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(SPSD II)

Part 4:
Mixed actions

FINAL REPORT

FEASIBILITY OF ECOLOGICAL NETWORKS: ECOLOGICAL, ECONOMIC, SOCIAL AND LEGAL ASPECTS (ECONET)

MA/01

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Research project ECONET (MA01 – Mixed actions)

FEASIBILITY OF ECOLOGICAL NETWORKS:

ECOLOGICAL, ECONOMIC, SOCIAL AND LEGAL ASPECTS

Final Report 2005

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RESOURCE ANALYSIS

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Foreword

This report includes results obtained from the ECONET research project, conducted between January 2003 and March 2005, under the auspices of the Belgian Science Policy. Ecological networks are among the most effective land planning tools that have been promoted in the last few years to control and reverse the actual dramatic process of biodiversity erosion. As such, they have a significant impact on the territory and can be viewed as competitors to other forms of landscape uses such as urban, industrial, agricultural, or recreational. However, they must not be seen as only competitors to such uses; in many instances, complementarities are possible. This implies that the problem of ecological network implementation should be analysed not only from the pure ecological perspective, but that account must be taken to legal preconditions, economic aspects, as well as perception and implication of local and societal actors. We believe our research is one of the first that attempted to integrate those complementary aspects, i.e., ecological, legal, social and economic, in the study of ecological network implementation, and even more particularly in the analysis of an actual case study.

During the research, we benefited from contacts with many persons, and more especially in the scope of a **Users' Committee**, with whom we organised four formal, productive meetings. Besides the representatives of the Belgian Science Policy, we would like to thank all members of the Users' Committee, among which those who supported us and/or participated in one or several of the meetings, namely

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- Sylvie Anciaux (Contrat de Rivière Dyle et affluents)
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- Bart Vercoutere (Vrienden van Heverleebos and Meerdaalwoud)

The report is composed of

- The complete final report;
- The appendices, in three parts, the first of which includes the references;
- The summary report.

CONTENTS

PART 1: INTRODUCTION

The conservation of biodiversity is a basic aspect of sustainable development. In a given landscape, the ultimate consequence of human settlement and exploitation of natural resources is the appearance of a mosaic of (semi-)natural habitats; scattered green "islands" in otherwise intensively cultivated (agriculture, forestry), urban or industrial surroundings. In order to reduce the number of endangered species, threatened with extinction due to habitat fragmentation, one solution is to (re)establish an ecological network, i.e. restoring a network of favourable habitats at the landscape scale. An ecological network can be defined as a set of interconnected zones representing a subdivision of the territory: "central zones" corresponding to long term sustainable habitats for animal and plant species where biodiversity preservation should be the top priority (nature and forest reserves), "restoration or development zones" where human land use is compatible with maintaining a given level of biodiversity, be it permanently or temporarily (extensive agricultural and forestry practices), and "corridors" (linear or stepping stones) providing physical connections (to facilitate migration of individuals between central zones; e.g., hedgerows). In essence, ecological networks aim to provide the physical conditions that are necessary for populations of species to survive in a landscape that to a greater or lesser extent is also exploited by human activities.

Establishing an ecological network means allocating the given resource - the territory - to certain, sustainable, land uses with a view to preserving biodiversity. At this, numerous actors will be directly involved, e.g. landowners, local authorities, nature associations, local people. The feasibility of restoring an ecological network will strongly depend upon their participation, one way or another. In particular, besides the "technical" nature conservation context, the legal, economic and social context of a restoration should be taken into account as well. The economic aspects include the trade-off between (direct/indirect, short term/long term) costs and benefits resulting from a given ecological network scenario. An important issue is the methodology by which such information will be processed (either by classical cost-benefit analysis, or through multi-criteria analysis, or modified analysis accounting for safe minimum standards). Regarding the social context, an important issue is the local stakeholders' perception of ecological networks in their neighbourhood. The way people or entities perceive these networks determines their attitude and willingness to cooperate, and precisely this public commitment is conditional for building the necessary partnerships for ecological networking (implying multi-functional land use) as well as keeping the balance between local, short term interests and global or collective, long term interests. Related to the question of interests and partnerships is the choice of proper instruments and the necessary process management for ecological network projects. Establishing ecological networks occurs inevitably within an extensive legal and regulatory framework. How the conservation of biodiversity, land use and land use planning are regulated at the local and the international level, must therefore be taken into account in this feasibility study. For instance European legislation, and more specifically the Birds and Habitats Directives, imposes on Member States to achieve definite conservation goals, as part of the "Natura 2000 Network". Legal instruments concerning land use planning, rural development, nature conservation and water resources management have to be put on in order to reach these objectives, and to create a solid legal framework containing ecological and socio-economic issues. In trans-boundary areas there may be the additional question of possible conflicting interests (be it economic, socio-cultural…) and laws on both side of the border. The formerly mentioned "perception" can be culture or nationality determined as well.

Until now, ecological networks, legal land use planning and socio-economic assessment of biodiversity have always been dealt with autonomously. This may impose serious constraints on the long-term success of ecological networks. If ecological networks are to be successful, they should incorporate both legal and socio-economic realities at the regional scale, as well as the possible interference between ecological networking and activities that generate revenues or welfare, and other interests. The objectives of this research project were to propose answers to these issues, both from a theoretical, comparative and from an applied perspective.

The way in which the research was designed can be illustrated by Fig. 1.1. It involves both theoretical aspects and application to a case study. Each of the teams (ecological: KULeuven and FUSAGx; legal and economic: UCL; social: Resource Analysis) had to go through the steps illustrated in the upper part of the figure. Based on these theoretical grounds, the case study was approached, first by the ecologists who studied the best way to define an ecological scenario that would be optimal, both in terms of species and in terms of biotopes. The other partners then came into play to evaluate this optimal scenario, in terms of legal, economic and social feasibility. The result of this would be a so-called "integrated scenario" for an ecological network for the case study under consideration. As we will see, the case study was performed at the scale of a relatively small, trans-boundary region, in which all of the aspects mentioned above were meaningful and workable within the time frame of the project, and would yield results that can serve in a more general context.

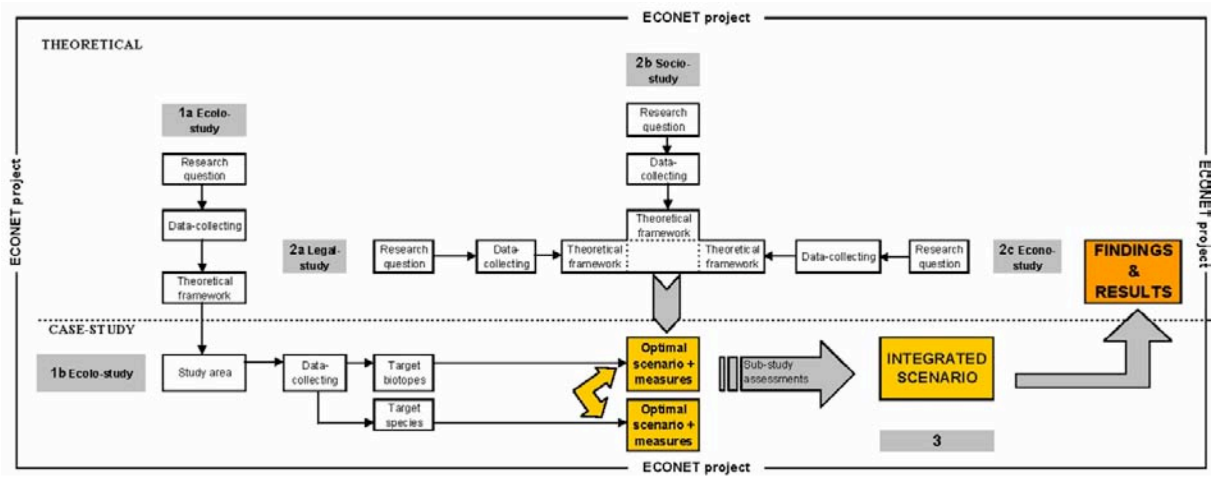

Fig. 1.1. – Design of the ECONET research.

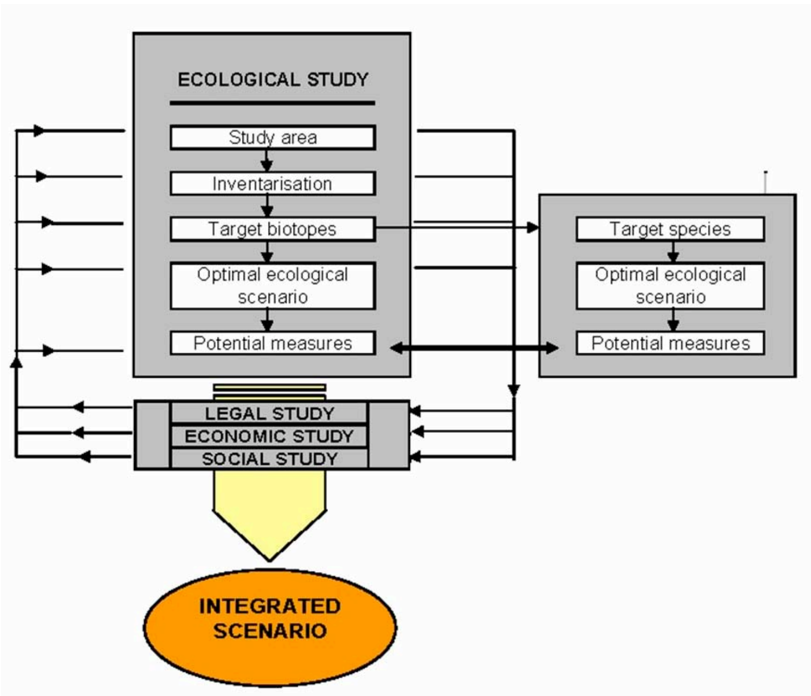

Fig. 1.2. – Dynamic and interactive process of defining an ecological network scenario.

Ideally the research process should be dynamic, integrated and interactive, as Fig. 1.2 illustrates. Legal, economic and social arguments would intervene in such a way as to assess the proposed optimal ecological scenario, each with its particular standpoints and with reference to the case study, resulting in a feedback loop and in a revision of the hypotheses made regarding the scenario. However, as will be seen, several circumstances prevented us from going through the process in a perfectly integrated manner. The result will be, rather, an ex-post assessment of the optimal scenario.

The study was subdivided into different work packages. The first work package included a structured overview of scientific reasons to conserve biodiversity and promote ecological networks, of social perception and economic assessment of these issues and project strategies and of legal instruments relative to ecological networks. The second work package dealt with a case study concerning ecological network restoration. Its ecological, economic, social and legal aspects have been studied thoroughly.

The report is organised as follows. Parts 2 to 5 will examine the approaches adopted by each of the participating teams, which include (1) theoretical perspectives, (2) application to the case study, (3) results obtained for the case study, and (4) discussion followed by a first set of conclusions from the standpoint of each disciplinary field. Part 6 incorporates more integrated conclusions and will discuss the outcomes of the research in terms of the interdisciplinary process, the drawbacks and difficulties encountered during the process. Finally recommendations and comments will be offered as to the possibilities to extrapolate the results of our research.

Part 2: Ecological aspects

2.1 Theoretical background

2.1.1. Context of ecological networks

2.1.1.1. The erosion of biodiversity

In June 1992, the Rio conference resulted in several conventions and proposals for actions in order to promote concrete implementation of the concept of "sustainable development". Sustainable development is a broad concept that refers to the whole set of interactions between economic, social and environmental aspects and seeks to reach an equilibrium between various requirements linked with these. It is a process of well-balanced development that "meets the needs of the present without compromising the ability of future generations to meet their own needs" (WCED 1987). In this perspective, the conservation of biological diversity appeared as one of the main preoccupations. During the last century, the rate of loss of biodiversity has accelerated at all levels, be it genes, species or communities, under the influence of anthropogenic pressure. The rate of species extinction is significantly higher now than all rates observed in paleontologic archives (Pimm & Raven 2000).

Many scientists are convinced that the concept of biodiversity cannot be dissociated from the idea of sustainable development (e.g. Barbier *et al*. 1994, Dolman 2000). Indeed, in sustainable development every human action should be undertaken in order to provide future generations with a set of options at least as large as the one from which we beneficiate today; biodiversity is one of the main components in this purpose (Jonas 1990, Vercelli 1996). Additionally, biological diversity may be essential for the stability and resilience of ecosystems, which provide the whole living world with various functions of vital importance at a local and global scale (climate regulation, soil conservation, food production…; Jeffries 1997). As a result, the conservation of biodiversity can be viewed as a necessary condition for sustainable development. On the other hand, people increasingly agree that effective conservation of biodiversity cannot be implemented without accounting for socio-economic decision and assessment processes as well.

Belgium, with his variety of physical environments, has a high biological diversity. This "natural biodiversity" has been strongly influenced by human activities such as agriculture and urbanization. The data available for mammals, birds, fishes, butterflies and for other categories of species, show that, nowadays in Wallonia, on 640 species, one third have disappeared (4.5%), in danger (8.6%) or vulnerable (20.2%). For fishes and butterflies, the percentage of extinct, threatened and vulnerable species exceeds 50%. In Flanders, the percentage of extinct, threatened and vulnerable birds reaches 43% (Anonymous 2002c). The main causes of this diminution of biodiversity are well known: 1) destruction and fragmentation of natural and semi-natural habitats; 2) overexploitation of biological resources; 3) introduction of exotic invasive species (Olivieri & Vitalis 2001).

2.1.1.2. Landscape dynamics and ecological processes

Ecological systems, whatever the human activities intensity that occur, are dynamic. Indeed, change is one of the intrinsic characteristic of landscapes: from long term changes related to geological phenomena, species evolution and their migration on earth, to short term changes related to physiological rates (flowering, leaf fall) and seasons (Burel & Baudry 1999). Human activities such as agriculture, forestry and urbanization have become, in many places, the main driving force of landscape dynamics (Baudry & Tatony 1993). This land transformation may severely compromise the integrity of ecological systems through loss of native species, invasion of exotic species, pronounced soil erosion and decreased water quality (Forman & Godron 1986). The remnants of native communities after such modifications are generally reduced in size and disconnected from adjacent, continuous habitat. As a result, the populations of plants and animals which occur in these remnants also are subdivided and reduced, which may either exclude some species immediately or increase their probability of extinction (Collingue 1996, Saunders 1991). Ecologists, conservationists, and land managers generally refer to these two components of land transformation, habitat loss and isolation, as 'habitat fragmentation'. Natural perturbations, such as glaciations, fire, floodings, hurricanes and volcanic eruptions are the first cause of fragmentation. In this sense, fragmentation is not a new phenomenon (Andrèn 1994). However, the dramatic increase of human population during the last century has gone with the intensification of commercial and residential development, agriculture and deforestation (Reed *et al.* 1996). In the strictest sense, "fragmentation" means the division of a whole into smaller parts (Forman & Godron 1986), *i.e.* the progressive division of a big and homogeneous habitat in a multitude of smaller and heterogeneous parts (Saunders 1991). Consequences of fragmentation include habitat loss for some plant and animal species, habitat creation for others, decreased connectivity of the remaining vegetation patches, decreased patch size, increased distance between patches, and an increase in edge at the expense of interior habitat (Reed *et al.* 1996). Such a structure modifies the spatial dynamics of population systems and hampers their survival (impossible migration, insufficient genetic exchange etc.; Opdam *et al*. 1993, Meffe & Carroll 1994). The change, fragmentation and disappearance of natural habitats are claimed to be the main factors causing a decrease in biodiversity in the world (Forman 1995).

The composition and structure of most landscapes reflect the interrelation between the physical environment and the land-use history (Grossi *et al.* 1995). Human contributions to spatial heterogeneity are significant ; current conditions usually result from 'layers' of past activities, each of them leaving a footprint on the landscape that persists long after the activity has ceased (Andersen *et al.* 1996). Land use influences the functioning of ecosystems as a whole, its self-purification capacity and the carrying capacity of the landscape (Kavaliauskas 1995). It also affects habitat quality for wild species and the potential for dispersal and migration that are vital for survival of populations especially in a fragmented landscape (Jongman & Pungetti 2004).

Present landscapes are dominated by man-made dynamic habitats, and the less dynamic habitats are small and isolated as are the populations in them. Habitat isolation and habitat loss prevent natural species from developing viable populations or let populations survive at different equilibrium levels (Hanski *et al.* 1985). Natural interconnections have declined with the disappearance of forested and river corridors, and with the development of human infrastructures. The strategy to overcome this is the redevelopment of ecological coherence through networks.

2.1.1.3. Landscape dynamics and population functioning

The landscape is heterogeneous and dynamic. It is a mosaic of habitats and a lot of species use, during their life cycle, several elements of this mosaic. Movements of individuals in the landscape are a key-process of the comprehension of ecological processes (Burel & Baudry 1999). Plants and animals are dispersed both by wind and water, with the help of other species, or by their own movements. Migration is a special kind of dispersal, directed to a certain site. Dispersal is essential in population survival and the functioning of biotopes. However, dispersal can only take place if there are means for dispersal and sites to disperse from and to. In general, restriction of species dispersal increases the chance of species extinction (Jongman & Pungetti 2004). A good dispersal capacity can allow a species to compensate for the fragmentation of its habitat (Kalkhoven 1993). The dynamics of fragmented populations directly depend on properties of landscape elements and their spatial arrangement in the landscape. The size, form and quality of the habitat patches govern the dynamics of local populations. The quality of "corridors" and the matrix hostility condition the possibility of species exchanges between patches, and thus determine at least part of the immigration process (Verboom *et al.* 1993). At landscape level, patch and corridor density and their spatial arrangement determine the connectivity of the landscape for a given species (Burel & Baudry 1999).

2.1.1.4. A short nature conservation history

Throughout the centuries, land use was adapted to the restricted technical ability of people to change the land. In Europe this led to a rather stable pattern of landscapes until the second half of the nineteenth century. Then, around 1850, the industrial revolution started. It means a revolution not only in the urban environment, but also in the rural environment. Machines were introduced as well as fertilizers and wire fencing. Semi-natural areas were converted into agricultural land and the scale of agricultural holdings increased. This process started on a small scale, but has continued until the present day.

The history of nature conservation and urban ecological networks started as a reaction to the industrial revolution. Already in the last half of the nineteenth century and the first half years of the twentieth century, nature was integrated into urban planning, for instance when the main axes of towns were developed into green boulevards, such as the Champs Elysées and the footpaths along the Seine in Paris (Searns 1995). The industrial revolution had a heavy impact on the cities and there was a growing need for urban green. Nature conservation and urban development joined forces in the 1920s. The statement was made that nature is important for outdoor recreation, for its scenic beauty and its intrinsic value. After the Second World War nature conservation focused more on the preservation of values within semi-natural landscapes. This was especially important in the northern states of Europe, where the decline of nature was alarming. After the first nature conservation year, 1970, changes took place in nature conservation in Western Europe. Nature conservation acts were revised in many western European states, in some cases by amending existing legislation and in other cases by formulating a new and more integrated nature conservation policy relating issues such as recreation, urbanization, regional planning and agriculture (Jongman & Pungetti 2004, Jongman 1995, Jongman *et al.* 2003). Traditionally, conservation efforts promoted a 'species or area approach', by implementing measures mainly aiming at protecting areas with populations of particularly vulnerable or threatened species (Primack 1998). This approach resulted in the design of sanctuaries, devoted to the protection of biodiversity with little attention paid to other parts or aspects of the territory or landscape matrix (e.g., Myers *et al.* 2000). In that period the linkage between protected sites did not seem to be crucial. This method is now recognised as being largely insufficient to *sustainable* preservation of biodiversity, as sanctuaries represent only a negligible part of the earth's surface (and hence a low portion of global biodiversity) and because of the additional effects of habitat fragmentation (Burkey 1989).

The general report of constant erosion of biodiversity leads to serious uncertainty about the objectives of nature conservation, both for earlier strategies and for future planning. In spite of the energy displayed by many actors, more than 30 years of struggle to protect sites have not stopped the decline of biodiversity. The emergence of new scientific paradigms has allowed the identification of the shortcomings of a strategy based on preservation of nature reserves.

2.1.2. The scientific basis of ecological networks

Scientists have approached the problem of habitat fragmentation for the past 25 years largely within the framework of two key theoretical developments in community and population ecology: the theory of island biogeography and metapopulation dynamics.

2.1.2.1. The theory of island biogeography

The theory of island biogeography was elaborated by MacArthur and Wilson in 1963 to explain species diversity of animal communities on oceanic islands. In particular, this theory postulated that the size of an oceanic island and its distance from a continental source of colonizing species would determine the number of species present on the island (Fig. 2.1). Islands close to a mainland would likely have higher immigration rates than more distant islands, while large islands would likely have lower extinction rates than small islands (Simberloff 1976). Thus, large islands close to continents were predicted to have a higher number of species than small islands more distant from continents. The authors suggested that while this theory focused on species diversity on oceanic islands, the predictions may be consistent for plant and animal communities inhabiting terrestrial ‘islands’ (MacArthur & Wilson 1967).

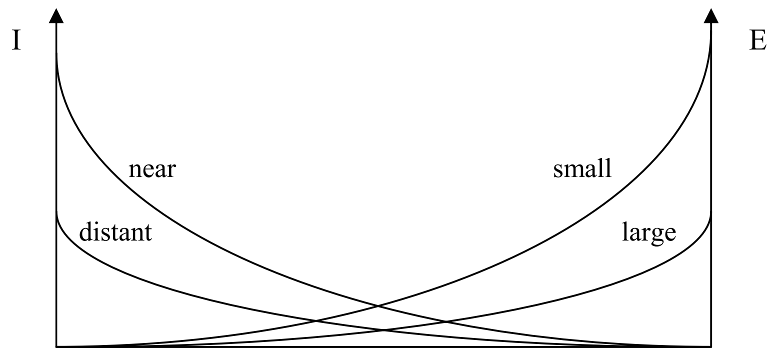

Fig. 2.1: The theory of island biogeography of MacArthur and Wilson (1967) (I: immigration rate, E: extinction rate).

The theory of island biogeography has lead to a more integrated vision of nature conservation where space has become more heterogeneous and where natural patches are separated by intensive agricultural zones, urban zones or barriers like communication and transport infrastructures. The reduction of extinction rates and the maintenance of immigration possibilities became thus objectives of nature conservation. Models based on this theory have allowed the delineation and the spatial distribution of natural reserves (Shafer 1990).

2.1.2.2. The metapopulation theory

The metapopulation theory was elaborated by Levins in 1969 to describe and predict the population dynamics of species occupying naturally patchy habitats, such as mountaintops. A “metapopulation” is a set of local populations that interact via individuals moving among them (Levins, 1970). For many species in man-dominated landscapes, natural habitats occur in small, spatially separated fragments. Species that inhabit these fragments may function as metapopulations. The metapopulation model describes the dynamics of an infinite number of identical patches, in which all occupied patches have a constant extinction rate, whereas all empty patches have a colonization rate that is proportional to the fraction of patches occupied and a colonization parameter (Verboom *et al.* 1993). In this model, local populations of organisms undergo periodic colonization and extinction, while the metapopulation as a whole persists indefinitely (Fig. 2.2). Ecologists have directly applied the understanding of the oscillations of such naturally transient populations to predict the persistence of species occurring in human-induced habitat fragments (Collingue 1996).

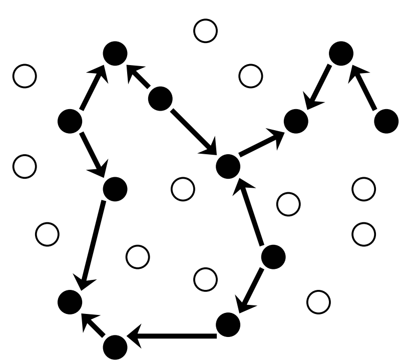

Fig. 2.2: Metapopulation model of Levins (1970). Black: ‘occupied’ patches, white: ‘unoccupied’ patches

In its basic formulation, the metapopulation states that the persistence of a group of populations in a region depends on two rates: an extinction rate with decreasing as a function of the surfaces of the sites with adequate habitats (the larger area, the lower the extinction rate) and a colonization rate depending on the isolation of the different sites (the more isolated, the lower the colonization rate). The colonization rate must thus be higher or at least equal to the extinction rate for species persistence at the landscape scale. In such a situation, if the isolation of the different populations is high, the patch area must be increased to lower the extinction rate. If, on the other hand, the patch area is small, the extinction rate will be high and thus the connectivity of sites must be increased to increase the colonization rate (Hanski & Simberloff 1997). To achieve this subtle equilibrium between extinction and colonization rates, that is what the implementation of ecological networks is aiming at.

However, studies about metapopulations indicate that species do not occupy necessarily all the favourable patches. Only a part of them can be colonized. This means that a species can be in danger even if it occupies all the favourable patches when the number of patches is low. On the contrary, in theory, an occupation of only 50% of more numerous sites lowers species extinction probability. The proportion of occupied patches by a metapopulation increases according to dispersal and migration facilities between the different patches. To favour these movements, the maintenance or the creation of links (corridors) are necessary as well as the amelioration of life conditions in the rest of the territory with intensive human activities (Verboom *et al.* 1993).

These theories constitute the theoretical framework for the implementation of ecological networks. They also underline the two main weaknesses of the traditional strategy of biodiversity conservation: 1) The place available for wild fauna and flora populations is clearly insufficient; 2) the individual management of sites does not integrate the effects of fragmentation on population dynamics at landscape level.

2.1.2.3. The ecological network: a landscape strategy

These ecological considerations make it increasingly important to think of nature conservation not only as species protection and site protection, but also as the management of coherent spatial structures (Jongman 1995). Today, it is clearly evident that biological diversity can only be preserved by a global management of the territory. From the theoretical paradigms cited above, emerges a new territorial planning strategy in the beginning of eighties: the **ecological network**. This concept appears as a functional response to habitat fragmentation effects on species communities. In this way, the ecological network is considered as a framework of ecological components, e.g. core areas, corridors and development zones, which provides the physical conditions necessary for ecosystems and species populations to survive in a human-dominated landscape (Jongman & Pungetti 2004). It stresses the importance of the size of habitat patches which guarantee the local population viability and on the maintenance of the elements that allow dispersion between the populations.

The former practice of nature conservation to focus on individual sites and manage them in respect with their unique communities has changed. Today, nature conservation perceives a site and habitat as part of a whole, as part of the complex landscape, and consequently bases decisions, priorities and targets evenly on reflections concerning the role of the site in the ecological complex of the landscape. Hence, the crucial task is to attain a functional connectivity maintained by biological and geophysical interrelations (Kuijken 2002). Planning is the key factor that allows these goals to be reached, since creating ecological networks requires an established planning system, be it regional or local.

In his application on biodiversity management, the concept of ecological networks is rather considered as a territorial planning tool aiming a partition of the landscape in objective zones. In general, at least three types of zones corresponding to three functions are considered:

- The **central zones** are zones containing species populations and habitats of high patrimonial value with a high conservation status, where preservation of biodiversity should be the top priority. These zones deserve a strong conservation status

- The **development zones** (or associated zones) are zones of lower biological value but they contain an important potential of biodiversity. These zones need a priori a smaller protection than central zones. In these zones the coexistence between different objectives is possible

- The **ecological corridors** are different landscape structures, with variable form and size (linear or tortuous, large or narrow) corresponding to structural links which, penetrating the landscape, maintain or restore the natural connectivity.

The protection of habitats and species, however, is affected by many factors interrelated with each other. From the latter, variations to the environment can be influenced not only by planning, but also by administrational decisions, community values, environmental attitudes, political and economic situations. After all the final goal of ecological network development is nature conservation; the former is only a tool to reach the latter. Ecological networks are one of the possible measures to tackle species and habitat conservation. Hence, a combination of several measures is the best practise for ensuring alternative solutions to the problem of environmental fragmentation (Jongman & Pungetti 2004).

2.1.3 The implementation of ecological network

2.1.3.1 Introduction

Restoring an ecological network requires that the whole set of landscape components (i.e. protected areas devoted to the conservation of biodiversity and areas directly affected by human activities) is taken into account and that their capacity to host, permanently or temporarily, populations of species, representative of local biodiversity, is assessed. The concept of ecological networks implies that biodiversity can only be maintained through concerted management of the entire landscape. Establishing an ecological network is then similar to allocating a given resource, the territory, to certain sustainable uses with the aim to preserve biodiversity.

2.1.3.2 Habitat and species conflicts

In practise, the implementation of the ecological network as a territorial planning and habitat restoration tool usually leads to the designation of the different zones without integrating the functional relations between these zones. To be functional, an ecological network in a landscape must integrate these constraints:

- At the landscape level, the conservation objectives are multiple and a source of conflicts. Indeed, the conservation objectives may differ for different habitats or species.

- Beyond these incompatibilities of objectives, within a certain habitat type, all species do not have the same needs for a functional ecological network. Because the requirements or characteristics of species can vary considerably, incompatibilities of functions of some zones may appear. Corridor zones and core areas for some species can be real obstacles for the movements of other species.

Thus, at a spatial scale, there is not ONE ecological network but SEVERAL ecological networks, i.e. thematic ecological networks. It is thus necessary to define precise objectives at the scale of the landscape when ecological networks are to be restored. In fact, much of these thematic networks will be limited at restricted regions where core populations will be maintained. Indeed, it is impossible to maximise the biological objectives on all the landscape but to guarantee the maintenance or restoration of natural processes in certain sites.

Moreover, ecological networks must be interconnected according to the spatial scale at which they have been defined. The spatial scale of these networks can be very different according to the biological objectives. A network for a target species with a low dispersal capacity or for a target habitat with a large distribution will result in different networks with different sizes and structures. A European ecological network is not a simple addition of national or regional ecological networks. It results from the choices of objectives (target habitat and target species) made at the European scale (Dufrêne 2003).

2.1.3.3. Different steps

a) Assessment of the current context

The implementation of ecological networks starts with the inventory of the current situation. In a first step, all the data is gathered to provide information about the current state of knowledge of the landscape. Through exploration of databases, maps and inventories, data on species distribution and the status of habitats is obtained. Data exploration is executed at different spatial scales: e.g. at a national/regional level for the delineation of large forests, at the scale of municipalities for the distribution of target habitats and species and at a local scale for the location of small landscape elements (hedges, lines of trees or isolated trees). With this step, we aim to have an overview of the study area in order to define the primary conservation objectives.

b) Identification of conservation objectives

To be efficient, an ecological network must answer at predefined pragmatic conservation objectives such as target-species or target-habitats. These species/habitats are chosen based on their high biological importance either because they are very threatened and need urgent direct actions, either because they are indicator species of high patrimonial habitats.

An indicator may be ‘a species, a structure, a process or some other feature of a biological system, the occurrence of which insures the maintenance or restoration of the most important aspects of biodiversity for that system. The interest in indicators has a long history within ecology (Hansson 2000). The earliest use was probably to manually demarcate various plant associations within phytosociology. Indicators have been commonly used in ecotoxicology to demonstrate possible toxic effects of environmental contaminants. Indicators have also been used to demonstrate general population trends. Finally, indicators have already been used in conservation biology, as *umbrella species* (usually large species with wide areal requirements, presumed to also cover the requirements of other species; Launer & Murphy 1994) or *flagships* (large appealing species attracting interest to their ecosystem, e.g. pandas; Noss 1990). The choice of indicator species for the implementation of an ecological network is a crucial step. The ecological network of the chosen target species would need to be also functional for all other species of the habitat. Ideally, these species will have a large home-range and a low dispersal capacity.

At a landscape scale, several possible combinations of target species and habitats may exist and they form ecological thematic networks. The spatial scale of these networks is largely different according to the biological objectives.

c) Analysis of the current situation

Here, the aim is to analyze the functionality of the current network in order to assess its current efficiency and to underline the priorities. At this stage, it is important to assess the biological potential of the landscape and to underline their complementarities and oppositions with respect to the predefined objectives. Conflicts between target habitats or between target habitats and target species may emerge at a landscape level. Hence, some of the objectives need to be reconsidered. Therefore it is important to take the potential of the habitats into account and not only their current state of conservation.

d) Planning of thematic networks

This stage consists of the conversion from current situation to the thematic ecological networks, i.e. the delineation of core areas, development zones and corridors. Several approaches and/or theories were developed to design a network:

- GAP-method:
This technique uses geographic information system (GIS) technology (maps of vegetation cover, species locations, existing reserves, land ownership,…) to identify gaps in an existing reserve network (Prendergast *et al.* 1999, Salem 2003).

- LARCH-model (**LA**ndscape ecological **R**ules for the **C**onservation of **H**abitat):
LARCH (a computer-approach) is designed as an expert system for the evaluation of ecological networks. It uses the concept of sustainable networks of habitat patches to assess the degree of viability for a species group). LARCH takes habitat characteristics (type & size) and species characteristics (dispersal) into account (Pouwels 2000, Bruinderink *et al.* 2003).

- Multi-criteria analysis:
Based on different requirements of the target species and on characteristics of the habitats (e.g. habitats, size and form of the patches, connectivity, isolation, state of conservation…) the suitability of different patches (areas) is evaluated. In a final step, a feasible and suitable connection between those patches is selected.

e) Combination of thematic networks

When thematic ecological networks are defined at landscape level, they cannot be considered separately, they must be shown as part as a whole. The thematic ecological networks must be integrated according to the selected spatial scale. Then, by combination of the several thematic ecological networks, the global one is defined.

f) Implementation and management

This crucial step concerns the implementation of thematic ecological networks on the field. This stage depends on the competence of nature conservation authorities and the corresponding legal tools and restrictions. It is based on scientific recommendations and on the realization of a dashboard. At this stage, it is important to also take into account the role of the matrix surrounding the existing set of nature reserves.

2.1.3.4. Examples in Europe

During the last decades of the twentieth century, ecological networks have been developed by authorities and scientific institutions in Europe, America and Australia (Bennett & Wit 2001). In Europe this has been partly a Europe-wide approach as a reaction to the Convention on Biological Diversity or as newly developed European policy, such as NATURA 2000 (Habitat and Species Directive, EC 92/34), the Emerald Network and the Pan-European Biological and Landscape Diversity Strategy (Anonymous 1996). These have been scientific studies and national initiatives varying from strategies for adaptation of conservation policies to development of ecological networks in a scientific approach. Nowadays, most of the ecological networks in Europe are part of national and regional nature conservation policies.

This chapter does not intend to present a complete picture of these developments, but to show common denominators and concepts to highlight common developments between countries and regions, and indicate where and when differences between countries and regions have to be taken into account when developing ecological networks. Understanding these differences and common issues is of utmost importance to find common principles and approaches and to know when differences are important enough to be maintained or when they can be neglected.

On the local and regional level, as well as on the national and international level, authorities and planners are developing instruments and are designating land in order to establish functional networks of ecosystems and habitats that are of relevance for the corresponding biodiversity. Throughout Europe, regional and national approaches are in different developmental phases. Ecological networks are interpreted in a variety of ways depending on different historical backgrounds of nature conservation planning, varying scientific traditions, different geographical and administrative levels and different land uses. Finally, the political decision-making is dependent on actors with different land use interests. This complex interaction between cultural and natural features results in quite different approaches to the elaboration of ecological networks.

Nowadays, three main **landscape ecological concepts** help to understand the different strategies for the elaboration of ecological networks in Europe: the ecostabilisation principle, ecological principles (dispersal and migration, connectivity and connectedness) and principles for river systems.

a) Ecostabilisation principle

The German and Eastern European tradition in applied geography has concentrated on regional relationships and has found applications in physical planning. Spatial planning in the Soviet era was subordinate to the rules of the planned economy. This kind of planning initiated large-scale technocratic projects and a mono-functional simplification of the collectivised agricultural landscape. Until the end of the 1980s nature did not figure in the spatial planning maps of countries such as East Germany, and zoning was directed by economic principles (Jongman & Pungetti 2004). For mitigation of the impact of economic planning in central and Eastern Europe, the ecostabilisation principle was developed in the early 1980s, resulting in planning concepts such as '*territorial systems supporting landscape ecological stability*'. In theory, the approach was based on the idea of a *polarised landscape* that was worked out in 1947 by the Russian geographer Rodoman (Mander *et al.* 1995). His principle is a functional zoning of the landscape elements into natural zones that antagonises the poles of intensive land use. The fundamental principle of the concept is strict delimitation of natural zones and zones for restoration as antagonistic poles from zones with intensive land use (agriculture, industry, urban areas), and uniting all natural zones into one coherent network (Jongman *et al.* 2003). It resulted in concepts such as '*nature frame*' (Lithuania), '*natural backbone*', '*ecological compensative areas'* (Estonia) and '*ecostabilising functions'*. Essential to these concepts are:

- The designation of landscape units to function as an ecological compensation to the territories that are heavily exploited
- The linkage of these compensative territories by zones with coherent land management
- The availability of sufficient space to create compensation zones and linkages between them.

The important principle behind the ecostabilisation concept for spatial planning is the acknowledgement of the importance of processes at the landscape scale, the presence of flows, the role of ecotones, and the use of the ability of nature to purify and restore (Jongmann & Pungetti 2004).

b) Ecological principles: dispersal and migration, connectivity and connectedness

Movement is itself the product of evolutionary pressures contributing in many ways to the survival and the reproduction of species (Hobbs, 1992). Restriction of species dispersal increases the chance of species extinction (Taylor et al, 1993). The main functional aspects of the landscape that are of importance for dispersal and persistence of populations are connectivity and connectedness. According to Baudry and Merriam (1988), connectivity is a parameter of landscape functioning, which measures the processes by which sub-populations of organisms are interconnected into a functional demographic unit. Connectedness refers to the structural links between elements of the spatial structure of a landscape and can be described from mappable elements (Jongman *et al.* 2003). Structural elements are different from functional parameters. For some species, connectivity is measured in the distance

Table 2.1: Criteria, basic data, type of networks and main functions of the ecological networks in Europe (source: Jongman, 1995; Jongman *et al.* 2003, Jongman & Kristiansen 2001, Brandt 1995, Burkhardt *et al.* 1995, Doms *et al.* 1995, De Blust *et al.* 1995, Kavaliauskas 1995, Mander *et al.* 1995, Troumbis 1995, Van Zadelhoff & Lammers 1995).

Network	Type of network	Main functions	Zones	Criteria	Basic data
National Ecological network (Netherlands)	Physical structure of areas with nature conservation interst (national level)	Ecological, river systems	Core areas, nature development areas, ecological corridors	Biodiversity, disturbance, geology, cultural history and aesthetics. For target species : international importance, negative trend in the national level and rarity at the national level	Areas of nature conservation interest, hydrology, cultural history, species changes, water quality
Nature frame of Lithuania	Hierarchical structure of natural areas at national, regional and local level	Ecostabilisation, river systems	Geoecological divides, Areas of inner stabilization, Migration corridors	Biodiversity, conservation status, location-spatial structure	Geology, hydrology, geography, ecology, economy, sociology and architecture
Territorial system of landscape ecological stability (TSLES), Czech Republic	Hierarchical structure of natural areas at national, regional and local level	Ecostabilisation, ecological	Biocentres, buffer zones, corridors	Biodiversity, representativeness, location and protection status	Geology, hydrology, species composition, vegetation structure, land use, climate, species diversity, edge composition, history
Territorial system of landscape ecological stability (TSLES), Slovak Republic	Hierarchical structure of natural areas at national, regional and local level	Ecostabilisation, ecological	Biocentres, biocorridor and interactive elements of supraregional, regional or local importance	Location, representativeness, protection status	Geology, hydrology, species composition, vegetation structure, land use
Network of compensative areas (Estonia)	Physical structure based on hexagonal units of intensive to extensive land use	Ecostabilisation	Core areas, buffer zones and nature development areas	Protected areas, large natural areas, size	Land use topography, geology, climate, soil, water, vegetation, forest, roads maps
Design of a nature reserve system in Greece	List of reserves and strategy on regional planning	Ecological	Core areas, natural corridors and buffer zones	Size of habitats, mutual proximity, shape, distribution in biogeographic districts	Vegetation maps, hydrographic network, land uses, slopes, species and habitats distribution, soil erosion
Green lungs of Poland	Action program for suistainable development including a network of nature conservation areas	Ecostabilisation, ecological, river systems	Nature conservation areas	Natural areas size, nature protection status, urban development	Not available
Ecological Network of Flanders, Belgium	Coherent structure of areas in which nature conservation policy in the main objective to be developed in the network and in its supporting network	Ecological	Core areas, nature development areas, corridor areas and buffer zones	Size, nature conservation value, , distribution, density, degree of human influence	Nature areas maps, biological evaluation maps, flora distribution maps, waterflow, topographic maps, soil, representativeness, management practises
Vermetzter Biotopsysteme Rheinland Pfalz, Germany	Physical structure of core areas and corridor zones and a list of priorities	Ecological, river systems	Core areas, corridor zones	Biotope types, selected species	Biotopes, land cover, forestry, water quality, species, natural vegetation
Ecological Networks of Denmark	Core areas and ecological corridors aiming at the creation of a coherent structure to facilitate dispersal of species	Ecological, river systems	Core areas and dispersal corridors	Importance for regional and (inter)national nature conservation	Nature types, flaura and fauna distribution, inventories

between sites, for other species the structure of the landscape, the connectedness through hedgerows represents the presence of corridors and barriers. Habitat area reduction will cause a reduction of the populations that can survive and in this way an increased risk of local extinction. It also will increase the need for species to disperse between sites through a more or less hostile landscape (Jongman & Pungetti 2004). Many European countries based their nature conservation strategies on this concept: Belgium (Ecological Network of Flanders), Denmark, Germany, Italy, The Netherlands, Portugal, Spain and UK.

c) Principles for river systems

Rivers do play a crucial role in the structure of ecological networks, mainly as a connecting landscape feature. A river itself is more than the sum of its parts and it is not a static body of water, but rather a continuum with a changing ecological structure and function. The river continuum concept is based on macro-invertebrates and it states that from the headwaters to the river mouth a continuous change in macro-invertebrate community takes place (Wenger 2002). Rivers are often not considered as one system. In general, several authorities and users decide on its management and use, because of the many borders that are crossed and the many interests that are involved. Owing to their role as transport mechanisms for nutrients, matter and species, rivers should be key systems in spatial planning and at least they are in the development of ecological networks (Jongman & Pungetti 2004). This principle is included in several strategies for the development of ecological networks in Europe, in particular in Denmark, Germany, The Netherlands, Poland and Portugal.

In most European countries, mapping of biotic and abiotic resources and conditions has been carried out and the results have been used in different ways and different scales as **criteria** for location of the network. In Central Eastern Europe, a large amount of data from geology, geomorphology, geography, hydrology, soil physics, etc. has formed the basis for design of the ecological network. In contrast, the development of ecological networks in Western Europe have focused more on the protection of valuable sites and threatened species with long dispersal ranges (Jongman *et al.* 2003). These differences in the choice of criteria, delineation of different zones and basic data are shown in Table 2.1.

2.2. Towards an optimal ecological network in the Dyle valley: a metapopulation capacity based approach

2.2.1. Introduction

The ultimate consequence of human settlement and natural resource exploitation is the appearance of a mosaic of (semi-)natural habitats, *i.e.*, scattered ‘green islands’ in otherwise intensively cultivated (agriculture, forestry), urban or industrial surroundings. The remnants of native vegetation left after such modifications are generally reduced in size and disconnected from adjacent, continuous habitat. As a result, the populations of plants and animals which occur in these remnants also are subdivided and reduced, which may either exclude certain species immediately or increase their probability of extinction in the long term (Collingue 1996, Wiegand *et al.* 2005).

