impact on pattern heterogeneity was slowed down dramatically since a major critique was raised on pattern measurements and their inferences on ecological functioning. Indeed, authors like Cale and Hobbs (1994); McIntyre and Hobbs (1999); Tischendorf and Fahrig (2000) raised awareness of the fact that different species, with different ecological niches, mobilities, behaviours, perceive their environment (the landscape, the ecosystem patches) in different ways, so spatial (and temporal) landscape patterns should be different from one species to another. Therefore pattern heterogeneity measurement was relevant only when measured according to a definite species or group of species with the same perception. This observation was an important breakthrough in landscape ecology because it helped further studying the link between spatial patterns and ecological functioning by adding functional aspects to spatial patterns assessments (Tischendorf and Fahrig, 2000). Ever since, very few studies still focused on the link between anthropisation and heterogeneity. Indeed, these studies now focus either on the impact of pattern heterogeneity or of anthropisation on species distribution either separately or without actually studying the link between anthropisation and heterogeneity (Blair, 1996; Mackey and Lindenmayer, 2001; Lindborg and Eriksson, 2004; Spooner et al., 2004; Pavlacky JR et al., 2009).

Even the few attempts of addressing the relationship between anthropisation and heterogeneity that occurred later did not deepen the subject enough, which leads to contradictory results (Bogaert et al., 2005; Pickett and Cadenasso, 2009). The existing studies indeed sometimes stated ascending or descending relationships, sometimes linear, sometimes logarithmic. They focused either on overall dynamics or only natural patches separately, and never compared the behaviour of the different components of spatial heterogeneity, such as compositional vs. configurational.

The objective of this paper is then to answer the following questions and hypotheses: (1) can the diverging conclusions of previous studies on the impact of anthropisation on heterogeneity be reconciled because heterogeneity would in fact respond in a bell shaped curve to increasing anthropisation? (2) Do natural and anthropogenic land covers follow different heterogeneity dynamics under growing anthropisation? (3) Do different components of heterogeneity follow different dynamics under growing anthropisation? In this paper, for the first time, the different components of pattern heterogeneity of both natural and anthropogenic land covers are studied jointly under a complete anthropisation gradient (from fully natural to fully anthropised landscape) to answer these questions. We use a set of 20 tropical African landscapes with different anthropisation intensities and patterns and we model their heterogeneity dynamics, distinguishing the different heterogeneity components, but also natural from anthropogenic dynamics. Verifying these hypotheses would tend to prove that such studies should also be performed in targeted research on definite species addressing the impact of either anthropisation or heterogeneity on organism distribution.

6.3 Material and Methods

6.3.1 Data acquisition

The option taken here to study the anthropogenic effect on pattern heterogeneity is to combine synchronic analysis with diachronic analysis (Baker and Billinge, 1982). A set of 11 different areas, here denoted as study sites were investigated. They are located in Democratic Republic of the Congo (D.R.C.), Ivory Coast and Benin. Six of the 11 sites present time series of landscape observation, with a total of 20 different landscapes (see Figure 6.2). The 20 landscapes are distributed along a wide gradient of landscape anthropisation. Moreover, the data set includes various biomes, presenting various forms of anthropisation coming from various pressure types occurring in tropical Africa (see Table 6.1).

The data were acquired during previous studies conducted by the co-authors. The metadata on acquisition and landscape characteristics are available in Table 6.1. Nearly all the data are classified LANDSAT images, with the same spatial resolution (30 m), except landscapes 18 and 19, which come from FAO's Africover cartography project (about 1000 m). Except for landscapes 3, 7 and 17, that were defined according to a smaller municipality upon data acquisition, the extents of the study sites are all the same (about 800 km^2). That extent was chosen to encompass the largest periurban zone investigated (Lubumbashi, D.R.C.) as well as its closest hinterland. The area of Lubumbashi in 2009 (landscape 20) is the most anthropised (see Section 6.3.2.1) landscape in our data set.

In their original version, the thematic resolutions (level of details of the image classifications) of the landscapes were designed according to the specific objectives of the previous studies they came from. In order to avoid bias, those were homogenised (Bailey et al., 2007). This was performed through merging similar patch types in order to adapt classification precisions to the less precise classifications of the image set : Bamba et al. (2008). The differences in numbers of patch types were reduced only when they were caused by thematic resolution, not by landscape richness. For example, for the landscapes in Benin, the whole forest-savannah gradient was covered in details using seven classes, while such details were not provided in Lubumbashi, where only savannah and woodland were distinguished. Therefore, the number of classes related to the forest-savannah gradient in Benin was reduced to 3.

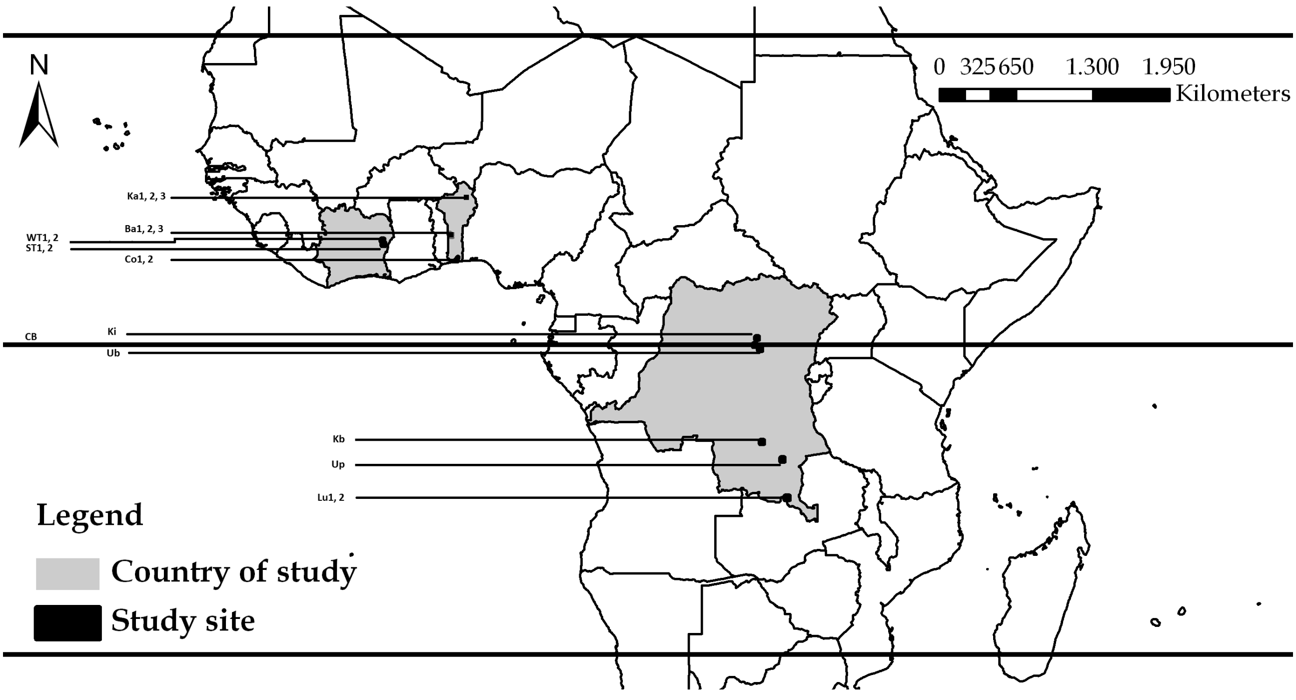

Figure 6.2 – *Location of the 11 study sites in D.R. Congo, Benin and Ivory Coast. The numbers refer to the 20 landscapes presented in Table 6.1, some of which correspond to times series on the same site. Bold horizontal lines represent the Equator and the tropics.*

Table 6.1 – *Metadata of the study sites. The list of land cover classes is here presented after thematic resolution harmonisation. The landscape numbers correspond to chronological order, but the landscapes belonging to the same time series were grouped in the table. (legend: see next table)*

Landscape number	Ba1	Ba2	Ba3	Lu1	Lu2
Country	Benin	Benin	Benin	D.R.C.	D.R.C.
Reference place	Bantè	Bantè	Bantè	Lubumbashi	Lubumbashi
Coordinate system	UTM31N WGS1984	UTM31N WGS1984	UTM31N WGS1984	UTM 35S WGS1984	UTM 35S WGS1984
Center coordinates (m)	375; 2687297534	375; 2687297534	375; 2687297534	551102; 8711134	551102; 8711134
Precision (m)	30	30	30	30	
Extent (km^2)	800	800	800	800	800
Shot date	1972	1986	2006	1984	2009
Classif. method	supervised	supervised	supervised	hybrid	hybrid
Natural classes	savannah, rainforest/ woodland, gallery forest	savannah, old growth forest, swamps, water	savannah, old growth forest, water	woodland, swamps, water	woodland, swamps, water
Anthropogenic classes	bare soils, crops/ buildings	bare soils, crops/ buildings, tree plantations	bare soils, crops/ buildings, tree plantations	bare soils, crops/ buildings, savannah	bare soils, crops/ buildings, savannah
Human activities	Semi-shifting slash and burn agriculture/pasture with fallow	Slash and burn agriculture/pasture, forest management	Slash and burn agriculture/ pasture, tree plantation/ logging, charcoal production	Slash and burn agriculture, mining, metallurgy, charcoal production	Slash and burn agriculture, mining, metallurgy, charcoal production
Landscape type	woodland and savannah landscape progressively degraded in savannah and bare soil due to urbanisation and climate change			woodland degraded in savannah and bare soil by urbanization and mining activities	
Map source	LANDSAT TM	LANDSAT TM	LANDSAT TM	LANDSAT TM	LANDSAT TM
Reference Publ.	a, c, g, i	a, c, g, i	a, c, g, i	h, i	h, i

Table 6.2 – *Metadata of the study sites (2). Table 6.1 continued. D.R.C: Democratic Republic of the Congo, hybrid: supervised/unsupervised classification, 1: based on typical relations between map scale (1/2 500 000) and geometric image resolution, after Spatial Industries Business Association, 20111, 2: FAO Africover mapping project. a: Bamba et al. (2010b); b: Bamba et al. (2008); c: Barima et al. (2010); d: Barima et al. (2011); e: Belgian Royal Museum of Central Africa (2008) ; f: Djibu Kabulu et al. (2008); g: Mama (2013a); h: Munyemba et al. (2008); i: Vranken et al. (2011); j: Vranken et al. (2013).*

Landscape number	Ka1	Ka2	Ka3	WT1	WT2
Country	Benin	Benin	Benin	Ivory Coast	Ivory Coast
Reference place	Kandi	Kandi	Kandi	NW of Tanda	NW of Tanda
Coordinate system	UTM31N WGS1984	UTM31N WGS1984	UTM31N WGS1984	UTM30N WGS1984	UTM 30N WGS1984
Center coordinates (m)	497536; 1240122	497536; 1240122	497536; 1240122	473057; 876654	473057; 876654
Precision (m)	30	30	30	30	
Extent (km^2)	800	800	800	800	800
Shot date	1972	1986	2006	1986	2002
Classif. method	supervised	supervised	supervised	hybrid	hybrid
Natural classes	savannah, old growth forest, swamps, water	savannah, old growth forest, swamps	savannah, old growth forest, swamps, water	rainforest, woodland, tree savannah	rainforest, woodland
Anthropogenic classes	bare soils, crops/ buildings, tree plantations	bare soils, crops/ buildings, tree plantations	bare soils, crops/ buildings, tree plantations	crop savannah, bare soils/ buildings	crop savannah, bare soils/ buildings
Human activities	Slash and burn agriculture/pasture, forest management	Slash and burn agriculture/pasture, forest exploitation	Slash and burn agriculture/pasture, forest management (charcoal, logging, plantations)	Forest management (fruit, log), slash and burn agriculture (with fallow)	Forest management (fruit, log), slash and burn agriculture (with fallow)
Landscape type	Forest and tree savannah facing urbanisation, agricultural conversion and savanisation			Forest-savannah transition, fragmented in crops, bare soils and urbanisation	
Map source	LANDSAT TM	LANDSAT TM	LANDSAT TM	LANDSAT ETM+	LANDSAT ETM+
Reference Publ.	a, c, g, i	a, c, g, i	a, c, g, i	c, d	c, d

Table 6.3 – *Metadata of the study sites (3). Table 6.1 continued.*

Landscape number	CB	Ub	Ki	ST1	ST2
Country	D.R.C.	D.R.C.	D.R.C.	Ivory Coast	Ivory Coast
Reference place	Congo basin	Ubundu	Kisangani	NW of Tanda	SE of Tanda
Coordinate system	UTM34N WGS1984	UTM34N WGS1984	UTM34N WGS1984	UTM30N WGS1984	UTM 30N WGS1984
Center coordinates (m)	945860; -3155	992583; -38344	965673; 55101	501755; 846681	501755; 846681
Precision (m)	30	30	30	30	30
Extent (km^2)	800	800	800	800	800
Shot date	2001	2001	2001	1986	2002
Classif. method	supervised	supervised	supervised	hybrid	hybrid
Natural classes	rainforest, water	rainforest, water	rainforest, water	rainforest, woodland, tree savannah	rainforest, woodland
Anthropogenic classes	crops/ buildings, tree plantations	crops/ buildings, tree plantations	crops/ buildings, tree plantations	crop savannah, bare soils/ buildings	savannah/ crops, bare soils/ buildings
Human activities	Small villages and agriculture following logging roads	Slash and burn agriculture (shortening fallow), forest management (log, charcoal, plantation)	Slash and burn agriculture (shortening fallow), forest management (log, charcoal, plantation), economic center	Forest management (fruit, log), slash and burn agriculture (with fallow)	Forest management (fruit, log), slash and burn agriculture (with fallow)
Landscape type	Almost undisturbed Rainforest	Small town in a rainforest landscape	Rainforest degraded in agrarian and urban zones but climatic reforestation	deforested rainforest and woodland by urbanisation and crops . Savanisation	
Map source	LANDSAT TM	LANDSAT TM	LANDSAT TM	LANDSAT ETM+	LANDSAT ETM+
Reference Publ.	a, b, i	a, b, i	a, b, i	c, d	c, d

Table 6.4 – *Metadata of the study sites (4). End of Table 6.1.*

Landscape number	Co1	Co2	Co3	Kb	Up
Country	Benin	Benin	Benin	D.R.C.	D.R.C.
Reference place	Cotonou	Cotonou	Cotonou	Kabongo	Upemba park
Coordinate system	UTM31N WGS1984	UTM31N WGS1984	UTM31N WGS1984	UTM35S WGS1984	UTM 35S WGS1984
Center coordinates (m)	425038; 71225	425038; 71225	425038; 71225	341074; 9182430	520716; 8944792
Precision (m)	30	30	30	1000[1]	1000[1]
Extent (km^2)	488	488	488	800	800
Shot date	1972	1986	2006	2007	2007
Classif. method	supervised	supervised	supervised	supervised	supervised
Natural classes	savannah, old growth forest, swamps	savannah, old growth forest, swamps, water	savannah, swamps, water	savannah, old growth forest, swamps, water	wood savannah, tree savannah, swamps
Anthropogenic classes	bare soils, crops/ buildings, tree plantations	bare soils, crops/ buildings, tree plantations	bare soils, crops/ buildings, tree plantations	crops/ buildings, secondary forest	crops
Human activities	Mangrove/ forest management (fruit, energy, log)	Mangrove/ forest management, agriculture, capital	Mangrove/ forest management (fruit and wood export), agriculture, capital city	Slash and burn agriculture	Traces of permanent agriculture
Landscape type	Savannah and lagune progressively affected by urbanisation and agriculture			Small town in a savannah landscape	Savannah reserve
Map source	LANDSAT TM	LANDSAT TM	LANDSAT TM	LANDSAT TM	LANDSAT TM
Reference Publ.	a, c, g, i	a, c, g, i	a, c, g, i	e, f, i	e, f, i

6.3.2 Data management

In order to model the relationship between anthropisation and different components of spatial heterogeneity, ten different kinds of metrics were computed for each of the 20 landscapes described in Table 6.1. The different components of heterogeneity that were calculated are: compositional heterogeneity (number and relative areal abundances of classes) and configurational heterogeneity (patch size, patch shape) (Mladenoff et al., 1993; Li and Reynolds, 1995). Overall indexes were first calculated, then they were divided into natural and anthropogenic components, i.e., the contributions of natural vs. anthropogenic patches to overall heterogeneity.

In order to apply these metrics to the 20 landscapes, the spatial data derived from the classified images were processed using the ArcGIS 9.3 and Fragstats (McGarigal et al., 2012) softwares. The dependence between anthropisation and the different heterogeneity components based on the metrics values of each landscape were represented on scatter plots for regression analyses. We used second order polynomial (quadratic) regressions to model the relationships because the bell-shaped curves fitted the scatter plot (Legendre and Legendre, 2012).

