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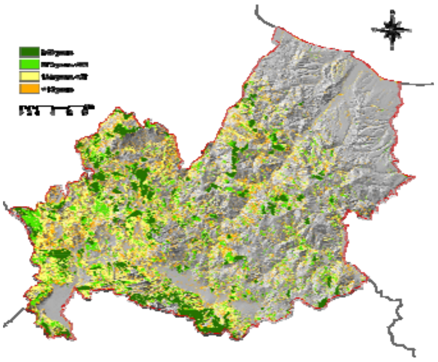
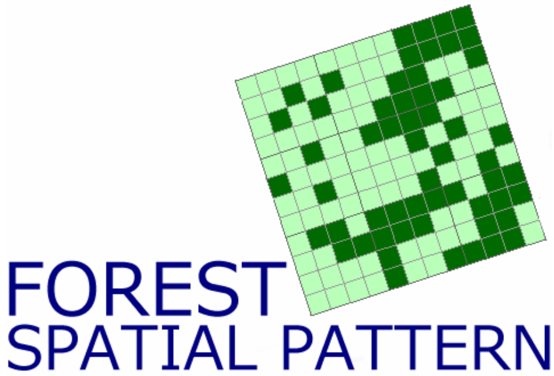
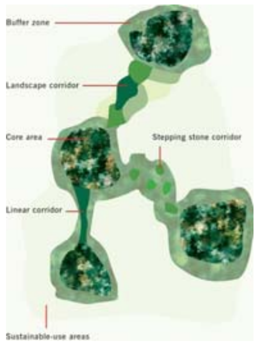
Linking and harmonizing forest spatial pattern analyses at European, national and regional scales for a better characterization of forest vulnerability and resilience

Tonti, D. (1), Estreguil, C. (2), Marchetti, M. (1), Oehmichen, K. (3), Chirici, G. (1),
Troeltzsch, K. (4), Watts, K. (5)







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European Commission
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Contact information
Address: Via Enrico Fermi,2749 Ispra,Varese- 21020 Italy
E-mail: christine.estreguil@jrc.ec.europa.eu
Tel.: +39 0332 785422
Fax: +39 0332 786561

http://ies.jrc.ec.europa.eu/
http://forest.jrc.ec.europa.eu/
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Linking and harmonizing forest spatial pattern analyses at European, national and regional scales for a better characterization of forest vulnerability and resilience

Project Consortium

University of Molise, ITALY, coordinator
Daniela Tonti, Gherardo Chirici, Marco Marchetti, Ugo Chiavetta

European Forest Institute, FINLAND
Katja Troeltzsch, Jo Van Brusselen

University of Hamburg, Institute for World Forestry, GERMANY
Michael Koehl, Katja Oehmichen, Aljoscha Requardt

Forestry Commission, UK
Kevin Watts

Preface

The present study was developed in the context of Regulation (EC) 2152/2003 on the monitoring of forest and environmental interactions, the so-called "Forest Focus" Regulation.

The Forest Focus regulation centered specifically on the monitoring of the effects of atmospheric pollution and fires on European forests, previously addressed by Council Regulation (EEC) No 3528/86 of 17 November 1986 on the protection of the Community's forests against atmospheric pollution and Council Regulation (EEC) No 2158/92 of 23 July 1992 on protection of the Community's forests against fire. Furthermore, “Forest Focus” aimed at encouraging the exchange of information on the condition of and harmful influences on forests in the Community and enabling the evaluation of ongoing measures to promote conservation and protection of forests, with particular emphasis on actions taken to reduce impacts negatively affecting forests.

In order to promote a comprehensive understanding of the relationship between forests and the environment, the scheme also included the financing of studies and pilot projects aiming at the development of monitoring schemes for other important factors such as biodiversity, carbon sequestration, climate change, soils and the protective function of forests. The EC launched and financed a series of seven studies dealing with the following topics:

1. *Climate change impact and carbon sequestration in European forests*
2. *Development of a simple and efficient method field assessment of forest fire severity*
3. *Use of National Forest Inventories to downscale European forest diversity spatial information in five test areas, covering different geo-physical and geo-botanical conditions*
4. *Harmonizing National Forest Inventories in Europe*
5. *Development of harmonised Indicators and estimation procedures for forests with protective functions against natural hazards in the alpine space*
6. *Linking and harmonizing the forests spatial pattern analyses at European, National and Regional scales for a better characterization of the forests vulnerability and resilience*
7. *Evaluation of the set-up of the Level I and Level II forest monitoring under Forest Focus.*

This study (topic 6 in the above list) aims at (1) applying the mathematical morphology based forest spatial pattern mapping tool available at the Joint Research Centre, (2) developing an index of forest vulnerability, resistance and resilience, and finally (3) addressing the link between forests spatial pattern and forest ecological functions related to vulnerability and resilience. Multi-scale and multitemporal spatial analyses were conducted in a harmonized way over case studies in five European ecological regions.

Ernst Schulte
Directorate General Environment

Jesus San-Miguel-Ayanz
Joint Research Centre

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Executive summary

The project “Linking and harmonizing the forests spatial pattern analyses at European, national and regional scales for a better characterization of the forests vulnerability and resilience” (JRC contract 382391 F1SC) covers one of the seven topics to be studied in the frame of the Regulation (EC) 2152/2003 on the monitoring of forest and environmental interactions, the so-called "Forest Focus" Regulation. It first demonstrates the application of the mathematical morphology based forest spatial pattern analysis tool developed at the Joint Research Centre (GUIDOS); it then develops one index related to potential forest vulnerability, resistance and resilience (FVRR index) on the basis of multi-criteria fuzzy modelling technics. Its final aim is to address linkages between forest spatial pattern and forest ecological functionality with emphasis on forest vulnerability, resistance, resilience.

This study was conducted by a European consortium coordinated by the University of Molise (Italy) and included partners from the University of Hamburg (Germany), the European Forest Institute (Finland) and the Forest Research (United Kingdom). The overall supervision of the project and the processing of forest spatial pattern were done by the Joint Research Centre.

Seven case studies were selected in five European ecological regions: two sites in Great Britain (Atlantic zone), two in Finland (Boreal zone), one in Germany (Continental zone), two in Italy (Alpine and Mediterranean zones) (figure 1). The analysis was conducted at two spatial scales with forest maps available at broad resolution (100m raster or 25 ha minimum mapping unit) and at fine resolution (25 m raster, 1ha minimum mapping unit). Multi-temporal historical data series were available as for example in the Mediterranean site (figure 2) to study trends in space and time for forest pattern and forest vulnerability, resistance and resilience.

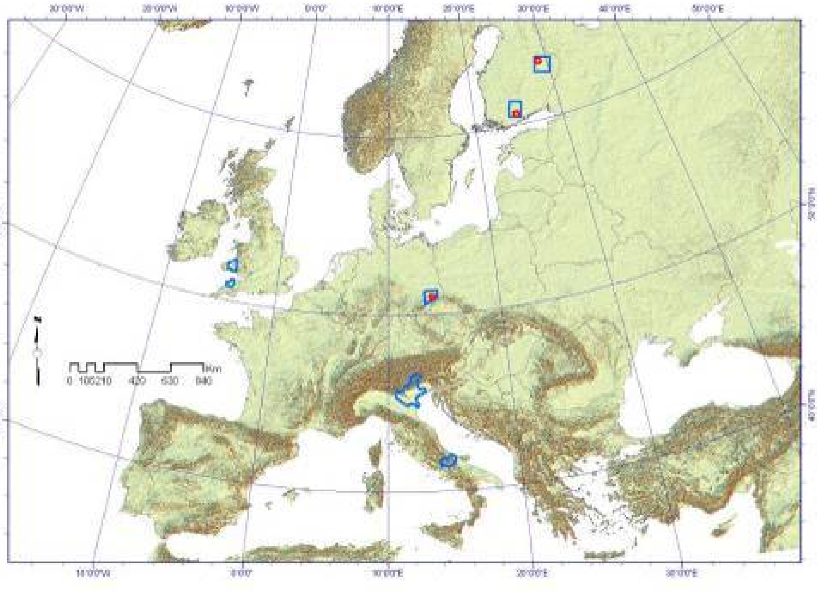

Figure 1: Location of test areas (boundaries of low resolution data in blu shade and high resolution data in red shades; large coverage for both resolutions in Italy and UK).

Landscape level forest spatial pattern refers to the spatial arrangement or configuration of forested ecosystems across the landscape. It was mapped by applying the mathematical morphology based freeware GUIDOS (Graphical User Interface for the Detection of Objects and Shapes) on all binary forest (forest type)/non forest maps. Forest pattern classes were core (interior forest area minus a fixed (100m) edge size), edge (external perimeter of core patch), perforation (perimeter of perforation in core patch), branches of edges, connectors between cores (bridge; loop when same core) and islet/fleck (isolated non-core forest patches). Pattern maps for the Molise site are illustrated in figure 2. Their use combined with forest type and age categories (derived from the historical analysis) and thus linking to potential forest ecological conditions is illustrated in figure 3.

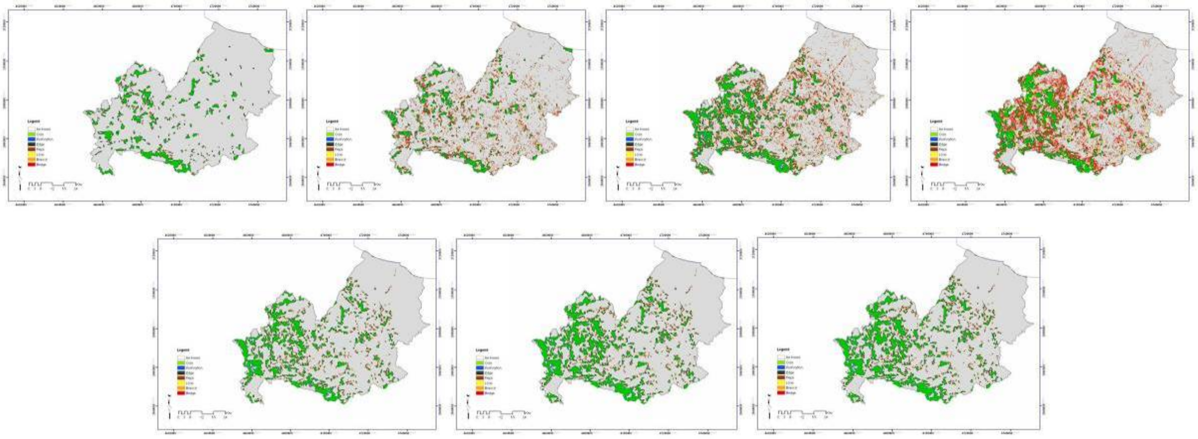

Figure 2: Forest spatial pattern maps generated with a 100m edge width in Mediterranean test area (Molise). Top: pattern from multitemporal high (25m) resolution forest maps (years 1936, 1954, 1992 and 2005 from left to right). Bottom: pattern from low resolution (100m) spatial pattern maps (years 1980, 1990, 2000 from left to right).

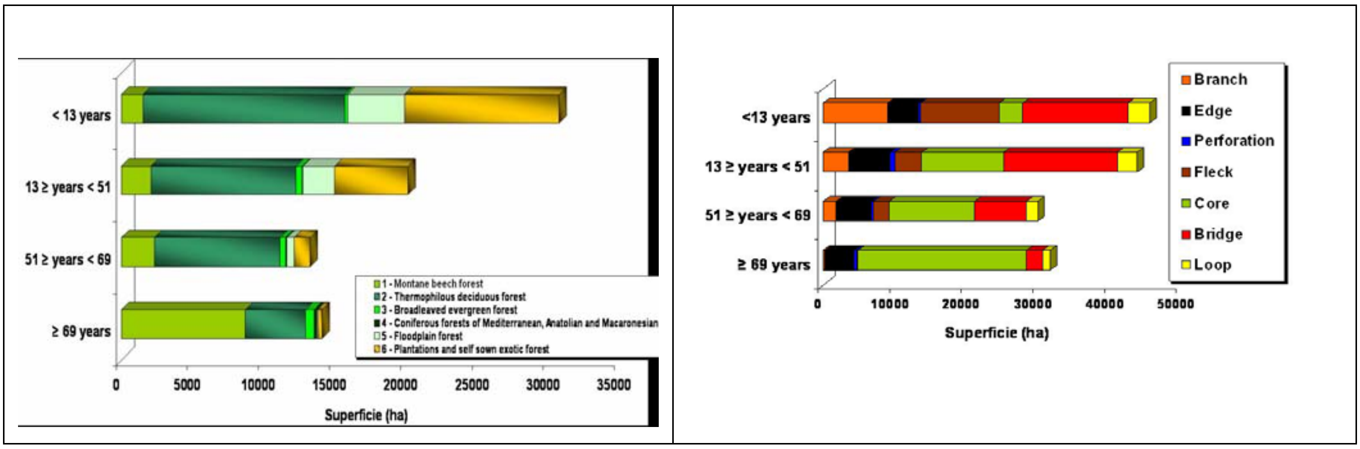

Figure 3: Mediterranean test area (Molise). Left: distribution of forest types per forest age class. Oldest forest are predominantly Montane beech while youngest formations include mainly thermophilous deciduous and plantations. Right: distribution of pattern classes in 2005 per forest age class. Oldest forests are predominantly in core areas. Connectors and islets/fleck include more young forest formations.

Forest vulnerability refers to the threat of forests to suffer natural or anthropogenic damages (urban development, fires, windstorms, etc.). Forest resistance is understood as the internal capacity of the forest to resist to the influence of disturbing factors remaining unaltered and conserving its ecological functionality. Forest resilience is understood as the internal capacity of the forest to recover after the damage. Data for each variable does not exist. The three variables (probably correlated as higher the level of resilience, higher the level of resistance and lower the level of vulnerability), were measured in a first approximation, by a single *FVRR* index. It would resume the ecological distance between the real and the optimum ecological and biological forest conditions depending on vulnerability, resistance and resilience related factors. FVRR would vary from low (low resilience and resistance, high vulnerability) to high (high resilience and resistance, low vulnerability).

Two approaches for modelling the FVRR index were tested:

1. The *species specific* approach was based on the use of different ecoprofiles (5 in total) related to four main umbrella species group (Figure 4): birds (lesser spotted woodpecker), small mammals (marten), large mammals (wolf and roe deer), butterflies (lesser purple emperor). Five models were designed in order to evaluate the ecological distance between actual and optimal conditions (one model per species ecoprofile). The final output variable is not a habitat suitability index but an overall value of FVRR for the five species.
2. The *species unspecific (forest generic model)* approach was based on the overall ecological and biological functionality of a forest habitat on the basis of generic rules. Similarly to the species specific model, the output value of FVRR is a measure of the ecologic distance between real and optimal conditions, but it does not refer to a single umbrella species but to the general functionality of the ecosystem.

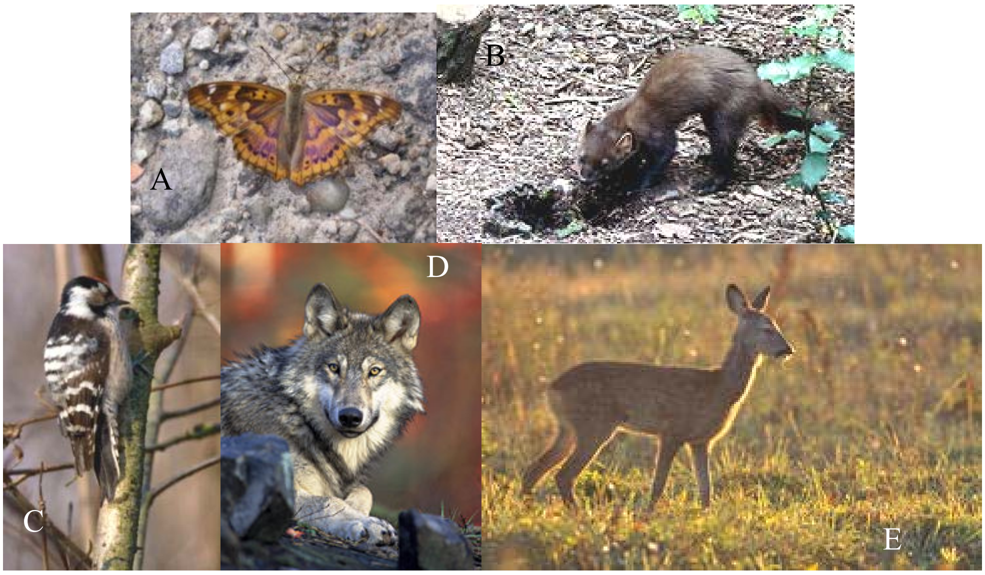

Figure 4: photos of the selected umbrella species (A: *Apatura ilia*, lesser purple emperor; B: *Martes martes*, pine marten; C: *Picoides minor*, lesser spotted woodpecker; D: *Canis lupus*, gray wolf; E: *Capreolus capreolus*, roe deer).

The FVRR index values were mapped by applying a GIS Multi Criteria Analysis based on a Weighted Linear Combination of different fuzzy factors. Input data were the forest and pattern maps (figure 2) and other vectorial data. Factors for the multi-criteria analysis were calculated per pixel (or per patch). Neighbouring landscape was defined by a 5 km circular window around each pixel or by the home range for the species-specific model. Factors were the forest and core forest patch sizes, the forest edge/core area ratio, the distance to neighbouring disturbing land uses like urban/settlements, roads, intensive agriculture, the disturbances by forest fires, the diversity and evenness of forest categories, the shape complexity and the vegetation naturalness. For each factor, a potential optimum condition was defined and each single forest pixel was given a value representing its ecological distance to this optimum using logistic fuzzy membership functions. The closer the pixel value was to 1 the higher was its potential level of resistance and resilience and lower was its potential level of vulnerability.

The FVRR indexes were calculated for the species-specific and the generic forest model in each test area at broad scale (low resolution data) and at finer scale (high resolution data) in order to evaluate the scale dependency of the analysis. The level of similarity of the results across scales would express the level of scale-dependency of the analysis, particularly of interest the loss of information in FVRR evaluation when applying the model with low resolution data instead of high resolution data.

FVRR index maps for the forest generic model show the distance between real and optimal ecological conditions of forest habitats (figure 5). The FVRR results tend to be less spatial scale dependent than traditional forest area measures and seem to capture interesting temporal trends in ecological functionality of forest habitats (figure 6).

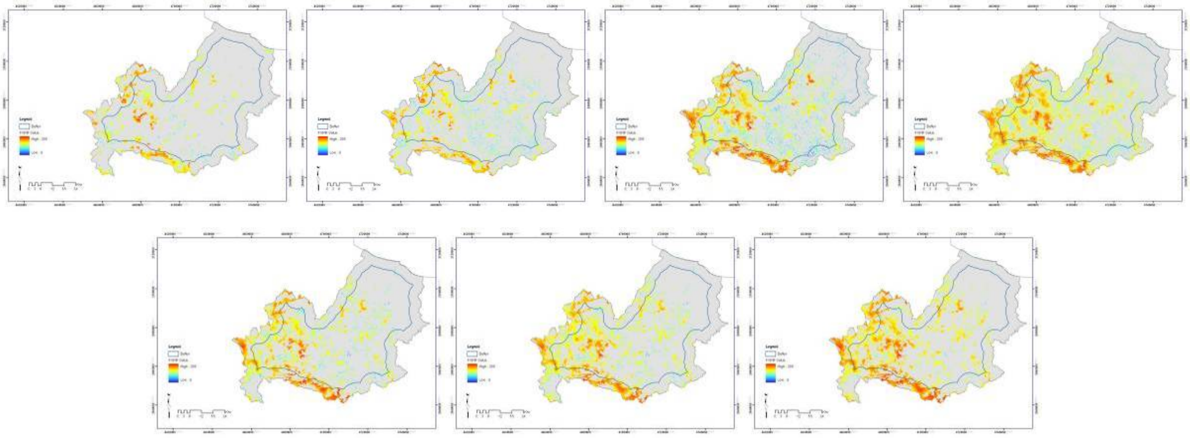

Figure 5: FVRR index maps for the forest generic model in the Mediterranean site (Molise), from blue shades for highest resistance/resilience and lowest vulnerability, to red shades for lowest resistance/resilience and highest vulnerability. Top: high resolution maps (years 1936, 1954, 1992, 2005 from left to right). Bottom: low resolution maps (1980, 1990, 2000).

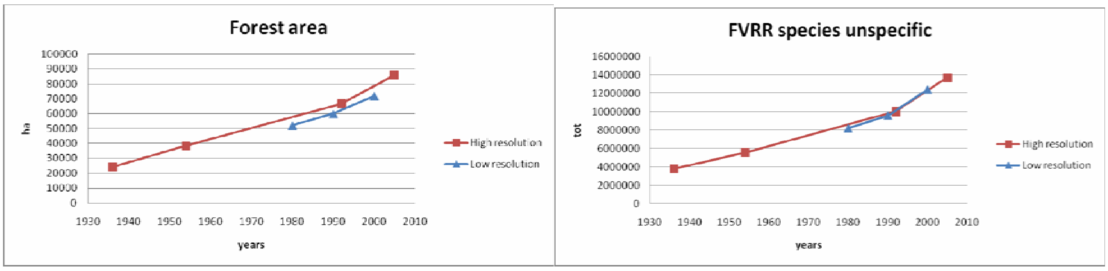

Figure 6: comparison between the temporal trends in forest area and in FVRR index calculated for the forest generic model in the Mediterranean site (Molise).

The temporal trends for each FVRR species specific models (and their aggregation) are similar to the ones for the forest generic model (figure 7). The FVRR index generally increased over time for all species. However, the models are not sensitive to each species traits. The results for the martens and butterflies on one hand, and the ones for the woodpecker and roe deer on the other hand are redundant. Results for the wolf show clearly a lowest index level with comparison to the others species.

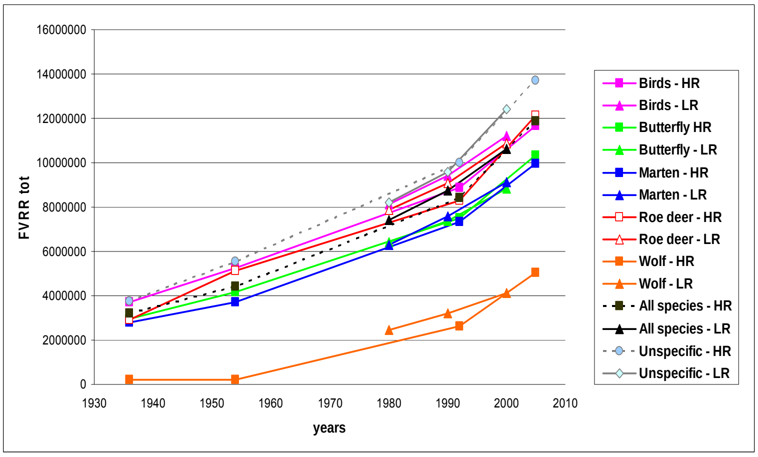

Figure 7: comparison of temporal trends of the species specific and forest generic (unspecific) model FVRR based both on low and high resolution data in the Mediterranean test site (Molise).

To characterize forest vulnerability, resilience and resistance across space and time, the FVRR index based on multi-criteria fuzzy analysis including forest pattern measures may be worth further investigation to test in large regions assessment at pan-European level.

1 Introduction

This report is the outcome of the project "Linking and harmonizing forests spatial pattern analyses at European, national and regional scales for a better characterization of forests vulnerability and resilience", also referred shortly to as "spatial pattern" (contract 382391 F1SC following tender 176-174131 launched by the Joint Research Centre (JRC) of the European Commission, Institute for Environment and Sustainability located in Ispra (VA, Italy), that entered in force the 21st of November 2006). In addition to this report, the DVD containing the digital database (the input dataset, the model output maps and tabular data) for the case studies was delivered to JRC at the end of the project.

The Kick-off meeting of the project was held at JRC in Ispra (VA, Italy) on the 18th of January 2007 and was followed by three interim meetings and a final meeting. All the technical and administrative documents are available on-line in the FTP project site (address: 193.205.110.240; user: ftpeco; password: geofor). The project ended in March 2009.

1.1 Objectives of the project

This study addresses the linkages between forests spatial pattern and forest ecological functionality with emphasis on forest vulnerability, resistance, resilience.

Forest vulnerability refers to the threat of forests to suffer natural or anthropogenic damages (urban development, fires, windstorms, etc.).

Forest resistance is understood as the internal capacity of the forest to resist the influence of disturbing factors remaining unaltered and conserving its ecological functionality.

Forest resilience is understood as the internal capacity of the forest to recover after the damage.

In this document we refer to forest vulnerability, resistance and resilience as FVRR. Higher is the level of resilience, higher the level of resistance and lower the level of vulnerability higher is the overall resulting potential ecological and biological functionality of the forest ecosystem.

This study has been designed to use the results of forest spatial pattern analysis as an input for a quantitative evaluation of FVRR status and trends in space and in time in different ecological regions in Europe and at different scale levels.

Since no field data related to real FVRR values exist, the study is carried out by modelling relationships between forest spatial structure and potential FVRR that are developed on the basis of a bibliographic review.

Two approaches for modelling potential FVRR are tested:

3. The *species specific* approach is based on the use of different ecoprofiles related to four main umbrella species (Figure 1): birds (lesser spotted woodpecker), small mammals (marten), large mammals (wolf and roe deer), butterflies (lesser purple emperor). The five models are designed in order to evaluate the ecological distance between actual and optimal conditions. The models are for this reason different from traditional habitat suitability models because the final output variable is not a suitability index but an overall value of FVRR for the four different umbrella species.
4. The *species unspecific* approach is based on the use of a more general ecoprofile oriented to describe the overall biological functionality of a forest habitat on the basis of general ecology rules. The output result of the model is expressed quantitatively in the same way than for the species specific approach and the output value of FVRR provides a measure of the ecologic distance between real and optimal conditions. Such a value is anyhow referred not to a single umbrella species but to the general functionality of the ecosystem.

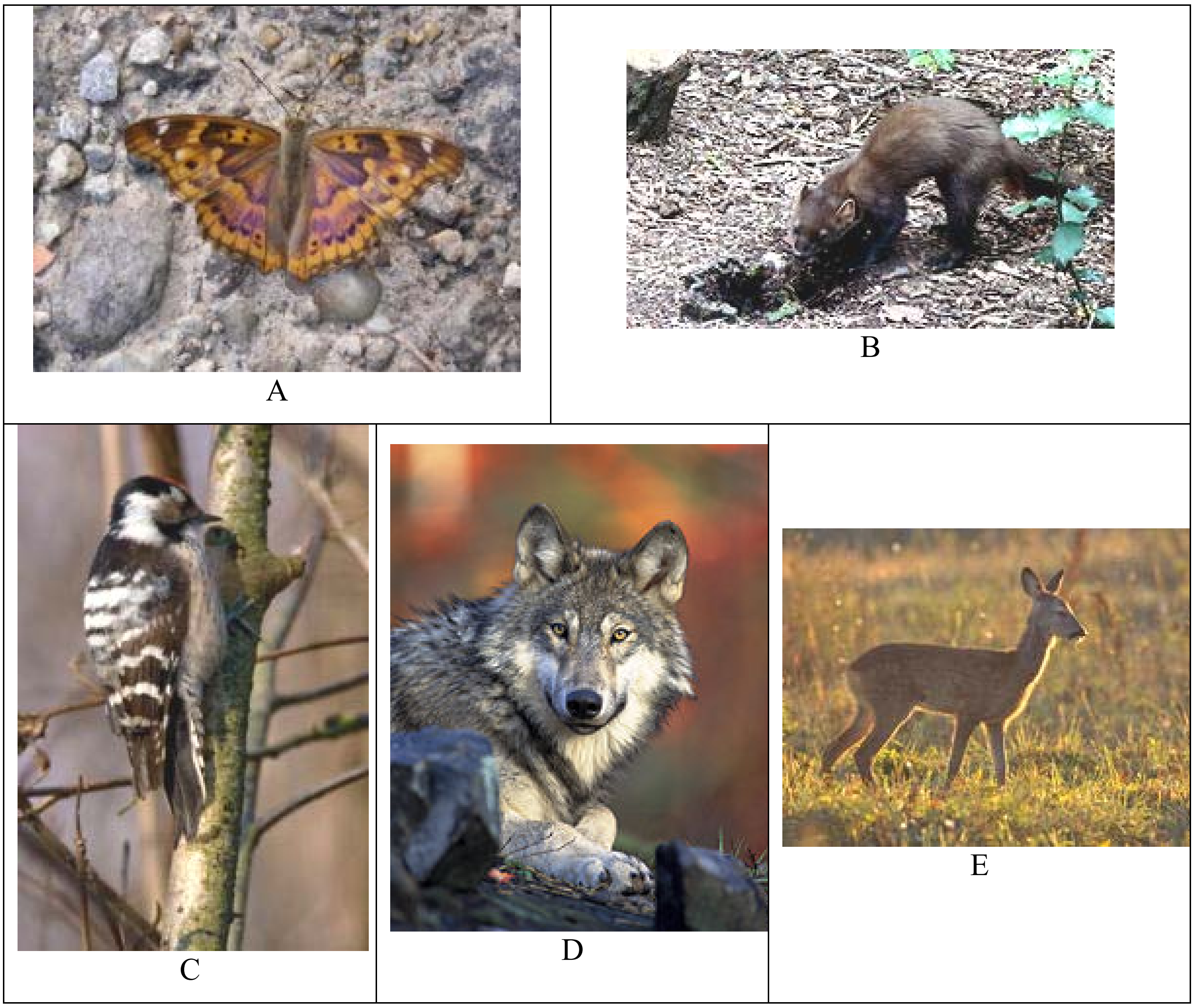

Figure 1: photos of the selected umbrella species (A: *Apatura ilia*, lesser purple emperor; B: *Martes martes*, pine marten; C: *Picoides minor*, lesser spotted woodpecker; D: *Canis lupus*, gray wolf; E: *Capreolus capreolus*, roe deer).

Test areas are designed specifically for a multiscale approach: large areas include sub-areas where data with finer resolution are available. The models for the estimation of FVRR will be applied in each test areas both on high and low resolution data in order to evaluate the level of scale dependency of the analysis. Results achieved with the same models in small test areas will be therefore available both on high and low resolution. If the results achieved at different resolutions in the same test areas with the same models will be similar we could infer that the analysis is not scale dependent and vice versa. The level of similarity will express the level of scale independency of the analysis. A high level of scale independency will enable a safe application of the proposed methodology on large areas (pan-European) based on low resolution data only. Under this point of view this project will provide a quantitative estimation of the amount of information lost in FVRR evaluation moving from high resolution data to low resolution data.

1.2 Time duration and structure of the work

The study was structured according to the following Work Packages.

The WP1 - Bibliographic synthesis on forest spatial pattern - is based on past experiences in forest landscape ecology with special regard to forest spatial pattern characterization. This task includes a bibliography review from currently on-going and recently concluded studies and a synthesis of available results. The review considers problems connected with the application of different methodologies, in different environmental conditions and on the basis of different scale of analysis.

The WP2 – Bibliography review on linking forest pattern and forest vulnerability/resistance/resilience includes a compilation of the state-of-the-art regarding the effects of forests spatial pattern processes (like fragmentation and connectivity) on the forests functions, forests vulnerability, resistance and resilience.

The WP3 - Case studies - aims at the development and implementation of an harmonized analysis method to define the potential forest vulnerability/resistance/resilience (FVRR) at two different scales, in different European biogeographical regions (Boreal, Continental, Atlantic, Alpine and Mediterranean) and address the link to pattern processes.

The project had a total duration of 24 months.

The first year of activities was devoted to:

- Complete the bibliography reviews (first two WPs) ; results are reported in the Chapters 2, 3 and 4 of the current report;
- Acquire and harmonize the spatial dataset in the selected five test areas ; the description of the available dataset is available in Chapter 5;
- Develop a consistent and harmonized model to analyse the dataset and link to the evaluation of the potential FVRR. The general approach is described in Chapter 6.

The second year of activities was devoted mainly to the model and methods implementation using the multi-temporal and multi-scale datasets:

- The final models developed for mapping the FVRR indexes are presented in Chapter 7 of the current report.
- Summary maps and statistics which are the outcome of the data analysis and models are presented in Chapter 8
- Chapter 9 is devoted to highlight the main conclusions of the project.

2 Bibliographic synthesis on pattern

2.1 Theory of landscape pattern and fragmentation

Landscape fragmentation receives increasing attention as a concept in landscape ecology since more than 20 years. Landscape ecology considers a landscape as a mosaic over which particular local ecosystems and land-uses recur and form a pattern (Forman 1998). It examines both abiotic and biotic interactions between spatial pattern and ecological processes occurring at landscape level. Landscape ecology seeks to understand the theory and behaviour of ecosystems and how their processes are maintained and changed by natural and human induced modifications (Betts 2000, Turner et al. 2001). Efforts in understanding the ecology of landscape mosaics have led to the emergence of a dozen principles (Forman 1995) of which the *patch-corridor-matrix model* is the central component in theory and practice (Betts 2000).

A *patch* as defined by Forman (1998) is a homogeneous area that differs from its surrounding. Forman (1998) noticed a correlation between patch shape and human activity; intense human activity leads to simpler, less convoluted patch shape. *Corridors* are strip-like patches differing from adjacent patches (Metzger & Décamps 1997). The *matrix* is the most extensive component of the landscape. It is highly connected, and controls regional dynamics (Forman 1998). The arrangement or structural pattern of patches, corridors, and a matrix that constitute a landscape is a major determinant of functional flows and movements through the landscape, and of changes in its pattern and process over time (Forman 1995). Together, they create a landscape pattern or landscape structure (Betts 2000). Landscape pattern thus refers to the number, size, and juxtaposition of landscape elements (Dunn et al. 1991).

Fragmentation is a spatial pattern process that refers to the “breaking apart” of a habitat (Betts 2000). Land is transformed by several overlapping spatial processes with varying effects on spatial pattern and ecological process. Landscape transformation can be due to human activity or/and natural disturbances. Four main land transformation sequences of which fragmentation is one are reviewed in literature (Forman 1995): (1) Perforation occurs when holes are made in a habitat, i.e. an extensive forest perforated by logged areas or blow downs. (2) Fragmentation is the breaking up of habitat into smaller parcels. (3) Shrinkage is the result of a decrease in the size of remaining patches or corridors. (4) Attrition is the disappearance of patches and corridors. Jaegger 2000 added two processes, related to how causing factors like linear elements like roads may fragment the landscape: (5) Dissection: habitat is divided by equal-width lines such as roads and railway tracks; and (6) Incision: occurs when a linear element enters a habitat type but does not separate the area into two patches (dissection). However, the term fragmentation is most often used in literature to encompass the four processes of land transformation. In the broader sense, the study of forest fragmentation is the study of habitat destruction (reduced habitat area) and isolation of the resulting remnants (change in spatial configuration) (Kupfer, 2006).

2.2 Forest fragmentation and spatial pattern in European policy processes

The problem of fragmentation of Europe’s forest landscape is recognized and explicitly addressed by several European policy processes (EC Biodiversity Communication 2006 on halting the loss of biodiversity by 2010 and beyond, with related Biodiversity Action Plan, EEA SEBI2010, MCPFE).

An important agreement is the *European Landscape Convention* which aims to promote European landscape protection, management and planning, and to organise European co-operation on landscape issues. Landscape related information needs which include landscape spatial pattern and fragmentation and connectivity assessments are identified through the *Convention on Biological Diversity (global* *level, and European Strategy)*, the *Pan-European Biological and Landscape Diversity Strategy (Council of Europe)* and the *Ministerial Conference on the Protection of Forests in Europe*.

Under the *Convention on Biological Diversity,* EU Heads of State or Government agreed to commit to 'halt the decline of biodiversity in the EU by 2010- and beyond- and to restore habitats and natural systems'. An EC Biodiversity Communication was adopted in 2006 with a list of indicators for monitoring progress towards this goal (the EC Biodiversity Communication, 2006). Under the CBD Focal area 'Ecosystem integrity and ecosystem goods and services', the indicator 'Connectivity/fragmentation of ecosystems' is listed as one of the 26 EU headline biodiversity indicators. The conference of the Parties considered fragmentation, along with species loss and habitat degradation, as factors that call for conservation, sustainable use and habitat restoration. A number of CBD Programmes of Work include activities to increase the understanding of the consequences of ecosystem and habitat fragmentation and to prevent and mitigate biodiversity losses due to fragmentation and conversion to other land uses (CBD/UNEP 2004). The initiative *Streamlining European Biodiversity Indicators 2010* (SEBI2010) from the European Environment Agency under request of DG-ENV aims to develop the agreed biodiversity headline indicators. SEBI2010 addresses fragmentation with the proposed indicator 'Fragmentation of natural and semi-natural areas' under the EU headline 'Connectivity/fragmentation of ecosystems' (EEA 2007). Within the SEBI2010 technical reports 4 and 5 (EEA, 2009), this indicator is reported for natural/semi-natural lands and mainly for forest. Its implementation was achieved according to three methods developed (or amended) at the Joint Research Centre (Estreguil and Mouton, 2009). European-wide maps aggregated per province were provided for the change in forest connectivity, for forest fragmentation and for change in natural/semi-natural landscape types. More information on the methods used and their implementation can be found in Estreguil and Mouton, 2009 (see also Estreguil 2005, Estreguil et al. 2007).

The *Ministerial Conference on the Protection of Forests in Europe* (MCPFE) has derived indicators defining different criteria for sustainable forest management at the policy level. They were adopted by the MCPFE as the "Improved set of Pan-European Indicators for Sustainable Forest Management" among which the indicator 4.7 addresses forest spatial pattern ('Landscape pattern: landscape-level spatial pattern of forest cover') (MCPFE 2007). An indicator and associated methodology derived from the JRC approach (Estreguil et al. 2007a and b) were for the first time proposed for the MCPFE 4.7 and implemented with the European dataset available (CLC 90 and 2000 over 21 countries) as a case study in the newly published report (MCPFE 2007, section 4.7 Parvianen and Estreguil, 2007). An important parameter is the trend of forest core area in Europe as core forests are an indicator for the overall stability of the forest ecosystem. Forest core area trends between 1990 and 2000 have been analysed at province level (NUTS 2/3 level) for 21 EU Member States. Results show that considerable changes in the forest landscape pattern took place in this short time period, they are mostly temporary due to forest management practices and few are permanent due to conversion to agricultural or urban lands (EEA, 2008; EEA, 2009; Estreguil et al, 2008ab; Estreguil and Mouton, 2009).

European-wide maps related to trends in forest pattern, fragmentation and connectivity can be queried and viewed from the European Forest Data Centre JRC web site (http://forest.jrc.ec.europa.eu/efdac/). The amount of existing indicators shows the high relevance of forest spatial pattern in the context of European biodiversity policies. Efforts are needed to implement these indicators on the European level. Operational methods of pattern analyses in forestry are available which provide data for the relevant indicators at a European scale (Estreguil et al. 2004, Estreguil et al. 2007b).

The *Pan-European Biological and Landscape Diversity Strategy* (PEBLDS) aims, among others goals, to counteract landscape fragmentation and promote ecological connectivity all over Europe. The strategy officially recognized the concept of ecological networks in Europe as an important approach for biodiversity conservation and recently, preliminary maps for the *Pan-European Ecological Network* (PEEN) have been proposed (PEEN taking Stock, Council of Europe, 2007). Also the Habitat

Directive from the European Union acknowledges the importance of landscape elements that would enhance the conservation status and the ecological coherence of the Natura2000 sites (Article 10). The EU ecological network of Natura 2000 protected sites consider sites protected under the habitat Directive and the Bird Directive.

3 Forest spatial pattern in relation to forest vulnerability, resistance and resilience

3.1 Effects of forest fragmentation on organisms

The fragmentation of landscape and habitats is among the major human-induced processes that are responsible for the loss of species diversity. Scientists suggest that habitat fragmentation may be one of the most important threats to biodiversity (Kranz et al. 2002, Honnay et al. 2005). An important research task is to understand how processes at the landscape level do influence biodiversity and to quantify the link between spatial landscape characteristics and biodiversity (Bengtsson et al. 2002). Fragmentation occurs in conjunction with loss of area and includes changes in composition, shape, and configuration of resulting landscape patches. It affects ecosystems by altering the conditions within a patch and the flow of resources among patches. Different species will respond differently to fragmentation. The differential responses will restructure the ecological community within patches, often to a state of lower species richness and high relative abundance of generalist species. Much study on fragmentation is taxonomically oriented, with a high bias towards studies on birds and mammals (Rutledge 2003, Fazey et al. 2005).

Causing factors of the fragmentation of European landscapes are agriculture, forestry, urbanization, construction of infrastructure (roads in particular) and tourism (Delbaere 1998). They create barriers for many animals and plants and so affect the dispersal of species. Populations are isolated from the main population which causes inbreeding and genetic losses (Kranz et al. 2002). Besides human-induced land use changes and forest harvesting (clear-cutting), also natural disturbances such as fire, windthrow or insects contribute to forest fragmentation. For example, the forest fires in Greece in 2007 affected 24 Natura 2000 sites and reduced territory size and habitat connectivity for forest dwelling mammal species such as the wild cat (EEA, 2008). Relevant achievements concerning the effect of past land cover changes in Europe and related pressures on habitats and their associated biodiversity have been produced within the BIOPRESS project "Linking Pan-European Land Cover Change to pressures on Biodiversity" (see http://www.creaf.uab.es/biopress/).

Habitat fragmentation implies four effects of the process of fragmentation on habitat pattern: (a) reduction in habitat amount, (b) increase in number of habitat patches, (c) decrease in sizes of habitat patches, and (d) increase in isolation of patches (Fahrig 2003). Besides patch size also boundary shape or characteristics of the surrounding landscape (landscape context) may have a significant effect on species presence or abundance in a habitat fragment (Turner 2005). In general, larger remnants and remnants, which are close to other remnants, are less affected by the fragmentation process. Small fragments of habitat can only support small species populations, which tend to be vulnerable to extinction. Moreover, small fragments of habitat have a reduced resource base for interior species compared to large fragments. Habitat along the edge of a fragment has different micro-climates and favours different species than the interior habitat. Small fragments are therefore unfavourable for those species, which require interior habitat and may lead to the extinction of those species (Turner et al. 2001). Species, which are specialized to particular habitats, and species, whose dispersal ability is weak, suffer from fragmentation more than generalist species with good dispersal ability (Turner et al. 2001).

According to a review by Rutledge (2003) fragmentation forces animals to increase the size of their home range to find enough resources to meet dietary needs. With fewer resources, individuals have fewer reserves for reproduction or combating parasites. Conversely, edge areas may actually support more species because edge areas tend to have attributes of both adjacent patches. Fragmentation changes the interaction among edge and interior species. Reproduction and mortality rates will change according to resource availability, mating opportunities, and the density of herbivores/predators. Social structures may be disrupted and reduce the opportunity for mating. At the population level, an overall increase in mortality rate implies a greater risk of extinction within the patch. Conversely, fragmentation may benefit certain species by providing a refuge if the predator or disease has difficulty moving among patches. Fragmentation often produces a series of isolated patches that remain connected through dispersal (functional connectivity). But effects vary widely depending on the process or organism (Rutledge 2003).

However, it should be noted that any findings on the effects of habitat fragmentation on species are very controversial in literature. Fragmentation may actually increase total species diversity: Although the fragmentation decreases e.g. the habitat of forest-interior species, the resulting heterogeneity increases opportunities for forest-edge and open-field populations. However, this increase in species diversity may not be desirable from a conservation perspective (Turner et al. 2001). It is important to distinguish between *habitat loss* and *habitat fragmentation*. While habitat loss has large, consistently negative effects on biodiversity, the effects of habitat fragmentation can be both positive and negative. According to a review of Fahrig (2003) the effects of fragmentation per se are as likely to be positive as negative. It is not possible to tease apart the factors that lead to positive versus negative effects. Fahrig (1997) tested the relative importance of habitat loss and habitat spatial pattern on population extinction and found that the effects of habitat loss far outweighed the effects of habitat fragmentation. In landscapes with more than 30% of suitable habitat, habitat fragmentation is primarily habitat loss. However, in landscapes with highly fragmented and low habitat amount (<10-30% of the landscape), patch size and isolation will exacerbate the effect of habitat loss and the loss of species or decline in population size will be greater than expected from habitat loss alone (Andren 1994). Lindenmayer et al. (2006) point out that species richness and populations of individual species decline more rapidly when less than 10–30% of habitat cover remained. Threshold levels are predicted to vary among landscape types and species from 60% for some taxa to 10% for others (Lindenmayer et al. 2006). Lindenmayer and Luck (2005) warn that species losses and population declines will still take place also above mentioned levels. Hypothesized 10–30% threshold levels in habitat cover will be an underestimate for many groups and some species could be lost above particular threshold_cover levels, simply as a function of habitat loss per se (Lindenmayer and Luck 2005). Hence it seems unlikely there will be generic rules for critical change points or threshold levels of vegetation or habitat cover (e.g. 10%, or 30%, or 70%) that can be applied broadly across different landscapes and different biotic groups (Lindenmayer et al. 2006).

In forestry, fragmentation leads to isolation of patches - isolation in the sense of increased distance between patches. The loss of connectivity or connecting elements in or between forest patches is also at issue when fragmentation is adressed. Also forest management contributes to fragmentation of forest types by converting primeval heterogeneous forests into a homogeneous forest landscape which, for example in the boreal region, is characterized by an increased isolation of broadleaved stands within a forest landscape dominantly coniferous. Forest fragmentation strongly effects forest species diversity. For example, cutting of primeval forest causes a decline of forest-interior species but in the same time results in the addition of many pioneer organisms that find the disturbed, open environment suitable for colonization (Franklin and Forman 1987). It is not possible to generalise fragmentation effects but important to consider both positive and negative responses of forest-dwelling species and their diverse habitat preferences when analysing the vulnerability of forest with respect to biodiversity when facing fragmentation. The following bibliographic synthesis gives a summary of results from studies on forest fragmentation effects carried out in Europe, North America, Japan and China.

3.1.1 Habitat requirements of interior and edge species

When addressing fragmentation effects on organisms, it is important to distinguish between ‘edge species’, ‘interior species’ and ‘generalists’. Following Villard (1998) the concept of habitat-interior preference was applied through the development of a classification of forest bird species among four categories: (1) forest-interior specialists, (2) interior-edge generalists, (3) edge species, and (4) field-edge species. Whitcomb et al. (1981) defined forest-interior specialists as species that ‘nest only within the interior of the forest and tend to avoid edge habitats.’ The vast majority of forest-interior species also are considered to be area-sensitive (Villard 1998). In contrary an edge species is primarily living in an ecotone, i.e. in a transition area between two adjacent ecosystems (Turner et al. 2001).

Bender at al. (1989) evaluated the conditions under which patch size effects are important determinants of local population density for animals (birds, mammals, insects) living in patchy landscapes. They showed that patch size has a strong effect on edge and interior species but is negligible for generalist species. Bender et al. (1998) conclude that (1) among generalist species which use both the edge and the interior of a habitat patch, the decline in population size associated with habitat destruction should be accounted for by pure habitat loss alone; (2) for interior species, the decline in population size associated with habitat fragmentation per se will be greater than that predicted from pure habitat loss alone; (3) for edge species, the decline in population size will be less than that predicted by pure habitat loss alone.

Caution is needed when classifying species as edge specialists. Imbeau et al. (2003) conclude from a study on edge preferences of birds that real edge species are quite rare and that a difference should be made between true edge species and species which in some landscapes, ‘happen to find their habitat requirements on edges’. They showed that most species considered to prefer edge habitats in agricultural landscapes are in fact only early-successional species that could not find shrubland conditions apart from the exposed edges of mature forest fragments.

Theory assumes that 10-ha patch size is under the minimum area required for some interior species, but that some forest interior species, such as certain salamanders and forest herbs, can survive in a 10-ha patch (Franklin and Forman 1987). According to Farina (1998), the sizes of fragments to which most species of insects, mammals and birds are sensitive are 1, 10 and 100 ha. The following sections will give more details on habitat size requirements for the different species groups.

The total area of forest interior in the landscape is related to both the total length and the width of forest edge. At boundaries between open areas and forest, the edge width is generally related to the height and structure of the forest. Franklin and Forman (1987) use a measure equivalent to two tree heights as a conservative rule-of-thumb to estimate the width of recently exposed edges. They modeled clear cutting processes and showed that the forest interior area disappears much more rapidly in landscapes with wide edges (120 m) than in those with narrow edges (20 m). With an edge width of over 160 m, no forest interior environment remains after the landscape is 50% cut over; i.e. remaining l0-ha patches are entirely edge habitat (Franklin and Forman 1987).

3.1.2 Edge effects

A concept closely allied to diversity is that of the *edge effect*. When two distinct vegetative life-forms (trees, shrubs, grass) or structures within life-forms (mature trees, saplings) meet, a boundary or edge is created between the two (Patton 1997). Edges between forest and non-forest habitats have significant effects on forest microclimate and resource availability (Gehlhausen et al. 2000). At the edge of a forested patch there is generally more light, a warmer and drier microclimate, and greater access for organisms that frequent open habitats. These characteristics have consequences on species composition and abundance. Forest edges provide habitat for a different suite of species than forest interiors (*edge species*) and may even have higher species diversity than core areas (Turner et al. 2001). Besides the species specificity, edges can also be sex- and age-specific (Barrett and Peles 1999). The species richness and composition changes across the edge to center gradient with changing abiotic attributes of the microenvironment (light, air temperature, soil moisture, humidity) (Gehlhausen et al. 2000). Biotic components, such as seed dispersal, may also give rise to changes in species composition from forest edge to interior (McDonald and Urban 2006). In fragmented forest landscapes simultaneous influences of multiple edges on remnant forest patches can occur. Their interaction can both enhance and lower the edge effectiveness (Harper et al. 2006). Knowledge of the edge phenomena is important for managing fragmented landscapes as advantageously as possible (Barrett and Peles 1999). Edge effects must also be considered when size and potential buffering habitat of forest preserves are planned (Gehlhausen et al. 2000).

Deng and Gao (2005) have shown that edge effects can negatively influence the nesting success of birds. Their study revealed that nesting success of some cavity-nesting birds (especially non-excavator species) is lower at the forest edge due to nest-site competitors and higher predation risk. On the other hand an American study demonstrated that in less fragmented forests cowbirds prefer edge locations for breeding (within 150-350 m from forest edge), whereas in a highly fragmented forest they utilize the entire forest and likely view it as ‘all edge’ (Howell et al. 2007). Magura (2002) revealed a significant forest edge effect on carabid species. Carabid species richness in a national park in Hungary was significantly higher in the forest edge than in the forest interior. Magura (2002) highlights the importance of forest edges as a source habitat for dispersal processes contributing to population re-colonisation in adjacent disturbed or destroyed habitats. Several studies have shown forest edge preference for white-footed mice (Cummings and Vessey 1994, Yahner 1992, Nupp & Swihart 1996). However, Wolf and Batzli (2002) point out that competition between mice for high quality edge habitat forces weaker animals to occupy the forest interior and hence can lead to higher mice abundance in the core forest. Forest edges may also change reproduction success of plant species. Jules et al. (1999) reports negative edge effects on a herbaceous perennial species which is close to extinction in North American conifer forests. They found lower seed production due to changes in pollination and increased seed predation by rodents. Another effect of forest edge is an often increased abundance of exotic species due to increase in resource availability near edges and increased dispersal into forest edges (McDonald & Urban 2006).

Furthermore forest edges can function both as traps and concentrators for wind-borne nutrients and pollutants from adjoining agricultural or urban landscapes and as effective concentrators of below-canopy chemical fluxes. Weathers et al. (2001) showed higher atmospheric input of sulfur and elevated through fall fluxes of dissolved inorganic nitrogen and calcium in forest edge zones compared to the forest interior. Weathers et al. (2001) indicate that these enhanced fluxes may have cascading effects on soil-nutrient cycling, microbial activity, seedling dominance, and other ecological processes near forest edges. The results of their study reveal that the structure of vegetation can greatly influence edge functioning and below-canopy nutrient fluxes. Removing the wall of lower-canopy vegetation that naturally occurs in older, unthinned edges eliminated the edge effect on below-canopy chemistry and fluxes, and instead extended it deeper into the forest. This indicates that the type of forest edge (soft/hard edge) influences the occurring edge effects and that the width of forest edge area influenced by edge effects varies. However, no generic measures and thresholds could be derived from the considered literature.