This process of **habitat loss** and **isolation** is commonly termed ‘**habitat fragmentation**’ which may be defined in this context as the progressive division of a large, homogeneous habitat into a heterogeneous mixture of much smaller patches (Saunders *et al.* 1991, Rutledge 2003). Consequences of fragmentation include habitat loss for some plant and animal species, habitat creation for others, decreased connectivity of the remaining habitats, decreased patch size, increased distances between suitable patches and severely increased edge/core habitat for smaller patches. Habitat loss is widely considered to be the principal threat to the long-term survival of species both locally and worldwide (Soulé 1986, Barbault & Sastrapradja 1995, Wiens 1996, Hanski and Ovaskainen 2000, Rutledge 2003). In order to reduce the number of endangered species, threatened with extinction due to habitat fragmentation, one solution is to (re)establish an **ecological network (EN)**, *i.e.*, restoring a network of favourable habitats at the landscape scale (Fahrig & Merriam 1985, Lefeuvre 1998, Honnay *et al.* 2002). According to Delescaille (1993), an **'**ecological network**'** can be defined as "*the whole of biotopes suitable for providing a maximum of species a temporal or permanent living environment, consistent with their requirements and guaranteeing their survival in the long term*". Ecological networks aim to provide the physical conditions that are necessary for populations of species to survive in a landscape that to a greater or lesser extent is also exploited by human activities. Only in such a way, a high level of biodiversity can be maintained or a higher level can be reached than would be observed in fragmented landscapes.

An ecological network can be seen in two different perspectives, as a purely '**structural**' network or as a more '**functional**' association of habitats. To sustain viable populations and by that enhance biodiversity, it is important that the network -a structure of patches in the landscape- can actually be used by species; that it is functional. By consequence, species-specific requirements (e.g. minimum area requirement, dispersal distance) may not be neglected in the process of establishing an ecological network (Wiegand *et al.* 2005). The way in which population dynamics are affected by landscape structure - especially the consequences of fragmentation and habitat loss - has become a major focus of ecological research (e.g. Wiens *et al.* 1993, Fahrig & Merriam 1994, Wiegand *et al.* 1999, Rutledge 2003). Ecologists and conservation biologists have used many measures of landscape structure to predict the population dynamic consequences of habitat loss and fragmentation, but these measures are in most cases not well justified by population dynamic theory (Hanski & Ovaskainen 2000, Rutledge 2003). Most of these indices should only be used to describe landscape pattern (Rutledge 2003). A new measure, the **metapopulation capacity (λ_M)** however, is rigorously derived from metapopulation theory and can easily be applied to real networks of habitat fragments with known areas and connectivities (Hanski & Ovaskainen 2000). A species is predicted to persist in a landscape if the metapopulation capacity of that landscape is greater than a threshold value determined by the properties of the species. Therefore, metapopulation capacity can conveniently be used to rank different landscapes in terms of their capacity to support viable metapopulations (Hanski & Ovaskainen 2000). As a first approximation, λ_M measures the amount of suitable habitat, but additionally λ_M takes into account the actual spatial configuration of the fragmented landscape ('the actual network'). Calculating how the metapopulation capacity is changed by removing habitat fragments from the existing network (*i.e.* habitat loss), by adding new ones into specific locations (*i.e.* reserve design, ecological network design), or by changing their areas could be one of the possible applications (Hanski & Ovaskainen 2000). Our approach (the development of an ecological optimal scenario) is based on that recent concept of Hanski and Ovaskainen (2000) (for more details, see *material and methods*).

The ecological objectives of the current case study are as follows:

- (1) Develop a more widely applicable approach for implementing an ecological network (EN) based on scientific literature background and expert knowledge;
- (2) Apply this methodology in the study area to propose an 'ecological optimal' scenario delineating an ecological network in the study area that should ensure the long term maintenance of certain priority biotopes in the landscape and the viability of a number of target species. The scenario will define land use throughout the study area so that it is compatible with the maintenance (at least temporarily) of a higher level of biodiversity;
- (3) Make a list of restoration and management measures that are necessary to build the ecological network;
- (4) Generalize the experience gained through the execution of this project, to allow the implementation of ecological networks in other areas or in other contexts.

The biological purpose of an ecological network and how it is intended to provide benefits to flora and fauna is an important first step and an essential basis for evaluating design and management of the landscape (Bolck *et al.* 2004). The feasibility of an ecological network however, depends on more than just ecological aspects. Indeed, establishing an ecological network means allocating the given resource - the territory - to certain, sustainable, land uses in order to preserve biodiversity. In this process, numerous actors will be directly involved, e.g. landowners, local authorities, nature associations and other stakeholders. The feasibility of restoring an ecological network will strongly depend upon the participation of these actors, one way or another. Moreover, apart from the "technical" nature conservation context, the legal, economic and social context of a restoration should be taken into account as well. Until now, ecological networks, legal land use planning and socio-economic assessment of biodiversity have always been dealt with autonomously. The objectives of this ECONET-research project are to propose answers to these issues.

The legal and socio-economic evaluations of the feasibility of the optimal scenario proposed by the team of ecological experts, are given in the following chapters. Here, an overview of the subsequent steps to generate an optimal ecological scenario in our study area (the Dyle valley) is given, as well as a proposal and full description of a more widely applicable methodology for the implementation of an ecological network.

2.2.2. Material and methods

2.2.2.1. Selection of a study area: the Dyle valley, a cross language border ecosystem

A series of unambiguous criteria was put forward to choose a study area: (1) the area should be of importance to the conservation of biodiversity and, in particular, should encompass habitats and/or species that are of conservational interest for both the Walloon and Flemish region in the context of NATURA 2000; (2) previous knowledge of the ecological structure, the habitats and the population biology of at least some species of interest would be an asset; (3) the selected area should present potential important interactions with regard to economic, social and legal issues; (4) the study area should allow for a comparison of opportunities and problems related to the implementation of an ecological network between both regions. Based on these criteria, the Dyle valley between Leuven (Flanders) and Wavre (Wallonia) was chosen (Fig. 2.3).

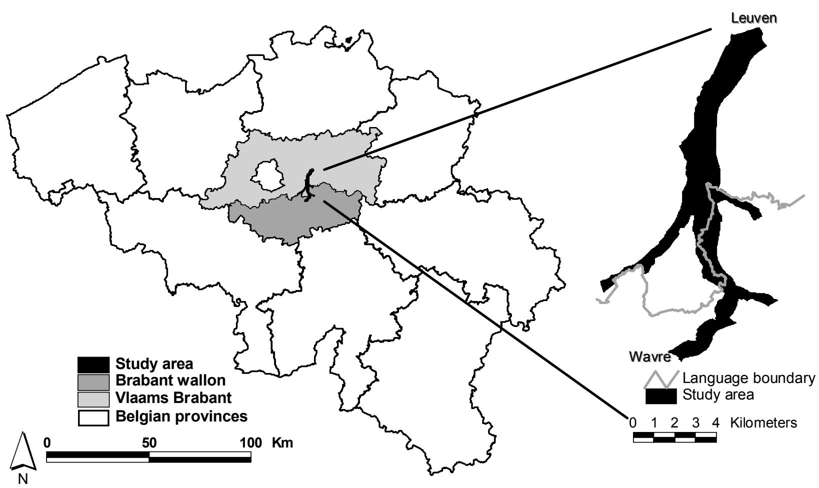

Fig. 2.3: Location of the selected study area for the case study

The whole valley-system can be divided into three different main geomorphological entities: the valley floor, the slopes and the plateau. Each of these entities has its own abiotic characteristics, biotopes and associated species. In the scope of this project, a limitation to one geomorphological unit was necessary. Therefore, the study area was restricted to the valley floor. In practice, boundaries were set along roads (western limit) and a railway (eastern limit), all of these situated at the foot of the slopes. Both roads and railways can be considered as serious obstacles for the dispersal of many plant and animal species. In the north (Leuven) and south (Wavre) the study area is bordered by urban zones representing a logical limit to our study area. Because the Laan valley and downstream sections of the

Nethen and Train valley belong to the same ecosystem as the Dyle valley, they were also taken into account.

Although the Dyle river itself shows no differences between Flanders and Wallonia, the land use in the valley does. The Flemish part is characterized by a mosaic of semi-natural habitats: relatively dry, extensively managed grasslands, wet grasslands, wet tall herb vegetations, marshes, alluvial forests and swamp forests. Here, the presence of agricultural activities is rather limited. On the other hand, intensively managed arable lands and grasslands on large fields dominate the Walloon part of the valley (Fig. 2.4). It is obvious that the current setting and the differences that exist between the two regions will influence the outcome of the study, not only on the ecological level but also on the socio-economical level (for more details, see socio-economical study; chapters 4 and 5). In addition, the juridical background also largely differs between Flanders and Wallonia (for more details, see juridical study; chapter 2).

As it is not essential within the scope of this report to go into detail on every aspect of the biotic and abiotic setting of the study area, only vegetation, soil and hydrology are (shortly) discussed below because of their importance for the elaboration of an ecological optimal scenario in a groundwater fed valley system (De Wilde *et al*. 2001). Other aspects as geology, land use history and geomorphological processes are fully described in literature (see De Becker & De Smedt 1994, Anonymous 2000, Verstuyft 2000, De Wilde *et al*. 2001, Circaète 1996, Deyries 1998, Duchatelet 2001, Docq 2002, Legast 2004).

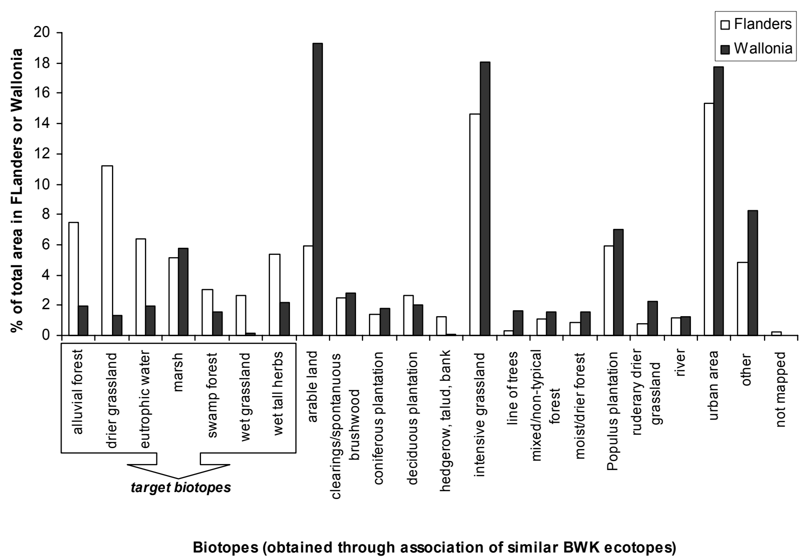

Fig. 2.4: Comparison of actual land use and vegetation types (*i.e.* associations of BWK-ecotopes into biotopes) between Flanders (white) and Wallonia (black), expressed as % of total surface of the study area in Flanders or Wallonia respectively.

2.2.2.2. A metapopulation capacity based approach

In order to prepare different scenarios for the implementation of an optimal ecological network, data on the actual ecological status (distribution of vegetation types in the study area) was needed. Due to time restrictions and limited resources, existing information had to be used as much as possible. In Flanders, the BWK ('*Biologische Waarderingskaart';* Guelinckx *et al.* 2002) an existing survey map depicting a combination of a vegetation and land use and covering the whole surface of this part of the study area could be used. Hence, in this area fieldwork could be limited to random verification of the accuracy of these data. Additional surveys allowed for extrapolation to the Walloon part of the study area. This survey was executed during late spring of 2003 (May 15th - June 15th).

a) Selection of target biotopes

For clarity reasons, similar BWK-ecotopes were clustered into larger units (further referred to as 'biotopes'; see App. 2.1a and 2.1b). In this way, the number of different ecotopes (more than 100) could be reduced to 20, a generalisation procedure essential for all further modelling efforts. Moreover, not all of these biotopes were typical for a valley system, and thus for the study area, or valuable enough to make a network for. Hence, a selection of a limited number of target biotopes was necessary. The combination of the BWK-ecotopes into larger biotopes was based on expert judgement and floristic similarity evaluated from a qualitative point of view (phytosociological method). Then, the selection of target biotopes was based on two criteria: they had to be (1) biologically valuable and therefore worthwhile to make a network for and (2) a typical biotope of a valley system. By reviewing literature and through the assistance of several experts seven target biotopes were chosen (Table 2.2; for an overview of their actual location see Fig. 2.10). 'Urban patches' on the BWK were seen as unchangeable entities and by consequence omitted from all further analyses.

Table 2.2: The seven target biotopes with indication of the BWK-ecotopes they enclose

Target Biotope	BWK-ecotope-codes
Eutrophic water	ae, ae-
Marsh	mr, mr-, mrb, mc, mc-
Wet tall herbs	hf, hfc, hfc-
Wet grassland	hc, hc+, hc-, hj, hj-
Drier grassland	hp+, hu, hu-
Alluvial forest	va, va-, vn
Swamp forest	vm, vm-, sf, sf-

b) Assessment of current conservation status of the target biotopes: quality scores

Restoration should only be considered if necessary. So an important step was to analyze the **current state of target biotopes** to evaluate the **current state of the ecological network** in the study area (so-called 'occupied patches'; Fig. 2.5). To do so, the help of Walloon experts (Bruno Nef and Marc Walravens, members of 'Amis du Parc Naturel de la Dyle') as well as their Flemish counterparts (Bart Vercoutere, member of 'Vrienden Heverleebos & Meerdaalwoud') was invoked. Based on their experience, they were able to provide a score (= **quality score, Q**) for each actual target biotope patch to estimate its state of conservation. Patches were scored from 1 up to 3; patches having quality score 1 being poorly conserved and 3 given to patches with a high conservation status. A quality score which was then rescaled between 0 and 1 for further use within the model.

c) Construction of potentiality maps

Throughout the study area there are other, currently 'unoccupied', patches that can potentially be converted into one of these target biotopes. To obtain maps with the potential distribution of all target biotopes in the Dyle valley, all factors influencing their occurrence needed to be investigated. The occurrence of different vegetation types in a valley system and the spatial distribution thereof mainly relies on **soil characteristics**, **groundwater regime** and **management type** (De Wilde *et al.* 2001). 'Management' is inherently linked to human activities and will be dealt with when restoration possibilities are discussed. Hence, the natural potential for a patch to sustain a target biotope will mainly depend on the soil characteristics and groundwater level on that particular site. Because of time restrictions, both were analyzed in a more pragmatic way (Fig. 2.5). Several ecohydrological studies have already been carried out, most of them in the Flemish part of the study area (e.g. Butaye & Deckers 1994, De Becker *et al.* 1999, Anonymous 2000, Huybrechts *et al.* 2000, De Wilde *et al.* 2001, Huybrechts *et al.* 2002). The majority of the alluvial soils in the valley have a loamy texture (‘A’ on soil map according to the Belgian soil classification system; App. 2.2). Locally, in some depressions on the valley floor, clay soils (‘E’) can be found. At the foot of the slopes, colluvial sandy loam depositions occur (‘L’). Typically for alluvial soils is their lack of a clear profile development (‘p’). Because of the low topographical location of the alluvial plane, the permanent presence of groundwater seepage in the floodplain depressions and the micro-topography with elevated natural levees and alluvial floodplain depressions, soil moisture within the valley shows large differences, varying from high to very high moisture content (drainage classes ‘e’, ‘f’, and ‘g’) till moderately drained situations (‘D’, ‘I’).

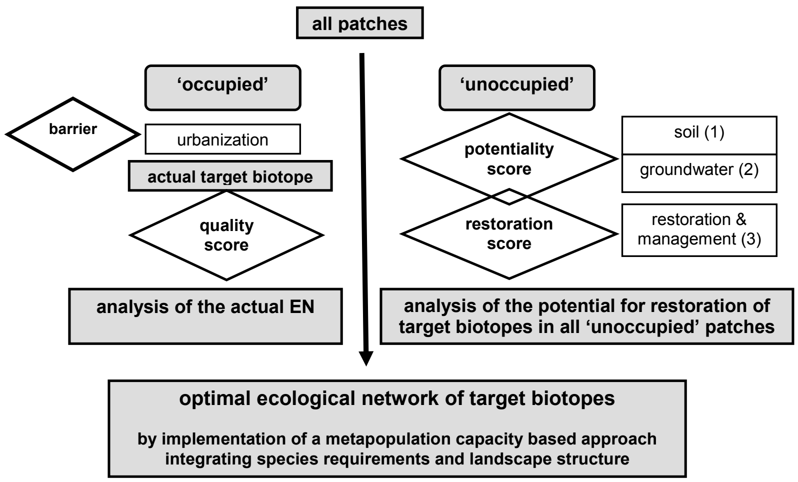

Fig. 2.5: Schematic overview of the elaboration of an ecological network and the approach we adopted in this case study

As mentioned earlier, one of the main factors that define the vegetation to be found in a valley system is groundwater level and quality. Through the study of De Wilde *et al.* (2001) in the Dyle valley and unpublished data for the Laan valley (Haskoning 2005), data on groundwater level were available. Indeed, a network of 247 piezometers was laid out in the Flemish part of the study area (Fig. 2.6a). A similar piezometer network did not exist in the Walloon part of the study area, so we had to work around this by linking groundwater depth to soil characteristics since these were mapped for the whole study area (App. 2.2). Hence, the relationship between soil type (in particular, texture and drainage class according to the Belgian soil classification system) and average groundwater levels measured in the piezometers on that type of soil was investigated. The majority of the piezomters is located on A-type (*i.e.* loamy) soils (Fig. 2.6b). To relate soil characteristics to groundwater depth, two assumptions were made: (1) as L, S, E, V –soils are very small in surface in the study area and/or often situated on valley slopes, it was decided to use the same values for groundwater depth as the ones corresponding to the different drainage classes on loamy soils; (2) The measured groundwater levels in the O-type soils (anthropogenic soils) cannot be used, because no clear link between both factors can be expected in this case. For patches on those soils, groundwater depths corresponding to soil types located around these patches were used. Median values of mean groundwater depths measured in piezometers on loamy soils with different drainage classes were calculated (Fig. 2.6c). By extrapolation through the soil map, we now had groundwater levels for the whole study area.

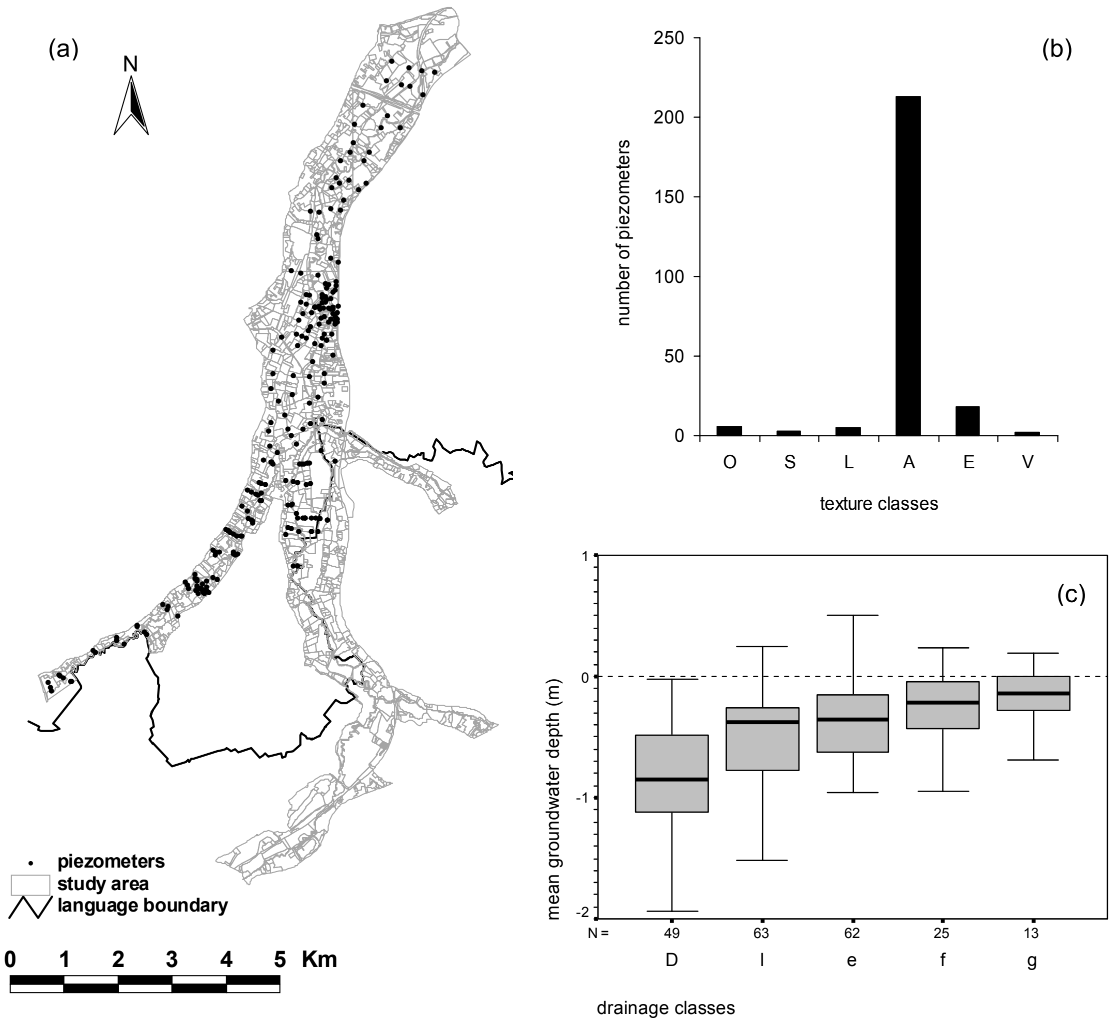

Fig. 2.6: (a) Depiction of piezometer locations in the Dyle valley, **(b)** distribution of piezometers (n = 247) over different soil texture classes according to the Belgian soil classification system (O: anthropogenic soils, S: sand, L: sandy loam, A: loam, E: clay, V: peat) and **(c)** variation in mean groundwater depth for piezometers located on loamy soils with different drainage classes according to the Belgian soil classification system (D-I-e-f-g: drainage classes, ranging from rather dry till very wet)

Next, the relationship between vegetation type and groundwater level was explored to obtain potential vegetation distribution maps. Huybrechts *et al.* (2000) performed a detailed analysis of this type in the nature reserve 'De Doode Bemde' which is located in the Flemish part of the Dyle valley. Based on their observations on the prevalence of different biotopes at sites having different levels of mean groundwater depth they produced 'response curves' (Fig. 2.7). The target biotopes, as they were selected in the ECONET-project, could largely be related to those studied by Huybrechts *et al.* (2000): swamp forest = T2; marsh = T3; drier grassland = T4; wet tall herbs = T5; wet grassland = T6 and alluvial forest = mean response of T4 & T5 (as alluvial forest can be seen as later succession stage of T4 and T5). Given the common geographical and abiotic setting, these response curves could be used to predict the potential vegetation distribution from mean groundwater level in the entire Dyle valley (P. De Becker pers. comm.; Fig. 2.7). As a target biotope 'eutrophic water' was omitted from all analyses because it can be assumed that throughout the valley floor obtaining this biotope would be simply a matter of digging.

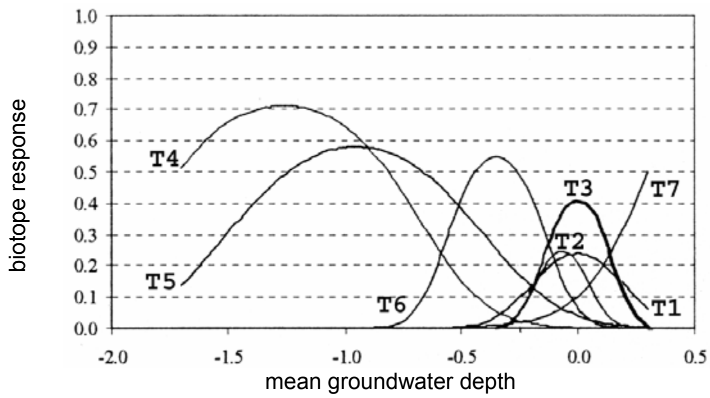

Fig. 2.7: Response curves for different biotopes in the 'Doode Bemde' nature reserve (T1: reed-land, T2: swamp forest, T3: large sedges with reed, T4: *Arrhenatheretum elatioris*, T5: *Filipendulion*, T6: *Calthion palustris*, T7: large sedges; Huybrechts *et al.* 2000).

By combining the relationship between mean groundwater depth and soil characteristics and the response curves depicting the link between mean groundwater depth and vegetation type, it was possible to relate vegetation type to local soil characteristics. Hence, it was possible to determine for each soil type the chance to find one or more of the target biotopes associated with it. Because the entire study area was mapped according to the Belgian soil classification system (App. 2.2), it was possible, by extrapolation, to attribute to every 'unoccupied' patch a '**potentiality score**' (*i.e.* the response score of Fig. 2.7, rescaled between 0 and 1) representing the natural potential of the site to restore each of the six target biotopes (ommiting 'eutrophic water').

d) Assessing feasibility of restoration

Obviously, as all target biotopes are typical for a valley-ecosystem, some conflicts could be expected for restoration of different target biotopes (having similar potentiality scores) at the same location. In that case, the target biotope that will preferentially be restored depends on extra criteria such as actual biotope, restorability, etc. So, again based on expert knowledge, for all possible transitions of actual non-target biotopes into target biotopes, a '**restoration score**' (*i.e.* a value indicating the feasibility of restoration) was given:

- **0 = no restoration**: actual biotope too valuable (e.g. mesophilic forest, valuable small landscape elements, actual riverbed,…).
- **0.25 = almost certainly impossible**: in spite of active restoration measures, the target habitat cannot be restored after 25 (open habitats) or 100 years (forests).
- **0.50 = difficult**: initial habitat restoration measures involve drastic operations (e.g. topsoil removal, modification of the hydrological properties of the area, clearcutting and tree removal on swampy soils,...), associated with very high costs. Afterwards, periodical management is needed for open habitats, without any guarantee that the target habitat will be restored after 25 years. For forest habitats, a lot of species typical for ancient forests will still be missing after 100 years.
- **0.75 = feasible**: initial restoration effort is relatively limited (e.g. clearcutting and/or removal of exotic species). Next, periodical management almost certainly leads to recovery of the target habitat after 10 to 25 years.
- **1.00 = easy**: after removal of the exotic species, natural succession processes lead to recovery of the target biotope after 50 years (for forest habitats) or extensification of agricultural practices are sufficient to restore the target biotope (for grassland habitats).

e) Construction of a model incorporating the metapopulation capacity approach

Finally, based on the work of Hanski & Ovaskainen (2000, 2002), the **model** was built in Matlab 6.5 (Anonymous 2002b), combining data on the actual ecological network of target biotopes, the potentiality maps, restoration feasibility, specific species requirements and landscape structure into one analysis. It was designed to calculate the '(actual) **metapopulation capacity**' of each target biotope (**λ_M**) based on conservation status (quality score, Q), specific species requirements (more specifically, minimum area requirement, β and dispersal distance, α) and several indices corresponding landscape structure (x-y-coordinates of patch centroids, patch area (A) and distances between patches of the same biotope, d_{ij}). This metapopulation capacity index is a measure for metapopulation persistence and hence indicates the current capability of the network to sustain metapopulations of species in a particular target biotope.

<u>Metapopulation capacity - Hanski & Ovaskainen 2000</u>

Metapopulation theory describes the dynamics of populations in fragmented landscapes. Long-term persistence of populations depends on landscape structure and on the ratio between colonization and extinction rates and: **P* = 1-(E/C) x 1/λm**

The most important **metapopulation (species-specific) processes**, **colonization (C)** and **extinction (E)**, can be related to the most important **structural features of fragmented landscapes patch areas (A)** and **spatial locations** by some assumptions: extinction rate $E_i = e/A_i$ (how smaller the patch, how higher the chance of extinction) and colonization rate $C_i = c\sum_{j\neq i}exp(-\alpha d_{ij})A_j p_j(t)$ (immigration to a patch is expected to increase with the number of neighbouring populations, with their sizes as reflected by the respective patch area, with their decreasing distances to the focal patch and increasing incidences of occupancy), where A_i (A_j) is the area of patch *i* (*j*), d_{ij} is the distance between patches *i* and *j*, $1/\alpha$ is the average migration distance, and *e* en *c* are species-specific constants (in our analyses both set on '1') (Hanski 1994, 1999, Hanski & Ovaskainen 2000). Instead of using patch area as a surrogate of expected population size, patch areas may be corrected for spatially varying habitat quality (Hanski & Ovaskainen 2000). In our case, a correction A*=ln(A.β.Q + 1) was used, with A = patch area, β = minimum patch area to sustain a viable local population and Q = quality score (Fig. 2.8). By applying this correction, the area of the patches is rescaled according to population size and probability of extinction. The latter can be represented by the minimum area requirement β (App. 2.3).

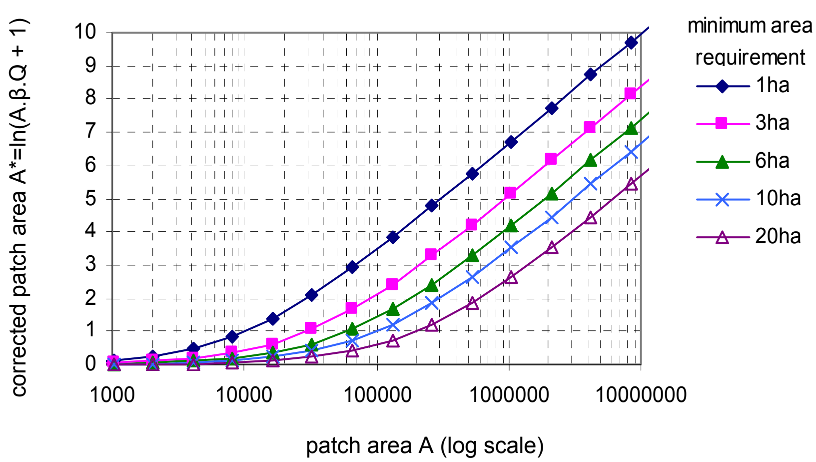

Fig. 2.8: Correction of patch area A for species having different minimum area requirements to account for population size and associated extinction probability in the metapopulation capacity approach

The **metapopulation capacity λ_M** of a fragmented landscape is a measure that captures the impact of landscape structure (the amount of habitat and its spatial configuration) on metapopulation persistence. In other words, it is an indication for the state of the network of a particular target biotope for a (meta)population of a certain species. Mathematically, the **metapopulation capacity λ_M** is the leading **eigenvalue of an appropriate 'landscape' matrix** (Hanski & Ovaskainen 2000). This 'landscape' matrix M is a square symmetrical matrix which consists of the elements $m_{ij}= exp(-\alpha d_{ij})A^*_jA^*_i$ for $j\neq i$ an $m_{ii}= 0$. These elements must be seen as a reproduction of the landscape in terms of colonization and extinction. Each element m_{ij} represents the contribution of patch *j* to the connectivity of patch *i* in ratio to the extinction in patch *i* (App. 2.3).

2.2.2.3. A pragmatic biotope approach versus a species-specific approach

One of the main goals of the project was the evaluation of the proposed optimal ecological scenario by the social, economic and legal partners. Because those partners needed sufficient time to evaluate the proposed ecological network, it was necessary to determine and submit an optimal ecological network and the restoration or management measures necessary to achieve this as early as possible. Therefore, the decision was made to adopt **a two-level approach.** First, as explained above, a more pragmatic approach was followed encompassing the proposal of **an EN based on target biotopes** and the list of restoration and management measures, both to be evaluated by the other partners. Next, **a more detailed ecological study** was carried out in which specific requirements of a set of selected target species were taken into account. Next, the first steps leading to this species-specific approach are shortly explained.

a) Target biotope approach

As the model needed the input of species-specific parameters, a procedure was to be devised to determine a set of parameters for a fictive species, the same for all 6 target biotopes. Therefore, a sensitivity analysis was carried out. Nine different scenarios were run: all possible combinations of three different dispersal distances ($1/\alpha$ = 100 m, 1 000 m and 10 000 m) and three minimum area requirements (β^{-1} = 1000 m², 10 000 m² and 100 000 m²). Based on this analysis, a scenario for a fictive species with a dispersal distance of 1000 m and an area requirement of 10 000 m² was chosen as this provided us with the highest sensitivity to demonstrate the possibilities of the model. It was mainly dispersal distance that seemed to be important: 100 m was too short; almost no patches could be reached from the existing patches with target biotopes. The current model calculates inter patch distances based on the x,y-coordinates of the centroids, which are almost always larger than 100 m. On the other hand, dispersal distances of 10 000 m were too long because the length of the study area is only 17 km (hence, distances between patches seldom exceed 10 km).

Next, by adding ‘unoccupied’ patches one by one to the existing network of a target biotope and subsequently recalculating metapopulation capacity index, differences between these indices and the one of the original network could be ranked to identify the most important ‘unoccupied’ patches for metapopulation persistence of a certain species in a certain target biotope. These calculations were carried out for all target biotopes separately. These metapopulation capacity increments were then rescaled between 0 and 1 and multiplied by the potentiality and restoration scores of that particular patch. For each patch, these values were then compared between target biotopes. To be able to assign just one target biotope to each ‘unoccupied’ patch, priority was given to the target biotope having the highest value for that patch. However, if the differences between the largest and second or third largest values were very small (less than 0.001) and if these second or third largest values corresponded to a target biotope that still covered only a small area in the Dyle valley (e.g. swamp forest and wet grassland), the patch was assigned to these target biotopes. Ultimately, this procedure resulted in an optimal ecological scenario depicting the ideal distribution of target biotopes in the Dyle valley.

To facilitate the evaluation of the proposed optimal ecological scenario by the other project teams, additional information was provided: **a matrix of restoration and management measures**, a procedure for **priority ranking for restoration** and, finally, **a delineation of core areas, development zones and corridors** in the study area.

First, a list of all necessary measures to restore and preserve each target biotope was established. Based on literature review and expert knowledge we constructed a matrix containing both the necessary restoration measures and subsequent management needed for conservation of the newly obtained target biotope for each possible restoration of an actual non-target biotope into a target biotope. In practice, these measures may differ between individual patches, even if the same target biotope is to be restored on patches currently holding similar non-target biotopes. However, only general measures were presented in this matrix as it was meant to function as a tool for other partners to evaluate legal and socio-economic feasibility of restoration activities (App. 2.4).

Next, there was a need to elucidate the importance of restoration of particular target biotopes on specific patches for researches lacking an ecological background. Therefore, a procedure was devised for priority ranking (in terms of work, budget, etc.) of the restoration of all 'unoccupied' in the study area. Restoration priority was based on the scores obtained through our metapopulation capacity approach (difference between metapopulation capacity before and after addition of the patch to the network; see earlier) multiplied by the 'potentiality score' and the 'restoration score'. This way, a value is obtained indicating contribution of each 'new' patch to the ecological network as it is restored. Ranking was executed for each target biotope separately and then for all 'unoccupied' patches over all target biotopes. Needless to say that conservation of the actual network of target biotopes received highest priority.

Up till now, the methodology did not employ the terminology "core areas", "development zones" and "corridors", usually mentioned in studies on EN frameworks (e.g. Delescaille 1993, De Blust *et al.* 1995, van Zadelhof & Lammers 1995, Mellin 1997), as clear definitions of these terms are yet to be devised. Nevertheless, the delineation of core areas, development zones and corridors (for each target biotope) seemed very important for the other partners because they depend on it to translate our priority ranking for restoration into their discipline-specific language and to relate ecological conservation priorities to specific legal tools. So, in a final attempt to enhance the interpretability of the optimal ecological scenario, the EN in the Dyle valley study area was divided into different zones representing core areas, development zones and corridors (which is, in fact, a kind of 'suboptimal' ecological scenario). To do so, the meaning of the terms had to be defined (based on the definitions of Delescaille 1993, De Blust *et al.* 1995, van Zadelhof & Lammers 1995, Mellin 1997) and then the different zones were delineated based on the outcome of our metapopulation capacity approach and restoration priority ranking:

- For each target biotope, **core areas** are the zones where the model projects rather large, continuous surfaces of same biotope with a combination of a high conservation status for 'occupied' patches and high priority scores for the patches to be restored. These zones are essential for the functionality of EN and therefore all legal tools should be applied and high (financial) effort should go into conservation of existing target biotopes and restoration of patches clustered around them.
- **Development zones** enclose all the existing patches of the target biotopes that are not in the core areas in the immediate vicinity. These patches should be maintained at least in their current state of conservation and improvement of the ecological value would be recommended. These zones contain all patches that are proposed for restoration into target habitat but that are outside of the core areas.
- **Corridors**, finally, are in this study defined as a set of patches that form stepping stones between two core areas. Obviously, in some cases they can be the same patches than in the development zones, but to make the corridor functional, it may be imposed that patches with a high priority score for restoration will effectively be restored. It is important to mention that other landscape structures that play an important role in connecting certain habitats can also act as corridors (e.g. the Dyle river can operate as a corridor for species of alluvial forests specifically adapted to hydrochory).

b) Species-specific approach

Landscape features do not directly act on biodiversity, but on performance of individual species (Verboom & Pouwels 2004). Therefore, in the context of ecological networks scaling down from landscape level to species level is essential. Because species differ in the way they use a fragmented landscape, solutions for improved connectivity may differ with the landscape settings and species (Vos *et al.* 2001). Indeed, depending on the spatial scale (continental, regional or local) as well as on the species of interest the requirements for a functional ecological network can diverge widely. So, in reality it may be expected that in a given landscape each species will have its own optimal ecological network. Metapopulation capacity approaches allow for considering species with long as well as short dispersal ranges and landscapes with spatial correlation at different scales (Ovaskainen *et al.* 2002). Indeed, as seen before, the metapopulation capacity model is also based on species-specific characteristics. However, as so far, we have worked with only one fixed set of species parameters (for a fictive species) for all target biotopes, which is, of course, an oversimplification of reality.

Table 2.3: Selected set of target species and their dispersal distances (Disp.) and minimum area requirements (Area) to be entered as parameters in the metapopulation capacity model (* Devolder *et al.* 2000, Daemen 2004; ** Broekmeyer & Steingröver 2001; *** Pouwels *et al.* 2002; **** K. Moreau & M. Hens, pers. comm.).

Alluvial forest	Disp.	Area	Wet grassland	Disp.	Area	Wet tall herbs	Disp.	Area
Anemone nemorosa	100m *	0.1ha *	*Rhinanthus angustifolius*	100m *	0.1ha *	*Achillea ptarmica*	100m *	0.1ha *
Adoxa moschatellina	100m *	0.1ha *	*Caltha palustris*	100m *	0.1ha *	*Filipendula ulmaria*	100m *	0.1ha *
Primula elatior	100m *	0.1ha *	*Lychnis flos-cuculi*	100m *	0.1ha *	*Cirsium oleraceum*	100m *	0.1ha *
Apatura iris	2000m **	50ha ***						
Limenitis camilla	2000m **	50ha ***	*Anthacharis cardamines*	1000m ***	50ha ***	*Aphantopus hyperanthus*	1000m ***	1ha ***
			Anas clypeata	25000m ***	9ha ****	*Locustella naevia*	20000m **	3ha ****
Dendrocopus minor	10000m ***	10ha ****	*Anas querquedula*	50000m ***	5ha ****	*Luscinia svecica*	10000m ***	4ha ****

By combining species with different functions in the ecosystem and operating at different scales, one gets a more complete view of the biodiversity than when investigating only one or two species. This concept is known as '*the multi-species approach'* (Lambeck 1997, Van Dyck *et al.* 1999, Van Dyck *et al.* 2001, Verboom & Pouwels 2004). This concept makes use of a relevant selection of ecologically well-known species drawn from different taxonomic groups. Hence, representative species of each relevant taxonomic group (plants, dragonflies, butterflies, grasshoppers & crickets, beetles, fishes, amphibians, reptiles, birds, mammals, etc., all having different home range sizes and dispersal distances) occurring in different habitat types have to be chosen. To support species selection in our case study, we again consulted field experts (Table 2.3). Selection was limited to three taxonomic groups (plants, butterflies and birds) and three of our target biotopes (alluvial forests, wet grasslands and wet tall herbs). Once the target species were chosen, a literature review was carried out to find the species-specific dispersal distances and minimum area requirements which could then be entered in the metapopulation capacity model (Table 2.3). Further analyses were similar to those performed for the 'target biotope'-approach. In contrast with the latter, all restoration possibilities resulting from running the metapopulation capacity model and all potentiality scores are presented for each patch. Steps involving restoration scores were omitted as it was not the objective to offer only one restoration option for each patch.

2.2.3. Results

2.2.3.1. Target biotope approach

a) Current conservation status of the target biotopes: quality scores

Actually, of the already existing set of target biotope patches (772.6 ha), 25.1% (194.1 ha) of the area is covered by drier grasslands. Together with wet tall herb patches (13.6%), they mainly occur in the central part of the study area (Fig. 2.9c and 2.10). Alluvial forests (16.6%) and swamp forests (7.5%) are mainly found in the north, the eastern part of the 'Doode Bemde' nature reserve and the upstream section of the Laan valley. In the current context, existing target biotope patches with marshes, wet grasslands or alluvial forests have a relatively better conservation status (Fig. 2.9a and 2.9b). Conservation status of drier grasslands or patches with eutrophic water on the other hand, is generally low, especially if their percentage of the number of target biotope patches is considered (Fig. 2.9a).

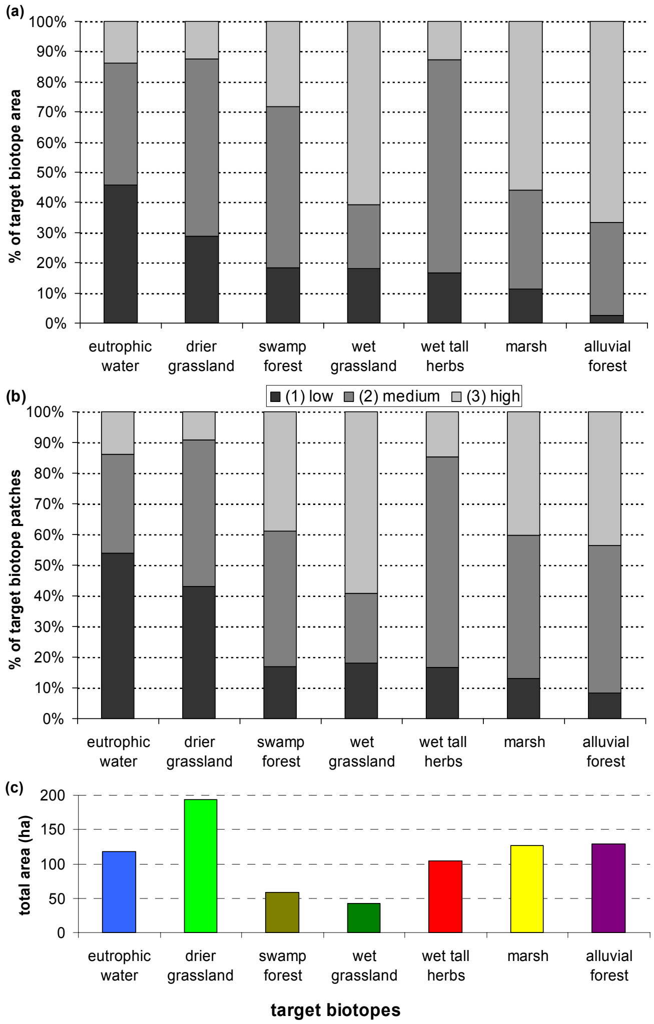

Fig. 2.9: Overview of conservation status (quality scores, Q) of patches currently occupied by one of the seven target biotopes: **(a)** expressed as % of actual target biotope area, **(b)** expressed as % of actual target biotope patches. **(c)** Total area (ha) of each target biotope in the study area.