6.3.2.1 Anthropisation

Anthropisation was measured using a very simple index. A is calculated as shown in Equation 6.1, where a_a is the area of the anthropogenic patch types in the landscape considered and a_{tot} the total area of this landscape. Note that $a_a + a_n = a_{tot}$, where a_n is the area of the natural patch types. This index is derived from the *U* disturbance index (O'Neill et al., 1988), transformed in order to range its variation in linear progression and with two borders: $0 \leq A \leq 1$. $A = 0$ represents an undisturbed landscape, while $A = 1$ indicates a fully anthropogenic landscape. For further computations, a threshold value of $A = 0.5$ was used to distinguish natural from anthropogenic landscapes. Determining which patch type is of natural or anthropogenic origin was performed according to in situ knowledge (see Table 6.1). This is necessary since, for different sites, the same land cover (e. g. savannah) may result from different dynamics, either natural or anthropogenic.

$$A = \frac{a_a}{a_{tot}} \tag{6.1}$$

6.3.2.2 Number and areal abundance of land cover classes

Compositional heterogeneity was quantified by the Simpson diversity index H (Renyi, 1961; Pielou, 1975; Gregorius and Gillet, 2008). It represents the probability that two patches sampled at random in a landscape do not belong to the same patch type (Pielou 1975; Gregorius and Gillet 2008). It is calculated by Equation 6.2, where p_i is the proportion of area of the i-th patch type ($p_i = a_i/a_{tot}$), a_i being the area of the i-th patch type, and N the total number of patch types. $0 \leq H < 1$. $H = 0$ indicates minimal heterogeneity: landscape dominated by one single patch type. $H \rightarrow 1$ represents maximal heterogeneity, many patch types evenly distributed.

$$H = 1 - \sum_{i=1}^{N} p_i^2 \tag{6.2}$$

The natural and anthropogenic components of H, called H_n, and H_a, were calculated using proportions of each natural or anthropogenic patch type ($p_{i,n}$ or $p_{i,a}$) in the total area of the landscape (a_{tot}). Instead of a value of 1, as in Equation 6.2, the partial Simpson index was calculated with the respective natural or anthropogenic area proportions. This was done so as for $H_n + H_a = H$. The indexes H_n and H_a are then calculated as follows, with N_n and N_a being the total number of respectively natural and anthropogenic patch types.

$$H_n = \frac{a_n}{a_{tot}} - \sum_{i=1}^{N_n} p_{i,n}^2 \tag{6.3}$$

$$H_n = \frac{a_a}{a_{tot}} - \sum_{i=1}^{N_a} p_{i,a}^2 \tag{6.4}$$

6.3.2.3 Patch size

Patch size is the first component of configurational pattern heterogeneity. It is here represented by a transformation of the Largest Patch Index or LPI (McGarigal et al., 2012). The original LPI represents the proportion occupied by the largest patch in the landscape. The version used here has been modified in order to render it directly proportional to heterogeneity. It is calculated using Equation 6.5, where a_{max} is the area of the largest patch in the whole landscape, and a_{tot} the total landscape area. $0 \leq LPI < 1$. The landscape is characterised by configurational homogeneity, dominated by one single patch when $LPI = 0$. $LPI \to 1$ represents extreme fragmentation: infinitely small patches dispersed throughout the landscape (Fahrig and Nuttle, 2005).

$$LPI = 1 - \frac{a_{max}}{a_{tot}} \tag{6.5}$$

As for LPI_n, the natural component of LPI, the size of the largest patch in the natural patch types ($a_{max,n}$) was picked up and divided by the total landscape area. The same method was applied to anthropogenic patch types (LPI_a):

$$LPI_n = 1 - \frac{a_{max,n}}{a_{tot}} \tag{6.6}$$

$$LPI_a = 1 - \frac{a_{max,a}}{a_{tot}} \tag{6.7}$$

6.3.2.4 Patch shape

Patch shape is here represented by the area-weighted mean shape index $AWMSI$ (McGarigal et al., 2012). In Equation 6.8, the patch shape index (a ratio between edge and area compared to the one of a square, that equals 4) is highlighted within brackets, while the rest of the formula represents its area weighted mean. M =number of patches, e_i = edge length of patch i. Patch shape complexity is minimal (homogeneity) when $AWMSI$ = 1 and $AWMSI \to \infty$ for complex patches.

$$AWMSI = \frac{\sum_{i=1}^{M} \left[\left(0.25 \frac{e_i}{\sqrt{a_i}} \right) .a_i \right]}{a_{tot}} \tag{6.8}$$

The natural ($AWMSI_n$) components of the $AWMSI$ are calculated as follows. M_n = number of natural patches. $e_{i,n}$ =edge of the i-th natural patch. The anthropogenic ($AWMSI_a$) component is calculated using the same method applied on anthropogenic areas.

As with the Simpson indexes (H_n and H_a), both shape indexes are weighted using total area, so that $AWMSI_n + AWMSI_a = AWMSI$.

$$AWMSI_n = \frac{\sum_{i=1}^{M_n} \left[\left(0.25 \frac{e_{i,n}}{\sqrt{a_{i,n}}} \right) .a_{i,n} \right]}{a_{tot}} \tag{6.9}$$

$$AWMSI_a = \frac{\sum_{i=1}^{M_a} \left[\left(0.25 \frac{e_{i,a}}{\sqrt{a_{i,a}}} \right) .a_{i,a} \right]}{a_{tot}} \tag{6.10}$$

6.3.2.5 Correlation between the components

In order to examine correlations between the different components of heterogeneity and wich one contributes the most to each landscape structure, a Principal Component Analysis (PCA) was performed on the 10 metrics for the 20 landscapes (Legendre and Legendre, 2012). This PCA represents each metric as an axis and each landscape as a dot. This meta-analysis is interpreted in the discussion.

A third component of configurational heterogeneity also exists: patch spatial arrangement. A choice has been made not to represent it here because the metrics measuring it are generally based on the distance between patches, which may lead to confusion as for connectivity measures (Tischendorf and Fahrig, 2000). This is further developed in the discussion.

6.4 Results

6.4.1 Number and relative abundance of land cover classes

The overall tendency is presented in Figure 6.3a and its distinct natural and anthropogenic contributions are presented in Figure 6.3b. The overall tendency shows a (convex) bell-shaped curve with a maximum at intermediate areal proportions of natural and anthropogenic land covers. The distinct contributions show opposite tendencies: ascending curve for anthropogenic land covers, descending for natural ones. Note that the scatter plot is more dispersed on the natural side (left of the graph) for natural land covers. However, confidence curves do not seem sensitive to this asymetric dispersion: they are quite far from the regression line, as confirmed by the lower coefficient of determination (Figure 6.3b). The intersection of the two curves is slightly off-centred towards the nature-dominated landscapes.

6.4.2 Patch size

The overall tendency is presented in Figure 6.4a and its distinct natural and anthropogenic contributions are presented in Figure 6.4b. The overall tendency is also a bell-shaped curve culminating at intermediate anthropisation stage. However, in the present case, the anthropogenic component is descending, while its compositional counterpart was ascending. The same contrast is found for the natural component, except that the scatter plot is less dispersed this time, which is confirmed by the narrower confidence interval displayed by the curves. There is also slightly higher dispersion of the anthropogenic component on the anthropised side. Moreover, the intersection between the two curves is here on the natural side, while it was on the anthropised side for compositional heterogeneity.

6.4.3 Patch shape

It was not possible to highlight any relationship between overall patch shape and anthropisation stage (very dispersed scatter plot, $R^2 = 0.123$, Figure 6.5a). The distinct natural and anthropogenic contributions to patch shape, however, showed clear relationships and are presented in Figure 6.5b. The tendencies between natural and anthropogenic land covers are still opposed, but this time the curves are concave. Though the quadratic terms of the curves are not significant, the R^2 coefficients were slightly better when using a quadratic regression rather than a linear one. As with compositional heterogeneity, the anthropogenic curve is ascending, and higher dispersion is found for natural land covers on the natural side. Meanwhile, as with patch size, the intersection is off-centred on the natural side.

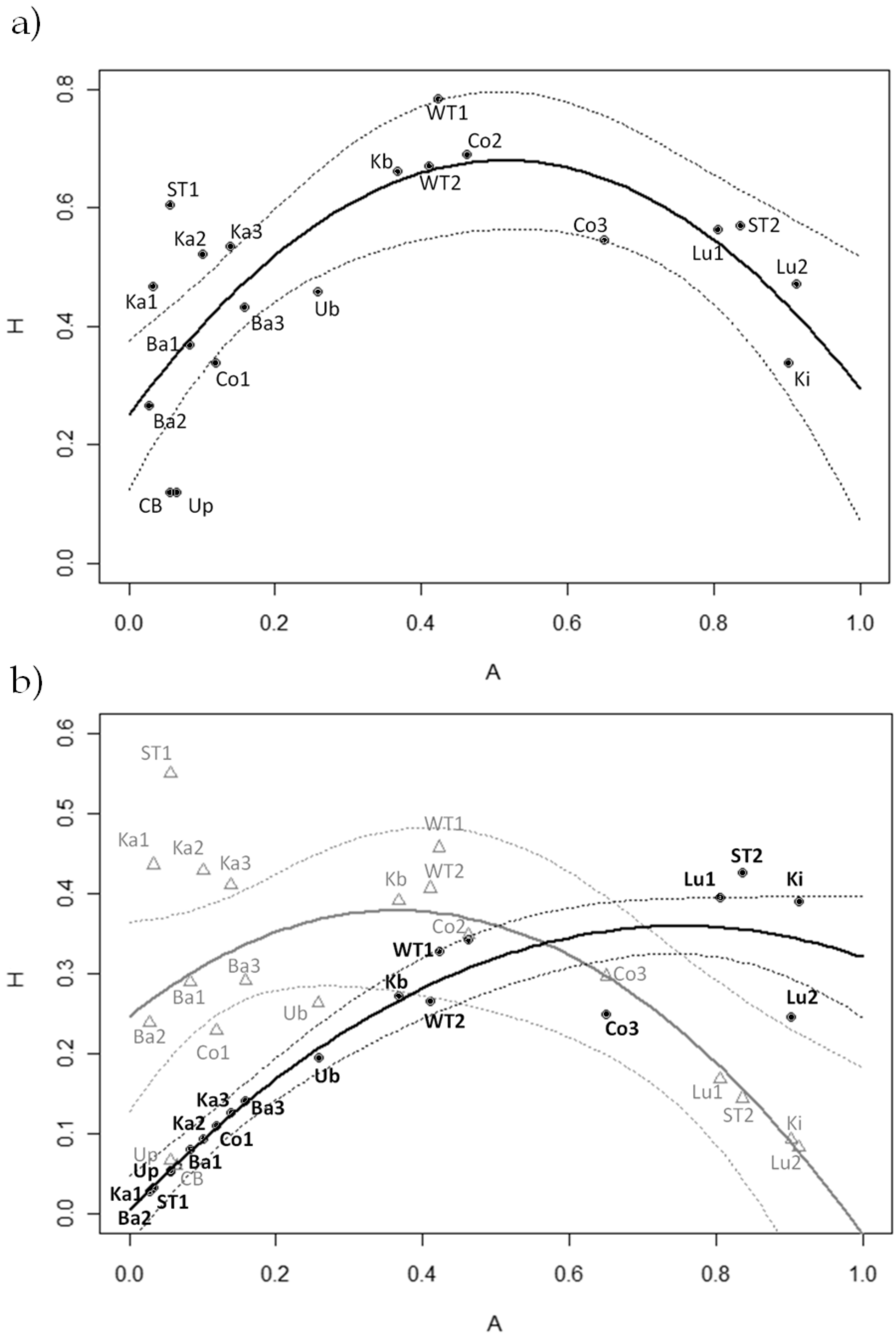

Figure 6.3 – *Relationships between overall (a) or natural vs. anthropogenic (b) compositional heterogeneity (Simpson diversity index, H) and anthropisation (anthropogenic areas, A) for the 20 studied landscapes. Dotted lines show the confidence of the regression curves. Equation of the 2nd order polynomial regression curve (a): $-1.630x^2 + 1.674x + 0.250$ ($R^2 = 0.531, F_{2,17} = 9.625, p < 0.01$), quadratic term: highly significant ($p < 0.01$). In (b), grey tiangles refer to natural land covers, while black circles refer to anthropogenic land covers. Equation of the natural 2nd order polynomial regression curve (b): $-1.001x^2 + 0.729x + 0.246$ ($R^2 = 0.383, F_{2,17} = 5.285, p < 0.05$), quadratic term: significant ($p < 0.05$). Equation of the anthropogenic 2nd order polynomial regression curve: $-0.269x^2 + 0.945x + 0.004$ ($R^2 = 0.903, F_{2,17} = 79.31, p < 0.001$), quadratic term: very highly significant (p<0.001). Data labels refer to the corresponding images (see Table 6.1).*

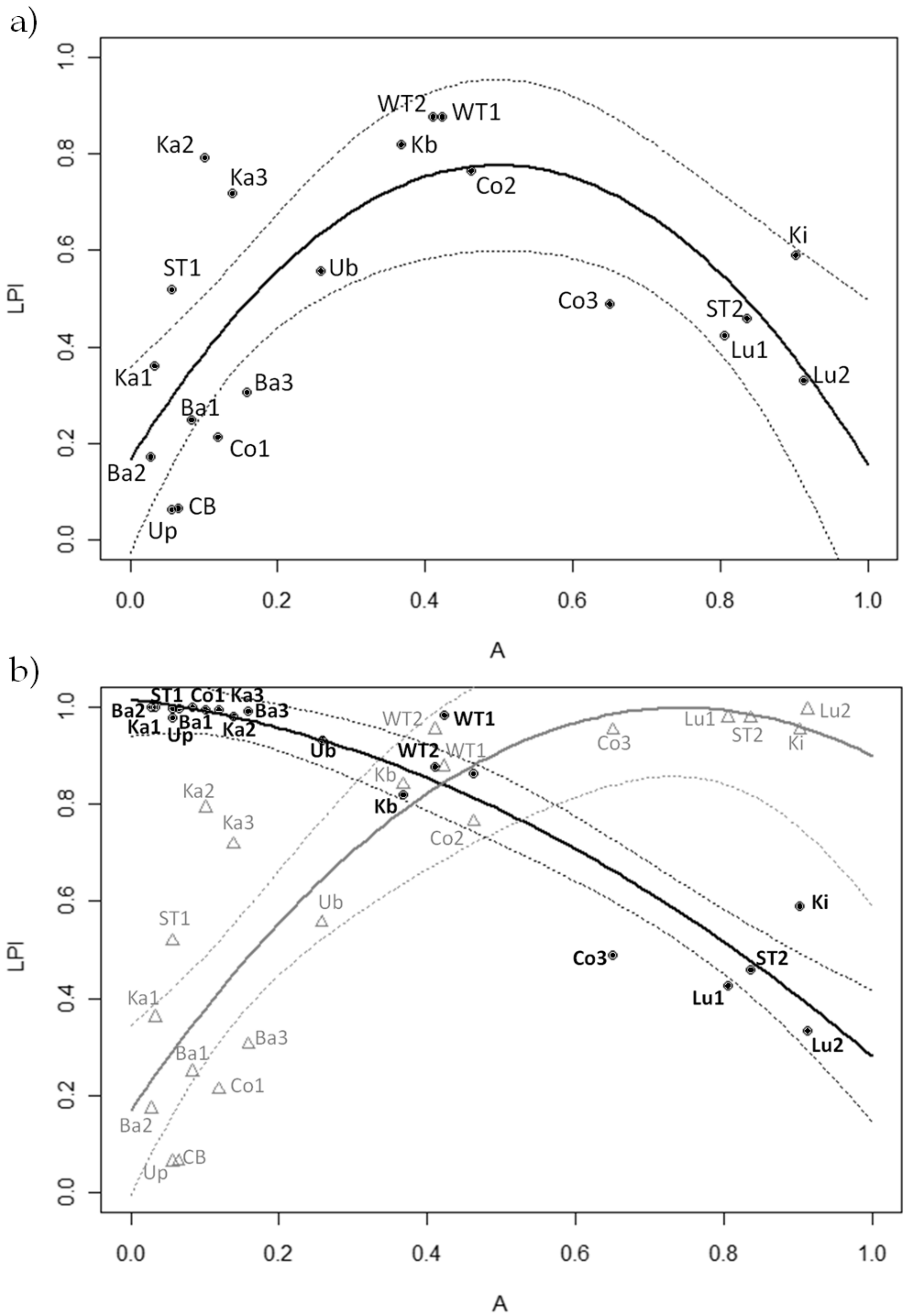

Figure 6.4 – *Relationships between overall (a) or natural vs. anthropogenic (b) patch size components of configurational heterogeneity (modified Largest Patch Index, LPI) and anthropisation (anthropogenic areas, A) for the 20 studied landscapes. Dotted lines show the confidence of the regression curves. Equation of the 2nd order polynomial regression curve (a): $-0.553x^2 + 1.182x + 1.015$ ($R^2 = 0.898, F_{2,17} = 74.84, p < 0.001$), quadratic term: significant ($p < 0.05$). In (b), grey triangles refer to natural land covers, while black circles refer to anthropogenic land covers. Equation of the natural 2nd order polynomial regression curve (b): $-1.501x^2 + 2.230x + 0.170$ ($R^2 = 0.752, F_{2,17} = 25.76, p < 0.001$), quadratic term: significant ($p < 0.05$). Equation of the anthropogenic 2nd order polynomial regression curve: $-0.553x^2 - 0.181x + 1.015$ ($R^2 = 0.898, F_{2,17} = 74.84, p < 0.001$), quadratic term: significant ($p < 0.05$). Data labels refer to the corresponding images (see Table 6.1).*

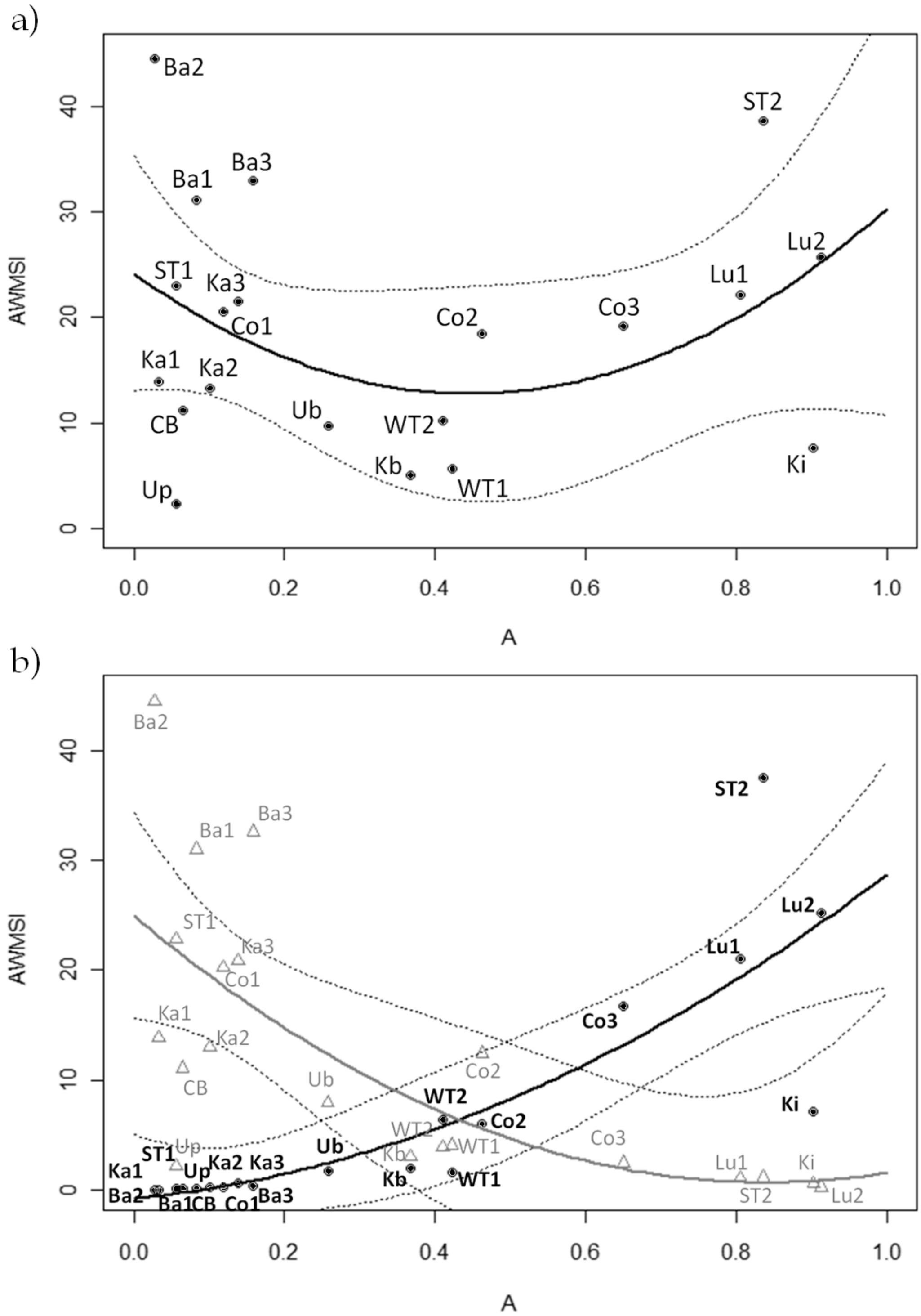

Figure 6.5 – *Relationships between overall (a) or natural vs. anthropogenic (b) patch shape components of configurational heterogeneity (Area Weighted Mean Shape index, $AWMSI$) and anthropisation (anthropogenic areas, A) for the 20 studied landscapes. Dotted lines show the confidence of the regression curves. Equation of the 2nd order polynomial regression curve (a):* $57.237x^2 - 51.086x + 24.13$ *(* $R^2 = 0.123, F_{2,17} = 1.188, p > 0.05$*), quadratic term: insignificant (*$p > 0.05$*). In (b), grey triangles refer to natural land covers, while black circles refer to anthropogenic land covers. Equation of the natural 2nd order polynomial regression curve (b):* $34.334x^2 - 57.738x + 24.923$ *(*$R^2 = 0.485, F_{2,17} = 8.005, p < 0.01$*), quadratic term: insignificant (*$p > 0.05$*). Equation of the anthropogenic 2nd order polynomial regression curve:* $22.700x^2 + 6.765x - 0.796$ *(*$R^2 = 0.705, F_{2,17} = 20.66, p < 0.001$*), quadratic term: insignificant (p<0.05). Data labels refer to the corresponding images (see Table 6.1).*