3.1.3 Impact of roads

Disturbance by an ever-increasing and expanding traffic flow has led to a further decline of mostly large animals, while traffic mortality is an important threat to specific animal groups such as mammals and amphibians (Verboom 2007). Even in densely forested areas, the fragmentation effects of roads are pervasive, significantly altering landscape structure (Saunders et al. 2002). In their study on road effects in a densely forested area in the Northern Great Lakes region, USA, Saunders et al. (2002) found that the densely forested landscape no longer retained any large, continuous forest when accounting for roads. The average size of mixed northern hardwood forest dropped from 860 to 89 ha, indicating a decline of connectivity and a substantial loss of habitat for interior-dependent species as e.g. the mean patch size used by wolf packs in the northern Great Lakes Region is 423 ha.

According to Forman and Alexander (1998), yearly 159.000 mammals and 653.000 birds are killed by road traffic in the Netherlands (status 1998) where the density of main roads is highest in Europe. The following findings of Forman and Alexander (1998) are of importance with respect to forest ecosystem vulnerability related to roads:

Concerning road kills amphibians and reptiles tend to be particularly susceptible on two-lane roads with low to moderate traffic. Large and mid-sized mammals are especially susceptible on two-lane, high-speed roads, and birds and small mammals on wider, high-speed highways. Nevertheless, except for a small number of rare species, road-kills have minimal effect on population size.

All roads serve as barriers or filters to some animal movement. Road width and traffic density are major determinants of the barrier effect, whereas road surface is generally a minor factor. Experiments showed that carabid beetles and wolf spiders are blocked by roads as narrow as 2.5 m wide, and wider roads are significant barriers to crossing for many mammals. The probability of small mammals crossing lightly traveled roads 6-15 m wide may be less than 10% of that for movements within adjacent habitats. Similarly, wetland species, including amphibians, commonly show a reduced tendency to cross roads.

A road density of approx. 0.6 km/km^2 appears to be the maximum for a naturally functioning landscape containing sustained populations of large predators, such as wolves, moose and bear. Road-effect zones generally exhibit lower population and breeding densities and reduced species richness. Average thresholds for road-effects (defined as ecological effects extending different distances from a road) are stated as follows:

- 100-200 m for large mammals in woodland with suitable surrounding habitat
- 50-100 m for large mammals in woodlands with less suitable surrounding habitat
- 50-100 m for forest interior birds with suitable surrounding habitat
- <= 50 m for forest interior birds with less suitable surrounding habitat

However, these thresholds vary depending on traffic density. The effect-distances for the most sensitive species related to high-speed roads were 305 m in woodland for a traffic density of 10.000 vehicles per day, and 810 m in woodland for a traffic density of 50.000 vehicles per day (Forman and Alexander 1998).

3.1.4 Effects on birds

Fragmentation effects on forest birds differ between species and can be both positive and negative. Several studies indicate that many forest bird species favour heterogeneous and fragmented areas (Atauri and Lucio 2001, Brotons et al. 2004, Howell et al. 2000, Lauga and Joachim 1992, Mayer and Cameron 2003, McGarigal and McComb 1999, Monteil et al. 2005). Heterogeneous area can be understood as an area with a high ratio of vegetation types per hectare (e.g. in Atauri and Lucio 2001 a heterogeneity of 0.01 provided highest bird species richness in the region studied when applying 37 vegetation types, including forest, natural grasslands, pastures, scrublands and olive groves). Fragmented forests provide more forest edges which is positively correlated to bird species richness (Mayer & Cameron 2001). However, other studies show that forest fragmentation also has clear negative impacts on numerous bird species (Bayne et al. 2005, Belisle and Desrochers 2002, Rosenberg et al. 1999, Åberg 1996). The majority of studies addressing fragmentation effects on forest birds concentrate on boreal landscapes. It is not possible to derive general conclusions on fragmentation effects on birds as results from local studies on single-species cannot be extrapolated to other species or regions (Rosenberg et al. 1999). Based on the literature reviewed it can be concluded that heterogeneity in landuse, forest patch size and total forest edge length are important criteria to consider when addressing forest fragmentation effects on birds.

Fragmentation resulting from clear-cutting as a harvesting method in Scandinavia impacts bird abundance. There has been a considerable debate as regards the maximum size of clear-cuts and impacts on biodiversity. As a consequence, sizes of clear cuts have been decrease in the past decade. As an example, in Sweden the clear cut size with highest share in total cutting area (ca. 23%) went down from 100 ha (1980-1990) to 40 ha (1990-2000) in a boreal forest landscape in Hälsingland. Only about 25% of total cutting area were connected to a clear cut size of 100 ha or more in the same region between 1990 and 2000 (EEA, 2005). Siffczyk et al. (2003) explain the population decline of willow tits in Finland during recent decades with such forestry practices. The study shows that willow tits avoid open areas (clear cuts and young sapling stands) and prefer mature forests. Åberg (1996) considers intensive forestry and the resulting changes in habitat structure as the main reason for the decline of hazel grouse in Europe. Huhta (1996) showed an increased risk of nest predation for pied flycatcher in Finland when clear-cut areas in the surrounding forest increased. This was explained with the penetration of mammalian predators from the clear-cuts to the breeding areas of pied flycatcher in search for alternative prey. According to Helle (1986) clear cutting decreased bird density by about 70% for a study site in Finland. In contrary Robinson (1999) showed for a study site in a mature deciduous forest in southern Illinois (USA) that the first cutting cycle in selective logging has only a minor effect on the forest bird community composition and creates a short-lived availability of habitat for gap species. However, fragmentation of one forest type within another due to intense forestry can have serious detrimental effects on forest-living species. Increasing isolation of deciduous forest patches within a coniferous forest is unfavourable for species reliant on deciduous forest (Enoksson et al. 1995). In the same way also the fragmentation of older forest connected with an increasing proportion of young stands due to intense forestry forms a threat to species like the forest grouse (Kurki 1997). The proportion of old-growth forests within forested boreal landscapes plays a positive role on bird distribution. Old-growth forest birds in Fennoscandia have sharply declined in numbers during the last decades as a result of commercial forest harvesting and fragmentation of old-growth forests (Brotons et al. 2003). Based on literature considered the identification of forest openings/perforation including temporary induced forest fragmentation such as clear-cuttings, and forest quality, in particular forest age, are relevant when addressing forest vulnerability. Using multitemporal datasets covering a time span of 10-50 years with a spatial resolution of 100 m or higher (e.g. data from BioPress project or Corine Landcover classification) it is possible to address effects of clear cutting on the boreal forest ecosystem based on clear cut density and clear cut cycle. The Corine Landcover classification assigns clear cuts to the class '324 Transitional woodland-shrub' which allows the identification cut-over areas. To evaluate forest quality and forest heterogeneity, applied data should at least specify the location of forest types if forest age data are not available.

Atauri and Lucio (2001) indicate that in Mediterranean landscapes the main factor explaining bird species richness is the heterogeneity of land use. Heterogeneous landscapes allow the coexistence of different communities of birds (forest, scrubland, urban), resulting in greater richness values. Similarly a study of Brotons et al. (2004) shows that species richness in Mediterranean forest fragments (Aleppo pine forest, fragments of > 1 ha size, surrounded by shrubland) does not differ from species richness in segments of continuous forests of equal area (coniferous forest here defined as forested area of at least 300 ha). The bird community of forest fragments got impoverished in some forest species but a higher proportion of species common in continuous forests were not affected by fragmentation. Brotons et al. (2004) also found that fragment communities had a significant proportion of common species that were scarce in, or absent from continuous forests. However, habitat preferences of the studied bird species (edge or interior species) were not specified in this study. On the other hand Santos et al. (2002) argues that forest birds most sensitive to fragmentation are missing in the Mediterranean. In a study for Central Spain they showed that bird richness in oak woodlands degraded from intensive use (grazing sheep and gathering firewood) was highly dependent on fragment area for all species regardless of isolation. They also found that species-area relationships of bird assemblages sampled in fragmented forests along a broad continental gradient (from Norway to southern Spain) showed that true forest birds only nest in woodlands >100 ha in southern Spain, whereas the full complement of forest species occurs in much smaller fragments in central-western Europe. This indicates that due to low habitat suitability of dry degraded Spanish woodlands the size of forest fragments is very important to forest interior species and should be at minimum 100 ha (Santos et al. 2002).

Also selected studies carried out in the Atlantic region do not imply negative impacts of forest fragmentation on birds. Lauga and Joachim (1992) assessed the abundance of six forest-depending bird species in South-western France in relation to forest fragmentation in a predominantly broadleaved forest landscape and found that local species abundance was higher in areas with high fragmentation value. However, the fragmentation was calculated based on a forest cover index and neighbouring forest cover index which did not consider forest fragment size. Monteil et al. (2005) studied links between bird species richness and forest fragmentation using neural network analyses for a study area in South-western France. Their results suggests that, for large forests (above 100 ha), increasing the area no longer has any effect on the species richness, when the regional pool of forest bird species has been reached. The study shows the strongest increase of species richness when forest fragment size increases up to 10 ha, and a further increase of fragment size (up to 100 ha) is connected with slower increase in species richness. Nour et al. (1998) examined the effects of forest fragmentation on the ability of parent birds (Great tit and Blue tit) to provide their young with an adequate food supply for study areas in Belgium. The study did not reveal suboptimal foraging or breeding conditions in small fragments compared to a nearby large forest.

A similar positive effect of forest fragments is demonstrated by Howell et al. (2000) who found greater species richness and diversity in oak-hickory forest fragments (340–880 ha) than in contiguous forest (18.000–30.125 ha) in a study site in Missouri, USA. However, there was a higher percentage of Neotropical migrants in the contiguous forest. The study indicates that the effects are highly species dependent. For example Rosenberg et al. (1999) concluded from a continent-wide study for North America that three widespread tanager species exhibit clear and similar negative responses to habitat fragmentation. However, the study does not specify critical forest fragment sizes. Howell et al. (2000) showed that forest interior species responded positively to forest core area or the mean size of forest patches, whereas other generalist species seemed unaffected by the level of forest fragmentation which was studied. Some species preferred edge-dominated sites. Also Villard et al. (1999) found that responses on fragmentation are very species-specific. They concluded from a study of 15 forest bird species in Canada that forest cover and configuration is an important predictor of bird distribution and bird species presence. In this study a total forest edge length of 20 km within a 6.25 km^2 landscape square seemed a critical value for some interior species (Hairy woodpecker, Least flycatcher and Veery) above which the species did not occur anymore.

In general the size of forest patches is positively correlated with bird species abundance across Europe. Dorp and Opdam (1987) assessed the impact of isolation on forest bird communities in small deciduous woodlots (0.1-39 ha) in the Netherlands. The study demonstrates that woodlot size is the main predictor of species number and probability of occurrence of most species. Also Monteil et al. (2005), Åberg (1996) and Robbins et al. (1989) show a strong relationship between forest area and relative bird species abundance. The correlation strength very much depends on the species. Lane et al. (2003) state that area-sensitive species (interior forest species), such as ovenbirds and scarlet tanagers, are most affected by forest patch size. They showed that in residential and agricultural areas in the USA, the abundance and reproductive success of these species depends on large forest patches (> 300 ha of mature forest for ovenbirds). On the other hand Larson et al. (2003) list the home range size for Ovenbirds in high quality habitat with 0.8 ha. Lane et al. (2003) also indicate that reproductive success of forest birds increases with patch size for some species, particularly mature forest species. In a case study on pied flycatchers Huhta (1996) explains the avoidance of small forest stands with the low amount of available food and high costs for food search. Robbins et al. (1989) conclude that most forest-nesting neotropical migrants in a study area in the Middle-Atlantic states of the US require hundreds of hectares of contiguous forest to reach a highest probability of occurrence whereas most forest-nesting short-distance migrants use small forests. Other permanent resident species in the same study area can nest in forests <20 ha. For population persistence of hazel grouse Åberg (1996) suggests a minimum habitat fragment size of 10 ha as the species was found to be strongly negatively affected by fragmentation. Larson et al. (2003) report the home range size of Ruffed grouse in high quality habitat with 5 ha. Virkkala (1991) concluded that in very large boreal forest reserves (> 1000 km^2), bird populations are buffered against impacts from the surrounding area.

Also the distance of forest fragments from continuous forest plays an important role for the occurrence of certain species in the fragments. For example in the study of Åberg (1996) hazel grouse were only recorded in forest patches closer than 100 m from continuous forest in a boreal landscape. According to Turner et al. (2001) also forest patch orientation influences bird species richness, presumably because patches that are oriented perpendicular to migratory pathways are more likely to attract birds than those aligned with migratory pathways.

Forest fragmentation also affects the competition between bird species. Mazerolle and Hobson (2001) showed that competition for breeding sites of Ovenbirds (Seiurus aurocapillus) in contiguous forest leads to a greater prevalence of individuals in better condition in these habitats. They found that males in contiguous forest were larger than males in forest fragments (9-40 ha) in a southern boreal mixed-wood forest in Canada. Therefore boreal forest fragments are considered to be of lower quality than contiguous forest for breeding Ovenbirds.

Forest bird movements are constrained in fragmented landscapes. Many forest bird species are reluctant to cross forest gaps (Lane et al. 2003). A Canadian study suggests that birds preferred to travel under forest cover rather than cross open areas (pastures and cereal fields), even when the forested detour conveyed a substantially longer route than the short cut in the open. In this study birds typically traveled under forest cover toward their destination and emerged in the open to shorten their route only when they could remain a safe distance (< 25 m) from the forest edge. This indicates that birds are unlikely to cross gaps > 50 m between two forest patches. Almost all of the studied species showed similar gap-crossing decisions This conclusion must however be restricted to the spatial scales at which the study was conducted (< 500 m) (Bélisle and Desrochers 2002). Similar Bélisle et al. (2001) show that movements of forest birds (Black-capped chickadees, Blackthroated blue warblers, ovenbirds) are constrained when they travel in deforested and fragmented forest landscapes outside migratory periods. The probability that a bird will fly in the open between two patches of forest decreases rapidly as the distance separating those patches increases (<200 m). In this study open habitats were 90% composed of pastures and cereal fields. On contrary, Norris and Stutchbury (2001) found that songbirds in a mosaic landscape of agricultural fields and forest fragments in Pennsylvania, USA, crossed open fields (maximum distance 465 m) even when corridors were present. However, longer distances inhibited male birds from traveling outside their woodlots in this study (Norris and Stutchbury 2001). Bayne et al. (2005) indicate that also forest dissection changes movements of birds. They found that ovenbirds in a boreal forest in Canada use seismic lines as territory boundaries and are less likely to cross them. Ovenbird occurrence also declined with increasing density of seismic lines. Similarly an American study showed that a transmission line functions as a barrier and territorial boundary for many forest bird species (Chasko and Gates 1982). However, regarding forest dissection by linear elements that create barriers, roads certainly have the strongest negative impact on birds (see section 3.1.3). Therefore, it is suggested to focus on roads within this project when addressing impacts of linear non-forest features in forests.

Concluding from the literature studied following parameters are considered to be of importance when assessing forest vulnerability with respect to birds:

- forest patch size (critical threshold between 10-100 ha)

- minimum distance from road (50-100 m; see section 3.1.3 Impacts of roads)
- isolation of forest fragments (critical thresholds depending on dispersal capacities of birds)
- heterogeneity of landscape and forest composition (heterogeneity usually correlated to higher species diversity)
- total length of forest edge in relation to forest patch size (high value critical for interior species, however no clear thresholds found from literature)
- density of clear cuts in boreal region (no critical thresholds found from literature)
- suggestion for edge width to apply: 60-80 m

Dispersal capacities of birds strongly vary depending on the species. Specialist as well as generalist species can be both good or poor dispersers. Lane et al. (2003) lists dispersal characteristics and habitat preferences for some forest species:

Species	**Habitat preference**	**Dispersal range**	**Type of disperser**
Red-eyed vireo	Generalist Deciduous and mixed forests as young as 8 years old, some residential areas with sufficient shrub and tree cover	Long No specific information	Good disperser
Black-capped chickadee	Generalist (forest, parks, thickets, suburbs)	Short Median of 200 m, recorded up to 20 km	Poor disperser
Northern goshawk	Generalist Upland coniferous and deciduous forest >25 years old	Long Mean of 15-20 km with limited data, known extreme of 100 km	Good disperser
Ovenbird	Interior forest Upland coniferous and deciduous forest	Long No specific information	Good disperser
Scarlet tanager	Interior forest Mature deciduous and mixed forest	Long No specific information	Good disperser
Boreal owl	Multiple subhabitats Requires mature mixed upland forest for nesting and lowland conifer for foraging	Long Frequently disperses long distances	Good disperser
Ruffed grouse	Multiple subhabitats Strongly prefers aspen forest of varying ages in close proximity	Rather short Mean of 6.4 km for males, 9.6 km for females	Poor disperser
Spruce grouse	Specialist Forests of jack pine and black spruce, usually young and dense	Rather short Mean 2.3 km	Poor disperser

Table 1: habitat preferences for some forest species (from Lane et al., 2003).

For the spatial analyses of this project it will be useful to address forest fragmentation effects on birds for different dispersal ranges using following three groups: Poor dispersers (< 3km), medium dispersers (3-10km), good dispersers (>10 km).

3.1.5 Effects on mammals

Mammals were grouped into the three classes small, medium-sized and large, depending on their size and dispersal distance. In general large mammals disperse farther than small mammals, and predators disperse farther than herbivores (Sutherland *et al.*, 2000). Many mammal species have larger home ranges than area-sensitive birds, and they may require extensive areas of habitat (Lane et al. 2003).

Small mammals

Tree squirrels are ideal models for investigations of ecological and genetic reactions of wildlife populations to habitat fragmentation, because they are usually habitat specialists and are strongly linked to woodlands. The presence of squirrels is affected by woodland size and/or the distance from the source area. Previous studies have indicated that large forest areas and/or connections between fragmented woodlots are important for the persistence of tree squirrel populations (Tamura and Hayashi 2007). Also Verboom and Apeldoorn (1990) found that the probability of red squirrel occurrence increases with woodlot size, with the amount of surrounding woods and with decreasing distance to large forest. Moreover, Tamura and Hayashi (2007) have shown that squirrel populations are affected by forest isolation. Populations in isolated woodlots host populations with lower genetic diversity compared to continuous forest. However, Delin and Andren (1999) could not find a negative impact of forest fragmentation on Eurasian red squirrels in boreal study site in south-central Sweden where the effect of habitat fragmentation seemed to be only pure habitat loss, i.e. halving the proportion of preferred habitat in the landscape should result in a halving of the red squirrel population. Bayne and Hobson (2000) demonstrated that the abundance of small mammals (11 species, including e.g. boreal red-backed voles, red squirrels, northern flying squirrels and deer mice) was lower in forest patches isolated by logging than in contiguous forest or farm woodlots surrounded by agricultural land. They found no significant difference in abundance of any of these species between small (10 ha) and large (>20 ha) farm woodlots. According to some studies voles and deer mice favour clear cut areas, whereas other studies show that voles prefer young forest (Lane et al. 2003). Altered forest landscvape structure has been suggested as a possible cause of the long-term decline in numbers of grey-sided vole in Fennoscandia. In a Swedish study Ecke at al. (2006) showed that population densities of grey-sided vole were positively and exponentially correlated with spatial contiguity of old-growth forest, indicating critical forest fragmentation thresholds (0.13 ha as minimum home range size for one reproductive female in the study area). The study indicated the negative effect of clear cuts on the density of grey-sided voles and hence the potentially negative sensitivity to increased clear cut area. Dispersal distances of small mammals vary depending on the species. Deer mouse as a generalist of woodlands and clearcuts has an average dispersal range of 50 m for males and 150 m for females. A European cogenitor of Southern red-backed vole has been recorded with a dispersal distance of up to 400 m. Moles can disperse up to 800 m. Water shrews, living in forested habitats near running water, have a dispersal range of up to 870 m. Roads and wide-open spaces tend to reduce the dispersal ability of many small mammals. (Lane et al. 2003).

Medium-sized mammals

Many forest carnivores are sensitive to fragmentation of their habitats. Chapin et al. (1998) tested the response of martens on clear-cutting in Main, USA, and demonstrated that martens prefer larger forest fragments (150-250 ha) and patches close to large fragments or an adjacent forest preserve. In their study the home ranges of martens were composed of more than 60% forest cover over 6 m in height. The average size of marten home range in California is 4 km^2 (Simon 1980, Spencer 1981). Another study showed that American martens respond negatively already to low levels of habitat fragmentation and were nearly absent from landscapes having less than 75% forest cover. Forested landscapes appeared unsuitable for martens when the average nearest-neighbour distance between open (non-forested) patches was less than 100 m (Hargis et al. 1999). Allen (1982) reports the minimum habitat area for martens with 259 ha. Thompson (2000) showed that indices of marten abundance in a study site in Ontario, Canada, were 90% greater in uncut forest than 3-40 year old logged forest. Similarly a study in Central Spain which analysed the use of forest fragments by medium-sized carnivores (red fox, badger, stone marten, genet and wild cat) demonstrates that large forest fragments are preferred by the species (Virgos et al. 2002). Lane et al. (2003) stated that hares decline following clear-cutting, but they re-colonize 6-7 years later, and increase to a peak at 20-25 years. Some examples for dispersal ranges of medium-sized mammals, according to the review of Lane et al. (2003): Martens are good dispersers and have been recorded with dispersal distances of up to 61 km. Hares are poor dispersers but can disperse up to 20 km. Male red fox can disperse in average about 22 km, females in average only 3 km, but both have been recorded also up to 300 km.

Large mammals

Most forest game species favor the open, early stages of forest succession and, therefore, probably will increase rapidly after initial clear cuttings. Many of these species make heavy use of edges or use two ecosystem types, forest and open land such as fields or clear cuts. For instance Saïd and Servanty (2005) showed that home-range sizes of roe deer decreased as forest edge density increased, which is consistent with the fact that edges are good browsing habitats for roe deer. The French study indicated that roe deer even benefited from storm damages caused by storm Lothar in 1999 as spatial heterogeneity of the landscape increased. However, game populations will decline with progressing clear cuttings of primeval forest due to loss of high-quality protective cover and winter habitat (Franklin and Forman 1987). Lesage et al. (2000) reports home ranges sizes for white-tailed deer of 1182 ha for adult males and 1102 ha for adult females. However, Maillard et al. (2002) reports only 157 ha for roe deer. For moose in Sweden Cederlund and Sand (1994) found home range sizes of 2590 ha for males and 1370 ha for females. Large carnivores usually avoid human disturbance and urban areas. Their primary habitats contain lower number of settlements, roads and railways, as urban areas and linear infrastructure create serious problems for dispersal. Collisions with vehicles and poaching are among the most important causes of mortality of large carnivores including big cats (Niedzialkowska et al. 2006). Beier (1995) reported that, in California, juvenile cougars dispersed through habitat corridors having ample woody cover, whereas they avoided highways, urban and night-lit area (Niedzialkowska et al. 2006). Increase of forest fragmentation and transportation infrastructure is also a serious threat to lynx populations. Lynx prefer undisturbed forest landscapes with more than 40% of forest cover, low fragmentation and short distances (mean distance of 19 km in this study) to contiguous forest with existing lynx populations (Niedzialkowska et al. 2006). Landscape fragmentation by streets is another serious problem for many animal species (Kranz et al. 2002). In a study on wolf dispersal pattern in Northwest Spain (Rodríguez-Freire and Crecente-Maseda 2007) show that a motorway in the study area acted as a significant barrier to wolf movement. Bona et al. (2006) point out that in recent years the major cause of mortality among a local wolf population in an alpine area in Italy has been collision with vehicles. Mikusinski and Angelstam (2004) showed that lynx occurred predominantly in regions with forest cover >50% in Europe, and Nielsen and Woolf (2002) found that core areas of lynx contained 61% forest cover in Illinois, USA. However, the Ibarian lynx prefers regions with tall shrubs (Palomares, 2001). Schadt et al. (2002) concluded that connectivity of forested and non-forested semi-natural areas is the most important habitat variable to describe lynx distribution in central Europe. In Poland, sub-adult lynx followed wooded areas during dispersal and the routes of their migration were strongly determined by the spatial distribution and connectivity of forests (Schmidt 1998, Niedzialkowska et al. 2006). Schmidt et al. (1997) stated the average home range of a female lynx in the temperate zone with about 8000 ha. Similar to lynx, also wolves avoid settled areas and roads. Away from developed areas, however, wolves likely respond positively to increased deer in diverse landscapes with a mixture of different land use types such as forests, fields and grasslands (Lane et al. 2003). The annual territory of wolf packs in Poland averages 20100 ha with a core area of territory covering in average 3500 ha (Jedrzejewski et al. 2007). A Finnish study showed a territory size of 60.000-100.000 ha per wolf couple (Kojola 2002). Also brown bears require large territories. A study of Mertzanis et al. (2005) reports that a Brown bear family in Northeastern Greece uses a home range size between 4000 and 25800 ha depending on the time of year.Dispersal ranges for large mammals are long, but depending on the species. White-tailed deer can disperse up to 9.6 km or even sometimes 40 km between summer and winter range. Wolves have long dispersal distances, typically between 80 and 160 km, but have been recorded up to 885 km. Mooses have been recorded in the USA with dispersal distances of up to 118 km. (Lane et al. 2003).

Concluding from the literature studied following parameters are considered to be of importance when assessing forest vulnerability with respect to mammals:

- forest patch size (suggested critical thresholds: small mammals 10 ha; medium-size mammals 100-200 ha; large mammals 4000-8000 ha core forest for lynx/wolf/bear/, 157 ha for deer)
- percentage of forest cover in the home range (critical threshold for some species 50-60%)
- minimum distance from road (100-200 m; see section 3.1.3 Impacts of roads)
- isolation of forest fragments (critical thresholds depending on dispersal capacities)
- distance from settlements
- suggestion for edge width to apply in the calculation of core areas: 100 m

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3.1.6 Effects on insects

Available literature on forest fragmentation effects on insects does not allow general conclusions as all reviewed studies are very much species specific. Divers habitat preferences of generalist and specialist species make it impossible to generalize. Moreover, the majority of studies address the boreal region. Dispersal ranges depend on the type of insect. Butterflies, depending on the species, can disperse over hundreds of meters or several kilometers whereas beetles usually have a dispersal of less than 10 m a month (Lane 2003). As butterflies are well studied and information on average habitat size is available for many butterfly species, this project will concentrate on those when addressing forest fragmentation effects on insects. Moreover, with the spatial data planned to use for the project it is not possible to evaluate effects on insects on large scale (10 m),

Reported effects on forest beetles are mostly negative, but dependent on local forest composition. It is important to note that beetle abundance is not only sensitive to local landscape characteristics but can reflect the amount of forest cover with a response radius of up to 2000 m (Holland et al. 2004). Rukke (2000) showed negative impacts of increased isolation and reduced habitat size on the presence of several beetle species in a fragmented forest landscape in Norway. The study, carried out both on tree level and forest island level, implies that larger forest patches (>20 000 m^2) house more populations of vulnerable species and reduce the chance of regional extinction of these species. Wolf and Gibbs (2004) found a strong relation between forest fragmentation and reduced diversity and abundance of forest-dwelling burying beetle. As for forest carabids, Koivula and Vermeulen (2005) suggest that decreasing patch size negatively affects their occurrence as well as overall species richness. However, Niemelä et al. (2006) point out that fragmentation of boreal forests and the size of a fragment are not crucial for the survival of the majority of carabid species, but that carabids requiring old-growth habitats suffer. Magura (2002) showed that carabid species richness is higher in the forest edge adjacent to grassland (about 10 m width) than in the forest interior. According to (Niemelä et al. 2006) clear-cutting in boreal forests dramatically changes the composition of carabid assemblages. Species restricted to mature forests disappear and open-habitat species invade, while habitat generalists survive at least in the short term. Koivula and Vermeulen (2005) show a negative impact of highways on carabids in forest landscapes for Finland and the Netherlands. Open habitats associated with road margins are dispersal barriers for forest carabids. Furthermore carabid assemblages of forest fragments are negatively correlated to traffic volume.

Forest composition and quality seem a more important factors for beetle diversity than forest fragmentation alone. Franc et al. (2007) showed that the occurrence of certain red-listed beetle species is very much related to the amount of deadwood and the existence of high-quality key habitats such as oak forest fragments within 1 km in the surrounding forest. They conclude that landscapes with many clustered woodland key habitats rich in oak have high priority for conservation of oak beetles.

A study of Ohwaki et al. (2007) indicates that forest interior keeps more butterfly diversity and specialists than openlands and has all endemic species. However, species richness was found to be highest in edge habitats (edge forest-grassland, considered edge width ca. 5m). The study implies that over-fragmentation of forests will cause loss of specialist and endemic butterfly species. Baz and Garcia-Boyero (1995) concluded for a study area in Spain that butterfly diversity is positively correlated with woodland area and negatively correlated with isolation. They studied woodlands of 3.6 up to 2115 ha size and included area of woodland within 1 km of the study plot as isolation criteria.

They found that round rather than long thin fragments are advantageous for the maintenance of butterfly diversity. In their study species diversity increased as patchiness of forest fragments increased. Natuhara et al. (1999) suggest that woodlands should be kept without fragmentation to preserve butterfly assemblage, but with glades or small grasslands, and with clearance of the shrub layer along the path. The BioScore project database (http://www.ecnc.nl/Bioscore/) holds a list of minimum habitat size requirements for a set of 77 focal butterfly species in Europe, i.e. species where baseline information was available and which occur between 10-20% of the surface area in at least one of the bio-geographical regions in Europe. According to these data habitat size required varies between 4 and 64 ha for the forest specialist species (i.e. species with medium or high habitat suitability for forests and only low habitat suitability for other land use classes). One forest butterfly species however (Camberwell beauty) is dependent on a habitat size of 4000 ha of broadleaved forest. For more than half of the forest butterfly species listed in the BioScore database broadleaved forest is the preferred habitat.

Sorvari and Hakkarainen (2006) indicate that anthropogenic changes in forest structures may have a potential to modify sex ratios of social insects and other forest-dwelling animals. The Finnish study demonstrates that the proportion of males of forest-dwelling ants was smaller in clear-cuts than in adjacent forests.

Insects can also be negatively affected by landscape fragmentation by streets. A German study showed that already motorways of 10 meters width prevent grasshopper species from extending their habitat to the “other side” (Kranz et al. 2002).

Concluding from the literature studied following parameters are considered to be of importance when assessing forest vulnerability with respect to insects (here: butterflies):

- forest patch size (suggestion for critical threshold: 15-65 ha)
- isolation of forest fragments (suggestion for critical threshold: 1 km distance)
- occurrence of broadleaved forest area
- total length of forest edge (high value assumably positive for species richness, however no thresholds found from literature)
- suggestion for edge width to apply: 25 m

Roads should serve as dissecting elements in the spatial analyses, i.e. continuous forest in the high-resolution data should be split into patches at the location of roads.

3.1.7 Effects on amphibians

Woodland amphibians are especially sensitive to human-caused transformation and fragmentation of their habitats. Low density, population variability, and high mobility coupled with restricted habitat needs predispose woodland amphibians to local extinction caused by habitat fragmentation (Gibbs 1998). In an American study along an urban-rural gradient Gibbs (1998) found critical fragmentation thresholds for selected woodland amphibians (at a mapping scale of 1 ha: 30% forest cover for spotted salamanders and wood frogs; 50% forest cover for red-spotted newts). These patterns are in contrast to the widely held notion that populations of the best dispersers are those most tolerant of habitat fragmentation. A review by Lane et al. (2003) indicates that amphibians, especially if they are poor dispersers and dependent on particular forms of sub-habitats such as the Four-toed salamander, are most sensitive to changes in the sizes of their sub-habitats. According that review, density and abundance of forest-dependent amphibians is reduced along high-contrast forest edges (e.g. clear-cut vs. forest interior). Negative effects along an edge gradient to a depth of 25-35 m have been found for four salamander species (Lane et al. 2003). Rosenburg and Raphael (1986) found that amphibian species' densities were lower in patches of Douglas-fir forests smaller than 10 ha. Also forest quality is of great importance for amphibians as they are often very specialist to certain habitats. Therefore clear-cutting causes dramatic reductions or local extinction of amphibians (Freedman et al. 1994). Waldick et al. (1998) show that the conversion of natural, mixed-species forests into conifer plantations reduces the amount of habitat available for amphibians. Furthermore Gibbs (1998) indicates that roads can hinder the movements of amphibian species.

Concluding from the literature studied following parameters are considered to be of importance when assessing forest vulnerability with respect to amphibians:

- forest patch size (critical threshold assumable between 1 and 10 ha)
- road density (no critical thresholds found from literature)
- density of clear cuts in boreal region (no critical thresholds found from literature)
- forest quality (natural forests vs. plantation forests)
- suggestion for edge width to apply: 30 m

Roads should serve as dissecting elements in the spatial analyses, i.e. continuous forest in the high-resolution data should be split into patches at the location of roads.

3.1.8 Effects on flora

Fragmentation has a stronger effect on plant species communities due to their lower dispersal capacities. Forest fragmentation is expected to affect patch occupancy patterns, population size and population viability of plant populations through changes in both patch area and isolation. Hamrick, J. L. (2004) addresses the problem of maintenance of genetic diversity in the face of extensive habitat fragmentation. Limited gene flow between fragments results in a genetic drift. Intra-fragment genetic diversity decreases while inter-fragment diversity increases. However, Hamrick, J. L. (2004) indicates that many forest trees may be buffered from the adverse effects of habitat fragmentation because their individual longevity, high intra-population genetic diversity and the potential for high rates of pollen flow counteract effects of genetic drift for tree species.

Influence of forest area and patch size on plant species richness are discussed controversially in literature. According to Berglund and Jonsson (2001) the fragmentation of old-growth forest habitat by modern forestry has become a major threat to plant and fungal species diversity in Fennoscandia. They showed in a study that total plant and fungal species richness as well as the density of lichens and red-list species in the forest interior is positively correlated to forest area. Fukamachi et al. (1996) point out that large patches of forest reserves tend to have relatively infrequent species. However, they could not detect a significant correlation between plant species richness and forest pattern. Guirado et al. (2007) concluded that large-scale effects attributable to landscape structure and fragmentation play a minor role for species richness in periurban forest patches in Spain.

Forest fragmentation strongly negatively affects reproductive success of pollination-specialist plant species. Aguilar and Galetto (2004) demonstrate in a Spanish study that quality and quantity of pollination for Cestrum parqui (a self-incompatible pollination-specialist flowering plant) was decreased due to fragmentation. They indicate a critical threshold of about 5-14 ha forest fragment size which still allows successful reproduction for this species. Similarly, a Belgian study on a common forest herbaceous plant species detected that patch area and isolation influences regional persistence of plant populations through altered colonization probabilities and reduced reproductive success of small populations. Plants from small populations had a significantly lower individual fitness than plants from large populations. Moreover, small populations produced significantly fewer seeds per fruit and per plant than did large populations (Jacquemyn et al. 2002). Tomimatsu and Ohara (2006) quantified the effects of forest fragmentation on populations of a representative spring herb in Hokkaido, Japan, and got similar results of reduced fertility. Small herb populations generally produced fewer seeds and experienced genetic loss and inbreeding. Kolb and Lindhorst (2006) noted reduced reproductive success for small populations of Sanicula in a German study on habitat fragmentation. They found evidence that largely out-crossing, non-clonal species are more sensitive to reductions in population size in terms of their reproductive success.

Forest fragmentation also effects the proportion of introduced species in forest patches. A Canadian study of Duguay et al. (2007) revealed that forest fragments in urban landscapes had about 40% more introduced plant species and a 50% greater proportion of introduced plant species than fragments found in a predominantly forested landscape. The results support the hypothesis that urban and suburban areas are important foci for spread of introduced plant species.

3.1.9 Conclusions on forest vulnerability, resilience and resistance

Forest biodiversity is one of the main forest functions – and probably the most important one. The forest ecosystem provides habitat for various forest-dwelling animal and plant species which form the forests biodiversity. Effects on the forest ecosystem – such as forest fragmentation – influence species' habitats and hence have an impact on species populations and biodiversity. As the previous sections have shown, especially animal species are dependent on minimum habitat requirements to survive in an ecosystem. If certain thresholds (see sections 3.1.3 - 3.1.8) are under-run, such as minimum forest patch size, the survival of a species is critical or even impossible as such forest fragments do not provide suitable habitat any longer. The vulnerability of forest species and forests biodiversity due to fragmentation is directly linked to forests vulnerability. The forest ecosystem is a functional unit of a dynamic complex of plant, animal and micro-organism communities and their abiotic environment. If components of this 'functional unit' are negatively influenced by forest fragmentation, this will harm the forest ecosystem as a whole and increase forests vulnerability.

Above critical thresholds a state of resilience and resistance of the ecosystem and hence of the forests biodiversity can be assumed. It is important that forests provide habitat characteristics above the critical thresholds pointed out in the previous sections (3.1.3 - 3.1.8) to assure and maintain an intact functioning forest ecosystem, and therefore to assure forests resilience and resistance.

3.2 Connectivity and population dynamics in fragmented landscapes

Connectivity has become one of the central concepts in landscape ecology. Patches are not isolated entities but connected to other patches by exchange of matter and organisms (Van Diggelen 2006). Landscape connectivity can be defined as the degree to which the landscape facilitates or impedes movement among resource patches (Taylor et al. 1993). Increasing connectivity has been frequently proposed as an effective strategy to address biodiversity decline within fragmented habitats (Bailey 2007). Model simulations for random landscapes have shown that ca. 60% of the landscape must be covered by suitable habitat in order to ensure organism movement between patches (Gardner et al. 1989). However, in real landscapes this percentage can be much lower if *corridors* exist which connect habitat patches (Van Diggelen 2006). Whether a habitat is connected or not is a threshold issue that depends on both the abundance and spatial arrangement of the habitat, as well as the movement or dispersal characteristics of the organism. Different species might perceive different thresholds in the same landscape. Hence, connectivity is a scale-dependent phenomenon (Turner et al. 2001).

It is important to distinguish between *structural* and *functional* connectivity, that means the measured spatial connectivity in the landscape (connectivity by adjacency of habitats) and the functional (or biological) connectivity experienced by species (connectivity by dispersal abilities of organisms) (Bengtsson et al. 2002, Brooks 2003, Van Andel 2006). While structural connectivity refers to the degree of habitat connectedness, functional connectivity – while related to structural connectivity – refers more directly to the ease with which organisms move across a landscape and is therefore of greater interest when assessing isolation effects. Functional connectivity of a species is depending on a range of factors that are not easily captured by existing landscape metrics (e.g. stepping stones, matrix characteristics, width and quality of corridors, gap crossing willingness of the species). Measures of structural connectivity are thus probably important only to the extent that they capture the underlying functional connectivity (Kupfer et al. 2004).

Corridors are an important factor for biodiversity conservation as they bridge gaps in fragmented landscapes. A corridor can be defined as strip-like patch differing from adjacent patches (Metzger & Décamps 1997) and connecting two or more larger blocks of habitat. Corridors aid in the flow between patches, facilitate animal dispersal and hence enhance or maintain the viability of specific wildlife populations in the habitat blocks. They provide the possibility for movement of individual animals from one habitat patch to another. Also non-connective corridors can be important for conservation issues, such as linear habitats that support breeding populations of many species even though they do not connect larger habitat patches (Beier and Noss 1998). However, functional corridors may not be discrete structures but can be diffuse and difficult to identify. Reduced contrast between habitat patches and the intervening matrix may enhance connectivity more than would a discrete corridor. Although protecting naturally existing corridors probably benefits regional and local biodiversity, the creation of linear patches may not provide such benefits (Gustafson and Gardner 1996, Rosenberg et al. 1997, Turner et al. 2001).

Connectivity by corridors is discussed controversially among scientists. Beier and Noss (1998) conclude that generalizations about the biological value of corridors will remain elusive because of the species-specific nature of the problem. However, Kupfer et al. (2004) indicate that a few generalizations can be made about the effectiveness of corridors. Given that forest fragmentation is associated with the loss of forest area and decreased connectivity, the most effective corridors will be those that support breeding populations and dispersal of their offspring as well as facilitate the general movement of biota. The value and effectiveness of corridors, however, also depends on habitat quality and degree of forest loss. Corridors would be most valuable at levels where functional connectivity exhibits the most rapid decreases with small changes in forest landscape structure (Kupfer et al. 2004). Bailey (2007) indicated a lack of firm empirical evidence that species diversity will increase following attempts to increase connectivity in fragmented woods. Biodiversity loss as a result of regional habitat loss rather than fragmentation cannot be significantly reduced by adjusting habitat pattern. Simberloff and Cox (1987) even discussed potential disadvantages of corridors, which include facilitation of the spread of disease, pests, fire and other disturbances, increased predation, and high costs of maintaining linear remnants with high edge-area ratios (Saunders et al. 1991).

The concepts of island biogeography and metapopulation study the dynamics of spatially structured (meta)populations, i.e. the population exchange between habitat patches. A metapopulation is defined as an assemblage of local populations that are connected by mutually dispersing individuals (Van Andel 2006). As for plants, seeds and pollen are the major dispersal units. Dispersal reduces the risk of extinction of a population as a whole, even if local subpopulations become extinct, provided that suitable sites are within travelling distance (*rescue effect*). For example, a study showed that white-footed mice have persisted in a remnant network of woodlot patches in North America, connected by migration routes. The level of migration influenced the growth rates of the population. This indicates that migration can buffer individual populations from negative events in their local environments (Van Andel 2006). It can be distinguished between source and sink populations, meaning that sink habitats are maintained by continuous immigration from the source habitats. However, dispersal capabilities are species-specific and can be low, especially for plant species. Studies in wind tunnels and in the field found that seeds hardly reach distances of more than 10 m from the parent plants (Van Diggelen 2006). But also the movement of animals is restricted. For instance Åberg (2000) showed that forest hazel grouse is a poor disperser and avoids open areas. He indicated that the threshold distance for movement of hazel grouse between forest patches in an agriculture-dominated boreal landscape is about 200 m. Richards et al. (2002) demonstrated that the dispersal success of wildlife species in fragmented forest landscapes increases with larger dispersal distances (dispersal capability) and smaller home range sizes.

Uezu et al. (2005) emphasize the importance of understanding species-specific perceptions of landscape when analyzing functional connectivity. They showed that two out of seven examined Atlantic forest bird species were affected by forest patch connectivity – either by the presence of corridors or by the distance between patches. The capacity to use corridors and open areas (i.e. functional connectivity) shaped the abundance pattern of White-shouldered Fire-eye. The open areas varied from a small road (10 m length) to gaps of up to 130 m. Also Belisle and Desrochers (2002) indicate that forest corridors play an important role for movements of birds. They showed in a Canadian study that forest birds prefer to travel under forest cover rather than cross open areas, even when the forested detour conveyed a substantially longer route than the short cut in the open. Most birds rarely ventured >25 m from forest edges despite having the opportunity to do so. D'Eon et al. (2002) suggest that old-growth forest associates in Canada, such as carnivorous birds, perceive the landscape as connected and are able to access all forest patches. Smaller, less vagile species, such as woodpeckers, chickadees, and nuthatches, may be affected by lack of forest connectivity (by e.g. harvest patches) at the scale of their interaction with old-growth patches. They identified the northern flying squirrel as of particular concern due to relatively weak dispersal abilities. Várkonyi et al. (2003) who studied movements of two noctuid moth species in Finland found that they favoured old-growth forest corridors and avoided entering the non-habitat areas of the surrounding landscape matrix during their movements. Bona et al. (2006) demonstrated that deer in an alpine area in northwestern Italy had strong preference for using the corridors. Hansson (2001) suggested the establishment of dispersal corridors to improve the quality of key habitats containing red-listed species in Swedish managed forests.

An answer to the problem of habitat fragmentation is the development of ecological networks which link nature reserves by means of corridors and small habitat patches (stepping-stones). Ecological networks aim to conserve or restore habitats and species populations by (re)establishing interconnectivity to counteract habitat fragmentation. They provide opportunities for dispersal and species migration and buffer habitats from potential threats. The development of ecologically coherent networks is part of European policy (Bern Convention, Habitat Directive, Natura 2000). During the Seventh Conference of parties of the Convention on Biological Diversity (2004) ecological networks were incorporated in the work program on protected areas as key conservation strategy. The Pan-European Ecological Network is an internationally agreed approach, built upon the ecological network concept. It is one of the most important implementation tools of the Pan-European Biological and Landscape Diversity Strategy (PEBLDS) but is not legal binding. The Natura 2000 Network consists of a network of special areas of conservation and protection for birds and habitats.

The *Gap analysis* approach, which analyses existing preserves and land cover types, helps to identify regions with high species diversity that remain unprotected (Turner et al. 2001). In the process of identifying such gaps, the extent of habitat fragmentation can be determined and this in turn will reflect the need for habitat corridors (Patton 1997). Opdam et al. 2003 propose *spatial cohesion* as sustainability indicator to determine whether the size and connectivity of ecosystem networks is sufficient for sustainable biodiversity protection. Rodriguez and Maseda (2006) studied the connectivity of a woodland bird species in Northwestern Spain within the Natura 2000 Network and suggest that the integration of non-protected patches into a regional network of identified corridors which interconnect spatial conservation areas of the Natura 2000 Network would further improve the network functioning. Van der Sluis et al. (2003) demonstrated that the ecological network of the Abruzzo region in Central Italy successfully counteracts fragmentation and hosts sustainable populations of several indicator species except for wolves. They point out that corridors effectively provide local connectivity and are essential to maintain the high quality of nature in that region. The study concludes that more corridors between high quality habitats are needed to increase connectivity within the network and to allow better movement for e.g. wolf and Brown bear populations. Van der Sluis et al. (2003) recommend the development of multiple use corridors (i.e. of combined habitat types). They also criticize the negative impact of highway traversing the network which forms a barrier for several species. Similarly Bona et al. (2006) and Woess at al. (2002) point out negative impacts of linear infrastructure crossing corridors and migration routes on ecological networks in Italy and Austria. Crist et al. (2005) stress the importance of roadless areas for connectivity. Roadless areas reduce isolation of habitats and help to maintain wide-ranging species movements. When relatively undisturbed and well-distributed among protected areas, roadless areas can be considered high-quality ‘habitat connections’. Noss (2001) suggests maintenance of habitat linkages parallel to climatic gradients and minimization of artificial barriers as a prudent strategy under any climate-change scenario.

The maintenance of connectivity is important for the proper functioning of ecosystems, which can be defined as dynamic complexes of plant, animal and micro-organism communities and their non-living environment interacting as functional units (Kettunen et al. 2007). Functional connectivity within landscapes is a result of a species’ use of the landscape. For functional connectivity to exist, landscape elements allowing the species’ use of the landscape, including movement of species within the landscape, need to be in place. The nature and scale of these elements can differ significantly between species and consequently species-specific requirements need to be carefully considered, for example, when developing suitable management/conservation strategies for species at a broader landscape level. Landscape connectivity is inherently neither good nor bad (Taylor et al. 2006). Through its effects on ecological processes, connectivity may positively influence population persistence for some organisms in some situations and negatively influence them in others..

The evidence that corridors provide benefits by increasing connectivity, rather than simply by providing additional habitat, are equivocal, largely because of the practical difficulty of distinguishing between these two effects and because of methodological shortcomings in previous research (Kettunen et al. 2007). Dawson 1994 concluded after reviewing previous studies that animals and plants prefer to move along corridors rather than cross the matrix habitat, but an approximately equal number found no detectable effects and few, if any, showed that recolonisation would not have occurred without corridors. None of the studies conclusively demonstrated that corridors act as conduits that prevent extinctions in patches, possibly because few were sufficiently rigorous to demonstrate unambiguous advantages. Overall, Dawson 1994 concluded that corridors:

1. ‘Sometimes allow individual animals to survive by allowing them access to sufficient habitat to meet their needs;

2. May maintain populations of some animal and plant species by replenishment; however, most species probably fail to use a corridor or can cross the gaps between patches of habitat adequately without its aid; and.

3. Can serve the needs of some migratory animals in their seasonal movements’. Others have come to similar conclusions (Davies & Pullin 2007; Donald 2005; Donald & Evans 2006). Wiens 1995, suggested that the ‘evidence that species do depend on corridors for their movements or that corridors have clear conservation value is limited and equivocal’. Little evidence was also found of the potential benefits of corridors in relation to movements required as a result of climate change (Davies & Pullin 2007; Wiens 1995).

Some studies have found some evidence of benefits from corridors. For example, Gonzalez et al. (1998), have demonstrated significant effects of corridors in preventing metapopulation extinction by providing an immigration ‘rescue effect’, and Mech and Hallet (2001) used genetic methods to argue that corridors increase connectivity for specialist mammals. Beier and Noss (1998) found convincing connectivity benefits of corridors, but in only around half of all published studies, largely because too few studies have included all the necessary demographic parameters. More recently a review by Debinski and Holt (2000) suggested that although the predicted positive relationship between species richness and fragment size is rarely apparent in empirical data from patches of natural habitat in fragmented landscapes, there is a consistent agreement across many studies that increasing connectivity increases species richness, and that movement is related to connectivity.

Despite these studies, it still remains unclear whether increases in movements and species richness are the direct result of connectivity, or simply because corridors provide additional habitat area. Haddad & Tewksbury 2006 further noted that the effect of corridors on population viability is little studied and the empirical understanding of the effects of corridors on community structure and diversity is still in its infancy. Although they find that support for corridor effects on population is growing, especially for smaller taxa with short generation times (because these are easier to study), there are many caveats.

Although there is little clear evidence that corridors directly provide clear population benefits, it might be prudent to assume that corridors should be maintained in accordance with the precautionary principle. This seems particularly prudent given the difficulties associated with demonstrating their impacts. Consequently Beier and Noss (1998), reviewing the complexity and intractability of this issue, suggest that 'those who would destroy the last remnants of natural connectivity should bear the burden of proving that corridor destruction will not harm target populations'. On the other hand, in the absence of conclusive evidence of the functional benefits of corridors, the costs of establishing them need to be compared critically against the costs and potential benefits of alternative conservation approaches (Simberloff et al. 1992).

This section has shown that lack of connectivity has a negative impact on the forests ecosystem as many forest-dwelling species are depending on functional connectivity between forest patches to successfully reproduce. If functional connectivity is not provided in a forested landscape, these species are vulnerable to local extinction, contributing to a loss of forest biodiversity. Decreased forests biodiversity is directly linked to forests vulnerability because it harms the functioning of the forest ecosystem. However, with increased connectivity the risk of biodiversity loss can be lowered and valuable functional habitat can be provided for species which would otherwise be vulnerable. This contributes to an increased forest ecosystem resilience and resistance. As the previous chapter has shown, general connectivity measures are difficult to define as they are species-specific. Generally, providing and maintaining functional corridors and 'stepping stones' for key species will increase the stability of forest ecosystems and hence lower forests vulnerability.

3.3 Effects of forest fragmentation on disturbance of the forest ecosystem and its resilience

Most ecosystems are subject to several disturbance regimes at different temporal and spatial scales. Natural disturbances usually are pulse disturbances with a characteristic magnitude and frequency, but human activities tend to transform pulse disturbances into press or chronic disturbances that reduce ecosystem resilience. Fragmented landscapes are likely to lead to a lower capacity to recover naturally after disturbances (Bengtsson et al. 2003) while spatial homogeneity often enhances the spread of disturbance (Risser et al. 1984). Low species diversity in managed homogeneous forest landscapes increases the likelihood of unexpected catastrophic change (Drever et al. 2006).

Van Andel and Grootjans (2006) distinguish three concepts of stability for an ecosystem:

- Stability: is the capacity of a system to maintain a relatively constant state in spite of changes of environmental conditions;
- Resistance: Systems are considered resistant if they show relatively little response to a sudden change in environmental conditions and maintain their structural and functional attributes;
- Resilience: Systems are considered resilient if they can be altered relatively easily but will return to the initial state more rapidly and regain structural and functional attributes that have been damaged due to changes in environmental conditions.