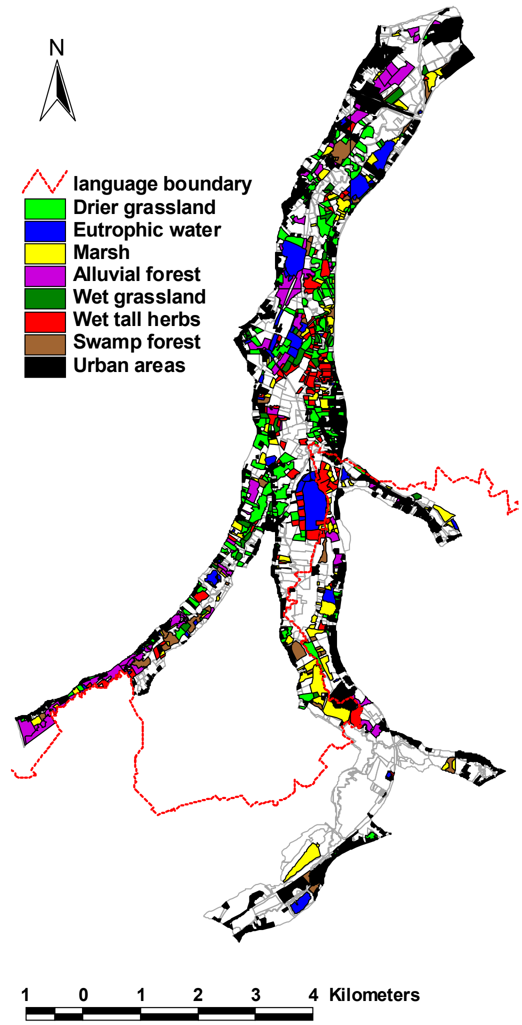

Fig. 2.10: Map of the study area showing the actual location of selected target biotopes

b) Potentiality maps

Throughout the Dyle valley, the natural potential to restore marshes and swamp forests is generally very low (*i.e.* only possible on the wettest sites), whereas the potential to sustain drier or wet grasslands is either very low or high (Fig. 2.11). Potentiality scores for the other target biotopes are somewhat more evenly distributed as their restoration is possible both to drier and wetter sites.

c) Feasibility of restoration

In general, for most patches restoration into one of the six target biotopes action is deemed to be ‘feasible’ to ‘difficult’ (Fig. 2.12). However, apart from restoration into drier grasslands (which mainly consists of reducing the intensity of current agricultural use), reinstatement of the target biotope is only very rarely considered to be ‘easy’. For 161 out of 1105 currently unoccupied patches (15%) it is advised that no restoration actions are undertaken since the actual biotope is too valuable.

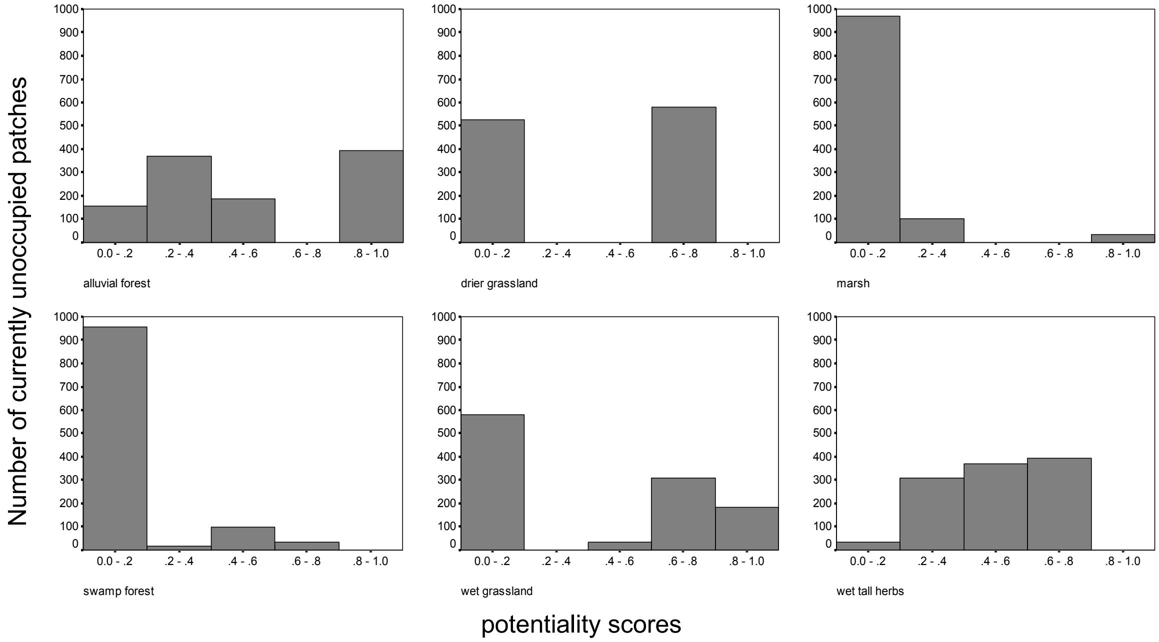

Fig. 2.11: Overview of the scores related to the natural potential for restoration of all actual non-target biotope patches (n = 1105) into each of the six target biotopes.

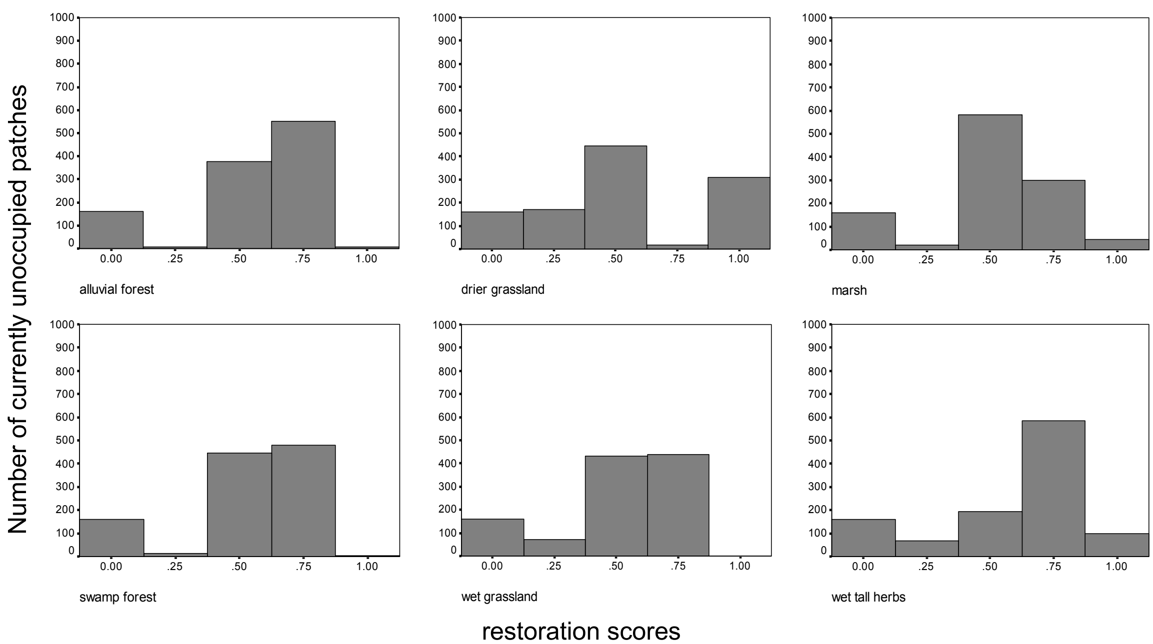

Fig. 2.12: Overview of the scores related to feasibility of restoration of all actual non-target biotope patches (n = 1105) into each of the six target biotopes (0.00 = no restoration, actual target biotope too valuable; 0.25 = almost certainly impossible; 0.50 = difficult; 0.75 = feasible; 1.00 = easy).

d) The optimal ecological network for the Dyle valley study area

The final outcome of the whole procedure is the optimal spatial distribution of all six target biotopes in an ecological network as it is depicted in Fig. 2.13. Based all criteria explained earlier, to each 'unoccupied' patch one target biotope can be attributed (Fig. 2.13a). In combination with the existing set of target biotopes patches in the study area (Fig. 2.10), restoration of these patches into the desired target biotope would result in the optimal ecological network for the Dyle valley at target biotope level (Fig. 2.13b). In general, it is clear that patches preferentially to be restored to a certain target biotope are more or less spatially clustered around already existing target biotope patches. Furthermore, implementation of this scenario would imply large forested areas in the northern and south-eastern part of the study area, whereas marshes would dominate the southernmost areas. Wet and drier grasslands would primarily be restored in the downstream section of the Laan valley and the central area of the Dyle valley.

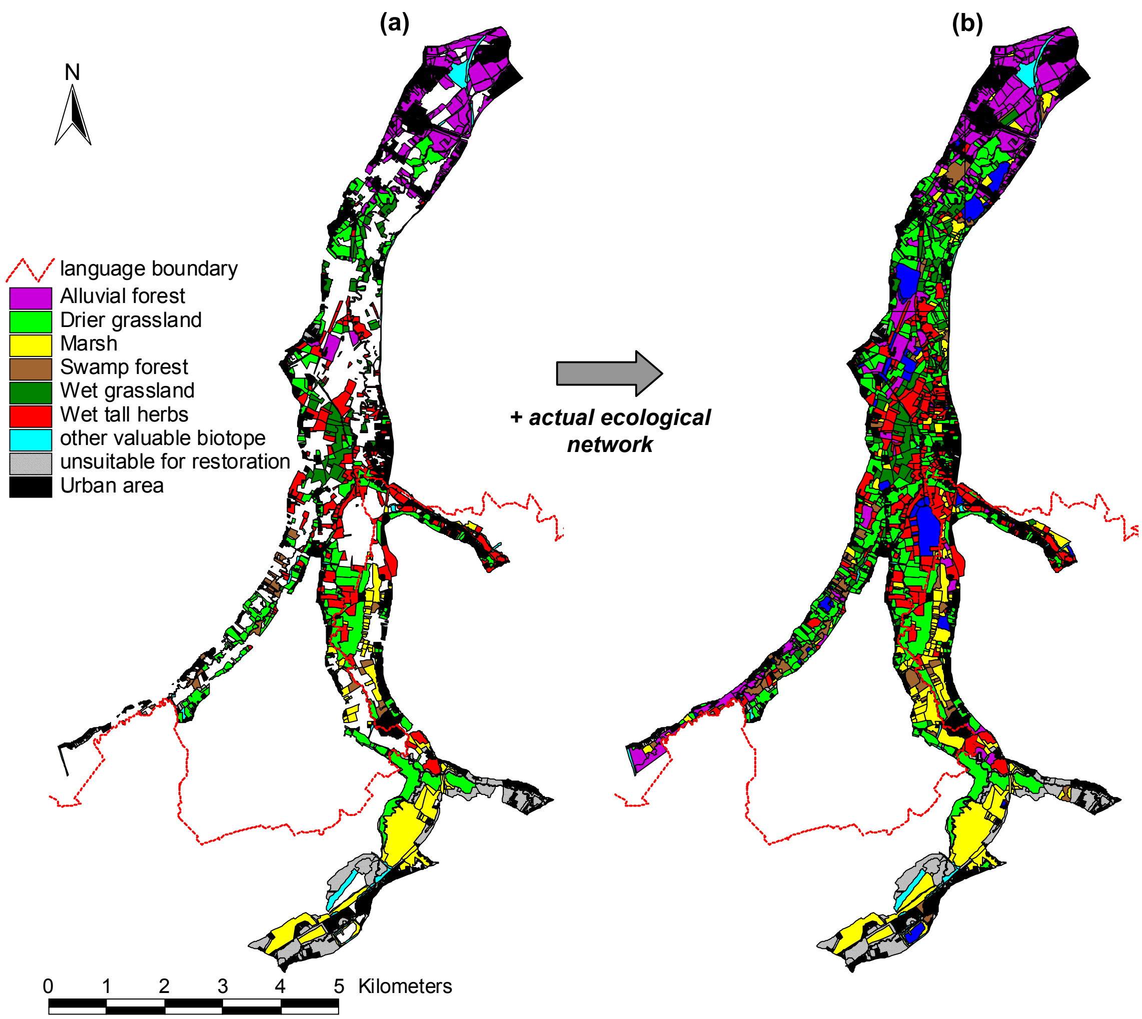

Fig. 2.13: Map of the study area showing the spatial distribution of all six target biotopes according to the optimal ecological scenario: **(a)** attribution of all currently 'unoccupied' patches to one of the six target biotopes **(b)** combination of actual ecological network (see Fig. 2.10) and (a) into the optimal ecological network for the Dyle valley (for a larger version of the optimal ecological network map, see App. 2.5).

e) Priority ranking

The procedure to delineate core areas, development zones and corridors, based on priority ranking for restoration and current conservation status, is illustrated by Fig. 2.14. This results in large core areas in the northernmost area of the study area, around the 'Doode Bemde' nature reserve and at the merging of the Dyle and Laan river. Development zones are mainly found in the Laan valley and the Walloon part of the study area.

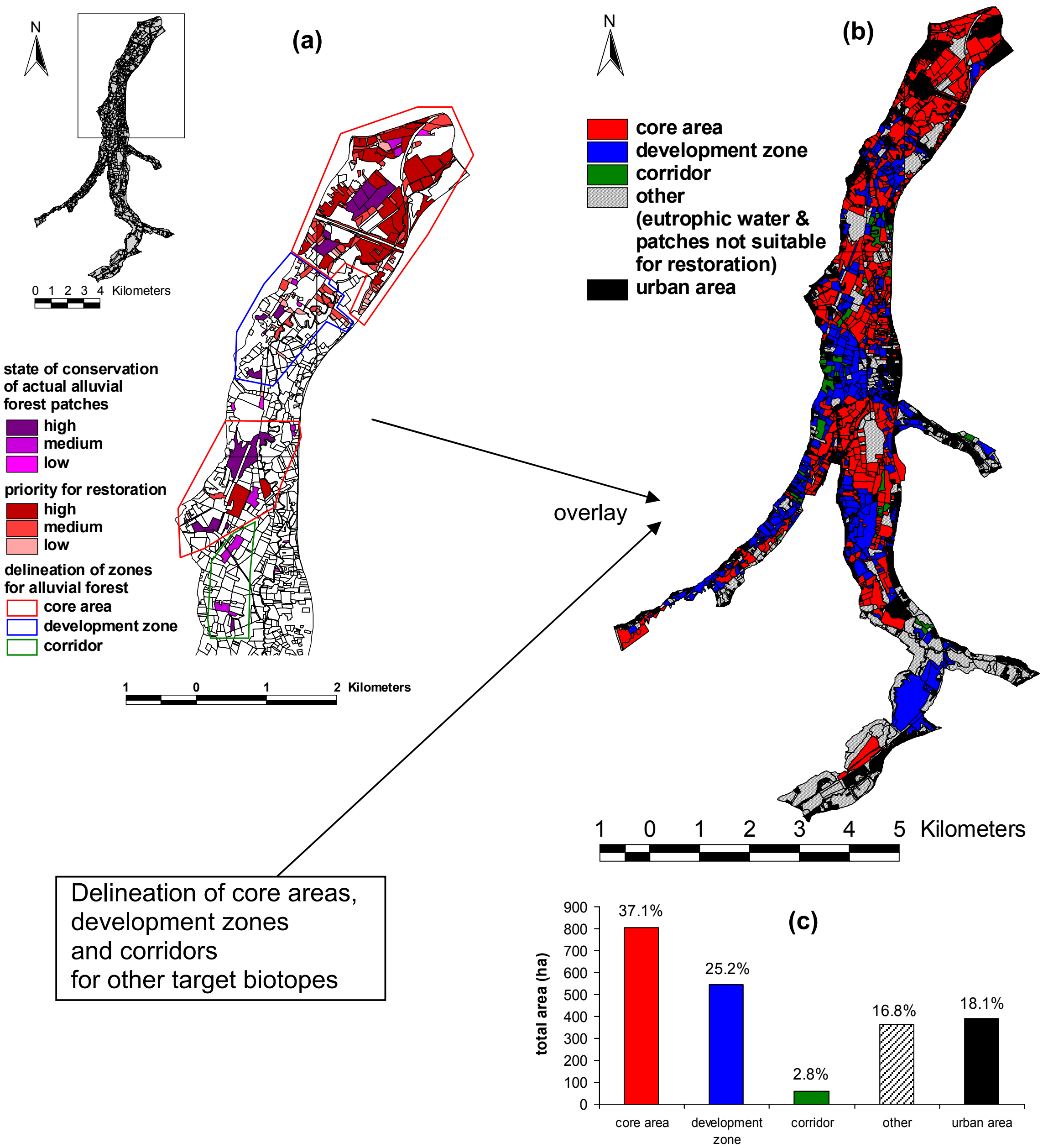

Fig. 2.14: Illustration of the process leading to delineation of core areas, development zones and corridors within the Dyle valley study area: **(a)** delineation of core areas, development zones and corridors for alluvial forests based on conservation status of actual alluvial forest target biotope patches and priority ranking for restoration of currently unoccupied patches into alluvial forest in the northernmost section of the study area; **(b)** resulting distribution of core areas, development zones and corridors throughout the whole study area through overlay of maps as depicted in (a) for all target biotopes; **(c)** total area attributed to core area, development zones and corridors.

2.2.3.2. Species-specific approach

As mentioned earlier, the main focus of the ecological team was the elaboration of the optimal ecological scenario at target biotope level. However, to complement these results and to demonstrate the influence of different species characteristics and requirements on the resulting desired ecological network, the results of a tentative species-specific approach are briefly illustrated. From Fig. 2.15 it may be clear that patches important to be restored into the same target biotope can differ between taxonomic groups (a vs. b and c vs. d). Moreover, some patches may be equally important to be restored into different target biotopes depending on the species of interest (e.g. a vs. c and b vs. d) resulting in conflicting management options.

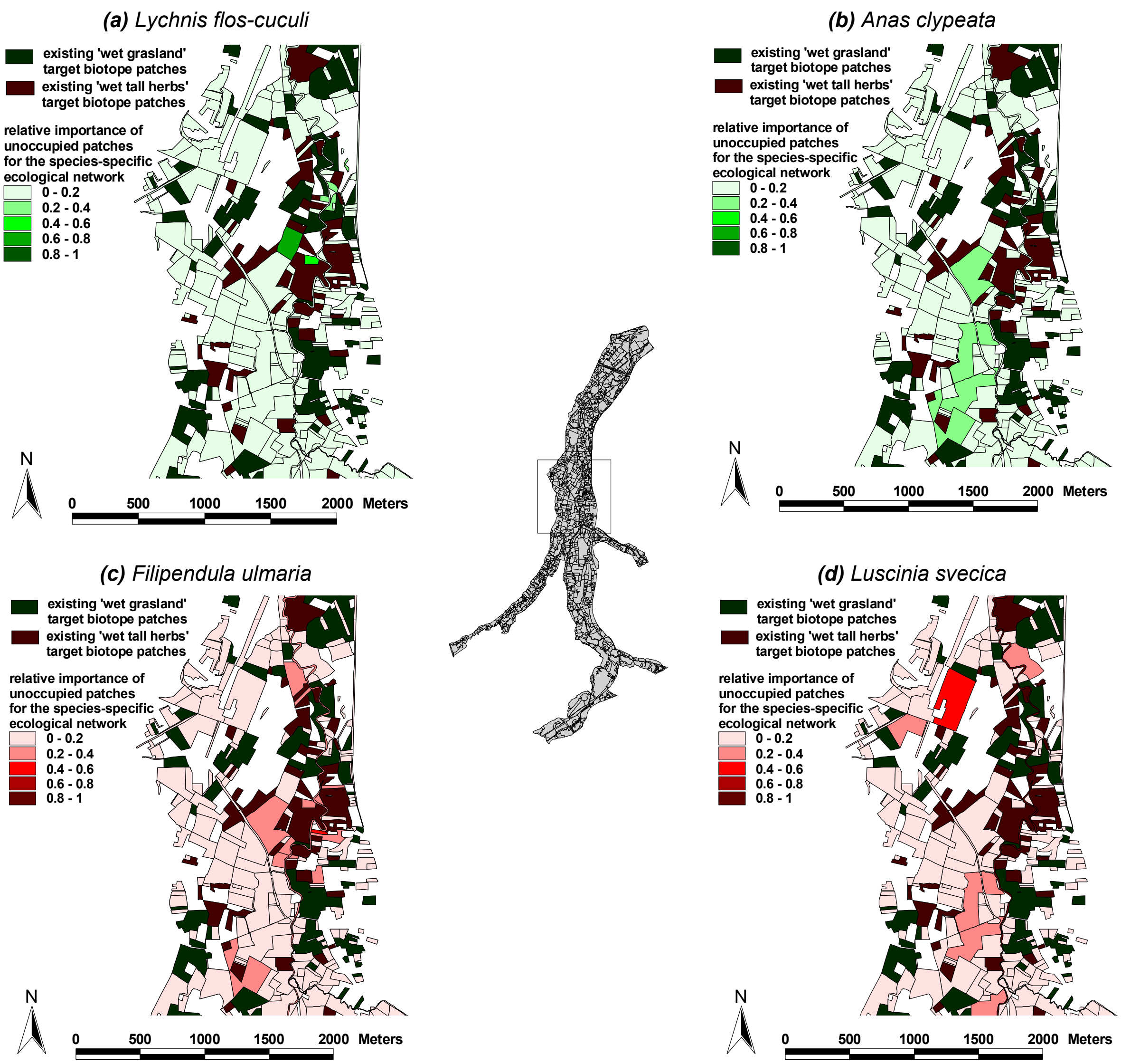

Fig. 2.15: Differences between species-specific ecological networks for the central section of the study area: **(a)** *Lychnis flos-cuculi*, a plant species of wet grasslands; **(b)** *Anas clypeata*, a bird species also thriving on wet grasslands; **(c)** *Filipendula ulmaria*, a typical plant species of vegetation types dominated by wet tall herbs and **(d)** *Luscinia svecica*, a bird species also occurring in wet tall herbs vegetation. 'Importance' of each patch (*i.e.* metapopulation capacity increment multiplied by potentiality score) for the ecological network of that particular species is indicated with a graduated colour (green for grassland species, red for species of tall herbs vegetation). For all species-specific dispersal distances and minimum area requirements see Table 2.3.

2.2.4. Discussion

2.2.4.1. A metapopulation capacity approach at biotope level: interpretation and possibilities for a wider application

By means of the metapopulation capacity approach built into a straightforward model we have deployed a simple step-by-step method for assessing functionality of ecological networks. With this method, ecological networks can be interpreted from a species perspective (see further) or, in a more pragmatic approach, from a target biotope perspective. Knowledge of metapopulation dynamics and landscape structure is incorporated as well. Although the biotope approach may seem rather coarse when it comes to the assessment of habitat quality and feasibility of restoration, a whole body of scientific data and expert knowledge can be incorporated as input data (depending on their availability). Moreover, it is based on the ecologically widely accepted metapopulation theory and its principles (Hanksi 1999, Hanski & Ovaskainen 2000 & 2002).

Hence, it may be clear that a more widely applicable methodology for implementing an ecological network based on scientific literature and expert knowledge was elaborated. Again, depending on the availability of detailed input data, the way of assessing potential for the chosen target biotopes or evaluating restoration feasibility can differ substantially. However, as illustrated by the study of Daemen (2004), the approach can easily be transferred to similar projects aiming at elaboration of an optimal ecological network. In particular, Daemen (2004) applied the metapopulation capacity approach as an indicative tool to asses the optimal distribution of *Caltha palustris* grasslands in the study area of the nature design project 'West-Vlaamse Scheldemeersen' (VLM).

2.2.4.2. A species-specific approach to overcome the shortcomings of the analysis at biotope level?

The effects of dispersal among a network of patches highly depend on the process or organism of interest (Rutledge 2003). Moreover, dispersal parameters also vary between different metapopulations of the same species and between landscapes. So, one has to be cautions with parameter generalization, even for the same species in a comparable landscape (Mennechez *et al.* 2004). Moreover, species-specific traits such as minimum area requirements and dispersal distances were hard to find in literature. Additionally, the combination of the spatial characteristics of species requirements (e.g. home-ranges of animal species) and the level of detail of the vegetation map (in our case the BWK) also limits species selection (Verboom & Pouwels 2004). All those things should be kept in mind while interpreting the results of our metapopulation capacity approach at species level. Results showed that there were indeed big differences in the optimal ecological network depending on the target species and the scale (for plants the scale of the BWK map is certainly too large). Furthermore, conflicts between actions taken for the restoration and management of different target species are to be expected: e.g. for forest plant species an 'unoccupied' patch should be restored into alluvial forest, while at the same location the model could predict restoration into wet grassland to be important for a bird species. While, at target biotope level, conflicting restoration options could be resolved objectively through incorporation of restoration feasibility scores, decisions on which biotope to restore for a particular target species would involve subjective preference for a certain species over another. Hence, restoration actions cannot always be based on the outcome of a model. The metapopulation capacity approach at species level can help to identify conflicting options, but the ultimate decision should depend on conservation and management priorities in that specific area.

2.2.4.3. Limitations of the model and suggestions for improvement

A disadvantage of the method, but inherent to ecological networks in a species' perspective, remains the relatively high level of uncertainty. The species-specific approach showed that the results of our model could be sensitive to small changes in the two species-specific parameter values (dispersal distance and minimum area requirements). By consequence, it can also be expected that the proposed ecological optimal scenario in our target biotope approach would be different if another set of parameters (*i.e.* another fictive target species) was used. Nevertheless, even though the model may provide a rather simplified view on the construction of ecological networks, the results can certainly offer useful indications and directions, as long as they are confronted with other relevant field data on species requirements and distribution.

The model, as it is currently written (App. 2.3), may also be prone to justified criticism. Unfortunately, within the scope of this BELSPO-project, time was too short to further refine the incorporation of a metapopulation capacity approach into a model. The most important shortcoming is the addition of actually 'unoccupied' patches to the existing network of target biotopes one at a time, without reckoning for the ones that were already added before. This may result in patches attributed to the same target biotope being more spatially clustered in our version of the optimal ecological scenario. In a more realistic scenario, step-by-step construction of the desired ecological network should be a sequential procedure since the 'actual' ecological network to which a new 'unoccupied' patch must be added changes if a formerly 'unoccupied' patch is restored. A process of iteratively adjusting patches and searching for the combination with the highest metapopulation capacity, *i.e.* the 'best' ecological network, would be more legitimate. Calculation of distances between patches can also be improved to come closer to reality. In the model as it exists now, Euclidian distances are calculated. Of course, that may not be completely correct in the field since, for their dispersal, a lot of species do not always follow the shortest distance between patches. In many cases, plant dispersal or animal movement rather adheres to (linear) landscape features such as small landscape elements (e.g. hedgerows, uncultivated field margins, etc.), rivers and small streams (Hermy & De Blust 1997) thereby hindered by or actively avoiding barriers (e.g. roads, unsuitable agricultural land or urban zones). These deviations are not yet incorporated in the calculation of dispersal distances. However, they may be difficult to account for in the metapopulation capacity approach at biotope level since dispersal trajectories are inherently species-specific traits.

To assess whether or not the developed network can guarantee the maintenance of a metapopulation of a certain species, the actual model may not provide sufficient detailed information. Indeed, a species is predicted to persist in a landscape if the metapopulation capacity of that landscape is greater than a threshold value determined by the properties of the species (Hanski & Ovaskainen 2000). Actually, this threshold value (e/c, with e and c species-specific parameters which determine respectively the extinction and colonization rates) is not taken into account, since we simply selected the ecological network having the highest metapopulation index. However, these species-specific parameters (e and c) are not known for most species and the necessary distribution data to calculate those values are lacking in most cases. In his study, Daemen (2004) estimated these parameters by using the SPONSIM-tool, a Stochastic Patch Occupancy SIMulator developed by Moilanen (1999, 2004). Even so, this approach also depends on availability of detailed distribution data which can only be obtained through long term monitoring programs. Therefore, since time is restricted to gather these data and as long as the results of the current model to build an ecological network are used to rank the restoration of 'unoccupied' patches in terms of their importance for the ecological network, they certainly provide useful suggestions for the elaboration of an optimal ecological network.

2.2.5. Conclusion

The lack of species-specific data (or the time necessary to gather these data through intensive fieldwork on a long term basis) is an acute problem in the context of the elaboration of an optimal ecological network at individual species level. Therefore, we adopted a more global approach at the biotope level which allowed for the assembly of an optimal ecological network of a selected set of target biotopes within the limited time frame. Moreover, the proposed method, which is based on preferential restoration of patches to obtain a landscape structure representing the highest metapopulation capacity, offers a framework for the elaboration of ecological networks in other areas and can certainly be applied in a wider context (*cf.* Daemen 2004).

In the case of the Dyle valley study area, results showed that for the elaboration of an optimal ecological network, expansion of already existing target patches with a high conservation status is desirable rather than spending time and resources on the restoration of patches which are currently isolated from the existing set of target biotope patches in the study area. However, to a certain extend this preference for clustering target biotopes may originate from the specificity of the model. In conclusion, we should keep in mind that modelling the desired ecological network through a metapopulation capacity approach may be a useful tool to decide on restoration and management priorities at target biotope level, but the model may not (yet) be powerful enough for realistic predictions for individual species in particular landscapes. Preliminary analyses at species level were performed to test for the model's sensitivity to species-specific parameters such a minimum area requirements and dispersal distances. However, as long as little or no information is available on other metapopulation-related variables such as extinction and colonization rates, results can only be interpreted in a comparative manner and they cannot be a sound foundation for explicit predictions about species persistence in a landscape.

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PART 3: LEGAL ASPECTS

3.1. General Introduction

La planification et la mise en oeuvre d'un réseau écologique ne sauraient être envisagées en dehors de son cadre juridique. L'utilisation des sols et la gestion des ressources sont régies par une foule d'instruments contraignants, incitatifs, fiscaux ou fonciers qu'il importe de prendre en considération lors de la mise en œuvre d'une politique de réseau écologique. L'intervention des pouvoirs publics – comme autorités ou comme gestionnaires – est indispensable à cette fin. La protection des sites et des populations ne peut être garantie à long terme que moyennant des mesures juridiques contraignantes, même si la maîtrise foncière d'un site peut suffire généralement à en préserver l'intérêt biologique. L'octroi de subventions publiques à la gestion ne peut par ailleurs être envisagée sans un cadre légal strict, permettant d'éviter tout abus. Enfin, le droit permet également de garantir le respect, par l'administration, des droits fondamentaux comme le droit de propriété lors de l'adoption de mesures contraignantes telles que Natura 2000. L'objet de la présente partie est donc d'évaluer, à la fois sur le plan théorique et sur le plan pratique, dans quelle mesure le droit positif en vigueur dans les deux régions permet-il de concrétiser le scénario écologique optimal proposé par l'ECO-team. Après un rappel du cadre juridique général (3.2.), l'on étudiera le cas pratique choisi, à savoir la vallée de la Dyle (3.3.). Des conclusions générales en seront tirées (3.4). Cette partie n'a pu être traduite en anglais, compte tenu des difficultés de traduire des termes juridiques précis dans une langue non officielle et du manque de temps imparti pour le legal team.

3.2. Theoretical background : legal framework for ecological network implementation

3.2.1. Cadre juridique général de l'utilisation des sols

Le sol (« *land* » en anglais), ainsi que la végétation (sur pied) qui s'y développe, sont considérés en droit comme des *biens immeubles* (art. 518, 520 et 521 C. civ.). Ceux-ci peuvent être répartis entre les biens appartenant aux particuliers (personnes de droit privé) et les biens appartenant aux personnes morales de droit public (Etat, régions, communautés, provinces, communes, etc.). Les particuliers ont, en principe, la libre disposition de leurs biens, conformément aux règles du droit civil tandis que les biens des personnes morales publiques sont administrés et ne peuvent être aliénés que suivant les règles et dans les formes propres au droit administratif (art.537 C.Civ.)[1].

Les biens immeubles dans le commerce font l'objet de droits réels, dont le plus complet est le *droit de propriété*. Celui-ci est défini comme « *le droit de jouir et disposer des choses de la manière la plus absolue, pourvu qu'on n'en fasse pas un usage prohibé par les lois ou par les règlements* » (art.544 C.Civ.). En d'autres termes, il ressort de cette disposition et de l'article 537 précité du Code civil que l'occupation et l'utilisation d'un sol en Belgique sont libres dans le chef de son propriétaire, sous réserve de toutes les restrictions que le législateur juge bon d'imposer à l'exercice de ce droit (« *servitudes légales d'utilité publique* »[2]). Si la privation du droit de propriété ne peut être imposée à un propriétaire que moyennant le respect des procédures d'expropriation[3] et le paiement d'une « *juste et préalable indemnité* » (art. 16 de la Constitution), il n'en est pas de même pour les restrictions apportées à ce droit au nom de l'intérêt collectif qui ne doivent être indemnisées que lorsque le législateur le prévoit[4].

[1] Les biens qui appartiennent aux personnes morales de droit public se divisent eux-mêmes en deux classes : les biens du domaine public (biens possédés par l'Etat dans l'intérêt de la collectivité et, à ce titre, placés hors du commerce, indisponibles, inaliénables, imprescriptibles) et les biens du domaine privé (biens possédés par les collectivités publiques au même titre que les particuliers et, par conséquent, susceptibles d'être l'objet de commerce juridique, notamment d'aliénation) ; HANSENNE, in COLL., *Guide de droit immobilier*.
[2] PAQUES, in COLL., *Guide de droit immobilier* ; PAQUES, 1983, p. 170.
[3] PAQUES, B., 2001, p. 51.
[4] Voy. sur cette question, PAQUES, 2002.

L'intervention des pouvoirs publics pour influencer l'utilisation des sols s'exerce dans le respect des *règles de répartition des compétences* fixées par la loi spéciale du 8 août 1980[5]. Les trois régions sont compétentes dans les principales matières qui touchent au réseau écologique, à savoir la conservation de la nature, l'aménagement du territoire et l'urbanisme, la protection du patrimoine immobilier, l'agriculture, la chasse, les forêts, l'eau, les travaux publics et les transports[6]. Cette unité des compétences est un gage de cohérence pour gérer le milieu à l'intérieur des frontières, mais peut poser des problèmes de cohérence en ce qui concerne des écosystèmes ou des corridors transrégionaux (comme la vallée de la Dyle). La coopération entre régions, par voie de concertation ou d'accords de coopération par exemple, est très peu organisée en conservation de la nature[7].

Une partie importante des interventions des pouvoirs publics sur l'utilisation des sols s'inscrit dans le cadre de *polices administratives*[8] comme le droit de l'aménagement du territoire, de l'environnement et de la conservation de la nature. Ces polices sont susceptibles soit de contribuer à la mise en place du réseau écologique, soit d'interférer avec celui-ci, selon les objectifs poursuivis et le degré de prise en compte de l'environnement. Parmi les instruments juridiques susceptibles d'être utilisés par les pouvoirs publics pour mettre en place un réseau écologique, figurent :

- les instruments contraignants, dont les servitudes légales d'utilité publique (par exemple le statut de site Natura 2000 ou la zone naturelle au plan de secteur) ;
- les instruments incitatifs et économiques (taxes, subventions, mesures fiscales, …) ;
- les instruments fonciers (expropriation, droit de préemption, échange de propriété, remembrement,…) ;
- les instruments de sensibilisation ;
- les instruments de gestion des biens publics (régime forestier, gestion des cours d'eau, de la voirie,…)[9].

La mise en œuvre intégrée d'un réseau écologique est susceptible de rencontrer d'importantes difficultés du fait que le *processus de prise de décision* en matière de contrôle de l'utilisation des sols par les pouvoirs publics s'avère, au sein même des régions, fortement *parcellisé* à la fois horizontalement et verticalement[10]. Horizontalement, il relève en effet de nombreuses législations indépendantes, poursuivant des objectifs différents. Conséquence immédiate, les administrations régionales chargées de les mettre en œuvre sont elles-mêmes historiquement fortement cloisonnées. Les mécanismes d'articulation entre législations ne sont pas nécessairement prévus. Verticalement, l'actuelle tendance à la décentralisation des compétences en matière d'urbanisme et d'environnement au nom du principe de subsidiarité[11], a pour conséquence que les décisions (permis, plans locaux, gestion des biens publics) sont prises par une multitude d'autorités aux ressorts territoriaux limités, notamment les communes et les provinces.

3.2.2. Planification stratégique du réseau écologique

Compte tenu de cette situation institutionnelle et administrative composite, la mise en place d'un réseau écologique doit nécessairement passer par une phase de planification (y compris spatiale), à toutes les échelles pertinentes. Cette étape primordiale doit jouer différentes fonctions, à savoir notamment identifier les problèmes, définir les priorités de conservation, repérer les lacunes du réseau d'aires protégées existant et établir une structure spatiale cohérente du futur réseau à l'échelle régionale[12]. Elle doit aussi permettre d'assurer une intégration homogène des considérations sur le réseau écologique dans les politiques sectorielles concernées, notamment en identifiant les instruments

[5] Voy. sur cette question, TULKENS, 1999.
[6] Voy. l'article 6, § 1er, I à X, de la loi spéciale du 8 août 1980.
[7] Seules les mesures applicables aux forêts ou aux nappes d'eau souterraine à cheval sur deux régions doivent faire l'objet d'une concertation (article 6, § 2, de la loi spéciale du 8 août 1980). Sur cette question, voy. MOERENHOUT, 2001.
[8] Voy. PAQUES, *Droit public des biens…*, 2002 ; MAST, A. et al., 2002.
[9] PAQUES, *Droit public des biens…*, 2002.
[10] SHINE et de KLEMM, 1999.
[11] NEURAY, 2001, pp.125-130.
[12] Sur les fonctions de la planification en conservation de la nature, voy. VAN HOORICK, 2000, p. 361 et s.

en vigueur incompatibles avec les exigences de conservation. Elle doit enfin se fonder sur une participation active des acteurs concernés et du public afin d'asseoir la légitimité des mesures de conservation prises et respecter les droits fondamentaux. Divers documents de planification à différents niveaux (régional, provincial et communal) sont susceptibles d'être utilisés à ces fins[13]. L'articulation entre les différents plans généraux et sectoriels est très complexe[14] et n'a pu faire l'objet de longs développements dans ce rapport. Il est renvoyé aux annexes pour plus de détails.

Au niveau régional, en *Région wallonne*, divers instruments de planification environnementale, à valeur indicative et régis par le décret du 21 avril 1994 relatif à la planification en matière d'environnement dans le cadre du développement durable[15] préconisent la mise en place d'un réseau écologique, à savoir le Plan d'Environnement pour le Développement Durable (PEDD)[16], et le futur Plan d'action pour le développement de la nature (PADN). Si ces plans sont élaborés sur une base participative, aucun n'établit une cartographie concrète du réseau. En aménagement du territoire, le Schéma de Développement de l'Espace Régional (SDER), adopté en 1999[17], mentionne expressément le réseau écologique comme objectif de conservation du patrimoine naturel[18], mais ne planifie pas le réseau dans l'espace et n'est pas légalement contraignant. Seule la procédure de sélection des sites Natura 2000 constitue en soi un exemple partiel de planification spatiale à valeur contraignante d'un réseau écologique en Région wallonne. Elle ne vise toutefois que la conservation des espèces et des types d'habitats d'intérêt communautaire présents en Wallonie et n'inclut pas explicitement de zones de liaison ou de développement, au sens des définitions ECONET[19]. Elle ne prévoit de surcroît aucune participation en amont de la procédure. En *Région flamande*, les instruments de planification mis en place en matière de conservation de la nature sont prévus par le décret du 21 octobre 1997 concernant la conservation de la nature et le milieu naturel[20]. Ils sont répartis entre le « *Natuurbeleidsplan* » et cinq «*deelplannen* » (Plan général de la nature et plans partiels), un « *Natuurrapport* » (Rapport de la nature) et des « *Natuurichtplannen* » (Plans directeurs de la nature). Ils permettent de fixer clairement les priorités de conservation et donc les espèces et habitats cibles pour mettre en place un réseau écologique. Sur le plan spatial, le «*Natuurontwikkelingsplan* » (NOP) de 1990 et surtout le « *Plan voor een Groene Hoofdstructuur* » de 1993, adoptés en dehors d'un cadre légal ont inspiré la mise en place du Vlaamse Ecologische Netwerk (VEN) et de l'Integraal Verwevings en Ondersteunend Netwerk (IVON), dont la mise en place est régie avec précision par le décret de 1997. La cartographie des zones de ce double réseau, réalisée sur la base des critères fixés dans le décret et soumise à participation, constitue une véritable planification spatiale du réseau, assez proche des concepts définis par ECONET. Elle est étroitement articulée avec la planification spatiale générale[21]. En matière d'aménagement du territoire, le « *Ruimtelijk structuurplan Vlaanderen* » (RSV) (Schéma de structure d'aménagement de la Flandre) [22], équivalent du SDER, mais dont certaines parties ont force obligatoire, prend en considération le VEN et l'IVON dans ses objectifs.

Au niveau provincial et communal, la planification locale stratégique peut être mise en place au travers de plusieurs instruments de planification. Certains relèvent de la législation sur la conservation de la nature (par ex. en Région wallonne, le Plan Communal de Développement de la Nature (PCDN) ou le Plan Communal d'Environnement et de Développement de la Nature (PCEDN) ; en Région

[13] *Amén.-Env.*, 1994; VAN HOORICK, 2000, pp.347-391; VAN HOORICK, 2004, pp. 91-116.
[14] Voy. sur cette question, JADOT, 2002.
[15] Décret du 21 avril 1994 relatif à la planification en matière d'environnement dans le cadre du développement durable (M.B., 23/04/94) modifié par décret du 22 janvier 1998 (M.B., 19/02/98).
[16] Arrêté du Gouvernement wallon du 9 mars 1995 adoptant le plan d'environnement pour le développement durable (PWEDD) (M.B., 25/07/95).
[17] Arrêté du Gouvernement wallon du 27 mai 1999 adoptant définitivement le schéma de développement de l'espace régional (SDER) (M.B., 21/09/99).
[18] « *la protection et le développement patrimoine naturel doivent s'appuyer sur la mise en place d'un réseau écologique. Sa concrétisation nécessite d'une part de reconnaître à certaine parties du territoire une vocation exclusive de conservation de sites de grand intérêt biologique, et d'autre part de tenir compte sur l'ensemble du territoire des conditions nécessaires au maintien et au développement des espèces animales et végétales* ».
[19] DUFRENE, M., GATHOYE, J.-L., in *Natura 2000 et le droit* (2004) qui indiquent cependant qu'un certain nombre de sites Natura 2000 ont été sélectionnés afin de jouer le rôle de zones de liaison ou de corridors. C'est le cas notamment de certains fonds de vallée
[20] Décret du 21 octobre 1997 concernant la conservation de la nature et le milieu naturel intitulé « *Natuurdecreet* » (M.B., 10/01/98)
[21] VAN HOORICK, G., 2004
[22] Arrêté du 23 septembre 1997 portant fixation définitive du Schéma de Structure d'Aménagement de la Flandre (M.B., 21/03/98)

flamande, les Plans d'Orientation environnemental communaux (« *Gemeentelijke milieubeleidsplan* ») ou les Plans Communaux de Développement de la Nature (« *Gemeentelijk natuurontwikkelingsplan* » - GNOP's). Les autres relèvent de celle sur l'aménagement du territoire (les schémas), qu'il conviendrait au demeurant de coordonner avec les premiers. D'intéressantes potentialités existent à cet égard.