6.5 Discussion and conclusion

6.5.1 Dynamics of the 20 African landscapes

On every overall figure, heterogeneity was maximal at intermediate anthropogenic effects. The variability of the anthropogenic components is lower than for the natural components, as shown by the scatter plot dispersion as well as the distance between regression and confidence curves. The case of Cotonou (labels Co1, 2 and 3) follows a time series with dominantly natural, intermediate and dominantly anthropised landscapes. The position of the three landscapes is in accordance with the general bell-shaped curves. This tends to confirm that the synchronic - diachronic combination displays consistent results.

However, the landscape series of Bantè (labels Ba1, 2 and 3) follow atypical anthropisation dynamics: the landscape in 1986 (Ba2) seems to be slightly less anthropised than in 1972 (Ba1). However, detailed classification as well as previous works Mama (2013a); Vranken et al. (2011) show that the area underwent severe forest loss and savanisation. This contradiction can be due to methodolical shortcomings linked with the land cover classification in anthropogenic or natural categories (see section 6.5.5).

Natural and anthropogenic heterogeneity tendencies showed opposed behaviour, but the way in which they are opposed depends on the heterogeneity component. Natural land covers are more diverse from a compositional point of view, while anthropogenic land covers are more diverse from a configurational point of view.

Following the existing landscape time series, the area situated to the south-east of Tanda (labels ST1 and 2), with strong forest exploitation pressures, is the one undergoing the widest anthropisation gradient. The area of Cotonou, which is the smallest study area and where the (economic) capital city (the administrative capital city being a neighbouring city called Porto-Novo) rapidly developed since the independence in 1960, is the one with the highest heterogeneity dynamics, and also has a strong anthropisation gradient. The area to the north-west of Tanda (labels WT1 and 2) is the only one with a (slightly) decreasing anthropisation gradient over time. Though this phenomenon could be due to methodoligal shortcomings as well, it can also be explained by the spontaneous woodland encroachment on savannah that, at least in terms of anthropogenic vs. natural surfaces, compensates for the development of anthropogenic activities.

Though the scatter plots of the different components appear more dispersed on their corresponding side of the graph (i.e.: natural land covers are more dispersed for natural landscapes), there is no clear outlier. That also shows that the two landscapes in Katanga (labels Kb and Up, Table 6.1), that were characterised by lower spatial resolution, do not seem significantly affected in their metrics values. They nevertheless represent some of the lowest heterogeneity values of the scatter plots.

A theoretical model could have been used to represent the behaviour of the different components of spatial heterogeneity under growing landscape anthropisation. To do so, simulations could be developed with different neutral models such as the ones produced by the SIMMAP software (Saura, 2003). Such a model would represent the different null hypotheses to be tested in this article. The parameters corresponding to the studied heterogeneity components could be controlled to study their specific contribution to landscape structure reaction to anthropisation. For example, as for the number of land cover classes, simulations could be performed with various numbers of classes, while the areal proportions of natural and anthropogenic land covers would be progressively inverted. Such simulations would also allow to distinguish theoretical or random evolutions from site specificities of the data set; i.e. in terms of maximum coordinates, intercept, curves intersection or general shape; and help to interpret such differences.

6.5.2 Multiple components, different tendencies

The bell-shaped curves correspond to the combination (a mere addition, except for LPI, by definition) of their natural and anthropogenic components following opposed tendencies. The relatively strong symmetry of the natural and anthropogenic curves for the $AWMSI$ explains that the overall curve was almost horizontal. This highlights the importance of decomposing the heterogeneity information into relevant categories regarding the independent variable, with the risk of missing strong relationships that seem to neutralise themselves. The fact that the different heterogeneity components follow distinct, even opposed tendencies also highlights the importance of decomposing the heterogeneity information into their different components.

These dynamics can be partially explained by the spatial transformation processes acting in each landscape (Bogaert et al., 2004). At the beginning of the anthropisation process, anthropogenic patch creation, enlargement and aggregation are first observed, provoking perforation, dissection and fragmentation of the natural matrix. This enhances landscape heterogeneity (Pielou, 1975; Forman, 1995) and results in ascending overall curves (Figure 6.3a and 6.4a). So natural compositional heterogeneity decreases (Figure 6.3) while natural patches get fragmented and shrink, which increases their configurational heterogeneity (Figure 6.4a and 6.5a) (Bogaert et al., 2004; McGarigal et al., 2012).

The phenomenon continues until natural and anthropogenic proportions are equal. Then spatial heterogeneity reaches its maximum (Figures 6.3a and 6.4a) and average patch size its minimum. Natural patch types of the mosaic, characterised by intermediary proportions of core and edge areas, evolve as different ecosystems (Forman, 1995; Farina, 2000a). Natural heterogeneities (compositional and configurational) culminate at intermediate proportions of natural and anthropogenic patch types areal proportions.

Further anthropisation processes will reverse the relative dominances of the landscape: anthropogenic types become the matrix, natural patches then encounter shrinkage and attrition (Bogaert et al., 2004). These simultaneous phenomena result in overall and anthropogenic configurational heterogeneity decrease, while anthropogenic compositional heterogeneity increases (Pielou, 1975; O'Neill et al., 1988; Fahrig and Nuttle, 2005). In contrast, natural land covers decrease in compositional heterogeneity and patch shape (shrinkage, attrition), while patch size decreases, which increases heterogeneity due to patch size.

6.5.3 Methodology justification: reconciling and completing previous studies

The few authors who addressed earlier the anthropogenic effect on pattern heterogeneity did so too simply and missed a whole range of understanding of the phenomenon: they stated linear or logarithmic relationships, sometimes with contradictory versions, sometimes addressing only natural land covers, but those relationships were always one-sided (Krummel et al., 1987; O'Neill et al., 1988; Kaufmann et al., 2000; Bogaert et al., 2005). The present study integrates all those results within a broader view that allows understanding every aspects of the anthropogenic effect on landscape structure and reconciles the previous results.

Indeed, this study shows anthropisation has, in fact, two effects on heterogeneity: anthropisation can either increase or decrease it, depending on the anthropisation stage of the landscape, which confirms our first hypothesis. The previous studies were addressing incomplete time series (anthropisation gradients), different land cover types and focused on different heterogeneity components. For example, Kaufmann et al. (2000) argued that anthropogenic effect tends to decrease diversity because they were focusing on compositional heterogeneity of forested land covers, which corresponds to (and is consistent with) Figure 6.3b. As a second example, Bogaert et al. (2005) argued that anthropisation increased overall compositional heterogeneity because they focused on a landscape dominated by natural land covers, which corresponds to the left end of the curve in Figure 6.3a. This also highlights the advantage of combining synchronic and diachronic studies in terms of better representativeness of anthropogenic effects on tropical African ecosystems due to easier data set acquisition than the diachronic alternative (Baker and Billinge, 1982).

It is important to justify how data harmonisation in our multiple-sources set reduces data comparison bias . Thematic resolution influences the number of patch types, but it is important to only exclude differences due to classification and conserve relevant ecological and structural information in terms of heterogeneity (Margalef, 1958; Bamba et al., 2008). Indeed, the number of classes in landscape composition has an ecological impact on species diversity and is therefore a relevant component of spatial heterogeneity (Forman and Godron, 1986a). Extent and spatial resolution, the two components of spatial scale, have a strong influence on landscape pattern (Turner et al., 2001), which explains that we harmonised scales as much as possible. It should be noted however that resolution and extent had only a slight influence (if any) on the results in the zones where they differed (see Section 6.5.1).

The 20 images used were sampled from different African areas in order to represent the widest possible variety of anthropogenic activities and ecosystems. However, sampling activities were constrained by data and area accessibility. Therefore, the fact that natural patches are more dispersed on the natural side, which induces lower coefficients of determination (see Figures 6.3b, 6.4b and 6.5) could be linked to site-specific differences within the present images set. For example, landscapes in the rainforest of the Congo Basin (landscape 10) and in the Upemba Park (landscape 19) are both very slightly anthropised, but they correspond to different climatic contexts and display different vegetation patterns. In addition, as nature-dominated landscapes were overrepresented in our sample, landscapes presenting more anthropogenic patch types should also be included in the dataset to disentangle this problem.

6.5.4 Correlation between the components

In Figure 6.6, the two first axes of the PCA explain almost 80 % of the observed variability. The overall metrics axes (arrows) are situated between their two components (natural and anthropogenic), but not bisecting the angle: it is closer to the natural component, probably because of the slight over-representation of natural landscapes in the data set. H and LPI are strongly correlated. Indeed, the information on dominance is present in both metrics, through the relative abundance of patch types in H and through the largest patch area (especially for high-dominance landscapes, where the matrix is a single patch) in LPI.

$AWMSIa$ and $AWMSIn$ are the most opposed components. This observation is consistent with the good symmetry between the two curves: they neutralise themselves in the overall curve (see Section 6.4.3), so the overall $AWMSI$ axis has little explanatory power (shorter axis), which is consistent with the fact that no landscape is situated close to this axis. This indicates the strong dependence of natural patch shapes on anthropisation, confirmed by the close position of the $AWMSI_a$ axis to the A axis. However, H_a and H_n seem to be almost independent (perpendicular axes). That information may have important implications for conservation, at least in terms of compositional heterogeneity. LPI_a and AWMSI_a are opposed, which can be explained by the fact that smaller patches often have a simpler shape (Krummel et al., 1987).

The position of the 20 landscapes on Figure 6.6 shows which landscape structures are more driven by which heterogeneity components. Four main groups can be distinguished. One group is strongly determined by the anthropogenic component (A axis) and is formed by the most anthropised landscapes. The other ones are dispersed around *LPI* and H; LPI_a and H_a; *AMWSI* and AWMSI_n. Landscapes belonging to the same time series seem to be close to the same axis direction, except Cotonou, which underwent strong anthropisation and heterogeneity dynamics during the period of study.

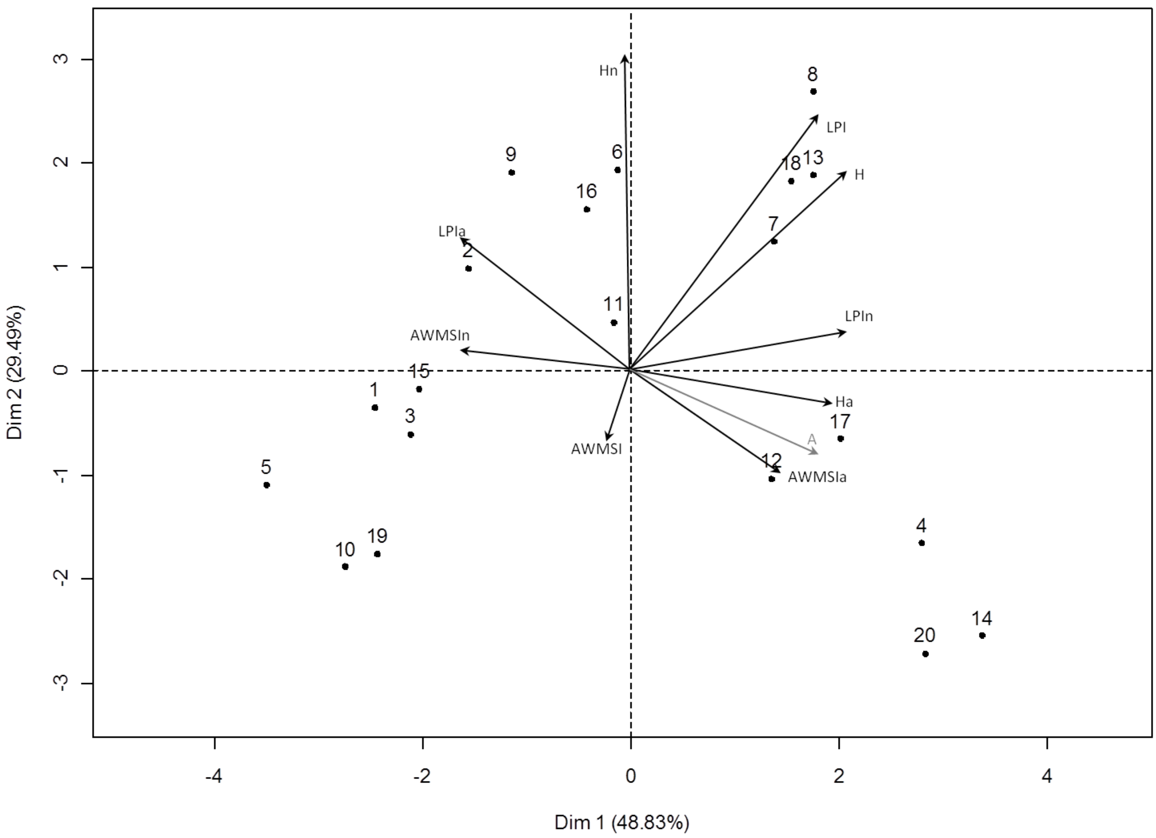

Figure 6.6 – *Correlation between overall, natural (n) vs. anthropogenic (a) components of standardised presented metrics (arrowed axes, see Section 6.3.2) applied to the 20 studied landscapes (dots in the scatter plot). Anthropisation (A, grey axis) is used as illustrative variable. Based on Principal Component Analysis. Percentage of explained variance by the third component: 16.32 %.*

6.5.5 Metrics limitations

The A index is easy to compute and use in order to study the relationship between anthropogenic effects and any other variables, especially for different data formats. Such a binary simplification eased computation and representation of the scatter plot and was required considering the amount of metadata available for the different study zones, but some moderately anthropised land covers or patches may have been misclassified due to this simplification. Moreover, this metric allows to distinguish similar patch types of different origins from one landscape to another, but not within the same zone. For example: savannah may be of natural origin in the North of Benin, but not in Lubumbashi, while in the forest-savannah transition zone of Ivory coast, some savannah patches may be natural while some other may be due to deforestation (Barima, 2010; Mama, 2013b). This shortcoming is probably also the origin of the unexpected anthropisation sequence in Bantè (see 6.5.1). Indeed, part of the local savannah is anthropogenic and replaced the forest in the area, but the classification did not allow distinguishing natural from anthropogenic savannah. Therefore, as savannahs were classified as natural, the savanisation phenomenon highlighted in Benin (Vranken et al., 2011) was completely missed, which returned inconsistent results for that landscape. This problem can be circumvented by choosing for a scaled anthropisation measurement instead of a binary classification, by studying patch dynamics to distinguish regressive from evolutionary series (Vranken et al., in preparation-a) or solved by better classification quality to distinguish the savannah types, which would require higher sampling efforts in the field (André et al., in revision).

The latter case may bias information on anthropogenic area proportions or diversity in some areas. For instance, in our data set, multiple origins of the same patch type were observed in landscapes 2, 6 and 16 in Benin (see Table 6.1), where savannah results from both climate change (partly and indirectly anthropogenic) and woodcuts (directly anthropogenic) for charcoal production (Mama, 2013a). Here, it also can be seen that anthropogenic effects may more or less directly affect patch types. Therefore, instead of being merely classified on the binary anthropogenic or natural side, land covers could be classified according to anthropogenic effect intensity, as with the hemeroby index (Steinhardt et al., 1999). In these conditions, it is possible that the area southwest of Tanda (landscapes 8 and 13, Table 6.1) will not be seen as undergoing an anthropisation decrease (see Section 6.5.1). How to refine the anthropisation quantification is further investigated in Vranken et al. (in preparation-a).

H is simple to compute and interpret as a spatial heterogeneity index. As richness represents relevant ecological information , we chose to use Simpson diversity and not evenness. It should be noted however that the number of classes has a stronger influence when comparing different sites than observing landscape dynamics on the same site: the complete loss of a patch type (attrition) is an extreme or ultimate destructive process less probable than areal proportion change. Simpson has been used instead of the Brillouin index, considered more precise when applied to a single zone, because here each of the 20 landscapes is considered as a sample (Pielou, 1975; Bogaert et al., 2005). The Simpson diversity index is also commonly called heterogeneity index (Bogaert et al., 2005). In order to avoid any confusion, it should however be noted that heterogeneity in this context does not represent heterogeneity according to its general meaning in landscape ecology, as exposed in the introduction, but only its compositional component.