Ecological resilience of a fragmented ecosystem – the capacity to absorb disturbances without undergoing fundamental change – is depending on the *ecological memory*. Bengtsson et al. (2003) distinguish two types of ecological memory: The internal or within-patch memory which consists of the biological structures that serve as foci for regeneration and allow species to colonize, and the external memory of surrounding habitats in fragmented landscapes which provides sources of and support areas for species colonizing disturbed patches (between-patch dynamics). Between-patch dynamics are based on dispersal and hence are limited for plant species. Another part of the external memory, determining the fate of disturbed areas, is organisms dynamically linking disturbed and undisturbed patches, e.g., herbivores and predators. The build-up of ecological memory in forests in the form of biological legacies and species in the mosaic landscape usually takes several forest generations (Bengtsson et al. 2003).

According to Malanson et al. (2007) resilience is determined primarily by the recovery of the best competitor, followed in importance by the remaining species as ordered in the competitive hierarchy. Results of a simulation study on ecological consequences of deforestation for plant diversity and community structure show that the initial degree of habitat alteration has little effect on resilience. Adaptation to an ongoing disturbance regime, which should improve resilience, is not evident when the disturbance itself continues. (Malanson et al. 2007).

The resistance and resilience of forests to climate change will likely be lower for fragmented forests compared to intact forests (Noss 2001). Fragmentation also has potentially serious impact on species migration caused by climate change. Pearson and Dawson (2005) indicate that in a fragmented landscape only those species with long-range dispersal capabilities might be able to migrate rapidly enough to keep pace with changing climate. By increasing the isolation of habitats, fragmentation is expected to interfere with the ability of species to track shifting climatic conditions over space and time. Weedy species, including many exotics with high dispersal capacities may prosper under such conditions, whereas species with poor mobility or sensitive to dispersal barriers will fare poorly (Noss 2001).

Connected to climate change, increasing frequency and severity of storms in Europe form a serious threat to fragmented forest landscapes. Franklin and Forman (1987) modeled ecological consequences of forest clear cutting in the western United States and showed that the potential for catastrophic windthrow in residual stands of primeval forest strongly relates to the forest landscape patchwork. Windthrow susceptibility increases with the amount of forest edges. Also Gratkowski (1956) stresses the disadvantage of forest harvesting by clear cutting as it has proved to enhance windthrow along the cutting edges, with most damages occurring within 200 feet from the edge. Results of a French simulation study indicate that the extensive damages caused to French forests by storm Lothar in December 1999 can not only be explained by the fragility of tree species or by local soil conditions, but rather by high levels of forest fragmentation which have induced very heterogeneous wind flow patterns resulting in the formation of local regions of particularly high turbulence intensity. A Canadian study showed that increased isolation of forest patches is significantly related to damages by ice storms (Pasher & King 2006). Heterogeneity within the forest decreases the risk for windthrow. Schütz et al. (2006) found that admixture of 10% or more broadleaved tree species reduce the vulnerability of spruce stands to storm damage by a factor of more than three.

Fragmented forests can also be more likely vulnerable to fires. Following Franklin and Forman (1987) sources of wildfire increase rapidly with development of access roads and creation of logging slash connected to clear cutting. The probability of fire spreading in residual primeval forest reflects the extent of edge influence in the patch. Cutover areas increase the probability of a fire spreading to a residual forest patch. On the other hand, Lloret at al. (2002) observed for a test site in Spain that fires were more prone to occur in homogenous forest areas with higher mean patch size. Franklin and Forman (1987) indicate that wildfire control should be easier in a landscape with greater heterogeneity (including firebreaks, such as roads).

Exotic species are now recognized as the second leading cause of species endangerment and extinction, after habitat loss. Habitat fragmentation is often viewed as providing a highway for invasion into forests and other habitats (Simberloff 2001). Broad-scale disturbances caused by habitat loss and fragmentation are believed to facilitate the spread of exotic species (With 2004). The study of With (2004) indicates that invasive spread may be enormously enhanced beyond some threshold level of habitat loss, which depends upon the species' dispersal abilities and the degree of habitat fragmentation. Colonization success of invasive species is predicted to be highest when more than 20% of the landscape has been disturbed.

Forest fragmentation may exacerbate outbreaks of caterpillars by decoupling them from their natural enemies (Roland 1993, Roland and Taylor 1997). Saunders et al. (1991) indicate that elevated springtime temperatures on trunks at the forest edge allow larvae to emerge before their parasites, which emerge from the cooler forest floor. This gives the pest a head start and results in population buildups. On the contrary, Holdenrieder et al. (2004) note that fragmentation might not be undesirable for the management of pathogens provided the goal is to limit their presence in the landscape. Condeso & Meentemeyer (2007) for instance have demonstrated that the emerging forest disease sudden oak death in North America shows increased severity within contiguous woodlands. According to Franklin and Forman (1987) responses of pests and pathogens to a developing patchwork are highly species specific, which reflects the interactions of their life histories (including dispersal mechanisms) and environment (including host distribution).

Whether forest fragmentation enhances spread of disturbance or not depends very much on the way the disturbance spreads: within the same habitat type (such as the spread of a species-specific parasite through the forest) or between different habitat types by crossing boundaries. In the first case forest heterogeneity and fragmentation will retard the process of disturbance whereas in the second case fragmentation and edge effects will enhance disturbance spread (Turner et al. 2001). The propagation of disturbance in heterogeneous landscapes depends on the structure of the landscape as well as the disturbance intensity and frequency (Turner et al. 1989).

Forest fragmentation also impacts other important forest functions as it strongly alters the fluxes of solar radiation, wind and water across the landscape. The energy balance of a fragmented landscape differs markedly from one with a complete cover of native vegetation, especially where the native vegetation was dense before clearing. Forest functions such as nutrient cycling and protection of soils may be affected by fragmentation (Saunders at al. 1991).

Fragmentation creates more forest edge which is often adjacent to patches with a more open physical structure such as pasture or urban areas. The edge areas tend to receive more solar radiation during the day and have a higher re-radiation at night which increases incidence of frost (Rutledge 2003, Saunders at al. 1991). Besides higher daytime temperatures, increased solar radiation produces drier conditions, particularly when coupled with increased airflow from surrounding open areas. The same processes can also affect soil conditions through heating and drying (Rutledge 2003). Malcolm (1998) modeled heat flows for a simulated fragmented forest landscape – though for a tropical region – and showed that the clearcut area served as a heat source and its temperature was maintained at a constant value higher than in the initially cooler forest. The surrounding neighborhood was sensitive to the clearcut temperatures and at high conductivities heat could flow far into the forest.

Forest fragmentation also modifies the local water regime. Clear-cutting leads to greatly reduced evapotranspiration and increased surface- and groundwater flows. The hydrological system in general becomes much less buffered with more extreme runoff events. Soil moisture levels change and the increased surface water flows result in increased soil erosion and transport of particulate matter and nutrients. Rises in water tables can bring stored salts to the surface and cause secondary salinity, with considerable impacts on remnant vegetation. However, these impacts greatly depend on the position in the landscape. Forest patches in run-off areas can be expected to experience more erosion, while those in run-on areas will experience more soil and nutrient deposition. Run-on areas with high deposition of nutrient-rich material can act as a focus for invasion by species requiring disturbance and/or nutrient enrichment for successful establishment (Saunders et a. 1991).

Fragmentation leads to a reduction in patch sizes and an increased isolation of the remaining patches. After the process of fragmentation, a forest remnant undergoes changes because it becomes, to some degree, an island, and being smaller and more isolated it cannot support all the species that it held as part of a larger habitat area (MacArthur & Wilson, 1967). Although the primary focus in fragmentation research tends to be on the pattern of fragmentation and the resulting effects on remnant forest ecosystems, the areas altered in fragmented landscapes are not static (Rudel et al. 2002) and may, through plant succession, come to increasingly resemble the forested remnants (Moran & Brondizio, 1998). This is especially likely if the disturbance is temporary and does not substantially alter site characteristics or resource availability (e.g., soil quality).

While persistent or irreversible changes in plant community structure or composition following disturbances were once regarded as rare incidents that only occurred following exceptional events or particularly extreme disturbances, research has shown that past human land uses and disturbances can be important determinants of species composition, biodiversity, community pattern, and ecosystem function even in landscapes that outwardly appear “natural” (e.g., Duffy & Meier 1992, Goodale & Aber 2001, Foster et al. 2003).

It is therefore important to understand not only the processes or steps associated with fragmentation, but also how these relate to the recovery of cut areas. The recovery of cleared areas will depend to some degree on the same processes and effects as the remnants, with the primary mechanism being plant succession. The linked processes of forest loss and recovery can be categorized by the degree of habitat destruction and the amount and rate of recovery (Malanson et al. 2006)..

In terms of ecosystem response to a disturbance, inertia (or resistance) describes how much a landscape changes given a specific force (e.g., a disturbance) (Milchunas & Lauenroth 1995). With respect to natural disturbances, differences in inertia are expected, dependent on the disturbance regime, on natural productivity, and on history. Forests subject to a fire regime with occasional stand-destroying crown fires (e.g., lodgepole pine communities) will have different inertia than stands characterized by a regime of frequent but low intensity surface fires (e.g., ponderosa pine communities) (Veblen et al. 2000). The inertia of forests to human fragmentation agents such as logging will be a function of characteristics of both the disturbance itself (e.g., logging type or intensity) and the response of the remnant vegetation to changes effected by the disturbance (e.g., altered microclimatic conditions).

Resilience or recovery of the landscape may be due primarily to the characteristics of the matrix if it is the most extensive area (Dale et al. 1998), but remnants can recover simultaneously, if also disturbed. Westman (1978) identified four components of resilience:

- Elasticity is the rate of recovery. It can be measured as the similarity to the pre-impact landscape over time.

- Malleability is the degree to which the recovered landscape differs from the pre-impact state. Some landscapes never approach a pre-deforestation condition, and changes in the abiotic environment may make exact recovery impossible. Also, pre-fragmentation forests may not have been in equilibrium with the environment.

- Amplitude is the amount of change that can occur before the landscape cannot recover toward its pre-impact condition. It identifies a system threshold, beyond which recovery is impossible. It thus depends on inertia.

- Hysteresis is the degree to which the path of recovery varies from the path of impact-change. For fragmentation, the concept can best be applied to examining differences among pathways of recovery due to dynamics.

According to Malanson et al. (2006) the initial degree of habitat alteration has little effect on resilience. Allowing succession to follow disturbance has a strong effect, which differs among species. Adaptation to an ongoing disturbance regime, which should improve resilience, is not evident when the disturbance itself continues. Fragmentation changes the spatial pattern of the landscape, and species respond differently because of their different dispersal abilities. The species in turn alter the spatial pattern.

The literature review has shown that several factors contribute to forest fragmentation and therefore contribute to increased forest vulnerability: Besides human-induced forest fragmentation by clear-cutting, urban spread and development of infrastructure also natural disturbances such as fires and storms cause forest fragmentation. The extent of forest fragmentation caused by natural disturbances is connected to the duration and severity of the disturbance regime. The literature study also showed that fragmented forests can be more vulnerable to natural disturbances. Effects of forest fragmentation are therefore not only connected to the survival capacity of forest-dwelling species but also to the capacities of forests to resist and overcome natural disturbances. As the scope of this project is to analyse fragmentation impacts on the forests ecosystem vulnerability, resistance and resilience, it should be focused on how the fragmentation of forests impacts forests vulnerability towards disturbance regimes rather than analyzing disturbance regimes as a cause of fragmentation. It can be concluded from the previous chapter that forest resistance and resilience towards natural disturbances is higher for forests with little fragmentation, whereas highly fragmented forests are more vulnerable to such disturbances. However, literature does not provide critical fragmentation thresholds related to natural disturbances. Moreover, also the forests composition is important to consider when addressing this problem as heterogeneous forest landscapes have higher resistance against storms e.g. than mono-cultural forest plantations.

Factors important to consider when addressing forests vulnerability towards natural disturbances:

- Forest edge length (long forest edges increase vulnerability towards storms)
- Forest patch shape (sharp and exposed corners contribute to vulnerability towards storms)
- Forest core area size and forest composition (large core areas of diverse foresttype composition lower the risk of storm damages or insect harm)
- Adjacent landcover types neighbouring forest patches (wood and shrubland as well as grassland can facilitate fire spread)
- Clear-cut density (high clear-cut density increases the risk of fire spread and insect harm)

3.4 Habitat suitability and habitat modelling

Habitat suitability (habitat quality) is the ability of the environment to provide conditions appropriate for survival, reproduction and viability of a species. Suitability is a continuous variable measured by the rate of population increase (Patton 1997).

Models predicting the spatial distribution of species (Boyce and Mcdonald, 1999; Guisan and Zimmermann, 2000; Manly et al., 2002; Pearce and Boyce, 2006) – sometimes referred as resource selection function or habitat suitability models or habitat evaluation procedure – are currently gaining interest in wildlife management issues (Hirzel et al., 2006). As these models often help understanding species niche requirements and predicting species potential distribution, their use has been especially promoted for conservation issues, such as managing species distribution, assessing ecological impacts of various factors (e.g., pollution, climate change), risk of biological invasions or endangered species management (Scott et al., 2002; Guisan and Thuiller, 2005). Habitat suitability models are developed on the basis of a large variety of methods (multi-variate analyses logistic regression, Gaussian logistic regression, discriminant analysis, nearest neighbours technique, neural networks). For an overview of applied examples we recommend, among others, Manel et al. (1999), Fielding and Bell (1987), Mccullagh and Nelder (1989), Smith et al. (2007).

When absence data are lacking; analyses of this nature allow comparison, in the multi-dimensional space of ecological variables, of the distribution of the sites where the species of interest was observed to a reference set describing the whole study area (Hirzel et al., 2002). For instance, Store and Jokimäki (2003) applied Multi Criteria Evaluation (MCE) on the basis of habitat preferences of various animal species and Clark et al. (2002) used the same approach to study red squirrel populations. Bayliss et al. (2005) developed a multi--species targeting approach for eight threatened bird species in UK while Smeins and Wu (2000) performed a similar analysis to develop landscape scale models for assessing the potential and present habitat suitability of eight rare plant species found in southern Texas. For a more detailed description of MCE techniques see § 6.

Habitat quality of an isolated patch may sometimes be more important for species diversity than patch size (Turner et al. 2001). Suitable habitat can be identified by habitat modelling with focus on statistical or expert-based habitat models, often linked to GIS (Van Andel 2006). Habitat suitability index modelling consists of combining spatially explicit land cover data with quantitative knowledge about the requirements of specialized species and building spatially explicit maps describing the probability that a species is found in a landscape (Angelstam et al. 2003).

Kenter et al. (2003) modeled habitat suitability for selected key species based on remote sensing and field data, in connection with habitat suitability indices derived from species-specific habitat requirements. They demonstrated that habitat models are a useful approach to monitor changes in habitat suitability at the landscape level over time. Luque et al. (2004) tested a simple habitat quality model which was applied to forest spatial pattern information derived from data of the Finnish national forest inventory. The proportion of high quality habitats was calculated based on criteria such as dominant tree species, forest stand age, volume of deciduous species within a stand and distance from roads and agricultural or populated areas. They showed that such an approach together with additional parameters on habitat requirements can be applied to compile habitat quality maps on the country level. Van Rooij et al. (2003) studied viability of metapopulations of representative species for three ecosystem types in a fragmented environment (woodland, wetland, grassland) by applying a landscape-ecological model. The model requires input in the form of habitat data (e.g. a vegetation or land use map) and ecological parameters (e.g. home range, dispersal distance, and carrying capacity for all habitat types). The results of the model present potential species distributions and habitat networks based on habitat suitability (optimal, sub-optimal, marginal). Such analyses help to design ecological networks or to identify gaps in existing networks. Holzkämper et al. (2006) investigated the effects of land use changes on habitat suitability for different species. Based on landscape pattern in connection with species presence and (pseudo-)absence data they applied a spatial optimization model which aims at maximizing habitat suitability for the focal species. Hirzel et al. (2002) propose a multivariate approach to study species distribution which does not require species absence data. The approach integrates multidimensional ecological variables at localities where the focal species was observed (*ecological niches*) to derive habitat-suitability maps for the whole study area. This method is useful in situations where species absence data are not available, unreliable, or meaningless (in case of invaders).

Analysis of habitat suitability at a European scale requires data on habitat preferences for a wide range of species. Within the BioScore project running under the EU Commission's Sixth Framework Programme (http://www.ecnc.nl/Bioscore/) a European database with ecological preferences of individual species has been compiled. Objective of BioScore is to develop a cost-effective tool that allows for monitoring and assessment of the impacts of key drivers and pressures from Community policies on biodiversity (species). The species-sensitivity database offers the possibility to interpret European-wide forest spatial pattern maps with regard to forest ecosystem vulnerability and critical fragmentation thresholds.

Ecological stability takes on any connotation in a continuum ranging from resilience (returning quickly to a previous state) to constancy (lack of change) to persistence (simply not going extinct). The precise definition depends on the ecosystem in question, the variable or variables of interest, and the overall context. In the context of conservation ecology, stable populations are often defined as ones that do not go extinct.

The “stability” of a community is thus characterised in one of the following ways:

A) Stable: a system is stable just in case all the variables return to their initial equilibrium values following a perturbation.

B) Resilience: how fast the variables return to their equilibrium following a perturbation.

C) Persistence: how long the value of a variable lasts before it is changes to a new value.

D) Resistance: the degree to which a variable is changed following a perturbation.

E) Variability: the degree to which a variable varies over time.

Species vary widely in their responses to changes in forest spatial patterns, and relying on species-by-species analyses of responses is complex and difficult to manage. Species can thus be categorized based on their response to spatial pattern changes. This way the effect of various spatial pattern changes on species can be anticipated. Habitat is a physical space that provides essential resources for a species. Shifts in patterns of available habitat result from natural processes, such as succession, fire, and drought, as well as from human-induced changes in land use. Depending on the scale of the changes in spatial pattern of habitat, there can be important consequences for a particular population of organisms, a species, or for several species. Analysing habitat for a certain species in a landscape requires using a patch type classification appropriate to that species. The appropriate classification might be traditional forest cover types defined by the dominant trees, forest age, forest cover type, soil characteristics or suitable breeding pools.

These characteristics are the most important determinants of sensitivity to change in spatial patterns: Natural abundance; Dispersal type; Interactions with other species; and Habitat specificity.

Species can generally be assigned to one of four types based on the habitat used:

- Generalist species: forest types, residential area, parks, thicket, tree cover, shrubs
- Interior Forest
- Multiple Subhabitats: swamps, marshes, ponds, creeks, rivers
- Habitat Specialist

The type of habitat a species uses, such as whether it is a habitat generalist or specialist, can be a primary influence in how forest spatial pattern affects that species’ persistence. Thus, analyses of habitat preferences for species groups (e.g. small mammals, large mammals, birds) are important when addressing forest ecosystem vulnerability, resistance and resilience. Sufficient amount of suitable habitat for key species provides ecological stability. This is directly related to the concept of umbrella species which is introduced in the next section.

Key variables to evaluate habitat suitability can be summarised as follows:

- preferred habitat (land use type / forest type)
- required minimum habitat size (forest patch size)
- required distance from disturbance (such as roads or edge)
- elevation

- bio-geographical region (climatic characteristics)

3.5 The concept of umbrella species

To prevent the further loss of species it is necessary to determine the composition, quantity, and configuration of landscape elements required to meet the needs of the species present. Lambeck (1997) developed an approach based on the use of focal or umbrella species, rather than on a single species, to define the attributes required to meet the needs of the biota in a landscape. The approach builds on the concept of umbrella species, whose requirements are believed to encapsulate the needs of other species. An umbrella species can be defined as a species whose conservation is expected to confer protection to a large number of naturally co-occurring species (Roberge and Angelstam, 2004). This implies that the species most sensitive to a certain threat is used to define the minimum acceptable level at which that threat can occur. In terms of habitat fragmentation the area requirements of the species most limited by the availability of particular habitats will define the minimum suitable area of those habitat types and the requirements of the most dispersal-limited species will define the attributes of connecting vegetation (Lambeck 1997). Beier and Loe 1992 indicate that an ideal umbrella species should be a species which makes use of habitat corridors. This will ensure that conservation efforts will consider both minimum area requirements and landscape connectivity (Beier and Loe 1992, Beier 1993). However, Roberge and Angelstam (2004) concluded in a review that single-species umbrellas cannot ensure the conservation of all co-occurring species because some species are inevitably limited by ecological factors that are not relevant to the umbrella species. Moreover, umbrella species from a given higher taxon may not necessarily confer protection to assemblages from other taxa. Therefore Roberge and Angelstam (2004) suggest a multi-species strategy based on systematic selection procedures including measures of population viability and data from many years.

Among the species suggested as potential umbrellas, most are large mammals and birds, but invertebrates are increasingly being considered (Roberge and Angelstam 2004). Carignan and Villard (2002) indicate that the use of invertebrates and plants as indicators must be considered with caution because they mainly react to disturbances at fine spatial scales and hence, would potentially be inadequate indicators for organisms that mainly react to larger-scale. Birds may offer a bridge between these two groups because they have been shown to respond to environmental changes over many spatial scales and are well suited for monitoring. Woodpeckers are often chosen as an umbrella for several birds and other species. Mikusinski et al. (2001) showed that the presence of the three-toed woodpecker and white-backed woodpecker is associated with high species richness of other forest birds. Martikainen et al. (1998) found that white-backed woodpecker also serves as an umbrella species for threatened saproxylic beetles in Finland, and that protecting woodpecker affords similar protection to the beetles. Both woodpecker species have potential for being used as focal species for the planning of networks of old-growth forest (Angelstam et al. 2003). Van der Sluis et al. (2003) present a long list of birds and mammals which are useful indicators to study functional connectivity, among them e.g. green woodpecker, great spotted woodpecker, red-backed shrike, dormouse, pine marten, brown bear, wolf and roe deer. Van Rooij et al. (2003) selected red-backed shrike (bird species) and dormouse (small rodent), both sensitive to fragmentation, as focal species for woodland to define requirements of an improved ecological network in Italy. Bona et al. (2006) used deer and roe deer as focal species when analysing connectivity as they are relatively common, represent important game species and may be associated with the concept of landscape species because of their requirements for large areas and a heterogeneous landscape. Maintaining corridors for deer also benefits large predators such as wolf and lynx. Martens have shown to be very vulnerable to forest fragmentation (Hargis et al. 1999) and can be considered as a useful indicator species for fragmentation impact analyses. For example, Linehan et al. (1995) chose fishers (a marten species) as an indicator in a study on wildlife corridors as their protection will buffer the effects of fragmentation upon other species.

It is important to consider different ecological profiles when analysing forest spatial pattern regarding their effects on biodiversity. For example within the EU-FP6 Integrated Project SENSOR ecological profiles of species with differing sensitivity to habitat fragmentation are applied to identify habitat network requirements related to spatial connectivity. The selected eco-profiles for the forest ecosystem vary in their dispersal capacity (10-15 km, 35-50 km), sensitivity for landscape permeability (barrier sensitive and non-sensitive) and minimum area requirements (key patch sizes of 5-10 km^2 and 50-150 km^2).

There is still a need to select relevant umbrella / focal species and ecological profiles for which critical thresholds in terms of forest fragmentation can be defined. It is not possible to generalise effects of habitat fragmentation in order to derive generic critical thresholds because responses of organisms are very much species-specific. Different habitat requirements of edge and interior species need to be considered in the selection process. European databases on species habitat preferences such as the BioScore species-sensitivity database may be a valuable tool for the selection of relevant focal species and the definition of critical thresholds.

Based on availability of information the following umbrella species which are distributed over most of the biogeographical regions in Europe have been chosen for the spatial pattern analyses of the project (listed thresholds are based on the literature review presented in sections 3.1.4 to 3.1.6.):

Species group	**Umbrella species**	**Habitat**	**Min. habitat size**	**Dispersal distance**
Small mammal	Marten	Forest	259 ha	60 km (home range ~4 km^2)
Large mammal	Roe deer	Forest, open fields, grassland	160 ha	10 km (home range ~10 km^2)
Large mammal (area demanding)	Wolf	Forest, open fields, grassland	4000 ha core forest	160 km (home range ~35 km^2)
Birds	Lesser spotted woodpecker	Deciduous forest	40 ha (which can be fragmented over a maximum area of 200 ha)	20% suitable forest in 2 km^2
Butterflies	Lesser Purple Emperor	Forest	64 ha	500 m (assumption)

Table 2: umbrella species selected for the project.

4 Forest spatial pattern analyses

4.1 Review of landscape metrics and indices

Landscape ecology is based on the notion that environmental patterns strongly influence ecological processes (Turner 1989). The habitats in which organisms live are spatially structured at a number of scales, and these patterns interact with organism perception and behaviour to drive the higher level processes of population dynamics and community structure (Johnson et al. 1992). A disruption in landscape patterns may therefore compromise this structure's functional integrity by interfering with critical ecological processes necessary for population persistence and the maintenance of biodiversity and ecosystem health (With 1999). Consequently, there is a necessity to develop methods to quantify landscape patterns to better understand the general relationships between changing patterns and processes. Since pattern affects process, indices of landscape pattern correlates with ecological processes providing a means to detect and monitor ecological changes (Rutledge 2003).

Landscape ecologists have developed different metrics to quantify landscape patterns and to better understand the relationship between pattern and process. The common usage of the term landscape metrics refers exclusively to indices developed for categorical maps. Landscape metrics are focused on the characterization of the geometric and spatial properties of categorical map patterns represented at a single scale (grain and extent).

Many different landscape indices exist in literature (Betts 2000; Rutledge 2003; McGarigal & Marks 1995; Oehmichen & Köhl 2006) with no single index capable to satisfactorily describe landscape pattern and composition (Betts 2000). This section summarizes the landscape metrics and indices most often used in landscape pattern analyses. Betts 2000 categorised landscape metrics into five major groups based on the effect of landscape fragmentation: habitat area/ landscape composition; patch size metric; edge metric; shape metric; and landscape configuration metric. A review of studies on the spatial requirements of species from a range of taxa (birds, mammals, amphibians, plants, and insects) carried out by Betts (2000) reveals that 'configuration' (55.5%), 'patch size' (39%), and 'total proportion of suitable habitat' (30.5%) are the most frequently cited landscape factors explaining distribution, movement and reproductive success.

4.1.1 Habitat area and landscape composition metrics

The total landscape area defines the extent of the landscape and is used for the computation of many class and landscape metrics. The area of each patch comprising a landscape mosaic is the most important and useful piece of information contained in a landscape (area metrics). Class area is indicative of the percentage of the landscape covered by a particular patch class.

Composition indices describe the basic characteristics of fragmentation. The two basic indices used to quantify fragmentation are number of patches and patch area, usually measured as mean patch area. Suitable species habitat is not always present in a single contiguous patch. Therefore, measuring the area of different habitat types is thus an effective method for determining landscape change (fragmentation). Gustafson (1998) described landscape composition by (a) the number of categories or classes in a map and the area associated with each; (b) the proportion of each class relative to the entire map; and (c) diversity.

Diversity measures are influenced by two components: richness and evenness (McGarigal and Marks 1995). Richness refers to the number of patch types present while evenness refers to the distribution of area among different types. Many indices have been developed to measure landscape diversity, most of which are based upon measures previously used to determine species diversity (Shannon's diversity index and Simpson's diversity index).

Shannon's Diversity Index: Shannon's diversity index is only available at the landscape level and is a relative measure of patch diversity, or the proportional abundance of each patch type within the landscape. The index will equal zero when there is only one patch in the landscape and increases as the number of patch types or proportional distribution of patch types increases (McGarigal & Marks 1995). Shannon’s diversity index reflects differences in patch richness.

Shannon's Evenness Index: Shannon’s evenness index quantifies evenness among landscapes. It is a measure of patch distribution and abundance, or the measurement of the distribution of area among patch types within the landscape. Shannon's evenness index is equal to zero when the observed patch distribution is low and approaches one when the distribution of patch types becomes more even. Shannon's evenness index is only available at the landscape level.

Diversity measures while useful for monitoring landscape change within a region over time or among regions are not relevant in isolation from other metrics. Variable patch type diversity may be the result of either human induced or natural factors (Betts 2000). Furthermore, diversity measures convey no information about the actual composition of a landscape (McGarigal & Marks 1995). According to Betts (2000) a landscape may have high diversity, but be characterized by many non-indigenous patch types (e.g. urban area, non-native tree species plantations).

4.1.2 Patch size metrics

Patch size is the simplest and most intuitive metric to determine especially if digital maps with patch categories are available. Rutledge (2003) reviewed the following indices related to patch size:

Number of patches index: This index measures the number of patches of a particular class. It depends on patch definition and data resolution.

Mean patch size index: The mean patch size is often used to determine a particular patch class (type) size within landscapes. This can be easily obtained if digital maps broken into patch categories are available.

Generally, the number of patch and the mean patch size indices provide an incomplete picture because the fragmentation concept also encompasses the relative sizes of the pieces that result (Rutledge 2003). Furthermore, the mean patch size is sensitive to the addition or deletion of small patches and can be problematic in landscapes were patches are not easily identifiable (Gustafson 1998).

Largest patch index: This index measures the largest patch of a given class as a percentage of the total landscape (With & King 1999, Saura & Martinez-Millan 2001).

Patch density index: The number of patches of a particular class per unit area.

Average patch carrying capacity: scales patch size based on a species’ area requirements (Vos et al. 2001).

Splitting index: It is a measure of the number of equal-sized patches of a particular class required to produce a desired degree of landscape division (Jaeger 2000).

Effective mesh size index: This index is a measure of the size of equal-area patch of a particular class required to produce a desired degree of landscape division (Jaeger 2000). Oehmichen & Köhl (2006) found the Effective mesh size as the most appropriate measure of forest fragmentation for two German test areas when evaluating several indicators.

Several of these factors obviously will bring similar messages and hence for an efficient analysis one would want to work as much as possible with complementary indicators. Neels et al. (2004) showed that Area-weighted Mean Patch Area Distribution and Effective Mesh Size are largely redundant and that they would not be used together, while however each can have utility due to their different interpretations and units. However Bogaert et al. (2002) reviewed that the use of statistical methods such as factor analysis to reduce the number of indices [Riiters et al., 1995; Cain et al., 1997; Bogaert et al., 1999; Herzog et al., 2001] does not render the ecological meaning of a metric to the analyst [Riiters et al., 1995]. Bogaert et al. (2002 concluded that generally, one should attempt to describe independent and fundamental components of a spatial pattern by utilizing a suite of metrics [Li and Reynolds, 1994; Riiters et al., 1995; Giles and Trani, 1999].

4.1.3 Edge metrics

Edge effect is highly variable and dependent upon habitat type, the composition of adjacent patches and the species under examination (Betts 2000). The following edge related indices are summarised in literature:

Total edge: The sum of all patch perimeters within a landscape (landscape level) or the sum of perimeters of a particular class (class level), in map units. It is directly affected by the landscape size. At the class level, high total edge implies that the class is made up of small or convoluted patches. At the landscape level, total amount of edge is directly related to the degree of spatial heterogeneity in that landscape (McGarigal & Marks 1995).

Perimeter/Area Ratio: The perimeter of a patch divided by its area. Perimeter/Area ratio is a very useful measure of fragmentation; it is a measure of the amount of 'edge' for a landscape or class. For a landscape, the sum of all perimeters divided by the total area. For classes, the sums for each patch type are used.

Edge density: It is a measure of landscape configuration and determines the amount of edge relative to the landscape area. It is equivalent to Perimeter/Area ratio, used in all datasets. Units are provided in metres/hectare for metric grids. Edge density is therefore a standardised metric useful for comparing landscapes of varying size. Laurence and Yensen (1991) established three steps to measure edge effect for a patch type in a given landscape: (i) identify species most sensitive to edge effect; (ii) determine edge function; and (iii) use a core area model to estimate the impact of edge on remaining habitat.

Core Area and Core Area Index: These are two ecological indices (McGarigal & Marks 1995). Core area is a simple measurement of area while core area index is a ratio of core area to patch area. Core area is defined as the total available habitat minus the edge effect (Betts 2000). Core areas indicate interior areas of a patch, which retain similar abiotic and biotic conditions to pre-fragmented conditions and do not experience strong influences from neighbouring patches (Rutledge 2003).

4.1.4 Patch shape metrics

Patch shape metrics attempt to quantify patch complexity, which can be important for different ecological processes (Forman 1998). Numerous metrics exists for evaluating patch shapes but this landscape feature remains the most difficult to measure effectively. Measuring patch shape has proven to be problematic in that no single measurement or index of shape can unambiguously differentiate all shapes (Forman 1998).

Mean Shape Index: The shape index is a measure of shape complexity. It represents the variation of a patch from a circle (Baskent & Jordan 1995). Mean shape index is equal to the sum of each patch's perimeter divided by the square root of patch area (hectares) for each class (class level) or all patches (landscape level), and adjusted against a square standard, then divided by the number of patches (McGarigal & Marks 1995).

Area Weighted Mean Shape Index: This is the average perimeter-to-area ratio for a class, weighted by the size of its patches.

4.1.5 Landscape configuration metrics

Landscape configuration metrics generally attempt to reflect the degree to which patches are isolated or connected across landscapes. Rutledge (2003) divides patch configuration indices into two categories: indices based on distances between patches and indices that compare the overall spatial pattern.

Distance Based Indices

Indices of patch configuration based on distance between patches vary in the degree to which they consider all other patches relative to a focal patch of interest.

Nearest neighbour statistics: This is the most commonly cited distance configuration metric (McGarigal and Marks 1995, Baskent & Jordan 1995, Hargis et al. 1998). It is the shortest, typically edge-to-edge, distance between a patch and the neighbouring patch of the same class. Mean nearest neighbour summarises the distance between each patch of a certain cover type (Betts 2000). This metric can be used along with information on the movement capabilities of various species to determine whether, on average, the distance between patches is close enough for organisms to interact (colonise, disperse, mate, migrate etc.). Nearest neighbour statistics provide an incomplete picture of the spatial realities of the landscape because very different landscapes can have the same mean nearest neighbour value (McGarigal & Marks 1995).

Mean Proximity Index: This metric measures the degree of patch isolation and fragmentation. Proximity indices as well as buffer indices measure configuration as the sum of the area of a similar patch class within a given distance (Hargis et al. 1998). Mean proximity index uses the nearest neighbour statistic.

Connectivity Index: This index measures the influence of all patches within the landscape as a function of their area and distance from the focal patch.

Pattern Based Indices

Pattern based indices of configuration attempt to provide a measure of the overall complexity of a landscape. Unlike distance measures they do not have a patch focus and are calculated for the entire landscape. The most widely used and often cited is contagion (Riitters et al. 1996; Hargis et al. 1998; McGarigal & Marks 1995).

Contagion Index: This index measures the degree of adjacency or “clumpiness” of a map based on adjacency of patches (Rutledge 2003). It measures the degree of adjacency between cells on a raster landscape.

Interspersion Juxtaposition Index: It is a measure of patch adjacency rather than pixels (McGarigal and Marks 1995). Interspersion juxtaposition index measures how intermixed patch types are within a landscape and is related to configuration of patches. It is calculated in percentage units and approaches 100% when all classes are equally adjacent to all other classes, and approaches zero when patch adjacency becomes uneven. Interspersion requires that the landscape be made up of a minimum of three classes. At the class level interspersion is a measure of relative interspersion of each class. At the landscape level it is a measure of the interspersion of each patch in the landscape.

Contrast Index: Contrast refers to the the magnitude of difference between adjacent patch types with respect to one or more ecological attributes at a given scale that are relevant to the organism or process under consideration. The contrast between a patch and its neighbourhood can influence important ecological processes (Forman & Godron 1986). Edge effects for example, are influenced by the degree of contrast between patches. Microclimatic changes are likely to extend father into a patch along an edge with high structural contrast than along an edge with low structural contrast (Ranney et al. 1981).

Patch per Unit Area Index: A measure of the degree of aggregation or connectedness of patches (Frohn 1998). It correlates well with dispersal success under a variety of conditions.

Patch cohesion: It was proposed by Schumaker (1996) to quantify the connectivity of habitat as perceived by organisms dispersing in binary landscapes. Patch cohesion is computed from the information contained in patch area and perimeter. Briefly, it is proportional to the area-weighted mean perimeter-area ratio divided by the area-weighted mean patch shape index (i.e., standardized perimeter-area ratio).

Aggregation index: It measures the degree of aggregation of a particular patch class on the landscape by comparing the number of shared edges with the total possible number of shared edges (He et al. 2000).

Degree of division index: This index is defined as the probability that two randomly selected locations do not occur within the same patch in the landscape (Jaeger 2000).

Lacunarity index: It measures the degree of gaps between features of interest on a map (Plotnick et al 1993). This is done by sliding a window of fixed size across a landscape and counting the number of cells of interest within the box.

4.2 Tools for spatial pattern analyses

Sustainable management of forests requires a harmonious and coordinated management of various resources taken into consideration the sustainable forest management paradigm. This necessitates a quantification of the variations in a forest. The incorporation of landscape ecological and fragmentation analyses within remote sensing science has expanded the inferential capabilities of such research. Griffiths & Lee (2000) demonstrated that it is difficult to ecologically interpret landscape structure based on plant diversity models alone. They highlighted the need to obtain data on both landscape quality and landscape structure. In order to effectively model spatial patterns, Schaffer & Leigh (1976) stressed the need of concise and interpretable description of patterns. Levin (1992) further stated the importance of resolution (fineness of spatial scale) in determining the spatial patterns. To better understand ecosystems across scales it is necessary to identify critical structuring processes and associated spatial and temporal scales (Risser 1995). Intense empirical investigation is needed to learn more about the complexities of spatial pattern

The number of environmental variables that can influence species distributions is potentially infinite (Ricotta et al. 2003). The relationships between landscape structure and species diversity distribution have often been analysed using field data coupled with statistical models: simple correlation and stepwise regression analyses (Balent & Courtiade 1992; Atauri & deLucio 2001; Telleria & Santos 1999; Pausas 2006; Pascher & King 2006). Nevertheless, a number of studies have demonstrated the importance of land cover as a major determinant of type and number (Harner & Harper, 1976; Tonn & Magnuson 1982). Furthermore, to satisfy requirements for comparability of data and indicators over large geographic regions, the input data for assessing landscape patterns are typically land cover maps derived from remote sensing data. The most widely used spatial pattern analysis program for quantifying landscape structure is FRAGSTATS (McGarigal & Marks 1995). FRAGSTATS offers a comprehensive choice of landscape metrics used to quantify the aerial extent and spatial distribution of patches (i.e. polygons on a map coverage) within a landscape.

In order to determine the distribution of diversity within a grid-based land cover map, some diversity indices need to be spatially referenced by calculating the index within a moving window that is passed across the map. Following this convolution operation, the computed index value at each point is displayed, allowing visualization of the spatial variability of the index across the analyzed land cover map (Gustafson 1998). The window defines the sample size and spatial extent of information that may be used in three stages - 1) computing features in a pre-classification stage, 2) evaluating neighborhood influences during classification, and 3) reclassifying pixels in a post-classification stage. The characteristics of the window used in any of these three stages are its size, shape, repetitions of application, and the dynamic nature of the size/shape characteristics in its use. Such windows are routinely used for raw spectral data, such as edge-enhancement, low and high pass filters, or texture measurements. Post-classification studies also use windows on nominal data.

The selection of window size is important not only for computation reasons, but most importantly for defining the relevant sample domain. Because shape, size and orientation of the window is normally fixed the subject of optimal window size as it relates to texture analysis and image classification has been an issue in many research papers. Gurney (1981) suggested that the selection of the size of a window is often based on either a preliminary test of classifier accuracy or on results from a different date or similar study area. Chavez & Bauer (1982) proposed that the ideal window sizes should be selected based on the spectral change between adjacent pixels. Franklin et al. (1996) suggested a technique for determining optimal window sizes based on semivariograms. Hsu (1978) noted the negative effects of using large windows (e.g. edge effects) and argued for small windows, such as a 3*3 or 5*5. However, Merchant (1984a) pointed out that too small a window, constrains adequate contextual analysis and recommended a window size that approximates the field-of-view of the visual image analyst.

The window size is generally assumed to be a static geometric window yet certain authors have advocated for various shapes, sizes, and even dynamic windows. Merchant (1984a and 1984b) first suggested the concept of a dynamic geographic window that changes size and shape to fit the application. Analogous to an n*n geometric window that includes the centred pixel and its neighbours, the geographic window includes the "field" (or patch) of interest and the neighbouring fields (or patches) of interest (Köhl & Oehmichen 2003). Hodgson (1991) and Dillworth (1991) also argued for a dynamic window size rather than a fixed size window. Hodgson (1991) demonstrated how multiple windows of a variety of shapes and sizes could be used simultaneously to build evidence for characterizing the homogeneity of a landscape. Dillworth (1991) also argued that no one geometric window size provides the best results for any image and suggested an adaptable window that dynamically changes for a given region. Such dynamic windows have been used on the classified data but are not known to have been used on spectral data.

Based on the basic unit of the image analysis two major approaches can be discerned: an adaptive geographic window and a traditional rectangular window. The adaptive geographic window proposed by Merchant (1984a) operates on neighbourhoods of patches (objects) instead of neighbourhoods of pixels. The traditional rectangular window performs well on data where the pixel is the basic unit of classification. Generally, pixel level classification permits mapping and monitoring of spatial patterns at the pixel level which provides a greater sensitivity to pattern changes over time. These methods are based on image convolution and do not require the identification of individual patches. A fixed area window, or kernel, is centred over each pixel on a land cover map and an index is calculated according to the amount and adjacency of a particular class in the window. This result is then assigned to the land cover class pixel located at the window centre, thus building a new map of the fragmentation index values.

The main drawback of the pixel based approach as pointed out by Merchant (1984a) and Dillworth et al. (1994) is the fact that it is considered unsuitable for characterising the spatial structure of classified data where attention is shifted from single pixels to land cover objects (patches). An example of a software based on the use of Merchant's adaptive geographic window is LaDy (Landscape Diversity Software) presented by Ricotta et al. (2003). The software is designed to operate on a neighbourhood of patches instead of a fixed rectangular neighbourhood of pixels (the conventional approach in image analysis). LaDy can be used to compute local landscape diversity profile on raster land cover maps. Such an adaptive window approach as applied in LaDy, is indeed less prone to the intrinsic problem of a convolution approach, but it does not solve this problem. The adequately analyze of the spatial structure of land cover objects in the entire image would require a window size which encompasses the entire map. This however is useless because it would provide only one value for the entire image and as such can not describe the spatial features of the image components (land cover patches).

An alternate approach for reliable pixel-level classification and mapping of land cover patterns is the morphological image processing (Soille 2003). Morphological filters are independent of the window size problem and are, most probably the only way to fully include the information of the neighborhood in the entire image, and still provide pixel level information. It is not a convolution approach and it considers the neighborhood of pixel and of objects. It is a technique based on a set theory for analysing the shape and form of objects. Vogt et al. (2006a) used the morphological image processing for classifying spatial patterns at the pixel level on binary land-cover maps. The algorithm to classify forest patterns is defined by a sequence of logical operations such as union, intersection, complementation, and translation using geometric objects called 'structuring elements' (SE) of pre-defined shape and size. The fundamental morphological operations are called erosion and dilation. The erosion operator shrinks regions of forest while the dilation operator expands them. Four classes of forest pattern were considered: (i) "core forest" is relatively far from the forest-non forest boundary; (ii) "patch forest" comprises coherent forest regions that are too small to contain core forests; (iii) "perforation" within a forest patch, defines the boundary between core forest and relatively small perforations and (iv) "edge forest" includes interior boundaries with relatively large perforations as well as the exterior boundaries of core forest regions. The four land-cover pattern categories 'perforated,' 'edge,' 'patch,' and 'core' were classified with higher spatial precision and thematic accuracy compared to a previous approach based on image convolution, while retaining the capability to label these features at the pixel level for any scale of observation. The description of the 4 classes refers to the initial definition of Vogt et al 2006a. This definition was revised and diversified to include connectivity by mapping connectors (Vogt et al, 2006b), then by identifying all perforations of core, not only the small ones, by discriminating between external type of core borders (edges, branch, bridge type of connector) and internal core borders (perforations, internal branch, shortcut type of connector.

At the Joint Research Centre 5 indices are available which are built on the basis of the pattern maps and calculated over reporting units of interest: quadratic average core area index to capture loss of area, sample index to capture loss of units per units size interval, external core border index to capture edge related issue, internal core border to capture perforations, connectivity index to capture connectors and core-connectors processes (Estreguil et al, submitted).

Evaluating indicators of connectivity are important aspects for biological conservation and biodiversity assessment. Corridors are strips of land differing from adjacent land on both sides (Metzger & Décamps 1997) and indicate structural connectivity between landscape patches. To automatically map corridors in northern Slovakia, Vogt et al. (2006b) used the morphological image processing (Soille 2003; Vogt et al. 2006a). They used forest maps derived from satellite imagery and applied the concept of skeletonization, a process which iteratively removes the boundary pixels of a region to its line representation. Corridors were mapped at multiple scales of observation. The approach successfully differentiated between relatively narrow ("line") and wide ("strip") structural corridors.

4.3 Spatial pattern analyses of forest composition

Plant variety provides a variety of food and water as well as different kinds of structural cover essential for the survival of wildlife. The size of a stand and its location relative to other stands, can determine whether wildlife habitat needs are met within the home range. Without variations in plant species and age classes many wildlife species, if not all, would find it hard to survive. Consequently, a diverse forest, with many different stands, is more able to meet the varying needs of many wildlife species than a uniform monoculture forest (Das and Nautiyal 2004).

In a forest, spatial heterogeneity is a universal feature and the spatial distribution of similar age classes or plant species is seldom random; they often exist as clusters or patches (Zahl 1974 cited by Das & Naitiyal 2004). A forest stand can be viewed as an aggregation of clusters or patches of trees without any loss of generality (Das & Nautiyal 2004). It is thus difficult to describe such patches-or non-random distribution of trees, in accurate terms. Many environmental conditions as well as demographic processes of plants within a community are reflected by non-random patterns (Das & Nautiyal 2004). Pattern as defined by Pielou (1965) cited in Das & Nautiyal (2004) is the spatial arrangement of plants on the ground or the arrangement of the members of one species relative to those of other species. Landscape patterns have two basic components: intensity and grain (Pielou 1977 cited in Das & Nautiyal 2004). Intensity is defined as the extent to which density of species changes from place to place while grain is the scale of patch size within a community. Grain reflects the important biological and environmental factors of spatial pattern. To develop a better understanding of patch distribution in a forest, it is of utmost importance to take a closer look at various species diversity indices and compactness indices. Whittaker (1977) distinguished four levels of diversity: (i) diversity of a microhabitat or a homogeneous habitat as point diversity or within-habitat diversity that is commonly known as alpha (α) diversity; (ii) the diversity at the landscape level or the gamma (γ) diversity; (iii) the "differentiation diversity" or the beta (β) diversity; and (iv) the total diversity of a group of areas of gamma diversity – the epsilon (ε) diversity. Beta diversity is most widely used particularly for forest landscapes. Compactness on the other hand refers to the closeness of patches in a forest to a circle (Das & Nautiyal 2004).

To analyse forests vulnerability, resistance and resilience it is important to study the patterns of forest composition. Heterogeneous forests with a mixture of different forest types can provide habitat for more key species than mono-cultural plantation forests. Moreover heterogeneous forests are more resistant towards natural disturbance regimes.

4.4 Review of concluded studies in different bio-geographical regions

All landscapes are characterised by degrees of heterogeneity (patchiness) at different scales. Differing substrates (soils, bedrocks), natural disturbances (fires, insect outbreaks), and human activity (forestry, road building) all create patchiness across a landscape (Betts 2000). Many research studies have been carried out to quantify landscape fragmentation and to understand the effect of pattern on process in different biogeographical regions. A review of landscape fragmentation studies indicates a large number of studies in the Mediterranean, boreal, continental and alpine regions; while very few studies have been documented in the Atlantic area.

4.4.1 Boreal region

The course of succession in Boreal forests is controlled by spatial dynamic processes: seed dispersal and selective foraging by mammalian herbivores. Knowledge of the links between disturbance process and resulting pattern will aid in directing forest management practitioners towards creating an ecologically sustainable environment (Wilson & Howard 2002).

The maintenance of habitat heterogeneity on a small scale is needed to preserve biodiversity in managed boreal forests. The main problem when adapting forest management to mimic natural processes and structures is the lack of detailed knowledge about historical forest conditions (Axelsson 2001). Using historical records Linder & Östlund (1998) analysed the changes in the structure and composition of boreal forests in Sweden. They demonstrated that fundamental changes due to commercial exploitation, intensive forest management and fire protection have reduced the number of habitats for many red-listed species considerably. They emphasised the need to restore and maintain natural biodiversity by re-creating the essential characteristics of the natural forest landscape. Axelson & Östlund (2001) performed a retrospective regional gap analysis in the middle boreal zone of Sweden using historical data. Forest surveys and cadastral maps from the 19th and 20th century were compared with a recent forest survey in a coniferous forest landscape. Changes in species composition, age distribution and landscape pattern were analysed at different spatial scales using a Geographic Information System (GIS). Spatio-temporal analyses were made to compare representative forest types. Clear-cutting has replaced fire as the most important factor influencing the landscape pattern.

Although habitat loss and fragmentation are widely regarded as major factors contributing to the decline of many populations, the relative importance of each phenomenon is seldom evaluated (Fiona et al. 1997). Fiona et al (2002) constructed simple empirical models of benchmark communities in boreal forests of Finland and Canada based on species composition, species abundance distribution, and habitat requirements, in order to identify features of bird species sensitive to the loss of older forests. They concluded that most responses may be attributed to pure habitat loss in landscapes where forest harvesting is the dominant land use practice. In these dynamic landscapes, total forest cover may not change, and predicting patterns of species decline requires identification of the habitats and species of concern.

Pasher & King (2006) used a generalized linear model to analyse relations between damage and fragmentation metrics representing patch isolation, edge density, and the relative size and distribution of patches in the landscape. The metrics were applied using spatial extents of 1*1 km and 4*4 km, following analyses of the variability of numbers of patches and of the lacunarity of forest patterns over a range of extents. The results showed that patch isolation, as measured by the mean Euclidean distance between patches was significantly related to damage.

Using remote sensing, geographic information systems and statistical methods to analyse the relationship between forest patch and landscape physical attributes and land-use, Daiyuan et al (2001) concluded that the role of landscape physical attributes on forest patch pattern has been modified by land use. In addition, physical attributes explain only a small proportion of the abundance of conifers on past abandoned land compared with land-use factors. Physical attributes only indirectly influence the forest pattern because they strongly influence the land-use practices. Kurki et al. (2000) examined the breeding success of forest grouse in relation to anthropogenic forest fragmentation in Finland employing Geographic Information Systems (GIS) and grouse data derived from Finnish wildlife triangle censuses. The breeding success of grouse was negatively correlated with both fragmentation of forest area and the decreasing proportion of older forest as a result of clear-cutting. The diminished breeding success of forest grouse as a result of increased forest fragmentation is a probable cause of population declines in forest grouse species during the past decades.

Pastor et al. (1999) presented spatially explicit models to demonstrate seed dispersal patterns in boreal forests. The results indicate that spatially explicit seed dispersal results in more clumped distribution of tree species and persistence of greater paper birch biomass than uniform seed rain across the landscape. Such results are consistent with current spatially explicit population models of dispersal and coexistence.

It can be concluded that in the boreal region one of key disturbance factors regarding forest fragmentation is clear-cutting. Special attention needs to be paid to the density of clear-cuts when addressing forest vulnerability. Many interior species are negatively influenced by a high clear-cut density (compare chapter 3). Clear-cut sizes vary in Scandinavia between a few hectares and hundreds of hectares. Landcover maps available for the project, with resolution of 25 m and 100 m, will both be feasible to detect clear-cut densities. As cutting cycles are about 60-80 years it will not be possible to analyse influences from clear cutting in a temporal aspect. However, the temporal resolution of the available data sets (10-50 years) will reflect the recovery of the forest in the clear cut areas. Clear-cut areas are associated with the Corine land use class ‘324 transitional woodland-shrub’ in the spatial data available for the project.