3.2.3. La mise en œuvre du réseau écologique

Les mesures concrètes visant à mettre en œuvre une politique de réseau écologique peuvent être rangées dans trois catégories : les mesures *de protection, de gestion et de restauration* à appliquer dans les zones centrales et de développement ; les mesures destinées à assurer la *connectivité* le cas échéant entre les zones du réseau ; les mesures visant à *intégrer* le réseau écologique dans le contexte global de l'utilisation des sols et de gestion des ressources (*infra*, 3.2.4). Ces mesures très diverses doivent être mises en œuvre nécessairement au moyen d'instruments juridiques, en particulier lorsqu'elles nécessitent l'imposition de contraintes ou l'octroi de subventions[23]. L'intervention des pouvoirs publics s'avère donc indispensable, même si les initiatives purement privées peuvent jouer un rôle considérable[24]. L'un des enjeux les plus importants tient dès lors dans le *degré de contrainte imposé aux autorités* dans la mise en œuvre du réseau écologique[25]. Si la volonté politique – ou une contrainte légale – existe, sa mise en œuvre peut prendre différentes voies, le cas échéant en combinaison – à savoir la voie contraignante, la voie incitative et la voie foncière[26] –, en fonction des instruments utilisés. La volonté politique ne saurait elle-même avoir de réels effets que si elle reçoit un soutien des acteurs concernés par le réseau écologique. A nouveau, la *participation* active des acteurs concernés et du public apparaît indispensable au stade de l'adoption de ces mesures, en particulier lorsque des contraintes fortes sont imposées unilatéralement par le Gouvernement ou quand celui-ci souhaite impliquer propriétaires et occupants dans la gestion des sites. Parmi les instruments à la disposition des pouvoirs publics pour créer un réseau écologique, l'on peut distinguer les zones protégées des autres instruments applicables dans l'ensemble du territoire.

3.2.3.1. Les zones protégées

A. En droit de la conservation de la nature

Les *zones protégées*[27] sont amenées à jouer un rôle fondamental pour le réseau écologique, à savoir créer les conditions du maintien ou de la restauration des différents éléments du réseau les plus menacés ou nécessitant une gestion spécialisée – la plupart des zones centrales, mais aussi, le cas échéant, des zones de développement ou de liaison. Elles jouent aussi le rôle capital de « *réservoirs* » de biodiversité, à partir desquels peut s'effectuer la recolonisation de sites voisins, le cas échéant après restauration.

En *Région wallonne*, la loi du 12 juillet 1973 sur la conservation de la nature[28] et ses arrêtés d'exécution habilitent le Gouvernement à créer des réserves naturelles domaniales ou agréées, des réserves forestières, des zones humides d'intérêt biologique (ZHIB) et des cavités souterraines d'intérêt scientifique (CSIS). Depuis 2001[29], elle l'oblige à désigner des sites Natura 2000 en vue de maintenir ou rétablir dans un état de conservation favorable les espèces et habitats d'intérêt

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[23] Pour une présentation des types d'instruments à la disposition des pouvoirs publics en matière d'environnement et d'aménagement du territoire, voy. PAQUES, M., 2002, p. 31 et s.
[24] Il est exact qu'une politique d'acquisition des sites par la Région ou des associations privées permet de ne pas devoir nécessairement prendre des mesures juridiques, le seul fait de disposer de la maîtrise foncière étant déjà un gage de protection et autorisant la gestion écologique. En l'absence de toute protection sanctionnée pénalement, les dommages liés aux atteintes extérieures ne peuvent cependant se voir sanctionnés que dans le cadre du droit de la responsabilité civile – qui peut requérir une remise en état des lieux –, ce qui est très insuffisant, notamment si le dommage est irréversible.
[25] A ce jour, force est de constater que les seules initiatives de mise en place d'un réseau dépassant le stade de la déclaration d'intention – tel que le VEN ou le réseau Natura 2000 wallon – résultent de telles contraintes.
[26] A ces trois voies s'ajoute une gestion intégrée des biens publics en vue d'établir le réseau.
[27] CEDRE (dir.), *Le zonage écologique*, actes du colloque de Gembloux du 29 mars 2001, Bruxelles, Bruylant, 2002.
[28] Loi du 12 juillet 1973 sur la conservation de la nature (M.B., 11/09/73) modifié notamment par le décret du 6 décembre 2001 relatif à la conservation des sites Natura 2000 ainsi que de la faune et de la flore sauvage (M.B., 22/01/02).
[29] Décret du 6 décembre 2001 relatif à la conservation des sites Natura 2000 ainsi que de la faune et de la flore sauvage (M.B., 22/01/02).

communautaire présents en Wallonie, conformément aux directives Oiseaux et Habitats[30]. Les *statuts « classiques » de protection* contribuent de façon cruciale à la préservation des sites les plus prestigieux du réseau, grâce aux mesures strictes de protection et de gestion qui s'y appliquent. Toutefois, leur sélection et leur désignation nécessitent souvent une maîtrise foncière[31], ce qui implique que, d'une part, la création de la zone protégée est rarement envisagée sans l'accord du propriétaire[32] et, d'autre part, la taille des parcelles protégées est assez faible en moyenne (quelques hectares).Elles ne peuvent donc couvrir d'importantes superficies d'un seul tenant, ni même en absolu[33]. Les mesures de protection sont par ailleurs limitées au périmètre du site protégé et ne permettent souvent pas de contrer les incidences extérieures du site, par exemple les épandages agricoles autour du site. Le *statut de site Natura 2000*[34] diffère des statuts classiques de protection à plusieurs points de vue. Ces sites sont appelés à couvrir une superficie totale nettement plus grande que les autres statuts réunis (+/- 220.000 ha) et les sites d'un seul tenant sont nettement plus grands (plusieurs centaines d'hectare en moyenne). Pour être protégé et géré, chaque site sélectionné doit être désigné par un arrêté du Gouvernement (« *arrêté de désignation* »), véritable plan de protection et de gestion du site. Chaque site fait l'objet d'un régime de « *conservation* », comprenant un régime de « *prévention* » et un régime de « *gestion active* ». En termes de protection, il oblige les autorités à prévenir tout *impact significatif*, quelle qu'en soit la cause, sur les habitats et espèces pour lesquels les sites ont été désignés. Il comporte, d'une part, des interdictions à valeur réglementaire et à « *géométrie variable* », devant être adaptées aux exigences écologiques des espèces et habitats cibles, tout en prenant en compte les exigences socio-économiques. D'autre part, il exige en effet qu'une « *évaluation appropriée des incidences* », spécifique à Natura 2000, soit réalisée pour tous les plans à valeur réglementaire et tous les projets soumis à permis « *susceptibles d'affecter significativement* » un site Natura 2000. Ce double mécanisme permet de n'imposer que les restrictions strictement nécessaires au droit de propriété pour atteindre les objectifs. Dans la même veine, concernant la gestion, le législateur a privilégié la voie contractuelle, en appelant les propriétaires et occupants – en particulier les agriculteurs et exploitants forestiers – à coopérer activement dans le cadre de « *contrats de gestion active* » subsidiés. Ce régime étant en grande partie à caractère réglementaire, les objectifs « *Natura 2000* » ont la priorité sur ceux d'ECONET.

En *Région flamande*, le décret du 21 octobre 1997 concernant la conservation de la nature et le milieu naturel[35] habilite le Gouvernement à créer des aires protégées « classiques » (réserve naturelle flamande, réserve naturelle agréée). Mais surtout il établit les bases légales du futur Réseau écologique flamand ("*Vlaams Ecologisch Netwerk*") (VEN) (qui correspond plus ou moins aux zones centrales ECONET) et du réseau intégral de soutien et d'imbrication ("*Integraal Verwevings- en Ondersteunend Netwerk*") (IVON) (qui correspond plus ou moins aux zones de développement ECONET). Le VEN est composé de deux types de zones, les « *Grote Eenheden Natuur* » (GEN) et les « *Grote Eenheden Natuur in Ontwikkeling* » (GENO). L'IVON comprend des « *zones naturelles d'imbrication et de transition* ». Dans le VEN, sont interdites toutes activités susceptibles de causer des préjudices irréparables à la nature, sauf en l'absence d'alternatives pour des raisons impératives d'intérêt public majeur, y compris de nature sociale ou économique si les mesures compensatoires et limitatrices nécessaires sont prises. Dans les zones GEN, l'autorité prend les mesures nécessaires pour préserver, restaurer et développer par priorité, par rapport à d'autres fonctions dans le site, la nature et le milieu naturel. Par ailleurs, depuis 2002[36], le législateur flamand a finalisé la transposition en droit interne des directives Habitat et Oiseaux[37]. Les sites Natura 2000 flamands sont appelés « zones spéciales de conservation » (« *Speciale Beschermingszones van Vogelrichtlijngebieden - SBZ-V* » et « *Speciale*

[30] Directive « *Oiseaux* » 79/409/CEE du 4 avril 1979 concernant la conservation des oiseaux sauvages ; Directive « *Habitats* » 92/43/CEE du 21 mai 1992 concernant la conservation des habitats naturels ainsi que de la faune et de la flore sauvages.
[31] Les statuts de ZHIB et de CSIS peuvent être octroyés en principe sans l'accord du propriétaire. En pratique, celui-ci est presque toujours demandé via une convention.
[32] Certains outils fonciers comme le droit de préemption, les subventions à l'achat de terrains par des associations privées ou l'expropriation permettent en effet de pallier en partie à ce problème.
[33] Comparez les chiffres in DUFRENE, 2001, aux superficies couvertes par Natura 2000.
[34] BORN, C.-H., 2005, à paraître ; COLL., *Natura 2000 et le droit,…*, 2004 ; COMMISSION EUROPEENNE, 2000 ; NEURAY, 2002 ; LAMBOTTE et NEURAY, 2003.
[35] Décret du 21 octobre 1997 concernant la conservation de la nature et le milieu naturel (M.B., 10/01/98).
[36] Décret du 19 juillet 2002 (M.B., 31/08/02).
[37] Voy. VAN HOORICK, 1997; VAN HOORICK, 2004.

Beschermingszones van Habitatrichtlijngebieden – SBZ-H »). Ils font l'objet d'un régime de « *conservation* » qui comprend des mesures de conservation nécessaires répondant aux exigences écologiques des habitats et espèces pour lesquels le site a été désigné et des mesures de conservation nécessaires pour éviter toute détérioration de la qualité naturelle et de l'environnement naturel de ces habitats et espèces ainsi que pour éviter toute perturbation significative de ces espèces. Comme en Région wallonne, la compatibilité entre les objectifs ECONET et les objectifs de conservation de chaque site doit être vérifiée.Comme pour les sites appartenant au VEN ou à l'IVON ainsi qu'aux zones « *vertes* » du plan de secteur, le Gouvernement flamand doit établir un plan directeur de la nature (« *Natuurrichtplan* ») pour ces sites Natura 2000, plan qui contient une vision du site, une description des mesures incitantes et contraignantes en matière de conservation de la nature et une énumération des instruments nécessaires pour concrétiser la vision du site. L'adoption de ces plans a pris beaucoup de retard.

B. En matière de protection du patrimoine immobilier : les sites classés

La protection du patrimoine immobilier[38] est une police ancienne qui vise à préserver le patrimoine immobilier bâti et naturel[39]. La principale mesure intéressant la conservation de la nature consiste dans le classement comme site sur base de certains critères esthétiques et scientifiques. Ce régime juridique, assez complexe, est très utile pour préserver certains paysages ou massifs forestiers ou encore certains éléments du paysage remarquables. Il peut être imposé aux propriétaires, ce qui permet la protection de sites d'une certaine superficie. En revanche, il n'est pas spécifique à la conservation de la nature, et, sauf exception, aucun lien exprès n'est établi avec la législation et la planification sur la conservation de la nature. L'administration compétente reste l'administration de l'urbanisme et non celle de l'environnement, même si des contacts sont généralement établis.

3.2.3.2. Les instruments applicables en dehors des zones protégées

Il ressort de ce qui précède que les zones protégées, même créées selon une planification régionale, ne sauraient suffire pour assurer la mise en place d'un réseau écologique viable[40]. Une politique efficace de conservation nécessite donc une approche à l'échelle de tout le paysage, impliquant le recours à divers instruments juridiques, contraignants ou non, permettant de peser sur l'utilisation des sols dans un sens favorable à la biodiversité.

A. En droit de la conservation de la nature

Certaines dispositions légales ou réglementaires protègent sur l'ensemble du territoire, *tant en Région flamande qu'en Région wallonne*, les habitats des *espèces protégées*[41]. Cette protection directe des habitats d'espèces (sans passer par la création d'une aire protégée) est une servitude légale d'utilité publique, qui s'impose à toutes les autorités et à tous les particuliers, sauf dérogation. Toutefois, l'efficacité de cette protection est très limitée dès lors que souvent, les inventaires manquent et les particuliers ne peuvent identifier eux-mêmes l'espèce protégée. Plusieurs *incitants* sont susceptibles de contribuer à la mise en place d'un réseau écologique, qu'on peut répartir entre, d'une part, les mesures de compensation et d'indemnisation (fiscales ou autres) et, d'autre part, les subventions à la gestion du milieu naturel. En *Région wallonne*, on dénombre plusieurs subventions[42] et des mesures d'exemption fiscale[43]. Des « *périmètres d'incitation* » peuvent être créés autour des sites Natura 2000 pour en favoriser la gestion active[44]. Ils pourraient être utilisés pour gérer des corridors écologiques entre ces

[38] DRAYE, A.-M., 2003; THIEL, P., in COLL., *Guide de droit immobilier*.
[39] Articles 185-231 du CWATUP ; Décret flamand du 16 avril 1996 portant la protection des sites ruraux (M.B., 21/05/96) modifié par le décret du 13 février 2004 portant des mesures de préservation de paysages patrimoniaux (M.B., 18/03/04)
[40] Les aspects à gérer à l'échelle du paysage sont notamment le maintien ou le rétablissement de sa connectivité (fonctionnelle et structurelle), la conservation des espèces utilisant de multiples habitats de type agricole ou forestier et de la gestion tant qualitative que quantitative des ressources en eau, qui ne peut s'envisager de façon réaliste qu'à l'échelle des bassins versants ou à tout le moins du lit majeur.
[41] Art. 2-5 de la loi du 12 juillet 1973 ; Chapitre VI du décret flamand du 21 octobre 1997
[42] A titre d'exemples, les subventions à la plantation de haies ou à la gestion active des sites Natura 2000 ou encore à la gestion de réserves naturelles agréées
[43] A titre d'exemple, l'exemption du précompte immobilier de biens immobiliers érigés dans certains sites (réserves naturelles, réserves forestières et sites Natura 2000).
[44] Art. 25, § 3, de la loi du 12 juillet 1973.

sites. En *Région flamande*, il est également à relever plusieurs subventions[45], des primes pour les sites protégés et des mesures d'exemption fiscale[46],… Vu l'influence du droit de propriété, les *instruments de politique foncière et contractuels ou d'application volontaire* sont précieux dans la mesure où ils permettent la gestion de certains sites respectivement en cas de refus de collaboration du propriétaire. Dans la première catégorie, on peut citer, en *Région flamande*, l'adoption d'instruments fonciers spécifiquement consacrés à la conservation de la nature, comme la possibilité d'échange de propriété[47] ou encore le Projet d'aménagement de la nature (« *Natuurinrichtingsproject* »), outil foncier très puissant inspiré de la procédure de remembrement et visant à restaurer une structure écologique à l'échelle du paysage[48]. En *Région wallonne*, en dehors de l'expropriation, on peut citer le droit de préemption applicable dans les réserves naturelles agréées et les subventions à l'achat de terrains octroyées à certaines associations de protection de la nature agréées à cette fin[49]. Bien que souvent décrié, le remembrement légal peut également constituer une opportunité. Dans la seconde catégorie, on peut citer, en *Région flamande*, les accords de gestion (« *beheersovereenkomsten* »[50]) et, en *Région wallonne*, les conventions de gestion des bords de route (opération « *fauchage tardif* ») conclues avec les communes.

B. En droit de l'aménagement du territoire

Les plans d'aménagement réglementaires

La *planification réglementaire de l'affectation des sols* en Belgique est d'une grande importance car elle réglemente de façon contraignante la répartition dans l'espace des grandes catégories d'utilisations physiques des sols (agriculture, forêts, urbanisation, loisirs, etc.) par la technique du « *zonage* » qui divise le territoire en zones au sein desquelles s'appliquent des restrictions. Elle est fixée dans un système hiérarchisé de plans ayant valeur réglementaire, dont le plan de secteur est le plan le plus élevé. Tous les plans de secteur ont été adoptés aujourd'hui, couvrant tout le territoire. Ils ont une influence considérable sur le contrôle de l'utilisation des sols, puisque tout permis (d'urbanisme, d'environnement) doit leur être conforme, sauf dérogation. La répartition des zones urbanisables et des tracés de grandes infrastructures routières a donc un impact direct sur le réseau écologique[51]. En fixant les zones non urbanisables, ils permettent de contenir, dans une certaine mesure, l'urbanisation. Certaines zones sont expressément réservées à la conservation de la nature (zone d'espaces verts, zone naturelle), d'autres contribuent seulement au maintien de l'espace ouvert et à la formation du paysage (zones agricole et forestière). Des règles procédurales strictes, comprenant une participation du public et une évaluation des incidences, sont prévues pour la modification de ces plans. En *Région wallonne*, la législation principale est constituée par le Code Wallon de l'Aménagement du Territoire, de l'Urbanisme et du Patrimoine (CWATUP)[52]. En *Région flamande*, la législation principale est constituée par le décret du 18 mai 1999 portant organisation de l'aménagement du territoire[53] et par l'arrêté royal du 28 décembre 1972 relatif à la présentation et à la mise en œuvre des projets de plans et de plans de secteur[54].

[45] A titre d'exemple les subventions à l'acquisition de zones en vue de la création de réserves naturelles, au boisement des terres agricoles, aux travaux de rénovation rurale.
[46] à titre d'exemple, l'exemption des droits de succession pour les bois, du précompte immobilier pour les terrains situés dans le VEN, les primes pour les sites protégés, …
[47] Art. 41 du « *Natuurdecreet* » du 21 octobre 1997.
[48] Art. 47 du « *Natuurdecreet* » du 21 octobre 1997. Le projet formule des objectifs de conservation à atteindre et suggère des mesures et des modalités d'exécution avec une estimation financière. Il s'agit notamment des mesures suivantes : échange de parcelles, travaux aux infrastructures et aux lots, adaptation des routes, mesures de préservation, levées temporaires des compétences, imposition de servitudes temporaires, travaux en lien avec les eaux urbaines, travaux du sol, déplacement d'entreprise ….
[49] Art. 15-19 de l'arrêté de l'exécutif régional wallon du 17 juillet 1986 concernant l'agrément des réserves naturelles et le subventionnement des achats de terrains à ériger en réserves naturelles agréées par les associations privées (M.B., 11/10/86) ; Art.6 et 31 de la loi du 12 juillet 1973 ; Art.175-180 du CWATUP.
[50] Par ex. dans les biotopes constitutifs d'oiseaux de milieu ouvert (« *weidevogelbeheer* »).
[51] Même lorsqu'il n'est pas effectivement construit, la haute valeur foncière d'un terrain à bâtir rend plus difficile ou à tout le moins beaucoup plus coûteuse son acquisition à des fins de conservation.
[52] BOUILLARD, 1994 ; BOUILLARD, 2002*;* DELNOY, 2002 ; HAUMONT, 2003; HAUMONT, 1996.
[53] M.B., 18/05/99.
[54] M.B., 10/02/73.

La police des permis d'urbanisme et de lotir

Tant en *Région wallonne*[55] qu'en *Région flamande*[56], différents actes et travaux tendant à modifier le milieu naturel et les petits éléments du paysage sont soumis à autorisation urbanistique (permis d'urbanisme, permis unique (ou équivalent) ou permis de lotir)[57]. Toute demande de permis doit être soumise à évaluation des incidences, ce qui implique en principe une prise en considération des aspects environnementaux par l'autorité compétente. Celle-ci garde cependant, sauf règle contraire (par ex. Natura 2000 ou le VEN), un pouvoir d'appréciation discrétionnaire pour délivrer le permis, ce qui ne garantit pas la protection de l'élément du paysage concerné.

C. En droit des établissements classés

La *police des établissements classés* assure un contrôle sur les établissements susceptibles de causer des nuisances à l'environnement en les soumettant à autorisation. De nombreuses activités à risque pour le réseau écologique (mais aussi des activités de gestion du milieu naturel) sont ainsi visées (captage d'eau, déversement d'eaux usées, élevage intensif, etc.). Toute demande de permis doit également être soumise à évaluation des incidences, laquelle doit tenir compte de l'impact sur le milieu naturel et Natura 2000 et parfois sur le réseau écologique en général. A nouveau, l'autorité garde, sauf règle contraire, un pouvoir d'appréciation pour délivrer le permis, même s'il porte atteinte au réseau écologique. En *Région wallonne*, le régime de permis d'environnement et de permis unique est régi par le décret du 11 mars 1999 relatif au permis d'environnement[58] et ses arrêtés d'exécution (dont ceux du 4 juillet 2002[59]). En *Région flamande*, le « *milieuvergunning* » (autorisation écologique) est organisé par le décret du 28 juin 1985 relatif à l'autorisation anti-pollution intitulé le « *Milieudecreet* »[60] et ses arrêtés d'exécution (Arrêté du 6 février 1991 fixant le règlement flamand relatif à l'autorisation écologique -VLAREM I[61]- et arrêté du 1er juin 1995 fixant les dispositions générales et sectorielles en matière d'hygiène de l'environnement -VLAREM II[62]-).

D. En droit de la gestion et de la protection des ressources en eau

La gestion intégrée qualitative et quantitative de l'eau est primordiale pour la conservation de nombre d'écosystèmes humides et aquatiques[63]. L'objectif d'une gestion intégrée du cycle de l'eau, qu'a imposé la directive-cadre n°2000/60/CE établissant un cadre pour une politique communautaire dans le domaine de l'eau, est repris, en *Région Wallonne*, dans le décret du 15 avril 1999 relatif au cycle de l'eau et instituant une Société Publique de Gestion de l'Eau (SPGE)[64]. Il prévoit l'adoption d'un « *programme d'action pour la qualité des eaux* » et des plans de gestion pour chaque bassin et sous-bassin. Cette matière a fait l'objet d'une codification depuis 2004 (Code de l'eau[65]), qui entrera en vigueur en mars 2005. En *Région Flamande*, le décret du 18 juillet 2003 relatif à la politique intégrée de l'eau[66] constitue le siège de la matière. Il met en place un régime ambitieux de gestion intégrée des ressources en eau.

La gestion hydraulique (« quantitative ») des cours d'eau (c'est-à-dire du bon écoulement de l'eau – curage, travaux de rectification, etc. – et des inondations – digues, …) est actuellement séparée de la gestion de la qualité des eaux. Elle est régie par la loi du 28 décembre 1967 relative aux cours d'eau

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[55] Art. 84, §1er du CWATUP. Sur la police des permis urbanistiques, voy. HAUMONT, 1996 ; LOUVEAUX, 1999 ; BARLET et VAN REYBROECK, 2003; LETELLIER, 2003.

[56] Art. 99, §1er du décret du 18 mai 1999 portant organisation de l'aménagement du territoire (M.B., 5/06/99) ; voy. VAN DEN BERGHE, 1987/6.

[57] Il s'agit notamment des constructions et installations fixes, de la modification sensible du relief du sol (remblai de zones humides, etc.), du boisement et déboisement, du défrichement et la modification de la végétation de certains éléments comme les haies, les alignements d'arbres ou encore, en *Région wallonne*, les habitats naturels d'intérêt communautaire proposés mais non encore désignés comme sites Natura 2000.

[58] M.B., 08/06/99 ; sur ces permis, voy. BARNICH, L., BELLEFROID, M., DELNOY, M., HAENEN, V., 2002 ; BASTIN, J., 2002 ; ORBAN de XIVRY, E., in COLL., *Le décret wallon relatif au permis d'environnement*, 2000, pp. 411 et s.

[59] M.B., 21/09/02.

[60] M.B., 17/09/85. Voy. DE PUE et al., 2003.

[61] M.B., 26/06/91.

[62] M.B., 31/07/95.

[63] ORBAN de XIVRY et BORN, 2002.

[64] M.B., 22/06/99.

[65] Décret du 27 mai 2004 portant le livre II du Code de l'environnement contenant le Code de l'eau (M.B., 23/09/04).

[66] M.B., 14/11/03

non navigables[67] applicable *tant en Région wallonne* (jusqu'à l'entrée en vigueur du Code de l'eau, dans lequel elle est intégrée) *qu'en Région flamande*. Les régions, les provinces[68] et les communes se partagent la surveillance et la gestion des cours d'eau non navigables, en fonction de la catégorie dans laquelle sont classés leurs différents tronçons. Cette législation n'intègre guère les exigences de conservation de la nature (sauf via l'évaluation des incidences de certains permis). Le contrôle des captages d'eau est effectué par l'exigence d'une autorisation d'environnement dans les deux régions, laquelle, pour rappel, est soumise à évaluation des incidences.

La protection « qualitative » des eaux de surface contre la pollution[69] repose principalement, outre sur la planification, sur la définition d'objectifs de qualité (« *normes d'immission* ») des eaux de surface dans des zones données et sur une réglementation des rejets directs et indirects de substances polluantes dans les eaux par des « *normes d'émission* » et un système d'autorisation pour les rejets. La définition d'objectifs de qualité stricts dans les zones protégées devrait être une priorité, comme le prévoit la directive-cadre sur l'eau. En *Région wallonne*, le décret du 11 mars 1999 relatif au permis d'environnement[70] soumet le rejet d'eaux usées à permis d'environnement, et donc à certaines conditions sectorielles[71] et à évaluation des incidences. En *Région flamande*, la réglementation relative aux déversements d'eaux usées a été intégrée dans le VLAREM II (autorisation environnementale). Elle prévoit des normes d'émission. Le décret flamand du 18 juillet 2003 relatif à la politique intégrée de l'eau prévoit la réalisation d'une évaluation des incidences spécifiques sur l'eau (« *watertoets* »).

L'assainissement des eaux urbaines résiduaires et industrielles, indispensable pour lutter contre l'eutrophisation et l'acidification des écosystèmes aquatiques, fait l'objet de la réglementation en matière d'égouttage, de collecte et d'épuration des eaux usées. Il peut cependant causer des dommages importants au milieu naturel en implantant des infrastructures dans des zones biologiquement riches (stations d'épuration et collecteurs en fond de vallée, etc.). En *Région wallonne*, l'assainissement public de l'eau usée est essentiellement régi par le décret du 7 octobre 1985 sur la protection des eaux de surface contre la pollution et par le règlement général d'assainissement[72], aujourd'hui tous deux intégrés dans le Code de l'eau. La mise en place de l'égouttage est réalisée par les communes par (sous-)bassin hydrographique[73], tandis que la collecte des eaux d'égouts et leur traitement sont réalisés par des organismes d'épuration agréés érigés en intercommunales et ce en exécution d'un contrat de service conclu avec la Société publique de gestion de l'eau (SPGE). En *Région flamande*, le décret du 18 juillet 2003 relatif à la politique intégrée de l'eau règle cette matière. Les travaux d'égouttage sont réalisés par la commune qui bénéficie de subsides régionaux pour ce faire.

E. En droit de l'agriculture

La police de l'espace rural[74], réglée par le Code rural, ne comprend que peu de dispositions d'intérêt pour le réseau écologique (distance de plantation, etc). Ainsi, les activités agricoles ne sont que peu réglementées par des mesures contraignantes, sauf en ce qui concerne la nécessité d'obtenir des permis d'environnement pour certaines installations ou activités, dont l'arrachage et la destruction de petits éléments du paysage, les rejets d'eaux usées ou les captages d'eau. La gestion de l'azote et l'utilisation

[67] M.B., 15/02/68.
[68] En Région wallonne, les provinces ont perdu leurs compétences au profit de la Région en 2004.
[69] Cette matière est régie par un nombre considérable de dispositions de nature législative et réglementaire. Le principal texte applicable en Région wallonne dans l'attente de l'entrée en vigueur du Code de l'eau – qui codifie l'ensemble des règles existantes – est le décret du 7 octobre 1985 sur la protection des eaux de surface contre la pollution (M.B., 10/01/86). La législation wallonne en matière de protection des eaux de surface est dès lors très complexe à mettre en œuvre de manière cohérente, malgré l'adoption du décret relatif au cycle de l'eau du 15 avril 1999 (censé rationaliser l'ensemble du secteur). Le nouveau Code de l'eau devrait améliorer cette situation.
[70] M.B., 08/06/99.
[71] Les arrêtés du Gouvernement wallon fixant les conditions sectorielles des établissements classés pour chaque type de secteur contiennent des normes d'émission susceptibles d'avoir un impact sur le réseau écologique. Elles ne sont cependant pas suffisamment strictes pour certains types d'écosystèmes ou certaines espèces.
[72] Art.33 du décret wallon du 7 octobre 1985 ; Arrêté du Gouvernement wallon du 22 mai 2003 relatif au règlement général d'assainissement des eaux urbaines résiduaires (MB, 10/7/2003).
[73] Les actuels plans communaux généraux d'égouttage (PCGE) sont destinés à être remplacés par les plans d'assainissement par sous-bassin hydrographique (PASH)).
[74] Voy. PAEME et al., 1998 ; PARADIS et HEYERICK, 1995.

de pesticides font l'objet de réglementations distinctes[75], généralement trop peu restrictives pour lutter efficacement contre l'eutrophisation et les excès de pesticides. Pour le surplus, l'action des pouvoirs publics, de nature incitative, se concentre autour, d'une part, du plan de développement rural, cofinancé par la Communauté européenne dans le cadre de la PAC et qui inclut les mesures agri-environnementales (MAE)[76] et, d'autre part, de l'écoconditionnalité (« *cross-compliance* »[77]), qui permet de soumettre l'octroi d'aides agricoles au respect de standards environnementaux.

F. En droit forestier

Le régime juridique de la gestion des bois et forêts en Belgique est principalement fonction de leur propriétaire. La plupart des bois appartenant à une personne de droit public (Région, commune) est soumise à un régime particulier de gestion – le « *régime forestier* » - par l'administration régionale. La forêt privée est gérée directement par le propriétaire, le cas échéant dans le respect des règles édictées par le législateur et le Gouvernement. Des dispositions de police (notamment sur la circulation) couvrent cependant l'ensemble des bois et forêts. En *Région Wallonne*, le Code forestier[78] demeure applicable aux bois soumis au régime forestier[79]. Cette législation a pour principal objet de réglementer l'exploitation du bois dans les forêts publiques et de régler la police en forêt, y compris la circulation. Le régime forestier repose fondamentalement sur un plan d'exploitation par massif appelé « *aménagement forestier* ». Depuis 1997, la fonction écologique de la forêt est également prise en compte[80]. Une circulaire « *biodiversité en forêt* » est en cours de rédaction. Dans les bois et forêts privés, aucune règle de gestion (plan de gestion, etc.) n'est obligatoire. En *Région flamande*, la plupart des dispositions du Code forestier ont été abrogées et remplacées par le « *Bosdecreet* » du 13 juin 1990[81], révisé en 1999, qui s'applique aussi bien aux forêts domaniales qu'aux bois privés, y compris les parcs. La législation flamande est beaucoup plus élaborée et favorable à une gestion durable des forêts que le Code forestier, y compris en bois privé. Des liens exprès avec le décret sur la conservation de la nature, y compris le VEN[82], sont établis, tandis que des exemptions fiscales importantes sont prévues.

3.2.4. L'intégration du réseau écologique dans son contexte environnemental et humain

Le réseau écologique ne constitue pas un système fermé, mais s'inscrit nécessairement dans une matrice paysagère globale où s'exercent des activités qui l'influencent directement et indirectement. L'intégration des objectifs du réseau écologique dans tous les secteurs d'activités pertinents – urbanisme, agriculture, chasse et pêche, eau, industrie, tourisme, énergie, transports, etc. – est dès lors indispensable, si l'on veut assurer son efficacité. Le rôle de la planification du réseau écologique est ici fondamental en ce qu'elle devrait identifier les problèmes. En droit de l'environnement, le *principe d'intégration*, aujourd'hui consacré tant en droit européen que dans les deux régions[83], prescrit aux autorités de prendre en considération l'environnement – et donc la conservation de la nature – dans l'élaboration et la mise en œuvre de leurs politiques sectorielles. Cette intégration peut s'opérer par des mesures transversales (évaluation des incidences et participation), par des mesures spécifiques prises dans chaque législation correspondante (par ex. l'écoconditionnalité en agriculture) ou par des mesures visant à améliorer la coordination et l'articulation entre les dispositions spécifiques à la conservation de la nature et les autres législations susceptibles d'interférer avec elles.

[75] Voy. Arrêté du Gouvernement wallon du 10 octobre 2002 relatif à la gestion durable de l'azote en agriculture (M.B., 29/11/02) ; Décret du 23 janvier 1991 relatif à la protection de l'environnement contre la pollution due aux engrais, dit « *Mestdecreet* » (M.B., 23/7/90) ; loi du 11 juillet 1969 relative aux pesticides et aux matières premières pour l'agriculture, l'horticulture, la sylviculture et l'élevage.
[76] Arrêté du Gouvernement wallon du 28 octobre 2004 relatif à l'octroi de subventions agri-environnementales (M.B., 29/12/04); Arrêté du Gouvernement flamand du 3 octobre 2003 relatif à l'octroi de subventions en vue de l'application de méthodes de production agricole respectueuses de l'environnement et à la préservation de la diversité génétique (M.B., 23/10/03).
[77] Sur cette question, voy. DWYER et al., 2000; de SADELEER et BORN, 2004.
[78] Loi du 18 décembre 1854 contenant le Code forestier (M.B., 22/12/1854).
[79] Sur ces questions, voy. ORBAN de XIVRY, in COLL., *Guide de droit immobilier* ; ORBAN de XIVRY, 1999.
[80] Sont encouragés par exemple par des subventions la gestion en futaie mélangée et jardinée, la régénération naturelle,…
[81] M.B., 28/09/90. Sur la gestion privée des forêts, VAN HOORICK, 1995-1996, pp. 625-634.
[82] STRYCKERS, 1999, pp.183-189.
[83] Art. 1.2.1. du décret flamand du 5 avril 1995 contenant des dispositions générales concernant la politique de l'environnement (M.B., 03/06/95); Code de l'environnement, Livre 1, art.3.

3.3. Case study: implementation of an ecological network in the Dyle Valley from the legal point of view

3.3.1. Methods

L'étude juridique a été réalisée en 3 étapes, 3 Work package (WP). Il y a lieu de relever que la méthodologie initialement fixée a connu des remaniements en fonction des questions et difficultés rencontrées en cours d'étude.

3.3.1.1. WP 1 : Analyse du Cadre juridique théorique

La première étape de l'étude a consisté nécessairement en un rappel du cadre juridique général de l'utilisation du sol et en un inventaire et une synthèse des principaux instruments juridiques nécessaires à la mise en place d'un réseau écologique, dans les différentes législations pertinentes (conservation de la nature, aménagement du territoire, eau, agriculture, forêt, …) (voir 3.2).

3.3.1.2. WP 2 : Cas pratique

A. Objet de l'étude

La deuxième étape a consisté en l'analyse, sous l'angle de la faisabilité juridique, d'un cas particulier de restauration d'un réseau écologique dans une zone transfrontalière relativement peu étendue (la vallée de la Dyle de Wavre à Louvain) en vue de déterminer sa faisabilité juridique. Elle a consisté à examiner si, sur le plan juridique, les mesures écologiques de protection, de gestion et de restauration proposées par l'ECO-team (scénario optimal écologique) pouvaient être mises en œuvre par des instruments juridiques existants et, dans l'affirmative, sous quelles conditions, compte tenu de la situation de fait et de droit sur le terrain. Des propositions ont été faites pour suggérer l'application de certains instruments juridiques à cette fin. Compte tenu de la taille de la zone d'étude, plusieurs sites échantillons ont été sélectionnés en fonction de leur représentativité des problèmes juridiques, sociologiques et économiques susceptibles de se présenter dans la zone d'étude. Le choix, qui s'était d'abord porté sur quatre sites, a été ramené à trois sites légèrement modifiés.

B. Etapes suivies dans l'analyse du cas pratique

Pour proposer un cadre juridique à la mise en œuvre du scénario écologique proposé par l'ECO-team (qui correspond à une planification scientifique du réseau, avec cartes du réseau écologique et mesures de conservation), nous avons procédé comme suit :

WP 2.1. : Analyse du scénario optimal proposé par l'ECO-team sous l'angle juridique
Le legal team a pris connaissance du scénario écologique optimal, comprenant les 8 cartes du réseau optimal (globale et par type d'habitat) et la liste des mesures de conservation et de restauration proposées pour le réaliser. Afin de pouvoir évaluer les nombreuses mesures proposées, il a fallu les classer en grandes catégories (protection, gestion, restauration) en fonction des législations qui les encadrent, de façon à pouvoir évaluer les possibilités juridiques existantes pour les mettre en œuvre (WP 2.2.). Les cartes du réseau ont permis de déterminer le champ d'application et le degré de priorité des mesures de conservation proposées.

WP 2.2.: Analyse de la faisabilité juridique théorique des mesures de conservation proposées par les écologues
On a d'abord vérifié que l'adoption des mesures de conservation proposées était possible en droit et à quelles conditions, c'est-à-dire testé la *faisabilité juridique théorique* du scénario proposé par l'ECO-team. Les instruments juridiques existants ont donc été évalués au moyen d'un tableau, sous l'angle de leur potentialités pour assurer la mise en œuvre des mesures de conservation proposées. Le tableau comporte en ordonnée les types d'instruments et en abscisse les types de mesures (P, G, R), chaque croisement indiquant par une croix (et un commentaire le cas échéant) si l'instrument peut être utilisé pour mettre en œuvre ladite mesure. Les instruments d'application volontaire ont été indiqués en

italiques. Ceci a permis d'en tirer des conclusions sommaires sur le caractère adéquat des instruments existants pour mettre en œuvre un réseau écologique. Toutefois, il est important de signaler que les mécanismes de contrôle et de sanction du non respect de ces instruments ne sont pas envisagés ici, étant considérés comme systématiquement appliqués, ce qui n'est pas le cas en réalité.

WP 2.3.: Analyse de la faisabilité juridique concrète des mesures de conservation proposées dans les sites échantillons – propositions juridiques de mise en œuvre
La mise en place du réseau écologique s'inscrivant dans une situation juridique existante très complexe, une analyse de la situation juridique existante dans les différents domaines pertinents (conservation de la nature, aménagement du territoire, eau,…) s'est avérée indispensable pour évaluer la *faisabilité concrète* (et plus seulement théorique) des mesures de conservation proposées. Sur base de cette analyse, il a été possible de faire des propositions pour encadrer juridiquement le scénario proposé par l'ECO-team. L'analyse étant impossible à l'échelle de l'ensemble de la zone, il a été décidé de sélectionner quatre, puis trois sites échantillons, un à cheval sur les deux régions (site I) et deux en Région flamande (sites II et III), en fonction de la diversité des situations qu'ils représentaient sur les plans juridique, social et économique. Deux phases se sont succédées :

- <u>Analyse de la situation juridique existante dans la zone d'étude et les sites échantillons</u>

Une analyse de la situation juridique existante dans la zone d'étude et dans chaque site échantillon a été réalisée. Le legal team a obtenu des données juridiques du Social team et les a complétées en prenant contact avec diverses personnes ressources. Une liste des instruments en vigueur dans la zone d'étude ainsi que des cartes de la situation juridique ont été réalisées avec l'aide de l'ECO-team (**appendice II**).

- <u>Evaluation de la faisabilité juridique des mesures proposées dans les sites échantillons</u>

Compte tenu des instruments disponibles (WP 2.2.) et de la situation juridique existante dans les sites échantillons (WP 2.3. A), on a évalué dans quelle mesure les mesures de conservation proposées pouvaient être réalisées concrètement. Lorsque la situation existante (aire protégée existante, zonage au plan de secteur,…) ne le permettait pas déjà, on a fait diverses propositions juridiques de mise en œuvre des mesures de conservation. Compte tenu du temps disponible, seules des propositions sommaires ont pu être faites. Elles ne visent que ce qui est directement nécessaire à la réalisation des objectifs ECONET, *sans passer par l'étape (pourtant capitale) de planification,* censée déjà effectuée au travers des travaux de l'ECO-team. La coordination des mesures (notamment avec l'aménagement du territoire, l'agriculture, etc.) n'a été envisagée que lorsque c'était strictement nécessaire[84]. Les propositions reposent sur plusieurs postulats et sont fondées sur plusieurs critères mentionnés en annexe (voir appendice II). Ainsi, les mesures d'application volontaire ont été proposées en priorité. Cette dernière phase a permis de dégager des conclusions sommaires sur la faisabilité juridique concrète de la mise en place du réseau dans les différents sites échantillons, moyennant la mise en œuvre des propositions juridiques.

3.3.2. Results

3.3.2.1. WP 2.1. : Analyse du scénario écologique optimal sous l'angle juridique

Les mesures de conservation proposées par les écologues ont été classées en catégories plus générales pour pouvoir être analysées plus facilement sur le plan juridique, à savoir : a) *protection contre les détériorations et perturbations et contrôle d'activités et processus néfastes*[85] ; b) *gestion du milieu biotique et abiotique*[86] ; c) *restauration des milieux et des populations*[87]. Les cartes du réseau

[84] Dans la réalité, de telles mesures de coordination devraient être envisagée dans une planification appropriée de la mise en place du réseau ECONET, qui combine l'aménagement du territoire, la conservation de la nature et la gestion du milieu physique (eau, azote) aux échelles appropriées.
[85] A savoir : protection contre les (contrôle des) atteintes directes au site (P DIR) ; protection contre les (contrôle des) atteintes en provenance de l'extérieur du site autres que via la gestion de l'eau (P EXT) ; protection contre les dépôts de déchets (P DECH) ; protection de la qualité de l'eau dans le site (y compris par des mesures hors site) (P QLE) ; contrôle des activités agricoles et sylvicoles autour et dans le site (C AGR) ; contrôle des populations d'espèces à risque (poissons – gibiers – espèces invasives) (C ESP).
[86] Gestion de la végétation ouverte existante (y compris l'élevage) (G HERB) ; gestion forestière (G FOR) ; gestion du milieu physique autre que l'eau (relief, sol) (G SOL) ; gestion de la quantité d'eau sur le site (gestion hydraulique et/ou hydrogéologique) (G HYDR).

écologique (indiquant les priorités de conservation – zone centrale, de développement, de liaison – pour chaque type de biotope cible) faites par l'ECO-team ont été utilisées pour évaluer la faisabilité juridique concrète des mesures dans les sites échantillons.

3.3.2.2. WP 2.2.: Analyse de la faisabilité juridique théorique des mesures de conservation proposées par les écologues

Dans le cadre du présent rapport, nous présentons seulement, à titre d'illustration, les résultats concernant quelques instruments juridiques spécifiques à la conservation de la nature qui fournissent le cadre juridique le plus adéquat et le mieux adapté à la plupart des mesures de conservation proposées. La planification elle-même n'a pas été envisagée, faute de temps. Elle est considérée comme acquise au travers des travaux de l'ECO-team, qui sont une forme de planification. Il importe de distinguer les deux régions. Il est renvoyé au cadre théorique pour des références légales.