The original LPI is a simple proxy for dominance, the modified version used here is directly representative for the patch size component of configurational heterogeneity (McGarigal et al., 2012). Its drawback is sensitivity to image resolution, but this is mitigated by the fact that it measures the largest and not the smallest patch. The $AWMSI$ has been chosen to represent the patch shape component of configurational heterogeneity because of its easy interpretation. The area weighted mean makes it more representative for the patches that dominate the landscape, considered as the ones with the highest ecological relevance in the landscape (McGarigal et al., 2012). The counterpart is that it reduces the influence of a large number of patches (small patches are often numerous) and that it introduces a correlation with landscape dominance, homogeneity (see Figure 6.6).

It is important to remind here that the aforementioned metrics were used for general pattern assessments, only assessing landscape state, and were not adapted to species with a specific mobility behaviour. The specific behaviour and metric values they present are then not to be taken as is for specific assessments or functional studies.

The choice of not representing patch spatial arrangement in order not to mix it with connectivity should also be justified. Closer patches ensure higher connectivity, but physically connected habitat are represented as a single patch, so this connectivity is not taken into account by such indexes. The Aggregation Index (McGarigal et al., 2012), based on pixel adjacencies, is not sensitive to that problem and would therefore be suitable to represent patch spatial arrangement, but it appears very strongly correlated to the shape index (*AWMSI*) and was criticised for that reason (Bogaert et al., 2002a). This index as well as the area weighted mean nearest neighbour distances were however tested on this data set (data not shown). The two metrics, though adapted to give higher values for higher "connectivity", followed opposed tendencies, as expected for the aforementioned reason. They also showed opposed tendencies from natural to anthropogenic land covers. In a general context, computing spatial arrangement metrics is of little value given the above, but in specific contexts, when studying targeted organisms, adding functional aspects to connectivity (connectedness, see Acosta et al. (2003)) should cope with the problem. It is therefore also important, in specific contexts, to examine the link between anthropisation and connectivity, which is of crucial ecological relevance and at the center of the research following the pattern / process paradigm.

6.5.6 Why heterogeneity-anthropisation relationships are worth addressing in targeted analyses

It has been proven in our study that, when addressing the impact of heterogeneity or the anthropogenic impact on species distribution, failing to study the anthropogenic effect on heterogeneity dynamics themselves removes an important perspective of understanding and environmental management for three reasons. First, general heterogeneity assessments showed a differentiated effect of anthropogenic disturbance depending on the anthropisation stage of the landscape studied, which is explained by spatial transformation processes and dominance studies (see p. 101). The anthropisation stage is of particular matter in tropical Africa, which, according to the 20 studied landscapes, still shows a large anthropisation range and strong landscape changes. The second argument in favour of addressing the heterogeneity-anthropisation link is that natural anthropogenic patch dynamics differ, which had not been previously highlighted. Thirdly, and most importantly, the different components of pattern heterogeneity (patch size, shape, spatial arrangement) respond in totally opposed ways to anthropogenic pressures, which was partially noted by Mladenoff et al. (1993).

Though the quantifications were here performed on general studies as they were before criticisms such as Cale and Hobbs (1994); McIntyre and Hobbs (1999); Tischendorf and Fahrig (2000), the observations, for their qualitative and comparative value, suggest that, for targeted populations sharing the same behaviour, differentiated anthropogenic effects on pattern heterogeneity dynamics should also exist. Given that, the present article can be the starting point for studying the relationship between anthropisation and heterogeneity in specific studies, instead of focusing either on anthropogenic effect or heterogeneity separately, as it is currently performed in the majority of cases.

These three observations should not only explain contradictory responses to anthropisation from the same group of species, but also help predicting future impact tendencies, depending on the current stage. The differentiated impacts on the heterogeneity components should also help understanding the distribution of targeted species group, that can be more sensitive to one component than to another.

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Chapter 7

The spatial footprint of the non-ferrous mining industry in Lubumbashi

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See also Appendix C.

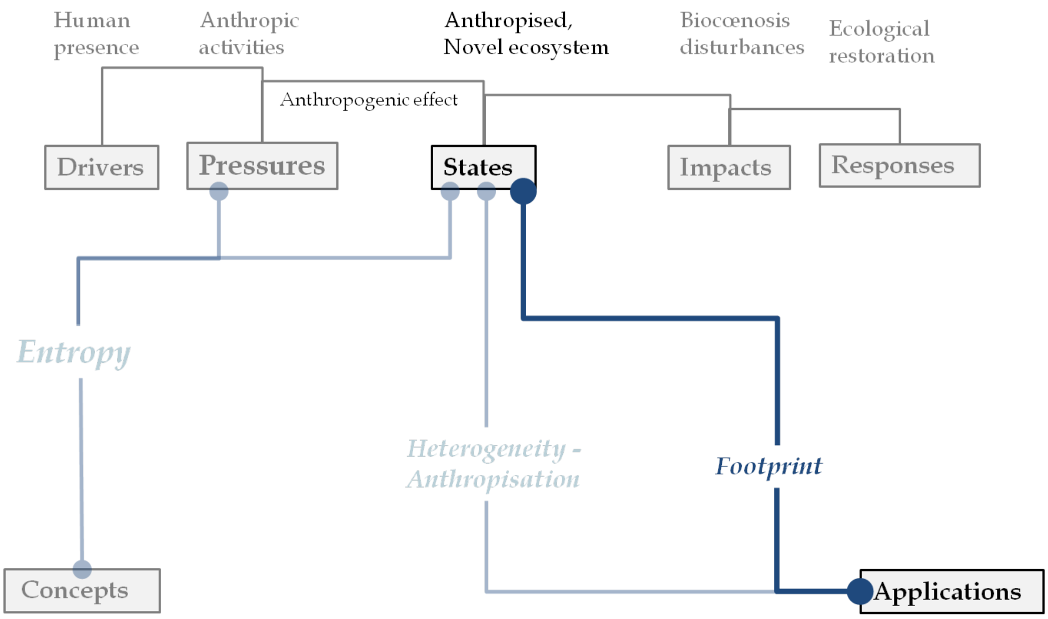

Figure 7.1 – *Location of the footprint article (deep blue theme) in the thesis mind map. Technical reading: states, thematic reading: heterogeneity (question of Section 1.2.2), logical reading: state. The hollow blue articles are thematically related to this one. This chapter uses spatial pattern assessment to test the hypothesis that it results from anthropogenic pressures.*

7.1 Summary

In the south-eastern part of the Katanga Province (Democratic Republic of the Congo), high concentrations of copper and cobalt are found in the soils of the well-known "Copper Belt". Due to dominant south-eastern winds, the metallurgic industry in Lubumbashi has been the source of spatially concentrated atmospheric deposits of non-ferrous metal particles and associated substances in a cone-shaped zone, situated north-west of the metal processing site. The existence of this zone has been evidenced using two different techniques: firstly, by means of landscape metric comparisons of the vegetation and bare soil patterns in two study areas, one inside the pollution cone and one outside; secondly, by means of the city perception theory developed by Kevin Lynch. Higher fragmentation and lower vegetation presence were observed inside the pollution cone, reflecting the negative impact of the atmospheric deposits. Those differences were higher for sites closer to the emission source. Lynch's approach outlined the negative impact of diverse industrial plants on the perception of the local population. Six pollution districts and several contaminated paths, limits, nodes and polluting landmarks were identified. Citizens even recognize them as part of the collective image of the city.

7.2 Résumé

Dans le sud-est du Katanga (République Démocratique du Congo), de hautes concentrations de cuivre et cobalt sont présentes dans les sols de l'Arc cuprifère. Suite à des vents dominants des secteurs sud-est, l'industrie métallurgique Lubumbashi a été la source de dépôts atmosphériques de métaux non ferreux et substances associées, concentrés en une zone en forme de cône au nord-ouest de l'usine. L'existence de ce cône a été démontrée par deux techniques: premièrement, au moyen de comparaisons d'indices de structure spatiale de la végétation et des sols nus; et ensuite avec la théorie de la perception urbaine développée par Kevin Lynch. Une fragmentation plus importante et une présence plus faible de végétation ont été observées dans le cône de pollution, ce qui reflète l'impact négatif des dépôts atmosphériques. Ces différences étaient plus élevées pour les sites plus proches de la source d'émission. L'approche de Lynch a mis en évidence l'impact négatif de plusieurs usines sur la perception de la population locale. Six quartiers de pollution et plusieurs voies, limites et nœuds contaminés ainsi que des points de repères polluants ont été identifiés. Les citoyens les reconnaissent même comme partie intégrante de l'image collective de la ville.

7.3 Introduction

Soil contamination by atmospheric deposits of non- ferrous metals has been described for ecosystems worldwide (Vangronsveld et al., 1995; Barcan and Kovnatsky, 1998; Ginocchio, 2000; Raven et al., 2000). In the south-eastern part of the Katanga Province (Democratic Republic of the Congo), high concentrations of copper and cobalt are present in the soils of the well-known "Copper Belt" (Faucon, 2009). This zone has been a place of intensive mining in the colonial period itself (until 1960) and afterwards, when mining activities and pyrometallurgic industry were continued by the Gécamines Company (Banza et al., 2009).

The analysis of landscape patterns is justified by the pattern/process paradigm, a central hypothesis in landscape ecology (Turner, 1989) linking emerging patterns to underlying processes. Landscapes close to metal processing sites generally display a distinct pattern of scattered vegetation patches embedded in a matrix of bare soil (Montgomery, 2003; Kozlov and Zvereva, 2007). Due to the dominant south-eastern winds, the metallurgic industry in Lubumbashi, capital of the Katangese Copper Belt, has been the source of spatially concentrated atmospheric deposits of non-ferrous metal particles and associated substances in a cone-shaped zone or "pollution cone", situated north-west of the metal processing site and characterised by degraded vegetation (Chapelier, 1957; Leblanc and Malaisse, 1978; Malaisse, 1997). At the landscape level, the footprint of this long-term metal processing activity should therefore be detectable as a zone with higher vegetation fragmentation and higher bare soil presence. Unfortunately, this footprint, and hence the existence of the pollution cone, has not yet been evidenced using landscape metrics (Bogaert and Hong, 2004). Therefore, this paper compares the patterns of vegetation and bare soil of two oppositely placed study areas, one inside the pollution cone and one outside.

According to Kevin Lynch (1960), environmental perception or legibility is central for every living creature capable of motion because perception determines the way they exploit their environment to subsist, depending on their movements across the landscape. Features leading to stronger images for citizens (high "imageability") then form the urban system, the analysis of which should be at the base of urban design (Lynch, 1960). Industrial landscapes are an issue of concern when a perceptive approach is applied since industrial residues may spread far beyond the industrial infrastructure itself, e.g., bare soils resulting from vegetation degradation due to environmental contamination (Lynch, 1960; Neuray, 1982; Amisi, 2010). The application of the Kevin Lynch theory (Lynch, 1960) is considered complementary to the aforementioned analysis based on landscape metrics, and is hence the second objective of this paper. Evidencing the ecological footprint of the non-ferrous mining industry is considered crucial to confront local decision makers with the negative impact of metallurgic industry within an urban context.

7.4 Material and methods

7.4.1 Landscape metrics

A map of Lubumbashi containing four land cover classes (vegetation, bare soil, built-up, water) based on a Quickbird image of 2005 was used (Munyemba et al., 2008; Munyemba, 2010); spatial resolution was set to 40 m to fit the recommended 30-100 m resolution range (O'neill et al., 1999) for pattern analysis. Two regions of interest (ROI) of 79 km^2 were defined and compared, one inside the pollution cone, situated north-west of the emission source, i.e. inside the deposit range (Koptsik and Koptsik, 2001) and one outside the pollution cone, situated south-east of the emission source (Figure 7.2). Each ROI was divided in three subzones of 13 km^2, 26 km^2 and 52 km^2 (with the smallest subzone closest to the emission source) in order to enable the detection of the influence of spatial scale. Total area, number of patches, average patch size and area of the largest patch of the vegetation and bare soil classes were calculated and noted as $a_v, n_v, \overline{a}_v, a_{max,v}$ and $a_s, n_s, \overline{a}_s, a_{max,s}$ respectively.

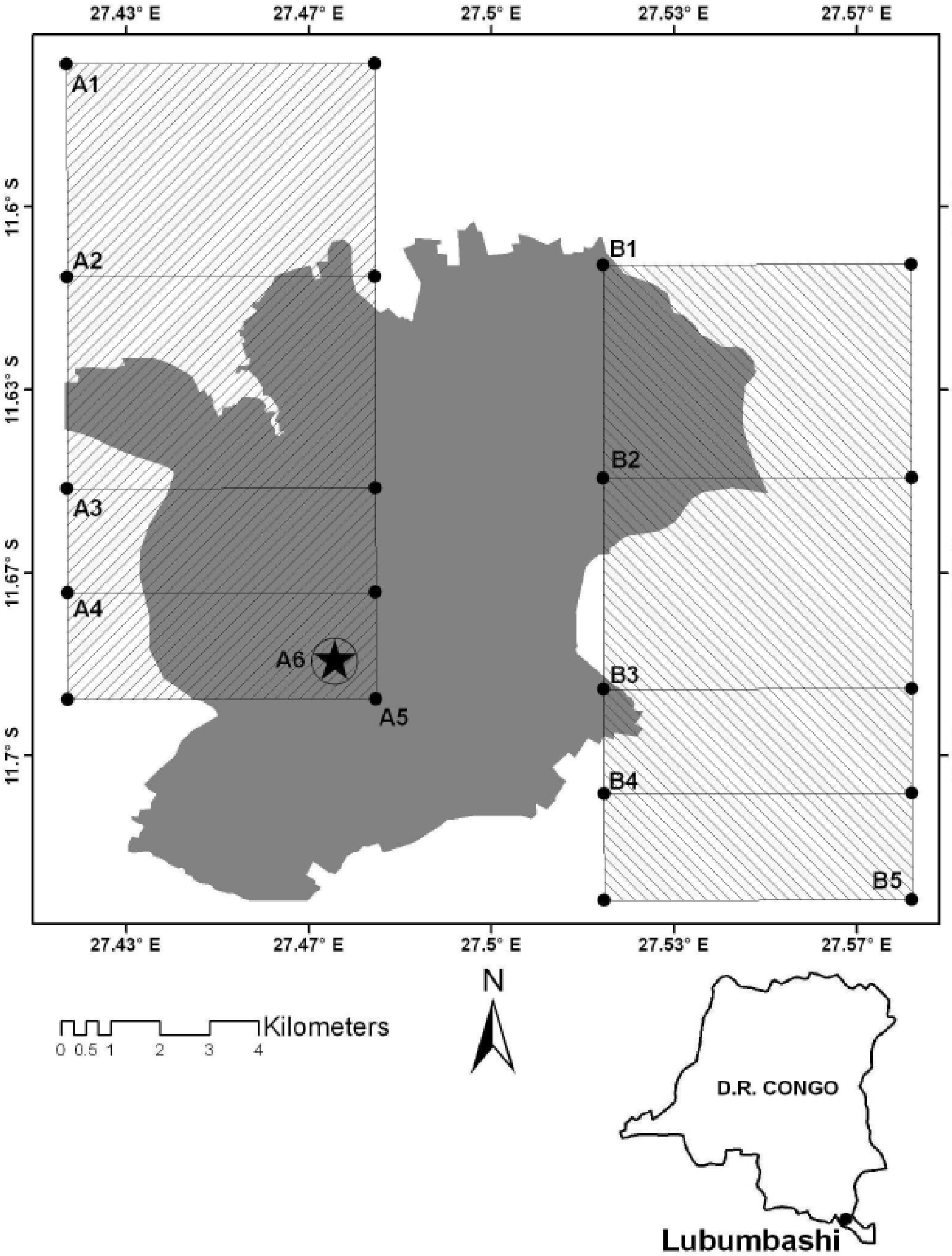

Figure 7.2 – *Map of the two study areas in Lubumbashi situated oppositely of the emission source (Gécamines smelter), indicated by a star symbol (A6). Each study area of 79 km^2 (rectangles A1/A5 respectively B1/B5) contains three subzones of 13 km^2 (rectangles A4/A5 respectively B4/B5), 26 km^2 (rectangles A3/A5 respectively B3/B5) and 52 km^2 (rectangles A2/A5 respectively B2/B5). As a consequence of the prevailing south-eastern winds, the pollution cone is expected to be situated north-west of the emission source. The grey zone indicates the central part of Lubumbashi, including the following municipalities: Katuba, Kampemba, Lubumbashi, Kamalondo, Kenya and Ruashi.*

To avoid any influence of the absolute areas of vegetation and bare soil on pattern measurement and to exclude non-pollution related differences between the zones, two ratios were calculated to compare the relative presence of vegetation and bare soil. Equation 7.1 expresses the dominance of vegetation over bare soil ($R_1 > 1$) or the dominance of bare soil over vegetation ($R_1 < 1$):

$$R_1 = \frac{a_v}{a_s} \tag{7.1}$$

R_1 is expected to be lower inside than outside the pollution cone because of the higher presence of bare soils due to toxic deposits.

Secondly, R_2 (equation 7.2) compares the average patch sizes of both classes:

$$R_2 = \frac{\overline{a}_v}{\overline{a}_s} \tag{7.2}$$

R_2 is expected to be lower inside the pollution cone and higher outside because of the supposed higher vegetation fragmentation in the cone.

The fragmentation degree of both land cover types was also measured by the index of the largest patch (Bogaert et al., 2002b) expressing the dominance (%) of the largest patch [D_v for vegetation (equation 7.3), D_s for bare soil (equation 7.4)] inside its class:

$$D_v = \frac{a_{max,v}}{a_v} \tag{7.3}$$

$$D_s = \frac{a_{max,s}}{a_s} \tag{7.4}$$

D_v was expected to be higher outside the pollution cone and D_s was expected to be higher inside the pollution cone.

7.4.2 Perception analysis

In order to analyse the "imageability" and legibility of the city, cognitive cartography was realised. The study was carried out in 2007 and inscribed inside the agglomeration of Lubumbashi according to perceived limits empirically defined during field prospection (Amisi, 2007). Two different methods were used, the first approach consisted of the identification of the structuring elements composing the city images: edges, paths, nodes, landmarks and quarters (Lynch, 1960; World-Bank, 2005).

Point edges (Amisi, 2007; 2008; 2010) have also been added to adapt the theory of Lynch (1960) to developing countries, where cities are less frequently limited by easily identifiable boulevards and where certain landmarks can be considered as edges. By walking, cycling or driving across the city, the presence, visibility, and interrelations between the structuring elements have been examined by four trained experts. Their observations are considered as repetitions of the observations made by a single person and hence form the "map of the observer" (Amisi, 2008). The second approach consisted of interviewing one hundred citizens. Sample size was verified using a saturation-accumulation curve (data not shown) in order to test whether all relevant elements had been cited. The point at which this curve, representing number of elements vs. number of individuals interviewed, levels off is accepted to correspond to the minimum sample size required (Horn, 1993; Amisi, 2010). To avoid any bias due to the interviewing sequence, 99 randomisations of this sequence were produced using the EstimateS software, which generated accumulation curves for each element with a 95% confidence limit (Colwell, 2005).