4.4.2 Atlantic region

Significant changes are currently being made to the territorial mosaic and biological diversity of Western European landscapes in general and to the landscape of south-western France in particular, as a consequence of changes in agricultural policies (Balent & Courtiade 1992). In the South-western France, the evolution of agricultural practices occurs in two opposite ways: the intensification of agriculture tends to simplify the landscape by hedgerows removal, grasslands ploughing and drainage for corn cultivation on the one hand and on the other hand, the decreasing numbers of cattle and sheep conduct the less fertile parts of the territory to evolve into fallow. These two processes are closely linked and important interactions exist between intensive agricultural areas and semi-natural communities. Passerine bird communities are most often used as an ecological indicator to understand the importance of these interactions and their role in ecological stability of landscapes (Balent & Coutiade 1992; Lauga & Joachim 1992).

Balent & Courtiade (1992) used stepwise and multiple regression analyses to measure the importance of the changes induced on landscape by a range of management practices differing in intensity. They compared the displacement along ecological gradients between 1983 and 1988. The changes occurring both in bird composition and landscape structure revealed the ecological impacts of the different management practices (hedgerow removal, drainage, and ploughing, decreasing grazing pressure). Lauga & Joachim (1992) modelled the response of forest cover dependent breeding birds to forest fragmentation. The forest cover was quantitatively estimated from a Landsat MSS scene. The scene was first reduced to a grid of 5865 quadrats, each 650 by 650 m. Two values were attributed with each quadrat: Quadrat Forest Cover (QFC) expressed in percent; and a local measure of forest fragmentation- the Neighbouring Forest Cover (NFC) - expressed on a 0-1000 scale. The distribution of six forest breeding species was sampled on 556 quadrats. For each species, the local abundance appears to be more correlated with the fragmentation-oriented NFC value than with the local QFC value. For three species out of six (song thrush, robin, chaffinch) an incidence model, based on the Logistic regression, was built. A correct fit was obtained. An incidence map of these species was then built up over the whole study area. Their regional status was then estimated, for a sampling cost of less than 10% of censoring all the area.

To better understand landscape biodiversity Monteil et al. (2005) proposed the simple neural network. They studied the links between bird species richness and forest fragmentation in South-western France and found out that the neural network improved the prediction accuracy compared to linear, log linear and logistic models.

4.4.3 Continental region

Land uses, especially harvesting and road building, are considered to be the primary cause of forest fragmentation in many parts of the world (Breese et al. 2004; Tinker et al. 1998), and are also known to affect animal populations (Jaeger & Fahrig 2004). Anthropogenically induced fragmentation of landscapes and habitats are considered the most imminent threats to biodiversity in continental landscapes. To facilitate adaptive management it is thus essential to assess the links between management activities and ecological consequences and thereby facilitate adaptive management. Tinker et al. (1998) implemented remotely sensed data and a Geographic Information System (GIS) to compare the effects of clear-cutting and road-building on the landscape pattern of the Bighorn National Forest, in north-central Wyoming. They analyzed several landscape pattern metrics for the landscape as a whole and for the lodgepole pine and spruce/fir cover classes and determined the relative effects of clear-cutting and road building on the landscape pattern. At both the landscape- and cover class-scales, clear-cutting and road building resulted in increased fragmentation as represented by a distinct suite of landscape structural changes. Patch core area and mean patch size decreased, and edge density and patch density increased as a result of clear-cuts and roads. Clear-cuts and roads simplified patch shapes at the landscape scale, but increased the complexity of lodgepole pine patches. Roads appeared to be a more significant agent of change than clear-cuts, and roads which were more evenly distributed across the landscape had a greater effect on landscape pattern than did those which were densely clustered. Bresee et al. (2004) further tested the effect of harvesting and road building on forest fragmentation in Wisconsin. Initially, they quantified changes and the rates of change in vegetative composition and structure using Landsat images, examined changes in landscape structure. They finally assessed changes within the area of road influence and investigated changes in landscape composition and structure within the context of forest management activities. The landscape was classified into six dominant cover types: mixed hardwood, jack pine, red pine, mixed hardwood/conifer, non-forested bare ground, and regenerating forest or shrub.

Forest patches surrounded by agricultural fields, but also fields surrounded by forest are typical continental landscape patterns. Storch (unpublished) described the effect of landscape fragmentation on the dispersal behaviour, demography, population genetics and species diversity of small mammals of the Black Forest at the habitat and landscape level scales. The effect of landscape alteration, such as reforestation of fields on small mammals was predicted. A comparison of small mammal populations and communities between fragmented and unfragmented plots using a mark-and-recapture design (dispersal, demography, species diversity) and population genetic analysis with microsatellite markers (gene flow, genetic diversity) was implemented. The results contribute to the identification of threats from landscape fragmentation to the biodiversity in the Black Forest region and help to reduce such threats.

Roads generally affect animal populations in three adverse ways: (i) they act as barriers to movement, (ii) enhance mortality due to collisions with vehicles, and (iii) reduce the amount and quality of habitat. Putting fences along roads removes the problem of road mortality but increases the barrier effect. Jaeger & Fahrig 2004 used a stochastic, spatially explicit, individual-based model of population dynamics to study the effect of road fencing. They investigated the conditions under which fences reduce the impact of roads on population persistence and concluded that a fence may or may not reduce the effect of the road on population persistence, depending on the degree of road avoidance by the animal and the probability that an animal that enters the road is killed by a vehicle.

For the atlantic and continental region it can be concluded, that the most important factor regarding forest fragmentations is the development of infrastructure (road construction, urban spread). Special attention needs to be paid to the location of roads and minimum road-distance requirements by key species. In contrary to the boreal landscape, forests are more fragmented but also more diverse (heterogeneous forest type composition) in the atlantic and continental region. Key variables to study forest fragmentation are therefore - besides distance from road - forest patch size, forest composition, forest connectivity.

4.4.4 Alpine region

Fragmentation of natural habitats especially alpine habitats has become a major conservation concern as vulnerable species become rarer and the red list of endangered species becomes longer (Trocme 2005). Knowledge of stand history and understanding of potential ecological transformations are essential for the correct application of close-to-nature silvicultural practices in alpine ecosystems. The importance of multiple sources of independent data to characterize the disturbances that have affected the origin and development of stands heavily impacted by humans has been highlighted by Motta & Edouard (2005). Elena-Rossello et al. (1997) studied changing patterns of land use classes of the mountainous regions of Spain from 1950s through the 1980s. Historical black-and-white aerial photography was analyzed in each land class during that period. Forest landscape pattern analyses were carried out using spatial analysis, including number, mean and maximum size of patches; fractal dimension of patch perimeters; and indices of dominance and contagion. Although a general trend is detected in land-use changes all over Spain during the last 50 years, spatial pattern analysis shows differential trends at land class level. The inner and highland poor land classes have increased their forest areas, due to farm and extensive grazing abandonment, and reforestation programmes.

Motta & Edouard (2005) studied size, age, and spatial structures in a mixed, multilayered forest located in the Upper Susa Valley in Piedmont, Italy, using complete stem mapping, dendrochronology, and spatial analysis. They realised substantial shifts in forest structure and species composition over the last 200 years, from an open structure with scattered regeneration to a dense multilayered structure with dense regeneration. Shifts in dominance and structure were found to be consistent with land-use changes rather than with disturbance history. Kozak et al. (2006) studied the changes in forest cover and forest pattern in the Carpathians over the last decades using bi-temporal Landsat images. Single-date forest-non-forest maps were derived by image segmentation and supervised classification, including the use of ancillary data (CORINE Land Cover and a Digital Elevation Model). Post classification comparison was then implemented to detect changed areas. Forest spatial pattern maps with four classes (core, patch, edge and perforated forest) were derived with morphological image processing. The study concludes however that CLC data did not reveal changes in forest spatial pattern. Due to the minimum mapping unit size set to 25 ha, CLC did not allow to delineate forest spatial pattern classes other than core forest area and forest edges.

The analysis of complex interactions between spatial distribution patterns of site factors and vegetation types is crucial for understanding high mountain ecosystems, especially in the view of a changing climate (Hörsch 2003). The effects of global warming on the spatial pattern and species richness of alpine ecosystems have been documented (Camarero et al. 2006 and Dirnböck et al. 2003). Climatic warming is expected to have profound effects on the species richness and spatial distribution of forest patches on mountainous landscapes. Dirnböck et al. (2003) assessed the potential response of alpine plant species distribution in north-eastern Calcareous Alps of Austria to different future climatic and land-use scenarios. Site conditions were simulated spatially using a Geographic Information System (GIS), a Digital Terrain Model, meteorological data, existing maps, and historical records on pastures. Ordinal regression models of plant species based on environmental constraints were used to assess land-use impacts on vegetation patterns in combination with climatic changes. The results confirm the fact that alpine plant species on mountain ranges with restricted habitat availability above the treeline will experience severe fragmentation and habitat loss, but only if the mean annual temperature increases by 2°C or more. Camarero et al. (2000) described the spatial structure of two contrasting subalpine Pinus uncinata forest-alpine grassland ecotones located in the Central Pyrenees as a preliminary step to infer the processes that produced their spatial patterns. Their results pointed out potential different responses of treeline populations to environmental changes according to the spatial pattern. In an effort to determine the responses of alpine flora to the expected upward shift of treeline ecotones due to climatic warming in the Spanish Pyrenees, Camarero et al. (2006) investigated species richness patterns of vascular plants at small spatial scales across elevational transects using the point method and Moran's *I* correlograms. Boundaries based on plant richness and tree cover were delineated using moving split windows and wavelet analysis. Plant richness increased above the forest limit and was negatively related to tree cover in the undisturbed sites. Moving split windows and wavelets detected the sharpest changes in plant richness above the forest limit at undisturbed sites. Most tree cover and plant richness boundaries were not spatially related. The upslope decrease of tree cover may explain the increase of plant richness across alpine treeline ecotones. However, the detection of abrupt richness boundaries well above the forest limit indicates the importance of local environmental heterogeneity to explain the patterns of plant richness at smaller scales.

4.4.5 Mediterranean region

Mediterranean landscapes are highly heterogeneous, fine-grain landscapes in which a large number of patches of different land-use and natural vegetation coexist. This heterogeneity has been attributed to topographical and climatic variability as well as human influence (Atauri & de Lucio 2000). Heterogeneity in Mediterranean landscapes is an important criterion for landscape planning and for the definition of management directives in order to maintain biodiversity. Atauri & de Lucio 2000 studied the relationships between landscape structure and species diversity distribution (estimated in terms of richness of birds, amphibians, reptiles and butterflies) in the region of Madrid, Spain using simple correlation and stepwise regression analyses. The independent variables were the landscape structure descriptors, the coverage of each type of vegetation and land use, the synthetic variables of the composition of land use conglomerates, and altitude; while the dependent variables were species richness of the considered groups. Landscape heterogeneity proved to be the most important factor affecting the distribution of birds and lepidopterans species, while the distribution of amphibians is more closely related to the abundance of certain land-use types.

Oak forests of the Iberian Peninsula have been heavily affected by human disturbance for a long time so that interior forest birds are now scarce or extinct and species adapted to shrubby forests form the bulk of these bird communities (Telleria & Santos 1999). Telleria and Santos (1999) evaluated the role of ecological densities (densities in a given habitat) in predicting the ability of forest passerines to occupy fragments of eight oak (*Quercus)* spp. in forest archipelagos of the Iberian Plateaux. Ecological density of individual species was the main predictor of their occurrence in fragments, whereas other biological traits (nesting site) and some potentially important landscape features (local cover of forests or distance to possible sources of individuals) were not correlated to fragment occupation. Their results confirm the empirical usefulness of random sampling hypothesis in predicting the ability of species to persist in fragments.

Guirado et al. (2007) assessed landscape structure and dynamics from patch metrics and patch history. A set of landscape metrics related to potential human accessibility to forests was also calculated. The results of multiple linear regressions indicated that the variance explained for non-forest species groups was higher than for forest species richness. Most of the main correlates corresponded to site disturbance variables related to direct human alteration, or to landscape variables associated to indirect human effects on forests. Potential human accessibility also affected the richness of most species groups. In contrast, patch size, patch shape and connectivity played a minor role, as did patch history. They concluded that the spatial configuration and dynamics of periurban forest patches play a minor role in determining plant species richness and assemblage compared to site conditions.

Fire is a key mechanism creating and maintaining habitat heterogeneity in Mediterranean landscapes by turning continuous woody landscapes into mosaics of forests and shrublands. The effect of fire on landscape pattern and consequently landscape dynamic processes in the Mediterranean has been widely documented. Pausas (2006) used simulation models to study the effect of landscape pattern on landscape dynamic processes under different fire regimes. The results suggest that some species increase and others decrease depending on the fire regime. However, the results also show that different landscape structures produce different dynamics and thus that there is a clear interaction between landscape pattern and fire regime. Due to the long historical role of fires in the Mediterranean, a negative effect of this type of perturbation on forest bird distribution at a landscape level is often envisaged. Herrando and Brotons (2002) used a multi-scale approach to assess the effect of landscape variables at increasing spatial scales on forest bird diversity in Mediterranean areas affected by wildfires. They concluded that mosaic-like landscapes shaped by fires in the Mediterranean basin are not strongly associated with negative effects fragmentation on forest birds other than those related with habitat loss.

The spatial distribution of all seedlings is also affected by the location and size of burned trees. Using field data and statistical analysis Ne’eman et al (1992) investigated the spatial distribution of seedlings of the dominant perennial plant species (*Pinus halepensis, Cistus salviifolius, Rhus coriaria*) after a wild fire in an eastern Mediterranean pine forest. It is suggested that variation in the heat of the fire, in the amount of ash between burned pine trees of different sizes, and in the distance from the burned canopy are responsible for the observed pattern of seedling distribution.

For the Mediterranean region there several key factors that need to be addressed when analyzing forest vulnerability towards fragmentation. Low forest cover of the Mediterranean landscape contributes to isolation of forest patches. On the other hand, high land use heterogeneity is often connected to higher species diversity. As Mediterranean forests are highly vulnerable to forest fires also adjacent land use classes neighbouring the forest patches need to be considered when addressing forest vulnerability.

Key factors that need to be studied are: Distance from road, distance between forest patches, forest cover per landscape unit, connectivity, landscape heterogeneity, surrounding land use classes of forest patches.

4.5 Scale dependency of pattern analyses

The range of spatial and temporal scales at which ecological problems are posed has expanded dramatically in recent years, and the need to consider scale in ecological analyses has often been noted (O'Neill et al. 1986; Addicott et al 1987; Meentemeyer & Box 1987; Morris 1987; Turner et al. 1989, Chust et al. 2004, Saura 2004; Uumea et al. 2005). Scale effects must be considered carefully when analysing responses of organisms to spatial pattern because concepts such as equilibrium and species persistence are scale dependent. However, understanding the biotic effects of spatial pattern at multiple scales is in its infancy and remains a high priority for ecology (Turner et al. 2001). There is no a priori justification of what spatial scale is biologically the most important for any given species. Fragmentation effects can occur for large-scale landscape properties such as the coverage of mature forests, but also for specific habitat properties at very local scale such as the occurrence of coarse woody debris within a forest (Kouki et al. 2001).

The impact of habitat fragmentation on species abundance varies greatly with scale. A study of Holland et al. (2004) demonstrated that different species of cerambycid beetles respond to the amount of forest cover at very different spatial scales, ranging from 20 to 2000 meters. Also Chust et al. (2004) stress the importance of a multiscale approach when studying responses to fragmentation. They identified scales at which forest spatial pattern causes changes in species richness and abundances of two different insect species groups (Dipteran and Homopteran) and found that effects on Homopteran richness occurred at finer spatial scale (6.25 ha landscape extent) than responses of the Dipteran species groups which were sensitive at a scale of more than 250 ha landscape extent. An American study demonstrated scale dependency of relationships between bird species richness and landscape characteristics. Mean patch size of forests explained bird diversity at small extents but became decreasingly important as landscape area increased whereas the number of forest patches and total forest edge explained more variance as landscape extent increased (Mayer and Cameron 2003). Bayne et al. (2005) showed that negative responses of ovenbirds on seismic line density in a Canadian study area were visible only at large scale but not at landscape level. Åberg 2000 suggests a scale of at least territory size (20-40 ha) when investigating fragmentation effects on hazel grouse occurrence in boreal forests. Fisher et al. (2005) found that the relationship between forest landscape structure and presence of North American red squirrels was not constant as the spatial extent of the landscape varied.

The mentioned studies show that parameters and processes important at one scale are frequently not important or predictive at another scale, and information is often lost as spatial data are considered at coarser scales of resolution (Henderson-Sellers et al. 1985; Meentemeyer and Box 1987). Ecological problems often require the extrapolation of fine-scale measurements for the analysis of broad-scale phenomena. The lack of comparability of fragmentation indices across spatial resolutions seriously limits the usefulness of integrating geographic data obtained at different spatial scales (Saura 2004). Therefore, the development of methods that will preserve information across scales or quantify the loss of information with changing scales has become a critical task. Such methods are necessary before ecological insights can be extrapolated between spatial and temporal scales. The spatial scale of ecological data encompasses both grain and extent. Grain refers to the resolution of the data, i.e., the area represented by each data unit. Extent refers to the overall size of the study area. In studies of landscape structure or function, information may be available at a variety of levels of resolution, data must often be compared across large geographic regions, and it may be necessary to extrapolate information from local to regional scales. The effects of grain and extent are thus of particular concern, and the response of ecological attributes measured on the landscape to changes in spatial scale is not known. Patches and their boundaries are sensitive to the scale at which they are viewed. The response of species to patchiness may depend on the resolution and on the extent by which the spatial pattern is perceived.

Gardner et al. (1987) demonstrated that the number, size and shape of patches in randomly simulated landscapes vary with extent, as indicated by the linear dimension of the landscape. Turner et al. (1989) concluded that the spatial scale at which landscape patterns are quantified influences the results obtained, and measurements made at different scales may not be comparable. Furthermore, qualitative and quantitative changes in measurements across spatial scales will differ according to how scale is defined. Thus the definition and methods of changing scale must always be explicitly stated. The notion of scale is thus of great importance in understanding ecological processes. Chust et al. (1987) concluded that the detection of optimal scales and the use of satellite images enable maps of important biotic indicators to be drawn up. If grain and extent are specified, information can be easily transferred across scales (Allen et al. 1987).

Turner et al. (1989) observed the effects of changing the grain and extent of landscape data on observed spatial patterns and identified some general rules for comparing measures obtained at different scales. They predicted the effects of changing grain size on the proportion of a landscape occupied by a particular land cover type analytically, and then tested against randomly generated landscapes. The grain and extent of real and simulated landscapes were also experimentally varied and indices of diversity, dominance and contagion were used to compare resulting landscape patterns. The indices of dominance and contagion decreased with increasing grain size, but the indices increased as extent increased. Dominance and contagion increased as resolution was decreased as long as the number of land cover types remained the same. Similarly, the number of land uses increased as extent increased, and dominance increased as the number of land cover types increased. Meentemeyer & Box (1987) also showed that increasing the number extent of the study area, increases the range of values for a landscape variable. Generally, apparent detail is lost with increased area (subject to density of patchiness), and decreased area involves newly apparent detail (Turner et al. 1989; Meetemeyer & Box 1987).

The apparent proportion of the landscape changes with grain. In general information about the less frequent land cover types is most easily lost, with the rate of loss depending upon their spatial arrangement. Rare cover types with patchy arrangements disappear more rapidly with decreasing resolution than contagious ones (Turner et al. 1989). Patch density is inversely related to the appropriate scale of analysis (Meetemeyer & Box 1987). Information is lost in coarser grain spatial data.

Responses of landscape metrics to changing grain size vary among landscapes and metrics. Frohn (1998) stated that the contagion index varies depending on spatial resolution. Uuemaa et al. (2005) investigated scale dependence of landscape metrics on artificial and real landscapes and confirmed that the mean Euclidean nearest neighbour distance and contagion are directly dependent on grain size. Saura (2004) examined the effect of spatial resolution on six common fragmentation indices: number of patches; mean patch size; edge length; largest patch index; patch cohesion; and landscape division. All metrics but patch cohesion indicated lower fragmentation at coarser spatial resolutions. An arbitrarily large value of patch cohesion was obtained by resampling the pattern to smaller pixel sizes. Landscape division and largest patch index were found to be the least sensitive indices to spatial resolution effects.

Even though, it is possible to identify simple relationships between landscape parameters measured at different scales, the exact relationship varies across landscapes and does not permit extrapolation from one region to another (Turner et al. 1989). Characterising the relationships between ecological measurements and the grain and extent of the data makes it possible to predict or correct for the loss of information with changes in spatial scale. More, importantly, the ability to predict how ecological variables change with scale may open the door to extrapolating information to larger scales and to comparing data measured in different regions.

Weng 2007 addressed the temporal dynamics of landscape change in the Dane County, Wisconsin. Transect analysis with temporal trend analysis were integrated and the changes of residential pattern in relation to forms of urban growth specifically discussed. Changes of landscape pattern from 1968 to 2000 were analyzed by FRAGSTATS with four metrics—percentage of landscape (PLAND), Shannon's evenness index (SHEI), patch density (PD), and mean patch size (MPS). Findings from metric analyses revealed that the degree of land-use diversity and landscape fragmentation is positively related to the degree of urbanization.

In this project the dependency of pattern analyses on both spatial and temporal scales will be studied by using datasets of 25m and 100m resolution respectively with time differences of up to 50 years. The pattern analyses at a temporal resolution of 10 years and more will reflect changes in the forest pattern in the different bio-geographical regions and conclusions can be drawn based on the results regarding critical developments for forest biodiversity. A temporal resolution of 10 years will reflect short-term changes in forest fragmentation resulting e.g. from fast-developing infrastructure or clear-cutting. Long-term changes such as urban spread, the abandonment of agricultural land (and connected afforestation) or forest fragmentation resulting from degradation of Mediterranean forests can be better addressed with datasets covering time spans of at least 50 years. This is possible with the availability of BioPress land cover data which include data on 1950 as well as 2000.

5 Test areas and data availability

All the geographic layers used in this project are projected in the ETRS-LAEA projection: pan-European CRS with datum ETRS89 in European azimuthal equal area.

Forest/land cover maps

This layers are derived by a number of different national or European projects such as Corine Land Cover and BIOPRESS. In order to make these maps comparable they have been harmonized in terms of geometric resolution and system of nomenclature.

Low resolution maps have a pixel of 100 m and are available in large test areas while high resolution maps have a pixel of 25 m and are available in small test areas.

Small test areas with high resolution data are located inside large test areas covered by low resolution data.

The nomenclature system adopted is derived from the EEA (2006) European Forest Types at category level integrated by basic information for non-forested lands:

Forest (FAO, definition)

1. Boreal forest
2. Hemiboreal forest and nemoral coniferous and mixed broadleaved□coniferous forest
3. Alpine coniferous forest
4. Acidophilous oak and oak□birch forest
5. Mesophytic deciduous forest
6. Beech forest
7. Mountainous beech forest
8. Thermophilous deciduous forest
9. Broadleaved evergreen forest
10. Coniferous forests of the Mediterranean, Anatolian and Macaronesian regions
11. Mire and swamp forest
12. Floodplain forest
13. Non riverine alder,birch, or aspen forest
14. Plantations and self sown exotic forest

15. Other Wooded Lands (FAO, definition)

For Non-forest areas the Corine Land Cover classes are adopted.

The forest maps prepared in the five test areas will be used as binary input for the GUIDOS software (developed by JRC). Here follow the description of test sites with a detailed list of available files that have to be processed by JRC with the GUIDOS software.

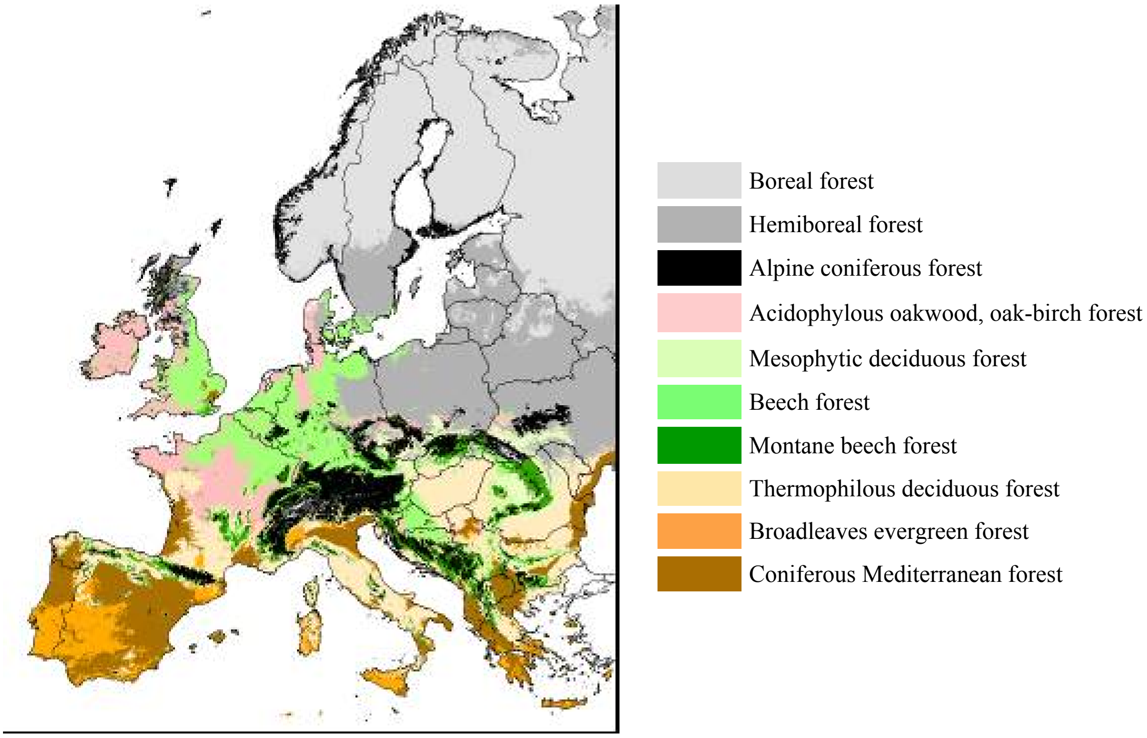


Figure 2: potential forest category distribution according to EEA (2006), available at http://forest.jrc.it/Forest_and_Climate_Change/Forest_Trends/Fcatresults/.

Test area	Resolution	Year	Status
Finland north	25	1950, 1990, 2000	OK
	100	1990, 2000	OK
Finland south	25	1950, 1990, 2000	OK
	100	1990, 2000	OK
Germany	25	1990, 2000	OK
	100	1990, 2000	OK
UK north	25	2002, 2008	OK
	100	2000	CLC1990 NOT AVAILABLE
UK south	25	2002, 2008	OK
	100	2000	CLC1990 NOT AVAILABLE
Italy north (Alpine)	25	1983, 2005	OK
	100	1980 (Belluno province only), 1990, 2000	OK
Italy south (Mediterranean)	25	1936, 1954, 1995, 2005	OK
	100	1980, 1990, 2000	OK

Table 3: data availability for the different test areas.

Roads

Vector files of main road networks have been acquired for the study areas. Since such information are used in the project to model the potential vulnerability from anthropogenic disturbances just main roads with real car traffic have been taken into consideration. This also due to the fact that:

- information regarding the location of "forest roads" are not available and its acquisition is very time consuming and out of the scope of the project;
- "forest roads" are always related to the main road network, it is possible to infer on the density of "forest roads" on the basis of the density of the main road network;
- "forest roads" are in general completely permeable under an ecological point of view so their real effect on habitat fragmentation is in general very low.

Vector roads have been rasterized both at high and low resolution. No multitemporal road maps are available so they are all dated to 2007.

Railways

Vector file of the railway network have been acquired for the study area. Vector rails have been rasterized both at high and low resolution. No multitemporal road maps are available so they are all dated to 2007.

Settlements

Main towns and cities are available in the forest/land cover maps already. In order to have a more detailed and complete evaluation of anthropogenic disturbances also smaller towns have been considered in the analysis. Vector maps have been rasterized both at high and low resolution. No multitemporal settlements maps are available so they are all dated to 2007.

Forest fires

The boundaries of main forest fires (larger than 50 ha) have been acquired from the EFFIS system of the JRC. Yearly data are available from the south of Italy starting from the year 2000. Original EFFIS files have been rasterized at low and high resolution.

Potential Natural Vegetation

Maps of potential natural vegetation at 1 km resolution are available as a result of a study of the JRC "Forest habitat suitability distribution and future trends under Climate Change". Maps used in this study are based on potential presence of different forest species and of European Forest Categories (EEA, 2006).

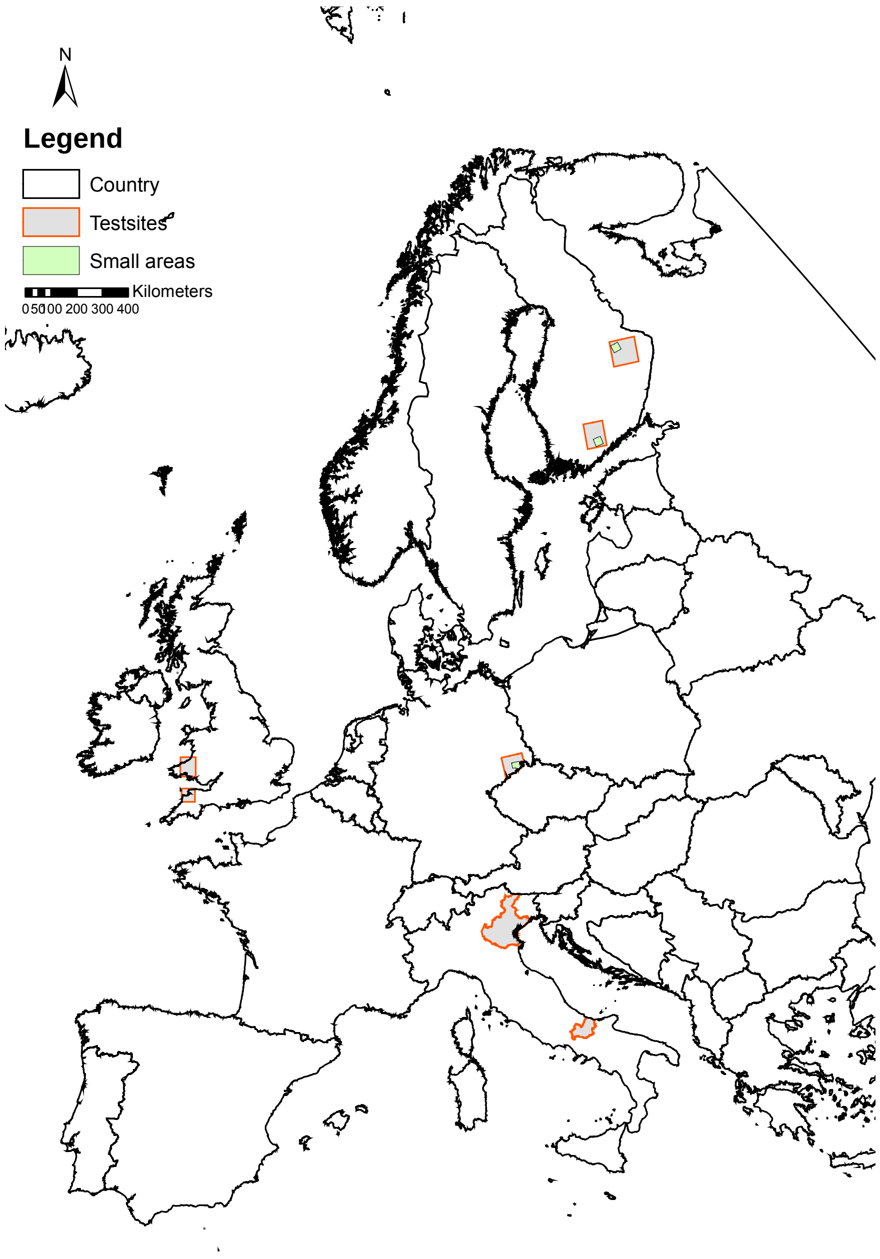


Figure 3: geographic location of test areas. When small areas are not visible (Italy and UK), small high resolution areas are the same of large low resolution areas.

5.1 Test data for the boreal test site (Finland)

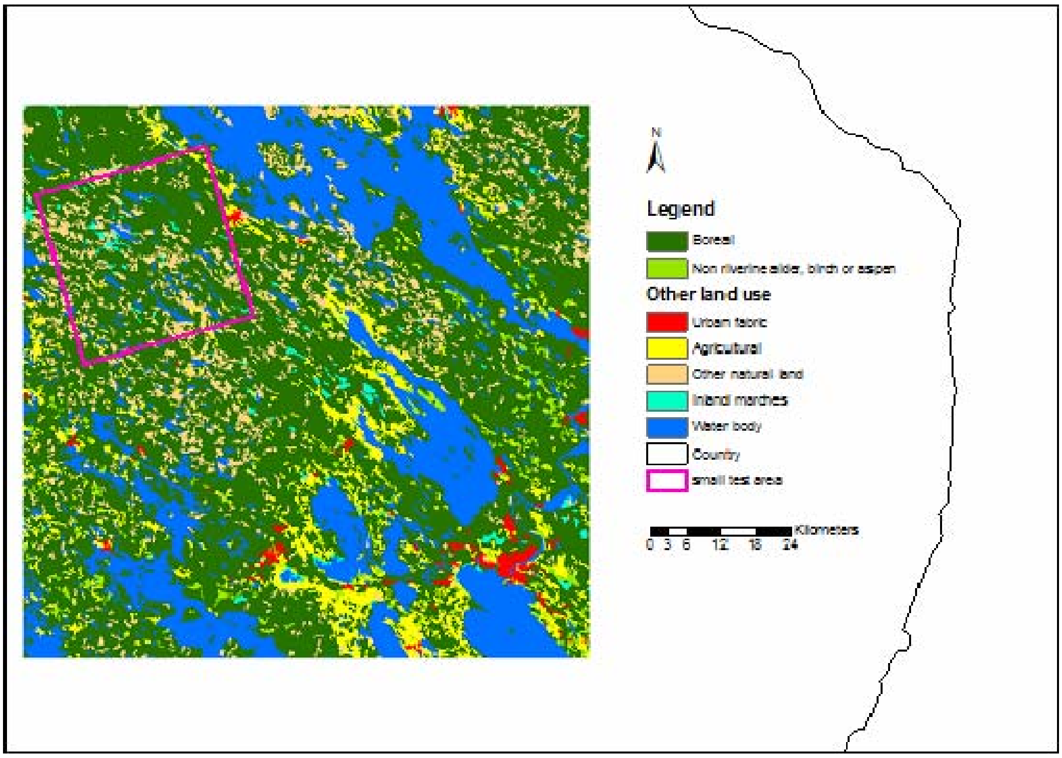


Figure 4: low resolution data in the northern test area in north Karelia/north Savo. The high resolution test area is in the upper left part.

In Finland two test sites are available:

- Northern test area: North Karelia / North Savo
- Southern test area: Häme-Uusimaa

The extensions of the large test areas are ca. 9000 km^2 (North) and ca. 7000 km^2 (South).

The low resolution multitemporal dataset is derived from CLC 1990 and CLC 2000 revised by manual photointerpretation of the original Landsat imagery.

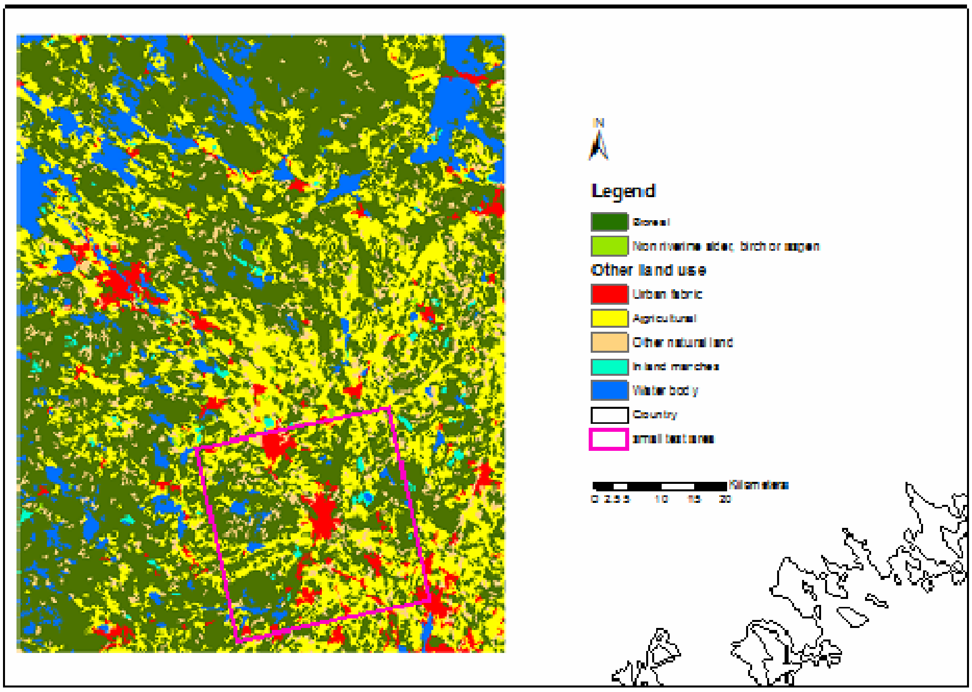


Figure 5: low resolution data in the southern test area in Häme-Uusimaa. The high resolution test area is in the lower part.

The high resolution multitemporal datasets are derived from: a project carried out by the Finnish Environment Institute SYKE, from the BioPress project and from forest stand data property of Forest Centre Joensuu. They are available in both test areas for the year 1950 and 1990.

5.2 Test data for the continental test site (Germany)

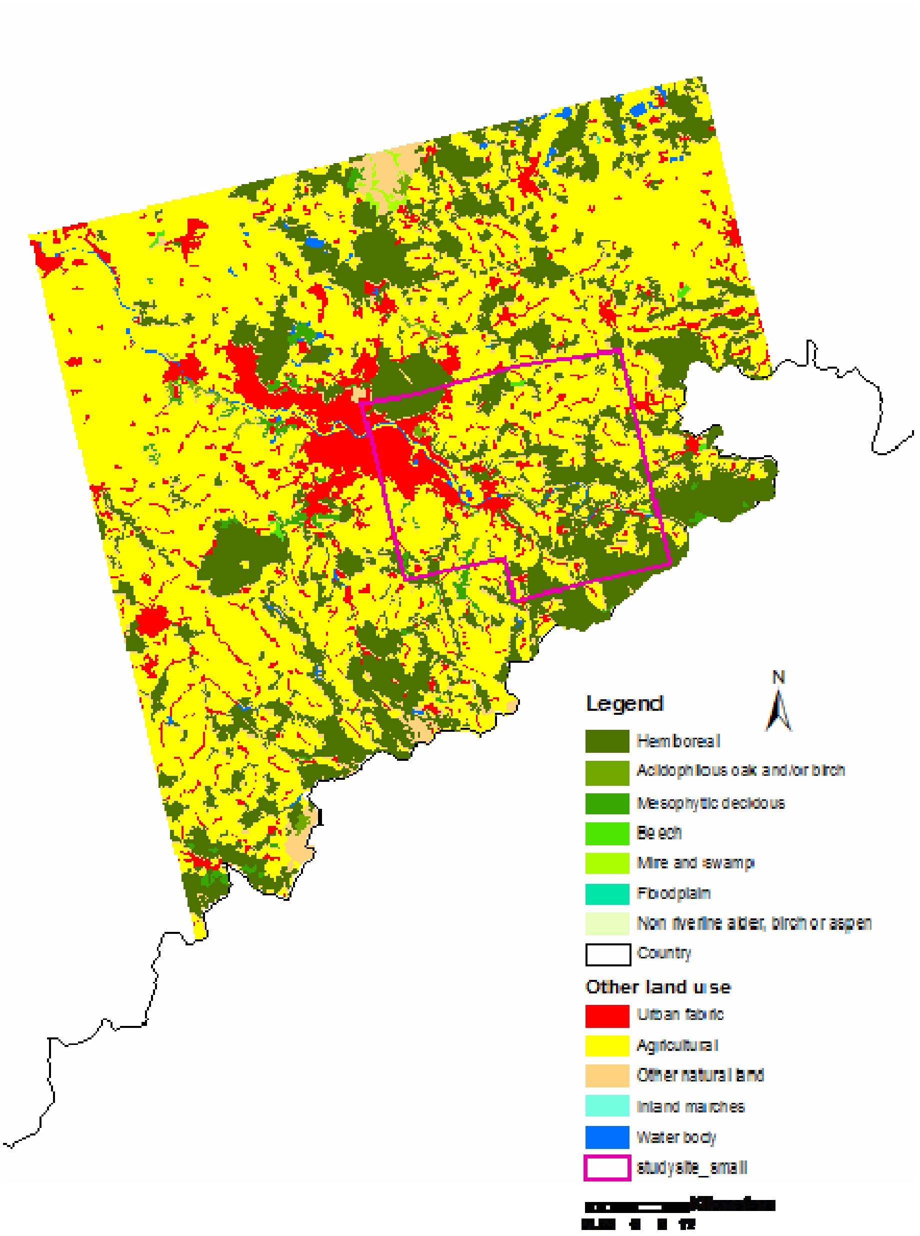


Figure 6: low resolution data in the test site in Saxony, Germany. The high resolution test area is in the middle of the image.

The test site is located in the administrative region of Bundesland Saxony and it approximately 5100 km² wide.

Low resolution data are coming from a revised version of the CLC dataset by manual photointerpretation of Landsat imagery assisted by the use of NFI and ICP plots.

The high resolution data are coming from a revised version of the BIOPRESS dataset with the use of one window and two transects of that project.

5.3 Test data for the Atlantic test area (UK)

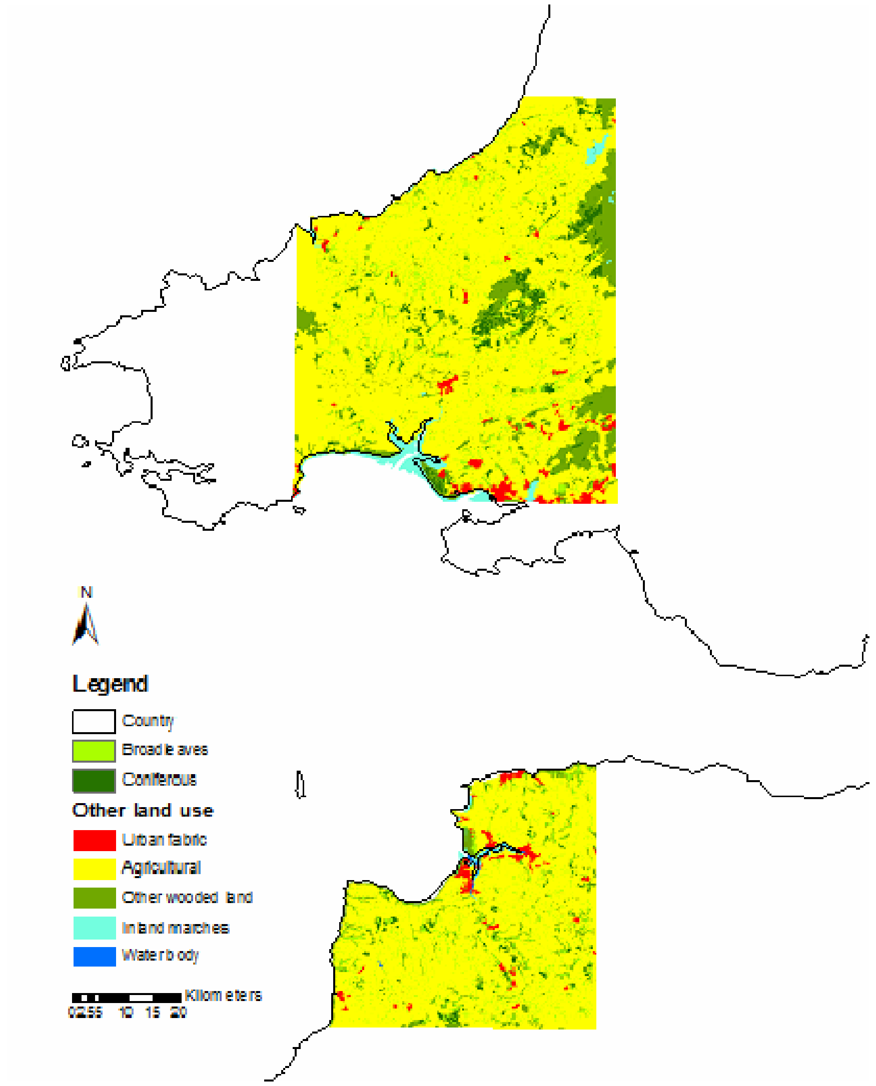


Figure 7: low resolution data in the two test site in UK. North Wales in the upper part and South England in the southern part. The high resolution test areas are the same of the low resolution one.

The low resolution data are coming from the LCM2000 project, Land Cover Map by the Centre For Ecology and Hydrology of the Natural Environment Research Council. They are derived by classification of multitemporal IRS and Landsat imagery.

High resolution maps are instead from Digital Woodland Map for England developed by Forestry Commission. Old maps are from digitalization of paper maps, new maps are instead produced by interpretation of aerial photos in scales between 1:10.000 and 1:25.000.

Both raw high and low resolution maps for UK are not developed on the species of detailed system of nomenclature (at least not for the purposes of the project). For this reason original maps on the basis of the information from the National Forest Inventory and from ICP plots have been reclassified accordingly to the EEA (2006) system of nomenclature. Part of this activity is still on running.

5.4 Test data for the Mediterranean test area (Italy)

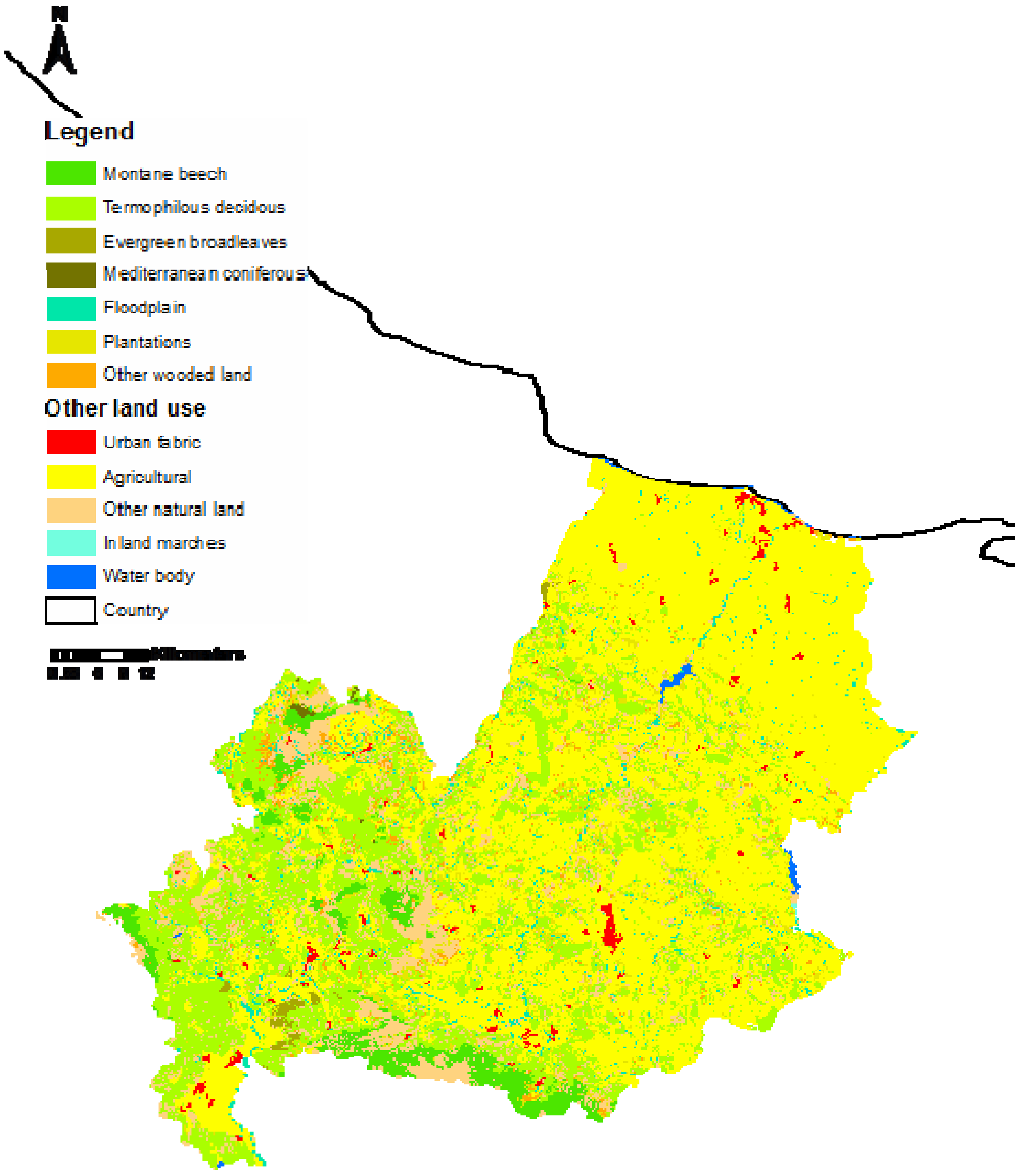


Figure 8: low resolution data in the Molise test site in Italy. The high resolution test area is the same of the low resolution one.

All the high and low resolution datasets are available for all the administrative Region of Molise (ca. 4438 km^2).

The low resolution multitemporal datasets are derived from the CLC data and are available for the years 1980 (Landsat MSS), 1990 (Landsat 5 TM), 2000 (Landsat 7 ETM+).

The high resolution multitemporal datasets are available for the years 1936, 1954, 1992, 2005.

The map at 1936 was derived by field work only, 1954 and 1992 maps are based by traditional photointerpretation of aerial photos while the new map at 2005 is based on pphotointerpretation of digital orthophotos. The systems of nomenclature of these maps are based on main tree species composition that have been reclassified accordingly to the European Forest categories (EEA, 2006).