En Région wallonne, le statut de **réserves naturelles (agréées ou domaniales)** et, dans une moindre mesure, de **réserve forestière**, permet de mettre en œuvre les protections contre les atteintes directes au site (P DIR) et contre les dépôts de déchets (P DECH). La protection contre les atteintes extérieures (P EXT) et le contrôle des activités agricoles et sylvicoles autour et dans le site (C AGR) est plus difficile. Elle est assurée en zone agricole et en zone forestière pour certains projets soumis à permis d'urbanisme[88], dans les sites Natura 2000 (via son régime de protection) et par la prise en compte de ces aires protégées dans l'évaluation des incidences sur l'environnement des projets, installations et activités classées. La protection de la qualité de l'eau dans le site (P QLE) n'y est pas assurée, sauf, le cas échéant, pour les réserves naturelles situées dans un site Natura 2000. Le plan de gestion de la réserve naturelle permet *a priori* de couvrir la plupart des mesures de gestion (G HERB, G FOR, G HYDR, G SOL) et de restauration (R HERB, R FOR, R AQU, R LIN) proposées par l'ECO-team. Pour les réserves privées, des subventions sont prévues, sous réserve des budgets régionaux disponibles bien entendu[89]. Le statut de **zones humides d'intérêt biologique** impose un régime de protection assez strict (P DIR) par le biais d'interdictions générales (interdictions de chasser ou capturer des animaux, de détruire ou endommager la végétation ou les plantes, d'utiliser des boues résiduaires, …) et d'interdictions particulières fixées par les arrêtés de désignation. La protection contre les atteintes extérieures (P EXT) et le contrôle des activités agricoles et sylvicoles autour et dans le site (C AGR) y sont assurées dans les mêmes limites que pour les réserves naturelles. Le plan de gestion éventuellement prévu par l'arrêté de désignation de la ZHIB permet de couvrir la plupart des mesures de gestion (G HERB, G FOR, G HYDR, G SOL) et de restauration (R HERB, R FOR, R AQU, R LIN) proposées. Il n'est en revanche pas prévu expressément de subventions, celles-ci étant fixées, le cas échéant par convention.

Le statut de **site Natura 2000** permet de prendre diverses mesures adaptées aux exigences des biotopes cibles. Il faut toutefois noter d'emblée qu'en principe, ce régime ne vise que des habitats et espèces d'intérêt communautaire, lesquels pourraient ne pas correspondre avec les biotopes cibles ECONET. Des incompatibilités sont donc possibles. En outre, le régime n'est applicable dans son entièreté que si l'arrêté de désignation a été adopté. La protection contre les atteintes directes au site (P DIR) y est assurée par le régime préventif qui comporte l'interdiction générale de détériorer les habitats et de perturber significativement les espèces pour lesquelles le site a été désigné ainsi que les interdictions particulières fixées par les arrêtés de désignation. La protection contre les atteintes extérieures (P EXT) et le contrôle de certaines activités agricoles soumises à permis (modification du relief du sol, certains déversements d'eaux usées, grands élevage) autour des parcelles reprises en zone centrale (C AGR) sont assurées par le régime Natura 2000 grâce, d'une part, aux interdictions particulières figurant dans l'arrêté de désignation (qui peuvent viser l'extérieur du site) et, d'autre part,

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[87] Restauration de nouveaux habitats aquatiques (R AQU) ; restauration de nouveaux habitats ouverts non aquatiques (R HERB) ; restauration de milieux forestiers (R FOR) ; restauration / recréation d'éléments linéaires et ponctuels du paysage (maillage écologique) (R LIN).
[88] Art. 452/35 et 452/42 du CWATUP.
[89] Ainsi par exemple, bien que la gestion des réserves forestières puisse être en théorie subsidiée par le Plan de développement rural, la mesure n'a pas été actionnée en 2004.

au mécanisme d'évaluation appropriée des incidences des plans et projets à risque : quel que soit le biotope concerné, aucune autorisation ne peut être octroyée par une autorité (permis d'urbanisme, d'environnement,…) si le projet demandé risque de porter atteinte à l'intégrité du site Natura 2000. Les autres mesures de protection (P DECH, P QLE) ou de contrôle (C ESP) sont également couvertes par le régime Natura 2000 pour autant qu'il y ait détérioration des habitats ou perturbations significatives des espèces pour lesquelles le site a été désigné ou pour autant que ces mesures soient indiquées dans l'arrêté de désignation. La gestion active prévue dans le site Natura 2000 permet de couvrir la plupart des mesures de gestion (G HERB, G FOR, G HYDR, G SOL) et de restauration (R HERB, R FOR, R AQU, R LIN) proposée par l'ECO-team. Ce régime de gestion active doit ainsi permettre d'atteindre les objectifs de gestion active précisés dans l'arrêté de désignation, grâce à des moyens choisis en concertation avec les propriétaires et occupants (contrat de gestion active). Le montant des subventions pour la gestion n'est toutefois pas connu, mais devrait couvrir les mesures. Les compensations fiscales devraient couvrir au moins en partie les moins-values éventuelles, à défaut d'un mécanisme d'indemnisation spécifique.

En Région flamande, le **plan directeur de la nature** (« *Natuurrichtplan* »), instrument assez proche de l'arrêté de désignation de site Natura 2000 en Région wallonne, est obligatoire dans les zones VEN et de l'IVON, et dans les sites Natura 2000. Il permet de planifier et de rendre obligatoire la plupart des mesures de protection (P DIR, P EXT, P DEC, P QLE), de contrôle (C AGR, C ESP), de gestion (G HERB, G FOR, G HYDR, G SOL) et de restauration (R HERB, R FOR, R AQU, R LIN) proposées par les écologues. Comme en Région wallonne, les statuts de **réserve naturelle (agréée ou flamande)** (« *erkende* » ou « *Vlaamse natuurreservaat* ») et de **réserve forestière** (« *bos reservaat* ») permettent également de couvrir la plupart des mesures proposées par le biais notamment du plan de gestion, subventionné dans les réserves privées.

Les statuts de **Zone de protection spéciale Natura 2000** (« *Speciale Beschermingszone SBZ-V (vogels) et SBZ-H (habitats)* ») et de **GEN** (« *Groot Eenheid Natuur* ») dans le VEN (« *Vlaamse Ecologische Netwerk* ») prévoient des régimes de protection (P), de gestion (G) et de restauration (R) suffisamment souples pour inclure la plupart des mesures proposées par les écologues, y compris concernant le régime hydrique. Toutefois, il importe que le « *Natuurrichtplan* » de la zone soit adopté et qu'il prévoie d'intégrer ces mesures pour qu'elles soient envisageables. Le régime n'exclut pas la prise en compte d'autres espèces et d'habitats que les espèces et habitats d'intérêt communautaire (contrairement à ce qui est le cas en Région wallonne). L'instrument foncier de gestion et de restauration dénommé **projet d'aménagement de la nature** (« *Natuurinrichtingsproject* ») permet de prendre des mesures plus radicales – mais coûteuses – en vue de restaurer la structure écologique dans le paysage, en jouant sur la structure foncière du site. L'ECO-team n'a pas proposé de mesures concrètes de déplacement d'entreprises ou de modification du réseau routier, bien qu'elles pourraient améliorer sensiblement la structure écologique de l'ensemble.

3.3.2.3. WP 2.3.: Analyse de la faisabilité juridique concrète des mesures de conservation proposées dans les sites échantillons – propositions juridiques de mise en œuvre

A. Résumé de la situation juridique existante de la zone d'étude

Seuls les éléments-clés de la situation juridique existante pour la mise en œuvre des mesures de conservation sont ici décrits. Pour le surplus il est renvoyé à l'annexe (**Appendice II**).

1. Situation administrative générale

La zone d'étude de la vallée de la Dyle se situe au sud de Louvain et au nord de Wavre sur un territoire transfrontalier Région flamande/Région wallonne. Les conséquences juridiques de cette situation sont considérables, dès lors que les compétences pour légiférer en matière d'utilisation des sols sont partagées entre les régions. Les instruments juridiques diffèrent donc d'une région à l'autre, tandis que le ressort des autorités compétentes pour gérer les ressources est partagé dans la vallée.

2. Situation foncière – structure de la propriété (publique – privée)

Les données relatives au caractère public ou privé de la propriété ont permis de déterminer la structure de la propriété. Celle-ci est cartographiée en **carte 1**. De cette structure dépend en partie l'applicabilité (sauf mesure foncière) des instruments d'application volontaire, en considérant que les terrains appartenant aux régions peuvent d'office entrer dans le réseau ECONET. En dehors des aires protégées appartenant aux régions (par ex. la réserve naturelle flamande des « *Vijvers van Florival* » ou la réserve naturelle domaniale du Bouli) ou à des asbl privées (ex. la réserve naturelle agréée « *De Doode Bemde* » de l'asbl « *Natuurpunt* »), la majorité des parcelles appartient à des particuliers ou à des personnes morales de droit public autres que les régions. Cette situation implique donc soit d'imposer des contraintes aux propriétaires (publics ou privés) soit de les convaincre de mettre en œuvre volontairement les mesures, soit encore de procéder à l'acquisition de terrains.

3. Conservation de la nature

Il ressort de l'analyse que la zone d'étude est en majeure partie couverte par des périmètres d'aires protégées, en particulier dans la partie flamande de la zone (**carte 2**). Ceci constitue une situation particulière dans la mesure où elle n'est pas nécessairement représentative de la situation juridique existante de l'espace rural en général. Les conclusions en termes de faisabilité juridique concrète pourraient donc difficilement être extrapolées à n'importe quelle situation en Belgique.

En Région wallonne, les sites « *Vallée de la Dyle en aval d'Archennes* » et « *Vallée de la Nethen* » sont candidats au réseau Natura 2000. Deux réserves naturelles sont présentes (1 réserve naturelle privée[90] (« *Laurensart-Grez-Doiceau* ») ; 1 réserve naturelle domaniale (« *Bois du Bouly-Archennes* »)). En *Région flamande*, l'entièreté de la vallée de la Dyle, « *De Dijlevallei* », a été désignée comme « *Speciale beschermingszones van Vogelrichtlijngebied (SBZ-V)* ». En 2001, le site « *Valleien van Dijle , Laan en Ijse met aangrenzende bos- en moerasgebieden* » a été proposé comme site candidat « *Speciale beschermingszones van Habitatrichtlijngebied (SBZ-H)* » au réseau Natura 2000. L'entièreté de la partie flamande de la zone d'étude se situe également dans le VEN et, plus particulièrement dans la GEN dénommée « *het Zoniënwoud, de Ijsevallei, de Dijlevallei, het Meerdaalwoud en de Molenbeek-Mollendaalbeek* ». Certains sites sont érigés en réserves naturelles ou forestières (4 réserves naturelles flamandes (« *Vijvers van Florival* » au sud, « *Vijvers van Oud-Heverlee* » au nord, « *Ijsevallei* » et « *Rodebos en Laanvallei* » à l'ouest) ; 1 réserve naturelle agréée (« *De Doode Bemde* ») ; 1 réserve forestière (« *Putten van de Ijzerenweg* »)). Le « *Natuurinrichtingsproject Dijlevallei ten Zuiden van Leuven*» (projet d'aménagement de la nature de la vallée de la Dyle) est actuellement au stade de l'approbation par le Gouvernement flamand. Le « *projectrapport* » formule des objectifs de conservation à atteindre et suggère des mesures et des modalités d'exécution avec une estimation financière. Signalons enfin, quant aux instruments de protection applicables en dehors des zones protégées, la présence d'espèces strictement ou partiellement protégées dont l'habitat ou certaines parties de celui-ci ne peuvent être détruits (par ex. : alyte, crapaud accoucheur, grenouilles verte et rousse, Vespertilion de Daubenton, Pipistrelle commune, Castor d'Europe, Hérisson, Perce-neige, Chabot,…).

4. Aménagement du territoire et patrimoine immobilier

En Région wallonne, la zone d'étude est affectée au **plan de secteur** (**carte 3**) à raison d'un peu plus de 50% en zones non destinées à l'urbanisation (à savoir en zone agricole, zone d'espaces verts et zone forestière). Aucune zone naturelle n'y est présente, ce qui contraste très fort avec la partie flamande de la zone, qui est majoritairement affectée en zone naturelle. Le reste de la partie wallonne de la zone d'étude est situé en diverses zones destinées à l'urbanisation, qui peuvent constituer des obstacles majeurs à la mise en place d'un réseau ECONET si des zones centrales y figurent. *En Flandre*, la zone d'étude se situe principalement en zone naturelle (« *natuurgebied* ») au plan de secteur, c'est-à-dire le zonage le plus protecteur. Ceci facilite grandement la mise en œuvre de mesures de protection et de gestion et évite de devoir envisager des révisions du plan longues et coûteuses. Une telle situation est cependant loin d'être représentative de la situation dans l'ensemble de la Flandre, où

[90] A notre connaissance, celle-ci n'est pas protégée par un statut officiel.

de nombreux sites d'intérêt biologique sont encore situés en zone urbanisable ou agricole. Les **sites classés** (**carte 2**) constituent des aires protégées sur base de certains critères esthétiques et scientifiques. Des sites classés sont présents tant dans la partie wallonne de la zone d'étude (par ex. le site du « *Marais de Laurensart* ») que dans sa partie flamande (« *Heverleebos en Meerdaalwoud* »).

5. Eau

La zone d'étude se situe dans le bassin hydrographique de l'Escaut et dans le sous-bassin hydrographique de la Dyle-Gette. Divers régimes juridiques sont applicables à la zone d'étude (**carte 4**). La **protection de la qualité de l'eau** n'est guère stricte dans la zone d'étude. *En Région wallonne*, seule une zone de protection est en vigueur, à savoir la zone d'eaux piscicoles cyprinicoles du Glabais et des ses affluents (jusqu'au confluent avec le Train), hors site échantillon. *En Région flamande*, certains cours d'eau non navigables de 1ère catégorie (gérés par la Région) sont soumis à des objectifs de qualité plus stricts (eau de consommation, eau piscicole).Pour le surplus, les normes de qualité de base applicables à toutes les eaux et les objectifs de qualité par type de substance dangereuse s'imposent aux pouvoirs publics dans les deux régions. La zone reçoit en amont les eaux épurées par la station d'épuration de Basse-Wavre, au moyen d'un traitement tertiaire, l'entièreté de la zone d'étude étant désignée comme « *zone sensible* » au sens de la directive européenne sur l'assainissement des eaux urbaines résiduaires. **Sur le plan quantitatif**, la zone d'étude comprend 4 captages d'eaux, 2 en Région wallonne (Pécrot et la Motte) et 2 en Région flamande (Korbeek-Dijle et Egenhoven). Des zones concentriques de protection ont été délimitées autour de ces captages, zones au sein desquelles les activités et installations sont réglementées. Il est possible que certains captages aient un impact pour les zones humides de la zone d'étude mais nous ne disposons pas de données. La **gestion hydraulique des cours d'eau** est partagée, entre les régions, les provinces et les communes, en fonction de la catégorie du cours d'eau. Cette gestion vise à faciliter l'écoulement des eaux, ce qui n'est guère favorable aux zones humides et aux berges, sauf mesures de gestion écologique.

6. Agriculture

La nappe des sables bruxelliens est désignée comme « **zone vulnérable** » à la pollution par les nitrates d'origine agricole tant en Région wallonne qu'en Région flamande. Ceci implique l'application de règles concernant les périodes durant lesquelles l'épandage de certains types de fertilisants est interdit, la capacité des cuves destinées au stockage des effluents d'élevage, ou encore la limitation de l'épandage des fertilisants. Nous ne disposons pas de données sur les éventuelles **mesures agri-environnementales** actuellement en cours dans la zone.

B. Evaluation de la faisabilité juridique des mesures proposées dans le site échantillon I

Pour des raisons d'espace, seul le site échantillon I est présenté ici (voy. l'**appendice II**).

Dans la partie wallonne du site échantillon n°1, la protection des biotopes existant et à restaurer situés en zones centrales est assurée principalement par la sélection du site comme **site Natura 2000**, qui couvre presque tout le site. L'arrêté de désignation définissant les interdictions particulières n'est pas encore adopté, mais son contenu a pu être approché dans un travail de fin d'étude sur la zone[91], qui préconise les mêmes mesures de protection que celles proposées par l'ECO-team, mais appliquées différemment dans l'espace. La gestion active prévue dans le site Natura 2000, d'après ce même travail, devrait correspondre en partie aux mesures de gestion et de restauration proposées par les écologues. Ces mesures pourraient être mises en œuvre, dans les parcelles privées, au moyen d'un contrat de gestion active (si accord du propriétaire) ou d'une expropriation (si désaccord du propriétaire) et, dans les parcelles publiques, au moyen d'une adaptation de la gestion actuelle via éventuellement l'adoption d'un plan de gestion (notamment dans la réserve naturelle du Bouli). Le problème est qu'actuellement l'arrêté de désignation n'est pas prêt, ce qui retarde l'entrée en vigueur de la protection et de la gestion.

[91] LEGAST, 2004.

L'affectation en zone d'espaces verts/naturelle au plan de secteur de la plupart des zones centrales est également une première protection contre l'urbanisation et contre les actes et travaux (soumis à permis) non destinés à la protection et à la régénération du milieu naturel. L'affectation de certaines zones centrales et de développement en zone urbanisable (zone de loisir, zone d'habitat, zone d'équipement communautaire) pourraient en théorie se révéler dangereuses (par ex. captages de l'étang de Pécrot). Toutefois, ceci n'est vrai que pour les parcelles non visées par le régime de protection du site Natura 2000, ce statut n'étant pas subordonné au respect du plan de secteur.

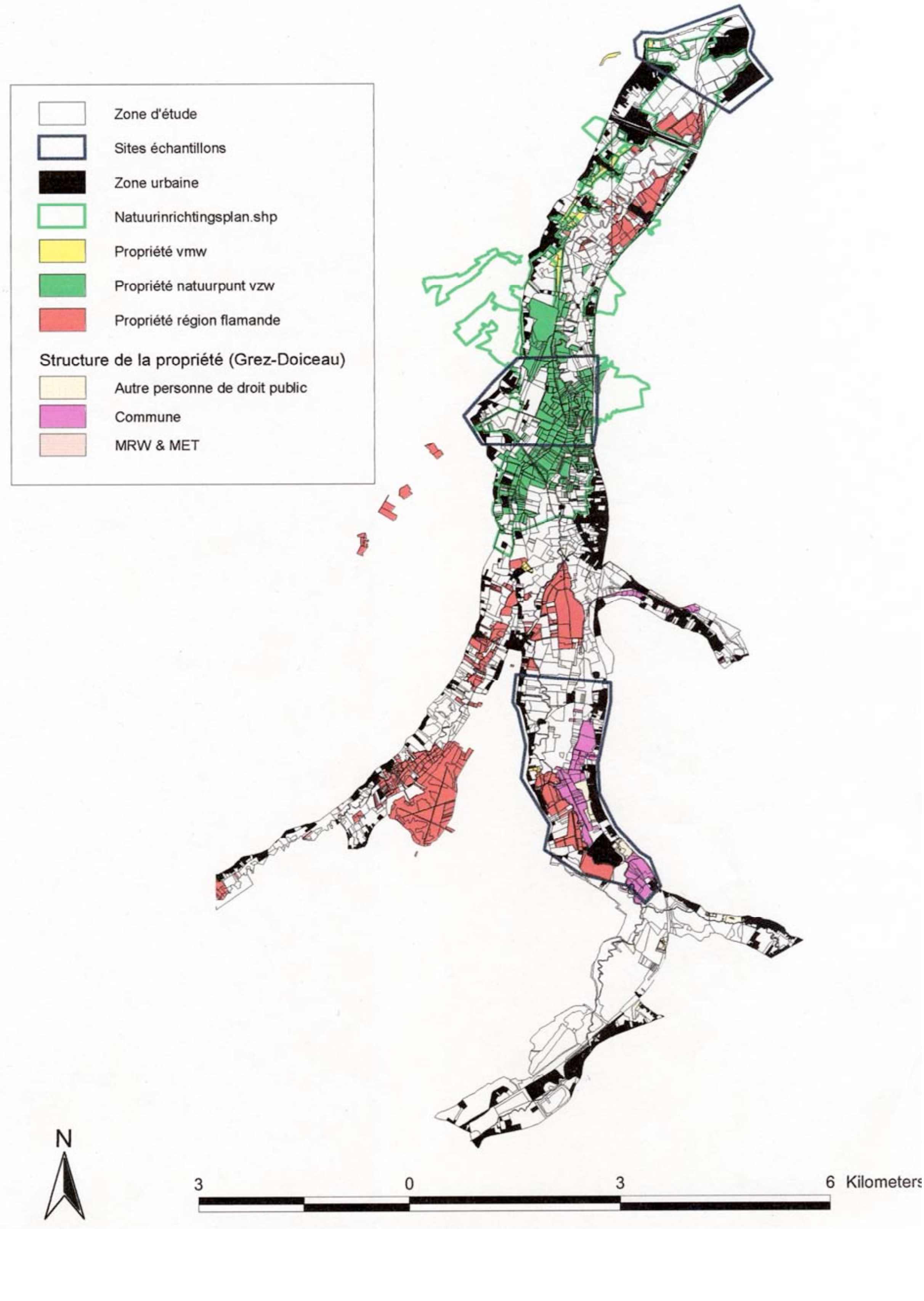
Pour les quelques zones centrales et de développement non protégées, c'est-à-dire non visées par le régime de protection du site Natura 2000, il est proposé, dans un ordre croissant de restrictions, une mise sous statut de **réserve naturelle** (domaniale ou agréée) moyennant l'accord du propriétaire et, en cas de désaccord de celui-ci, soit une **modification du plan de secteur** en vue de leur affectation en zone naturelle (mesure moins contraignante et ne couvrant pas toutes les mesures préconisées), soit la désignation unilatérale d'une **zone humide d'intérêt biologique**, soit, en dernière extrémité, une **expropriation** en vue de créer une réserve naturelle domaniale (mesure très contraignante, mais plus efficace et plus complète). La désignation d'un **périmètre d'incitation** autour du site Natura 2000 permettrait en outre au Gouvernement de prendre des mesures subventionnnées pour « *favoriser la gestion active du site Natura 2000* ». La conclusion de **contrats de gestion** avec des personnes publiques (commune) ou privées (entreprise, particulier) et l'engagement des agriculteurs concernés dans des **MAE** sont d'autres façons d'atteindre les objectifs fixés.

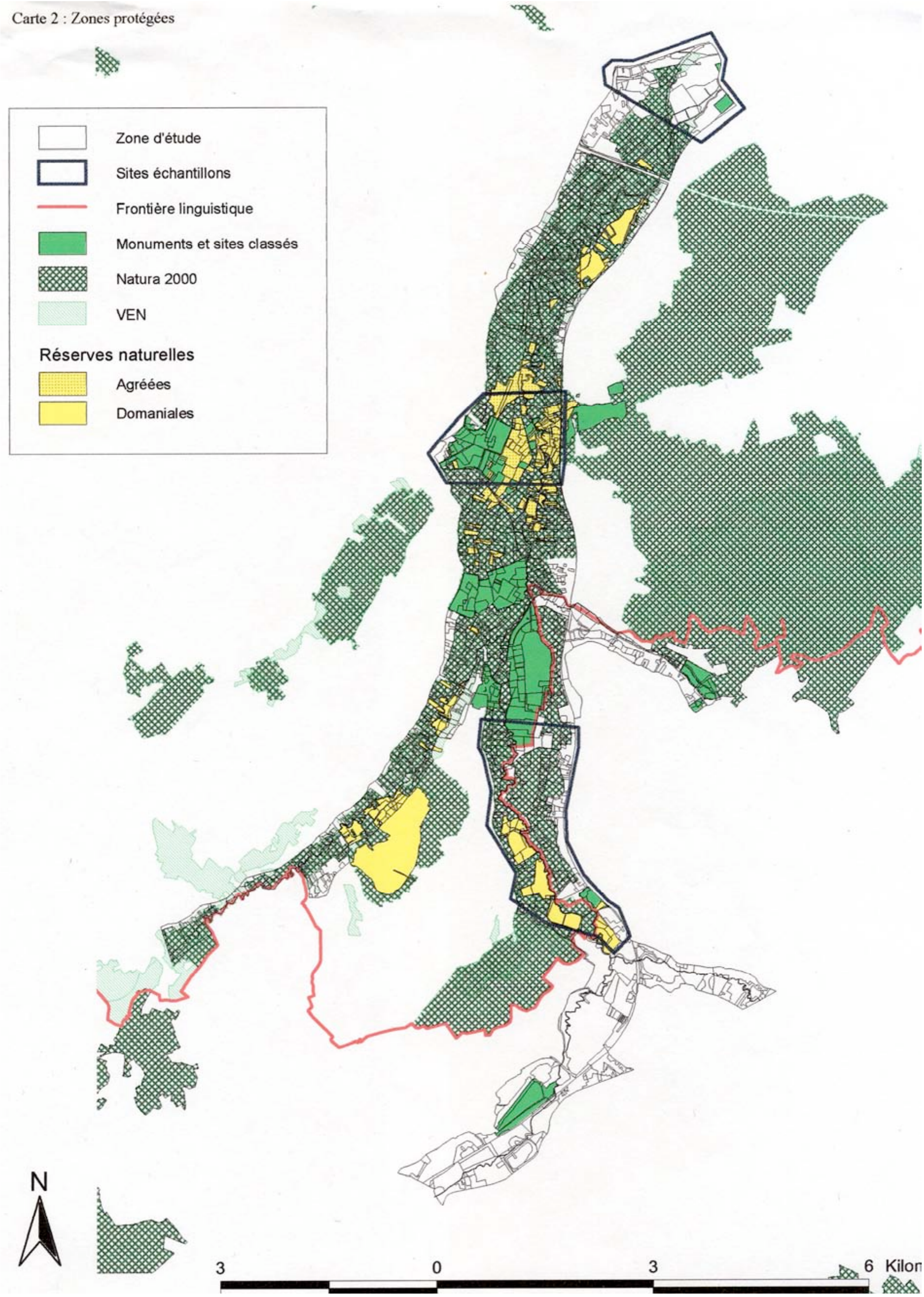
Dans la partie flamande du site échantillon n°1, la protection des biotopes existant et à restaurer situés en zone centrale est assurée principalement par les statuts de **zone de protection spéciale Natura 2000** et de **GEN** dans le VEN, qui couvrent presque tout le site, à condition toutefois que soit adopté le « *Natuurrichtplan* » qui devrait intégrer les mesures nécessaires à la préservation du site et proposé des moyens (incitatifs ou contraignants) pour les réaliser. Tant que ce « *Natuurrichtplan* » n'est pas adopté, des mesures provisoires de gestion pourraient être intégrées dans des **accords de gestion** conclus par la Région dans les biotopes constitutifs d'oiseaux de milieu ouvert, des **MAE** pourraient être activement promues dans le site pour réduire les intrants, une **zone d'extension de la réserve naturelle flamande** pourrait être désignée afin d'y permettre l'exercice du droit de préemption, une gestion écologique des bois pourrait être promue par des subventions, … Le **projet d'aménagement de la nature** (« *Natuurinrichtingsproject* ») de la vallée de la Dyle, qui englobe la majeure partie du site, permet de prendre des mesures plus radicales (déplacement d'entreprises ou modification du réseau routier) en vue de restaurer la structure écologique dans le paysage, en jouant sur la structure foncière du site. Les mesures prévues par l'ECO-team ne prévoient cependant pas de telles mesures.

L'affectation en zone verte au plan de secteur de la plupart des zones centrales, est également une première protection contre l'urbanisation et contre les actes et travaux (soumis à permis) non destinés à la protection et à la régénération du milieu naturel. Les deux législations étant coordonnées, il n'y a pas en principe d'incompatibilité entre le VEN et le plan de secteur.

En matière de protection de la **qualité de l'eau**, les mesures existantes s'avèrent notablement insuffisantes dans les deux régions. Des mesures spécifiques précises (création d'une zone de protection des eaux cyprinicoles, création d'un périmètre d'incitation autour du site Natura 2000, promotion des mesures agri-environnementales, adaptation des conditions d'exploiter des permis de captage consistant en une réduction du volume d'eau autorisé, adaptation des permis de déversement d'eaux usées industrielles et agricoles, finalisation des travaux d'égouttage, de collecte et de traitement des eaux résiduaires …) devraient être adoptées à l'échelle du bassin versant de la Dyle, en concertation avec les administrations concernées et dans le cadre d'une coopération interrégionale. Les nouveaux cadres juridiques mis en place en Flandre et en Wallonie depuis 2003 devraient permettre une amélioration et assurer la prise en compte à tout le moins des sites Natura 2000, conformément à ce qu'exige la directive-cadre sur l'eau.

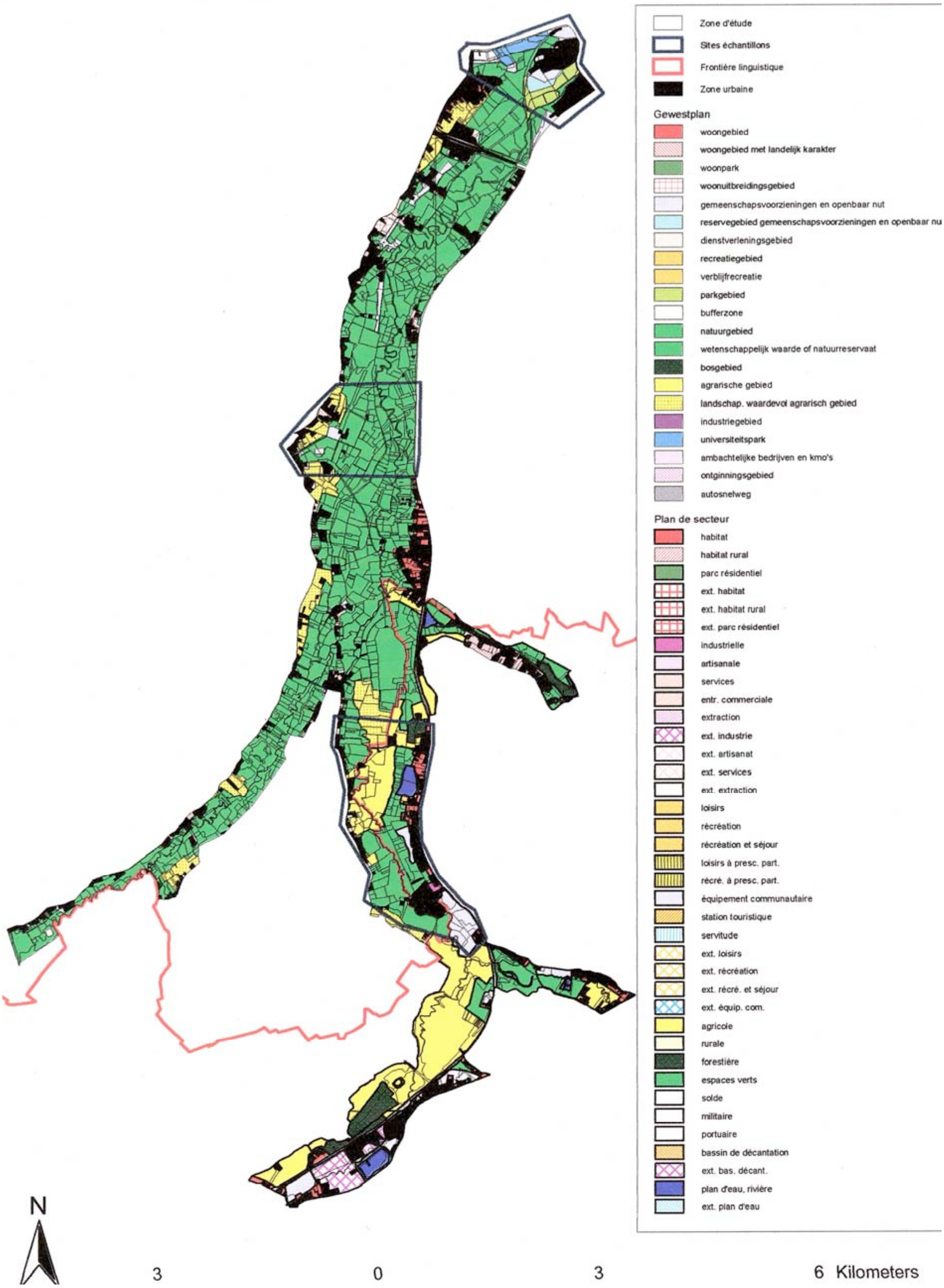
Carte 1 : Structure de la propriété



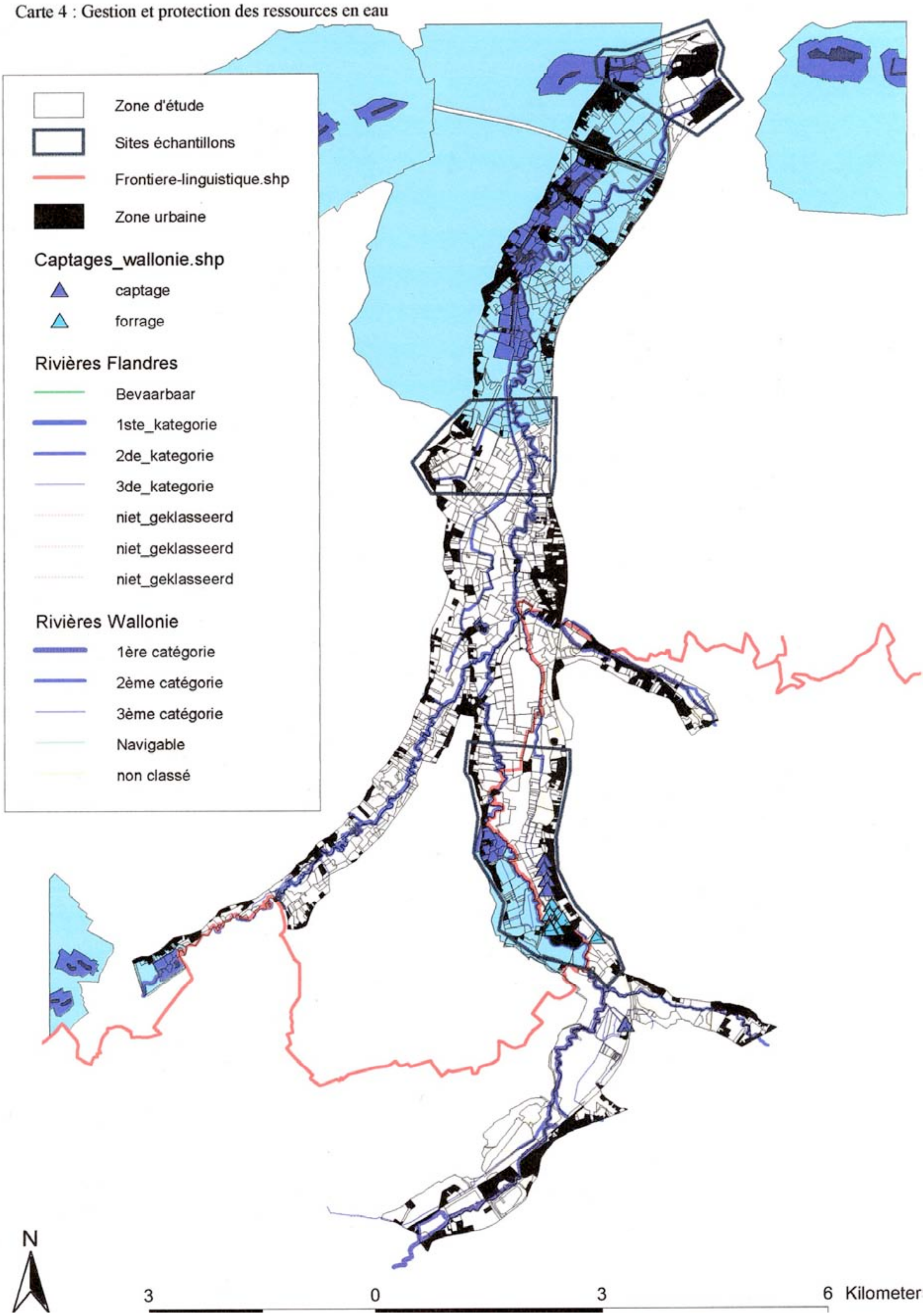


Carte 2 : Zones protégées
Zone d'étude
Sites échantillons
Frontière linguistique
Monuments et sites classés
Natura 2000
VEN
Réserves naturelles
Agréées
Domaniales
N
3
0
3
6 Kilon

Carte 3 : Affectation au plan de secteur



Carte 4 : Gestion et protection des ressources en eau



3.3.3. Discussion

Compte tenu des moyens juridiques proposés pour mettre en œuvre les mesures de conservation au sein des sites échantillons dont question ci-avant, des conclusions sur leur faisabilité juridique sont formulées.

3.3.3.1. Dans la partie wallonne du site échantillon n°1

De façon générale, le contexte juridique est raisonnablement favorable à la mise en place d'un réseau écologique, du fait principalement de l'inscription du site en site Natura 2000. Ce régime est très intéressant vu la structure de la propriété dans la zone car il permet de prendre diverses mesures adaptées aux exigences des biotopes cibles à l'échelle de la vallée, y compris à l'égard d'activités extérieures au site, et ce indépendamment de l'accord des propriétaires et occupants, ce qui aurait été impossible via les procédures de création de réserve naturelle ou forestière en cas de désaccord du propriétaire, sauf expropriation à grande échelle, sur l'ensemble du site. Il ne doit pas non plus respecter l'affectation au plan de secteur, alors que certaines affectations posent des difficultés (Pécrot). Deux problèmes majeurs se posent, à savoir, d'une part, le fait que l'adoption de l'arrêté de désignation est fortement retardée et, d'autre part, le fait que ce régime est strictement focalisé sur les espèces et habitats naturels d'intérêt communautaire et qu'il ne peut viser d'autres espèces ou habitats si ceux-ci n'ont pas les mêmes exigences que les premiers. Il se peut dès lors que ses objectifs entrent en contradiction avec les objectifs ECONET (par exemple la restauration en biotope de type « marsh » d'une forêt alluviale), auquel cas Natura 2000 a priorité. Il importe de prendre d'autres mesures juridiques de conservation (compatibles), comme la création de réserves naturelles agréées, si le régime Natura 2000 ne couvre pas les propositions ECONET[92]. En dehors du site Natura 2000 (et du périmètre d'incitation le cas échéant), les instruments sont plus limités. Sauf recours au classement comme site (peu spécifique à la conservation de la nature) ou à l'expropriation et hormis le cas particulier de la création d'une zone humide d'intérêt biologique[93], ils nécessitent généralement un accord du propriétaire (au moins informel) et ne couvrent que la superficie des parcelles pour lesquelles cet accord a été donné. Il manque aussi d'un instrument contractuel spécifique de gestion en faveur de la conservation de la nature en dehors des zones protégées et des terres agricoles, ainsi que d'un instrument foncier de type « remembrement » pour permettre des échanges de propriété de grande ampleur.

Le principal problème posé sur le plan juridique tient sans doute dans la protection de la qualité de l'eau et la gestion des apports (atmosphériques et autres) en azote, qui sont très difficiles à assurer pour un cours d'eau aussi important que la Dyle. *A priori*, seules des mesures intégrées et prises à l'échelle du bassin versant (combinant approche contraignante et volontaire), en coopération entre les deux régions, peuvent permettre d'améliorer la situation. La gestion du régime hydrique est également compliquée par l'existence de captages d'eau potable et la dispersion des compétences en matière de gestion hydraulique des cours d'eau.

En tout état de cause, les limites des budgets disponibles peuvent induire des restrictions dans la gestion des sites, même lorsqu'un mécanisme de subvention est prévu.

3.3.3.2. Dans la partie flamande du site échantillon n°1

Le contexte juridique est globalement très favorable à la mise en place d'un réseau écologique dans ce site, tant au plan de secteur (en majeure partie en zone verte) que du point de vue de la conservation de la nature (VEN, Natura 2000, réserve naturelle flamande) ou du point de vue foncier (droit de préemption de la Région flamande dans certaines zones, possibilité de créer des zones d'extension de réserve, Natuurinrichtingsproject). Le régime Natura 2000 flamand n'exclut pas, semble-t-il, la prise

[92] Mentionnons par ailleurs que certaines interdictions générales proposées par l'ECO-team (par ex. interdiction de tout curage des cours d'eau) semblent par ailleurs excessives par rapport à l'objectif de conservation poursuivi et aux nécessités de protéger le droit au respect des biens (droit de propriété).
[93] Ce statut peut être imposé au propriétaire, mais c'est très rarement le cas en pratique.

en compte d'autres espèces et d'habitats que les espèces et habitats d'intérêt communautaire (contrairement à ce qui est le cas en Région wallonne), ce qui est un avantage. Des mesures (limitées) de contrôle de l'épandage d'engrais sont en outre prévues dans le VEN. L'on ne voit pas, dès lors, quel instrument pourrait encore être utilisé pour mettre en œuvre le réseau écologique, pour autant bien sûr que ceux qui sont déjà en vigueur soient effectivement appliqués. En effet, comme en Région wallonne, un retard s'accumule pour l'adoption du plan d'aménagement de la nature (« *Natuurrichtplan* ») de la GEN / de la SBZ. De plus, il n'est pas encore dit que la gestion sera prise en charge par les propriétaires et occupants si ceux-ci refusent de collaborer. Ceci pourrait nécessiter des mesures plus strictes d'acquisition de la propriété (notamment pour les zones centrales les plus fragiles). Le « *Natuurinrichtingsproject* » devrait permettre une plus grande souplesse dans l'aménagement du site et autoriser des travaux de restauration importants (après acquisition de la propriété par la Région flamande par échange de parcelles).

Comme en Région wallonne, un problème majeur est lié à la protection et la restauration de la qualité de l'eau, qui ne dépend pas entièrement du Gouvernement flamand puisque la Dyle prend sa source en Région wallonne. De plus, cette protection implique des mesures à l'échelle du bassin versant qui ne peuvent être prises que moyennant une approche intégrée impliquant les différentes autorités compétentes (eau, mais aussi aménagement du territoire, permis d'environnement,…). Le nouveau décret flamand de 2003 pourrait améliorer cette situation.

La remarque relative aux budgets disponibles est également d'application.

3.3.3.3. Remarque générale sur la représentativité de la situation juridique de la zone d'étude

Il faut préciser que cette situation juridique très favorable à la conservation de la nature est loin de se présenter partout en Belgique. Il se pourrait donc que la vallée de la Dyle soit peu représentative à certains égards de la situation juridique générale des biotopes cibles dans les deux régions. Le fait que plusieurs des habitats cibles soient aussi des habitats d'intérêt communautaire (par ex. les forêts alluviales) permet de nuancer cette observation puisque des sites doivent être créés pour les préserver dans toute la Belgique.

3.4. Conclusions générales

En dépit de certains aléas méthodologiques liés au caractère multidisciplinaire de l'étude et des contraintes de temps[94], la présente étude a permis de dégager plusieurs enseignements sur la faisabilité juridique d'une stratégie du réseau écologique. Tout d'abord, elle met en évidence la *complexité du cadre juridique* dans lequel cette stratégie s'inscrit nécessairement, à savoir les différents régimes juridiques et fiscaux encadrant l'utilisation des sols et la gestion des ressources aux différentes échelles. La structure de la propriété, l'éclatement vertical et horizontal des compétences dans le processus de décision, le manque d'articulation entre les législations, le cloisonnement des administrations, la rigidité des normes eu égard à la dynamique et à la variabilité des systèmes écologiques constituent autant de freins à la mise en place d'un réseau écologique intégré. Le nécessaire respect des droits et libertés fondamentaux exige enfin des autorités qu'elles n'adoptent que des mesures proportionnées et non discriminatoires, ce qui réduit encore leur marge de manœuvre.

Ensuite, l'examen des principaux instruments juridiques existant dans les deux régions et leur application au cas de la vallée de la Dyle ont permis d'évaluer *dans quelle mesure l'arsenal des règles en vigueur est à même d'encadrer la création et la mise en œuvre concrète, sur le terrain, de la stratégie de réseau écologique*, qui implique, d'une part, une planification approfondie et intégrée et, d'autre part, l'adoption de mesures extrêmement diverses, de protection, de gestion et de restauration. Une difficulté particulière est liée au caractère transrégional du projet ECONET, qui a nécessité l'analyse de deux systèmes juridiques très différents (flamand et wallon). La complexité d'une approche intégrée n'en est que renforcée.

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[94] Les conclusions méthodologiques relatives au projet ECONET sont intégrées dans la conclusion finale du rapport.