Citizens were firstly asked general questions about the image evoked by Lubumbashi. Consequently, they were requested to describe the paths they usually took to walk across the town, and put this into words as if they had to explain it to someone not familiar with Lubumbashi. This method allows an identification of the city elements from a citizen's point of view and the affect he associates with them. Secondly, pictures of representative elements of Lubumbashi, including metallurgic infrastructures or sites close to them where shown. The people interviewed were asked to describe their feelings about the metallurgic industry in their city, and were requested to identify which of the six aforementioned structuring elements allowed to recognize such sites and their environmental impact. All cited elements were used to compose a map forming the collective image of the city.

The map of the observer and the collective map were compared using the Sørensen similarity index S_s (Jaccard, 1900; Sørensen, 1948; Legendre and Legendre, 2012) (equation 7.5):

$$S_s = 100\frac{2a}{2a+b+c} \tag{7.5}$$

with a the number of elements present in both maps, and b and c the number of elements mentioned in only one of both maps. A chorema (Brunet, 1980) combining the elements linked to the mining industry and characterised by environmental impact from both collective and observer's maps was also produced to highlight the negative impact of the mining industry on the town and to compare it with the aforementioned spatial statistic approach (Brunet, 1980; Munyemba, 2010; Amisi, 2010)

7.5 Results

7.5.1 Landscape metrics

Inside and outside the pollution cone, a dominance of vegetation over bare soil ($R_1 > 1$) was observed, but this dominance was higher outside ($R_1 \approx 3.5$) the pollution cone than inside ($R_1 \approx 2$), as expected (Figure 7.3a). The average patch area of vegetation was found to be higher than the average patch area of bare soil both inside and outside the pollution cone ($R_2 > 1$; Figure 7.3b), but the difference was more pronounced outside ($R_2 \approx 13$) than inside the pollution cone ($R_2 \approx 5$).

Hence, vegetation patches are actually smaller inside the cone than outside. The relative positions of the curves representing the fragmentation metrics (Figure 7.3) confirm that vegetation cover is less fragmented outside the pollution cone than inside; analogously, bare soil is less fragmented inside the pollution cone than outside. The metric curves with respect to the inside and outside of the cone come closer to each other as the study zone extents increase. Hence, the differences between vegetation and bare soil patterns decrease when considering larger study zones, including areas located further away from the emission source (Munyemba et al., 2008; Munyemba, 2010; Vranken, 2010).

7.5.2 Perception analysis

Accumulation curves (data not shown), showed that after 45 interviews all city structuring elements had been cited (Amisi, 2007; 2008; Vranken, 2010). No significant elements have consequently been ignored.

Comparison of the elements composing the observer's map with those composing the collective map evidences a high similarity (Table 7.1) when compared to Lynch (Lynch, 1960). The perceived elements which could be linked to the negative (visual) impact of

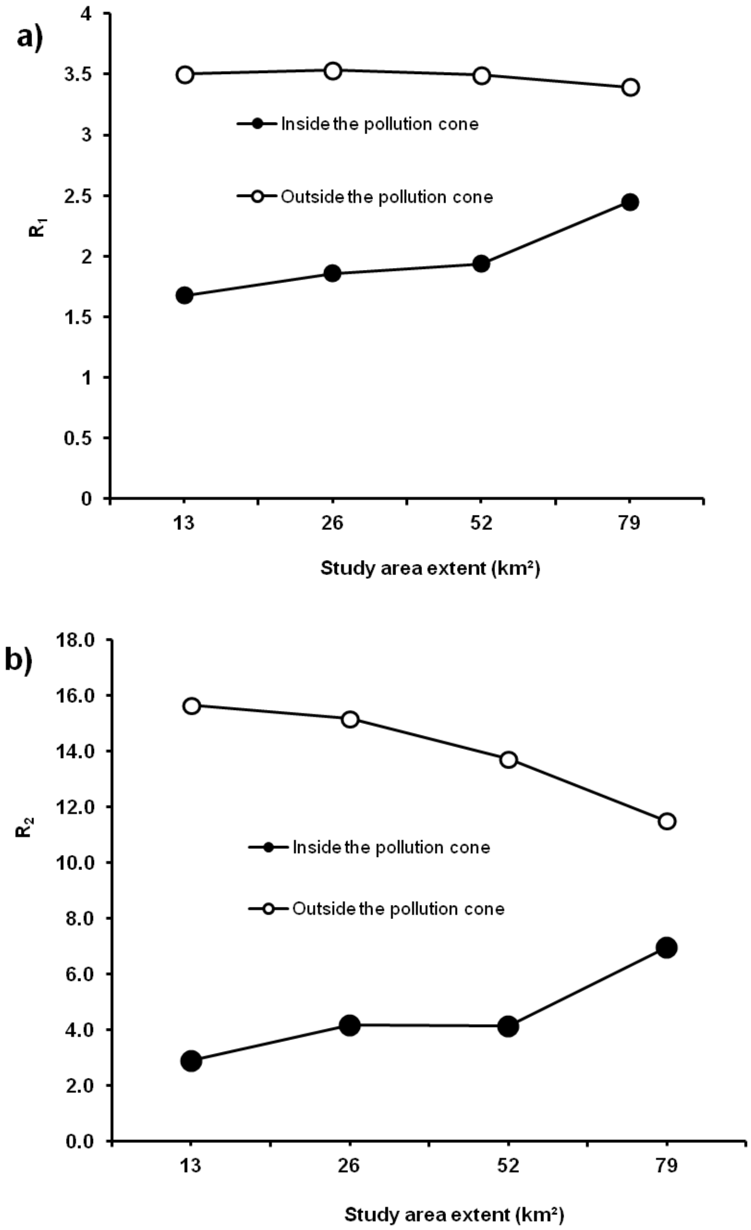

Figure 7.3 – *Pattern analysis of vegetation and bare soil by means of landscape metrics in Lubumbashi. Two study areas are compared, one situated inside the pollution cone, one situated outside. (a) Dominance of vegetation or bare soil in the study areas as measured by the ratio of the area of the vegetation to the area of bare soil (R_1). (b) Comparison of the average patch area between vegetation and bare soil by means of the ratio R_2.*

mining (in both the observer's and the collective maps) have been represented on a synthetic chorema (Brunet, 1980) (Figure 7.4).

Each of the six elements composing the image of the city was reported as environmentally altered by the mining industry. The map clearly identifies the pollution cone linked to the activity of the Gécamines industry, and more recently the STL plant, situated on the same site. Nevertheless, it also shows that many other mining companies installed in the periphery of the town induce perceived environmental impacts. In the northern part of the city, recent industries, i.e. South China Mining, Zong Hang, Zhong Hua, Cota Mining, Ngapura Mining, Huachim Society, Congo Loyal and Congo Julian were cited. In the eastern part, Ruashi mining is situated in the centre of a large mining

Table 7.1 – *Number of elements composing the image of Lubumbashi, for the trained investigators (observer) and the interviewed citizens (collective). S_S: Sørensen coefficient of similarity between both maps.*

		Edge	Point edges	Paths	Nodes	Landmarks	Districts	Total
Observer	Elements	10	7	65	54	45	19	200
Collective	Elements	8	5	63	52	49	19	196
Common	Elements	7	5	59	44	40	19	174
S_s	%	77.8	83.3	92.2	83.0	84.9	100.0	87.9

district. Other industrial plants were also mentioned, situated between the railway and the Naviundu River: Chemaf, STVD, Ciment Kat and Exaco. Even if most of those recent industries only occupy the peripheral zone of the city, their impact is also mentioned in the city centre, notably through the citation of transportation axes and rivers which were reported to be contaminated by these mining activities; the embankment areas of two out of five rivers (Naviundu and Lubumbashi) were also reported as polluted. Polluted drinking water was mentioned in the interviews as well. Decantation pools and fumes were denoted as pollution sources. The presence of bare soils and metallophyte vegetation were frequently cited as consequences of those polluting activities. Next to the different metallurgic plants from which the city originated, contamination and pollution are nowadays part of the image of the city of Lubumbashi.

7.6 Discussion

Higher vegetation presence and lower fragmentation outside than inside the pollution cone confirmed our hypothesis. This is consistent with observations from soil science and botany carried out in the area, though their extent could not be made as large as those in this study for logistic reasons. Indeed, higher copper and cobalt concentrations were found in surface horizons of soils (Shutcha et al., 2010) and land cover changes have been observed over time, replacing miombo with savannah, metalliferous grass species, and bare soils for the extreme contamination cases (Chapelier, 1957; Leblanc and Malaisse, 1978; Mbenza et al., 1989; Malaisse, 1997; Leteinturier et al., 1999; Shutcha et al., 2010). When both study zones are enlarged, they appear to become more similar. This may be understood because the area extensions are situated further away from the emission source and are less influenced by atmospheric deposits. This tends to confirm that vegetation is affected by those deposits, as vegetation cover increases when deposit amounts and densities decrease with distance. Furthermore, pattern change with changing extent is common in large scale ecological studies since pattern features have been found to appear or disappear with changing spatial scales (Benson and Mackenzie, 1995; Farina, 2000b).

The differences observed between the maps composed by trained observers, on one hand, and the collective maps, on the other hand, are assumed to be originating from differences in instruction level between local populations and experienced investigators (Amisi, 2007; 2008). Since the social contrast between rich and poor citizens is generally larger in developing countries compared to developed ones (World-Bank, 2005), a lower similarity could be expected, as evidenced by this study. Indeed, mobility, or "motility", i.e. the ability to move across a city, using different means of transportation, or to find one's way

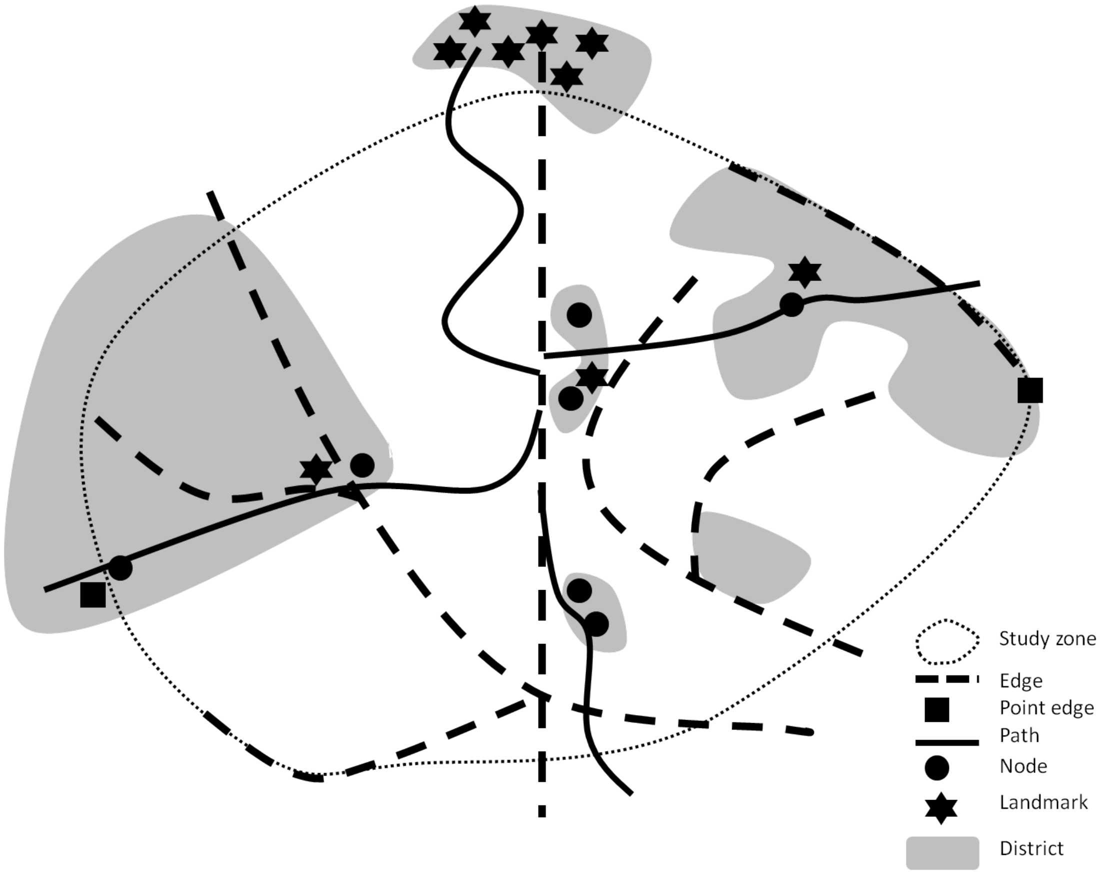

Figure 7.4 – *Chorema of the perceived negative impacts of mining and industry in Lubumbashi (synthesis of observers' and collective maps). Each of the six elements composing the city has been represented according to its association with perceived environmental impacts related to mining and/or metallurgic industry. Edges are contaminated main roads or rivers, point edges being industrial plants. Landmarks are also industrial buildings or plants. Paths correspond to the main roads to mining sites. Nodes are often mining sites connected to paths related to pollution. Districts are mainly characterised by contamination, deforestation, bare soils and metallophyte vegetation. Northern zone: district of recently developed industries such as South China Mining, Zong Hang, Zhong Hua, Cota Mining, Ngapura Mining, Huachim Society, Congo Loyal and Congo Julian. Eastern zone: district with Ruashi mining zone and metal processing factories. Central zone, situated in between the railway (West) and the Naviundu River (East): districts marked by presence of Chemaf, STVD, Ciment Kat, and Exaco. Western zone: Gecamines and STL industrial plants with their pollution cones.*

on maps, is reported higher for upper social levels, having higher so-called "socio-spatial" competences (Bret, 2009; Orfeuil, 2004).

The current study shows that Kevin Lynch's urban picturesque analysis methods are applicable to developing countries as well when slightly adapted. New concepts like point limits have been introduced to enable this, but the method remains sufficiently general to apply elsewhere as well. A major critique on Lynch's work was the small number of interviewed individuals to establish cognitive representations of the city (Lynch, 1960). To avoid this drawback, Kevin Lynch suggested a huge census, which would have been harsh to conduct. Therefore, the accumulation curve method used in the current study allows to compose a complete collective image of the city and, at the same time, with a limited sample size (Amisi, 2007; 2008). It is also interesting to note that, although mining zones and roads were also perceived as major structuring elements of the town itself and of its development (Amisi, 2007; 2008; Vranken, 2010), they were also perceived as highly polluted.

7.7 Conclusion

Landscape analysis using pattern metrics confirms the hypothesis that, due to prevailing south-eastern winds, higher concentrations of non-ferrous metal particles have been deposed north-west of the Gécamines metal processing site, causing lower relative vegetation presence and higher vegetation fragmentation. The area and cost-efficiency of remote sensing and pattern analysis, complementary to specific on-site investigation, is also highlighted in this study, as previous small extent botanical or soil studies, though abounding in this sense for years, have not been able, till now, to evidence the very presence of a uniform and large pollution cone downwind of the Gécamines plant. Still, its depth and widths remain to be defined. Application of Kevin Lynch's methodology taught that (industrial) mining activities largely influence the image of the city, also outside the pollution cone. The location of new mining and metal processing activities in what is currently the city periphery, will not solve the problem, since urban sprawl due to population growth will establish new contact areas between habitat and industry (Nzuzi, 1991; Khonde and Rémon, 2006).

Evidencing the existence as well as the ecological and human impacts of the pollution cone is crucial in convincing local decision makers of the negative impact of metallurgic industry in urban living conditions and perception.

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Part IV

Discussion

CHAPTER 8

Answering the questions

As has been developed in Chapter 2, anthropisation has major impacts on ecological functioning that affects biodiversity (Vranken et al., 2011). Landscape ecology studies these impacts by measuring the consequences of human activities (pressures) on landscape composition (deforestation, agriculture), but also configuration (roads) by assessing typical spatial transformation processes occurring on natural land covers during the anthropisation process. However, what most of these assessments (except the study on rodents) were actually measuring is landscape spatial structure, basing their ecological impacts on hypotheses built on typical pattern / process relationships (Turner, 1989; Ness et al., 2010). The interpretations in the next chapters highlight some situations where deductions based on the pattern / process paradigm may be overreached and which conceptual amendments or interpretation advice could be provided in order to prevent such overreaching.

In this general discussion, we try to reach a further level of abstraction by federating the different contributions to answer questions that hold together within a transversal framework. This chapter is structured according to the thematic reading (see Section 1.2). This discussion can be understood without reading the previous articles thanks to the glossary, p. 133.

8.1 Anthropised landscape: a state conditioned by its measurement

This section answers the first question of this thesis (Section 1.2.1, p. 4): "What is landscape anthropisation and how can it be quantified? "

8.1.1 Towards a new framework

This section answers the first objective of this thesis: define landscape anthropisation and associated concepts, evaluate its quantification methods and propose an upgrade (Vranken et al., in preparation-a), see p. 4. The *Anthropisation* publication (Chapter 3) showed four main results. (1) A plethora of concepts were used to describe different aspects of anthropisation from different disciplines, referring to system states, pressures and ecological impacts without distinction. (2) Anthropisation quantification strongly depends on the reference states, which are practically complex to determine for specific sites. (3) Anthropisation quantification also strongly depends on the data formatting and the data management methods, all of which have different capabilities. (4) Data formats could be combined to design a new assessment method ensuring more precision and comparability, integrating individual patch dynamics, while relying on easily accessed data.

When combined, the different concepts used to assess anthropogenic landscape and ecosystem change appear suitable for integration in an approach oriented towards response to such changes. Within this view, the DPSIR framework is also subject to critique. To scientists, it can appear arbitraty and oversimplifying the interactions between entities at stake (especially human vs. natural dynamics) and crossing hierarchical boundaries to frame a logical chain. Moreover, many of the relationships between the human system and the environmental system are not sufficiently understood (Smeets and Weterings, 1999). On the other hand, to policy makers, it can appear that scientists using the DPSIR framework do not use it in a sufficiently transdisciplinary way, do not address possible responses or are not really oriented towards a definite decision maker, which should influence the conceptualisation of the issues assessed (Tscherning et al., 2012). However, the scientists refering to the DPSIR framework in their publications rarely criticize it.