5.5 Test data for the Alpine test area (Italy)

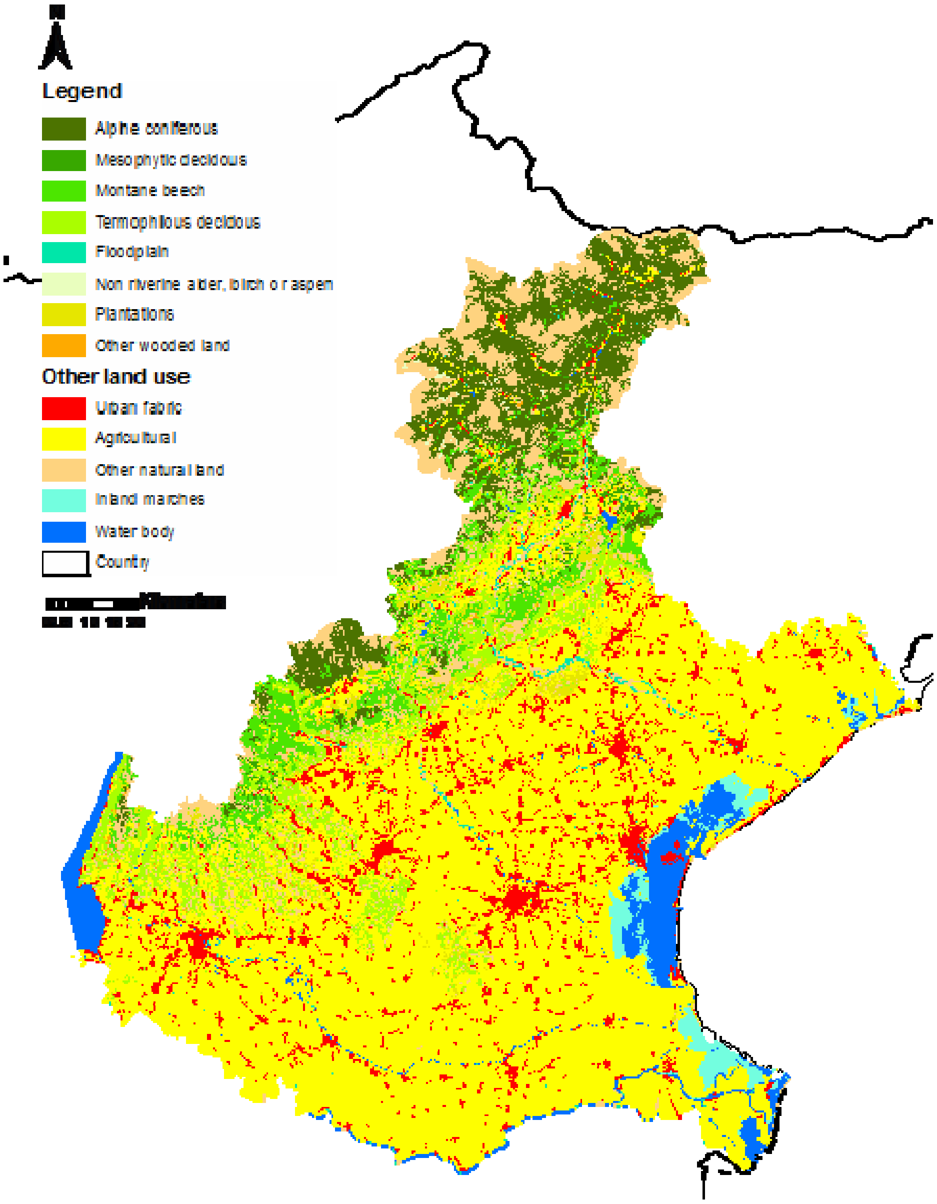


Figure 9: low resolution map in the Veneto test site (same coverage for high resolution).

F

All the high and low resolution datasets are available for all the administrative Region of Veneto (approx. 18378 km^2) with the exception of the low resolution map at the year 1980 that is available just for the Province of Belluno (northern alpine part, 3678 km^2).

The low resolution multitemporal datasets are derived from the CLC project and are available for the years 1980, 1990, 2000.

The high resolution multitemporal datasets are available for the years 1936, 1983, 2005.

Procedures and system of nomenclature adopted are similar to those ones described for the Molise Region.

6 Multi Criteria Evaluation: an introduction

6.1 Introduction

Land evaluation is a powerful tool to support decision-making in land use planning and deals with the assessment of the (most likely) response of land when used for specified purposes. It requires the execution and interpretation of surveys of climate, soil, vegetation and other aspects of land in terms of the requirements of alternative forms of land use. Land suitability assessment (LSA) can be regarded as a specific case of land evaluation and is an appraisal of land characteristics in terms of their suitability for a specific use (FAO, 1976). The basic concept behind LSA is that suitability for a specific and sustainable use of the land is the synthetic result of complex relationships between different land environmental qualities (e.g.,, climate, soil characteristics and slope). Suitability for a specific use is therefore evaluated by matching requirements for that use with characteristics and qualities of land components.

Land suitability is usually expressed by a hierarchical system organised into orders and classes (FAO, 1976). The orders indicate whether the land is suitable or not for a given land use; two main orders are distinguished:

- S (suitable): land on which sustained use is expected to yield benefits which justify the inputs, without unacceptable risk of damage to land resources;

- N (not suitable): land whose qualities appear to preclude sustainability for the considered land use; the limitations are so severe that they preclude the successful application of the given land utilization type.

Classes reflect degrees of decreasing suitability within the order "suitable". Most often three classes are applied:

- S1 (highly suitable): land which has no significant or only minor limitations to the sustained application of a given land use;

- S2 (moderately suitable): land which has limitations that are moderately severe for sustained application of the given land use; the limitations will reduce productivity or benefits and will increase the required inputs;

- S3 (marginally suitable): land which has severe limitations for sustained application of a given land use.

LSA applications are common in many fields of land use planning. In urban planning, for example, LSA is undertaken to determine the suitability of land for housing within a municipality territory, whereas in rural planning LSA is often used to assist the zonation of a rural region into a mosaic of land units, with each capable of sustainably supporting specific farming systems. The same holds for forest planning: if the ecological requirements of a given forest tree species are sufficiently known such that auto-ecological behavior of the species can be plotted reliably, maps of land suitability for that particular forest species can be potentially drawn from geodatabases of environmental factors.

LSA is also frequently used to support the creation and management of protected areas. An example is the identification of the location and boundaries of new protected areas or for mapping different levels of environmental protection inside an existing protected area. In biological conservation, LSA is frequently applied when evaluating habitat for endangered fauna or flora species although applications of this nature rely upon expert-based knowledge of the habitat preferences of species as well as on data relating to actual habitat use. Such an approach is also referred to as habitat suitability modeling (Hirzel et al., 2006).

Geographic Information Systems (GIS) are powerful tools within which LSA can be performed, particularly as geographical databases on land qualities can be handled and combined. For this reason, and particularly in the past few decades, LSA has become one of the most common applications of

GIS in land planning and management (McHarg, 1969; Hopkins, 1977; Brail and Klosterman, 2001; Collins et al., 2001; Malczewski, 2004).

In its first applications, GIS-based LSA was basically carried out by overlaying thematic maps in a digital format. Land suitability was derived from the input thematic layers using simple map algebra or map logic operators. Over the last two decades, LSA issues have been addressed with increasingly complex conceptual models by developing specific solutions of data handling and processing. Amongst others, the so called GIS-based multi--criteria evaluation (MCE) or multi-criteria analysis (MCA) procedures have been widely adopted for various applications (e.g., Banai, 1993; Jankowski and Richard, 1994; Joerin, 1995; Barredo, 1996; Beedasy and Whyatt, 1999; Malczewski, 1999; Barredo et al., 2000; Mohamed et al., 2000; Bojorquez-Tapia et al., 2001; Dai et al, 2001; Joerin et al., 2001, Church, 2002).

The basic idea of the “spatial pattern” project is to develop a methodology derived from the GIS-based MCE land suitability application for modeling potential FVRR.

Within this framework, the main objectives of this chapter are to i) outline the basic principles of different GIS-based LSA approaches and ii) present the methodology to be applied for spatial modeling of potential FVRR.

6.2 GIS-based land suitability assessment

A GIS can be defined as a computer-assisted system for the acquisition, storage, analysis and display of geographic data (Eastman, 2006). As such, the GIS can provide essential management information or be used to develop a better understanding of environmental relationships. In recent years, considerable interest has grown around the use of GIS for land suitability mapping and modelling and two main groups of approaches to GIS-based land suitability can be distinguished: (i) overlay mapping and (ii) multi-criteria evaluation methods (Collins et al., 2001).

6.2.1 Overlay mapping

The computer-assisted overlay techniques were developed in response to limitations of manual methods of mapping and combining large datasets in paper format (MacDougall, 1975; Steinitz et al., 1976). Overlay mapping is quite simple; input thematic layers (e.g.,, precipitation, soil depth, pH and/or slope) required to evaluate land suitability for a specific use are acquired and transformed into input factors (or criteria). These are stored in the form of interval data in that pixels (raster data) or polygons (vector data) are assigned with relative ranking values based on well-known relationships between land use requirements and land qualities. For example, to assess land suitability for a specific crop, single environmental factors (e.g., rainfall, soil drainage, soil texture, pH and temperature at germination) are ranked into classes of suitability (cf. § 1); different criteria are then combined using logic or algebraic functions to obtain the final suitability map (Lyle and Stutz, 1983). Such a simple approach is basically the reproduction in a computer environment of techniques developed for the overlay of paper maps (Tomlin, 1990). The major criticism to the conventional map overlay approach concerns the inappropriate use of methods for standardizing suitability maps and untested or unverified assumptions of independence among suitability criteria (Hopkins, 1977; Pereira and Duckstein, 1993). This limitation can be overcome by integrating GIS and multi-criteria decision making (MCDM) methods.

6.2.2 Multi-criteria decision making

The MCDM procedures (or decision rules) define a relationship between the input thematic information and the output suitability map that is more complex than logic or algebric relationships adopted in overlay mapping. The suitability model can include decision maker’s preferences and can be based on various manipulations of data and preferences according to specified decision rules. All the input thematic layers are transformed into constraint or factor criteria:. A constraint serves to limit the land use options (or alternatives) under consideration (Eastman, 2006). Examples include the exclusion of protected areas from housing development or the exclusion from farming of areas with slopes exceeding a 30 % gradient. A factor is a criterion that enhances or limits land suitability for a specified land use option (alternative). It is most commonly measured on a continuous scale. For example, the steeper a slope, the more severe are the limitations for the establishment and sustainable management of forest productive plantations. A number of multi-criteria decision rules have been implemented in the GIS environment for tackling land-use suitability issues. The decision rules used to aggregate the input criteria (constraints and factors) into a final suitability map can be classified into multi--criteria (or multi--attribute) and multi--objective decision-making methods (Carver, 1991; Malczewski, 1999). In the multi--criteria analysis, several input factors have to be aggregated to derive one final suitability map for a single specific objective. In multi--objective approaches, the analysis is oriented towards considering several different possible objectives, whether conflicting or otherwise, within a given area and can be considered as being the GIS answer to address the inherent conflicts of land planning.

6.2.3 Multi-criteria methods

In the last decade, a number of multi-criteria evaluation methods (MCE) have been proposed for GIS based land suitability analysis (Eastman, 1999). Amongst others, the most common are (i) Weighted Linear Combination (WLC) or (ii) Ordered Weighted Averaging (OWA).

6.2.3.1 Weighted linear combination

In WLC, each input thematic layer (criterion) is assigned with a weight indicating the relative degree of importance each criterion plays in determining the suitability for an objective. Input criteria are standardized to a numeric range, quantifying scores of suitability, and then combined by means of a weighted average. Criteria weights are assigned according to decision-maker preferences and determine how each criterion will trade-off relative to other factors: a criterion with a high weight can tradeoff or compensate for poor scores on other factors (Eastman, 2006). In contrast, a criterion with a high suitability score but a small weight can only weakly compensate for poor scores on other factors. There are, however, fundamental limitations associated with the use of WLC in a decision-making process which are comprehensively discussed by Jiang and Eastman (2000) and these authors regard the WLC approach as just an extension to, and generalization of, the conventional map overlay methods in GIS.

6.2.3.2 Ordered weighted average

OWA is a more complex class of multi-criteria methods (Yager, 1988) that enables the degree of tradeoff between the criteria to be governed; in the OWA, two sets of weights (criteria and order) are applied: Criteria weights are defined as in the WLC, while order weights determine the overall level of tradeoff allowed. Unlike criteria weights, order weights do not apply to any specific criterion but are defined on a pixel-by-pixel basis according to the ranking order of criteria scores (order weight 1 is assigned to the lowest ranked criteria of a given pixel, order weight 2 to the next higher-ranked criteria for that pixel, and so forth). To understand how order weights influence final results, consider the case where criterion weights are equal for three criteria A, B and C (factor weights = 0.33); the score of these factors for one pixel is respectively 100, 50 and 200. When ranked from minimum to a maximum value, the order of these factors for a given pixel is B, A, C. Any combination of order weights can be defined in OWA that sum to 1. In the combination [1, 0, 0] the criterion with the minimum value in the set (B) will receive all the possible weight and no trade off is possible with other higher ranked criteria (result = 50); this solution is from a decision-making standpoint risk-averse: the final aggregated score is more or less close, depending on criteria weights, to the suitability value of the lowest ranked criteria; similarly, the combination [0, 0, 1] makes irrelevant the contribution of lower ranked criteria (result = 200); it can be regarded as a risk-taking solution because the final aggregate score is more or less close to the suitability value of the highest ranked criteria. As all the combinations in the continuum between these two extremes are possible, different degrees of tradeoff between the ranked criteria can be set for determining the suitability for an objective. The OWA, therefore, allows a variety of land use strategies to be developed with these ranging from a risk-averse through all intermediate the neutral-towards-risk strategy (corresponding to the conventional WLC) to a risk-taking strategy. Thus, OWA can be considered as a generalization of WLC.

6.2.4 Multi-objective methods

Multi-objective methods are applied when the suitability for different land uses (or more in general for different objectives) have to be defined in a given area. The different objectives may be non-conflicting (complementary) or, much more frequently, conflicting (Eastman, 2006). They are complementary when different objectives may share the same land unit (e.g., a forest plantation in a protected area); they are conflicting when they are mutually exclusive (for instance, the cultivation of two different agricultural crops in the same parcel). The solution of multi-objective complementary problems is, in general, quite easy: a number of land suitability analyses equal to the number of the objectives is performed and the results are aggregated to find the optimal solution by analysing the degree to which each land units meet the considered objectives (Voogt, 1983). Multi-objective conflicting models are often tackled by converting them to single objective problems and then solving the problem using the standard linear programming methods in order to find a compromise solution (Feiring, 1986; Diamond and Wright, 1988; Aerts, 2002; Malczewski, 2004). An advantage of the model (and of the linear programming approaches in general) is the ability to map the patterns and opportunity costs in addition to the optimal land suitability pattern. This added information could be used for evaluating the robustness of land suitability patterns and identifying areas where modifications could be made without significant impacts (Cromley, 1994; Cromley and Hanink, 1999).

6.3 *Examples of GIS-based LSA applications*

A full overview of the applications of the LSA is beyond the scope of this chapter. Instead, this section provides only an excerpt for a circumstanced comprehension of the applications developed so far focusing specifically on agriculture and forestry, habitat suitability, urban planning and land use planning.

6.3.1 Agriculture and forestry

In agriculture and forestry, land suitability has a very long tradition with various kinds of applications to support the sustainable agricultural development in term of selection of crops and cultivation techniques in a given area.

Many studies (e.g.,, Liengsakul et al., 1993; Kalogirou, 2002) have focused attention on land suitability for agricultural crops, whilst others have considered woods and forest plantations (e.g., Chirici et al., 2002). In most applications, LSA has been applied to support the improvement of the agricultural sector in relatively poor rural areas (e.g., Thailand, Mexico and NW Spain) by introducing new crops or new cultivation methods. The majority of the applications have focused on GIS overlay techniques although MCE with fuzzy classifications is increasingly being used over the last five years. Data input layers relate typically to the physical environment (mainly climate, topography and soils). In many cases, expert knowledge has been integrated and some studies (e.g.,, Cools et al., 2003) have also considered farmers' opinion. In general, the outputs from such modelling has led to improvements in agricultural land use and also changes in related policies. The results of several studies indicate the usefulness of fuzzy modeling in classifications based on farmers knowledge of agricultural land suitability, which could provide useful information for optimizing land-use planning. Frequently, one of the main limitations in such application is the definition of the weight to be applied to different factors. The most common approach is to use questionnaires to experts/farmers to define factors ranking. Answers are transformed into weights by simple linear relationships (the higher is the average importance associated to a given factor and the higher is its weight in the MCE).

6.3.2 Habitat suitability

Habitat suitability evaluation is generally based on multi-variate analysis but when the relationships between ecological variables and habitat suitability are well-known LSA methods can also be applied to habitat suitability mapping. For a detailed description of such methods refer to § 3.4.

6.3.3 Urban planning

In urban planning LSA is frequently used to identify future settlements areas (Sui, 1992; Kaiser, 1995). In such applications, both environmental and socio-economic input factors are used to map possible future development of periurban areas, frequently handling multi-objective problems of conflicting interests of stake-holders. The methods used in these applications range from linear programming (Cromley and Hanink, 1999), MCE (Bannet et al. 2005) and Analytical Hierarchical Process (Dyer, 1990; Thapa and Murayama, in press) to mixed approaches (Ligtenberg et al., 2001). It is noteworthy that in this field, LSA is routinely applied, though with simple map overlay approaches, to support urban planning activities.

6.3.4 Land planning

LSA is frequently applied in land planning in ways similar to those applied to urban planning. In general, however, analyses are performed at a coarser spatial resolution, over wider territories and for more heterogeneous applications (Overmars et al., 2007); these frequently involve supporting decision-making in the selection of multiple land uses alternatives to be promoted in a given area (Bognar et al. 1998; Malczewski et al. 2003; Guo et al., 2006) or of sites where new infrastructures, such as greenways (Collins et al., 2001) or wind turbines (Meentemeyer and Rodman, 2006), are to be developed.

7 Modeling forest vulnerability, resilience and resistance

In traditional LSA approaches each single landscape element (one pixel of a raster map) is classified accordingly to its potential belong to the fuzzy set of elements suitable for a given land use. In other words within this project we propose the application of fuzzy MCE techniques to spatially evaluate the level of forest vulnerability, resilience and resistance that can be there for considered a special case application of the more general land suitability problem.

In the proposed approach each single forest pixel will be classified on the basis of its ecological distance from its potential local optimum. The closer the pixel value will be to 1 the higher will be the potential level of resistance and resilience and lower will the value of vulnerability.

Ecological optimum will be defined both for the five species specific approaches and for the species unspecific approach for a total number of six models that will be run on multitemporal dataset in all the test areas at two different spatial resolutions.

The models will be developed with a good balance between precision, to have a detailed model able to produce scientifically sound results, and generalization to enable the application of the models on large areas and in different environments.

The idea is to model the relationships fount in the literature review between spatial pattern information and FVRR with *fuzzy membership functions* (Van Ranst et al., 1996; Groenemans et al., 1997).

Fuzzy membership functions relate selected *forest spatial pattern factors* to an overall value of FVRR on the basis of the ecological distance between real condition and optimum potential conditions. FVRR will be expressed between a minimum of 0, related to a potential negative condition for FVRR and high distance from local ecological optimum (high vulnerability, low resilience, low resistance) and a maximum of 1, related to a potential optimal FVRR condition and limited distance from local ecological optimum (low vulnerability, high resilience, high resistance).

The application of fuzzy membership functions can be considered a normalization techniques that make possible the comparison between very different environmental variables.

The normalized (between 1 and 0) fuzzy maps, one for each of the selected forest spatial pattern factors, will be aggregated with different MCE techniques in order to have a sensibility analysis of the approach. The aggregation will produce one unique fuzzy map of FVRR.

The entire procedure will be repeated for:

- the five different test areas;
- within each test area, for the different study years;
- within each test area and study year, at different spatial resolutions.

7.1 Species unspecific model

The raw input data used in the work are described in § 5. On the basis of such data a number of different ecological factors were created.

The model was developed for one pilot study area (Italy, Molise). The methods and the variables used in the models were optimized on the basis of the first results obtained in the pilot study area in order to produce the final models that have been applied in all the study areas.

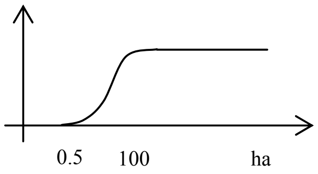
7.1.1 Input factors

Here follows the list of ecological factors that have been calculated from the original input raw data. All the output maps resulting from the MCE analysis are expressed in the 8 bit range between 255 and 0. A value of 255 means that the given pixel fulfils at 100% the biological requirements of local ecological optimum, for a given factor the level of resilience and resistance is maximum and the vulnerability is the minimum. At the opposite a value of 0 represent a biological condition very far away from local ecological optimum. The factors sometimes are calculated per pixel sometimes per patch. All the following graphs have the 8 bit range 0-255 in the Y axis.

The GUIDOS settings adopted are: 8,1,1 for low resolution (pixel 100 m) and 8,4,1 for high resolution (pixel 25 m).

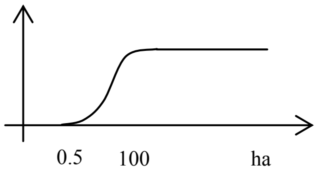
Patch size *(per patch)*

The area of each forest patch was calculated. An increasing logistic fuzzy membership function was used in order to evaluate the ecological functionality in terms of the size of the forest patch. A minimum threshold was fixed to 0.5 ha and a maximum potential optimum was fixed at 100 ha (see § 3 for ecological meanings of these values).



Core area size *(per patch)*

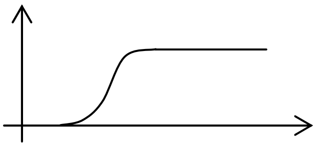
The area of each core forest patch was calculated. An increasing logistic fuzzy membership function was used in order to evaluate the ecological functionality in terms of the size of the forest patch. A minimum threshold was fixed at 0.5 ha and a maximum potential optimum was fixed at 100 ha (see § 3 for the ecological meanings of these values).



Distance from neighbor disturbing sources

Agricultural areas, pastures, urban areas and settlements determine a different disturbing factor of forest ecosystems. Forest edges have in fact more relevant ecological values when they are connected with rural areas (pastures and non-intensive agricultural lands). To take into account such a proximity effect from all non-forest land uses will be reclassified on the basis of their potential level of disturb against natural and semi-natural ecosystems. The relative disturbing factors considered in the final aggregation are used to evaluate the disturbing effect of the proximity of non-natural land uses. So continuous urban areas and highways have a disturbing effect that is four times that of agricultural areas and pastures, two times of intensive agricultural areas and so on.

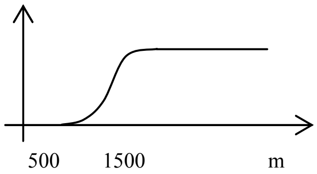
Very high potential disturbing level: continuous urban areas,



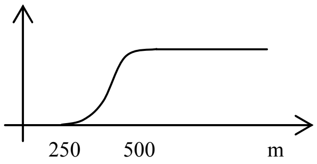
highways. Relative disturbing factor equal to four.

500 1500 m

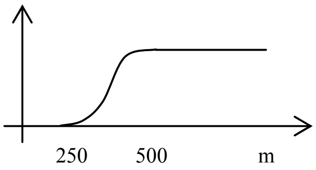
High potential disturbing level: discontinuous urban areas, small settlements, roads, railways. Relative disturbing factor equal to three.



Moderate potential disturbing level: intensive agricultural areas. Relative disturbing factor equal to two.

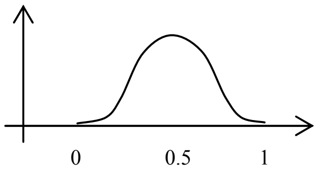


Low potential disturbing level: non-intensive agricultural areas, pastures. Relative disturbing factor equal to one.



A distance map was calculated from each disturbing source. Different logistic fuzzy membership functions were applied to transform the distance from disturbing sources in the ecological disturb against forest ecosystem. The four disturbing factors have been aggregated by weighted sum (adopted weights are equal to the relative disturbing factors) to obtain one single disturbing layer.

Forest edges/core area distribution *(per pixel)*

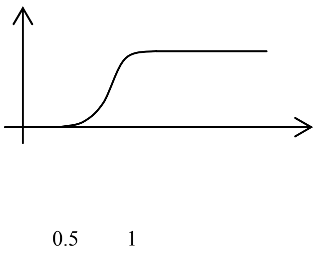


For a given part of a landscape good balance between core areas and forest edges should exist. These environments have in fact very different ecological conditions and are niches for different animal and plant communities (see § 3). For this reason an index for the evaluation of the distance from the optimum ecological balanced distribution of the two environments will be applied. On the basis of a simplified GUIDOS core/non-core map a circle moving window with radius of 5 km was used to calculate for each forest pixel both the ratio of core areas and of forest edges area on the total number of forest area within the considered window. Such values were contrasted with potential optimal values based on a 50% presence of the core and 50% presence of edges. Gaussian fuzzy membership functions was used to reclassify forest edge area and core area ratio against total forest area in order to evaluate the biological proximity to potential ecological optimum.

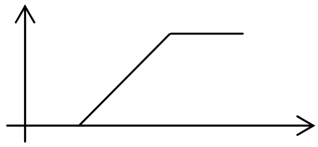
Potential percolation *(per pixel)*

Potential percolation (per pixel)

In order to emphasize the ecological role of *stepping stones* areas where forest patches are not physically connected but are close enough to enable the movement of animals or the genetic dispersion. To evaluate such a potential connectivity we will use the local percolation theory (Farina, 1998). When a forest pixel has in its surrounding a large area covered by forest pixels it is probable that the analyzed pixel is potentially connected with the other forest pixels, even if it is not physically adjacent. A circular moving window of 5 km radius was used in order to evaluate, for each forest pixel, the percent of the area covered by other forest pixels. This raster map was then elaborated with a logistic increasing fuzzy membership function as proposed by Farina (1998) having a minimum threshold of 50% and a maximum threshold of 100%.



Diversity & eveness of forest categories *(per pixel)*



Habitat diversity is strictly related to the stability of forest ecosystem, its resilience, resilience and vulnerability (§ 3). For this reason higher is the local forest habitat diversity and eveness and higher is considered the potential ecological functionality. Optimal diversity is anyhow limited by ecological factors such as climate and soil conditions. For these reasons on the basis of Forest Categories (EEA, 2006) potentially present in the same biogeographical area the target optimal condition was defined. The potential forest categories presence was evaluated on the basis of the results of the JRC project "*Forest habitat suitability distribution and future trends under Climate Change*" (Figure 2; available on line at http://forest.jrc.it/Forest_and_Climate_Change/Forest_Trends/). An increasing linear fuzzy membership function between one forest category and the maximum potential number of forest categories in the biogeographical area was used to reclassify t *Index* (*SHDI*) to forest ecosystem fun

$$SHDI = \sum_{i=1}^{m} (P_i \cdot \ln P_i)$$

Where P_i is the area proportion of the i-th forest category in the considered 5 km radius circular moving window.

A similar approach was followed for the eveness index (*SHEI – Shannon Eveness Diversity Index*). A potential optimum area distribution of the maximum number of forest categories in the biogeographical area was defined and a fuzzy increasi s used to reclassify eveness in to p functionality.

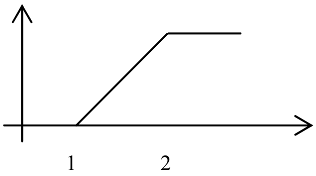
$$SHEI = \frac{-\sum_{i=1}^{m} (P_i \cdot \ln P_i)}{\ln \cdot m}$$

Where P_i is the area proportion of the i-th forest category in the considered 5 km radius circular moving window.

Shape *(per patch)*

A complex shape of forest patches increase the relative area extension of forest edges (ecotones) that have high potential ecological value (see § 3.1.2). The shape complexity was evaluated on the basis of fractal dimension (*FD*) of patches calculated as the ratio of logarithmic transformed perimeter and area values:

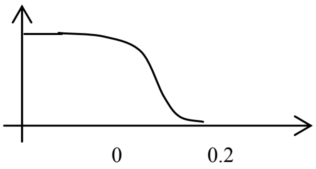
$$FD = \frac{2 \ln(0{,}25 \cdot perimeter)}{\ln area}$$



The potential increasing value of forest patches of more complex patches was modeled with a linear increasing fuzzy membership function applied within the fractal dimensions potential minimum (1) and maximum (2).

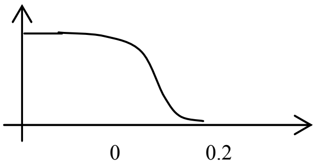
Disturbance by roads and settlements *(per pixel)*

On the basis of urban areas a distance buffer map of 250 m was created (see § 3.1.3) in order to prudentially take into account the effect on all the different animals species. On the basis of a circular moving window of 5 km radius the percent amount of land impacted by anthropogenic disturbances was calculated for each forest pixel.



Disturbance by forest fires *(per pixel)*

On the basis of a circular moving window (radius of 5 km) filter yearly forest fires maps from EFFIS were elaborated. For each year and for each forest pixel the percentage of the landscape (circle of 5 km radius) effected by a forest fire was calculated calculated.



Yearly maps were aggregated in order to evaluate, for each forest pixel, the average percentage of the landscape (circle of 5 km radius*) effected by a forest fire in the time period for which the EFFIS service was available.

Such raster map was analyzed on the basis of a logistic decreasing fuzzy membership function within the 20% and 0% of disturbed area.

Vegetation naturalness *(per pixel)*

Forest/land use maps were contrasted with potential vegetation maps in order to assess if the main dominant forest categories (EEA, 2006) are natural for the study areas. For each study area the ecological distance from real to potential vegetation will be calculated and the inverse of the ecological distance assumed as the level of biological functionality. The ecological distance between real and potential vegetation will be based on the application of the phytosociological hierarchical system approach (vegetation series, Blasi et al. 2000). Potential vegetation maps were provided by JRC on the basis of the project "*Forest habitat suitability distribution and future trends under Climate Change*" (available at http://forest.jrc.it/Forest_and_Climate_Change/Forest_Trends/).

Unfortunately the coarse resolution of the maps do not make possible a specific analysis based on the full system of nomenclature for European Forest Categories (EEA, 2006) so the analysis was mainly carried out on the main land cover.

The cross-tabulating matrix adopted is based on a quantitative value of FVRR between 0 and 60. These values were then linearly rescaled in the usual 8-bit interval to make it comparable with the other factors.

Real \ Potential	3 - Alpine coniferous forest	6 - Beech forest	7 - Montane beech forest	8 - Thermophilous deciduous forest	10 - Coniferous forests
7 - Montane beech forest	60	60	60	60	60
8 - Thermophilous deciduous forest	60	60	60	60	60
9 - Broadleaved evergreen forest	60	60	60	60	60
10 - Coniferous forests	60	60	60	60	60
12 - Floodplain forest	60	60	60	60	60
14 - Plantations	30	30	30	30	30
15 - Other wooded lands	45	45	45	45	45
16 - Other natural lands (Pastures)	40	40	40	40	40
17 - Urban fabric	0	0	0	0	0
18 - Water bodies	0	0	0	0	0
19 - Inlands marches	0	0	0	0	0
20 - Agricultural	15	15	15	15	15

Table 4: cross-tabulation matrix used to reclassify the combination between potential and real land use land cover. The values of FVRR are in the range 0-60.

7.1.2 Factors aggregation

The input factors were aggregated on the basis of the fuzzy MCE approach. After several reiterative tests the MCE was based on the *Weight Linear Combination* with equal weights for all the factors. Since the interpretation of the results is always carried out in a comparative way (different resolutions, different years, different test areas) the relative differences never vary if all the tests are always carried out with the same methods.

7.2 Species specific models

On the basis of the bibliographic review (§ 3.5) five different umbrella species were selected, for this project. The approach followed for species specific models is similar to that one applied in the unspecific approach of the previous chapter.

The analysis produced five different spatial analysis showing the biological distance between real habitat environmental condition and optimal habitat condition. The method is based on the use of fuzzy membership functions applied to a number of environmental factors defined on the basis of the ecological requirements of the five umbrella species. The membership functions related ecological factors to optimal conditions in order to produce a number of normalized maps of Habitat Suitability. The set of selected ecological factors and fuzzy membership functions for each umbrella species are defined as *ecological profiles* or *ecoprofiles*. All resulting maps for each environmental factor and for each species will be all expressed, as for the unspecific approach, in the 8 bit range range 0 – 255. A value equal to 0 means that the site (the pixel) is very far away from optimal ecologic condition while a value of 255 express that the pixel is optimal for the selected species.

Normalized suitability maps for each ecological factor were aggregated in order to produce one Habitat Suitability map for each of the selected umbrella species on the basis of the fuzzy MCE procedure.

The five different maps were then aggregated to produce an overall value (see the following chapter for the aggregation rules) that was compared for the different resolutions, different test areas and different study years also with species unspecific model.

The general approach is based on the thesis that the extension and the spatial distribution of areas where the overall ecological functionality of the habitat is high (minimum distance from the optimum for the five umbrella species) are proxy variables of the potential disturbances and the possibility of the habitat to resist and to recover after disturbances.

7.2.1 Ecoprofiles

Please note that since ecoprofiles are based on real ecological species requirements the spatial thresholds cannot be directly applied because of the coarse resolution of the input layers. In these cases the closest feasible value was adopted.

Marten (Martes martes)

On the basis of the ecological information available for marten and taking into consideration the real availability of data in the study the ecoprofile for marten is the following.

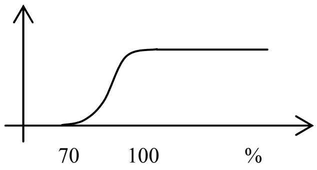
Definition of core area: at least 100 m from the forest patch border (settlements, roads and railways will be considered as interruptions).

Type of forest

Martens prefer old growth or mature coniferous forests. Especially in winter, the use of broadleaf forest is limited. Coniferous HIS will be equal to 1, broadleaf forest will be 0.5.

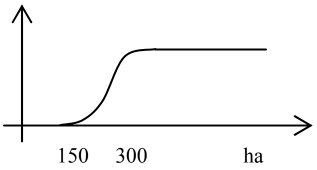
Forest core area in the home range (per pixel)

On the basis of a circular moving window of 4 km^2 (home range) the percentage of core forest will be calculated on the total land. A logistic increasing fuzzy membership function will be used with a minimum threshold of 70% of core forest in the dispersal range and an optimum of 100%.



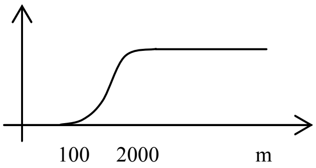
Forest patch size (per patch)

A logistic increasing fuzzy membership function will be applied on the basis of forest patch size with a minimum of 150 ha and an optimum of 300 ha.



Anthropogenic disturbances (per pixel)

On the basis of the distance from settlements, urban areas, roads and railways a logistic increasing function with a minimum of 100 m and an optimum at 2 km.

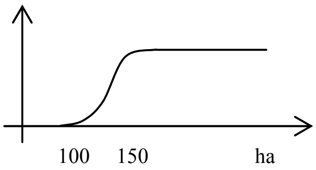


Roe deer (Capreolus capreolus)

Definition of core area: at least 100 m from the forest patch border (settlements, roads and railways will be considered as interruptions).

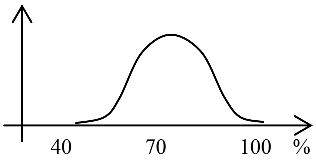
Forest patch size (per patch)

A logistic increasing fuzzy membership function will be applied on the basis of forest patch size with a minimum of 100 ha and an optimum of 150 ha.



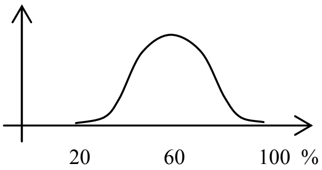
Forest core area in the home range (per pixel)

On the basis of a circular moving window of 10 km^2 (home range) the percentage of core forest will be calculated on the total land. A gaussian fuzzy membership function will be used with a minimum threshold of 40% of core forest in the dispersal range an optimum at 70% and a second minimum of 100%.



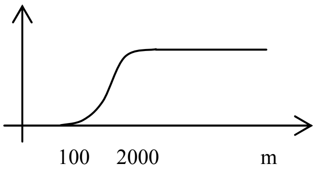
Forest with open grassland/agriculture in the home range (per pixel)

On the basis of a circular moving window of 10 km^2 (home range) the percentage of agricultural/pasture areas will be calculated for each forest pixel. A gaussian fuzzy membership function will be used with a minimum threshold of 20% of core forest in the dispersal range an optimum at 60% and a second minimum of 100%.



Anthropogenic disturbances (per pixel)

On the basis of the distance from settlements, urban areas, roads and railways a logistic increasing function with a minimum of 100 m and an optimum at 2 km.

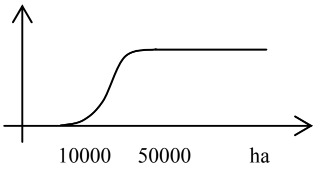


Wolf (Canis lupus)

Core area set at 1 km. Settlements, roads and railways will be considered as interruptions.

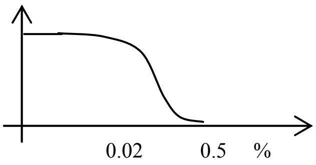
Forest patch size (per patch)

A logistic increasing fuzzy membership function will be applied on the basis of forest patch size with a minimum of 10000 ha and an optimum of 50000 ha.



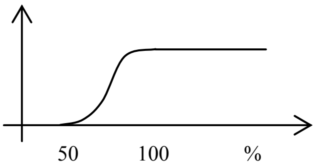
Anthropogenic disturbances (per pixel)

On the basis of the percent presence of settlements, urban areas, roads and railways in a moving window of 35 km^2 a logistic decreasing function will be applied with an optimal minimum of 0.02% and a maximum of 0.5%.



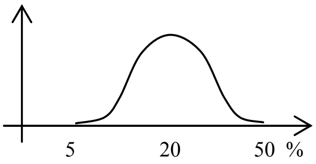
Forest core area in the home range (per pixel)

On the basis of a circular moving window of 35 km^2 (home range) the percentage of core forest will be calculated on the total land. A logistic increasing fuzzy membership function will be used with a minimum threshold of 50% of core forest in the dispersal range an optimum at 100%.



Forest with open grassland/agriculture in the home range (per pixel)

On the basis of a circular moving window of 35 km^2 the percentage of agricultural/pasture areas will be calculated for each forest pixel. A gaussian fuzzy membership function will be used with a minimum threshold of 5% of open areas in the range an optimum at 20% and a second minimum of 50%.

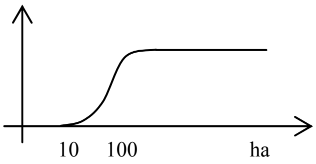


Birds (Picoides minor)

Core area set at 80 m. Deciduous Forest only. Settlements, roads and railways will be considered as interruptions.

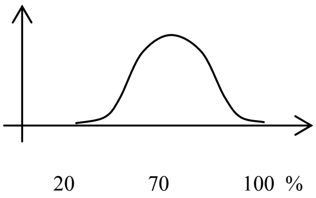
Forest patch size (per patch)

A logistic increasing fuzzy membership function will be applied on the basis of forest patch size with a minimum of 10 ha and an optimum of 100 ha.



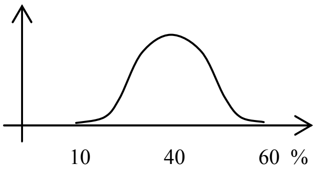
Forest core area in the home range (per pixel)

On the basis of a circular moving window of 2 km^2 (home range) the percentage of core forest will be calculated on the total land. A gaussian fuzzy membership function will be used with a minimum threshold of 20% of core forest in the dispersal range an optimum at 70% and a second minimum of 100%.



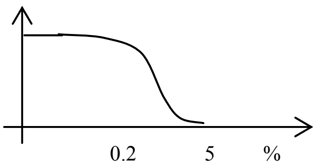
Forest with open grassland/agriculture in the home range (per pixel)

On the basis of a circular moving window of 2 km^2 the percentage of agricultural/pasture areas will be calculated for each forest pixel. A gaussian fuzzy membership function will be used with a minimum threshold of 10% of open areas in the range an optimum at 40% and a second minimum of 60%.



Anthropogenic disturbances (per pixel)

On the basis of the percent of buffer zone of 100 m around settlements, urban areas, roads and railways in a moving window of 2 km^2 a logistic decreasing function will be applied with an optimal minimum of 0.2% and a maximum of 5%.



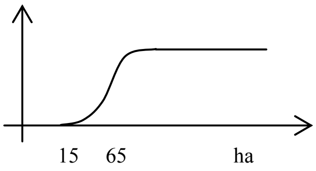
Distance from human activities (per pixel)
Buffer area of 100 around settlements, urban areas, roads and railways is not suitable.

Butterflies (Apatura ilia)

Forest core area set to 25 m. Settlements, roads and railways were considered as interruptions.

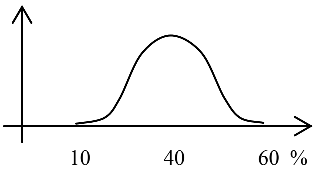
Forest patch size (per patch)

A logistic increasing fuzzy membership function will be applied on the basis of forest patch size with a minimum of 15 ha and an optimum of 65 ha.



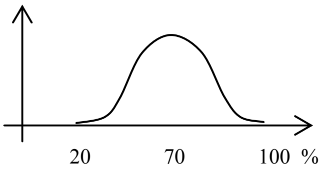
Forest with open grassland/agriculture in the home range (per pixel)

On the basis of a circular moving window of 0.8 km^2 the percentage of agricultural/pasture areas will be calculated for each forest pixel. A gaussian fuzzy membership function will be used with a minimum threshold of 10% of open areas in the range an optimum at 40% and a second minimum of 60%.



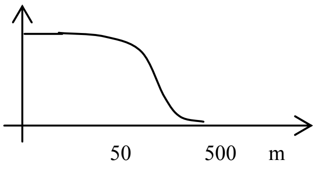
Forest core area in the home range (per pixel)

On the basis of a circular moving window of 0.8 km^2 the percentage of core forest will be calculated on the total land. A gaussian fuzzy membership function will be used with a minimum threshold of 20% of core forest in the dispersal range an optimum at 70% and a second minimum of 100%.



Anthropogenic disturbances (per pixel)

On the basis of the distance from settlements, urban areas, roads and railways a logistic decreasing function with a minimum of 50 m and an optimum at 500 m.



7.2.2 Factors aggregation

As for the species unspecific model after a reiterative test carried out in the pilot test area the input factors were aggregated for the five umbrella species by fuzzy *Weighted Linear Combination*.

Please note that the resulting maps for the five umbrella species are used, as for the species unspecific approach, to evaluate the spatio-temporal trends of forest spatial pattern in the selected test areas. They do not express or they are not related to the real or potential presence of the five selected species.

8 Results

The methodology adopted to present the results of the project is based on the following steps.

Once that all the FVRR maps resulting from the species specific and species unspecific models based both on high and low resolutions maps were ready the average values of core and non-core pixels from the GUIDOS maps were calculated together with their relative standard error. Averaged FVRR statistics are therefore stratified in three different classes: forest area, core forest area, non-core forest area. In the analysis non-core areas are all the resulting classes from MSPA analysis not belonging to the core class.

Following the same approach the total value of the FVRR indexes was also calculated summing up all the pixels once again stratified in the same three classes as above.

While the *average* statistics is used to discuss about the theoretical relationships between the models adopted, the scale of the analysis (high vs. low resolution maps) and the type of forest areas (core vs. no core), the *sum* statistics are used to describe the temporal trends since they are influenced both by the change in the forest area and the change in the results of the FVRR models.

Since some of the input factors used in the FVRR models are based on moving window filters the results are comparable just in those areas fully inside the window kernel. Statistics are therefore extracted just in a buffer area inside the study area. All the maps presented in this chapter report both the boundary of the study area and the internal boundary of the buffered zone used in the extraction of the statistics.

The final aim of the analysis is to support the discussion about the possible use of a FVRR species unspecific model versus more complex species specific models. The scale dependency of FVRR indexes and their spatial and temporal trend in the different study areas used in the project (Figure 10).

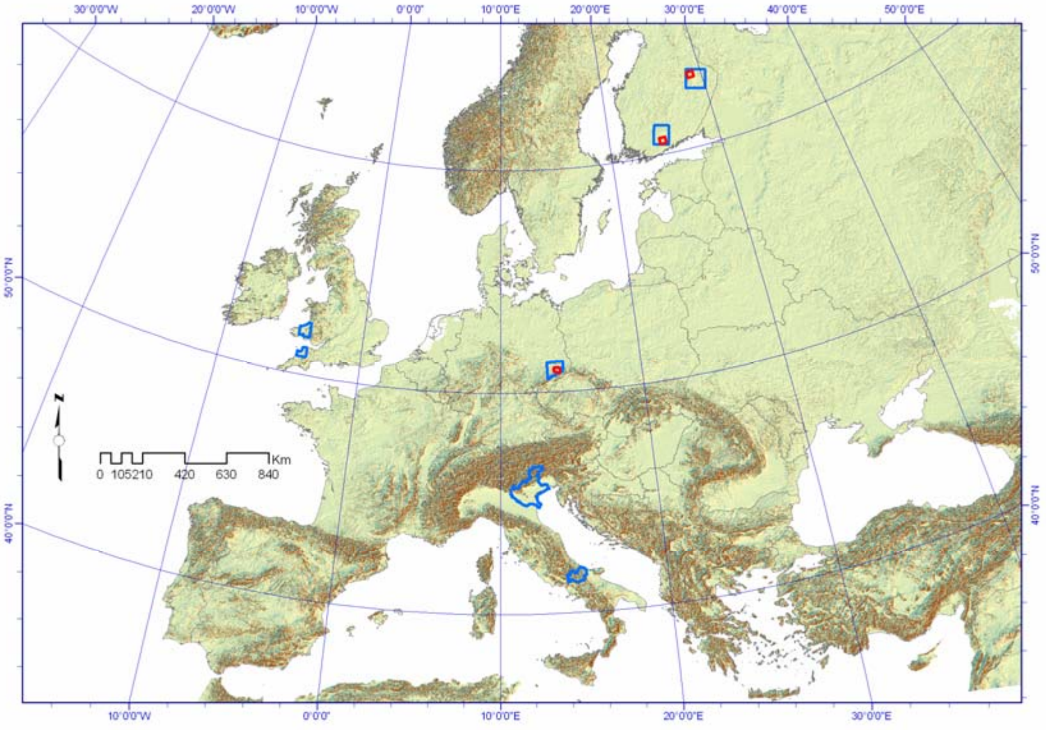


Figure 10: location of the test areas. LR areas in blue, HR areas in red (when they differ).

8.1 Mediterranean test area

The Mediterranean test area has the higher availability of data: four dates for high resolution maps (1936, 1954, 1992 and 2005) and three dates for low resolution maps (1980, 1990 and 2000). It was used as a pilot study area for the development of the models and for this reason the data and the discussion for this area are more complete.

The original test area correspond to the administrative area of the Regione Molise 4441 km^2 wide, because of the effect of the moving window filters used in the analysis data are fully comparable just in a smaller area. The study area is the same for both high and low resolution maps.

The area is interested by a nearly linear increasing trend of forest area. On the basis of high resolution maps forest augmented in the period 1936-1954 of the 3.4% yearly, of the 1.9% in 1954-1992 and of the 2.2% in the period 1992-2005. Low resolution maps confirm a similar trend with an yearly increase in forest area of the 1.5% in the period 1980-1990 and of 2.0% in the period 1990-2000.

As a result the forest area, on the basis of high resolution data, passed from 239 km^2 in 1936 to 860 km^2, an increase of 260%.

Low resolution data tend to underestimate forest area if compared with high resolution maps. The two dataset are not fully comparable because none of the high/low resolution maps are acquired at the same date. The difference can be estimated around 5000 ha. Considering high resolution maps more accurate than low resolution ones, the low resolution maps underestimate forest area of approximately 9% (Figure 11).

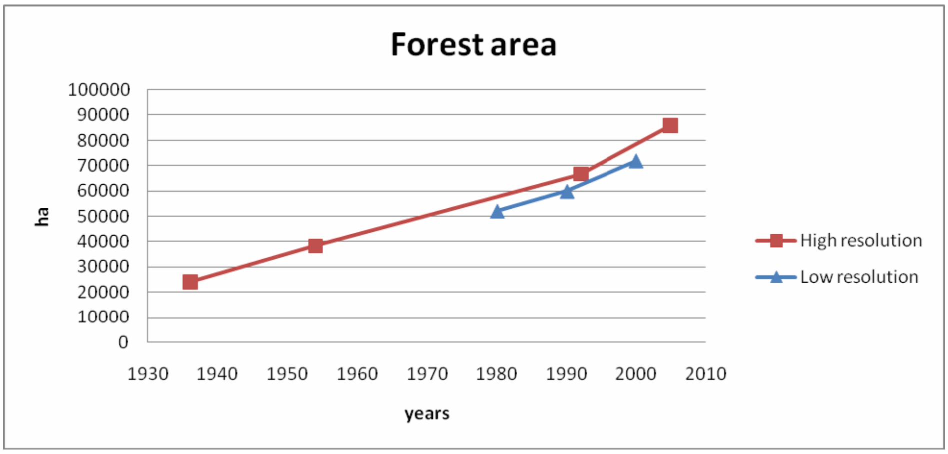
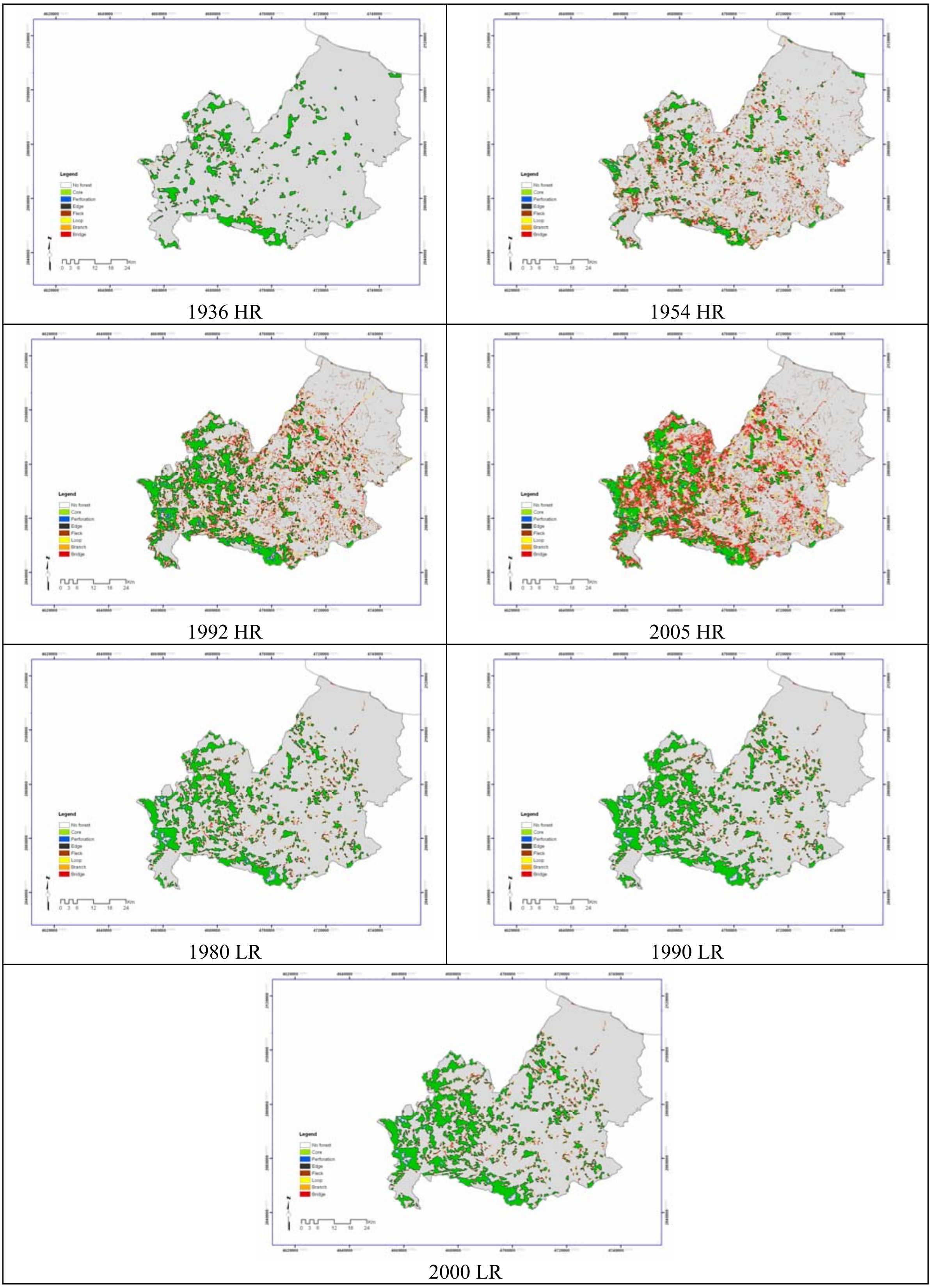


Figure 11: comparison of trends in forest area comparing low and high resolution data.



1936 HR

1954 HR

1992 HR

2005 HR

1980 LR

1990 LR

2000 LR

Figure 12: result of the GUIDOS analysis on multitemporal low (LR) and high resolution (HR) forest maps.

The trends in forest area are slightly different if core and no core areas are considered separately. The trends in high and low resolution maps for no core areas are similar to the general forest area trend while for core areas low resolution maps tend to overestimate forest area. As a result the ratio of core and no core areas is strongly affected by the maps resolutions (Figure 13).