En termes de *planification spatiale du réseau*, il ressort de l'étude que les régimes du VEN et Natura 2000 sont les deux principaux instruments de planification spatiale spécifiques organisant de façon contraignante et sur des bases scientifiques la mise en place d'un réseau écologique fonctionnel à l'échelle régionale, ou à tout l'ossature de celui-ci. A l'échelle du site, les régimes du VEN et de Natura 2000 comprennent en outre des outils de planification (y compris spatiale) des mesures de conservation fort élaborés, mais complexes à appliquer (arrêté de désignation, Natuurrichtplan,…). Les régimes classiques d'aires protégées (réserves naturelles et forestières, etc.) comprennent aussi des plans de gestion (plus facilement gérables), bien que ne couvrant pas des superficies aussi grandes. L'articulation du plan du réseau avec la planification spatiale de l'aménagement du territoire, indispensable, est systématique en Région flamande (avec certains inconvénients), mais n'est prévue que dans une certaine mesure en Région wallonne.

Une grande différence entre les deux régions est que, dans la Région wallonne, l'objectif du réseau est strictement focalisé sur les espèces et habitats d'intérêt communautaire (Natura 2000), le statut de site Natura 2000 ne pouvant en principe être utilisé pour préserver d'autres espèces et habitats menacés en Wallonie. Ceci rend la planification du réseau incomplète, même si nombre de sites Natura 2000 abritent d'autres espèces et habitats menacés. De surcroît, l'absence de procédure de participation dans le processus de sélection et de désignation des sites risque de susciter de vives oppositions. Le futur Plan d'action pour le développement de la nature (PADN) vise certes l'ensemble du patrimoine naturel et fait l'objet d'une large consultation, mais n'offre pas de vision spatiale du réseau. De ces considérations il ressort qu'en Région wallonne, la faisabilité (théorique) d'un réseau « complet » serait grandement améliorée par un instrument de planification *spatiale* du réseau couvrant toutes les espèces et habitats menacés en Wallonie, de préférence contraignant pour les autorités (et articulé avec la planification spatiale générale) et soumis à participation. L'étude du cas pratique n'a cependant pas porté sur cet aspect de la création d'un réseau.

S'agissant de la *mise en œuvre concrète* du réseau envisagé sur le plan juridique, trois voies peuvent être suivies à cette fin par les pouvoirs publics, le cas échéant en combinaison, à savoir la *voie réglementaire*, la *voie incitative ou contractuelle* et la *voie foncière*, consistant à agir sur la propriété. Chacune repose sur l'utilisation d'instruments spécifiques ou combinant plusieurs approches. Il ressort de l'analyse théorique et du cas pratique que si les instruments existants sont globalement satisfaisants dans les deux régions, le droit flamand prévoit toutefois plus d'instruments et d'habilitations spécifiques pour les autorités publiques (notamment au travers du VEN et des mécanismes associés) que le droit wallon, qui est en général plus ancien. Il ressort de l'étude que, dans certains cas, leur application peut dépendre fortement de la structure du droit de propriété et impliquer un pouvoir d'appréciation discrétionnaire de l'administration (par ex. dans les permis de modification des petits éléments du paysage) qui rend plus aléatoire la protection théorique qu'offrent ces instruments.

Plus précisément, les *statuts d'aires protégées*, de nature contraignante, offrent sans doute le cadre juridique le plus approprié pour assurer la protection (à l'intérieur du site), la gestion et la restauration de la plupart des *zones centrales* du réseau et, le cas échéant, de certaines zones de développement plus fragiles. Les statuts de réserve naturelle (le plus strict) et forestière présentent un grand intérêt à l'échelle de la parcelle, mais sont généralement insuffisants pour couvrir les importantes superficies qu'occupe le réseau, compte tenu de la maîtrise foncière que requiert, sauf expropriation, leur mise en place. La création d'aires protégées indépendamment de l'accord du propriétaire (classement comme site, zones humides d'intérêt biologique, site Natura 2000, VEN) s'avère donc indispensable pour protéger les zones centrales, sauf à procéder à de coûteuses mesures d'acquisition à l'échelle de toute la vallée. Les régimes Natura 2000 (dans les deux régions) et du VEN (en région flamande) sont particulièrement élaborés, contraignants et intégrés et opèrent à l'échelle du paysage plutôt que de la parcelle, ce qui leur confère un intérêt considérable pour une stratégie de réseau écologique telle qu'envisagée dans la vallée de la Dyle. Leur mise en œuvre, obligatoire pour les autorités, s'avère cependant d'une grande complexité et est fortement retardée. Il n'est pas dit non plus qu'elle emportera l'adhésion des acteurs si leurs préoccupations, notamment financières, ne sont pas prises en considération. Enfin, l'objectif du réseau Natura 2000 étant strictement focalisé sur les espèces et habitats d'intérêt communautaire, d'autres mesures complémentaires, comme la création de réserves, peuvent s'avérer nécessaires en dehors, voire à l'intérieur même des sites. En cas d'incompatibilité, Natura 2000 a toutefois priorité, ce qui peut empêcher la réalisation d'un réseau centré sur d'autres objectifs. Il en ressort que le recours à d'autres instruments, contraignants et/ou fonciers, reste nécessaire pour créer un réseau complet et efficace.

La création de zones protégées dans les zones centrales est toutefois loin de suffire pour créer un réseau écologique. En dehors des zones protégées, dans les *zones de développement*, l'affectation au plan de secteur en zone non urbanisable (zone forestière, agricole, naturelle, d'espaces verts,…) combinée avec des incitants attractifs (y compris négatifs comme l'écoconditionnalité) en faveur d'une gestion du milieu (agricole, aquatique, urbain et forestier) compatible avec le maintien d'une certaine biodiversité devrait permettre d'assurer le développement du patrimoine naturel « ordinaire ». La révision thématique des plans de secteur fondée sur une planification spatiale du réseau et l'affectation de budgets plus importants aux mesures écologiques de développement rural devraient à cet égard constituer une priorité. Dans les *zones de liaison*, des mesures combinant incitants et protection contraignante devraient être adoptées, comme l'autorise par exemple le droit wallon (périmètre de liaison écologique et périmètre d'incitation entre les sites Natura 2000). Leur planification serait toutefois nécessaire pour éviter le gaspillage de ressources. La protection du *maillage écologique* et l'obligation de *compenser* systématiquement sa destruction pourraient également améliorer la connectivité du paysage. Les polices de permis impliquant un pouvoir d'appréciation discrétionnaire des autorités compétentes, des règles de protection ou au moins des directives spécifiques devraient être adoptées pour orienter la prise de décision.

Enfin, l'étude fait ressortir l'importance capitale d'une *approche intégrée du réseau écologique* afin d'obliger toutes les autorités concernées à prendre en considération le réseau dans les politiques sectorielles. En effet, ceci ressort notamment des difficultés rencontrées dans la zone d'étude pour mettre en œuvre, sur le plan juridique, des mesures de protection efficace de la qualité de l'eau et des mesures de contrôle des impacts des activités agricoles autour du site. L'approche intégrée peut recourir à différentes techniques procédurales ou institutionnelles (évaluation des incidences, mécanismes de coordination des décisions, organes consultatifs,…) mais nécessite aussi l'adoption de mesures substantielles, comme la planification du réseau et l'adoption de règles de protection contraignantes applicables à tous. L'approche intégrée de la gestion des ressources en eau, récemment promue par les législateurs communautaire, flamand et wallon, est un pas important en ce sens, même ces nouveaux régimes n'ont pu être analysés de façon suffisamment approfondie.

En conclusion, l'on constate que *le droit wallon et flamand offre, certes de façon inégale entre les deux régions, aux autorités publiques un arsenal d'instruments, de type contraignants, économiques, volontaires ou fonciers, susceptibles d'être utilisés pour créer un réseau écologique*. Aires protégées, permis, subventions, mesures fiscales, écoconditionnalité constituent autant d'outils potentiellement intéressants, *pour peu que la volonté politique de les mettre en œuvre existe*. Dans la vallée de la Dyle, nombre d'entre eux sont au demeurant déjà d'application, ce qui rend d'autant plus « faisable » le réseau ECONET, sous réserve de ce qui a été dit ci-avant. Des améliorations pourraient toutefois être envisagées, notamment en Région wallonne – elle manque par exemple d'un outil foncier comparable au Natuurrichtingsproject et d'instruments de planification stratégique performants à l'échelle du paysage.

Ces outils, même améliorés et jouissant d'un appui politique, ne sauraient cependant permettre la mise en place d'un réseau écologique efficace et accepté par le public qu'à différentes conditions, à savoir notamment :

- l'élaboration d'une planification unique et cohérente, fondée à la fois sur des *bases scientifiques* et sur une *participation des acteurs*, et comportant une délimitation spatiale claire du réseau à l'échelle du paysage ;
- le recours au principe de précaution en cas d'incertitude scientifique, tout en promouvant la recherche pour ajuster au fur et à mesure les mesures prises ;

- la mise en œuvre d'une approche intégrée, impliquant la participation et la coopération de tous les acteurs concernés, sur la base de la planification précitée ;
- le respect du droit de propriété et des principes de proportionnalité et d'égalité / non discrimination dans les mesures prises, le cas échéant au travers de mesures d'indemnisation et de taxation correspondant ;
- l'exercice d'un contrôle efficace et l'application de sanctions adéquates en vue d'assurer l'effectivité des instruments mis en œuvre (ceci était considéré comme acquis dans l'étude);
- le dégagement de moyens financiers, techniques et humains suffisant pour garantir la mise en œuvre effective des mesures envisagées et la prise en charge par la collectivité des coûts liés au maintien du « patrimoine commun » que constituent les espèces et habitats sauvages.

PART 4: SOCIAL ASPECTS

4.1. General introduction

This part deals with the social context of the implementation of an ecological network. Within the ECONET-project, implementing an ecological network is considered as allocating the territory or pieces of land to certain sustainable land uses, aimed at preserving biodiversity. This allocation may imply the execution of – more or less far-reaching – nature development and management measures, in combination with a certain legal protection. A significant part of the territory in which an ecological network is to be established, however, no longer is "virgin" land. Ecological networking demands a multifunctional use of the common, currently often mono-functionally used, space. As the restoration of a network may displace or diminish long-established activities, the 'social context' comes forward. It is generally acknowledged that the social context plays an essential part in the durable success of programmes aiming at the conservation of biodiversity, including those pertaining to ecological networking (Nowicki et al., 1996). The 'technical' approach seems largely insufficient for a sustainable preservation of biodiversity. The major challenge is involving other types of land users, convincing them to co-operate in the sustainable management of their own environment; reconciling nature conservation with socio-economic terms where possible. The implementation of programmes towards the conservation of biodiversity often conflicts with socio-economic customs or acquired rights and with economic imperatives. For instance, with respect to claims on the land: some pieces of land will have to be re-allotted, other pieces will need a considerable modification of the current land use in order to be compatible with the restoration of an ecological network. Co-operation of local land users is therefore essential for the maintenance of biodiversity on a larger scale and in the long term. The social context is indeed important.

By taking into account the social aspects of the implementation of an EN, this sub-study aimed at delivering the 'software' to complement the 'hardware' of the ecological approach in the multi-disciplinary ECONET-project. Economic and legal aspects of the implementation are the other 'software components' in this project. These aspects were considered in parallel sub-studies.The central research question for the sociological sub-study was quite straight: "***which social aspects determine the feasibility of the implementation of an ecological network***?". According to us, this question could be answered by studying three aspects as being determinative for the social context of ecological networks and nature conservation, namely:

(1) man's perception of nature (and nature conservation);
(2) social impacts of the implementation of ecological measures; and
(3) the process or organizational structure of implementing an EN.

For the sociological research process, a deductive approach is followed, whereby out of the general research question a theoretical framework is designed on the basis of a literature study (§ 4.2). This framework is used as foundation and guide for the further research activities. The insights out of the theoretical framework are used to explore the social context of the implementation of the scientifically developed ecological network in the Dyle valley (§ 4.3). In § 4.4 we sketch the major conclusions from a sociological point of view.

4.2. Theoretical framework

4.2.1. Introduction

From a sociological point of view, a theoretical framework is indispensable to cover and frame the study of the complex issue of interrelated natural-physical and societal components of the implementation of an ecological network. The construction of this framework was therefore a crucial phase in the sociological sub-study. It describes how the reality of the implementation of an ecological network can be described and approached from a sociological point of view. It also explores if and how impact, perception and organizational structure interrelate in the context of the implementation of an ecological network.

In § 4.2.2 the general framework used for the sociological study is presented. It enables a representation of a vision on society in general and (the relation between) natural (ecological) and societal structures in particular. The general framework will give an insight in how perceptions of people are formed (§ 4.2.3), impacts of actions are described (§ 4.2.4) and organizational structures are designed (§ 4.2.5). Next to a conceptual story of how these phenomena are formed, the theoretical framework also contains a more practically focussed part in which the research concerning perception, impacts and organizational structures in relation to nature (conservation) is explored. A profound description of the theoretical framework can be read in Annex 'Theoretical Framework'.

4.2.2. General framework for the study

As the basis for the framework, the "systems theory" of Niklas Luhmann (Blom, 1997; Laermans, 1997; Noe et al., 2003; Vancoillie et al., 1999) was used. This theory was combined with other sociological theories, like the dramaturgical model (Burns, 1992; Drew et al., 1988; Goffman, 1971, 1982, 1987; Vancoillie et al., 1999), aspects of structuralism (Gielen, 2001; Laermans, 1984; Pels et al., 1989), Actor Network Theory (Callon, 1986; Latour, 1986, 1987, 1988, 1993, 1999; Law, 1986) and the civilization theory (Elias, 1990; Vancoillie, 1999). Using the systems theory, reality (i.e. the social research field) can be envisioned as being a 'system', itself consisting out of several sub-systems, which can be psychic, organic, or social (cf. infra). Between these sub-systems relations and interactions may exist, in one way or another, and this can change the (successive communicative) operation(s) of these sub-systems. The participants or members of the respective sub-systems can mediate the influence of one system to another[95]. Their personal and professional background (cf. infra: *habitus*) will serve as a filter through which the stimuli out of the environment will pass, before altering the system somehow or other. A characteristic of the systems theory is that the system is dynamic; nothing is fully determined.

The ***social sub-systems*** form the primary focus in the social sub-study. The systems theory of Luhmann distinguishes three kinds of social systems, namely interactions, organizations and the society in itself. For the research on the social feasibility of the implementation of an ecological network, only the theoretical insight into the first two mentioned is relevant. Interactions arise through the mutual observation of people who are at a certain place. Simplified, interactions are similar to (but are not completely the same as) communications between people or groups. Organizations are a more structured form of social systems that exist even without direct interaction. Examples of organizations in the research field of the ECONET-project are local and regional authorities, government departments, nature associations, companies, etc.. Human beings are 'members' of one or more of these organizations. Goffman (1982) pays attention to an important aspect concerning organizations, namely the distinction between 'frontstage' and 'backstage'. The frontstage of an organization represents the official position of the system, for instance, of a farmers union towards nature development. In the backstage of the organization, the different personal attitudes of the members of the organization and their (power) position towards others become, among other things, apparent[96]. Another important characteristic of a social system related to group dynamics and communication – though this counts for psychic and organic systems as well – is that the system is autopoietic (see also § 4.2.5).

The mind or ***psychic system*** is, according to the systems theory, a temporalized system in which mental representations succeed each other selectively. The elements of psychic systems are feelings, thoughts, intentions, etc. – in short: mental impressions or representations. Perception can thus be viewed in terms of the systems theory as the operation of the psychic system. The actual operating of the psychic system goes together with the creation of a new thought or, in general, of a new mental representation.

[95] Actually, the formation of system-internal operations or elements on the basis of system-external operations in the environment is mediated by the psychic systems in the nearby surrounding of the social system. They serve as a sort of filter. This way, the social system decides itself if it takes up the elements produced in the environment as input for the own operating.

[96] More elaboration on this frontstage-backstage subject can be found in Annex 'Theoretical Framework'.

When framing nature conservation in general, and the implementation of an EN in particular, there is, according to the systems theory, an important third sub-system, the ***organic system***, i.c. the ‘organic members’ of the ecological network. This organic system is also characterized by means of a number of individual smaller organic systems, for instance specific types of fauna and flora. In the ECONET-project, these individual organic systems are the so-called ‘target habitat types’ for the Dyle valley (cf. the ecological sub-study). The organic system is described in the ecological sub-study[97].

The value and power of the systems theory is that it is not only valuable to observe the structural aspects of a certain phenomenon, but also to observe the process in itself. Another advantage of this vision on society is that society is not seen as fixed and predetermined, but as a social system that is contingent, and therefore never certain, never fixed.

Below, the three social aspects as included in our research questions are framed in the spirit of this theoretical framework. In § 4.2.6 the use of the general framework as framework for the exploration of the social feasibility of the implementation of an ecological network is discussed. The following hypothesis has been investigated: ‘the systems theory forms an adequate theoretical framework for the social research concerning an ecological network, as it is able to integrate the three major aspects, i.e. perception, impact and organizational structure’.

4.2.3. Perception

Local stakeholders’ perceptions of nature and nature conservation, and the biased mutual representation of each other (as ‘insiders-outsiders’; ‘them-us’; ‘right-wrong’) play a prominent part in the feasibility of the restoration of an ecological network (Bogaert, 2004; Aarts, 1998; Filius et al., 2000; Klumpers, 2001; Resource Analysis, 2003; …). Perception partly determines the attitude and willingness of stakeholders to co-operate, and precisely this public commitment and public support will be conditional to build the necessary partnerships for ecological networking (implying multi-functional land use) as well as keeping the balance between local, short-term interests and global or collective long-term interests.

General theory

In terms of the systems theory, perception is understood as the operation of the psychic system or mind of a member or participant of a social system. The psychic system mediates between different social systems. It observes, processes and filters ‘communications’ coming from the (social) environment. Thus, perception is seen as being formed through interaction of the psychic system with the surrounding social and natural environment. The operations (or communications) in the environment of the psychic system are observed and processed in the operating of the psychic system itself. However, these mental impressions are also filtered through and formed by someone’s present situation and personal background (e.g. habitus[98], ground-attitude (cf. infra), etc.).

Jacobs et al., (2002) give a practical model that reflects the above theory of perception as a result of a filtered interaction with the environment (Figure 4-1). According to Jacobs et al., perception (of nature, organization, …) can be seen as one’s position or attitude[99] towards a specific

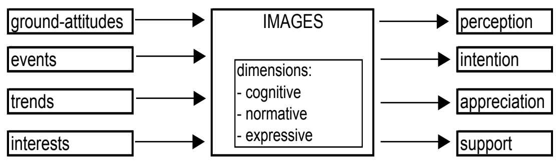

Figure 4-1 Perception: basic influential factors and effects

[97] Note that ‘organic system’ has a different meaning in the systems theory of Luhman (sociological perspective) compared to the ecological perspective. For Luhman an organic system is a collection of living things (fauna & flora). For ecologists the organic system is broader, consisting out of biotic (living) and abiotic (non-living) components.

[98] Habitus, a concept defined by Pierre Bourdieu, is the total existential environment of a person. This includes the person's beliefs and dispositions, and prefigures everything that that person may choose to do. The concept of habitus challenges the concept of free will, in that within a certain habitus at any one time, choices are not limitless. A person is not an automaton, for there exists flexibility in a habitus, but neither is there complete free will.

[99] Attitude is an important thing that helps people to frame their social world and reduce complexity. It helps us to define how we perceive and think about others, as well as how we behave towards them. Given the theory of cognitive dissonance, the

object (e.g. nature conservation measure, organization, …), existing in the empirical world. Man's perception is a result of many aspects. 'Events', or impact/effects as a consequence of change, are one type of these determining factors. Another important influencing aspect of someone's perception is his or her 'ground-attitude'. The so-called '*ground-attitude'* or *fundamental attitude* of a person can be defined as his or her general view on things. This fundamental attitude of a person is quite stable compared to, for instance, perception, as it is mainly formed through long-term socialization and learning processes and the specific personality of each individual. 'Ground-attitude' can, in the systems theory, be defined as the whole (synthesis) of all past operations of that psychic system. *Images* can be seen as the vision of individuals or groups on how an object should look like and on what feelings are associated with certain states or forms of the object. Images are concerned with the meaning people put on objects and serve as frameworks that direct and structure the perception and appreciation of an object in cognitive, normative and expressive respect.[100]

Perception of nature (conservation)
A number of environmental philosophers elaborated on the description of ground-attitudes towards nature. De Groot (in Jacobs et al., 2002) made a classification of four different ground-attitudes with respect to the relation between man and nature:

- ➔ Man as dominant master of nature: people have the right and the knowledge to dominate nature; without people, nature is of no value;
- ➔ Man as nature's guard or manager: nature in itself is valuable, but is also – and especially – a resource for people, who therefore must manage nature well;
- ➔ Man as nature's companion: nature is 'something else', outside of people, something you may know and with which you can build a respectful and reciprocally equivalent relationship;
- ➔ Man as participant to nature: nature is superior to people. Human beings, as being a component of nature, cannot decide on nature affairs.

Aarts (1998) configured a direct link between these ground-attitudes and existing nature (conservation) images. For instance, regarding the nature-image of dominant masters, nature is a robust system, which will always recover spontaneously. These 'masters' are not interested in nature in itself. Guards of nature are afraid of nature. Hence, nature must be controlled. Companions of nature want to protect nature because of it's vulnerability; while participants to nature believe that nature is robust, can develop on it's own, but still needs protection (wild nature). Other research (Filius et al., 2000; Jacobs et al., 2002; de Boer et al., 2002) tried to link the above and other nature-images to specific stakeholders. In Figure 4-2[101] the results of a meta-analysis, performed during our research process, are represented. Depending on the ground-attitudes towards nature, the nature-image(s) and the context (e.g. events, trends, …), people will perceive nature somehow or other. Buijs et al. (in Bogaert, 2004) identified the following perceptions of nature: 'nature disserves protection', 'nature is for recreational use', 'nature is beautiful', 'nature is impressive', 'nature is a counterweight for society', 'nature is a source of food and medication'.

	Wilderness	Autonomy	Broad	Decorative	Functional
Farmers			x		x
Recreational people	x	x	x	x	
Hunters			x		x
Nature associations	x	x	x		
Anglers / fishermen				x	
Public authorities	x	x	x		

Figure 4-2 Nature-images related with actor-groups

In conclusion it can be stated that '*the*' nature does not exist. 'Nature' has a different meaning for different people. These differences deserve particular attention when implementing nature conservation strategies, e.g. restoring ecological networks.

relationship between attitudes and behaviour becomes questionable. Nevertheless, attitudes still are one of the strongest indicators for possible future behaviour.

[100] The cognitive dimension is concerned with the question of how people define the object. The normative dimension is concerned with the question of how to act in respect to the object. The expressive dimension is concerned with the question of what people perceive as beautiful. (Jacobs et al., 2002).

[101] The darker a field is shaded, the more relevant it is for that specific actor-group.

Ground-attitudes, images and perceptions concerning nature of different stakeholders in the Dyle valley were analyzed in the case study (see § 4.3).
The following hypothesis has been investigated: 'different stakeholders will have different nature-images, and these nature-images may differ substantially from the scientific nature-image'.

Differences in ground-attitudes, images and perceptions related to nature generally lead to different images and perceptions related to nature conservation as well (cf. supra: Aarts, 1998).

The concept of ecological networks is nowadays widely accepted within the nature conservation sector as the appropriate policy for the conservation of biodiversity. In most European countries it became the leading principle of nature conservation policy (Jongman & Kristiansen, 1998). A European survey among professional nature conservationists shows that both environmental scientists and NGOs are considered as most outspokenly favourable to the concept of ecological networking, closely followed by nature conservation authorities and owners/managers of currently protected areas (Rientjes et al., 2003). On the other hand, the inquired conservationists perceive farmers and other landowners as rather critical to the concept of EN. Nevertheless, still according to this survey, during the process of network development these 'opponents' of ecological networks have been known to move from 'critical', to 'neutral' and even 'supportive'. A survey within the Department of Nature and Forest Conservation in Flanders confirmed some of these results (Resource Analysis, 2003: see Annex 'Survey among the Flemish govermental departments and the ECONET user-group'). This seems logical, as it may be assumed that most of the respondents – all out of the environmental sector – have similar ground-attitudes, nature-images and perceptions (see Figure 4-1). However, a stakeholder analysis in preparation to the demarcation of the Flemisch Ecological Network (VEN) showed that, besides the nature conservationists, the representatives of the agricultural organizations as well were favourable to the demarcation of the network. They mentioned the "clarification of farmers' future prospects" as a reason for this positive attitude[102]. Other stakeholders, on the other hand, like fishers, hunters and foresters showed a rather negative attitude towards the demarcation. This seems to be a result of misconceptions and non-communication, as negative impacts for these types of stakeholders are usually rather small or non-existent (Resource Analysis, 2002).

Although nature conservationists seem to agree on the concept of ecological networks as the leading principle of nature conservation, the conceptualisation and (views on the) practical implementation of these networks did develop differently throughout Europe, as a result of different geographical, natural, economic, political and social conditions. Different definitions of ecological networks can be found in the scientific literature[103]. The named objectives of ecological networks differ from purely ecological reasons (to conserve nature, to protect certain species, ...), to landscape conservation and even recreational or cultural reasons (Jongman & Kristiansen, 1998). The survey of Rientjes et al. (2003) further shows that there is no consensus on the necessity of having corridors, stepping stones and buffer zones within an ecological network. Related to this, nature conservationists also have different opinions concerning the (juridical) implementation of an ecological network (Resource Analysis, 2003; Rientjes et al., 2003).

[102] However, this positive attitude only counted in the beginning of the process, and only for the *concept* of EN. Once the public inquiry on the demarcation took place – and public sentiment was roused – the farmer organizations clearly and publicly opposed the demarcation of the VEN. The vagueness of procedures for implementation, the lack of clarity concerning compensation commitments and incentives, and, still, the uncertainty about legal protection had fuelled the resistance.

[103] In the special issue on ecological networks of the journal "*De levende natuur*" (104, n° 6), an ecological network is defined as "an unbroken system of areas, where nature can develop and plants and animals can thrive without disturbance". Some other definitions mentioned in the scientific literature are:

- ➔ "an ecological network is composed of core areas, (usually protected by) buffer zones and (connected through) ecological corridors" (Bischoff & Jongman, 1993);
- ➔ "ecological networks, developed from the concepts of island theory and metapopulation dynamics, aim to provide the physical conditions that are necessary for ecosystems and species populations to survive in a landscape that, to a greater or lesser extent, is also exploited by economic activities" (Bennett, 1997);
- ➔ "the whole of biotopes suitable for providing a maximum of species a temporal or permanent living environment, consistent with their requirements and guaranteeing surviving in the long term" (internal ECONET definition, after *Delescaille L-M. 1993. Le maillage écologique et l'espace rural. Annales de Gembloux 99: 61-69*).

In the case study perceptions of the implementation of an ecological network in the Dyle valley, and views on the concept of EN, were analyzed.
The following hypothesis has been investigated: 'in the ECONET-project the ecological partners and end-users do have the same idea about the goals of an ecological network and the way an ecological network should be developed and implemented'.

The perception of social systems, and of their members or participants, can be characterized in the same way as the perception of the organic environment (e.g. nature). In the case of the implementation of an ecological network, these kinds of perceptions may be as important as the perception of nature or nature conservation in itself. In § 4.2.5 some further elaboration has been done on these perceptions of social systems and their members.

4.2.4. Impact

The social aspect 'impact' is concerned with the description of the social impacts of the implementation of an EN on society and its members, like farmers, recreational users, and so on. Social impacts are the real and perceived impacts felt by people (at individual and collective level) as a consequence of biophysical and/or social change processes, which are caused by planned interventions. It is important to analyze the potential impacts of the implementation of an EN to make clear how (much) the current social reality in the area will change (for the better or the worse) as a consequence of that implementation. The bigger the change, the bigger the effect on the social feasibility of an EN.

General theory
In terms of the systems theory, impact can be understood as the entering of certain operations (mostly communications coming from social systems) of one system into another. The affected system may or may not take the communications of the other system as content for its own communications and is – in the former case – influenced by this, by the other system. The 'entering' can be implicit as well, in that the selection of a communication is (partly) based on the operations in the environment of the system, without the operation in the environment being the subject of that new communication. In this case the communication in the system would have been different if a certain event in the environment did not happen. Thus, following the systems theory, impacts can be measured through an analysis of the (altered) communications. Both forms of entering of the environment into the system can either be identified as real or perceived impacts. In the case of perceived impacts, no real change in a social or organic system appears (as is the case for real impacts). Yet, a change in a psychic system can be detected, because a person might be afraid (perception) of certain operations in the environment.
Thus, (the impact of) certain actions (e.g. nature conservation measures) in the environment can have an influence on the perception of the environment (i.c. the ecological network and the other stakeholders). In other words: if an actor is influenced by the implementation of an ecological network, his or her attitude towards the ecological network, towards other stakeholders and towards nature in general may change, one way or another.

For the measurement and assessment of social impacts, different methods (for instance cost-benefit analysis, impact analysis, etc.) can be used. 'Impact assessment' is a general term used to cover a whole set of tools and processes to help the inventory, analysis and presentation of relevant information to decision makers – whether being governments, companies or individuals. Impact assessments are systematic processes for the identification, prediction and evaluation of the positive and negative consequences of activities. 'Social impact assessment' or SIA is "the process of analyzing (predicting, evaluating and reflecting) the intended and unintended consequences on the human environment of planned interventions (policies, programs, plans, projects) and any social change process invoked by those interventions, so as to bring about a more sustainable and equitable biophysical and human environment" (Becker & Vanclay, 2003). A SIA gives an overview of the social morphology of the considered area on the one hand, and of the foreseen measures on the other hand, and of how both aspects affect each other. A SIA has a surplus value towards other analytical methods in that it also adresses distributional aspects (e.g. social justice), social dynamics, social welfare, liveability and public and political support. It uses different social research techniques (e.g. surveys, focusgroup discussions, discourse analysis, …) and specific evaluation methods.

Impact of nature conservation

Nowadays, in response to the opposition of several societal stakeholders to nature development and conservation, nature conservationists and policy makers focus on the global societal benefits of nature development. There are a number of reasons identified why biodiversity could be important for mankind (e.g. Rientjes, 2001; ten Brink, et al., 2002; Turner, van den Bergh & Brouwer, 2003; …). Usually a distinction is made between *tangible* (e.g. building and clothing material, medicines, food for domesticated animals, food, timber, wildlife, genetic resources, …) and *non-tangible* benefits (relaxation and leisure area, environmental services, informational and evolutionary source, aesthetic or cultural value, …). Those benefits are often used as the social legitimation of nature conservation actions and plans (like, for instance, the implementation of an ecological network). At the moment researchers all over the world try to assess this value (the positive impacts) in economic terms (for instance Constanza et al., 1998; Witteveen & Bos, 2004; Ruijgrok, 2002; Moons et al., 2000). However, potential negative impacts (often local) or social/societal costs are usually left out of this political and scientific discourse, although it seems important to focus on these effects as well, as they determine for the most part the social feasibility and public support of an EN. To address this issue, a social impact assessment (SIA) seems to fit best. In a SIA emphasis is laid on local – both positive and negative – effects of a plan and on sociological impacts, like the alteration of social dynamics and social cohesion, or social erosion.

In the case study, a social impact analysis related to the implementation of an ecological network in the Dyle valley was performed.
The following hypothesis has been investigated: 'a SIA is an appropriate methodology to gain a clear insight into the social feasibility of ecological networks'.

4.2.5. Organization of the process

The implementation of an ecological network involves – among other things – the execution of restoration and management measures. Most of the land that is of value for the implementation of an EN neither is public property, nor owned and/or managed by conservation organizations. In almost all cases other groups are to be involved if the quality of nature is to be maintained or improved. Neither government authorities nor nature associations can successfully protect nature on their own. They depend on the co-operation of a wide group of people and organizations whose actions directly or indirectly affect nature: land owners, visitors to protected areas, hunters, farmers, foresters, tourism operators, etc. The feasibility of restoring an ecological network will therefore also depend on the participation, be it at different levels, of these actors (Rientjes, 2000; Bogaert, 2004; Overbeek & Lijmbach, 2004, …). To get support or co-operation, an appropriate organization of the (communication) process will be necessary. From a sociological point of view, the interaction (communication) between persons, organizations, … must be emphasised. This perspective is concerned with, for instance, whom to involve, at what stage in the process, what 'degree' of involvement, 'rules of the game', financial resources, etc..

General theory

In accordance with the systems theory, communications are the most elementary entities of 'the social' and, a fortiori, of society[104]. Successive communications form an own, independent reality or 'a system'. As for a so-called *'interaction system'*, the system will vanish or will cease to exist when the members of the system quit communicating and leave the event. Simplified, interaction systems are similar to (but not fully the same as) conversations: they demand the presence of actors who are able to observe each other (be it 'live' or via virtual media). If there is also a more structural relation, besides the mere physical presence or connection, Luhmann speaks about '*organizational systems*'.

[104] 'Communications' may be verbal or nonverbal, like for instance actions, postures or gestures of a person.

In relation with communications between different organizational systems, we should at first mention the autopoiesis – or *recursive self-referentiality* – of social systems. Out of this recursive self-referential production of new elements (i.e. communications) of a system, it follows that communications out of the (social) environment are filtered and, if accepted by the system, translated in terms of the own communication. Every organization[105] maintains its own reference image of reality. True communication and interaction is necessary to transcend one's own fixed image. Non-communication leads to the enhancement of the own reference image (stereotyping and stigmatization), and to the enlargement of the 'gap' between different parties. Through this mechanism, positions of different stakeholder(group)s become more and more rigid and negative towards each other, resulting in a still growing gap. This gap between the own group and the surroundings (cf. in-group and out-group: Elias, 1990; Aarts, 1998; Annex 'Theoretical Framework') particularly occurs when a group (tries to) increase(s) its internal cohesion, thereby making the distinction between '*us*' and '*the others*'(i.e. non-members of the group, or other groups) more explicit. Through this introspection, a system emphasises and even reinforces itself. This is often accompanied by negative attitudes and behavior towards the social environment (e.g. stereotypes[106], stigmatization: Goffman, 1971).

Thus, problems with communication among organizations arise when the above mechanisms come into play. Consequently, the relations between parties may change considerably in that their commitment and motivation to achieve a common solution completely disappears. However, this declining process can be altered by organizing an effective (intercultural) communication (Aarts, 1998). Through participation and open (clear, univocal) communication, other systems in the environment can be made more receptive to, and attentive for, the operating (communication) of the own system, as temporary, but more structural links between the system and the environment will be realized. Proactive participatory planning seems to provide a way to resolve potential (communication) conflicts by integrating values and expertise that exist within different professional and public groups (Gobster & Hull, 2000). The purposes of such a participatory approach are then, on the one hand, achieving public support, and, on the other hand, exchanging knowledge (scientific environmental knowledge versus lay knowledge, values, beliefs) and by doing so trying to obtain a better solution for the conflict or problem (Blunt & Warren, 1996; Hirsch & O'Hanlon, 1995).

Concerning communication and negotiations, Aarts (1998) makes a distinction between 'reactive compromises' and 'creative compromises'. According to Aarts, a 'reactive compromise' is the result of a negotiation process that is distributive in nature. The way in which actors operate in such negotiations is characterized by a high degree of reticence with respect to one's own position, by an emphasis on positions, by overcharging in the hope of ending up in the middle, and by a lack of care for the other party. Hence, a reactive compromise is the result of reacting upon each other's proposals. In the context of a planning process, this would mean that interaction with or among stakeholders only takes place in the end phase of the process. In Vandenabeele (2004) this is called 'the concept of the acceptance logic'. On the other hand a 'creative compromise' results from an integrative process of negotiation, in which actors show both a high degree of openness with respect to one's own situation and intentions, and a certain level of care for the other party in addition to self-care. This enhances the possibility that a new common problem is formulated – reframing may be needed (cf. translation[107]) – in which the problems of the various parties are incorporated (e.g. both the environmental problem (loss of biodiversity) and the agricultural problem (threat to social security)). The common problem then forms the basis for achieving a creative compromise. A creative compromise may lead to more accepted, sustainable solutions. Vandenabeele (2004) talks about 'the concept of the deliberative logic' in this context. Here, interaction would take place right from the start of the planning process untill the end. Both negotiation processes however are ideal types and do not, or hardly, exist in the pure form. A 'swinging' between the two types of processes – reactive vs. creative, or top-down

[105] Still in terms of Luhmann, an 'organization' is a more structural form of an 'organizational system'.
[106] Stereotype - A rigid, oversimplified belief that is applied to all members of a group or social category. (http://www.webref.org/sociology/s/stereotype.htm).
[107] For more information about the 'model of translation' in the Actor-Network Theory, see Annex 'Theoretical Framework', § 1.2.5. Power.

enforced commitment (e.g. the initiation of a restoration project) vs. interactive communication and co-operation – seems to be valuable in policy making (Aarts, 1998; van Woerkum, 2000).

Interactive planning and policy making, however, is not the solution to every problem in policy development and policy implementation. Hisschemöller (1993, in: Vandenabeele, 2004) stated that problems which can be characterized as 'unstructured'[108] are favourable to a broad interactive policy making. Apart from the problems associated with interactive policy making as such, not every situation is suited to start an interactive (policy) process. Rientjes (2000) listed some critical prerequisites for an open, participatory approach. First of all, people must be aware of their being a stakeholder in the specific problem (problem awareness). Secondly, there must be a feeling of mutual dependency (awareness of need of others to reach the own objectives). If people think that the others have 'nothing' to offer, or if they think that they have absolutely no chance of reaching their objectives themselves, they will not become involved. Power-relationships have to be more or less equal. Thirdly, the people involved must use the same language (interpret important concepts in the same way). If this is not the case, sufficient time must be available in the process to be able to learn to speak this same language.

Question is whether the planning and implementation of an ecological network could benefit from one or another interactive planning method. Therefore, the following hypothesis has been confronted with existing literature: 'the planning and implementation of an ecological network in general is a typical object of participatory planning'.

Above, the involvement of stakeholders in a process is treated from a sociological – even psychological – point of view. A practical translation of this concept is needed to be able to design the process. Other aspects as well are of importance for the practical implementation of the process. At least three design principles appear to be crucial when analyzing organizational structures for projects, namely the actors involved (cf. supra), the 'rules-of-the-game' and the nature of the project in itself (Agrawal, 2002; Baland & Platteau, 1996; Blumenthal & Jannink, 2000; Leach, 2002; Leach et. al. 2002; Ostrom, 1990, 1999; Ostrom et. al. 2002). The rules should govern the interaction between the actors involved (i.e. govern the substantive and procedural terms of co-operation and define the relationship between the parties involved in the process). Types of rules are, for instance, decision rules, financial rules, monitoring rules, conflict resolution rules, sanction or pay-off rules and entry-exit rules. Finally, some characteristics of the project itself can be important in the assessment of the performance of the organizational structure (e.g. geographical, social and historical context). The practical translation of these design principles has not been tested in the Dyle valley, since there was no 'real' network to be implemented – no real process of implementation was to be initiated. Elaborated research on this topic will be carried out in another Belgian Science Policy-project: "*The design-parameters of governance structures at the local level – case: the implementation of ecological networks*".

The organization of a process aiming at nature conservation
As already stated above, clear communication is a crucial aspect in nature conservation in general, and during the implementation of an ecological network in particular. Neither government authorities nor NGOs can successfully restore a network on their own. Surveys carried out in the nature conservation sector (among environmental scientists, NGOs and government authorities) show that even the nature conservationists themselves are aware of this (Bogaert, 2004; De Zitter, 2002; Resource Analysis, 2003). Local people, people who use the land, people who use nature for leisure activities, land owners, foresters, farmers, hunters, anglers, road constructors, (local) authorities, government, … are all mentioned in the surveys as 'needed to be actively involved' in case of an ecological restoration

[108] Unstructured problems are defined as problems with neither consensus on values (standards), nor on knowledge (facts). If there is consensus on the values, but not yet on the facts, then some kind of interaction will be necessary as well, be it less broad, with a major role for scientific experts. If there is consensus on the knowledge, but not on the values, then interaction will also be needed to be able to solve the problem, with a major role for civic organizations (like nature conservation and farmer organizations).

project (cf. enrolment[109]). Hence, it seems that nature conservationists support an interactive approach for the planning and implementation of ENs. Based on another survey carried out among the Flemish administration for nature conservation (Resource Analysis 2003), it can be stated that farmers, foresters, nature conservationists and landowners are perceived as the most important stakeholders when considering the feasibility of an ecological network. Recreational users, like for instance riders, bikers, hikers and anglers, as well as industrial companies are observed, overall, as to rather marginally affect the feasibility. The same survey shows that farmers, besides being 'most needed for success', are also perceived to show the most negative attitude towards an EN. Nature conservationists, for their part, are seen as the most favourable to the implementation. It is noteworthy, however, that some parties which are mentioned in the surveys as 'most important to succeed' are hardly – and often even not at all – involved in policy preparation and implementation (Resource Analysis, 2003; Leroy, 2004).

The practical organization of the process of implementing an EN varies throughout Europe[110] (Nowicki et al., 1996). However, in most cases the initiative to establish an EN will follow a top-down approach from central government (Nowicki et al., 1996; Mougenot & Roussel, 2002). These top-down initiatives then at first depend upon support at the highest political level in order to secure the necessary enabling legislation and financial resources beforehand. The approach to many ENs is, to a large degree, based on pure biological surveys. The restoration goals are decided by environmental scientists. This approach has been criticised lately for being 'ecocentric' and based on a too narrow set of values. It has not provided enough opportunities for combining nature conservation with other forms of land use such as agriculture, forestry or tourism. In several countries this led to difficulties as regards the co-operation of local stakeholders (Jongman & Kristiansen, 1998). The re-distribution of resources is not perceived to be legitimate by the affected interest groups, and the outcome often considered as unfair and unjust (Nowicki et al., 1996). Nevertheless, the 'ecocentric' approach remains to be widely spread within nature conservation (policy). Often, these plans are a result of a 'D-A-D-strategy' (Decide-Announce-Defend). Interaction with stakeholders is limited. In different cases this approach led to the development of the phenomenon of in- and out-group (Aarts, 1998; Bogaert, 2004; Resource Analysis, 2003; Annex 'Theoretical framework').

From a sociological point of view, this unilateral 'ecocentric' approach could be opposed. Restoration goals should not be set exclusively by scientists (ecologists) and nature associations. For a particular site in a network, usually more than one state of nature can be created or restored. In that case, those restoration goals should be chosen which are likely to succeed ecologically as well as socially, as this restored nature is most likely to be maintained and thus survive in the long-term (Gobster & Hull, 2000). Contributions from the social context will be needed to help to decide the restoration goals, and to plan, implement and maintain the desired states of nature. Nature and restoration goals which fit better in the nature-images of local stakeholders are more likely to succeed (Bogaert 2004), and the willingness of stakeholders to co-operate in the restoration and management will be larger. What's more, integrating local ideas while designing or restoring ecological networks will help creating or/and enlarging local 'ownership' of the conservation of biodiversity. And this ownership will also determine sustained public support, and hence the success of an ecological network. An interesting idea about how to organize the co-ordination and development of ecological networks had been put forward by Mougenot & Roussel (2002). According to them, in shifting from 'sanctuary-based' nature management to 'ordinary natural area-based' nature management, the ecological network concept already broadens the scope of questions as to *what* is to be protected and *why*. Mougenot & Roussel suggest a classification covering three modes of project co-ordination. Involvement of stakeholders and used resources differ from one category to another. The first category of co-ordination – "negotiating nature" – is based on exclusion (notion of reserve), by re-negotiating (bargaining) the rights and obligations of current users, in terms of space and time. Main driving forces in negotiation

[109] For more information on the process of enrolment in the 'model of translation' of the Actor-Network Theory; see Annex 'Theoretical Framework', § 1.2.5. Power.
[110] Note, however, that to date, it has been the *planning* (mapping) of the ecological network in particular which prompted most thought and activity (Nowicki, Bennett & Middleton, 1996; Nowicki, 1998 in: Mougenot & Roussel, 2002; Rientjes & Wolters, 1996).

are economic incentives (price for expropriation, rent, subsidies, …). The second category – "sharing nature" – focusses on cohabitation, by linking different perceptions, functions and uses. Natural assets become common "border elements", permitting the coexistence of perception frameworks from the economic, cultural or social spheres as well. The third co-ordination pattern – "reconfiguring natural and human networks" – involves successive retranslations, which produce a new subject for joint management for all partners. In this case, the territory concerned is not only the support for different coexisting uses or functions; it becomes a fulcrum for preservering natural and also identity-linked or economic resources: it is elevated to the rank of 'heritage'. The authors furthermore suggest that these three co-ordination patterns may be seen as complementary, and may possibly be superimposed.