Natural science, social science and policy making work according to their own (and distinct, sometimes divergent) paradigms and objectives. It is then not unexpected that a bridge between the three is not fully compatible with these disciplines. However, such generalisations should not be taken for what they are not: an environmental impact assessment is not meant to provide predetermined policies to address the problem (Tscherning et al., 2012). Meanwhile, organising such an assessment within the DPSIR framework does not mean giving an exhaustive assessment of all the interactions occurring between biological and cultural systems at any scale, which would be practically infeasible. It aims at providing scientifically relevant guidelines to policy makers and minimal knowledge of the issue, so that they understand environmental and human stakes. The decisions to be taken in response to environmental problem assessments still belong to policy makers. In turn, the applied policies will be followed and appropriated by scientists as for their implementation in the form of restoration or conservation project on specific sites, for example.

The DPSIR framework is one of the most popular Conceptual Ecosystem Models (CEM), but other CEMs also exist. Such general frameworks all aim to integrate biophysical and societal issues within a transdisciplinary view to find a tradeoff between human development and ecosystem health (Kelble et al., 2013). Some of them, such as the PPD (Press-Pulse Dynamics) framework (Collins et al., 2010), focus more on the monitoring of socio-ecological systems functioning, some others on the institutional context that would stimulate appropriate responses to environmental problems (Cumming et al., 2013). All of them provide more implementation details to their level of focus, but they are all very generic and do not target operational details linked to the research or the management processes, which is to be adapted case by case. Cumming et al. (2013)'s social learning method is based on the idea that political institutions should be designed so as to favour innovation and experiment at each hierarchical level they control while installing a close monitoring of such experiments. Such practices would help developing and selecting the best practices to match the scales of ecosystem services supply and demand, which are the key to ensure landscape sustainability (Cumming et al., 2013). To compare this framework with the DPSIR, it can be considered as nesting the DPSIR: designing an even more general paradigm (scale matching of ecosystem service supply and demand) to frame research and another framework (social learning) to favour the development of response-oriented research. Its first advantage compared to the DPSIR framework is that it allows to take the issue of business relocation, as distant drivers of environmental pressures into account, issue that was not easy to address within the sole DPSIR framework. Its second advantage is that it integrates the notion of ecosystem services. Such a framework shift allows to open perspectives of environmental justice, because the ecosystem services concept is designed to fulfil human needs, and highlights the capacity of human activities to enhance the sustainability of ecosystem service production (Cumming et al., 2013; Kelble et al., 2013). This argument is at the basis of the development of a new CEM based on the DPSIR framework, the EBM-DPSER (Ecosystem-Based Management - Drivers Pressures State Ecosystem services Responses) framework (Kelble et al., 2013). This framework provides more detailed instruction on how to conduct socio-ecological research thanks to ecosystem-based management, while it introduces the ability of the system to provide ecosystem services (instead of mere negative "Impacts") to a framework already well known to scientists and managers. This new framework offers interesting perspectives for the development of integrative action-oriented research and management in order to ensure sustainable development. It also facilitates the adaptation of previous research, such as the present thesis, to this new framework, which could be considered. In that case, it should be adapted to landscape ecology's pattern / process paradigm, as it was done with the DPSIR, in order to frame landscape ecological research on the EBM-DPSER steps.

The definition of the reference states that will serve as achievable response strongly influences the measure of the anthropogenic effect intensity itself. Currently, only unachievable (virtual naturalness) or unambitious (potential naturalness) reference states existed, so restoration ecology lacked a more optimistic, yet realistic option. The most ambitious achievable goal for restoration tackles pressures in the DPSIR. The *restored indigenous naturalness* (Vranken et al., in preparation-a) consists in completely removing local anthropogenic influence and redirects the ecosystem toward its previous dynamics: this is the most one can do to respond to anthropogenic disturbances. This should be the ultimate goal for restoration. Restoration is however not the only option considered in *Anthropisation*: artificial analogues serves as reference for management, in a more ecosystem service-oriented approach (Vranken et al., in preparation-a). It should be noted that the reference state definition is dependent on the existing techniques, that precondition the design of response plans. Moreover, once the reference state has been chosen, determining its specific characteristics for application to a definite site remain practically problematic, which has been illustrated in our study case in Katanga (André et al., in revision).

Once this reference is taken, anthropogenic effect assessment consists in evaluating how and how much the studied landscape or ecosystem is remote from this reference, which is the only normative dimension of our proposed assessment framework. It does not intricately imply that anthropisation necessarily has a negative impact on ecosystem functioning. This assessment favours large-scale anthropisation evaluation because it aims at limiting field surveys to identify hemeroby levels based on floristic composition. To do so, it combines the strengths of the existing data formats, acquisition methods and analyses. This method is of particular importance for the harmonisation of anthropogenic change assessment for comparison in different countries because it aims at being standard while taking specific patterns or processes into account. The originality of the method we propose lies within three facts. First, it integrates landscape dynamics in order to create location rules and refine anthropogenic vs. natural origin classification, which helps highlighting individual features of specific patches. Secondly, the combination between categorical mapping and point data analysis could also be performed on the basis of synchronic overlay. This forms an approach more specific to patch individual features or indirect effects like distance to anthropised patches. Thirdly, combining the previous aspects produces an integrated framework implementable to areas were few data are available *a* *priori*. Considering that those areas are often less accessible, thus less disturbed, particular conservational perspectives are at stake. As for the application of this method on a real case, it has been seen that field surveys are necessary to address the link between hemeroby and local biophysical parameters for the different ecosystems present in the area, then between these biophysical parameters and the corresponding spatiotemporal structure identified by remote sensing. Without a deep knowledge of such a link, that would allow to spatially identify the hemeroby level of each ecosystem patch, the patch dynamics approach here proposed would remain too speculative.

8.1.2 The challenge of assessing overall anthropisation patterns in an area with few data sources

This section is the answer to the second objective of this thesis: test the application of a new anthropisation metric to an African landscape André et al. (in revision), see p. 5)? This first attempt to test the anthropisation quantification method elaborated in *Anthropisation* (Chapter 3) showed four main results. (1) The methodology is applicable on an African city as long as relevant data representing ecological and anthropogenic processes are taken into account, which requires a relatively good field knowledge. (2) To discern individual patch dynamics, a better spatial resolution (than 30 * 30 m) for the satellite images is required to ensure better classification precision and thematic resolution. (3) Including fire dynamics highlighted synergies between land cover and fire practices. (4) This new method highlighted strong peri-urban dynamics.

This first application of the aforementioned framework includes site specificities that could be integrated to the methodology without modifying the framework. It consisted in adapting an hemeroby scale to Lubumbashi, which was a first attempt in Africa, and crossing the information with fire regime in order to compare anthropisation assessment with and without fire influence.

This test application also illustrated that, when adapting the chosen reference state to a specific site, more than one reference state (restored indigenous naturalness) was possible in the same landscape. The number of restored indigenous natural ecosystems depends on the number of *virtually natural* ecosystems in the area, but also on global anthropogenic pressures. Basically, these reference ecosystems correspond to all the ecosystems listed in the zero-level of the hemeroby scale we used in our methodology. In the case of Lubumbashi, those *restored indigenous natural* ecosystems were determined as Muhulu, Dembos (see p. 71) , swamps and copper hills.

It should also be noted that this implementation is actually a simpler version of the framework elaborated p. 50. This simplification was required by the lack of thematic and spatial resolution as well as classification precision of the data. The latter prevents to correctly represent all the ecosystems described in the hemeroby scale of Lubumbashi. As a consequence, dynamic aspects were explored only in the final step of the map building, in a general assessment rather than following the trajectory of individual patches, as previously announced in the *Anthropisation* chapter. Further implementations of a more elaborated version of change detection should be tested using post-classification comparison to focus on individual patches.

This observation also shows that spatial resolution can interact with thematic resolution and classification precision. In African landscapes, such interaction could be seen, for example, in the case of smallholder tree plantations. First, as such parcels are generally quite small (a few hectares), their surface and texture could be mixed with natural forest or thinned forest if the image spatial resolution is not sufficient to distinguish their spatial structure or texture (Boyd and Danson, 2005). Secondly, too coarse spatial resolution to detect tree plantations may lead to merge the parcels into an irregular form and wrongly identify them as a natural feature (Bogaert et al., 2014). Moreover, plantation, thinned forests and old-growth forest do not have the same species or spatial distribution of trees, but image resolution may not make such discrepancies appear (Bastin et al., 2014), classification may not be able to distinguish such classes, though such distinction would be relevant for anthropisation assessment.

In the case of *Anthropisation Katanga*, LANDSAT images did not appear of suitable spatial resolution for our scale of study to distinguish crops from savannahs or continuous from discontinuous built-up, but their spectral resolution (seven bands) could help isolating the spectral signatures of these land uses. Therefore, pan-sharpening with images of higher spatial resolution, such as SPOT images (up to 1.5 m spatial resolution in panchromatic, 6 m in multispectral (four bands) mode (SPOT, 2013)) could be tested. Field knowledge also highlighted that apparently similar land covers were not always equally affected by anthropogenic effects. What is observable in the field but not by remote sensing is that *miombo* woodlands developed on lateritic soils are natural and do not have the same morphology than *miombo* developed on deep red soils, that result of the interaction between agricultural practices and ecological succession. Soil data, which are not available to that extent at appropriate grain, or better knowledge of natural and anthropogenic *miombo* spatiotemporal structure would allow to distinguish them using the patch dynamics approach described in Vranken et al. (in preparation-a).

The reader should keep in mind that the aforementioned results are available for general assessment only and should not be taken as is to infer specific impacts or pressures on definite groups of species, that can be more sensitive to certain pressures or patterns than others (Cale and Hobbs, 1994). The application of the new quantification framework focuses on states: landscape spatial structure, but also relies on the definition of pressures and impacts, that are considered when building the hemeroby scale.

Data comparisons, but also methodology comparisons could be performed on different anthropisation assessment sites and methods. If other small-scale anthropisation quantifications have not yet been performed in Africa, studies covering the whole world already exist, such as Sanderson et al. (2002). Comparing the results of such studies with the ones obtained from the application of our methodology on different landscapes would allow highlighting the effect of scale and methodology differences on anthropogenic landscape change assessment. This would show the differences in the values obtained, and their consequences on possible responses to be addressed.

8.2 Pattern heterogeneity: its pending thermodynamic meaning, its multiple influence on anthropisation

This section answers the second question of this thesis (Section 1.2.2): "What is pattern heterogeneity? How is it quantified? How is it influenced by anthropisation? ".

8.2.1 Why entropy fails to explain landscape spatio-temporal heterogeneity

This section answers the third objective of this thesis: define the various existing landscape entropy interpretations and characterise their links with thermodynamics (Vranken et al., 2015), see p. 5. The review performed in this goal gave four main results. (1) The three existing definitions of entropy in landscape ecology follow the logic of thermodynamics or information theory, but these two theories represent different phenomena. (2) Spatial heterogeneity is based on information theory and has therefore no relevant thermodynamic interpretation. (3) Unpredictability could be properly interpreted in thermodynamic terms if energy transfer measurements were performed at the appropriate level. (4) Complexity theory explains that scale dependence of patterns is not likely to be thermodynamically determined.

The reason why the current thermodynamic interpretations of spatial heterogeneity, unpredictability and scale dependence are erroneous are basically due to two causes (Vranken et al., 2015). First, the tools used to describe these three features do not always come from thermodynamics, but generally from information theory, that used similar terminologies as thermodynamics but are deprived of physical correspondence with thermodynamic system behaviour.

The second reason is linked to system complexity, appears less definite and could help to find perspectives for the assessment of thermodynamic processes underlying landscape spatio-temporal dynamics. The current thermodynamic interpretations at ecosystem and landscape levels are based on process description occurring at physical corpse levels (cf. *macrostate*, p. 55). However, according to complexity theory, such inferences are inappropriate because system behaviours are not self-similar across organisational levels. Therefore, describing processes accross different hierarchical levels of matter organisation would first require studying all the energy and matter exchange of the system elements (at subsystem level) with each other and their environment: appropriate data collection at this level would require such a huge infrastructure that it would be practically infeasible. Moreover such study would lead to more than one possible results at system level (Green and Sadedin, 2005).

In order to properly investigate the thermodynamic behaviour of ecosystem processes, the way it is affected by anthropogenic pressures and its role in influencing landscape spatio-temporal dynamics — the state of the system — , the quantification of energy exchanges should first be performed at landscape level, and explore interactions with the direct lower and upper levels : ecosystems and region. A perspective to perform such measurements, could be infra-red albedo acquisition through passive remote sensing. With such thermal information, energy exchanges and entropy production could be computed and relationships could be drawn with landscape spatial structure and its evolution.

8.2.2 What anthropogenic and spatial pattern impact assessment are missing

This section answers the fourth objective of this thesis: characterise the link between anthropisation and spatial heterogeneity based on 20 African landscapes (Vranken et al., in preparation-b), see p. 5. This first attempt to study such a link by combining synchonic and diachronic approaches based on a large landscape sample gave three main results. (1) Opposite anthropogenic effect on pattern heterogeneity depending on the amount of anthropised surface (bell-shaped influence) (2) Opposite anthropogenic effects between natural vs. anthropogenic heterogeneity. (3) Opposite anthropogenic effects between the various heterogeneity components.

General assessments of spatial heterogeneity have been dismissed because they neglected functional aspects of spatial patterns (Cale and Hobbs, 1994), but in the meantime, important analyses were thrown out with the bathwater. Not only general heterogeneity assessments were dismissed, but also the relationship between (compositional) anthropogenic landscape change and pattern heterogeneity was neglected since then. Though still incomplete and contradictory at that time, they were however not useless: failing to study the anthropogenic effect on heterogeneity dynamics removes an important perspective of understanding and environmental management when studying anthropogenic impact on ecosystem health. This shortcoming has been proven in our study (Vranken et al., in preparation-b).

First, general heterogeneity assessments showed that anthropogenic effects on landscape structure was bell-shaped: the tendency depends on the starting point (anthropisation dominance) from which landscape dynamics is considered. This can easily be explained by spatial transformation process and dominance studies (Bogaert et al., 2004). The second proof could have been deduced from studies prior to Cale and Hobbs (1994) criticism, yet it was still relatively unexplored since then: natural (and semi-natural) land covers follow different dynamics than anthropogenic land covers, generally opposed. Thirdly, and most importantly, the different components of pattern heterogeneity (number and relative areal abundance of land covers, patch size, shape, spatial arrangement) respond in opposed ways to anthropogenic pressures. For example, while anthropogenic effect decreases natural land cover diversity, it increases configurational heterogeneity of these natural patches due to their smaller size (see Chapter 6).

The natural vs. anthropogenic decomposition highlighted the impact of thematic resolution on spatial transformation processes assessment : antagonist dynamics for different land covers may seem to neutralise themselves at a more general level. The anthropisation stage of the landscape considered plays an important part in the results in tropical Africa, which, according to the 20 studied landscapes, still shows a large anthropisation gradient. European landscapes, known to be almost fully transformed, should be situated at the right end of the heterogeneity / anthropisation curves (Figures 6.3 to 6.5).

The binary (natural vs. antrhropogenic) classification appears to oversimplify the results and sometimes introduce biases, when different anthropogenic dynamics correspond to the same land cover. However, in order to perform a rather precise anthropogenic effect assessment such as the one described in (Rüdisser et al., 2012), the thematic resolution of the 20 study sites used in *Heterogeneity-Anthropisation* was not satisfactory. Therefore, information on underlying processes of the patterns studied and anthropogenic effect intensity for each land cover could not be spatialised. For this reason, no hemeroby scale (André et al., in revision) was built: a simpler index, with a binary classification (anthropogenic vs. natural) was preferable. At this level of thematic precision, less field knowledge is required to correctly classify the land cover classes by anthropisation level and our knowledge, fed by contact with the authors of the aforementioned studies, were considered sufficient to achieve that goal. The large number of landscapes also mitigated the "noise", as the identifiable anthropisation dynamics mistake (in Benin) did not change the general shape of the curve.

As with *Anthropisation Katanga*, those results were obtained from general pattern assessments, only assessing landscape state, and were not adapted to species with a specific behaviour. The specific behaviour and metric values they present are then not to be copy-pasted for specific assessments or functional studies. However, the highlighted differentiated dynamics depending on anthropisation stage show this issue should be studied when assessing the anthropogenic impact on ecological processes in more specific studies as well. This should not only explain divergent tendencies for different heterogeneity components from the same group of species, but also help predicting future impact tendencies on the structure of their environment. Such information is capital to infer ecological impact of anthropogenic landscape change (Naiman et al., 1988) and design responses to these anthropogenic impacts, guiding the most appropriate environmental management measure depending on the context.

Synchronic and diachronic studies were combined to illustrate pattern heterogeneity dynamics under a gradient of anthropisation instead of following the evolution of a single area in a diachronic study. Synchronic study circumvents the practical difficulty to find long data time series for the same area. Indeed, remote sensing has only existed for a few decades and the spatial and spectral resolution of the captors has strongly evolved during this period. Moreover, in the case of remote sensing, image classification would require access to ground reality, which is not always practically feasible with past images. When comparing maps, thematic resolution (classification precision) consistency problems also occur. It should also be noted that, to obtain a full anthropisation gradient in a diachronic study, the considered landscape should be currently fully anthropised, while not all the landscapes are in this state. Moreover, for conservation purposes (see the priorities of restoration ecology, Chapter 3), landscapes that are not yet fully novel should be studied as well.

The "fictive" evolutive series obtained in this way is thus only a proxy, and suffers from noise linked to site specificities. Not every landscape follow the same dynamics, even when they share similar departure conditions. A diachronic study following the same landscape throughout its anthropisation gradient, from originally natural to completely novel, would avoid such a noise, but also be over-fitted to the studied case.

Therefore, studying several complete and incomplete time series following different landscape dynamics, as it was performed in *Heterogeneity-Anthropisation*, combines the advantages of synchronic and diachronic analysis. In that case, particular attention is recommended to the aforementioned resolution problems. This combination allows generalisation if the landscape dynamics are not too divergent, but at the least this allows comparing different landscape trajectories, for example landscapes from different climates, location or culture.

The quadratic polynomial regressions used in *Heterogeneity - Anthropisation* are a simple example of the possible existing techniques to build calibration models, but other techniques also exist. For example, a quadratic variant of the PLS (Partial Least Squares) regression could also have been used (Wold et al., 1989; Ni et al., 2014). This modelling method allows predicting various correlated (Y) variables depending on various correlated (X) variables (Tenenhaus, 1998). In our case, it could be used to build a single predictive model assembling the variables representing the different components of heterogeneity altogether, then separately, and examine their relationship with the anthropisation gradient and the interactions between the different components. Such a technique could also allow the comparison of different anthropisation assessment methods. This technique is frequently used in for chemical analyses but does not seem to be applied in landscape ecology, probably because the noise is more important in biological than chemical systems.