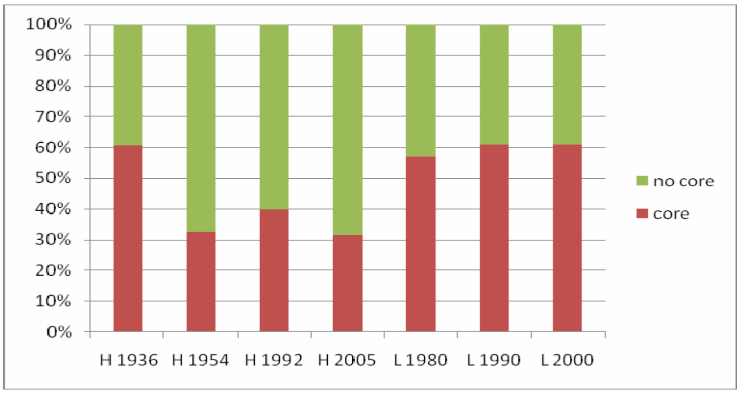


Figure 13: Relative area of core and no core forest areas in the different maps available in the Mediterranean test area (H for high resolution and L for low resolution).

As expected on the basis of the high resolution maps the positive trend of forest area is prevalently due to the positive trend of no core areas (Figure 14). The forest area in Regione Molise is in fact increasing meanly because of the abandon of marginal agricultural and pasture areas in mountain regions. These new forest areas are prevalently no core areas.

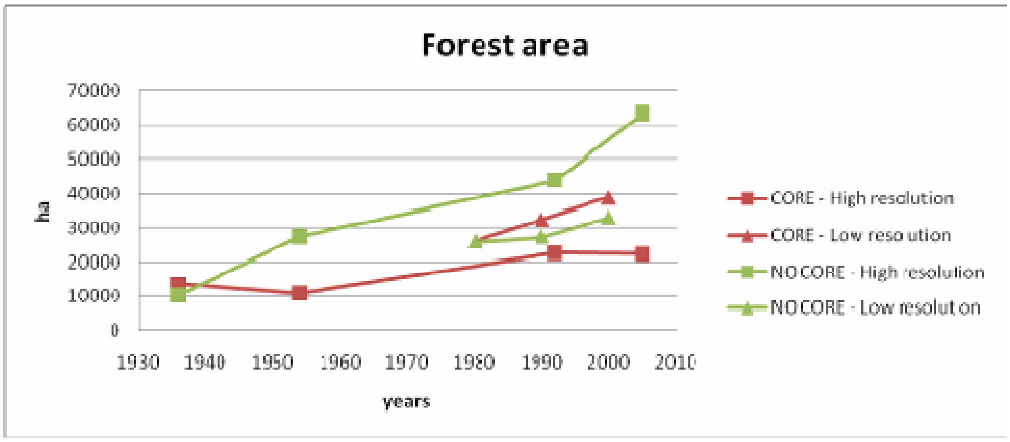


Figure 14: comparison of trends in forest area comparing low and high resolution data for core and no core forest areas.

8.1.1 Species unspecific analysis

The results of the application of the species unspecific model to high and low resolution maps are in Figure 15. In the Mediterranean test area the FVRR index is confirmed to be not scale dependent since the temporal trends of low and high resolution maps is nearly the same.

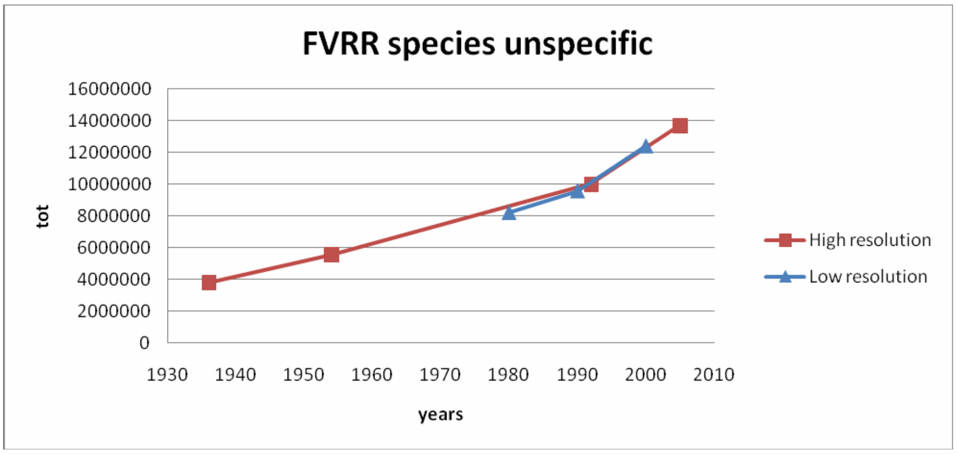


Figure 15: comparison of trends in species unspecific index of FVRR comparing low and high resolution data in forest areas.

As expected the average value of the index FVRR is higher in core areas than in no core areas (Figure 16) for all the study years and for all the adopted resolutions.

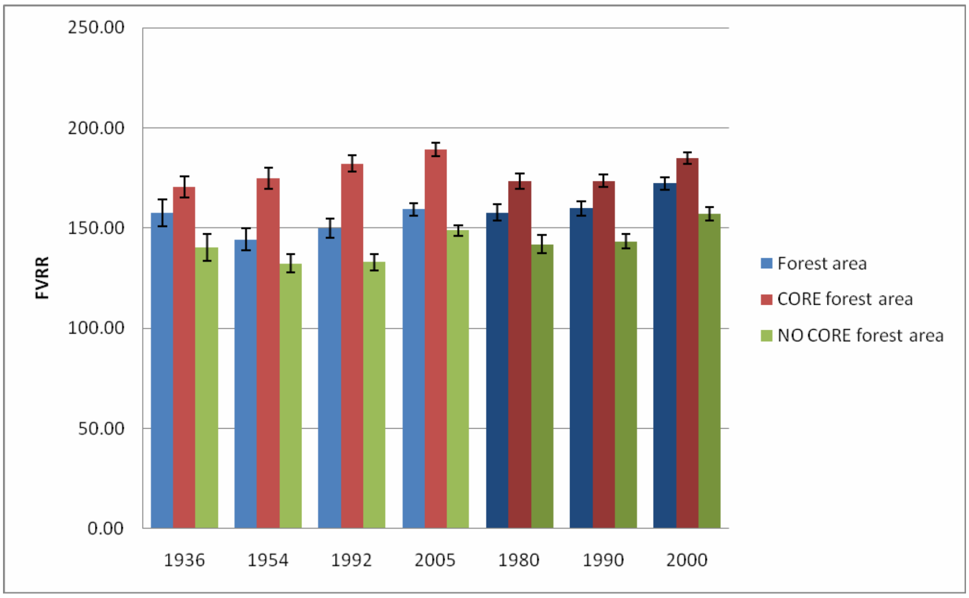


Figure 16: average values with standard error of the species unspecific index of FVRR comparing low and high resolution data for core and no core forest areas. Data from 1936, 1954, 1992 and 2005 are high resolution (brighter), data from 1980, 1990, 2000 are low resolution (darker).

When observing the overall trend of the index FVRR in the high resolution maps the values in core areas are lower than in no core areas. This can be explained looking at the very strong presence of no core areas in this test (Figure 17). Looking at low resolution maps the effect is the opposite and core areas has an overall FVRR value higher than in no core areas.

In core areas (at both the resolutions) the temporal trend is weaker than in no core areas that demonstrate instead a very strong change rate in time (always increasing).

The trend of the FVRR index in Figure 17 is due to the cumulative effect of the increasing forest area (expecially of no core areas, at least in high resolution maps) and of the average values of the index

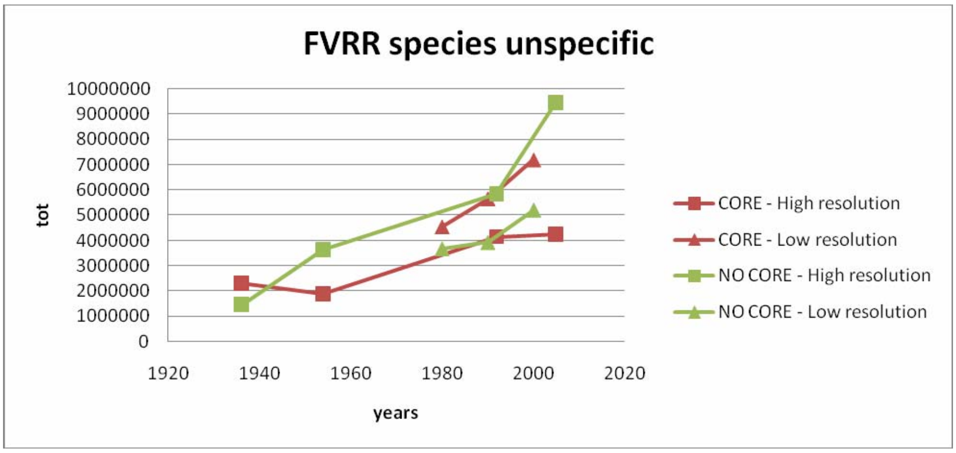
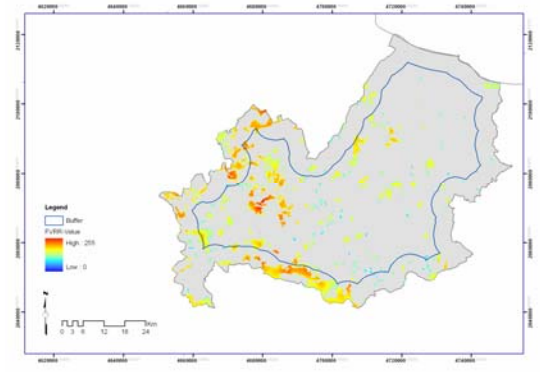
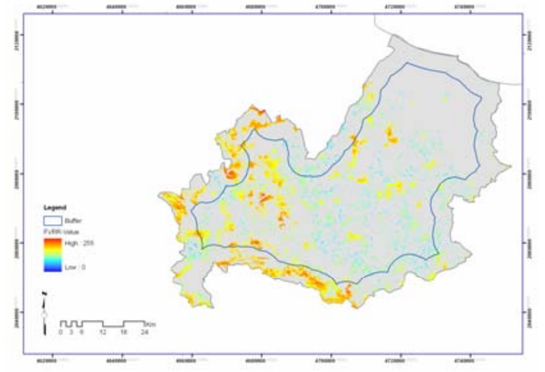
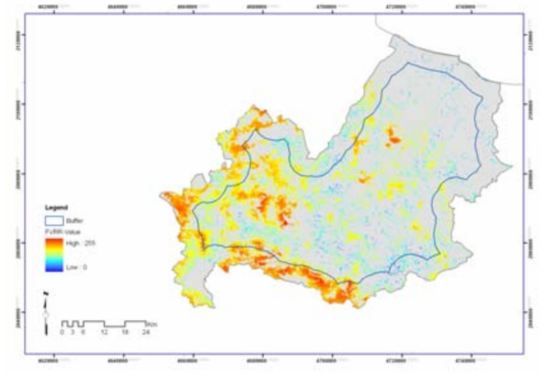
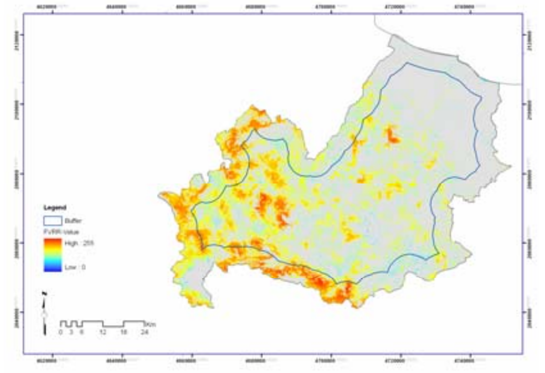
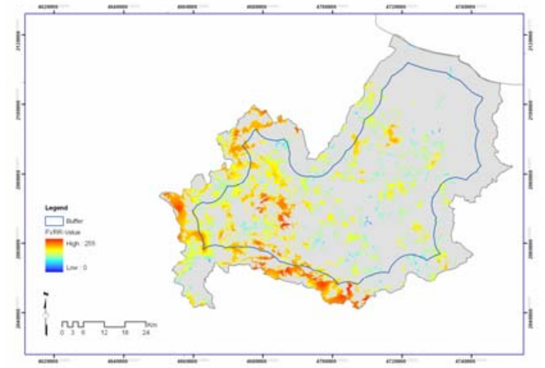
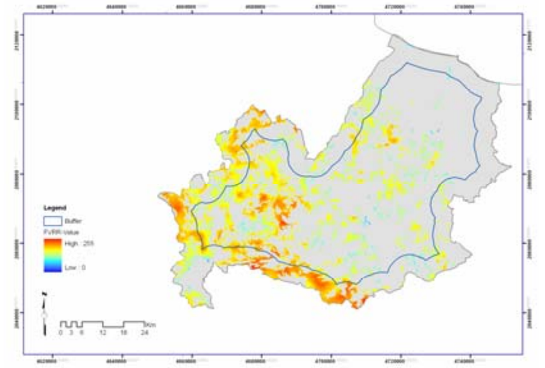


Figure 17: comparison of trends in species unspecific index of FVRR comparing low and high resolution data for core and no core forest areas.

 1936 HR	 1954 HR
 1992 HR	 2005 HR
	

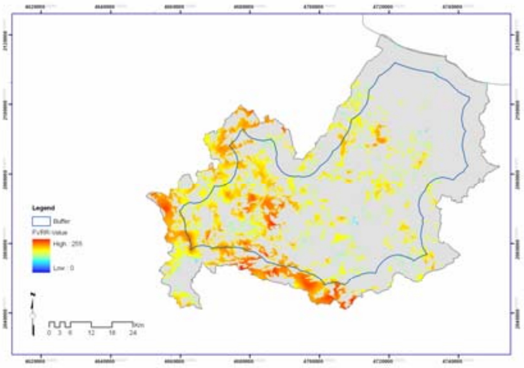
1980 LR	1990 LR
 2000 LR	

Figure 18: the index FVRR based on multitemporal high (HR) and low resolution (LR) forest maps.

8.1.2 Species specific analysis

The results of the index FVRR was calculated for birds (Figure 32), butterflies (Figure 33), marten (Figure 34), roe deer (Figure 35) and wolf (Figure 36). The five species were also aggregated summing up all the indexes (Figure 37).

As expected from the theoretical ecological background for birds the average value of FVRR in core areas is always much greater than in no core areas (Figure 19) in both high and low resolution maps.

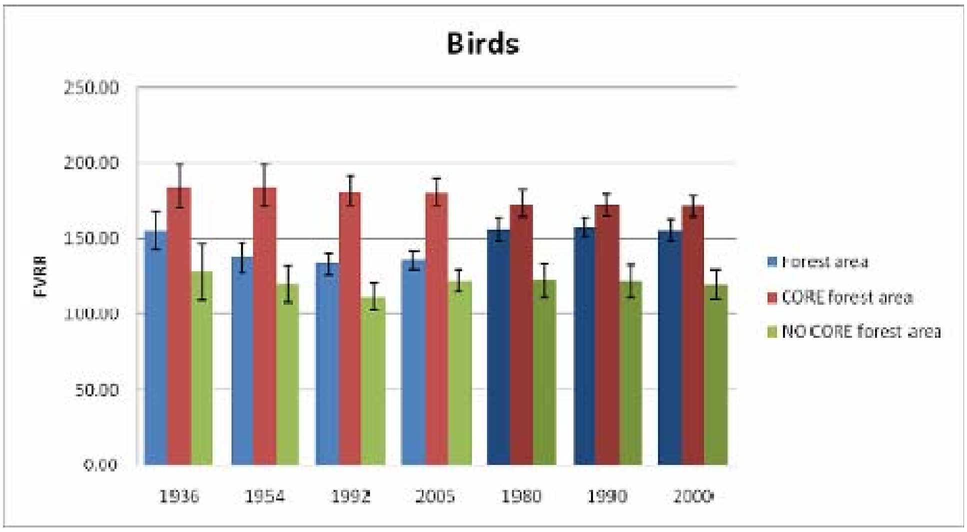


Figure 19: average values with standard error of the birds index of FVRR comparing low and high resolution data for core and no core forest areas. Data from 1936, 1954, 1992 and 2005 are high resolution (brighter), data from 1980, 1990, 2000 are low resolution (darker).

The temporal trend of FVRR for birds is always increasing. In high resolution maps the trend in core and no core areas is almost the same with the exception of 2005 when the great augment area in foresst no core areas determined higher values of the bird index in this type of areas than in core areas. In low resolution maps the differences between the two resolutions are different. Since the area of forest no core is lower than in high resolution maps, the higher average value of core area determined a higher value in these areas (Figure 20).

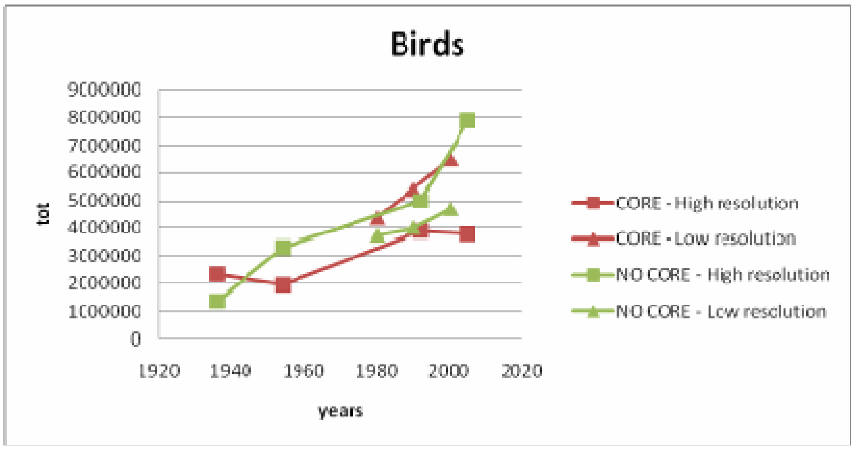


Figure 20: comparison of trends in birds index of FVRR comparing low and high resolution data for core and no core forest areas.

For butterflies the relationship between the FVRR index and forest area type (core and no core) is not linear, also demonstrated by the higher variability of the index within the different forest area types (Figure 34). In high resolution maps such a variability tend to decrease moving from 1954 to 2005. In general in high resolution maps the index is higher in no core areas while in low resolution maps the index is greater in no core areas.

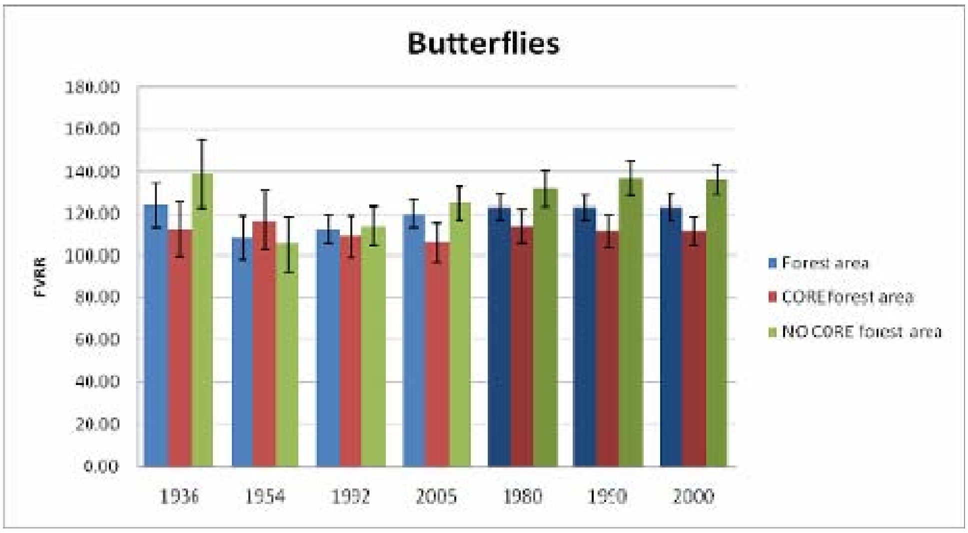


Figure 21: average values with standard error of the butterflie index of FVRR comparing low and high resolution data for core and no core forest areas. Data from 1936, 1954, 1992 and 2005 are high resolution (brighter), data from 1980, 1990, 2000 are low resolution (darker).

The temporal trends of the butterflies index clearly shows higher values of the FVRR index in forest no core area both in high and low resolution maps. In core areas the index has a stronger incresing rate moving from 1936 to 2005 than in no core areas. No core areas seems therefore to have ecological conditions closer to the potential optimum for the selected umbrella species.

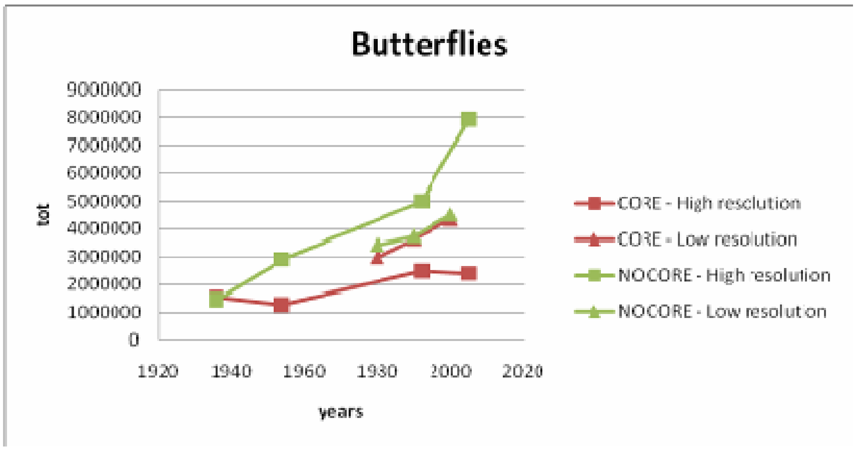


Figure 22: comparison of trends in butterflies index of FVRR comparing low and high resolution data for core and no core forest areas.

For marten the average values of the FVRR index (Figure 23) in core areas is always higher clearly connected with the ecological preferences of this umbrella species. The differences in the average values between forest classes (core and non core) and their internal variability are similar for all the considered study years and resolution adopted.

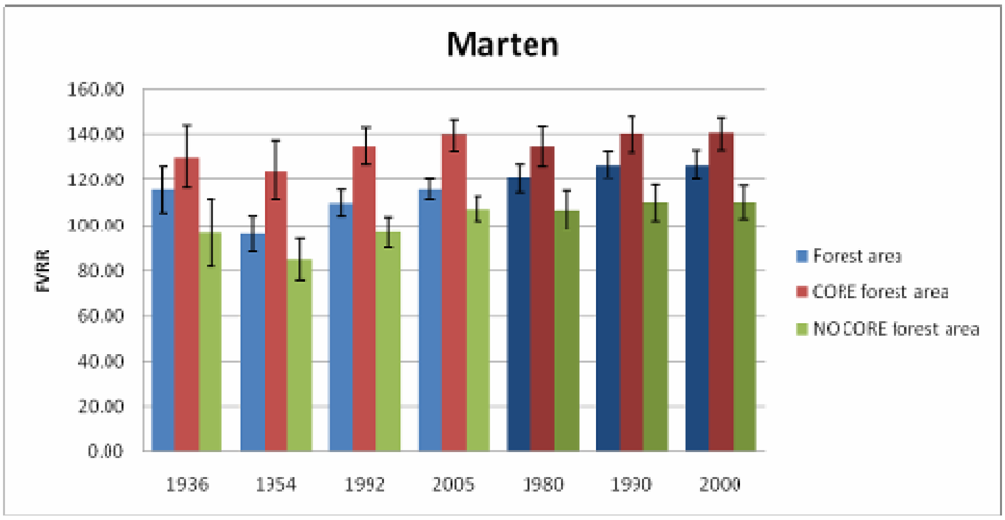


Figure 23: average values with standard error of the marten index of FVRR comparing low and high resolution data for core and no core forest areas. Data from 1936, 1954, 1992 and 2005 are high resolution (brighter), data from 1980, 1990, 2000 are low resolution (darker).

The temporal trend of the marten FVRR index (Figure 24) in the Mediterranean study area is always increasing, despite of the resolution adopted. The trend is very similar to that one of the birds FVRR inde: for high resolution maps trends are similar with the exception of the year 2005 when the strong increase in forest no core area determined a conseguente increase rate of the index in these areas from 1992 to 2005. In low resolution maps the values in core areas are always higher because of the limited spatial extension of no core areas.

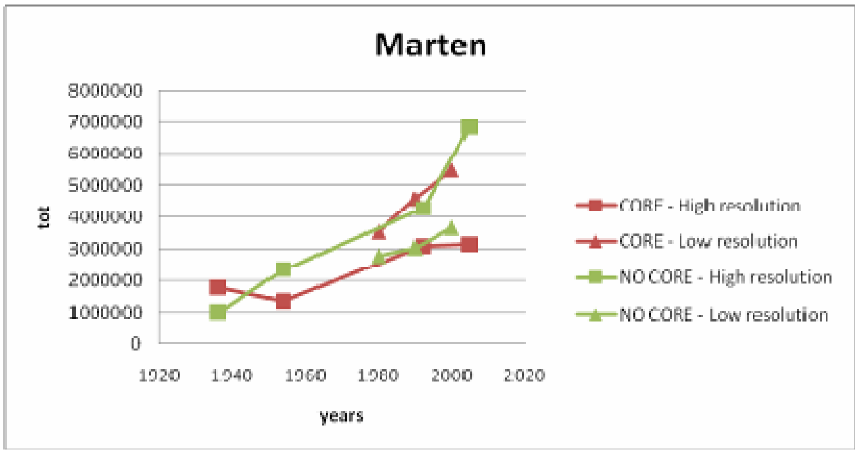


Figure 24: comparison of trends in marten index of FVRR comparing low and high resolution data for core and no core forest areas.

Accordingly with the ecoprofiles adopted in the modeling procedure the average values of the roe deer FVRR index are always higher in core areas than in no core areas for all the study years and for all the considered resolutions (Figure 25).

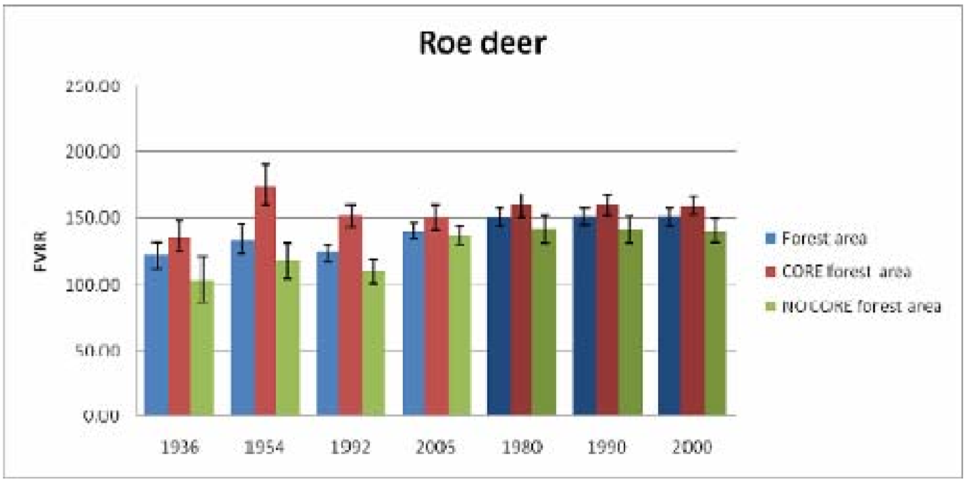


Figure 25: average values with standard error of the roe deer index of FVRR comparing low and high resolution data for core and no core forest areas. Data from 1936, 1954, 1992 and 2005 are high resolution (brighter), data from 1980, 1990, 2000 are low resolution (darker).

The temporal trends of the roe deer FVRR index (Figure 26) is similar to those ones of birds and of marten, the ecological considerations already done for those species are valid therefor also for roe deer.

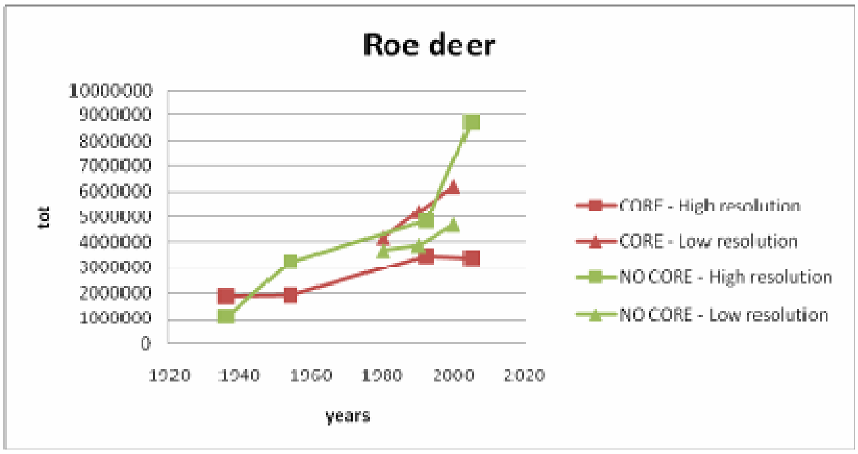


Figure 26: comparison of trends in roe deer index of FVRR comparing low and high resolution data for core and no core forest areas.

The wolf FVRR index is based on an ecoprofile different from all the others. Since the wolf is very demanding for large undisturbed continuous core areas the values of the index in the years 1936, and 1954 when the forest area was still very limited (Figure 27). In the other years the average values of the index are greater in core areas. In core areas from low resolution maps the variability of the index is very high but the values are constantly increasing from 1980 to 2000.

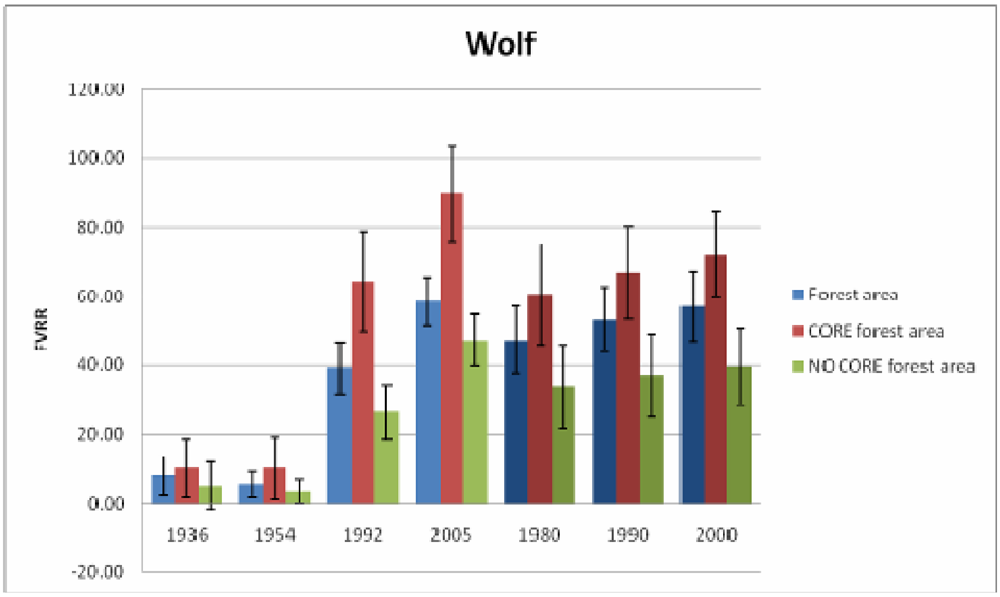


Figure 27: average values with standard error of the wolf index of FVRR comparing low and high resolution data for core and no core forest areas. Data from 1936, 1954, 1992 and 2005 are high resolution (brighter), data from 1980, 1990, 2000 are low resolution (darker).

The temporal trends of the wolf FVRR index (Figure 28) is strongly increasing after the year 1980, the increasing rate is higher in core areas than in no core areas and the trends are similar for high and low resolution data.

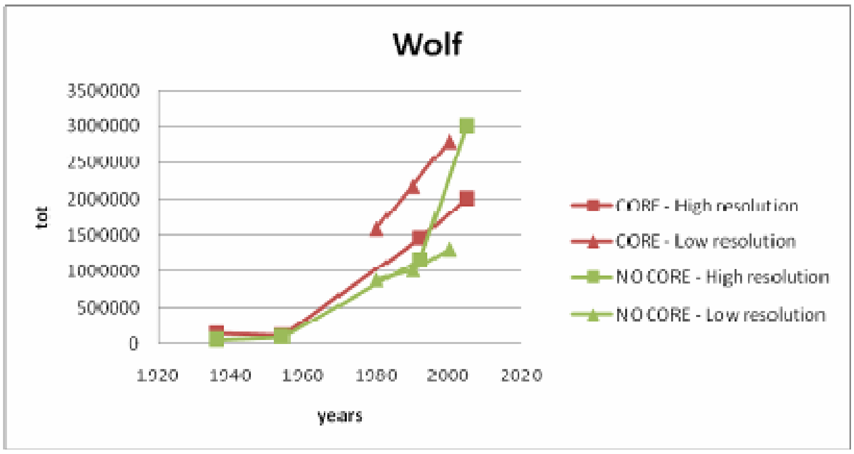


Figure 28: comparison of trends in wolf index of FVRR comparing low and high resolution data for core and no core forest areas.

The results achieved in the Mediterranean test area clearly show that the strong increasing rate of forest cover determined a nearly linear increase for all the considered umbrella species. The FVRR index of wolf, compared to the other species is still lower, since the wolf is very demanding for large undisturbed forest areas. For this reason forest areas area for this umbrella species is more vulnerable and less resilient and resistant to external disturbances. Note that the average trend of all the considered species is also reported in Figure 29 (black lines).

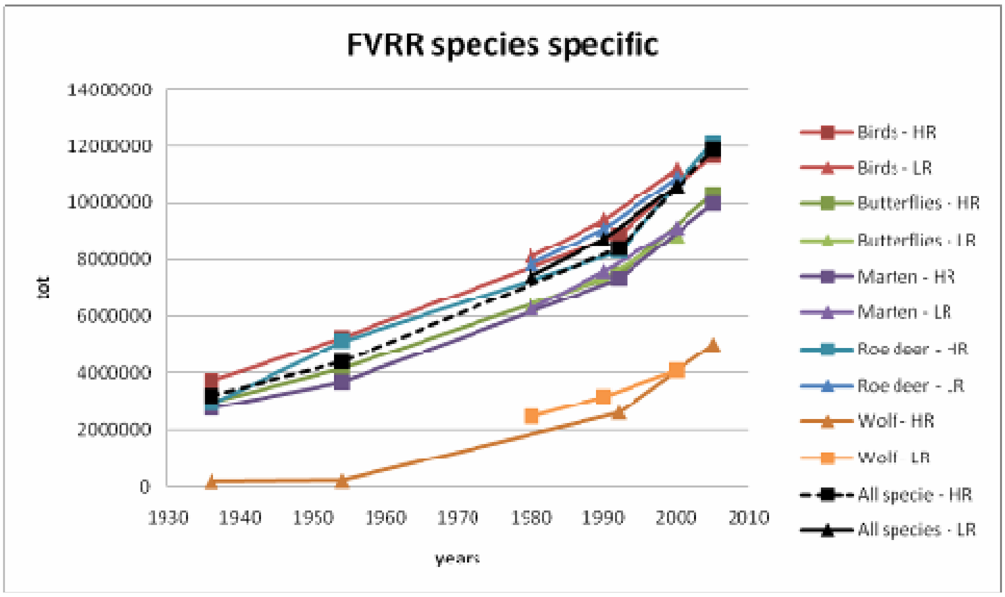


Figure 29: comparison of trends in all the umbrella species index of FVRR comparing low and high resolution data.

The annual average percent change rate of the FVRR index for the considered time periods in the Mediterranean test area for both high and low resolution data of four of the considered species (wolf excluded) demonstrate a linear relationship with the percent change rate in forest area (Figure 30).

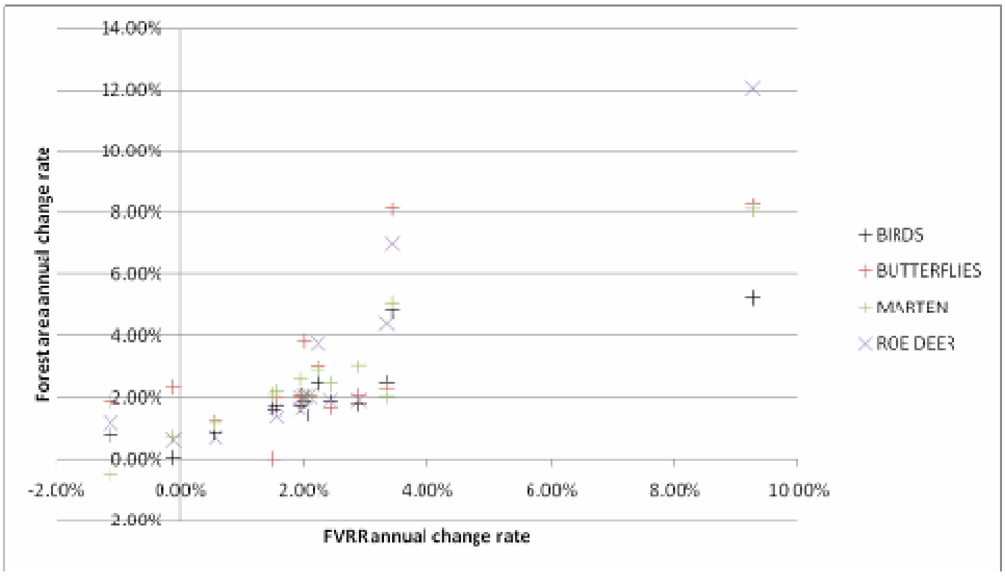


Figure 30: relationship between the annual trend of FVRR for four umbrella species and the annual trend in forest area.

The same umbrella species demonstrate also a close linear relationship with the FVRR species unspecific index (Figure 31).

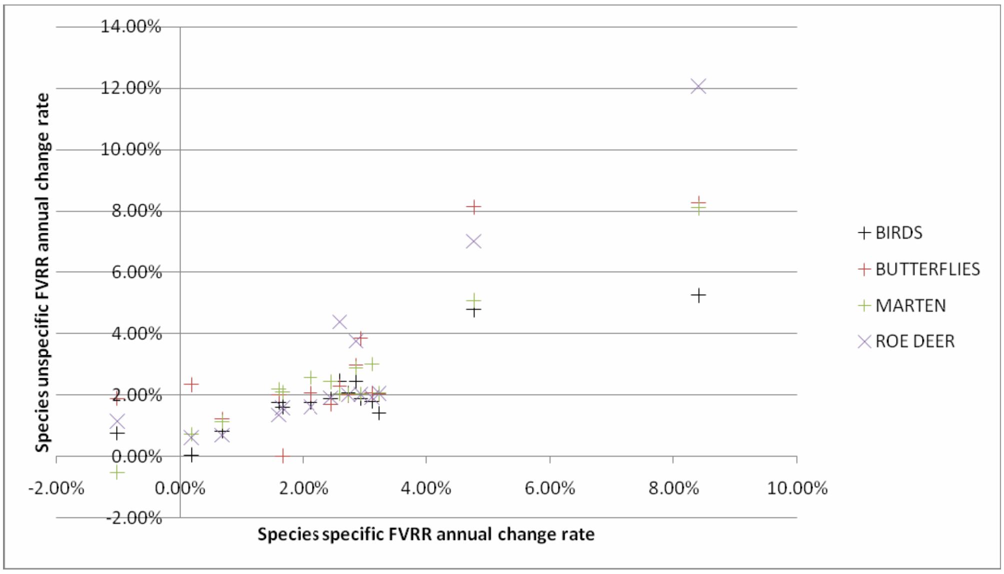
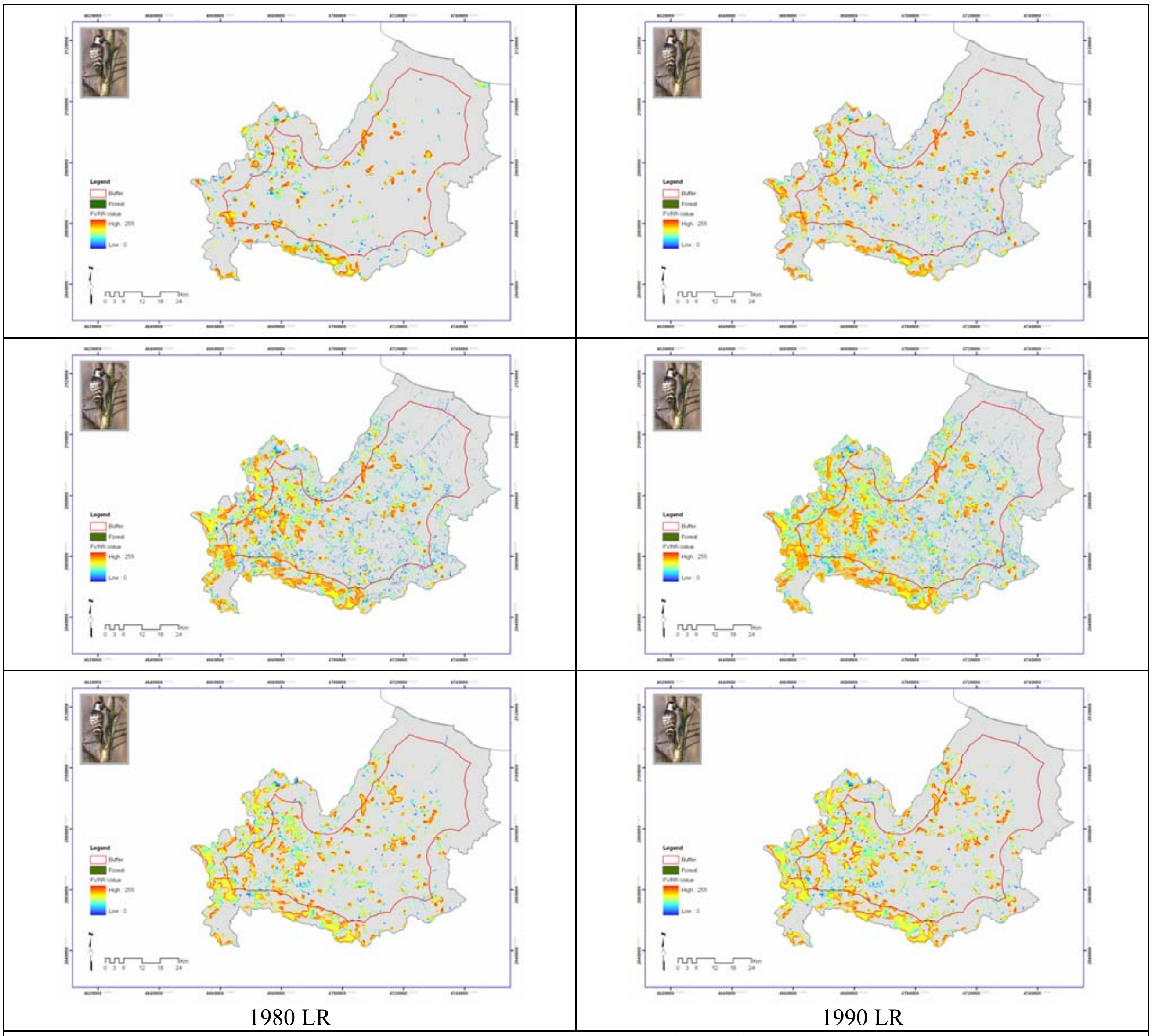
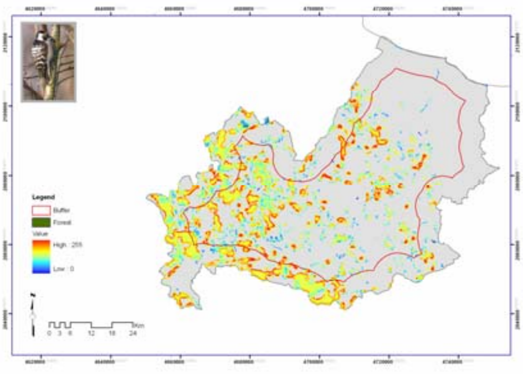


Figure 31: relationship between the annual trend of FVRR for four umbrella species and the annual trend of species unspecific FVRR index.



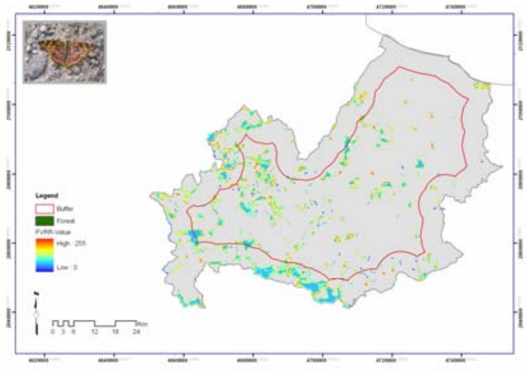
1980 LR

1990 LR

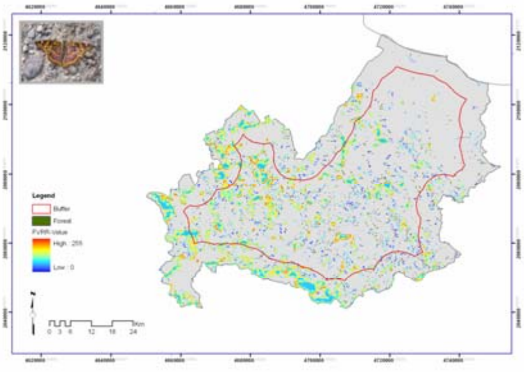


2000 HR

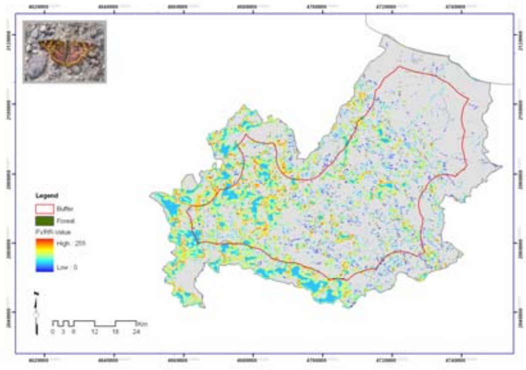
Figure 32: the index FVRR for birds umbrella species based on multitemporal high (HR) and low resolution (LR) forest maps.



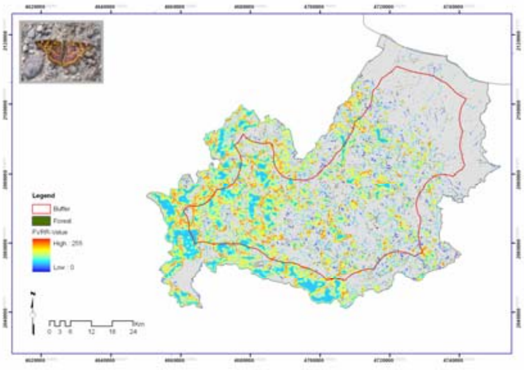
1936 HR



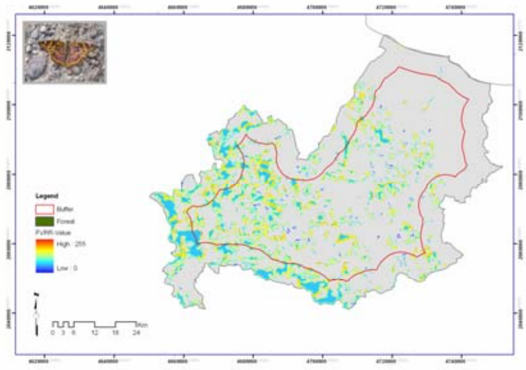
1954 HR



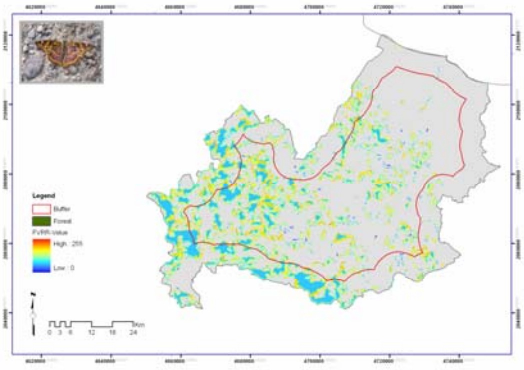
1992 HR



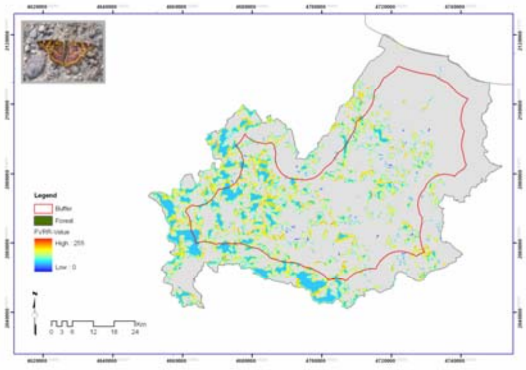
2005 HR



1980 LR

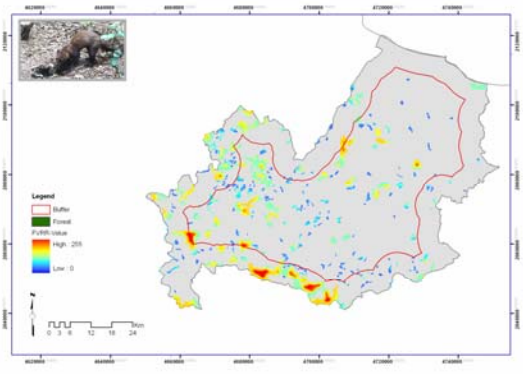


1990 LR

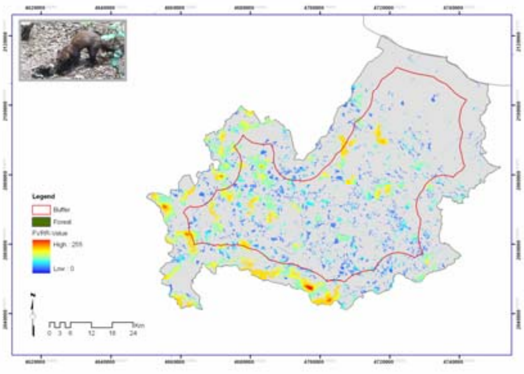


2000 LR

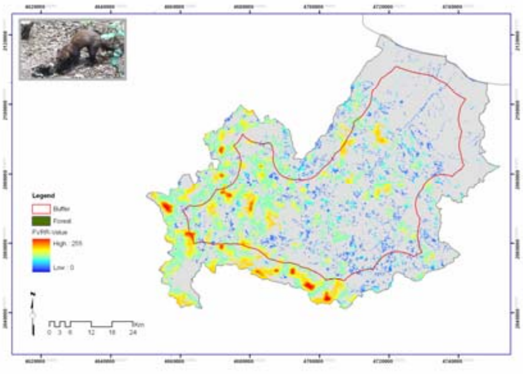
Figure 33: the index FVRR for batterflies umbrella species based on multitemporal high (HR) and low resolution (LR) forest maps.