In conclusion, it can be stated that good and univocal communication is of vital importance in a process of nature conservation (Rientjes, 2000; Nowicki et al., 1996; Aarts, 1998). Question is within which process-design (reactive vs. creative) this communication must be set up. In the theoretical framework, elaboration has been done on the surplus value of an interactive 'creative' style. However, Leroy (2004) shows that good results can also be achieved by using other process-designs. Taking into account that most nature conservation projects involve actors with different perceptions and nature-images[111], we still dare to suggest that the creative process-design will be the most appropriate in most cases. In practice, the current (social) state of the area and the objectives of the project must be analyzed before choosing one or another process-design.

The effect of organizational structure could not be actively tested since there was no 'real' network to be implemented in the Dyle valley. Some recommendations concerning the organizational structure however could be made, taking into account the social survey of the Dyle valley and the sociological observation of the research in the ECONET-project.

4.2.6. Conclusions on theoretical framework

A theoretical framework has been developed to study the concepts of social perception, social impacts and organizational structure in relation to the implementation of ecological networks. Based on this enquiry, the research questions were made more specific for the Dyle case. Whether the selected sociological theories (see § 4.2.2) provided a useful framework to assess the social feasibility of the implementation of ecological networks, was also object of study.

Based on the theoretical elaboration on the three mentioned aspects in the context of nature and ecological networks, it can be stated that the general framework does enable the study of the feasibility of ecological networks. A sociological description of the mechanisms behind the aspects impact, perception and organizational structure could be given in terms of the framework. Moreover, an advantage of the general framework is that it not only enables the observation of the structural aspects of a certain phenomenon, but also the observation of the process in itself (evolution in time, etc.). By this, it seems suitable for the implementation of an ecological network, which is a clear dynamic theme. By means of the general framework, we also managed to explore (theoretical) connections and interrelations between impact, perception and organizational structure. Social impacts and the way people get involved in a process (=organizational structure), highly determine their perception of nature (policy). Impacts, however, can be real as well as perceived. With an appropriate process organization, and true communication, wrong perceptions (about potential impacts) may be adjusted and, by this, impacts extenuated. It may also help to develop a common nature-image. In conclusion we can state that, when assessing the feasibility of the implementation of ecological networks, all three above-mentioned aspects should be considered, as they all will determine the feasibility of an EN (see § 4.2.3 – 4.2.5). In this, their mutual coherence must be taken into account as well.

[111] Differences in perception and nature-image can be understood as differences in values. Like pointed out above, interaction will be necessary with a central role for civic organizations (see also footnote 108).

4.3. Case study: the Dyle valley

The 'theoretical' findings of chapter 4.2 were to be tested – and, if possible, improved – in the case study, i.e. the implementation of an EN in the plain of the Dyle valley, between Leuven (Flemish region) and Wavre (Walloon region). Three sample sites were chosen for the assessment of the feasibility of this implementation, namely site 1: transboundary-Pécrot, site 2: Doode Bemde-Neerijse and site 3: Egenhoven-university campus. The social research included a survey in the valley (questionnaire, exploration of sites), face-to-face interviews with users and a thorough literature review. Below, the results for the three social aspects (perception, impact and organizational structure) are briefly presented from the 'Dyle valley point of view'. The detailled results for the case study can be read in annex.

4.3.1. Perception

a) Perception of the natural environment

In general, people in the Dyle valley tend to have rather moderate nature-images, showing only few extremes. The decorative and broad nature-image can be found the most. Still, a significant part of the population, and farmers in particular, have a functional nature-image. Regarding the ground-attitudes, most people have the 'partner' ground-attitude, aside from the specific nature-image they might have. Respondents with the 'guard' ground-attitude often show a functional nature-image, while respondents with a 'participant' or 'partner' ground-attitude rather have a decorative nature-image.

Only minor differences could be detected between Walloon or Flemish respondents. The differences are larger between the three sample sites[112]. In site 1, the population is mainly concentrated on the countryside, less on the city. A considerable part of this area still is farming land. Consistently, the nature-image of the inhabitants is more function-minded, and the broad and functional nature-images are observed the most, in contrast with the other sites where the aesthetic value is more important. For the respondents in site 2, the natural environment is especially a décor/scenery, in which they live, dwell and spend their free time. Here, the aesthetic value is more important than the functional capacities of the area. In site 3 half of the people have the decorative nature-image. The other half has either the broad and functional nature-image, or the wild and autonomous nature-image. This could be a consequence of the mixed population in this site from a socio-demographic point of view (gender, socio-economic status, age, etc.).

b) Perception of the social environment

Little difference can be observed among the sample sites. Farmers, in general, seem to have a neutral attitude towards hikers and other recreational users in the area. Only the so-called 'eco-fundamentalist hikers' cause resentment. The relations with other farmers and hunters are assessed as neutral. Worth mentioning is the listing by one farmer of 'positive relations with the nature conservationists in the area'. Inhabitants show to have difficult relations with mountainbikers, quad-riders, motor crossers and hunters. Relations with other (soft) recreational users and other stakeholders in the area are experienced as to be 'neutral to positive'. Attitude of inhabitants towards farmers, however, is said to be 'neutral to negative'. Recreational users, for their part, show a positive attitude towards almost everyone they meet in the area. From time to time, hikers do come into conflict with the 'harder' recreational activities, such as mountainbiking.

4.3.2. Impact

To identify the social impacts of the implementation of an EN in the Dyle valley, a 'Social Impact Assessment' (SIA) has been performed. The other considered social aspects – i.e. perception (cf. supra) and organizational structure (cf. infra) – were used as mediating variables for the analysis of the impacts. Themes which are important in a SIA on nature conservation/development are: health and social wellbeing, quality of the living environment (liveability), family and community impacts, economic impacts and material wellbeing. Some summarising results of the SIA are presented below.

[112] To make the results of the analysis in the case study more relevant, the differences between the three sample sites have been artificially enlarged, be it based on the collected data.

A thorough description of the SIA methodology, of the studied criteria, the inventoried data in the sample sites and the results can be found in Annex 'Social Impact Assessment of the Dyle valley'.

In general can be stated that economic stakeholders in the valley, like farmers and owners of forests, will be most drastically affected by the implementation of the EN. The impact on inhabitants and recreational users will be rather marginal, as the EN has been restricted to non-urbanised areas and the nature conservation measures, as proposed by the ecologists, do not imply such things as, for instance, restrictions on accessibility of areas.

Site 1 is the most 'rural' of the three chosen sample sites. Agricultural use is prominent. The most important and clear social impact that can be expected in this site is the restriction of freedom imposed on a young intensive farmer, and on the owner of a large economically viable populus plantation. Hence, the implementation of the EN might trigger some social upheaval in this site. People mainly have a functional nature-image, with a rather reluctant attitude towards restrictive measures. The change of the landscape due to the nature conservation measures will, however, in general be perceived in positive terms, because of the transformation towards a more open landscape, which appears to be well appreciated by the actors in this area.

In Site 2, a considerable part of the area nowadays already is dedicated to nature development. So, the implementation of an EN seems to be socially feasible in this site. To a large extent, this can be attributed to the presence of the nature area 'The Doode Bemde'. Functions and activities in the area already are concentrated on, and adapted to, nature conservation. Economic activities, like farming, already are obliged to work under certain restrictions. Nevertheless, when implementing the network, the proposed extra restrictions on farming land in the nature area 'The Doode Bemde' will lead to a non-viable situation for the farmers present.

Site 3 is the most urbanized of the sample sites, with a significant part of the area in use of the University of Leuven. The current construction works in the northern part of the site (science park), and a recent extension of the IMEC-site, could impose a serious restriction on the feasibility of the EN. In addition, most patches in this site are proposed to be transformed into alluvial forest. This kind of intervention will impose restrictions on all kinds of land use. A slight deterioration of the attractiveness of the landscape might occur, since the landscape will become more monotonous after execution of the measures. Although people in this site appear to be the most tolerant towards nature development measures, wettening of the soil still may lead to a NIMBY-response from the highly educated and financially strong inhabitants. Disappearance of the science park and the newly developed part of the IMEC-site will cause social unrest, and does not appear to be socially feasible.

4.3.3. Organization[113]

a) Identification of stakeholders

The Dyle valley between Leuven and Wavre is, both in the Flemish as in the Walloon part, characterized by a relatively low degree of urbanization and a relatively high number of patches in nature area and agricultural use. Still, on the edge of the valley there are several small and moderate rural centers (e.g. Oud-Heverlee, Korbeek-Dijle, Neerijse, Nethen, Archennes, …). The Dyle valley falls within the range of influence of Brussels, which affects land and property prices, traffic in the area (recreation, tourism), etc..

The more important stakeholder groups in the Dyle valley are landowners, forest owners and farmers. These groups have not always been approached decently by nature conservationists. For example, the latter are known to have changed the water level in some areas without consulting or even mere informing other users. However, overall, not many conflicts nor opposition among stakeholders have been observed in the Dyle valley up to now. Nature conservationists and farmers even appear to collaborate quite well (Bart Vercoutere, pers. comm.). Another important activity in the area is

[113] Note that not all theoretical insights could be tested actively since there was no 'real' network to be implemented in the Dyle Valley. Therefore, we chose to stay 'on the sidelines' concerning our case (i.e. a *virtual* restoration project) in order to avoid fermenting needless trouble or interfering with on-going nature conservation or land use planning processes.

recreation (hiking, biking, fishing, riding, etc.). Most of this recreation is located in and around the nature reserve 'de Doode Bemde'. Worth mentioning is that most of these recreational users do not live in or around the study area. Nowadays, one could even observe an 'internationalisation' (the '*eurocrates*' from Brussels) of the recreation in the Dyle valley (Bart Vercoutere, pers. comm.).

When asked about their willingness to co-operate on the implementation of nature development measures, the inquired stakeholders in the Dyle valley showed a rather positive attitude[114]. This positive attitude especially counts with respect to management measures. Towards restoration measures, farmers show a more negative attitude. In § 4.3.1 the results are presented of the perception of the stakeholders towards nature on the one hand, and towards other actors in the area on the other hand.

b) Observation of own research process

During the process of each project, several decisions need to be made by project members and sponsors. These decisions give direction to the project. The content of decisions, in combination with the process of decision-making, give insight in the group process (rules, power, attitude, …) and the relations or communication between the different project partners.

From a sociological point of view can be stated that the following decisions guided the ECONET-project:

- Goal of the research was to develop a (integrated) method to analyse the feasibility of a scientific, theoretical ecological network. It was decided that interaction with local stakeholders, except for the nature conservation sector, was neither possible nor opportune in this virtual network planning. One of the consequences was a mono-sectoral users' committee.
- To test the method a case area was used, i.e. the Dyle valley between Leuven and Wavre. A lot of energy had to be spent to fully inventory and analyse the current situation in this transboundary study area, although it only concerned a fictive (virtual) network implementation. During this process, the methodological approach to create an integrated EN disappeared. In the end, only a multidisciplinary assessment of an optimal ecological network was performed.
- Demarcation of the case area: the case was limited to the valley bottom of the Dyle valley between Leuven and Wavre. Valley slopes and plateau were not incorporated as they represent different geomorphological entities (ecological landscapes). The Laan-valley and small parts of the Nethen- and Train-valley belong to the same ecosystem as the Dyle-valley and were taken into account. Limits were put on roads (in the west) and a railway (in the east), both at the foot of the valley slopes. In the north (Leuven) and in the south (Wavre), urban zones form the limit of the study area. Nature already occupies a favourable position in this case area (many nature areas, a nature reserve, …). It could be expected that this would influence (reduce) the number of conflicts with other users or uses in this area.
- Definition of the concept of an 'ecological network' as "*the whole of biotopes suitable for providing a maximum of species a temporal or permanent living environment, consistent with their requirements and guaranteeing surviving in the long term*" resulted in an emphasis on 'ecotypes'. Actual ecological barriers, like for instance roads or urban areas, were no longer relevant on this scale. Impacts only are patch-related (e.g. no restrictions on recreational uses).
- The planning (mapping) of the 'optimal ecological network' was done by the ecologists and restricted to areas which appeared (to them) to be legally and socially feasible at first sight. Hence, urbanized zones, transport infrastructure, etc., were not considered in their ecological analysis. This caused that one no longer could speak of a 'truly optimal' ecological scenario for the valley, and that certain social, economic and legal aspects in

[114] Given the large degree of positive answers in the survey carried out in the Dyle valley with respect to the co-operation to the execution of measures, in contrast with the real situation in Belgium, a misunderstanding of the question probably occurred in the survey.

relation to the feasibility, which could be expected to be important in a real situation, could not be investigated.

- The type of nature which was to be restored in the considered patches when implementing the network – the so called 'target biotopes' – was decided on by the ecological team, based on pure ecological criteria ('biologically valuable'; 'typical biotope in valley system'; 'worthwhile to make a network for from an ecological point of view'), and depending on the soil and groundwater regime on a particular spot. No contributions from the social sciences (society) were used at that point in research.
- The use and demarcation of sample sites in the case area to analyse the juridical, economic and social impacts. Assumptions concerning the feasibility of the whole network in the valley no longer were possible. On the other hand, by means of the sample sites it became possible to test the methodology 'in situ' in different situations.
- As agreed upon among the partners, the ecological team proposed a list of necessary (technical) measures (restoration and conservation), needed to reach the particular goal ('restore the actual non-target biotope into a target biotope') in a patch. Measures are thus focussed on patch level (cf. supra), and exist out of both restoration and management actions. No information was given on whom they expect to implement the measures (e.g. nature conservationists or current users), nor how (e.g. voluntarily vs. obligatory). The measures do not include restrictions on accessibility of areas (e.g. no restriction on recreational uses).

Looking back to the ECONET-project in general, communication as such was not always easy. First of all, the research team consisted out of Walloon and Flemish researchers, while using English as the working language. It was sometimes difficult to clearly express the things as intended, or to express slight distinctions – not unimportant in a context were a 'common language' is of utmost importance (cf. supra). Secondly, the ECONET-team consisted out of people with different backgrounds and different expertises, and, consequently, different interpretations (and sometimes misunderstanding) of certain concepts were not uncommon.
Another remarkable aspect concerning the group process was that rules, for instance about decision-making, were absent. This lack of rules sometimes led to long and difficult discussions, and confusion about the outcome. On the other hand, everyone was free to give his or her opinion during the research process.

4.3.4. Conclusion: feasibility of EN in the Dyle valley

When taking into account the results of the Social Impact Assessment, it can be stated that the impact of the proposed restoration of an EN in the Dyle valley, and more specific in the three sample sites, remains relatively limited, aside from some bottlenecks. The most significant impacts are to be found in the agricultural and sylvicultural sectors, as could already be expected from the nature of the proposed (technical) measures, since farmers and foresters do use the valley for economic purposes. Another important impact is to be found in the northern part of the case area. Buildings are currently under construction in a patch that should become 'alluvial forest' in the network. The intended economic use would no longer be possible. However, social impacts will of course also vary depending on the judicial and socio-economic instruments used for implementation.

Thus, as already said above: in general, the proposed scientific EN could be feasible from a sociological point of view. This is to a large extent due to the fact that, apart from economic activities as agriculture and sylviculture, overall little change for human activities is to be expected. Little to no effects were observed for, for instance, tourism, inhabitants, industry, transport and mobility, etc.. If ecological measures were more drastic (e.g. limited accessibility of certain areas, removing existing roads (barriers), …), this social feasibility would probably decrease.

Because of the virtual aspect of the project (the designed ecological network will not be implemented in practice), it was difficult to analyze the aspects of perception and organizational structure of the social feasibility. Perception of nature and nature-images of (economic) users of the valley are quite

different from perception and nature images of the scientific designers (ecologists) of the network. Surprisingly, however, is the willingness among stakeholders in the Dyle valley to co-operate to the implementation of an ecological network. Question is whether this willingness will still be as large in practice, when a certain nature-image (restoration goal) would be imposed. If the EN in the Dyle valley is to be implemented for real, enough attention should be paid to the organization of the process of implementation, and the perception of local actors of the policy process. It seems that more moments of harmonization of ecological and other interests (social, economic and legal) may be needed in order to be able to succeed in a socially feasible implementation of the network. Without true interaction, the feasibility of the proposed ecological network might be much lower than could be expected, based at the SIA in the sample sites.
To conclude we can state that the ECONET-research process in itself was rather multidisciplinarily approached – instead of interdisciplinarily –, with an emphasis on the ecological aspects. The process appeared to be a succession of reactive compromises.

4.4. Discussion and conclusions

In the sociological sub-study the question of 'which social aspects determine the feasibility of the implementation of an ecological network?' has been explored. The hypothesis has been investigated that social feasibility is mainly determined by the following tree aspects: social impact, perception and organizational structure. Below, a summarizing answer will be given to the general research question and the different hypotheses that have been assessed in relation to the three aspects will be discussed.

a) Perception
Overall, people appear to have rather moderate nature-images, showing only few extremes and little difference between different stakeholder groups. In the survey carried out in the Dyle valley, for instance, the decorative and broad nature-image happen to occur the most. The functional nature-image is also found within a significant part of the population, and especially among farmers. Only minor differences were detected between different areas and subpopulations. Nevertheless, as the concept of ecological networks in principle starts from a more 'pure' image of nature, i.e. 'wilderness' and a more autonomous nature-image, only little agreement on the basic ideas about nature and nature conservation (restoration) can be observed between, on the one hand, policy-makers and other initiators of EN implementation (e.g. nature conservationists, environmental scientists), and, on the other hand, local stakeholders and the general public (cf. De Zitter et al., 2002; Annex "Experiment with user-group"). The first hypothesis in § 4.2.3 can therefore be endorsed. Harmonisation of opposite nature-images or altering of conservation practice appears to be needed to come to a feasible implementation of an EN, i.e. with support and understanding from the side of (local) stakeholders. Or, in terms of the systems theory, it can be stated like this: communications and decisions of the political sub-system will not spontaneously be adopted in the communication of local social sub-systems (stakeholder groups). Both ECONET-researchers and -users had, more or less, a similar nature-image. In particular, there was consensus about the goals of an EN as to preserve biodiversity. The concept of an EN consisting out of core areas, corridors and bufferzones, each with an own function or goal for biodiversity conservation, however, was a subject of discussion among the team. Some people considered the case area of the Dyle valley as one big core area, while others made (or wanted to make) spatial differences between core areas, bufferzones and corridors. Moreover, the ecological team used a new discourse regarding EN. Therefore, the second hypothesis in § 4.2.3 can be largely endorsed as well.

The results of the sociological research carried out in the ECONET-project are, to a large extent, coherent with other existing research results. A lot of research already has been performed on perception and the way perception determines nature conservation. A lot of suggestions already have been done on how to handle this perception from a theoretical perspective (for instance: Bogaert, 2004; Aarts, 1998; etc.). The question remains how to practically integrate the subject of perception in a planning process. Do the ecological goals need to be altered – for instance: make them fit in the nature-image of local stakeholders –, or can sociological themes be taken into account when defining restoration goals? How to use the contributions from the social sciences to help decide restoration goals, and to plan, implement, and maintain desired states of nature? Further research on this in empirical situations is clearly needed.

b) Impact

The methodology of the social impact assessment (SIA) is developed to gain an insight into potential social impacts, which might occur as a consequence of the implementation of a certain project or policy. A crucial aspect of a SIA is that it considers more than pure socio-economic impacts (as does, for instance, a cost-benefit analysis). A SIA also incorporates concepts of social cohesion, social tension, etc., and takes into account the background of an area and its stakeholders (i.e. profiling step). For instance, information about local stakeholders' perception of nature and their perception of other actors can be used as a 'background' or framework for the assessment of potential impacts. Question was then if such a SIA in particular could help to gain an insight into the feasibility of ecological networks.

Methodologically, it was not easy to determine the crucial social impacts of the implementation of an EN. The SIA-method in general is still developing and there is no – or not yet – universal agreement on impacts, criteria and indicators among scientists. Moreover, until now expertise on how to perform a SIA is mostly gathered during the assessment of global social impacts of big infrastructure works, like for instance the construction of dams or roads. Defining the right criteria and usable indicators in the case of the assessment of the implementation of an ecological network on local scale was therefore quite a difficult exercise. The current research gives an initial impetus to a framework for the assessment of the impacts of nature conservation. Due to the typical nature of the (technical) restoration and management measures, the SIA was more or less limited to the socio-economic effects of the conversion of land uses. Due to the virtual character of the case, assumptions were needed to estimate the impacts, since direct interaction with stakeholders was not always feasible. Hence, it is difficult to point out the exact value of a SIA in the context of the implementation of an ecological network. Anyhow, to affect the end result (and obtain a better result, i.e. a more feasible EN) the SIA should be performed *during* the planning process, and not just at the end. Results of the SIA could then be integrated in the ecological design. As such, the SIA would become more than a mere product and social input would be used in the whole planning (mapping) phase of the EN. In conclusion could be stated that the SIA as such is a useful tool to analyze the social feasibility of the implementation of an EN. Integrating the philosophy and approach of a SIA (indicators, tools for inventory and assessment) in the planning process appears to be useful to come to a socially feasible ecological network.

c) Organization

Different sources (public authorities, local users and scientific literature) emphasize the importance of the involvement of certain stakeholder groups – namely farmers, foresters, nature conservationists and land owners – in the process of implementing an EN. These groups are mentioned as 'to be actively involved' in case of an ecological restoration project. Both recreational users, like for instance bikers, hikers and anglers, as well as the industrial sector, appear to have less impact on the feasibility of the network. Combining this knowledge with the fact that many of the former actors have a nature-image which is in itself quite opposite to the nature-image on which the concept of an EN is based (cf. supra), it seems indeed of utmost importance to involve these actors in the process and their needs and interests deserve particular attention. In terms of the systems theory, it can be put like this: a more structural link between the mentioned social systems is needed and these systems should be made part of one larger social (organizational) system. Doing so, all actors would become member of the new social system, which will improve the observation of the mutual communications. These communications will affect the communications in the respective social system of each stakeholder and, consequently, become a topic in this system (which may, e.g., increase the awareness of shared needs and issues etc.). Hence, it can be stated that an interactive planning process will probably increase the feasibility (sustainability) of an EN. However, the specific situation of the case – current (social) state of the area, history regarding nature conservation, objectives of the project, and so on – should always be assessed first, before choosing an appropriate process-design. There is no such thing as 'the' process-design, which will be suitable in all cases.

It was not possible to clearly analyse and assess the aspects of process-design in the case study, as it was a virtual case; no real process of implementation was to be initiated. However, the observation of the ECONET-research process in itself already confirmed some points of particular interest of a multidisciplinary approach. All 'players' had their own language (jargon), perception and an own way of doing things. Further research should point out how to achieve optimal harmonisation. Looking back upon the process, one could say that the ultimate goal of the ECONET-project – a feasible and integrated design of an EN in the Dyle valley – was rather aimed at by means of a 'reactive compromise' instead of a 'creative compromise' (Aarts, 1998). The ecological perspective and findings formed the starting point, on which the other disciplines consequently hung their own research. The demarcation of the case area, for instance, was also done by the ecological team. One could further say, in line with Aarts (1998) and Vandenabeele (2004), that there was a strong emphasis on the different positions in this project (different scientific perspectives). The true interaction actually took place at the end of the process. This makes that one cannot speak of a real *inter*disciplinary approach, but rather of a *multi*disciplinary approach. Misunderstandings among the partners could have been prevented if the need for a common language (Rientjes, 2000) would have been recognised from the beginning. The problem of the absence of a 'common language' and rules could perhaps be solved by involving an independent facilitator in this research process.

d) General conclusions

From the theoretical framework, it can be derived that both perception, social impact ànd organizational structure indeed are important to gain an insight into the feasibility of the implementation of an ecological network. Furthermore, the three aspects are clearly interwoven, as can be concluded based on our theoretical and empirical results. Question remains which factor is the 'driving force', when considering the feasibility of the implementation. German research concluded that it is not the competition for land nor the lack of knowledge which are the 'core' issues of the feasibility of nature conservation, but the *perception* of the different actors involved (Stoll-Kleemann, 2001). Question is, however, if this is also the case in the Belgian context where pressure on land is higher. Further research on this is needed by means of observation and analysis of 'real' cases of ecological network implementation.

Besides this, it can also be stated that the integration of perceptions of stakeholders and (real, true) communication between these actors is crucial for the feasibility of the implementation of an EN. Question is to what extent ecological (restoration) goals may be altered to come to a sustainable ecological network from an economic, social, legal ànd ecological point of view. Depending on the answer on this question, one of the models of co-ordination and approach of nature conservation, as presented by Mougenot & Roussel (2002), could be chosen.

Due to the virtual aspects of the research project, the construction and exploration of a theoretical framework and the elaboration on a methodology for integrating social feasibility in the implementation process have been emphasized in this social sub-study. In general, it can be concluded that the theoretical framework seems appropriate to frame and assess the social feasibility of the implementation of an ecological network, even in an inter- or multidisciplinary setting. The coherence and interrelations between the three different social aspects, which can be used to assess the feasibility and can be found in the empirical world, are also framed by the sociological theory. In addition to this, the systems theory (be it combined with other sociological theories) also addresses in particular the aspects of progress of a policy-process. Nevertheless, the practical translation of the theoretical findings into the Social Impact Assessment, including the used criteria, could be improved. Further research and practice in real situations of restoration projects is necessary to test and optimize the approach and theory used in the ECONET-project.

PART 5: ECONOMIC ASPECTS

5.1. General introduction

Economic aspects of environmentally-oriented projects are crucial. One can argue about the inappropriateness of associating a monetary value to environmental goods, on ethical and/or ecological grounds. However, in our society, most decisions are taken on economic bases; ignoring such a fact in environmental projects might simply be misleading or lead most actions to inexistence. Economic assessment of environmental projects often provides for essential information towards decision.

The goal of the economic part of the project is thus to evaluate the costs and benefits linked with implementation of ecological networks (ENs), and to assess the economic feasibility of such a project. An essential assumption, made throughout this chapter, is that the reasoning will not be based on absolute values, but instead, on differential values between the existing situation and the one that would result from implementation of the projected EN.

In the theoretical part, we first review the concepts of biodiversity and ecosystem valuation. We next identify all costs and benefits linked with EN implementation. In the case study part, we first explain our methodological approach. Next we enumerate the collected data and their treatment in relation with the selected sample sites. Finally, we conclude on the economic feasibility of EN implementation and propose recommendations to facilitate acceptability of such a network by all stakeholders.

5.2. Theoretical background

5.2.1. Economic aspects of biodiversity

a) The components of the value of environmental goods

The value of environmental goods can be decomposed into several components. A first subdivision appears between use and non-use values, which can be further decomposed into direct use value, indirect use value, option value, bequest value, and existence value (Turner et al. 1993; see also Appendix 1).

b) Biodiversity valuation

Several methods exist for biodiversity valuation (Hanley & Spash 1993; see also Appendix 2). OECD established a matrix of environmental sectors and valuation techniques that are best suited for each of them[115].

c) Links between biodiversity and ecological networks

Ecological networks allow for the improvement of surfaces and connectivity of natural areas. This has a positive effect on biodiversity and ecosystem functioning. Indeed, ecological networks improve the viability of the populations that constitute the ecosystems. They contribute to enhance their stability and functions, and therefore, they allow improving the services rendered and the benefits provided by ecosystems (Jongman & Pungetti 2004 - see Fig. 5.1).

In the following we shall try to associate costs and benefits to the different steps depicted in Fig. 5.1 :

- Implementation of ENs implies costs, both direct (installation of physical measures) and indirect (linked with lost opportunities for agriculture, forestry, industry, … whose activities are, as such, incompatible with the presence of ENs);

[115] See Appendix 3

- Improvement of biodiversity has direct benefits, linked to tourism, recreation, employment in the environmental sector, and to the existence value of species;
- Improvement of ecosystem functions also offers indirect benefits, connected to services such as carbon sequestration, hydrological effects, species habitats, etc (Costanza et al. 1997).

This can be illustrated by Costanza et al.'s study (2004): there exists a positive correlation between the number of species present in a given territory and net primary production, and this at different geographical scales[116]. Further, one can see that the net primary production of a territory is positively correlated with the economic revenue that results from this[117].

Implementation of ENs also has a significant socio-economic impact, in terms of land use and activities that can be developed. It is thus necessary to evaluate all costs and benefits of ENs.

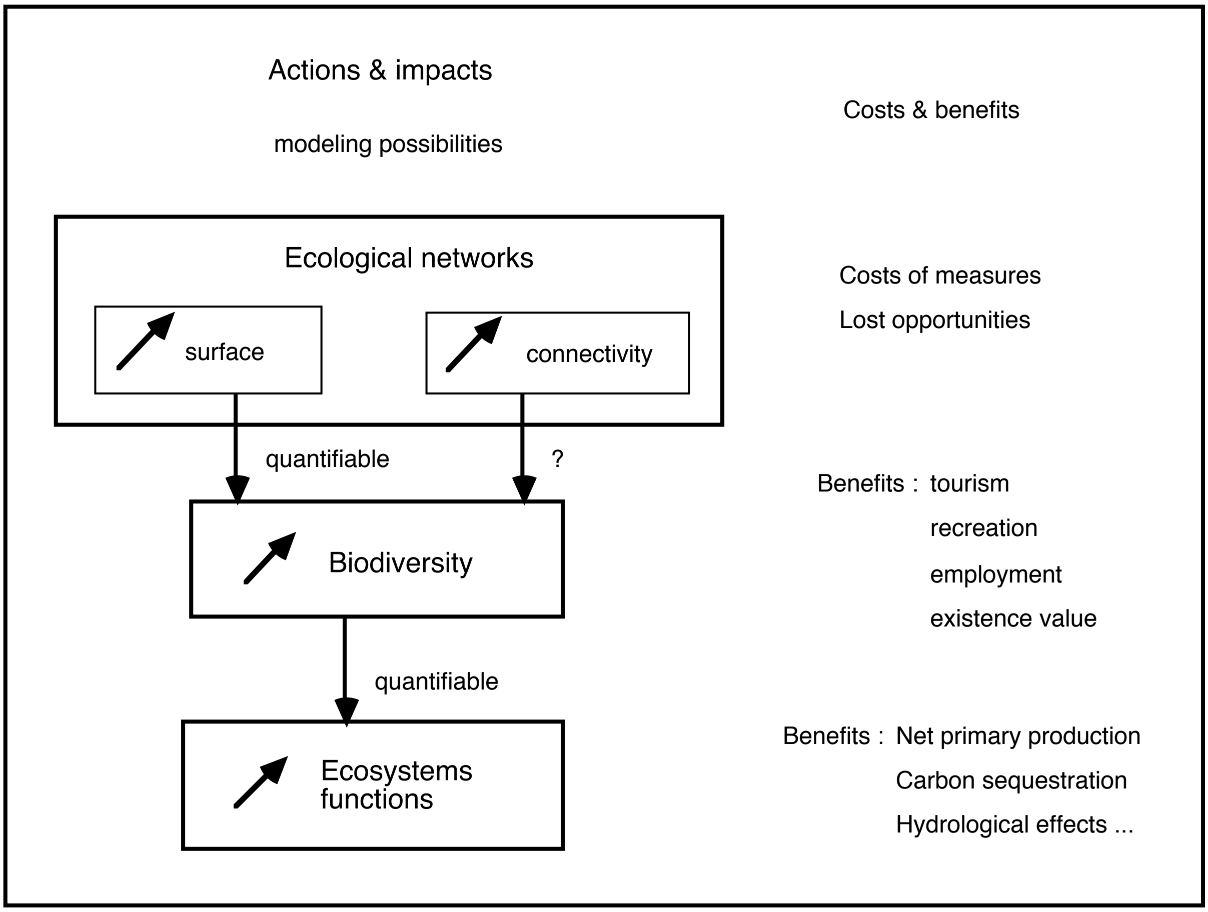

Fig. 5.1. – Actions and their impacts on biodiversity and ecosystems, modelling possibilities, and corresponding costs and benefits.

5.2.2. Costs and benefits of ecological network implementation

Here we take the standpoint of society as a whole, i.e., we envisage all costs incurred or supported, and all benefits perceived, independently of who bears or receives them. The situation might be extremely different if we take the point of view of one particular actor: this point will be discussed subsequently.

[116] See Appendix 6.
[117] See Appendix 7.

a) Costs

1. Implementation and direct costs

Direct costs are supported by the landowner, which may be a public authority, an organisation or a private owner. In the latter case, compensations and subsidies (see below) can be perceived to cover incurred costs. Costs of EN implementation can be divided into four categories, corresponding to the four phases of EN installation[118].

1. Predesignation phase

The constitution of ENs is preceded by scientific studies to designate and delimitate the sites that will make part of the network. This preliminary diagnostic phase will also serve to define management objectives of the site. The costs linked with this phase are essentially salary costs as well as cost of the material that will be used by the researchers. Obviously they are dependent upon the quantity of work accomplished as well as duration of the research; therefore, they may vary from site to site.

2. Management planning and administration

This phase is essentially administrative. It is necessary to prepare and examine the projects and strategies and to define management plans of the sites. Next comes the work necessary for consulting and meetings with public authorities and municipality representatives. Those costs are essentially due to salary, representation, meetings, etc. As for the first phase, they vary from case to case, depending on the progress of research, areas to be covered, etc.

The costs of those first two phases correspond to non-operational charges, i.e., salary, scientific personnel, office material, etc. Their assessment is difficult, due to the diversity of stakeholders, actions to be taken and material to be purchased, and depend on the actual situation. However, they can be estimated to lie between 10 and 25 % of the global implementation cost[119].

3. Implementation of the EN

This represents the most important costs for the community. It includes the financing of maintenance and restoration measures of the various zones of the EN. Generally speaking,, management is accomplished by the owner of the land, who bears the costs and signs a management contract with authorities. Such a contract incorporates the tasks to be accomplished by both parties, as well as financial support granted to the owner (i.e., subsidies). When public authorities are the owner, they will incur the financing of the project.

The work activities to be conducted in those zones are of two types: **conservation** and **restoration**.

Conservation of the zones

This approach consists in maintaining the natural and semi-natural habitats in their present state. This constitutes one of the most expensive activities of EN implementation. It is a hard and continuous task that requires qualified personnel as well as adequate machinery.

Restoration of the zones

In some cases the habitats can be in a bad state and must be restored. The costs generated by such a restoration are often important at the outset. However, in subsequent phases, costs tend to decrease since after restoration everything that is required is site management.

[118] Form the final report on Natura 2000 Financing, by the Task Group on article 8 (November 2002).
[119] Colas S., Herbert M., « Le coût de la gestion courante des principaux milieux naturels ouverts », *Espaces Naturels de France*, 2 février 2002, p. 1.

4. “Occasional” management costs

Those costs cover situations that were unforeseen in management and expense planning. They would be incurred by owners or by public authorities. Their importance varies from place to place; however, they only represent a small part of total costs.

Costs of phases 3 and 4 are not easy to estimate because they depend on the biotope concerned, the state of the land, available material, etc. They can be assessed through reference to other similar projects or previous studies. Care must be taken to make the characteristics of existing projects correspond to those of the new project to be evaluated (in terms of biotope and required measures).

2. Negative impacts – indirect costs

Here we must account for the loss of revenues linked with the forsake of particular activities which are judged incompatible with the implementation of an EN, such as intensive farming, polluting industrial activities, unsustainable forest management, etc. This has consequences on employment and economic development of the region under concern.

We can estimate those costs using the opportunity cost method (i.e., considering the revenue that will be lost due the cessation of those activities), or through the amount of compensation that should be paid for this sake, or by calculating the loss of real estate value of the field.

3. Conclusion

We can say that the costs related with the implementation of ENs are mainly of two types : administrative costs and those due for restoration and management of the sites. Administrative and restoration costs are those that will require the most important budget at the outset, but these costs will tend to decrease as the implementation of the network progresses. Indeed, as soon as the EN is implemented, the management costs will be the most significant to incur. In Appendix 8 we present a recapitulative table with the different costs associated to ENs in the Walloon Region[120]. We see the situation from the standpoint of expenses made by the Region; this is why the acquisition costs and the premiums granted are taken into account.

From that table one can notice that the most significant costs are thus those relative to land exploitation, which includes maintenance but mainly the compensations paid by the Walloon Region, which are indeed the most significant expenses incurred by the Region (see below). The land purchase costs will go on to decrease, because the number of fields to be acquired will diminish, at least in the long term. On the contrary, the conservation costs are expected to go on increasing in the short term, due to increase in the surface area to manage, but will tend to stabilise in the long term. Those figures must be considered carefully, although they provide us with a rough estimate of the budgets that will need to be devoted to ecological networks.

b) Benefits

1. Socio-economic, direct benefits

Socio-economic benefits of a particular site are not restricted to the site itself; they are also a source of profit for the local and regional economy. Indeed, the expenses of a local economy beneficiate to many persons through the purchase of goods and services. A study was undertaken by the European Commission in the scope of the Natura 2000 network (Institute for European Environmental Policy – IEEP, 2002), to assess the types of benefits or opportunities that can be derived from the implementation of ENs. Hereafter we present a non-exhaustive overview of that study.

[120] See Appendix 8.

1. Employment level

A non-negligible number of employments can be created through the installation of an EN[121]. The creation of new work positions is more a social benefit; however it implies strong economic repercussions as well. Employment opportunities include those that are directly linked to site management, such as guardians, the personnel in charge of conservation, and educators. Those jobs may also incorporate the work connected with agriculture and land production, fisheries, as well as the workforce in charge of services on the site, such as hotels, guest houses, restaurants, etc.

In this case, we do not obtain a direct monetary value, but instead, a number of job positions created, which is a good indicator of the surplus transmitted to the region by the project.

2. Tourism

Quite viable tourism activities are compatible with the management of most nature protection sites[122]. Tourism can generate high revenues and employment in the region. Generally, secondary benefits have a stronger economic impact than the benefits obtained directly from management of the site. This can be explained for example by the expenses made by the tourists outside of the site (hotels, transportation, food, other goods and services). Of course, this depends on each particular site, on its location and attractiveness, etc. All of those opportunities for additional revenues provide local populations with economic incentives, in such a way as to enable them to implement nature management practices that are more favourable to local flora and fauna.

Those benefits can be estimated in comparison with those generated by existing projects of tourism-related exploitation of natural sites, or through simulation of the number of entrances as function of the fee.

3. Production and exploitation

The majority of benefits result form exploitation of resources of the network as such (Moons et al. 2002). Depending on the situation, they originate from agricultural or forest exploitation, fishing, or hunting. Indeed, among the main advantages of ENs is the fact that many human activities are still perfectly compatible. Due to the diversity of EN components, it is certainly not possible to give a comprehensive view of all such activities. In the following points we will concentrate on two of those aspects, i.e., agriculture and forestry[123].

What must be assessed is the revenue increase allowed by EN implementation. Substantial benefits can be derived from natural sites conservation. Indeed, many consumers show growing interest towards biological, recycled, land products[124], ... For example, in Belgium, the turnover resulting from biological agriculture is estimated at 62.5 million Euros per year. The number of biological farmers is continuously growing as well. The surface area occupied by biological agriculture, even if still limited, shows significant increase (500 ha in 1985, 5000 ha in 1996, 18000 ha in 1999)[125]. Moreover, various surveys show that consumers are willing to pay more for products with a particular identity (system of quality label, products from the land, etc)[126].

[121] European Conference: “Promoting the Socio-economic Benefits of Natura 2000”, Brussels, 2002.
[122] European Conference: “Promoting the Socio-economic Benefits of Natura 2000”, Brussels, 2002.
[123] Those two sectors represent the highest proportion among the various components of the network. For example, only in the Walloon Region, forest areas (32%) and useful agricultural surface (45%) account together for 77% of the territory[123].
[124] Nature et Progrès Belgique : http://www.natpro.be/articles24.htm.
[125] See Appendix 9.
[126] Moons et al., op. Cit., p.6.

It is also worth insisting on the importance of sustainable forestry, yielding significant revenues. Thus, the European forest sector generates an annual turnover of 400 billion Euros, provides work to 3.5 million employees and contributes as much as 9 % of the value added by the manufacturing sector[127]. The Belgian forests are among the most productive in Europe. Their timber production generated 3.8 million m^3 in 2003[128]. Unsustainable forest management might therefore lead to destruction of this revenue source in the future.

To estimate agricultural and forest benefits connected with EN implementation, one has to evaluate the revenue surplus generated by additional activity after EN installation. For example, one can consider biological agriculture revenues, minus traditional agriculture revenues (i.e., the situation that prevailed before the EN); the difference yields the surplus. One can also assess those benefits through the opportunity cost method; i.e., unsustainable management of forest would lead to its disappearance, and hence, to revenue losses in the long term.

2. Environmental, indirect benefits

Indirect benefits pertain to goods and services provided by ecosystems. These are the functions fulfilled by ecosystems that allow for the maintenance of life on Earth and sustain economic activities such as thermal regulation, climatic regulation, oxygen regulation, water cycle, etc. Implementation of ENs contributes to quality and quantity improvement of natural ecosystems and thus allow for improving the benefits associated with those. Such indirect benefits exert their effects on the (very) long term and participate in the increase of the quality of life of the whole Planet. Beneficiaries are therefore society in general, at the world level and on the long term.

Advantages of ENs can be quantified by using direct valuation methods such as contingent valuation, in order to perceive how much people would be willing to pay to maintain or improve the services rendered by natural habitats and ecosystems. This method, however, entails several biases and limits as we described in Appendix 2. Another method consists in considering as benefits the costs avoided thanks to measures taken to prevent a pollution or loss of habitat. As a final alternative, for each considered ecosystem, one can attempt to identify the services they provide for, to adopt appropriate methods and to estimate their monetary value increase linked with EN implementation. Costanza et al. (1997) have developed a methodology that allows calculating the average annual specific (i.e., per ha) values for several types of ecosystems, accounting for the services they render. We can use their values by considering that the implementation of ENs will allow for substituting one given biotope (B0) by another biotope (B1) of improved ecological quality over the corresponding area, and by calculating the corresponding increase of value when one passes from B0 to B1.

3. Conclusion

The benefits associated with EN implementation will yield their beneficial effects mainly on a mean to broad scale: the region in the case of socio-economic (direct) benefits and the world's society in the case of environmental (indirect) benefits. We would like to insist on the difficulties to estimate such benefits: no single universal evaluation method exists; instead, choices must be made and assumptions must be emitted for every particular situation. Only after resolving the case study we will be able to know which category of benefit is predominating; there exists no a priori method to detect this beforehand.

Finally, we should bear the attention of the reader to the fact that the list of benefits such as we established is certainly not comprehensive and reflects only a part of the value that we can attribute to environmental goods. Indeed, we account herein mainly for direct and indirect use value but we omit or under-estimate the option and non-use values (bequest and existence).

[127] CEPF, COPA-COGECA, ELO, USSE, « Un cadre politique pour le modèle européen forestier », 23 septembre 2003, Bruxelles, 6 p.

[128] Herren G., «Le bois depuis la forêt jusqu'à l'habitation » : http://www.bois-habitat.com/presse/gerdherren/gerdherren.html.

c) The Net Present Value (NPV)

After all components of costs and benefits have been evaluated, we can compute the Net Present Value (NPV) of the project, i.e., over an adequate time horizon, we sum up the net yearly benefits (benefits minus costs), each multiplied by the appropriate discount factor.