8.2.3 Testing the relationship between anthropisation and heterogeneity

This section answers the fifth (and last) objective of this thesis: identify anthropogenic disturbances through the study of spatial patterns (Vranken et al., 2013), see p. 5. The study of vegetation spatial patterns and pollution perception in Lubumbashi gave three main results. (1) The anthropogenic effect of mining activities on vegetation structure are vegetation fragmentation due to the enlargement and aggregation of bare soil patches. (2) This anthropogenic effect decreases with the distance from the emission source. (3) The impact of this pollution is well perceived in the urban matrix by the citizens.

In the *Heterogeneity - Anthropisation* article discussed in the previous section (Vranken et al., in preparation-b), we saw that Lubumbashi was dominantly anthropised and we can infer the underlying spatial transformation processes from its position (Bogaert et al., 2004). In *Heterogeneity - Anthropisation*, two images of Lubumbashi (1984 - 2009) showed increasing patch complexity and aggregation for anthropogenic landuses, due to anthropogenic patch aggregation. Considering the natural land covers, this anthropisation stage corresponds to decreasing patch aggregation due to dissection processes and slightly increasing patch size, due to attrition processes (the smallest patches disappear first). The other heterogeneity components were relatively stable at this thematic scale. More precise *a posteriori* tests showed that, during the 1984 - 2009 period, bare soils underwent aggregation, while vegetation was dissected.

This information could be reused as a calibration model to distinguish natural from anthropogenic pattern dynamics of two land cover classes. In the case of the pollution cone due to metallurgic activity in Lubumbashi (Vranken et al., 2013), vegetation is presumably natural and bare soils presumably anthropogenic. The increasing anthropogenic pressure was materialised by comparing one study site presumably outside the influence zone of the Gécamines smelter chimney and another site downwind the processing plant. Comparisons of the spatial patterns (state) confirmed the hypothesis emitted from the observation of the *Heterogeneity-Anthropisation* curve .

Other data management methods could have been used in complement with the ones performed in Vranken et al. (2013). For example, statistic tests could have been used to test the significance of the differences between landscape spatial structure inside and outside the pollution cone. In that case, the χ^2 test could have been applied on each couple of index values. Its application *a posteriori* showed that, except for the total area of vegetation vs. bare soils, all the metrics displayed highly significantly different values between the control zone and the disturbed zone for the closest test areas (13 km^2). To isolate the effect of distance and scale, two methods could have been applied in parallel. First, rectangles of increasing size, all centred on the Gécamines plant could have been used to highlight the effect of scale. Secondly, to account for the effect of distance from source, rectangles of the same size could have been studied, each one of them situated at an increasing distance from the Gécamines plant.

Chapter 9

Opening the debate

9.1 Limits and perspectives: functional aspects

9.1.1 Genericity vs. function: mutually exclusive representations

The answers to the scientific questions asked in this dissertation, such as "how intense is anthropogenic landscape change in a given area?" (see Section 1.2.1) or "what is the link between heterogeneity and anthropisation?" (see Section 1.2.2) could not have been obtained otherwise than through the use of generalised assessments.

However, such generalised measures and interpretations for ecological processes have been strongly criticised lately because they fail to take into account functional aspects of spatial patterns, especially related with organism (fauna, but also flora) mobility across the landscape (Cale and Hobbs, 1994; Tischendorf and Fahrig, 2000; Mander et al., 2005; Fahrig et al., 2011). Indeed, as mentioned in *Heterogeneity - Anthropisation*, the ability to link pattern and process depends on the way patterns are characterised. However, landscape patterns are not perceived the same way by all its abiotic and biotic components. Generally speaking, landscape structure does not have the same effect on every ecosystem component and these effects do not operate at the same scale, depending on the mobility of the elements considered: different species, but also water or wind and mass transport elements in the case of erosion, or different organic or mineral pollutants that can diffuse through soil or water (Burel and Baudry, 2003). Therefore, pattern definition, in accordance with the studied element, strongly influences the perception of their behaviour and subsequent ecological processes (Cale and Hobbs, 1994; Burel and Baudry, 2003). Consequently, it is not possible to address functional aspects of landscape patterns in a generalised form, using a single measure. This fact represents a major constraint in addressing the link between spatial patterns and ecological processes.

The relevance of generalised assessments has been thus questioned during the last decades. However, landscape ecology as well as environmental assessment reports keep on making general assessments. So why is it still performed and should it be stopped? An hypothetical answer to this question consists in putting these divergences in a scale perspective: generalised studies are especially suitable for large-scale, even global evaluations, that are in turn of particular importance to bridge science and policy, that handles responses to environmental problems. From the policy point of view, there is a need for clear and specific information (Smeets and Weterings, 1999).

As outlined by complexity theory (see Chapter 4), the phenomena studied at one level do not allow defining which phenomena will occur at the upper or lower levels: here, the same problem is observed. As a consequence, general pattern assessments are not suitable to quantify ecological processes, that occur at a lower hierarchical level, nor to characterise the behaviour of definite groups of species or mass transport. General landscape pattern quantification should focus on patterns, not processes. Such assessments can only give an idea of the general underlying processes, such as the fact that the shape of a habitat patch influences effective habitat (core) area through edge effect, but any further conclusion, for example as for edge depth or core area or shape, should be verified at the spatio-temporal and thematic scales corresponding to the ecosystem component of focus. The greatest danger of generalisation is to make overreaching conclusions.

9.1.2 How to include functional aspects in the present work

Functional aspects of spatial patterns, particularly connectivity and *connectedness* (Acosta et al., 2003), wich represent a major determinant of species distribution, have already been and are still addressed in several studies, for example: Baudry (1989); Petit and Burel (1998); Tischendorf and Fahrig (2000); Bélisle (2005); Fahrig and Nuttle (2005); Ludwig et al. (2005); Baguette and Van Dyck (2007); Fahrig et al. (2011). Within this view, the opacity of the matrix and the mobility of the (biotic and abiotic) ecosystem elements accross the landscape are of particular importance. In order to integrate such aspects to the methods developed in this thesis relies, both anthropisation as well as heterogeneity assessment methods should be adapted to the studied ecosystem element. Attention should however be raised to the fact that such adaptation makes the assessment unsuitable for generic appraisal.

Considering the methodological framework developed in *Anthropisation*, the adaptation in the case of species mobility would consist in focusing on the alteration processes occurring in the preferred habitat of the studied species and on how much it would affect their behaviour. As for the degraded habitats or other land cover/uses in the landscape, their "opacity" should be assessed in accordance with these species: to which extent they can settle, feed or cross the other land covers (Benton et al., 2003; Bélisle, 2005). To do so, the *distance to nature* (Rüdisser et al., 2012) gradient would be calculated as a functional distance, according to the opacity of the matrix and the mobility of the studied species (Petit and Burel, 1998). Those specificities can also be included in a "personalised" pattern heterogeneity assessment. More specifically to the incidence of patch shape, edge width could be taken into account, that would be measured according to the biophysical gradient affecting the studied species, directly or indirectly through its habitat characteristics (Burel and Baudry, 2003; Baguette and Van Dyck, 2007).

A consequence of these new aspects is that it is possible to nuance the formerly binary vision of habitat patch vs. hostile matrix (Puech et al., 2015). Functional corridors are not necessarily the same land cover as habitat patches. Occupied or exploited land may even be used for species mobility, resource consumption, even temporary or permanent habitat, as highlighted by the concept of *analogous ecosystems* in restoration ecology (see Chapter 3). In the case of combining the functions of *agroecosystems*, particular attention should be paid to the anthropogenic pressures occurring in the matrix (Benton et al., 2003). For example, in agricultural landscapes, different cultivation practices may have differentiated impacts on ecological processes: for species mobility, but also for their resource consumption, even their settlement (Benton et al., 2003; Bengtsson et al., 2005). In this way, trees in an orchard or in an agroforestry parcel may be used as habitat or relay by bird species if the harvesting operations do not represent significant pressure to them (Bhagwat et al., 2008). This reminds of the fact that anthopogenic effects are not systematically detrimental to biodiversity or ecosystem health (Vranken et al., in preparation-a).

From the above, we know that scale dependence on pattern (particularly heterogeneity) measurement is at the center of the issue to address functional aspects of pattern assessment for all the ecosystem components (Cale and Hobbs, 1994; Benton et al., 2003; Cushman, 2015; Puech et al., 2015; Vranken et al., 2015). The study of the propagation of pattern-process relationships across space and through time is constrained by the complexity of the systems studied, where studying one hierarchical level does not allow to infer the processes happening at other levels (Cushman, 2015; Vranken et al., 2015). Therefore, in order to better understand the interaction between abiotic factors and species with different behaviours (trophic level, mobility, ecological valence, etc.), each one of them should be first studied at the appropriate scale, as suggested in the two previous paragraphs. Only then can correlation analyses be performed on the spatio-temporal distribution of species, their habitat, their food, at different scales.

9.2 Conclusion: addressing land scarcity

Growing landscape anthropisation creates land scarcity that addresses a double challenge linked to sustainable development: maintaining ecological processes while supporting the needs of a growing human population with increasing needs. For example, in the South, industrialisation and export-oriented agriculture as well as rural outmigration combined with demographic growth in cities are challenging food security. Meanwhile, the pressures on natural and semi-natural land keep on growing (Bogaert et al., in preparation). Bogaert et al. (in preparation) propose a new transdisciplinary field to assess the problems of land scarcity: *choralogy*, the science of land. In this context, anthropogenic effect assessment to address adequate responses to land scarcity takes all its sense.

Facing land scarcity, two approaches are generally taken: land sparing (keeping a maximal amount of preserved areas) and land sharing (combining resources exploitation while maintaining ecological processes) (Green and Sadedin, 2005; Fischer et al., 2014). An example of response corresponding to land sparing consists in defining wildlife reserves at landscape or even regional scale, while reserving the rest of the land to industry, agriculture and urban areas, as it is done, for example, in the United States. On the other hand, the exploited areas like agricultural land are generally managed in an unsustainable way: focused on food production only (Fischer et al., 2014). Land sharing corresponds to what is addressed in agroecology, mulfifunctional and urban agricultures, as outlined by the term *agroecosystem*, that combines "biological" and "cultural" processes at ecosystem scale (Gliessman, 2007; Huang et al., 2015).

In the context of this thesis, land sparing would use restored indigenous naturalness as a reference state for the spared land, while land sharing would take artificial analogous ecosystems as reference. Tough these options may appear mutually exclusive, they can actually be combined in a multi-scale approach (Fischer et al., 2014). In that context, the role of landscape ecology to assess the impact of landscape heterogeneity combined with the impact of anthropogenic disturbances on ecosystem processes, as described throughout this thesis, appears as a missing step that can reconcile land sparing with land sharing. For example, remote and relatively undisturbed areas could be preserved, while a mosaic of natural land covers (*restored indigenous natural*) areas and eco-intensive exploited land (*artificial analogous*) could take place elsewhere. Studying such a structure would combine the study effect of landscape structure and anthropogenic activities on ecological processes. Indeed, heterogeneity, for example in agricultural landscapes, is said to enhance biodiversity and this issue is frequently addressed in landscape ecology (Naiman et al., 1988; Benton et al., 2003; Fahrig et al., 2011; Scariot, 2013).

This multiscale combination of land sparing and sharing can even be deepened at finer scales: at the parcel scale, agroforestry can be considered as land sharing while treelines at the border of the parcel would be considered as land sparing, but at the landscape scale, treelines, hedgerows and other uncultivated landscape elements can be considered as land sharing practices (Puech et al., 2015). The temporal dimension should also be taken into account in terms of sparing and sharing, especially in Africa, due to their fast land cover change linked with, *inter alia*, slash and burn practices (van der Werf et al., 2008).

Particularly in populated areas of Africa, land sparing and sharing combinations at landscape and parcel scales could appear a promising response to environmental disturbances and food insecurity because it can ease decentralised land management. This scale of land administration would be more adapted, compared to the global infrastructure management problems encountered by most African countries (see the example of Lubumbashi: Vranken et al. (2014); André et al. (in revision)) and the abundance of smallholder farms (Altieri, 2002) in rural areas. Indeed, conventional agriculture (Altieri, 2002) as well as wildlife reserves (Diallo et al., 2011) have shown their limits in this context. Moreover, the issue of food sufficiency, for example to mitigate rural outmigration (Altieri, 2002; Fischer et al., 2014), is dependent on connectivity between food production and food consumption. This issue could also be addressed with tools from landscape ecology, combined with transport studies and human geography.

Such perspectives show the integrative power of landscape ecology to address the socio-ecological tradeoff of sustainable development.

Glossary

Aggregation: see: "Spatial transformation processes" (Bogaert et al., 2004). The action or process of collecting units or parts into a whole; to bring or gather together into a whole; to fill gaps or open space.

Analogous (artificial) ecosystem: state in which a managed ecosystem is when its anthropogenic distrubances are compensated or its ecosystem services enhanced thanks to the development of functionally similar abiotic and biotic features to natural landscapes occurring in the same region (its natural analogue) (Lundholm and Richardson, 2010).

Anthropisation: degree of influence of human activities on landscape composition, configuration and dynamics (Bogaert et al., 2011). The corresponding landscape state is *anthropised*.

Attrition: see: "Spatial transformation processes" (Bogaert et al., 2004). The reduction or decrease in the number of patches; disappearance of patches.

Biocultural (landscape): landscapes generated by both natural and anthropogenic processes (Bogaert et al., 2014).

Categorical mapping: map data formatting in which the features classified in the same category are assumed to represent similar ecosystems (Gustafson, 1998).

CEM: Conceptual Ecological Model, general research and management framework designed to integrate and synthesize scientific knowledge in a manner familiar to managers and policymakers (Kelble et al., 2013).

Chorema: schematic and synthetic representation of the studied characteristics of a geographic space (Brunet, 1980).

Composition: see "Structure".

Configuration: see "Structure".

Connectedness: functional connectivity, the fact that two habitat patches are linked, from the point of view of one or more individual or species, so that it / they can move from one patch to another even though they are not physically connected (Burel and Baudry, 2003).

Connectivity (physical, structural): the fact that two habitat patches are adjacent, physically close, contiguous in space (Burel and Baudry, 2003).

Conservation, conservation management: action of preserving an area from anthropogenic disturbance in order to protect local biodiversity or contribute to its recovery, after restoration or management (Weddell, 2002; Science and Policy Group, 2004).

Corridor: elongated patch that helps connecting two habitat patches of similar kind by enhancing species mobility through the matrix. Those corridors can physically connect those patches or act as stepping stones (Tischendorf and Fahrig, 2000).

Creation: see: "Spatial transformation processes" (Bogaert et al., 2004). The formation of new patches, which results in an increase of the total number of patches; the act of causing to exist of patches; patch genesis.

Diversity: see "Composition", "Structure".

Disclimax, paraclimax: (obsolete term) stable vegetation developed after an anthropogenic disturbance (Henderson et al., 1960). See also "Future naturalness".

Dissection: see: "Spatial transformation processes" (Bogaert et al., 2004). The carving up or subdividing of an area or patch using equal-width lines; sectioning of an area or patch; area or patch (sub)division.

Deformation: see: "Spatial transformation processes" (Bogaert et al., 2004). The change of patch shape, without patch size change; patch disfigurement.

Diachronic analysis: analysis of the dynamics of a phenomenon by following the evolution of a single study case along all the development stages of the studied phenomenon (Baker and Billinge, 1982). See also "Synchronic analysis".

District: according to Kevin Lynch's picturesque perception theory of urban space, medium-to-large two-dimensional section of the city, which the observer mentally enters "inside of," and which is recognizable as having some common, identifying character (Lynch, 1960).

Dominance: major areal relative abundance of one patch or one land cover in a landscape, inverse of evenness (Forman and Godron, 1986a).

DPSIR framework: acronym for Driver Pressure State Impact Response analysis framework, used to classify environmental indicators and analyse environmental problems regarding the relationship between ecological and human dynamics in a comprehensive and transdisciplinary analysis (Smeets and Weterings, 1999). See "CEM".

Edge, edge effect: transition zone, more or less abrupt and wide, between two adjacent ecosystems, such as forest and field, due to the biocoenosis (mainly vegetation) and biotope variations (light, temperature, moisture, etc.). This notion is linked to the notion of ecotone in ecology (Naiman et al., 1988; Forman, 1995).

Edge (picturesque): according to Kevin Lynch's picturesque perception theory of urban space, linear element not used or considered as paths by the observer: boundary between two phases, linear break in continuity, more or less penetrable: shore, railroad cut, edge of development, wall (Lynch, 1960). See also: "Path".

Emergent property: singular characteristic of a complex system that is novel, different compared to the characteristics of the elements composing this system (Wu and David, 2002).

Enlargement: see: "Spatial transformation processes" (Bogaert et al., 2004). The increase of patch size; patch size expansion.

Evenness: also called *equitability*. Equal, even relative abundance of all the elements composing a system. In landscape ecology, it corresponds to an equal areal distribution of the classes, different ecosystems composing the landscape. See also "Composition".

Fragmentation: see: "Spatial transformation processes" (Bogaert et al., 2004). The breaking up of an area into smaller parcels, resulting in unevenly separated patches; the breaking up of extensive landscape features into disjunt, isolated, or semi-isolated patches.

Future naturalness: see "Naturalness" (Vranken et al., in preparation-a). Also called *anthropogenic naturalness*. State that the system eventually reaches after human influence ceased and after following complete ecological succession (Peterken, 1996).

Hemeroby: qualitative scaled measure of the difference between a reference (natural) state and the anthropised state of a system. The system considered varies from plant species (Jalas, 1955) to landscapes (Renetzeder et al., 2010). Sometimes called *degree of naturalness*.

Heterogeneity (spatial pattern): intricateness of the landscape spatial pattern. This heterogeneity can be compositional (many classes evenly distributed) or configurational (many patches, complex shapes, dispersed)(Turner et al., 2001).

Hierarchical level: see "Scale".

Landmark: according to Kevin Lynch's picturesque perception theory of urban space, external point-reference, frequently used clues of identity and even of structure that are relied upon in a journey. They are usually a rather simply defined physical object: building, sign, or mountain. Landmarks can be distant or local (Lynch, 1960).

Landscape (landscape ecology): heterogeneous land area composed of a cluster of interacting ecosystems (Forman and Godron, 1986a). It corresponds to a hierarchical level between the Region (upper) and the ecosystem (lower) (Burel and Baudry, 2003).

Location rules / criteria : set of georeferenced parameters, the value of which allow to identify and locate an area of interest.