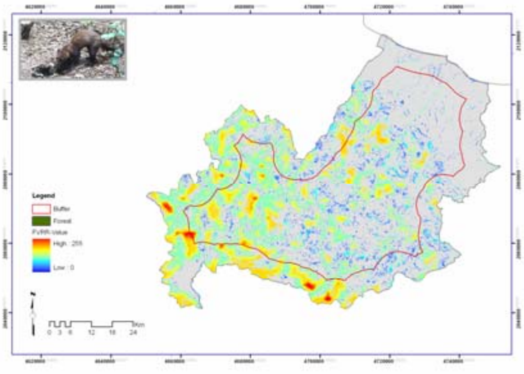
1936 HR



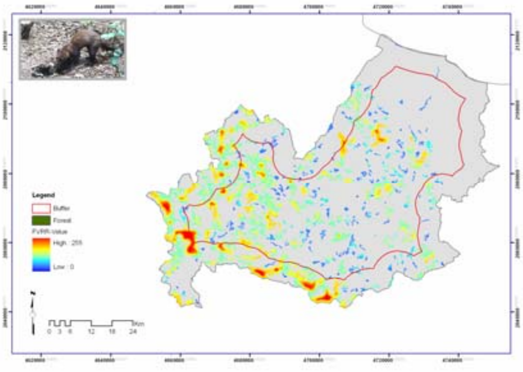
1954 HR



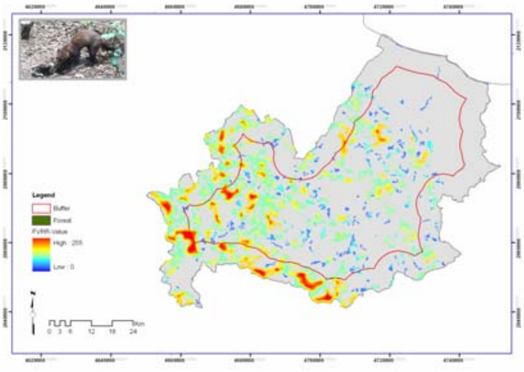
1992 HR



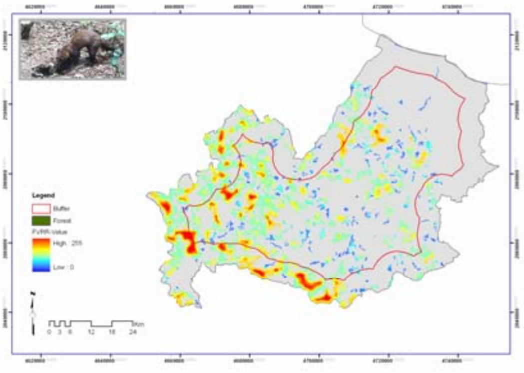
2005 HR



1980 LR

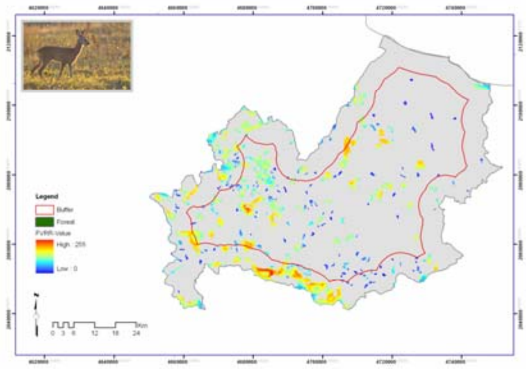


1990 LR

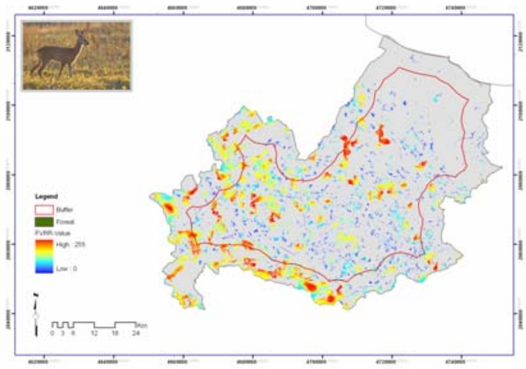


2000 LR

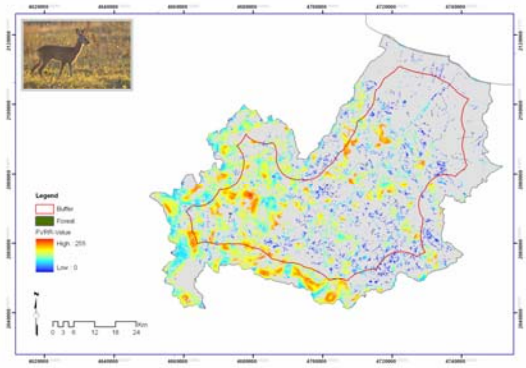
Figure 34: the index FVRR for marten umbrella species based on multitemporal high (HR) and low resolution (LR) forest maps.



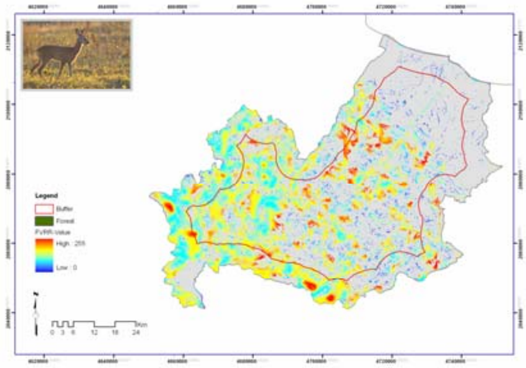
1936 HR



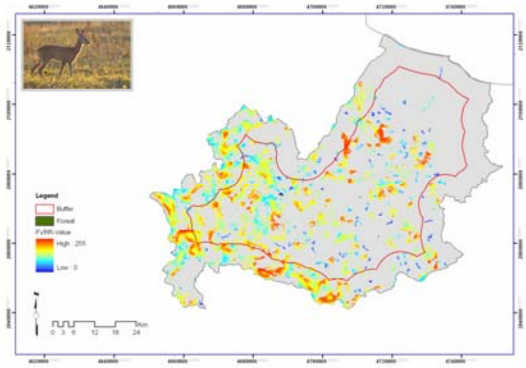
1954 HR



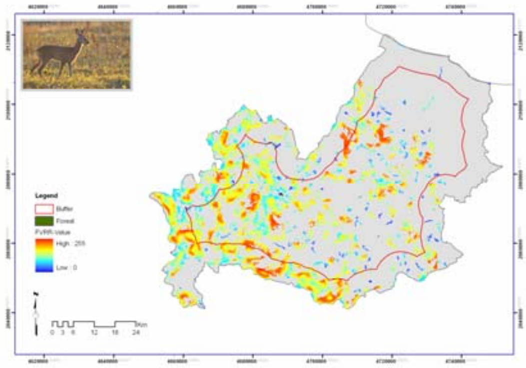
1992 HR



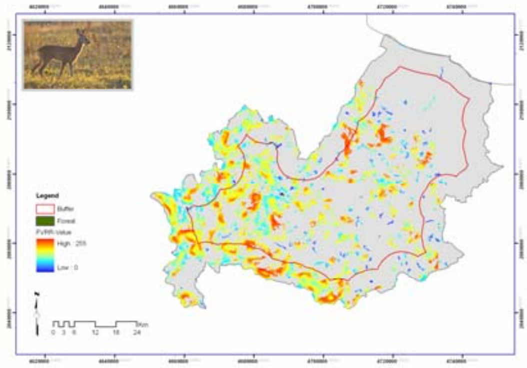
2005 HR



1980 LR

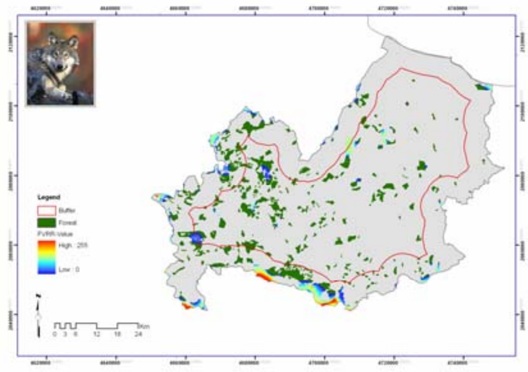


1990 LR

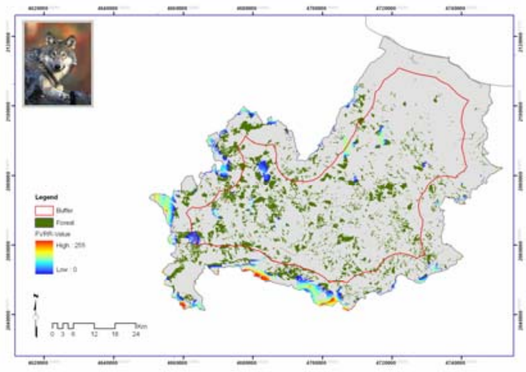


2000 LR

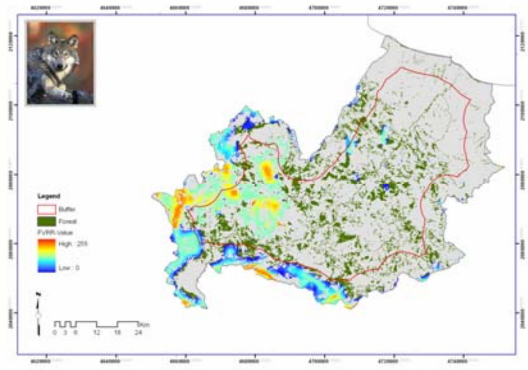
Figure 35: the index FVRR for roe deer umbrella species based on multitemporal high (HR) and low resolution (LR) forest maps.



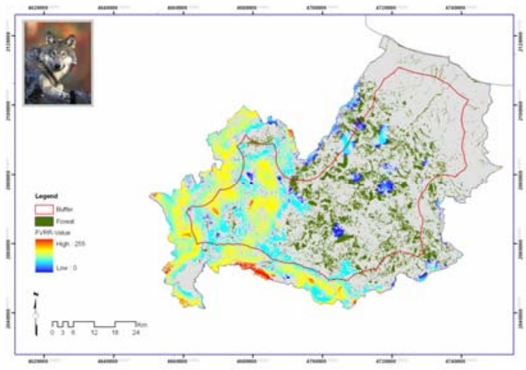
1936 HR



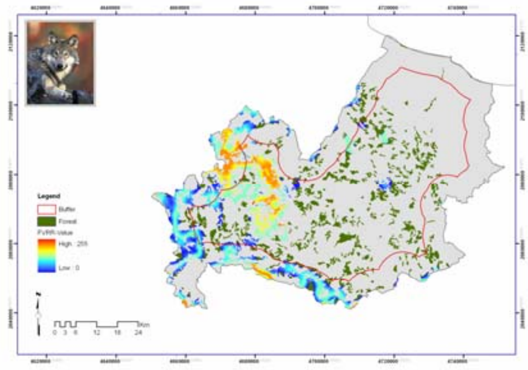
1954 HR



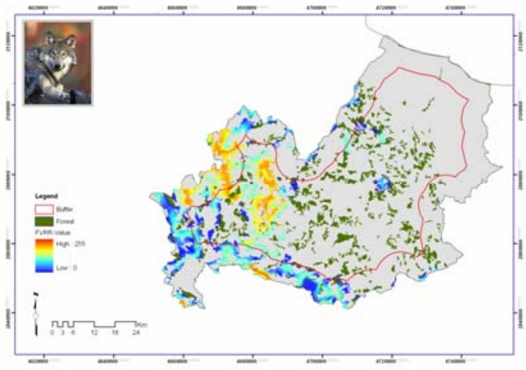
1992 HR



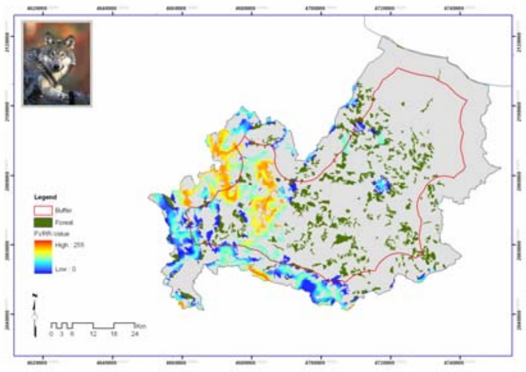
2005 HR



1980 LR

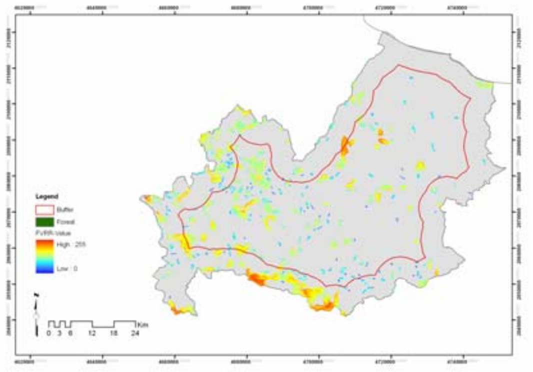


1980 LR

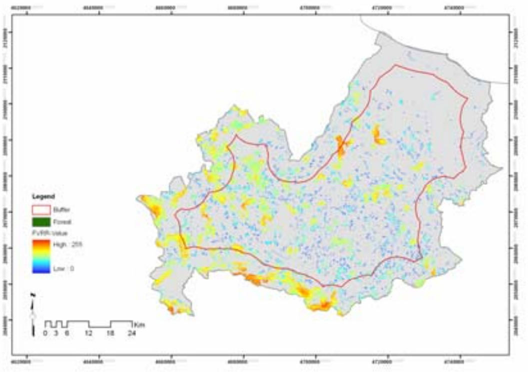


2000 LR

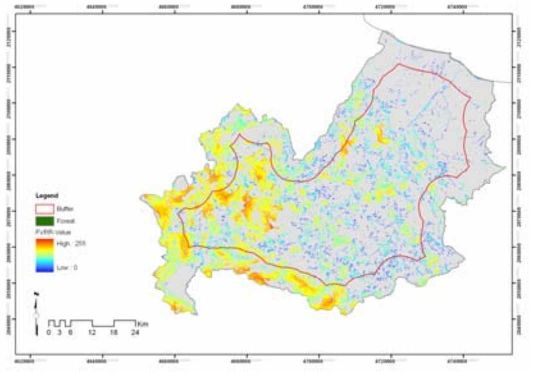
Figure 36: the index FVRR for wolfs umbrella species based on multitemporal high (HR) and low resolution (LR) forest maps.



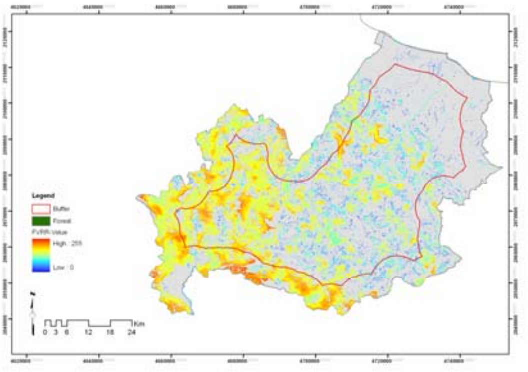
1936 HR



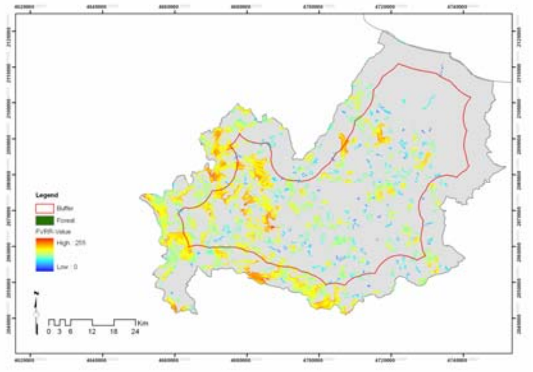
1954 HR



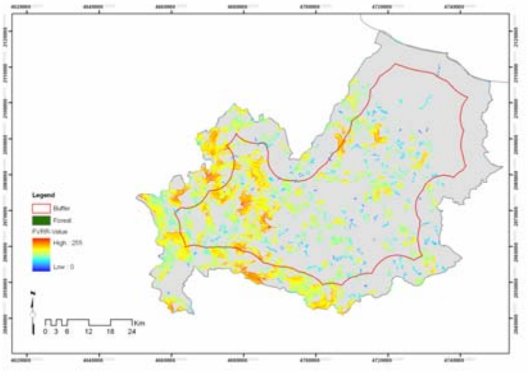
1992 HR



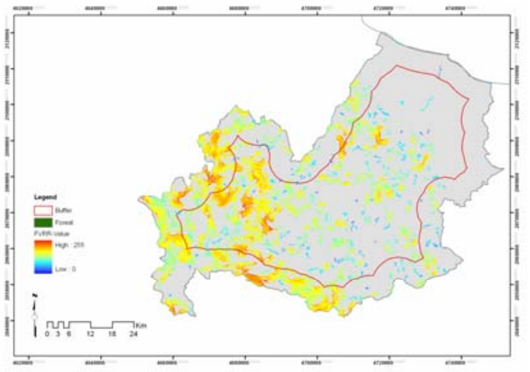
2005 HR



1980 LR



1990 LR



2000 LR

Figure 37: the index FVRR summing up all the five umbrella species based on multitemporal high (HR) and low resolution (LR) forest maps.

8.2 Alpine test area

The Alpine test area is located in Italy in the administrative Region of Veneto and it is 18400 km^2 wide. For the year s1990 and 2000 both low resolution and high resolution maps are available for the whole area (Figure 61). The low resolution map at the year 1980 is instead available just for the administrative Provincia of Belluno.

On the basis of low resolution maps the area is interested by a very limited increasing trend of forest area, just 0.08% per year in the period 1990 – 2000 on the basis of low resolution data and of 0.25% on the basis of high resolution maps. On the basis of low resolution data the trend is increasing for core (0.47%) but decreasing in no core areas (-0.70%), the opposite for high resolution data, core areas decreased of 1.26% yearly while no core areas increased of 2.23%.

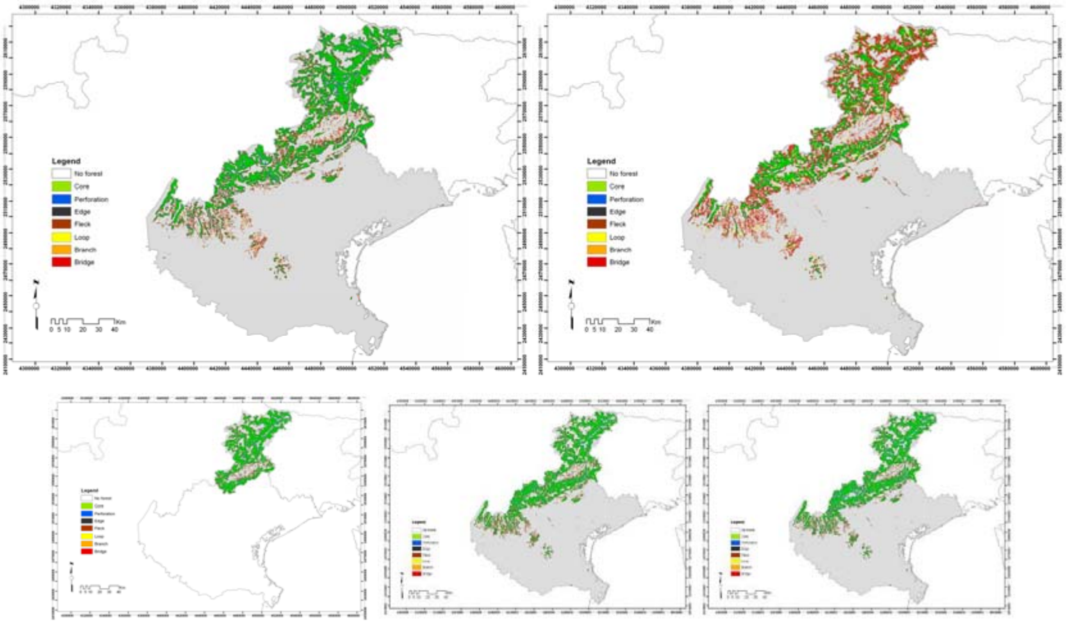
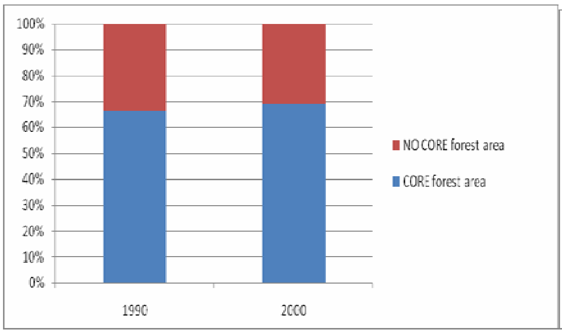
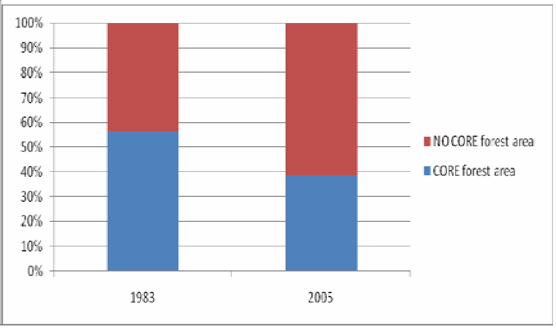


Figure 38: result of the GUIDOS analysis in the Alpine test area. Above on the basis of high resolution maps (year 1983 on the left and 2005 on the right) and below on the basis of low resolution maps (from the left: year 1980 for the Belluno Province, 1990 and 2000).





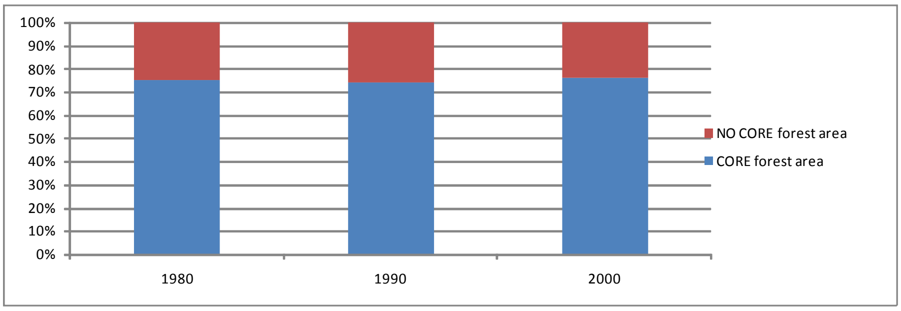


Figure 39: relative area of core and no core forest areas in the different resolution maps, above in the Alpine test area (low resolution on the left and high resolution on the right) below on low resolution data for the Belluno Province only.

The temporal trend in forest area in the Regione Veneto is positive both on the basis of high resolution (+ 0.2% yearly) and on the basis of low resolution maps (+0.1% yearly).

In the subarea of the Belluno Province the trend on the basis of low resolution data was negative in the period 1980-1990 (-1.2% yearly) but positive in the period 1990-2000 (+ 0.1% yearly). The same trend in the last decade was confirmed by high resolution maps (+ 0.1 % yearly).

On the basis of high resolution maps the trend of core forest areas is negative but positive in no core areas. The trends on the basis of low resolution maps are much less strong but they show an opposite trend.

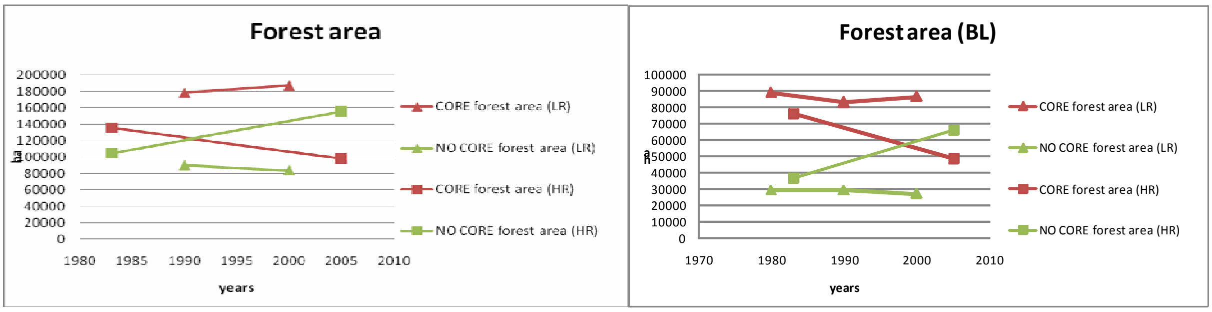


Figure 40: comparison of trends in forest area on the basis of low and high resolution data for core and no core forest areas for the Regione Venetao (left) and Belluno Province (right).

8.2.1 Species unspecific analysis

As expected the average values of the index FVRR is higher in core areas than in no core areas at least on the basis of high resolution maps. The difference is significantly just in high resolution maps.

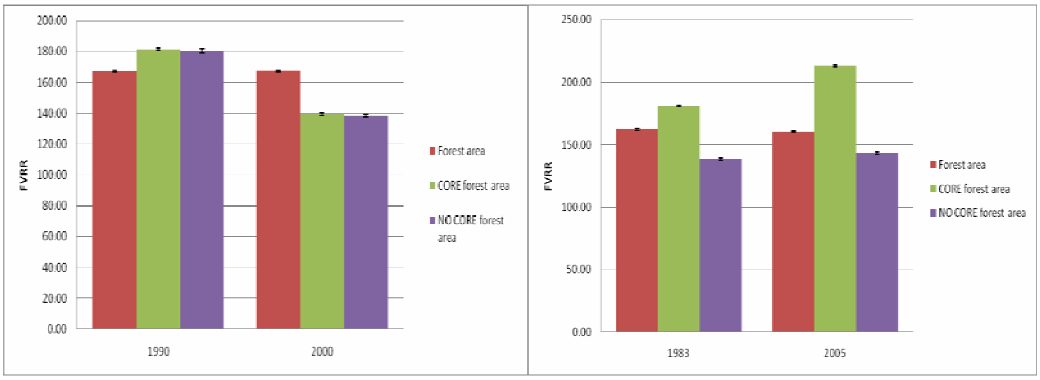


Figure 41: average values with standard error of the species unspecific index of FVRR from low resolution data for core and no core forest areas.

When observing the overall trend of the index FVRR in the low resolution maps the values in core areas are lower than in no core areas.

The trend of the FVRR index is due to the cumulative effect of the increasing forest area and of the increasing average values of the index.

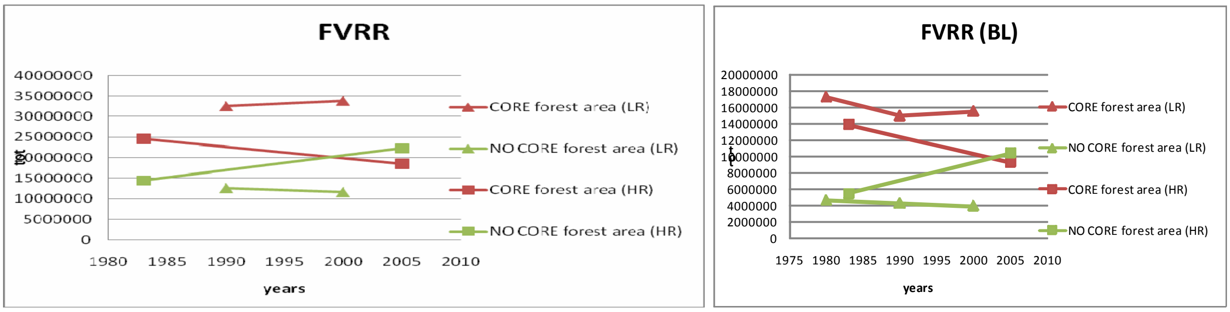


Figure 42: trends in species unspecific index of FVRR for low and high resolution data in forest areas. For Regione Veneto on the left and for Belluno Province on the right.

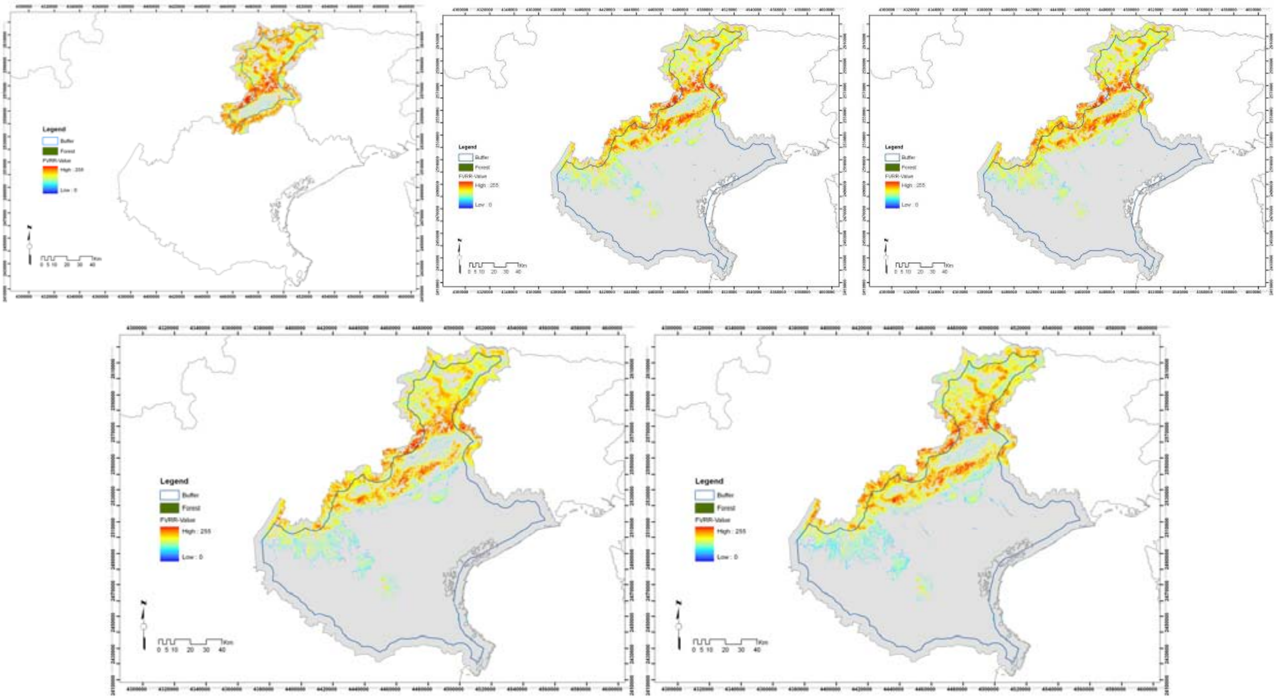


Figure 43: the index FVRR based on multitemporal low resolution forest maps above (from the left: 1980 (Belluno Province), 1990, 2000) and on high resolution maps below (from the left: 1983 and 2005).

8.2.2 Species specific analysis

Here follow the results of the calculation of the FVRR index for the five umbrella species.

As expected from the theoretical ecological background for birds the average value of FVRR in core areas is always greater than in no core areas.

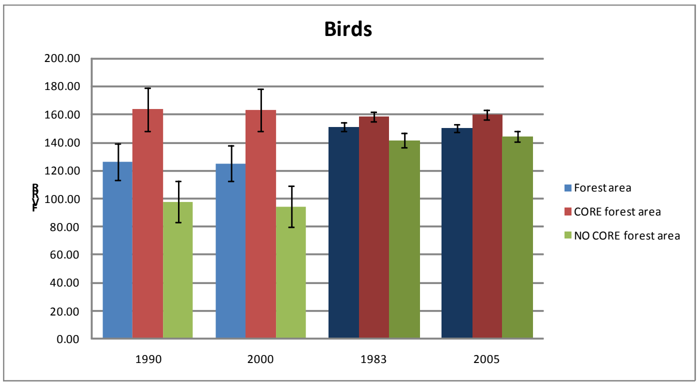


Figure 44: average values with standard error of the birds index of FVRR from low and high (darker) resolution data for core and no core forest areas.

The temporal trend of FVRR for birds is very limited because limited is the change in forest area.

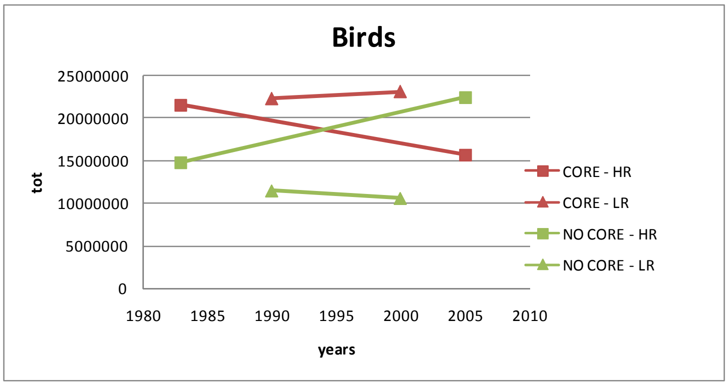


Figure 45: comparison of trends in birds index of FVRR on low and high resolution data for core and no core forest areas.

As expected from the theoretical ecological background for butterflies the average value of FVRR in no core areas is always greater than in no core areas.

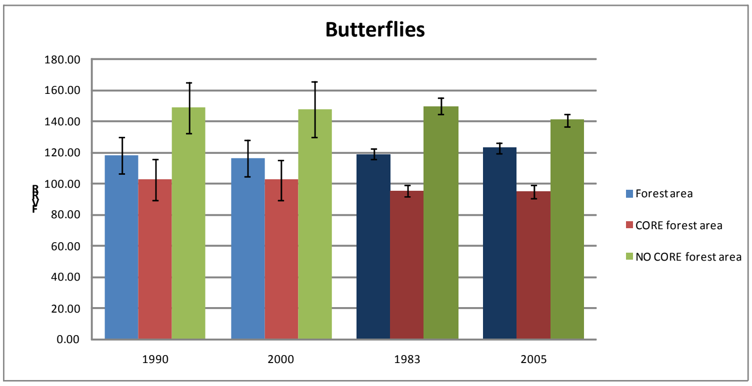


Figure 46: average values with standard error of the butterflies index of FVRR on low and high resolution data for core and no core forest areas.

The temporal trend of FVRR for birds is very limited because limited is the change in forest area.

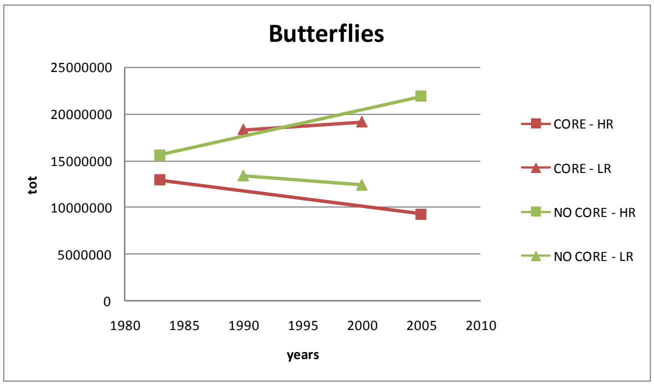


Figure 47: comparison of trends in butterflies index of FVRR on low resolution data for core and no core forest areas.

As expected from the theoretical ecological background for birds the average value of FVRR in core areas is always much greater than in no core areas.

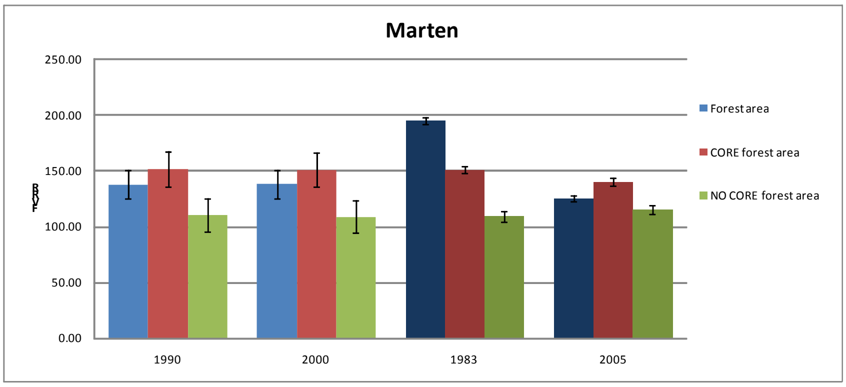


Figure 48: average values with standard error of the marten index of FVRR on low and high resolution data for core and no core forest areas.

The temporal trend of FVRR for marten is very limited because limited is the change in forest area (Figure 76).

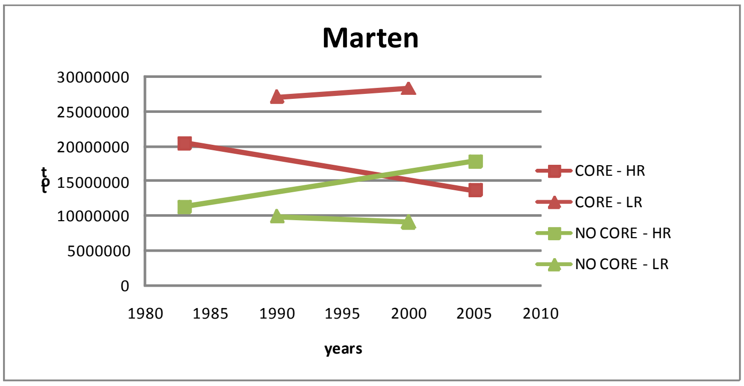


Figure 49: comparison of trends in marten index of FVRR on low and high resolution data for core and no core forest areas.

As expected from the theoretical ecological background for roe deer the average value of FVRR in core areas is always greater than in no core areas.

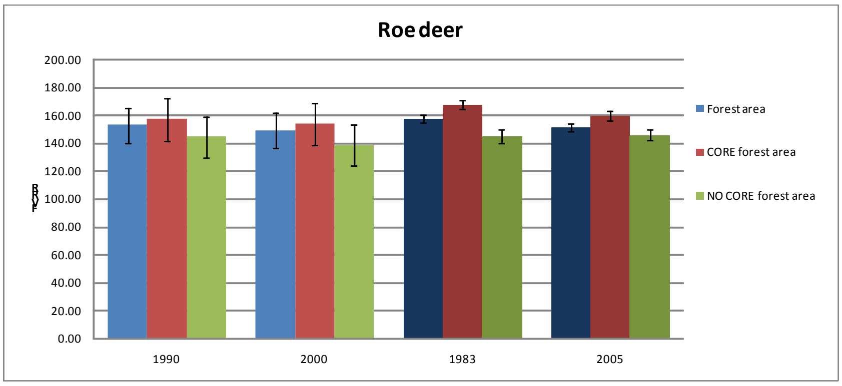


Figure 50: average values with standard error of the roe deer index of FVRR low and high resolution data for core and no core forest areas.

The temporal trend of FVRR for roe deer is very limited because limited is the change in forest area.

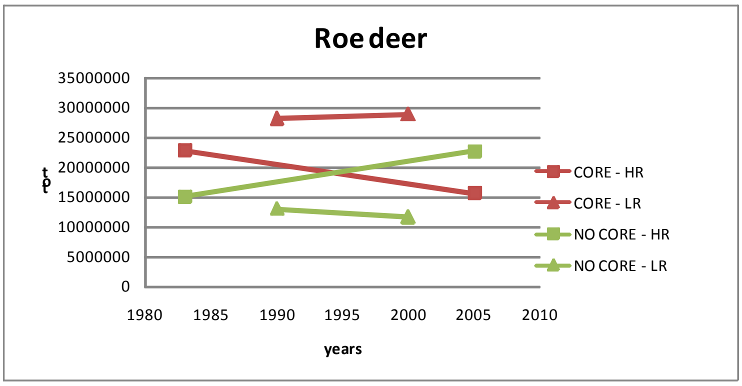


Figure 51: comparison of trends in roe deer index of FVRR on low and high resolution data for core and no core forest areas.

As expected from the theoretical ecological background for wolf the average value of FVRR in core areas is always greater than in no core areas (Figure 78).

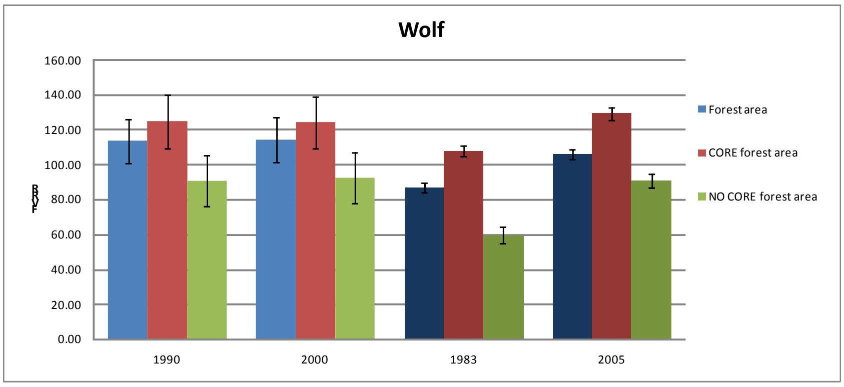


Figure 52: average values with standard error of the wolf index of FVRR low and high resolution data for core and no core forest areas.

The temporal trend of FVRR for wolf is very limited because limited is the change in forest area (Figure 80).

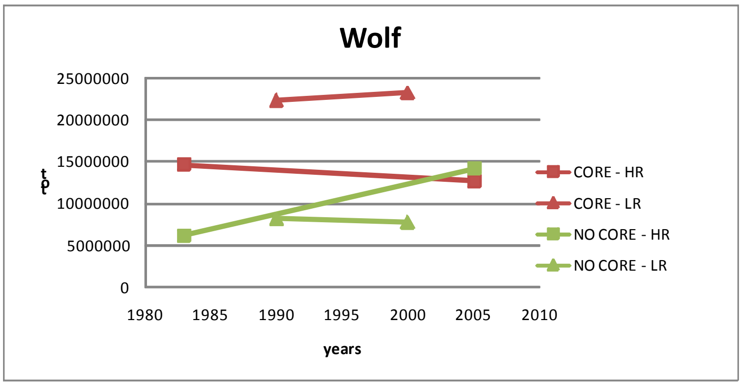


Figure 53: average values with standard error of the wolf index of FVRR low and high resolution data for core and no core forest areas.

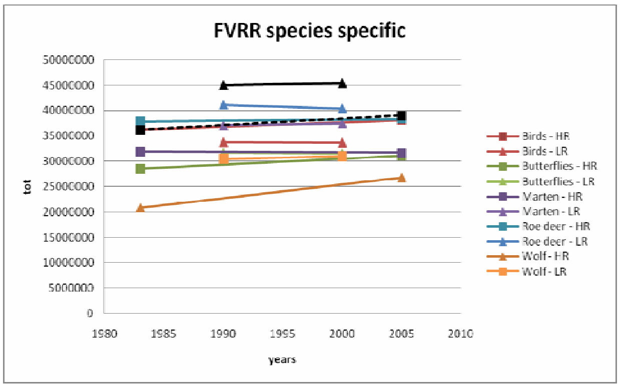


Figure 54: comparison of trends in all the umbrella species index of FVRR comparing low and high resolution data.

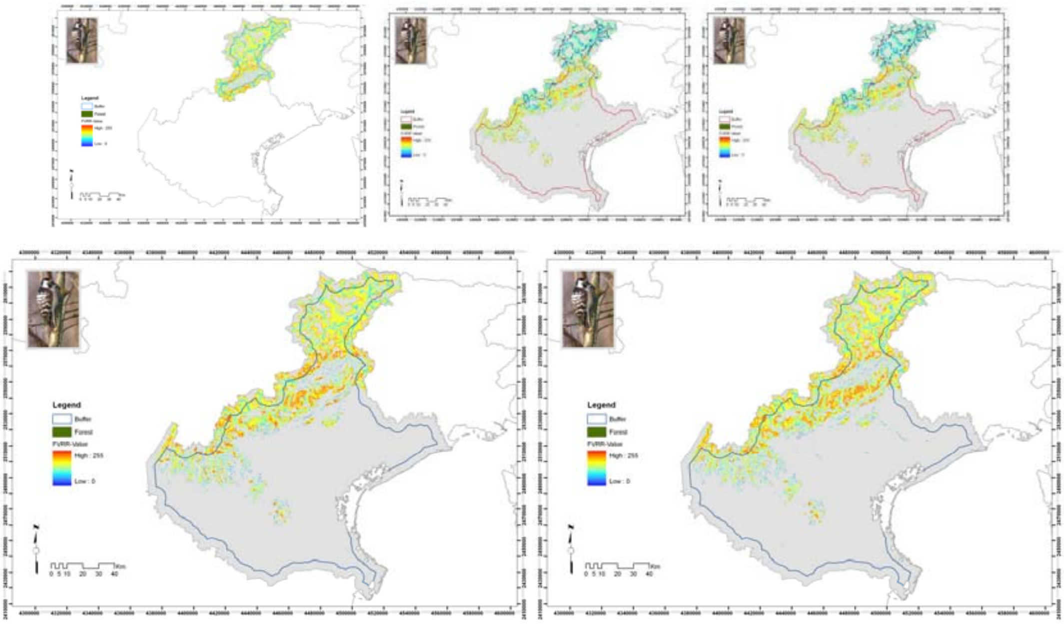


Figure 55: the index FVRR for birds umbrella species based on multitemporal low (above from the left: 1980, 1990, 2000) and high (above from the left: 1983 and 2005) resolution forest maps.

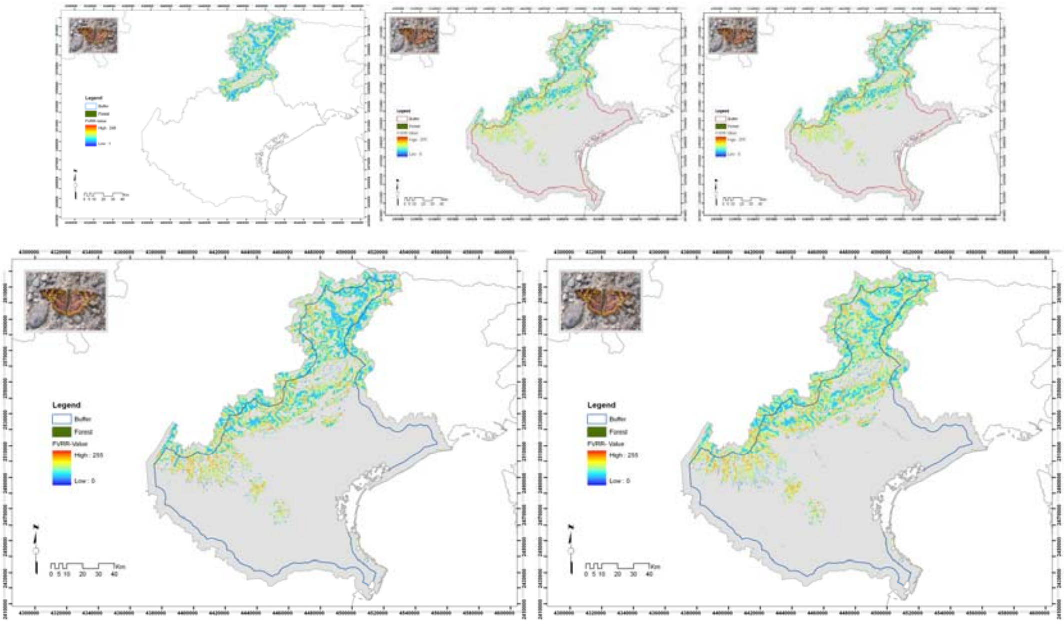


Figure 56: the index FVRR for butterflies umbrella species based on multitemporal low (above from the left: 1980, 1990, 2000) and high (above from the left: 1983 and 2005) resolution forest maps.

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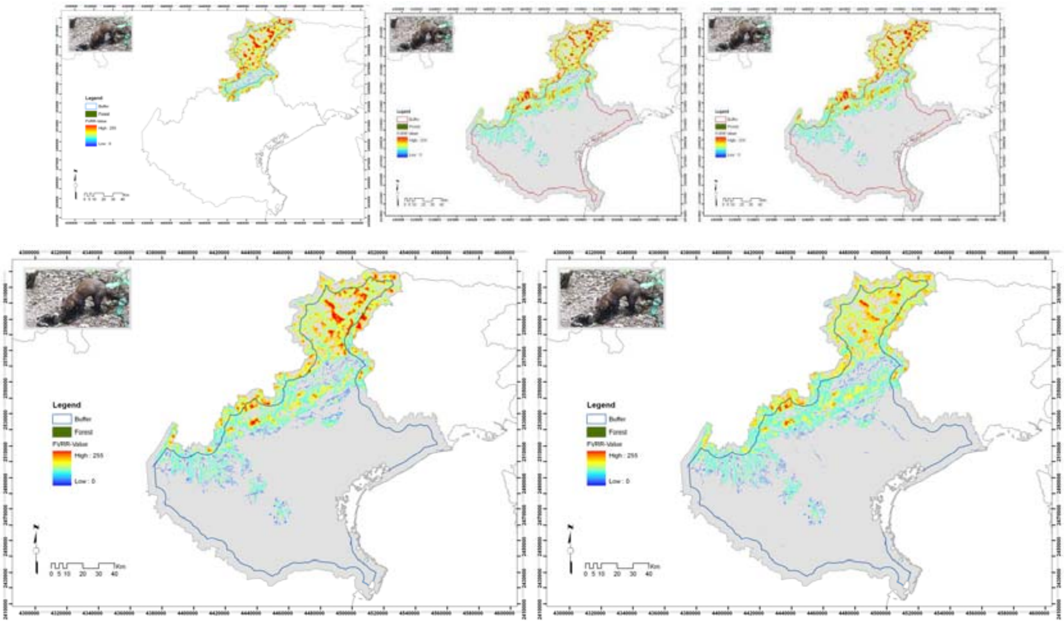


Figure 57: the index FVRR for marten umbrella species based on multitemporal low (above from the left: 1980, 1990, 2000) and high (above from the left: 1983 and 2005) resolution forest maps.

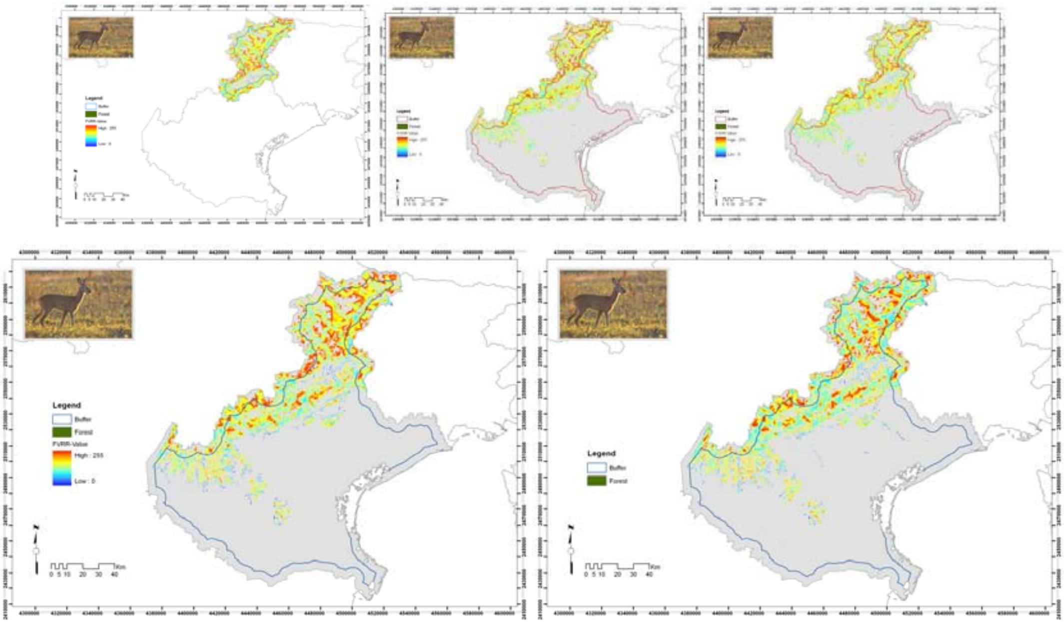


Figure 58: the index FVRR for roe deer umbrella species based on multitemporal low (above from the left: 1980, 1990, 2000) and high (above from the left: 1983 and 2005) resolution forest maps.