Discounting is necessary for homogeneity of economic valuation. It allows for putting at the same level present and future amounts of money. The discount rate represents the social preference with respect to time: the higher the discount rate, the most important is the depreciation of the future. The general formula of the NPV is as follows:

$$NPV = \sum_{t=1}^{n} \frac{B_t - C_t}{(1+r)^t}$$

- C_t , B_t – marginal costs and benefits associated with EN implementation, with respect to the initial situation;
- r – discount rate;
- n – number of years.

Discounting is often criticized in environmental management. Indeed, when the rate value is high, the result is to strongly decrease the present value of future amounts. A high discount rate discourages investment, especially when it implies high initial expenses, while benefits would occur only after some years. Expenses made in the scope of environmental conservation purposes, because they would have their positive effects only in the long term, would thus be discouraged.

One can assume that public authorities who would subsidise a given project, would expect a return at least equivalent to other alternative investments. The interest rate on financial markets is often taken as a reference towards discounting[129]. However, a commonly advocated drawback of the interest rate is that it only reflects the preferences of present generations with respect to time. Therefore it can be reasonably admitted, for public projects, to take into consideration a lower discount rate that will account more adequately for general long term welfare[130].

Another drawback in environmental economics is the scale difference between payers and beneficiaries. Persons who bear the costs are often local communities (public authorities and residents), who pay the costs here and now, while beneficiaries are much more dispersed in space and time, and might include society as a whole, at the world level, on a much larger time horizon.

d) The situation of target actors

Hereafter we shall discuss financial elements that are not incorporated in the previous enumeration of costs and benefits, because they constitute a cost for one actor and a benefit for another one. Incorporating these in our previous analysis would therefore imply double accounting of certain elements (for example, premiums are granted to compensate management and restoration costs). Nevertheless, those elements are of uppermost importance when seen from the standpoint of a particular actor. Therefore, for each element reviewed hereafter, we will identify to whom it beneficiates and to whom it costs.

[129] Hanley et al., op. cit., p.16.
[130] Hanley et al., ibid., pp.129-130.

1. Subsidies[131]

Premiums are a cost for public authorities who pay them. In Belgium, those who pay and manage such premiums are the Regions. They are a financing source for the beneficiaries (farmers, forest owners, local authorities, etc). Indeed, if we wish to favour management and operation measures that are more oriented towards environmental conservation, this often implies profitability loss for the owner. This loss is compensated by the system of premiums. Among these, in the Walloon Region, are agri-environmental measures[132], premiums granted in the scope of sustainable forest management[133], and other operations and decisions in the Region[134].

2. Real estate acquisitions[135]

In cases where it is absolutely essential (from an ecological standpoint) to intervene on a given area, whereas the owner shows strong opposition, public authorities may decide to buy the land in order to exert appropriate management. This thus implies a cost for public authorities, and a revenue for the owner.

3. External funds[136]

Site management and development are supported by local, regional or national funds and investments, but also by European funding programmes. These are thus a financing source for site owners and managers, but a cost for the paying authority or organism.

4. Synthesis

To evaluate the economic situation of EN implementation from the standpoint of a target actor, we would have to consider all costs and benefits that have significance for him/her, and thereafter calculate the net present value for that actor. In summary,

- Direct costs: land owner, public authorities who deal with the project;.
- Indirect costs: land owner, residents (local economy);
- Direct benefits: owner, residents (local economy);
- Indirect benefits: society as a whole;
- Premiums: cost for public authorities, benefit for owners;
- Acquisition: cost for public authorities and organisations, benefit for the owner;
- External funds: cost for paying organisation, benefit for the owner.

5.3. Case study

5.3.1. General methodology

Let us consider the example of a territory composed of five patches, on which an EN has to be implemented. Preliminary studies conducted by biologists, agronomists, etc, allow for assessment of the initial conditions of those patches, the target biotopes adopted for the EN and the potential habitat of every patch (see Table 5.1 – it might also be proposed to keep the patch in its initial state whenever it is already a target biotope). This corresponds to the approach adopted by the teams of ecologists.

Next, some hypotheses are made:

[131] See Appendix 10.
[132] See Appendix 10.
[133] Ministère de la Région Wallonne - Direction de la Nature et des Forêts and see Appendix 10.
[134] Ministère de la Région Wallonne – Direction des ressources naturelles et de l'environnement and see Appendix 10.
[135] See Appendix 11.
[136] See Appendix 12.

- Corridors are not taken into account (hypothesis also made by the other teams);
- Biotope 0 is considered as having no (ecological) value;
- The global (ecological) value is optimal for the first choice of ecological teams ("potential 1");
- Every transition has specific costs and benefits per unit of territory.

In a further step, the ecologists determined the measures that should be taken in every patch to allow for the transition from a given initial biotope towards a target biotope. Hence, for every possible transition (or restoration) from Bi to Bj, the following elements must be determined (Table 5.2):
- List of measures to be taken;
- List of costs and benefits per territorial unit;
- Data collection;
- Cost and benefit assessment (towards NPV calculation).

Costs are associated with each category of measures, while benefits are the result of the whole biotope transition.

Table 5.1. – Potential transitions for a hypothetical territory composed of five patches.

Patch	Initial state	Potential 1	Potential 2	Potential 3
1	Biotope 0	Biotope 1	-	-
2	Biotope 2	Biotope 3	Biotope 4	-
3	Biotope 1	Biotope 1	Biotope 6	Biotope 6
4	Biotope 1	Biotope 5	Biotope 7	-
5	Biotope 2	Biotope 4	-	-

Table 5.2. –Sample list of costs and benefits associated with the transition from biotope Bi to biotope Bj for a hypothetical patch.

<table>
<tr><td colspan="5">Transition k: Bi → Bj</td></tr>
<tr><td>Measure 1</td><td>Implementation
(direct) costs:
Investment
Maintenance
Personnel
Restoration
Compensation</td><td>Indirect costs:
Revenues lost from industrial, agricultural, ... activities</td><td rowspan="3">Direct
benefits:
Tourism;
Recreation;
Employment;
Existence</td><td rowspan="3">Indirect benefits:
Net primary production;
CO_2 sequestration;
Hydrological effects;
…</td></tr>
<tr><td>Measure 2</td><td>Implementation
(direct) costs</td><td>Indirect
costs</td></tr>
<tr><td>Measure 3</td><td>…</td><td>…</td></tr>
</table>

5.3.2. Methodology for the case study

The site selected for our case study is the Dyle valley. The ecologist teams have determined the initial habitats of the whole site, the target biotopes selected for the EN, as well as the potentials for every patch to sustain each of the target biotopes. Hence we were confronted with a very large amount of possible transitions, each of which would have to be calculated. For time availability reasons, the ecologists have then decided that only the first transition had to be selected (best ecological value). Thus they defined the optimal ecological network for the Dyle valley. In the following we will proceed to economic valuation of that network only; hence for every patch we will quantify only one transition.

Even so the task was enormous (data collection and economic valuation). In agreement with the legal and sociological teams we decided to restrain the evaluation to three sample sites, selected within the case study site. The data collection and processing will thus pertain to those three sample sites.

For each of these, we began by enumerating all patches and identifying, for each of them, the initial habitat and the selected target biotope, so that all transitions were identified. The ecologists detected 19 different (initial) biotopes within the site, which were grouped into five categories, namely, forests, meadows, plantations, agricultural lands and other (gardens, parks, etc). Such a grouping is relevant, because proposed measures for the transitions are rather similar within each category. For target biotopes, we adopted the seven target biotopes selected by the ecologists, i.e., dry grasslands, wet tall herbs, alluvial forests, wet grasslands, marshes, swamp forests, and eutrophic water. The information available for each sample patch includes the list of initial biotopes (one of the five categories) Bi and their area Ai, the list of target biotopes Bj and their area Aj, and the list of all transitions Bi → Bj with the area covered by each transition, Aij (in ha).

a) Costs

Direct costs:

We grouped the four phases identified in § 5.2 into two sets: the costs of the preliminary study (phases 1 and 2) and the implementation costs (phases 3 and 4). We begin with implementation costs.

Implementation costs

It is meaningful to separate management costs from restoration costs. Restoration costs must be accounted for only one period, because after restoration has taken place, the only tasks remaining include management of the habitats. By contrast, conservation costs are annual because that phase needs to be repeated every year.

Restoration

For each identified transition, the set of measures to be taken has been established by the ecologist teams. For every transition Bi → Bj, we have several measures, noted Mij_k, k = 1, …, p. For every measure Mijk, the cost was estimated per hectare, i.e., Hij_k. To perform this we used existing data from other projects or researches, which are specific to the type of biotope and measure. Multiplying this cost per hectare by the area of transition Aij, one obtains the total cost of each measure k:

$$Cij_k = Hij_k * Aij$$

Summing over all measures to be taken for the transition that will lead to the restoration of the initial biotope i into the target biotope j, we obtain the total cost of transition i → j:

$$Cij = \sum_k Cij_k$$

The total restoration cost for the sample site is then obtained by summing over all restorations, i.e.,

$$C1 = \sum_{ij} Cij.$$

This has to be taken only over the first period.

Management

For every target biotope Bj, the list of measures to be taken has been identified, i.e., Mj_k, k = 1, … , p. For every measure, the cost can be estimated per hectare and per year, namely Hjk. Those costs can be derived from the same data as for restoration costs, and are also specific to the type of biotope and measure. In the available studies, the authors already grouped the annual costs per hectare of all measures required for management of the target biotopes:

$$Hj = \sum_k Hj_k$$

This cost, multiplied by the surface of biotope Aj, will yield the total annual management cost of every biotope, i.e.,

$$Cj = Hj * Aj$$

Summing over all biotopes, we find the total annual management cost of the sample site:

$$C2_t = \sum_j Cj \qquad t = 1, \dots, n$$

By discounting those yearly costs and summing over the time horizon n, with a discount rate of r, we obtain

$$C2 = \sum_{t=1}^{n} \frac{C2_t}{(1+r)^t}$$

Total implementation cost

$$CM = C1 + C2.$$

Costs of preliminary study

The cost of the preliminary study is estimated between 10 and 25 % of the restoration implementation cost (see above). In order not to under-evaluate that cost, we can apply the highest percentage in the interval, yielding

$$CE = 25\ \% * C1$$

That cost will be incurred only once.

Total direct cost

$$CD = CM + CE = 1{,}25 * C1 + C2$$

Indirect costs

Industrial revenue losses

In our case study, the ecologist partners suggested not to convert urban zones into natural zones; hence, no cost will be associated to loss of industrial activities.

Agricultural revenue losses

By contrast, it was proposed to transform agricultural areas into natural areas, by forsaking totally all kind of agricultural activity on the corresponding territories. Therefore the revenue loss in these zones is total. Hence, we will consider the average annual agricultural revenue per hectare, i.e., RA_t for every year t = 1, … , n of the time horizon. This revenue can be calculated from data collected from the Belgian agricultural sector (see data in Appendix 13). More accurate data can also be used, according to the type of agricultural activity performed on each specific territory, but those were not available in our case. Thus, if A designates the total area of restored agricultural land in the sample site, the total annual loss of agricultural revenue for the site will be

$CA_t = RA_t * A$

We must account for this loss yearly; hence, the total agricultural revenue loss over the time horizon can be assessed as (r is the discount rate and n the total number of years)

$$CA = \sum_{t=1}^{n} \frac{CA_t}{(1+r)^t}$$

Forest revenue losses

The ecologist teams also proposed to convert plantations and forest exploitation zones into natural zones by forbidding wood cutting and other forestry-related activities. The annual revenue loss is therefore also total for those areas. The average annual forest revenue per hectare, RF_t (t = 1, … , n) can be calculated from data collected from the Belgian forest sector. If B designates the total area of restored forests and plantations, the total annual loss of forest revenue will be obtained as

$CF_t = RF_t * B$

This loss must be accounted for every year; hence, the total forest revenue loss over the time horizon can be assessed as

$$CF = \sum_{t=1}^{n} \frac{CF_t}{(1+r)^t}$$

Real estate value loss

The various restorations proposed by the ecologist teams have to be put into practice from a legal point of view. Accordingly, the proposal of legal partners is to change the affectation of the corresponding areas in the sector plan (FR: "plan de secteur"; NL: "gewestplan"). Then this legal solution must be assessed from an economic standpoint. Thus, for each transformation of real estate land into natural area in the sample site, Pi → Pj , one has to consider the real estate value per m^2 of land categories i and j, i.e., VIi, and VIj, respectively. Those data being specific to the land category and to the geographic zone of the sample site, we based our analysis on the prices of real estate sales according to the district ("arrondissement"). From those data it appears that VIi > Vij . Considering each transformation Pi → Pj and the corresponding surface area Sij on which this takes place, the real estate value loss can be assessed as

CVij = (VIi-VIj) * Sij

Summing over all such transformation over the sample site, we calculate the total real estate value loss as

$CV = \sum_{ij} Cvij$

That category of cost has to be incurred only once for all.

Total indirect cost

CI = CA + CF + CV

Total cost

C = CD + CI

b) Benefits

Direct benefits

In this research two categories of direct benefits will be considered, namely the benefits from tourism-related exploitation of natural sites, and those related to natural resource exploitation in the EN. It is not certain that tourism-related activities will be developed in the EN, but we consider such an eventuality as possible, due to the growing public interest for such natural areas. Normally the benefits from the EN natural resources exploitation would include the increase of agricultural and forest revenues thanks to EN implementation (see above). However, in our case, the measures proposed by ecologists imply cessation of those activities. Hence there cannot be exploitation benefits, except for the benefits of clear-cutting in the case where a forest biotope or a plantation is converted into another kind of biotope. Such cuttings would occur only once, as would the benefit that would be derived.

Benefits from wood cutting

Two categories of revenues per hectare for wood cut can be distinguished, namely from coniferous species and from broadleaved species, which we designate by RC1 and RC2, respectively. These revenues can be calculated using data collected from the Belgian forest sector (see data in Appendix 13). If A1 and A2 are, respectively, the surface areas of coniferous and broadleaved trees cut within the sample site, the total revenue from wood cutting will be

$$BC = A1 * RC1 + A2 * RC2$$

Benefits from tourism-related activities

We consider the annual revenue per hectare from tourism-related activities, i.e., for year t (t = 1, … , n), RT_t. If S is the surface area of the natural zone under consideration, on which tourism-related activities are likely to occur, the total annual tourism revenue for the sample site will be

$$BT_t = RT_t * S$$

Hence, the total tourism revenue over the whole time horizon can be described as (r and n as above)

$$BT = \sum_{t=1}^{n} \frac{BT_t}{(1+r)^t}$$

Total direct benefit

$$BD = BC + BT$$

Indirect benefits

For every biotope transition Bi → Bj, we consider the superficies affected, i.e., Aij. For each year t considered in the time horizon (t = 1, … , n), we have annual values per hectare attributed to the initial biotope and to the target biotope, i.e., VEi_t and VEj_t, respectively. In most cases, $VEj_t > VEi_t$. Hence for every transition we may calculate the total annual indirect benefit as

$$BIij_t = (VEj_t - VEi_t) * Aij$$

Summing over all transitions found in the sample site, the total indirect annual benefit is

$$BI_t = \sum_{ij} BIij_t$$

And over the total time horizon,

$$BI = \sum_{t=1}^{n} \frac{BI_t}{(1+r)^t}$$

Total benefit

$$BT = BD + BI$$

5.3.3. Data

The data necessary to quantify our case study are contained and described in Appendix 13. For some of the costs and benefits, several different sources may be available. Whenever this is the case, we decided to take into consideration the most pessimistic hypothesis, i.e., the highest costs or the lowest benefits among the available data.

5.3.4. Results[137]

Results of the quantification appear in Tables 5.3 to 5.5 below.

The highest costs are those related to implementation and real estate value loss. For the benefits the situation is not as clear: indeed, depending on the sample, indirect benefits or tourism-related benefits are the most important. Indirect benefits vary much from sample to sample, because they are strongly dependent on the biotope. Thus, in the first sample, many biotopes are restored into marshes, whose estimated monetary value is by far higher than those attributed to other biotopes (see Appendix 13).

Attention should be drawn on the amount of real estate value lost in Sample 3: this value is prohibitively high due to the fact that it is proposed to convert a patch with a very high economic value (real estate) into a natural zone. This reflects well what could happen whenever we wish to implement an ecological network into urbanised or industrial areas, whose prices are even higher. Nonetheless, such an option has not been adopted in Belgian proposals (Natura 2000, VEN).

Other value variations among the samples are essentially due to differences of surface areas.

Summing up those findings, except for the first sample, the results are not much in favour of ecological networks. However, we should remind that the benefits are often under-evaluated:

- First, ecologist teams have rejected any kind of agricultural or forest exploitation in preserved sites, which prevents from beneficiating from a potential important revenue source;
- Next, indirect, social benefits are often under-estimated. The data exploited herein were estimated using different methods, which only give a very rough assessment of truly expectable amounts. Moreover, in our case, we calculated the value increase due to the various biotope restorations, whereas one could say that without implementation of the ecological network, many biotopes are likely to be degraded. Hence, indirect benefits are likely to be higher.

[137] See Appendix 14 for details

Table 5.3. – Costs of the ecological network, in EURO.

COSTS	Sample 1	Sample 2	Sample 3
Direct			
Restoration (1)	351 487	191 190	157 685
Conservation (/year)	129 983	63 748	6 383
Preliminary study (1)	87 872	47 798	39 421
Indirect			
Agric. revenue losses (/yr)	10 562	4 153	11 125
Forest revenue losses (/yr)	14 867	7 496	5 496
Real estate value losses (1)	99 383	94 073	14 066 715

Table 5.4. – Benefits of the ecological network, in EURO.

BENEFITS	Sample 1	Sample 2	Sample 3
Direct benefits			
Wood cutting (1)	4 598	2 348	1 054
Tourism (/yr)	50 463	24 874	12 653
***Indirect benefits* (/yr)**	545 473	31 511	6 860

Table 5.5. – Total costs and benefits of the ecological network, in EURO.

Samples	1	2	3
Fixed costs (1)	538 742	333 061	14 263 821
Annual costs (/yr)	155 413	75 397	23 004
Fixed benefits (1)	4 598	2 348	1 054
Annual benefits (/yr)	595 936	56 385	19 513

Discounting of results

The net present value (NPV) from Samples 2 and 3 (annual costs > annual benefits) will be negative for any discount rate. Only for the **first sample** does discounting make any sense:

- With r = 3 % and n = 30 years, we obtain NPV = 8 540 828 EUR.
- With r = 1 % and n = 30 years, we obtain NPV = 11 275 271 EUR.

The difference between those two values is not highly significant: both cases are “easily profitable”, due to the fact that annual benefits are almost four times higher than annual costs for this sample; hence fixed costs would be rapidly compensated.

Nonetheless, the choice of the discount rate might be quite determinant in environmental projects, especially whenever fixed costs are higher (example of real estate costs in Sample 3) and annual benefits come closer to annual costs. In such a case, selecting a high discount rate value might hamper the feasibility of the project.

5.4. General conclusion

This work allowed us to develop a method of cost – benefit analysis for the implementation of ecological networks. Beyond the final results obtained for the case study, this development turns out to be valuable, because it can be exploited in similar projects, using different data.

The choice we made, to take into consideration the general social standpoint for our evaluation is relevant, because environmental issues have a very broad spatial and temporal dimension, which can even be considered universal. However, the standpoint of target actors of the EN should also be accounted for, to estimate the feasibility of such networks. This is still an important research field that remains to be explored. We provided a beginning of method to tackle this, in the theoretical part of this report.

As we have seen, benefits have an important reach and significance (society over the long term), whereas costs are supported now, by local communities. Those who bear those costs have therefore to be compensated. The amount of compensations can be reasonably based on assessment of those costs. Hence, whenever an owner has to implement management measures on his/her field, the compensation he/she will perceive should be equal to the cost of measures. Whenever a farmer ceases to exploit, the compensation should be equal to his/her revenue losses.

Thus we propose to account for cost values, case by case, to determine the compensations or premiums allocated to target actors. For example, if a farmer has to cease agricultural exploitation of part of his/her field, the loss he/she will undergo will strongly depend on the agricultural activity that was formerly developed. In the same line, restoration and management measures to be taken, as well as their costs, significantly depend on the nature and state of the biotope considered. Hence we think it is important to personalise the premiums and compensations instead of fixing an arbitrary amount.

Additionally, we think it is important to permit some level of sustainable activity on those territories. Thus, biological agriculture, sustainable exploitation of forests, sustainable tourism … are generally compatible with significant economic returns, while at the same time allowing for the implementation of ENs. We think it is little feasible (at least economically) to suppress all kind of human intervention on most parts of the ecological network.

It makes also little economic sense to convert urbanised or industrialised areas into natural zones to make part of an EN. Those areas not only have a high value in terms of real estate, but also provide for important social functions. The cost of such conversions is generally prohibitive.

Finally, besides the satisfaction one can get from the economic exercise and the obtained results, attention should be maintained on the essential role played by ENs for the whole society in the long term. It is interesting and important to evaluate the costs of EN implementation in the perspective of budgetary management and planning, but certainly not with the goal of seeking absolute economic profitability.

PART 6: INTEGRATED CONCLUSION

6.1. Ecological sub-study

6.1.1. Methodological conclusions regarding the optimal ecological scenario

For the case study, the main focus of the ecological team was the elaboration of an 'optimal ecological network' in the Dyle valley. The lack of species-specific data (or the time necessary to gather these data through intensive fieldwork on a long term basis) did hamper the construction of an optimal ecological network at individual species level. Therefore, we adopted a more global approach at biotope level which allowed for the assembly of an optimal ecological network of a selected set of target biotopes within the limited time frame. The development of a method incorporating the comparison of landscape metapopulation capacity increments as a result of adding currently unoccupied patches to an existing network of target biotope patches, resulted in a more widely applicable approach for the elaboration of ecological networks in other areas. However, the current approach can certainly be improved, as restored patches are not taken into account for further calculation of metapopulation capacity of the growing ecological network. Hence, in a more realistic scenario, clustering of target biotope patches may not be as pronounced as it is in the current optimal ecological network. Other aspects of the methodology may also be prone to justified criticism: both the scores for conservation status and for restoration feasibility were obtained through consultation of several experts acquainted with the study area and the vegetation types occurring in this valley system. While this approach certainly holds a certain degree of subjectivity, it may be assumed that combination of the judgment of several experts produces a score that corresponds to the actual conservation status or restoration feasibility.

To demonstrate the model's sensitivity to species-specific parameters such as minimum area requirements and dispersal distances, optimal ecological networks for several target species were constructed, illustrating possible conflicts for the restoration of the same patch into different target biotopes. Hence, we should keep in mind that modelling the desired ecological network through a metapopulation capacity approach may be a useful tool to decide on restoration and management priorities at target biotope level, but the model may not (yet) be powerful enough for realistic predictions for individual species in particular landscapes.

6.1.2. Comments on the ecological feasibility

We mainly encountered problems when matching data on vegetation, soil and hydrology. The general lack of GIS data layers providing background data on abiotic conditions, distribution of vegetation types, hydrological properties in Wallonia and its deviation from the data standards in Flanders did also restrict the production of uniform maps and spatial analyses for the whole study area. Also, different traditions and accents of both labs resulted in a few discussions. Though time consuming, these discussions were fruitful and they unquestionably added value to the final result.

6.2. Legal sub-study

6.2.1. Methodological aspects

As far as methodological aspects are concerned, although many very interesting findings came out of the study, the global impression is that all objectives initially assigned could not be reached as first scheduled. No "integrated scenario" was drawn from all sub-studies. Different reasons can be outlined.

Our first conclusion is that, unsurprisingly, time was lacking, and, especially, *legal part was underrepresented in the study* (only 20 % part-time researcher) in comparison with other parts (min. 1 full-time researcher). Ecological aspects were given much attention, whereas legal aspects were considered secondary when submitting the project. This imbalance has had many consequences:

- As the legal context was very complex (two distinct legal orders: Flemish and Walloon law; numerous legislations involved (water, agriculture, forests, industry,…); frequent modifications of legislations), no profound analysis of these regimes and of their interaction with legal instruments for nature conservation was possible, although this interaction could greatly hamper the legal feasibility of the EN.
- Legal proposals for implementation of ecological scenario were limited to direct implementation (creation of protected area, …) – as only these measures have direct consequences on people – and were not focused on planning, although this part of EN implementation is crucial in our opinion (as it is the way to involve stakeholders and population and to integrate EN in sectorial policies).
- An integrated scenario was not really produced in the end of the ECONET project. Only conclusions on feasibility of ecological optimal scenario could be given.

A second conclusion is that timing of such study was made difficult, as each non ecological sub-study had to wait the results of ecological sub-study (ecological scenario). Furthermore, many legal implementation measures (expropriation, designation as protected area,…) had socio-economical consequences (diminution of real estate value, land-owner refusal to cooperate to implement conservation measures,…). So, economic and sociological teams had to include in their own evaluation the results of the legal study, which only came in December 2004. Also, ecological outputs were sometimes difficult to interpret in their first form, as no scientific priorities were given, especially in the spatial dimension. The map of the EN (with central and development zones and corridors), delivered later, was really helpful as it depicted the strategic priorities and the spatial dimension of the ECONET project. In that way they were equivalent to the crucial planning phase in an ideal public EN policy.

Finally, we should mention that the study area (Dyle valley) may be not representative for a "normal" EN situation (from a legal point of view) – e.g. an EN in agricultural or semi-urban regions of Occidental Flanders or Hainaut – as nearly all this area is protected and under nature conservation regime. This is not a critique – as technical requirements justified this choice –, but should be taken into account when reading conclusions on the global feasibility of implementing an EN.

6.2.2. Conclusions on legal feasibility of EN in the Dyle valley

The present sub-study, in both its theoretical and practical aspects, allowed to draw some conclusions on the legal feasibility of EN, in general and in the Dyle valley.

The first conclusion draws attention on the complexity of the legal context of EN strategies, as the whole public real estate and administrative law is involved. Not only nature conservation law is to be used to implement EN, but also legislation on all other land use regimes (land use planning, water management, agriculture, forestry, industry, transport infrastructures,…) as they influence habitats and structure, functioning or dynamics of populations. EN feasibility is thus function of integration of strategic goals and objectives into sectorial land use policies – via procedural mechanisms like impact assessment and coordination of decision processes, but also substantial measures like planning and protection rules –, in order to reduce their impact on EN.

A second conclusion concerns the availability of relevant instruments to implement, on the one hand, planning and, on the other hand, protection, management and restoration measures as proposed in the ecological optimal scenario. Actually, since some years (1997 in Flanders, 2001 in Wallonia), nature conservation legislation was revised and improved to include specific mechanisms of planning of a regional EN or at least its "backbone" (VEN, Natura 2000). In the Walloon Region however, the EN is focused only on Natura 2000 species and habitats, which is a restrictive procedure. Articulation with general land use planning is provided at different degrees in both regions. To implement protection, management and restoration measures, many instruments – obligatory, incentives or property rights-based – are available to public powers, even if improvements could be desirable, especially in the Walloon Region. More problems concern integration of EN considerations into sectorial policies like water resources management or rural development. Actually, it is not the specific and integrated instruments that are lacking, but rather the political will to implement them and public support to accept them. Basically, these problems arise frequently because of, on the one hand, excessive costs for the land-owner or occupant, which raise social contestations and, on the other hand, the poor conservation budgets which cannot afford large-scale specialized management and indemnities to compensate property rights restrictions. Legal feasibility can also be hampered by considerations on equality or property rights protection, but generally any *proportionate* conservation measure can be implemented without sanction based on these principles.

In the Dyle valley itself, legal feasibility of ECONET-project seems to be quite high, as nearly the entire zone is under protection and/or management regimes favourable to nature conservation. The only real obstacle is the protection of water quality and the control of eutrophication in such a big catchment as the Dyle valley. These problems can only be tackled by an integrated approach of nature conservation and resources management, based on unique, scientific and coherent planning.

6.3. Sociological sub-study

6.3.1. Methodological conclusions

Within the social research part focus lay, due to the virtual aspect of the study, on the construction and exploration of the theoretical framework to gain insight into the methodology of integrating the social feasibility in the designing process. In general, it can be stated that the sociological theoretical framework seems appropriate to frame and assess the social feasibility of the implementation of an ecological network, even in an inter- or multidisciplinary setting. The coherence and interrelations between the three different social aspects (impact, perception and organizational structure) - that were used to assess the feasibility and can be found in the empirical world - also are framed by the sociological theory. In addition to that, the systems theory of Niklas Luhmann combined with some other sociological theories, also addresses the process-like character of a policy-process.

The methodology of the social impact assessment is developed to give insight in possible or potential social impacts that might occur as a consequence of a certain project or policy. A crucial aspect is that it looks further than the purely socio-economic impacts like in a cost-benefits analysis. It integrates also concepts like social cohesion, social tension, … Next to observing the potential impacts it also integrates the background of the area and its stakeholders in the analysis thanks to the profiling step. So, for instance information on the perception of people on nature and other actors can be used as a background for the assessment of certain impacts. Question was if a SIA could help to give an insight in the feasibility of an ecological network. Methodologicaly it was not easy to determine the crucial social impacts of the implementation of an EN. In this research, a first proposition of an assessment framework for the impacts of nature conservation and development was suggested. Due to the nature of the measures the SIA was more or less limited to socio-economic effects of land use transitions. Due to the virtual nature of the case study, assumptions were necessary to estimate the impacts and direct interaction with the involved actors was not feasible. So, it is difficult to point out exactly the value of a SIA in the context of the implementation of an ecological network. Nevertheless, to have an effect on the end product it is crucial that one should perform the SIA during the planning process and not just at the end. Results of the SIA could then be integrated in the ecological designing process. As such the SIA becomes more than a product but it is used to give social input in the designing process.

6.3.2. Conclusions on social feasibility

On the basis of the theoretical study of the three aspects in relation to nature and ecological networks, it can be stated that implementation of an ecological network is clearly a dynamic process. Through the general framework, we also managed to theoretically make a direct connection and interrelation between impact, perception and organisational structure. Impacts and the way people are involved (=organisational structure) largely determine the perception of people towards nature (policy). Impacts can however be real as well as perceived. With a good organization, false perceptions (about impacts) can be enfeebled and impacts extenuated. Also, a common nature-image can be designed. Thus, it can be stated that when studying the feasibility of the implementation of ecological networks, the three above-mentioned aspects should be given the necessary attention. They determine each on their own turn the feasibility. Besides that, it is also important to consider their mutual coherence. Question is still, which of the factors is the 'driving force' in relation to the feasibility of the implementation of an ecological network. German research came to the conclusion that it is not the competition for land or lack of knowledge that is the 'core' issue in relation the feasibility of nature conservation but the perception of the different actors involved (Stoll-Kleemann, 2001). Question is if this is also the case in the Belgian context where the pressure on land is higher. This has to be further investigated by observations and analysis of real cases.

When strictly looking at the results out of the Social Impact Assessment it can be stated that the impacts in the Dyle valley and more specifically in the three sample-sites remain relatively limited, aside from some bottlenecks. In general, the feasibility of the proposed scientific EN from a sociological perspective stays within limits when taking into account the results of the Social Impact Assessment. This can for a large part be attributed to the fact that, apart from economic activities as agriculture and sylviculture, overall little change for human activities could be detected. Little to no effects could be observed for, for instance tourism, inhabitants, industry, transport and mobility, etc. If the ecological measures would be more drastic (e.g. accessibility of the area, removing roads, …) , the social feasibility would probably decrease. Also, social impacts and the social feasibility will differ dependable of the juridical and socio-economic instrumentarium that will be used.

Finally, because of the virtual character of the project (the designed ecological network will not be implemented in practice) it was difficult to analyse the feasibility, taking into account perception and organisational structure. Perception of nature of the actors involved is quite different from the perception and nature images of the scientific designers. If one would implement an EN in the Dyle-valley for real, attention needs to be paid to the organizational aspect of the process and the related perception of the local actors on the policy-process. To come to a socially feasible implementation, probably much more moments of harmonisation between the ecological design and the preconditions of the other aspects (social, economic and legal) would need to occur. Without such an interaction, the feasibility of the proposed ecological network would be low. Question is to what extend ecological goals could be changed to come to a sustainable ecological network from an economic, social and ecological point of view. Depending on the answer to this question one of the models of co-ordination like presented by Mougenot & Laurence (2002) could be chosen.

6.4. Economic sub-study

This research allowed us to develop a method of cost – benefit analysis for the implementation of ecological networks. Beyond the final results obtained for the case study, this development turns out to be valuable, because it can be exploited in similar projects, using different data. The method is the result of many interactions among the teams. Thus, the choice to work by implementation measures for estimating direct costs comes in a straightforward manner from interaction with ecologist teams, while evaluation of some indirect costs was based on conclusions from the legal team.

The choice we made, to take into consideration the general social standpoint for our evaluation is relevant, because environmental issues have a very broad spatial and temporal dimension, which can even be considered universal. However, the standpoint of target actors of the EN should also be accounted for, to estimate the feasibility of such networks. This aspect is also extremely meaningful from the perspective of the sociologists' research. This is still an important research field that remains to be explored.

We have to deplore that we worked in a multidisciplinary rather than interdisciplinary manner. Thus, certain choices were made "upstream" of the economic contribution by other partners, which strongly oriented some aspects of the economic sub-study. For example, the decision of ecologist partners not to allow for any agricultural or forest activity over the territory selected for the ecological network significantly reduced the potential economic benefits that might be obtained from such areas. Such a choice would be hardly defendable on purely economic grounds.

It makes also little economic sense to convert urbanised or industrialised areas into natural zones to incorparate them into an EN. Those areas not only have a high value in terms of real estate, but also provide for important social functions. The cost of such conversions is generally prohibitive.

In a more interdisciplinary process, such conclusions might have repercussions on the ecological and legal choices that would be implemented.

Finally, besides the satisfaction one can get from the economic exercise and the obtained results, attention should be maintained on the essential role played by ENs for the whole society in the long term. It is interesting and important to evaluate the costs of EN implementation in the perspective of budgetary management and planning, but certainly not with the goal of seeking absolute economic profitability.

6.5. Multidisciplinary research-process: conclusions

In the beginning of the research process, a dynamic, interactive and interdisciplinary process was designed (See Fig. 6.1). The intention was that the ecological perspective would be the starting point of the research, onto which the other disciplines would found their research. Nevertheless, the development of the optimal ecological scenario and the preparing steps would all be done in close collaboration and discussion with the other disciplines. Out of this iterative and interdisciplinary process, an optimal ecological scenario would arise that would be assessed by the non-ecological disciplines, out of which an integrated ecological scenario would arise.

Now, at the end of the project, it can be stated that these goals were not fully achieved.

First of all, the process was characterized by multidisciplinarity instead of interdisciplinarity. Looking back, one could say that the ultimate goal of this research-project – a feasible and integrated design of an EN in the Dyle valley - was aimed at by means of a 'reactive compromise' instead of a 'creative compromise' (Aarts, 1998). For instance, the demarcation of the area for the case-study was performed by the ecological team. For that demarcation, certain preconditions where used to set the boundaries of the EN, like urbanised zones, transport infrastructure, etc. This made that one could no longer speak of a truly optimal ecological scenario and that certain aspects for the social, economic and legal feasibility no longer could come into play. Because of this, certain valuable insights might be lost. One could say, in line with Aarts (1998) and Vandenabele (2004), that there was a strong emphasis on different positions (different scientific perspectives). The interaction actually took place at the end of the process. For instance, the choice to use three sample-sites within the case-study area was made after the delineation of the optimal ecological network. These were chosen by the legal, economic and sociological teams, to allow a more relevant analysis.

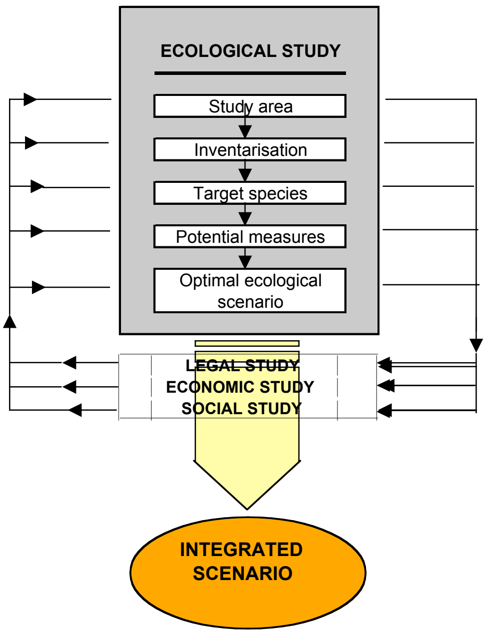

Fig. 6-1. – Original design of the research process.

One of the consequences of this multidisciplinary approach instead of an interdisciplinary approach was that some decisions made earlier in the process had to be revised later on, because they were incompatible with the research approach of the other disciplines who started their more focussed research later in the process. Also, a more interactive planning could have prevented communicative problems. For instance, there were different concepts of an ecological network between the different disciplines and even within one discipline. Also, the different languages (Dutch, French and English) and professional jargon within the research team made that the harmonisation was more difficult. Misunderstanding could have been prevented when the need for a common language (Rientjes, 2000) would have been more recognised.

Second, the integrated scenario seemed a step to far within this project. The research and the results stop at the assessment of the optimal ecological scenario by the three non-ecological disciplines: i.e. legal, economic and sociologic. Input about the feasibility from this team to the ecological teams was therefore minimal. On the other hand, the ecological teams changed their approach somewhat by taking in account suggestions provided by the other partners. It is difficult to say why the proposed target was not reached. At the start of the project a structured research-plan was developed. During the process the methodologies and approaches of the different disciplines seemed to be too different to come to an integrated result. A reason therefore could be that too little attention was paid to generate a common language. A solution here could be to effectively design an integrated plan at the beginning that afterwards is assessed from an ecological, economic, social and juridical point of view.

Finally, in addition to the minor language difficulties mentioned earlier, there was definite problem with the use and exchange of discipline-specific terminology. Specific terms well known to ecologists were unknown to the other partners whereas discipline-specific economical and sociological terms where new for researchers having an ecological background. While these problems could be overcome rather easily, it resulted in a lot of time being spent on communication or on the construction of necessary tools to facilitate the interpretation of the optimal ecological scenario by the other teams (e.g. the matrix of management and restoration measures, the presentation of the optimal ecological network as a combination of core areas, development zones and corridors, …). The fact that most partners were working at different locations added to high time investment in communication and discussion. Moreover, the different teams did not start on the same date and were not all involved in the project on a full-time basis. Therefore, we have two suggestions to future similar multidisciplinary projects:

- to bear these extra time needs in mind while defining the necessary project time. If possible, temporarily working at a same location during execution of the project would be an advantage.
- to carefully plan timing and intensity of the workload so that human resources can be deployed more adequately for the different disciplines.

6.6. Conclusive remarks

6.6.1. Positive returns from the research

Our research was probably the **first attempt** to approach the feasibility of ecological networks from the perspective of **complementary** disciplines, i.e., ecological, economic, social, and legal. These are supposed to cover the whole spectrum of so-called "exact" and "human" sciences that are applicable to such a problem area. We can say that results were indeed produced as to the **feasibility** of an ecological network scenario, although it should be acknowledged that these were obtained from individual standpoints (i.e., ecological feasibility, economic feasibility, social feasibility, legal feasibility), but with rather limited interactions and limited feedback among partners.

Nonetheless, this research provided us with reasonable insight into the **complexity** of the double task

- to plan an interregional ecological network in an integrated manner, and
- to obtain a realistic/pragmatic and functional ecological network, at least on paper.

This certainly allows us to emphasize how careful must be the **planning phase** in the implementation of ecological networks.

Obviously, there were interesting, meaningful and productive **interactions** among our teams. Some examples of these have been given above in this integrated conclusion. However, difficulties to reach an integrated scenario in time show how important legal, economic and social aspects are in order to reach the final objective of an "integrated" project of ecological network.

6.6.2. Comments on the transposibility of results

From a strict ecological point of view, we worked at a target biotope level rather than individual species level. Such an approach is more widely applicable for the elaboration of ENs than an approach that would be based on target species. This is because, among others (e.g. lack of species-specific data), it allows more easily adding currently unoccupied patches to an existing network of target biotopes, whereas a species oriented approach would be more specific to the local conditions and hence less widely applicable. Similarly, the approaches used by the other disciplines (legal, social, economic) and adapted to the study of ecological networks in the manner we proceeded, can be transferred in a straightforward manner to the study of other cases of ecological networks.

The legal descriptions of land-use regimes and the theoretical feasibility assessment of EN from a legal point of view can be used already without being applied to case studies, to start discussions on improvement of the legal framework in both regions (eg on interactions between water regimes and EN). The problem is that we lacked time to obtain more than superficial conclusions on these aspects.

A pecularity of our study deserves to be pointed out, namely, that it was applied to a rather heavy densely populated area. Obviously this implied severe limitations in the choices that were possible (as we have seen with the economic and social aspects) and the approach would be significantly different in cases where the initial situation entails less inhabited, more "natural" areas from the outset.

Finally, although we must recognize that conclusions are rather limited as regards the actual ecological, economic, social, legal situations (in the sense that nothing can be transposed as such), one of the main interests of the research lies, from the **scientific point of view**, in the clarification of theoretical concepts (in all sub-studies) and their use in a case study and, **from a methodological point of view**, in the detected drawbacks and the lessons that were derived, as indicated below.

6.6.3. Difficulties, drawbacks and recommendations

As pointed out above, we have been confronted with communication problems. Those arouse from language differences (we had to work in three different languages, i.e., French, Dutch, English), different vocabularies, methodologies and approaches used by the various partners, and from the fact that we were working at different locations. Indeed, a few intense "integrated meetings" showed the interest we might have gained from more frequent and intense interactions. Having decided to work "accross the (linguistic) border" also induced some difficulties, from the fact that we had to work with two different legislations using two different languages, and with different databases regarding vegetation, soil, hydrology, biotope characterization and nature conservation priorities.

About the significance and reach of our research, as also already commented, our study turned out to be **multidisciplinary** instead of **interdisciplinary** as it should have been. It also suffered from the fact that it remained essentially **theoretical**, in the sense that it was not intended for immediate implementation. This implied that there was little if any interaction with local actors, which would have made the study much richer.

Finally, we must admit that the research process has been essentially **sequential** and not truly **integrated** as it should have been; i.e. the ecological partners paved the way for other partners, after which the legal team intervened first, followed by the economic and social partners. This largely hampered an interactive process as illustrated in Fig. 6.1, in which legal, social and economic aspects play like a filter and allow for feeback loops to take place, leading to a truly **integrated scenario** for the ecological network. Instead, due to the sequential process, the result we came to turned out to be an ex post **feasibility study** (from legal, economic and social standpoints) of the **optimal ecological scenario**. The outcome appears more as a **reactive compromise** than a **creative compromise**.

The drawbacks and difficulties we just mentioned, may originate from one essential aspect, namely the **lack of sufficient time** to devote to such a research, in the way it would have deserved. We must realise that the teams should work in more organised sequences, with periods of overlapping work and feedbacks, and that they do not necessarily have to work with the same intensity during the entire time devoted to the project. This process should be spread on a longer time horizon than we actually had. To come to a good compromise, we can estimate that at least three to four years would be necessary to organise a research like ours, with methodological aspects, development of a case study and a truly integrated process in a well organised sequence.

We also realised that legal aspects have been largely under-estimated in the design of the research. We would recommend a equilibrium among partners as reflected in Table 6.1.

Table 6.1. – Observed and recommended equilibrium among partners
(figures in % equivalent full-time researchers).

Partner	Representation in project	
	ECONET	Recommended
Ecological	50	40
Economic	20	20
Social	25	20
Legal	5	20