Matrix: large patch in which the other patches are encompassed and that dominates the landscape (see: "Patch) (Forman, 1995).

Mosaic (land): group of patches forming the landscape when there is no specific dominance of a certain patch (matrix) in the landscape (Forman, 1995).

Motility: psychic ability to move across a city, using different means of transportation, or to find one's way on maps (Bret, 2009; Orfeuil, 2004).

Naturalness: characteristic of what is natural, reference state of the system (ecosystem, landscape), when no human activity has influenced it (yet). Different definitions exist, that are used as references to measure anthropisation or as a restoration goal:*original naturalness, virtual naturalness, potential naturalness, future naturalness, managed indigenous naturalness* (Peterken, 1996; Kowarik, 1999; Vranken et al., in preparation-a).

Node: according to Kevin Lynch's picturesque perception theory of urban space, strategic point in a city, convergence of paths. Points may be primarily junctions, places of a break in transportation, a crossing, moments of shift from one structure to another, or simply concentrations, as a street-corner hangout or an enclosed square (Lynch, 1960).

Novel ecosystem: ecosystem that has undergone enough anthropogenic pressure to cross ecological, environmental and social thresholds beyond which their historical abiotic, biotic and social components will never be recovered. The evolutive dynamics of this ecosystem will differ from its virtual naturalness (Hobbs et al., 2013).

Original naturalness: see "Naturalness" (Vranken et al., in preparation-a). State of the system before the first anthropogenic land transformations (Peterken, 1996).

Pan-sharpening: remotely sensed image treatment that consists of merging a panchromatic image of high spatial resolution with a multispectral image with coarser resolution, in order to obtain an image with high spatial and spectral resolution (http://www.pancroma.com).

Patch: elementary unit of a landscape, ecosystem different from its surroundings (Forman, 1995).

Patch dynamics: study of the individual properties and evolutionary trajectories of the patches composing a landscape (Wiens et al., 1993)

Pattern / process paradigm: causal connection between landscape spatial patterns or structure (composition and configuration) and ecosystem processes occurring therein (Turner, 1989). That powerful connection allows to study landscape patterns to infer the underlying ecological functioning that drives the (spatial, temporal) distribution of organisms of different species (Fahrig and Nuttle, 2005).

Path : according to Kevin Lynch's picturesque perception theory of urban space, channel along which the observer moves. Paths may be streets, walkways, transit lines, canals, railroads (Lynch, 1960).

Perforation: see: "Spatial transformation processes" (Bogaert et al., 2004). The process of making holes in an area or patch; gap formation; interruption of land cover continuity by formation of openings.

Potential naturalness: see "Naturalness" (Vranken et al., in preparation-a). Also called *potential natural vegetation/community*. State that would develop if human influence disappeared and if the resulting succession were finished instantly, self-sustaining ecosystem in the disturbed abiotic condition of the site (Peterken, 1996).

Resilience (landscape): capacity, for a landscape that underwent structure change due to a disturbance, to recover, to go back to this previous structure (Pimm, 1984).

Resistance (landscape): capacity, for a landscape, to face a disturbance without structural change (Pimm, 1984; Vranken et al., 2015).

Restoration (ecological): process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed. This supposes the existence of a reference ecosystem that will serve as model (Science and Policy Group, 2004).

Restored indigenous naturalness: see "Naturalness". Natural system (ecosystem, landscape) present if there had never been any local disturbance, even abiotic (Vranken et al., in preparation-a).

Scale, hierarchical level: spatial or temporal focuses for the observation of a phenomenon. Scale refers to a continuous property measured in common units, whereas hierarchical level refers to a discrete property with various entities studied at each level (Cushman et al., 2010).

Scale dependence (pattern): effect of the spatial and temporal resolution of observation on spatial and temporal patterns measurement (Riiters et al., 1996; Vranken et al., 2015).

Shift: see: "Spatial transformation processes" (Bogaert et al., 2004). Patch repositioning; patch translocation.

Shrinkage: see: "Spatial transformation processes" (Bogaert et al., 2004). The decrease or reduction in size of patches, without "attrition"; progressive reduction of the initial land cover patch, ideally maintaining its original shape.

Spatial transformation processes: pattern dynamics undergone by a landscape in transformation, generally described at class level. They are classified in ten categories: *perforation, dissection, fragmentation, shrinkage, attrition, deformation, shift, creation, enlargement* and *aggregation* (Bogaert et al., 2004). The five first processes represent class (land cover) degradation, while the three last processes represent the appearance of new land covers.

Stretched values: map data formatting used to represent the spatial distribution of data under continuous variations, generally represented using raster maps in which each pixel has a definite value (Gustafson, 1998).

Structure: spatial pattern displayed by the landscape. It consists of landscape composition (number and relative abundance of ecosystems/ classes) and configuration (number, shape, size and spatial arrangement of patches) (Forman and Godron, 1986a; Li and Reynolds, 1995).

Synchronic analysis: analysis of the dynamics of a phenomenon using a multitude of case studies in different development stages of the studied phenomenon (Baker and Billinge, 1982). See also "Diachronic analysis".

Thematic resolution: level of details included in a map legend or image classification to distinguish the different land cover classes in the represented landscape (Bailey et al., 2007).

Unpredictability (pattern dynamics): instability of landscape pattern evolution, irregularity of landscape pattern change over time, incapacity to predict towards with state (composition, configuration) the studied landscape will evolve (Vranken et al., 2015).

Virtual naturalness: see "Naturalness" (Vranken et al., in preparation-a). Also called *present naturalness*. Hypothetical current state of the system if human beings had never had any impact on it (Peterken, 1996; Lecomte and Millet, 2005).

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Appendix

APPENDIX A

Anthropogenic effects in landscapes: historical context and spatial pattern

Authors: Jan Bogaert, Isabelle Vranken, Marie André

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A.1 Abstract

Bio-cultural landscapes are characterized by anthropogenic pattern features. Historically, noticeable anthropogenic effects are accepted to have appeared in landscapes after the invention of agriculture and further trends of landscape change could be linked to the development of agriculture. Through time, a sequence of landscape dynamics with three stages is expected, in which a natural landscape matrix is initially substituted by an agricultural one; urban patch types will later on dominate the matrix as a consequence of ongoing urbanization. The importance of the development of agriculture and its productivity for the evolution of settlements, villages and cities is emphasized. Anthropogenic change of landscapes confirms the status of geographical space as a limited resource.

Keywords: Agriculture, Anthropogenic effects, Domestication, Land cover dynamics, Landscape metrics, Urbanization.

Appendix B
Appendix to chapter 5

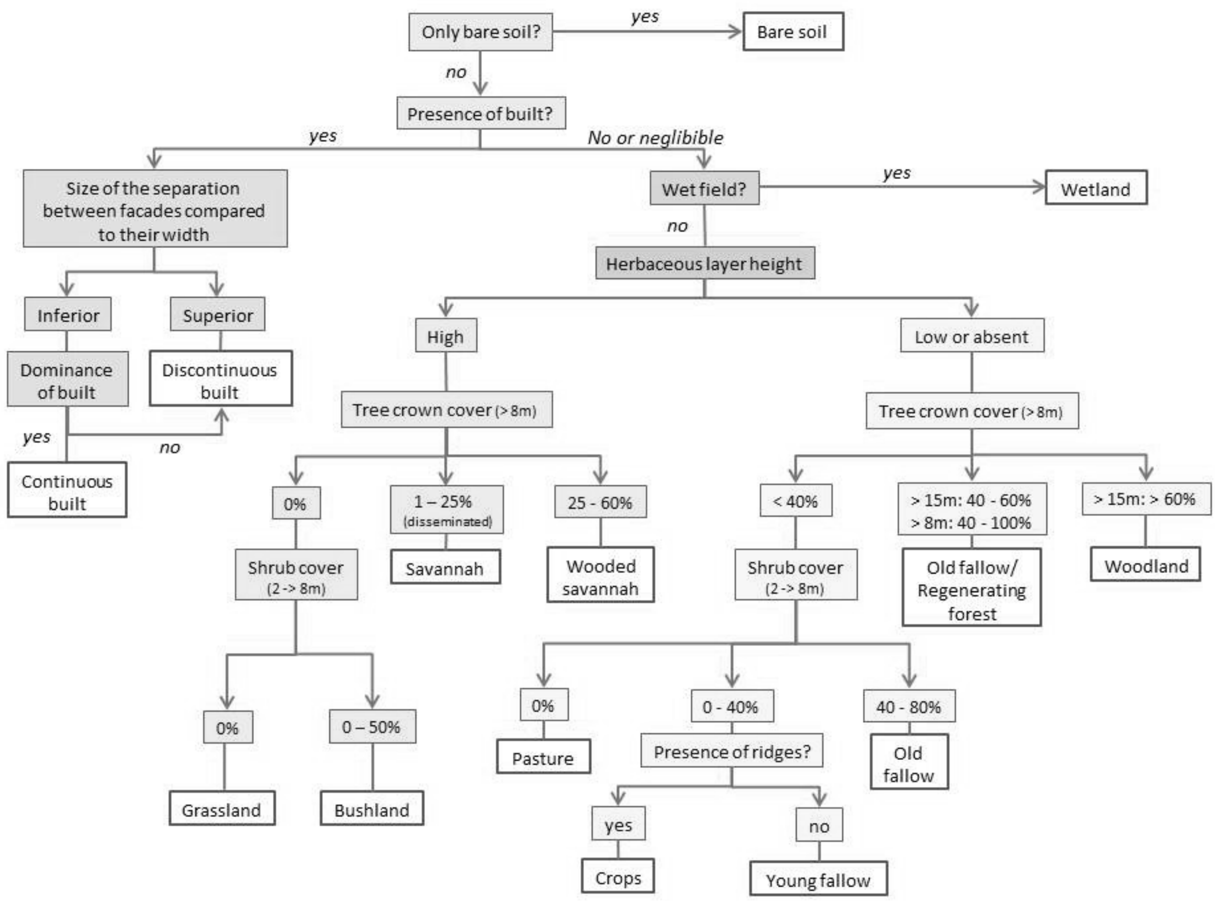

Figure B.1 – *Decision tree used by the analyst in the field to deduce the land cover of the samples points.*

The analyst placed herself in homogeneous imaginary circles of 10 m radius. Inside that circles, the land cover was evaluated. The iterative evaluation begins with the consideration of the presence of bare soil: if it covered the entire circle, then the sample point was identified as “Bare soil”. Otherwise, if built surfaces were present and if the size of the separation between constructions was superior to their width or if there was no dominance of built surfaces, the sample point was assigned to “Discontinuous built”. Alternatively, it was attributed to “Continuous built”. If the amount of built surfaces was null or negligible, then the analyst evaluated the moisture of the field. In case of wet field, the sample point was assigned to “Wetland”. Otherwise, the height of the herbaceous layer was evaluated (plants with a height inferior to 2m are considered as “herbaceous”). If it was considered as high, the tree crown cover was evaluated (a “tree” is considered as a wooded plant with a height superior to 8m). If it was null, then the shrub cover was considered (“shrub” are considered as plants with a height between 2 and 8m). If the shrub cover was null, then the sample point was attributed to “Grassland”. Otherwise (shrub cover of 0-50%), it was attributed to “Bushland”. If the tree cover ranged between 1 and 25%, the sample point was assigned to “Savannah”. If it ranged between 25 and 60%, it was attributed to “Wooded savannah”. When the height of the herbaceous layer was low or when this cover was absent, as previously mentioned, the tree crown cover was evaluated. When inferior to 40%, as aforementioned, shrub cover was evaluated. When null, the sample point was assigned to “Pasture”. When superior to 0 but inferior to 40%, when the presence of ridges on the ground was recognized, the sample point was attributed to “Crops”. Otherwise, it was assigned to “young fallow”. When the shrub cover ranged between 40 and 80%, the point was assigned to “Old fallow”. When the tree crown cover was superior to 40%, then the height of trees was also considered for the segregation. Indeed, when the trees with a height superior to 15m had a crown cover superior to 60%, then the sample was attributed to “Woodland”. Otherwise, it was assigned to “Old Fallow/Regenerating forest”. The criteria used in the decision tree were documented in Trochain (1957), Letouzey (1982) and Bellefontaine (1997). No sample points could be collected in the field for the following classes: Natural grassland, streams, savannah/crop mosaic, recurrent burned areas, reservoirs and slag heap. These land cover classes were identified via Google Earth.

Classified data	Reference data										User's accuracy (%)
	Continuous Built	Crops, pastures, grassland and young fallow	Natural grassland	Discontinuous built, bare soil	Woodland	Savannah, bushland	Savannah/ crops mosaic	Water	Wetland	Wooded savannah	
Continuous Built	7	1	0	12	0	0	7	0	1	0	25.0
Crops, pastures, grassland and young fallow	2	10	1	7	0	3	5	1	4	0	30.3
Natural grassland	0	2	7	0	0	1	0	3	1	4	38.9
Discontinuous built, bare soil	5	1	0	10	0	0	1	0	1	0	55.6
Woodland	0	0	6	0	15	3	1	0	0	8	45.5
Savannah, bushland	0	0	0	0	0	0	0	0	0	0	0.0
Savannah/crops mosaic	0	0	0	0	0	0	0	0	0	0	0.0
Water	0	0	0	0	0	0	0	11	0	0	100.0
Wetland	1	0	0	0	0	0	0	0	8	0	88.9
Wooded savannah	0	0	1	0	0	8	0	0	0	0	0.0
Burned area	0	1	0	1	0	0	1	0	0	3	
Producer's accuracy (%)	46.7	66.7	46.7	33.3	100.0	0.0	0.0	73.3	53.3	0.0	

Overall accuracy: 41.2%; Kappa statistic: 0.397

Figure B.2 – *Confusion matrix for the classification of the 2002 image*

	Continuous Built	Crops, pastures, grassland and young fallow	Natural grassland	Discontinuous built, bare soil	Woodland	Savannah, bushland	Savannah/ crops mosaic	Water	Wetland	Wooded savannah	
Continuous Built	15	2	0	9	0	0	0	2	1	0	51.7
Crops, pastures, grassland and young fallow	0	6	0	2	1	4	9	3	3	0	21.4
Natural grassland	0	0	5	0	0	0	0	2	2	2	45.5
Discontinuous built, bare soil	0	1	0	15	0	0	0	1	2	0	78.9
Woodland	0	2	4	0	8	6	0	1	0	5	30.8
Savannah, bushland	0	1	0	1	0	0	1	0	0	0	0.0
Savannah/crops mosaic	0	0	0	0	0	0	1	0	0	0	100.0
Water	0	0	0	3	0	0	0	5	0	0	62.5
Wetland	0	0	0	0	2	0	0	1	5	0	62.5
Wooded savannah	0	2	5	0	4	3	4	0	1	3	13.6
Burned area	0	1	1	0	0	2	0	0	1	5	
Producer's accuracy (%)	100.0	40.0	33.3	50.0	53.3	0.0	6.7	33.3	33.3	20.0	

Overall accuracy: 38.2%; Kappa statistic: 0.364

Figure B.3 – *Confusion matrix for the classification of the 2013 image*

APPENDIX C

Appendix to Chapter 7

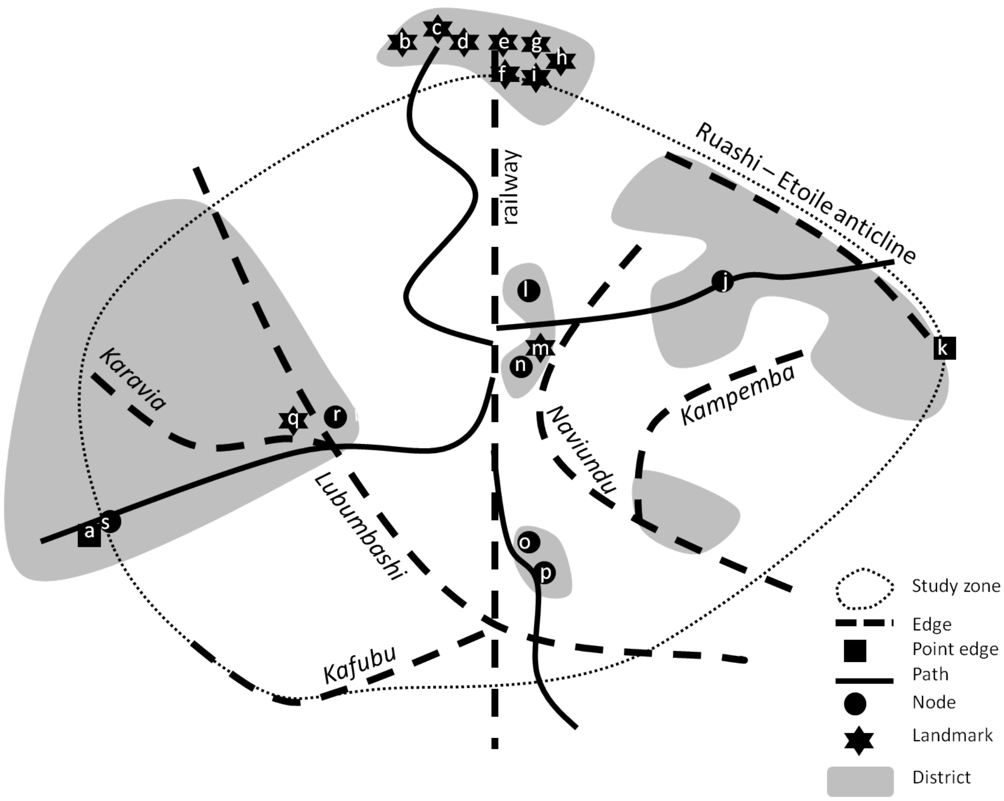

Figure C.1 – *Detail of Figure 7.4: Chorema of the perceived negative impacts of mining and industry in Lubumbashi (synthesis of observers' and collective maps). Each of the six elements composing the city has been represented according to its association with perceived environmental impacts related to mining and/or metallurgic industry. Edges are contaminated main roads or rivers. River names are indicated in italics. Nodes are often mining sites connected to paths related to pollution. Point edges are industrial plants, as Landmarks are. The letters correspond to the following factories or mines: b) Congo Loyal ; c) Congo Julian ; d) Zong Hang ; e) Ngapura Mining ; f) Zong Hua ; g) South China Mining ; h) Cota Mining ; i: Huachim ; j) Ruashi Mining ; k) Etoile mine ; l) Chemaf ; m) Ciment Kat ; n) Simba brewery ; o) STVD ; p) Exaco ; q) Gécamines ; r) STL ; s, a) Somika. Paths correspond to the main roads to mining sites. Districts are mainly characterised by contamination, deforestation, bare soils and metallophyte vegetation.*