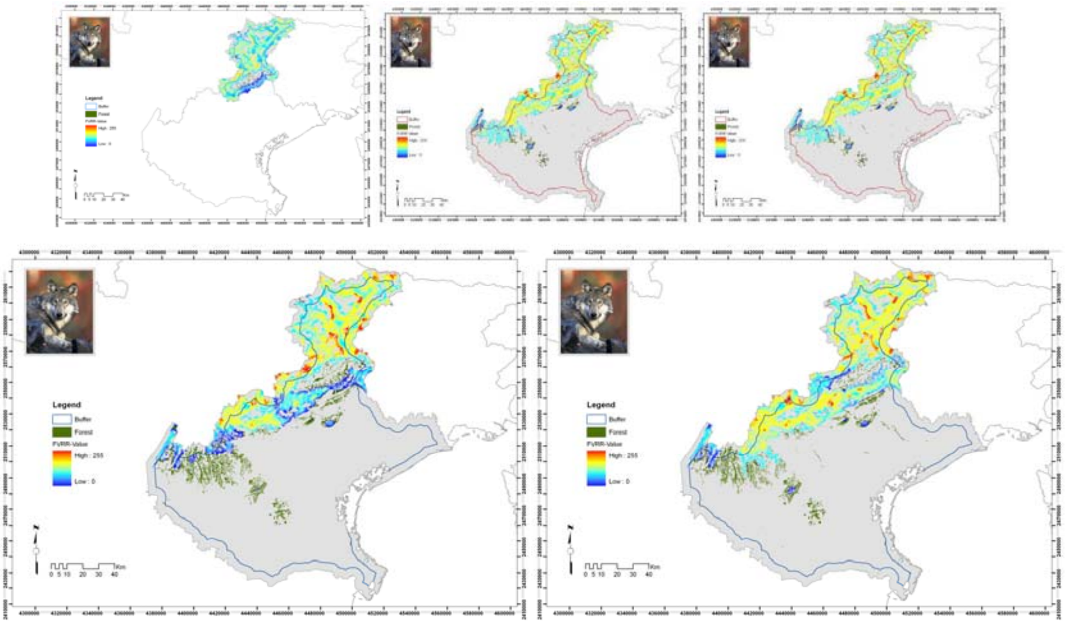


Figure 59: the index FVRR for wolf umbrella species based on multitemporal low (above from the left: 1980, 1990, 2000) and high (above from the left: 1983 and 2005) resolution forest maps.

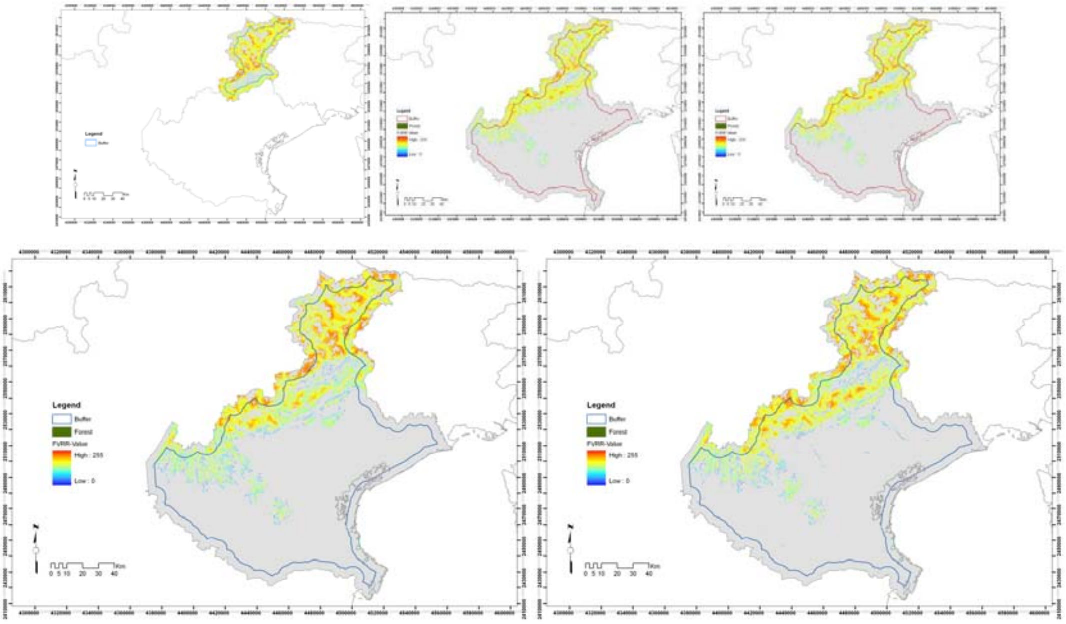


Figure 60: the averaged index FVRR for all the five umbrella species based on multitemporal low (above from the left: 1980, 1990, 2000) and high (above from the left: 1983 and 2005) resolution forest maps.

8.3 Continental test area

The Continental test area located in Germany (Saxony) covered by low resolution data is 5020 km^2 wide. For the year s1990 and 2000 both low resolution and high resolution maps are available.

On the basis of low resolution maps the area is interested by a very limited decreasing trend of forest area, just -0.23% year in the period 1990 - 2000. Such a trend is both for core (-0.22%) and no core areas (-0.24%) (Figure 62).

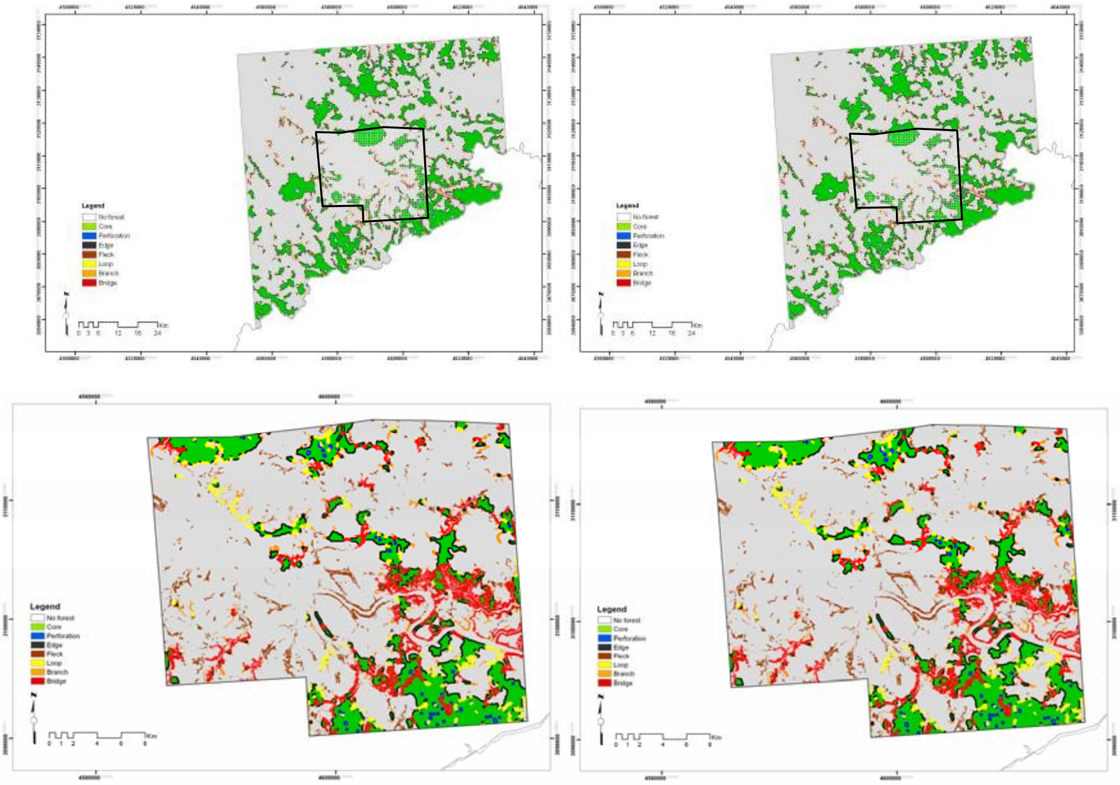


Figure 61: result of the GUIDOS analysis on multitemporal low (above) and high (below) resolution forest maps. Both data are from the year 1990 (on the left) and from the year 2000 (on the right). The high resolution maps are from a subsample on the test area represented in the above high resolution maps.

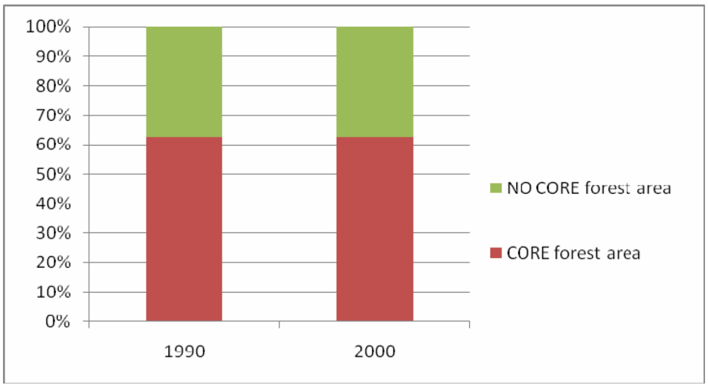


Figure 62: relative area of core and no core forest areas in the different low resolution maps available in the Continental test area.

In the subset of the Continental test area covered also by high resolution maps the relative proportion of core areas is higher in low resolution data than in high resolution data.

Figure 63: relative area of core and no core forest areas in the different low and high resolution maps available in the Continental test area.

The temporal trend in forest area is very limited both for high and low resolution data and both in core and no core areas.

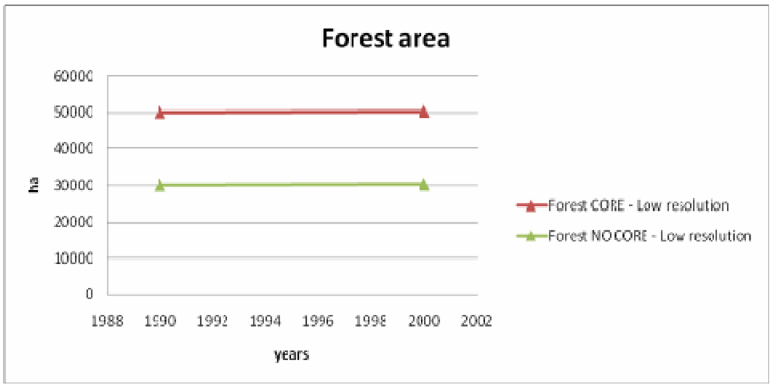


Figure 64: comparison of trends in forest area on the basis of low resolution data for core and no core forest areas.

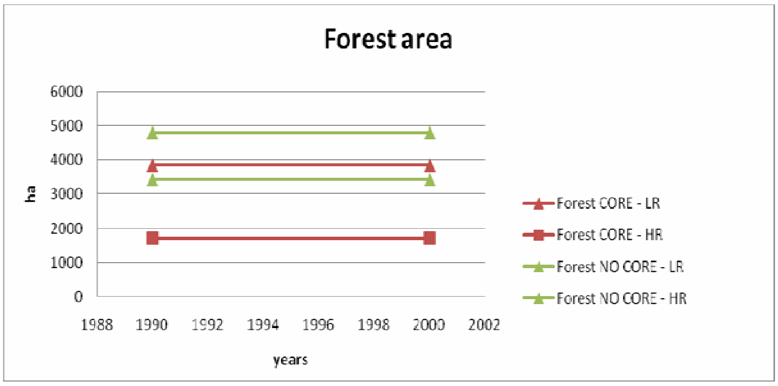


Figure 65: comparison of trends in forest area on the basis of low and high resolution data for core and no core forest areas.

8.3.1 Species unspecific analysis

The FVRR species unspecific model was applied both on high and low resolution maps available for the year 1990 and 2000.

As expected the average value of the index FVRR is higher in core areas than in no core areas for both the study years.

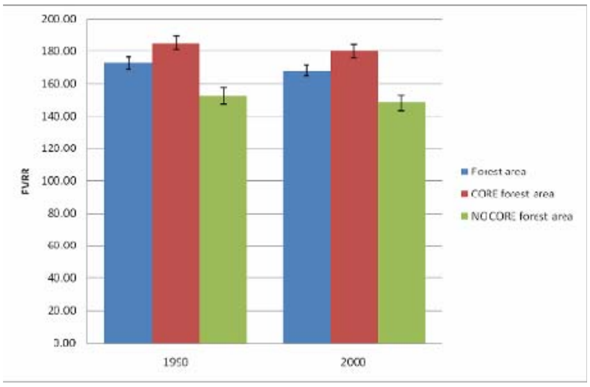


Figure 66: average values with standard error of the species unspecific index of FVRR from low resolution data for core and no core forest areas.

The same relationship was found in the subarea where both high and low resolution maps were available.

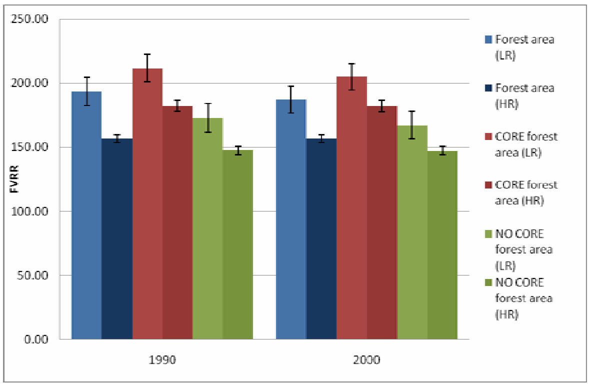


Figure 67: average values with standard error of the species unspecific index of FVRR from high and low resolution data for core and no core forest areas.

When observing the overall trend of the index FVRR in the low resolution maps the values in core areas are lower than in no core areas.

The trend of the FVRR index in Figure 68 is due to the cumulative effect of the decreasing forest area and because of the decreasing average values of the index.

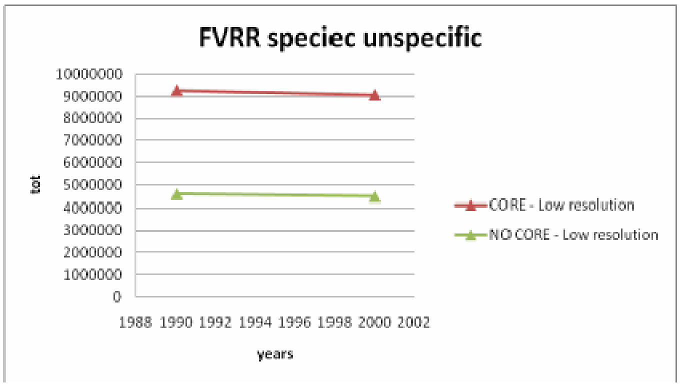


Figure 68: trends in species unspecific index of FVRR for low resolution data in forest areas.

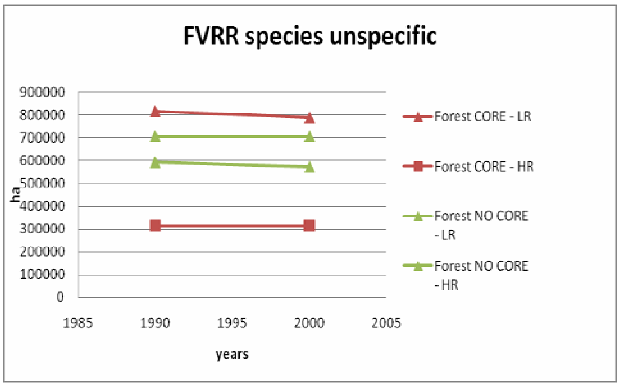
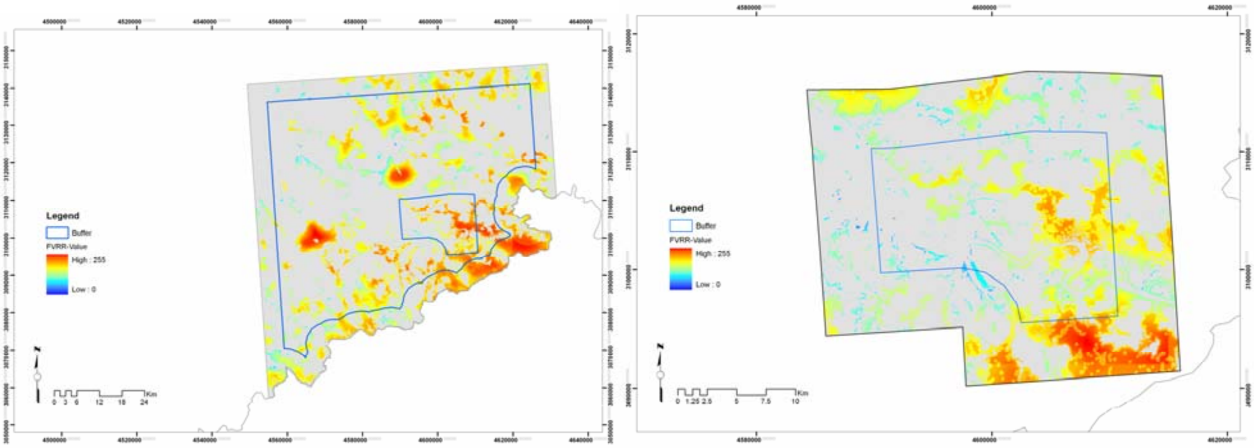
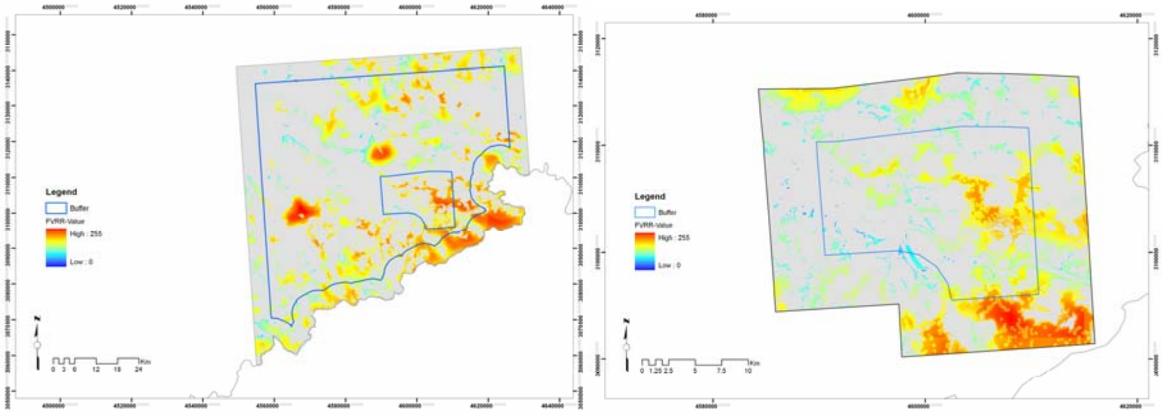


Figure 69: trends in species unspecific index of FVRR for low and high resolution data in forest areas.



1990



2000

Figure 70: the index FVRR based on multitemporal low (left) and high (right) resolution forest maps.

8.3.2 Species specific analysis

The results of the index FVRR was calculated for birds (Figure 82), butterflies (Figure 83), marten (Figure 84), roe deer (Figure 85) and wolf (Figure 86). The five species were also aggregated summing up all the indexes (Figure 87).

As expected from the theoretical ecological background for birds the average value of FVRR in core areas is always much greater than in no core areas (Figure 71).

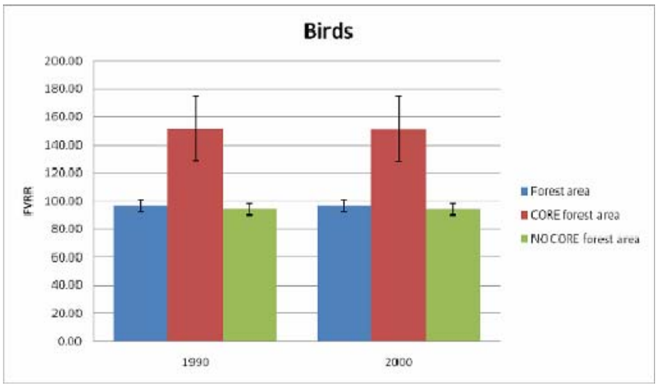


Figure 71: average values with standard error of the birds index of FVRR low resolution data for core and no core forest areas.

The temporal trend of FVRR for birds is very limited because limited is the change in forest area (Figure 72).

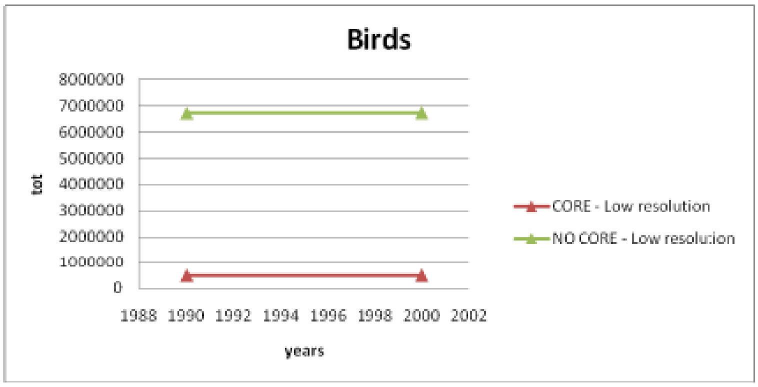


Figure 72: comparison of trends in birds index of FVRR on low resolution data for core and no core forest areas.

As expected from the theoretical ecological background for butterflies the average value of FVRR in no core areas is always greater than in no core areas (Figure 71).

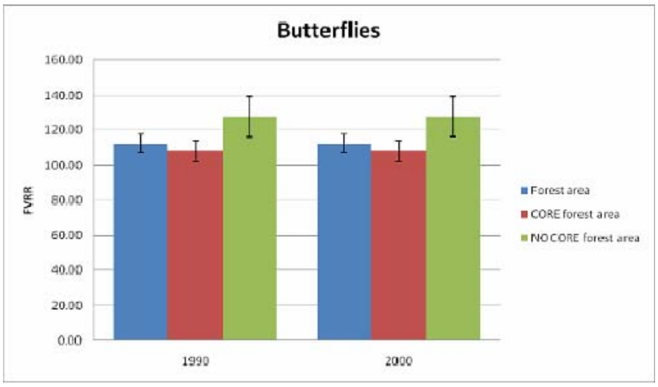


Figure 73: average values with standard error of the butterflies index of FVRR low resolution data for core and no core forest areas.

The temporal trend of FVRR for birds is very limited because limited is the change in forest area (Figure 74).

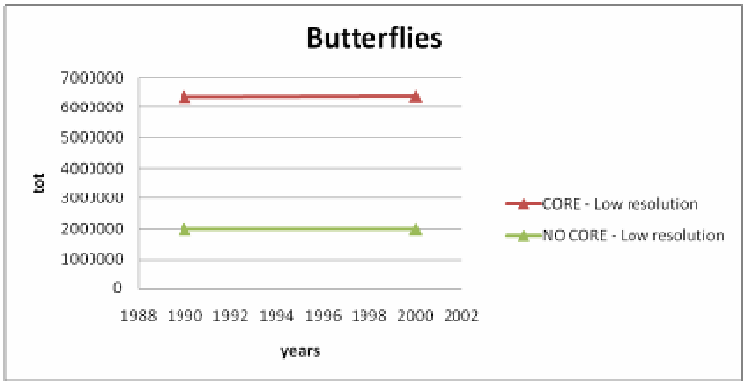


Figure 74: comparison of trends in butterflies index of FVRR on low resolution data for core and no core forest areas.

As expected from the theoretical ecological background for birds the average value of FVRR in core areas is always much greater than in no core areas (Figure 75).

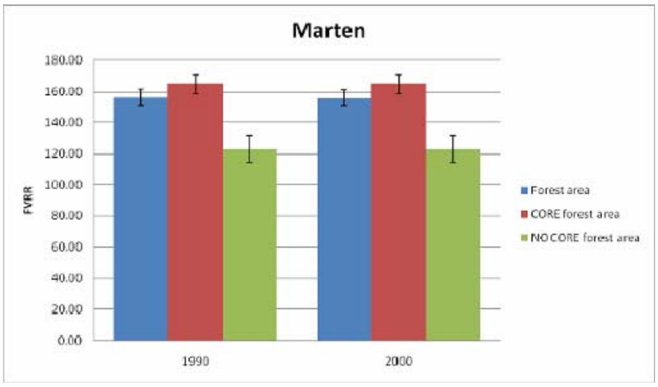


Figure 75: average values with standard error of the marten index of FVRR low resolution data for core and no core forest areas.

The temporal trend of FVRR for marten is very limited because limited is the change in forest area (Figure 76).

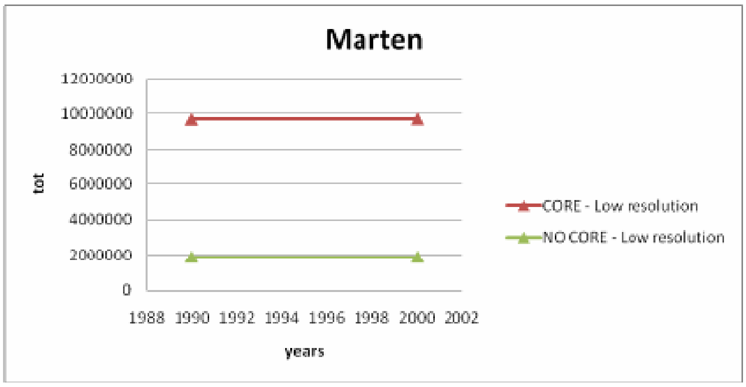


Figure 76: comparison of trends in marten index of FVRR on low resolution data for core and no core forest areas.

As expected from the theoretical ecological background for roe deer the average value of FVRR in core areas is always greater than in no core areas (Figure 77).

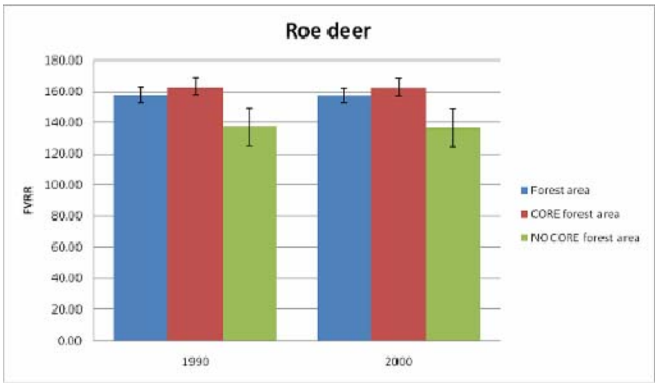


Figure 77: average values with standard error of the roe deer index of FVRR low resolution data for core and no core forest areas.

The temporal trend of FVRR for roe deer is very limited because limited is the change in forest area (Figure 78).

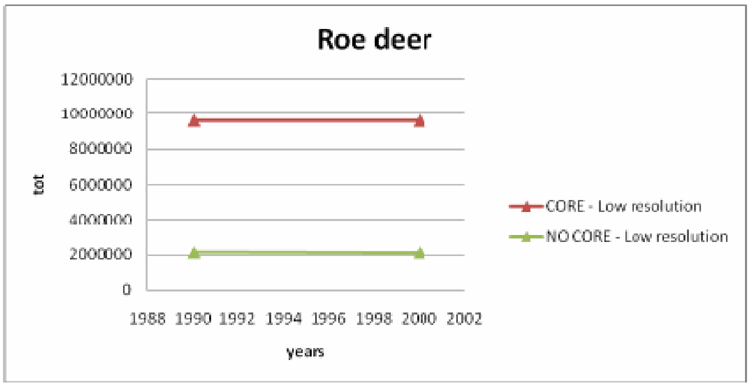


Figure 78: comparison of trends in roe deer index of FVRR on low resolution data for core and no core forest areas.

As expected from the theoretical ecological background for wolf the average value of FVRR in core areas is always greater than in no core areas (Figure 78).

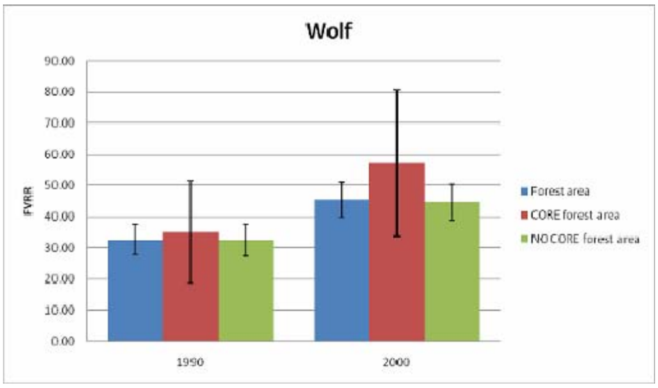


Figure 79: average values with standard error of the wolf index of FVRR low resolution data for core and no core forest areas.

The only one temporal trend of FVRR is that one for wolf, at least in no core areas. Although a limited change in forest area occurred it was enough to change the sensible FVRR index (Figure 80).

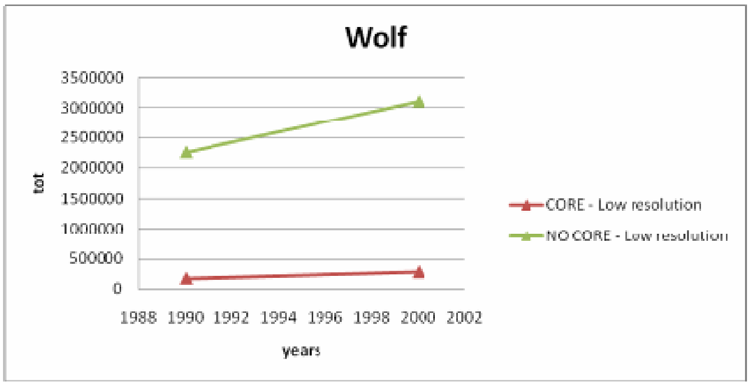


Figure 80: average values with standard error of the wolf index of FVRR low resolution data for core and no core forest areas.

The results achieved in the Continental test area clearly show that the very limited change in forest cover did not determined any change in none of the considered umbrella species with the exception of wolf. Note that the average trend of all the considered species is also reported in Figure 81 (black lines).

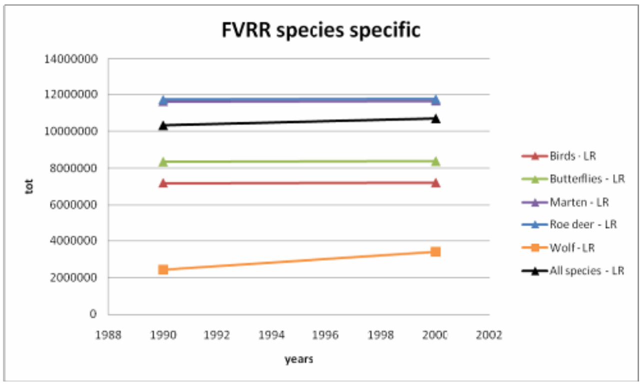
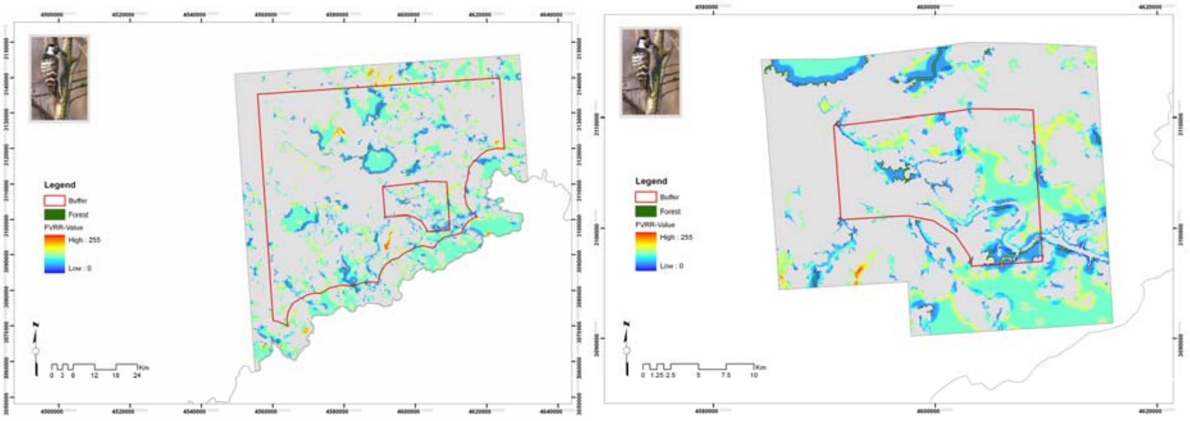
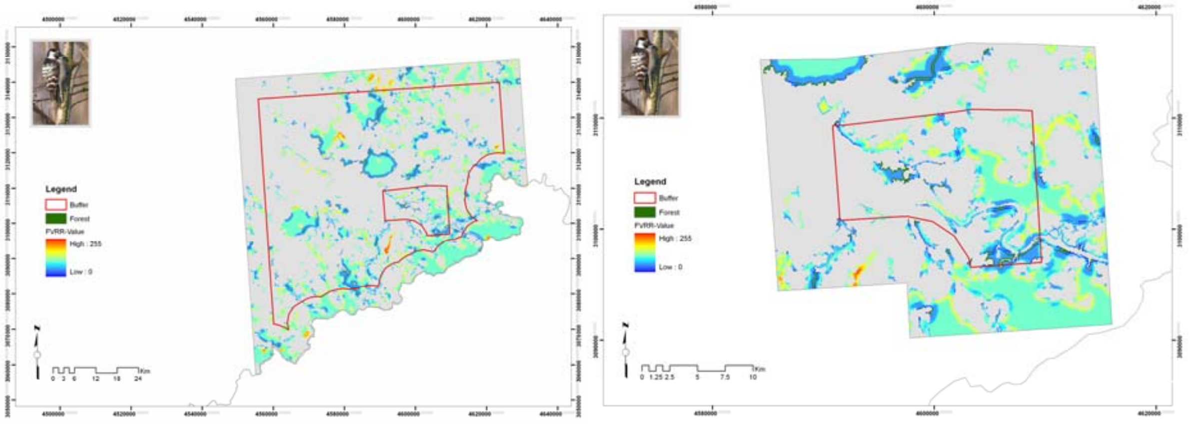


Figure 81: comparison of trends in all the umbrella species index of FVRR.

Here follows all the species specific FVRR maps based on low resolution data.

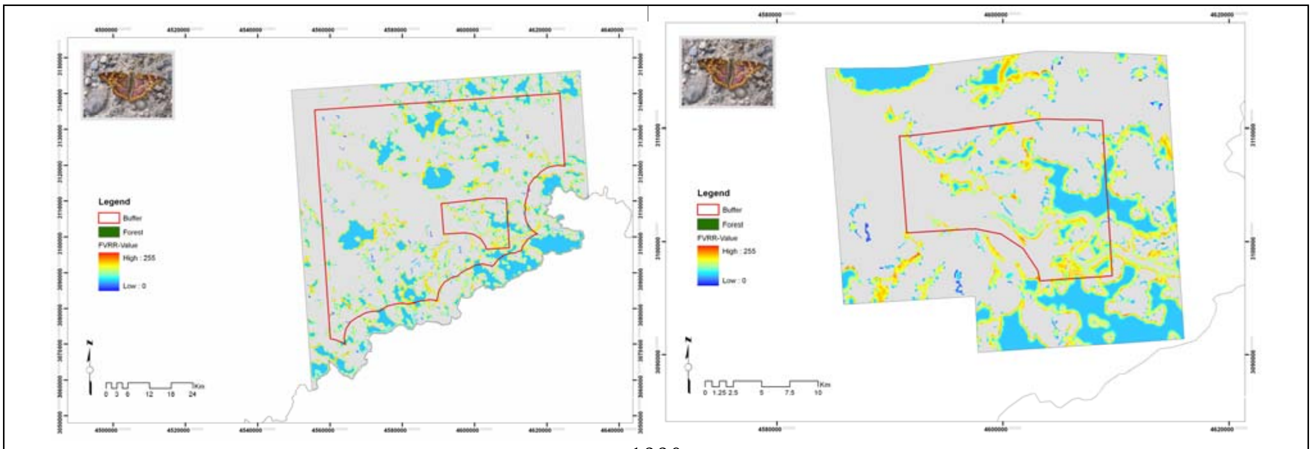


1990

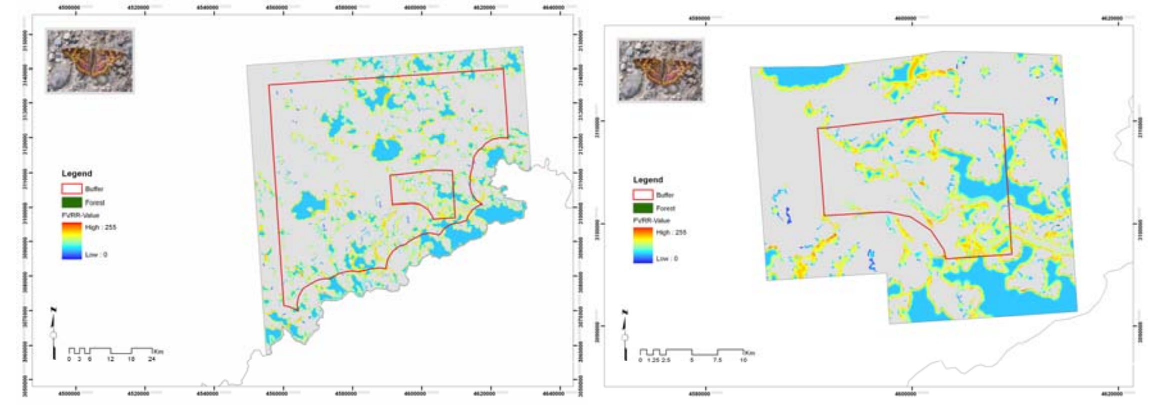


2000

Figure 82: the index FVRR for birds umbrella species based on multitemporal low (on the left) and high (on the right) resolution forest maps.

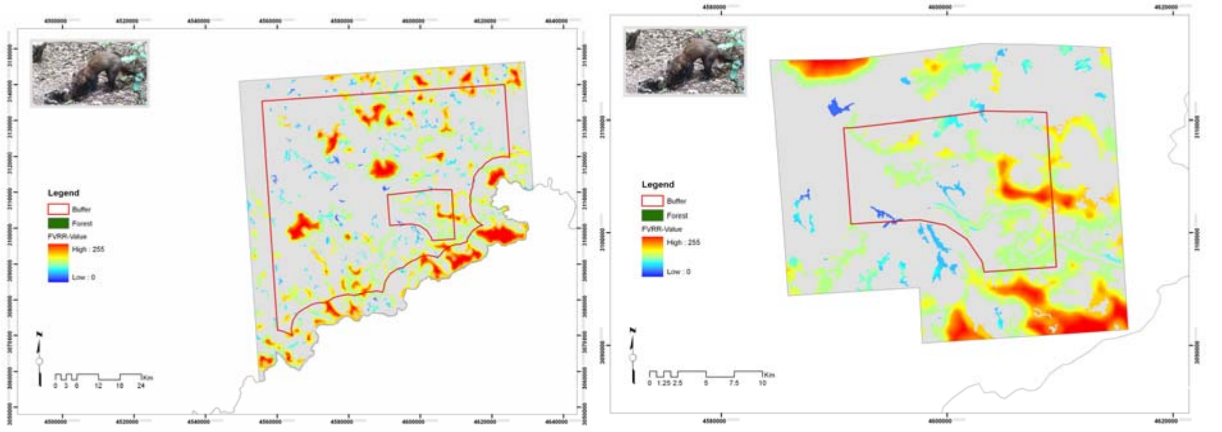


1990

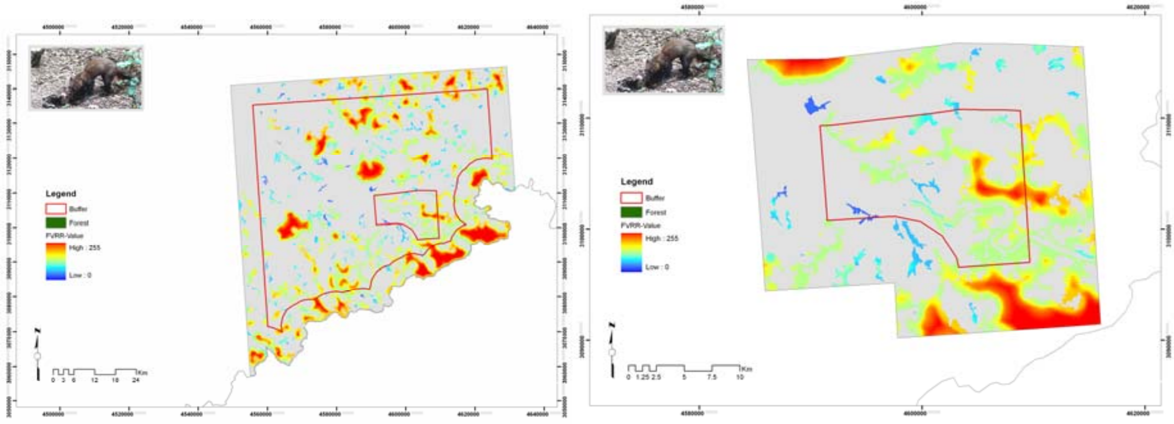


2000

Figure 83: the index FVRR for butterflies umbrella species based on multitemporal low (on the left) and high (on the right) resolution forest maps.

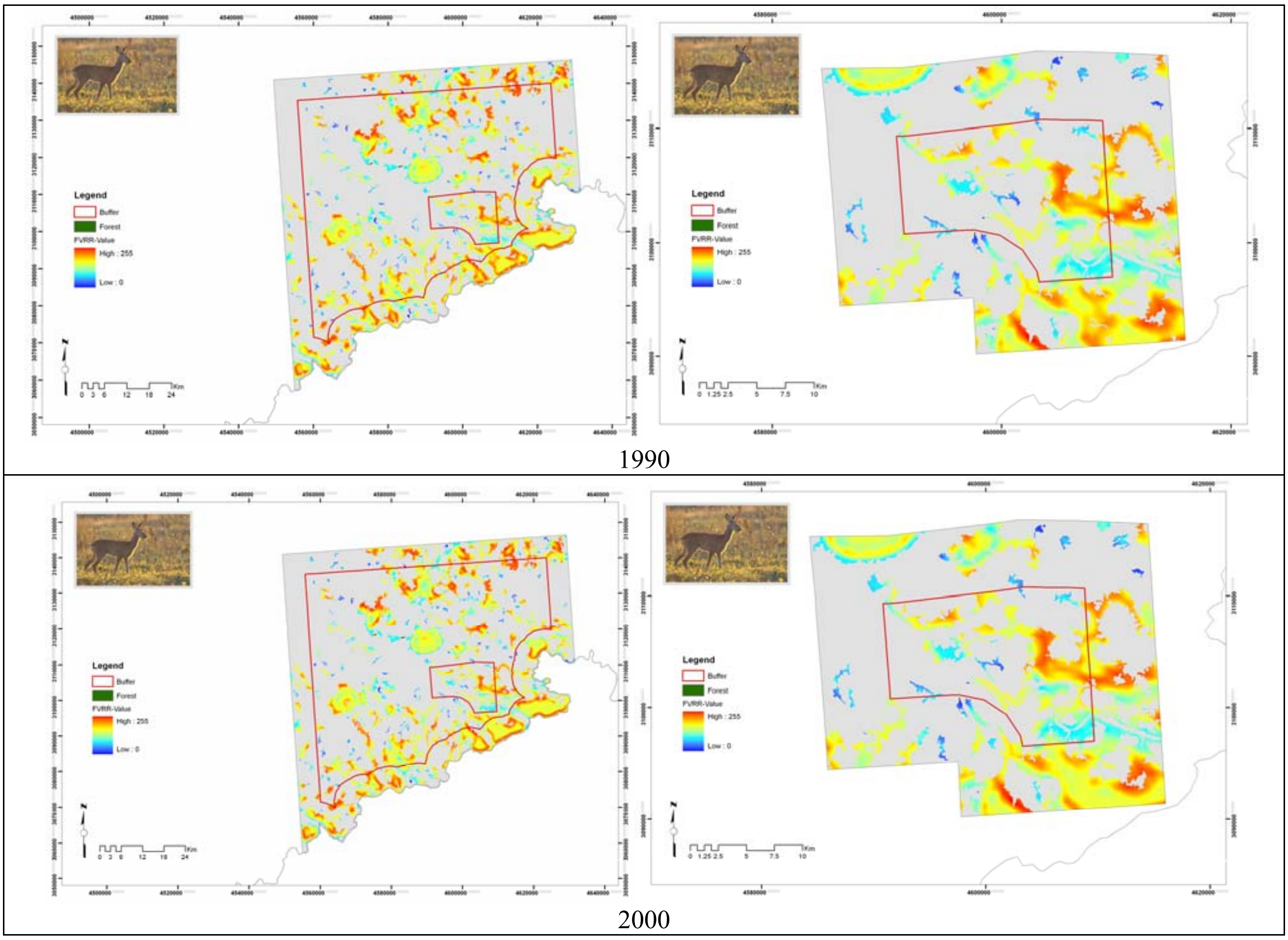


1990



2000

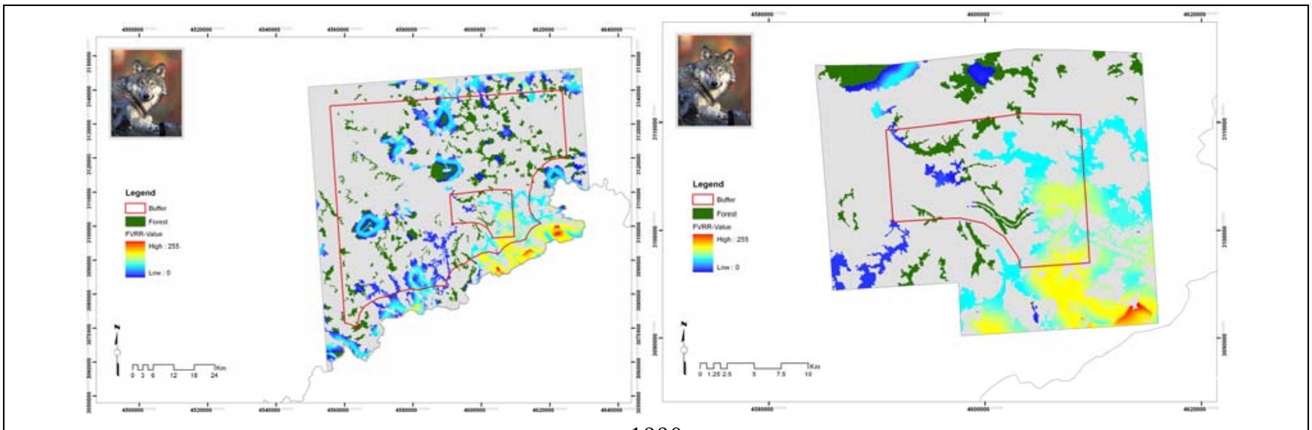
Figure 84: the index FVRR for marten umbrella species based on multitemporal low (on the left) and high (on the right) resolution forest maps.



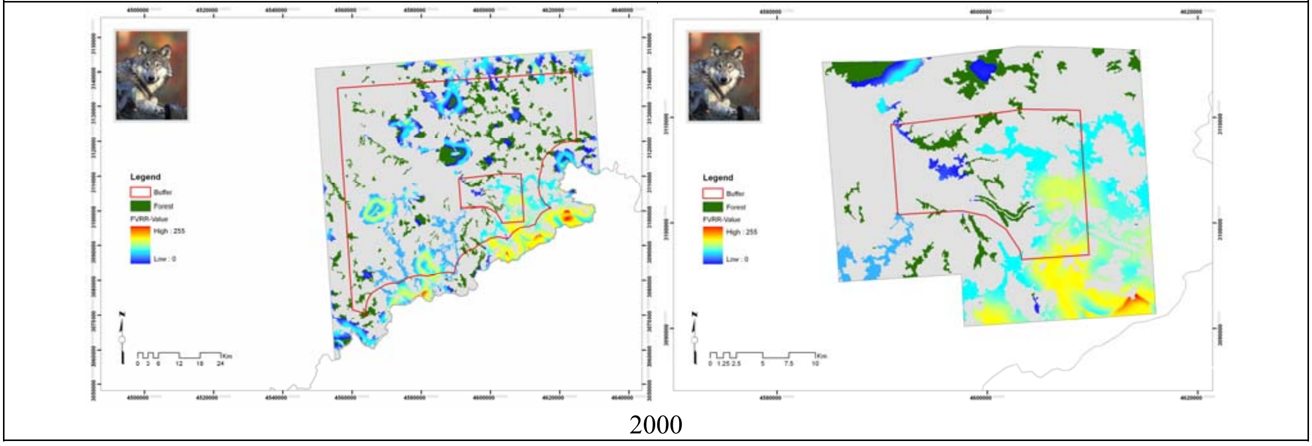
1990

2000

Figure 85: the index FVRR for roe deer umbrella species based on multitemporal low (on the left) and high (on the right) resolution forest maps.

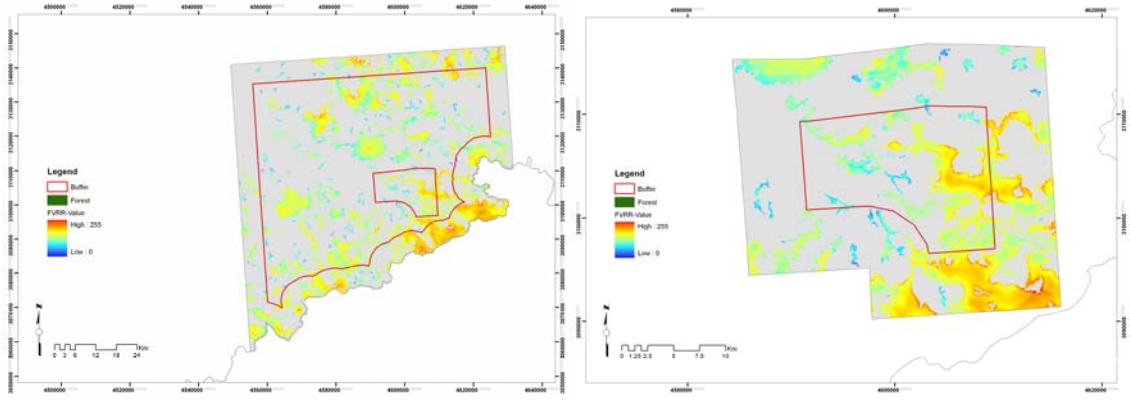


1990

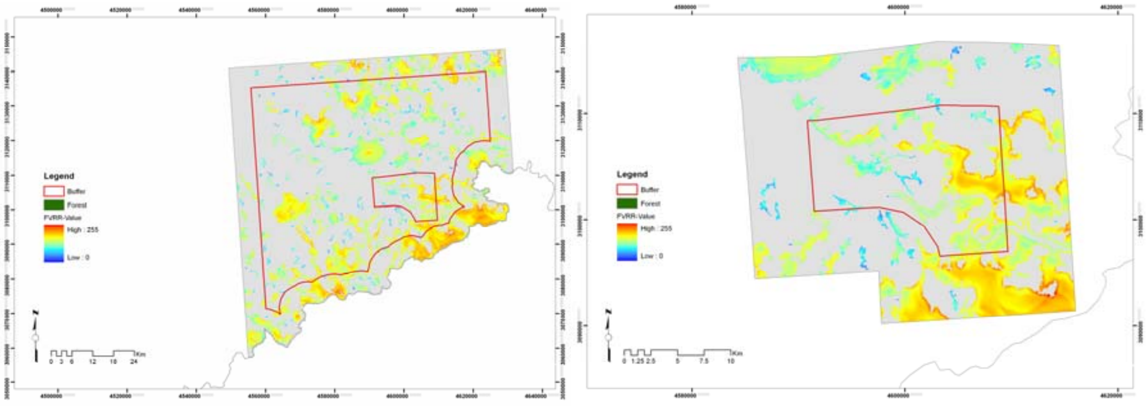


2000

Figure 86: the index FVRR for wolf umbrella species based on multitemporal low (on the left) and high (on the right) resolution forest maps.



1990



2000

Figure 87: the index FVRR summing up all the five umbrella species based on multitemporal low (on the left) and high (on the right) resolution forest maps.

8.4 Atlantic test area

The Atlantic test area is divided in two subareas on in north Walles (3097 km^2 wide) and one in south England (1663 km^2 wide). In this area the multitemporal analysis was not completed since the low resolution 1990 maps was not available. UK did not participated to the Corine Land Cover project at the year 1990 and for this reason the acquisition of data layers is more complex.

Here follow the GUIDOS map based on low resolution data at the year.

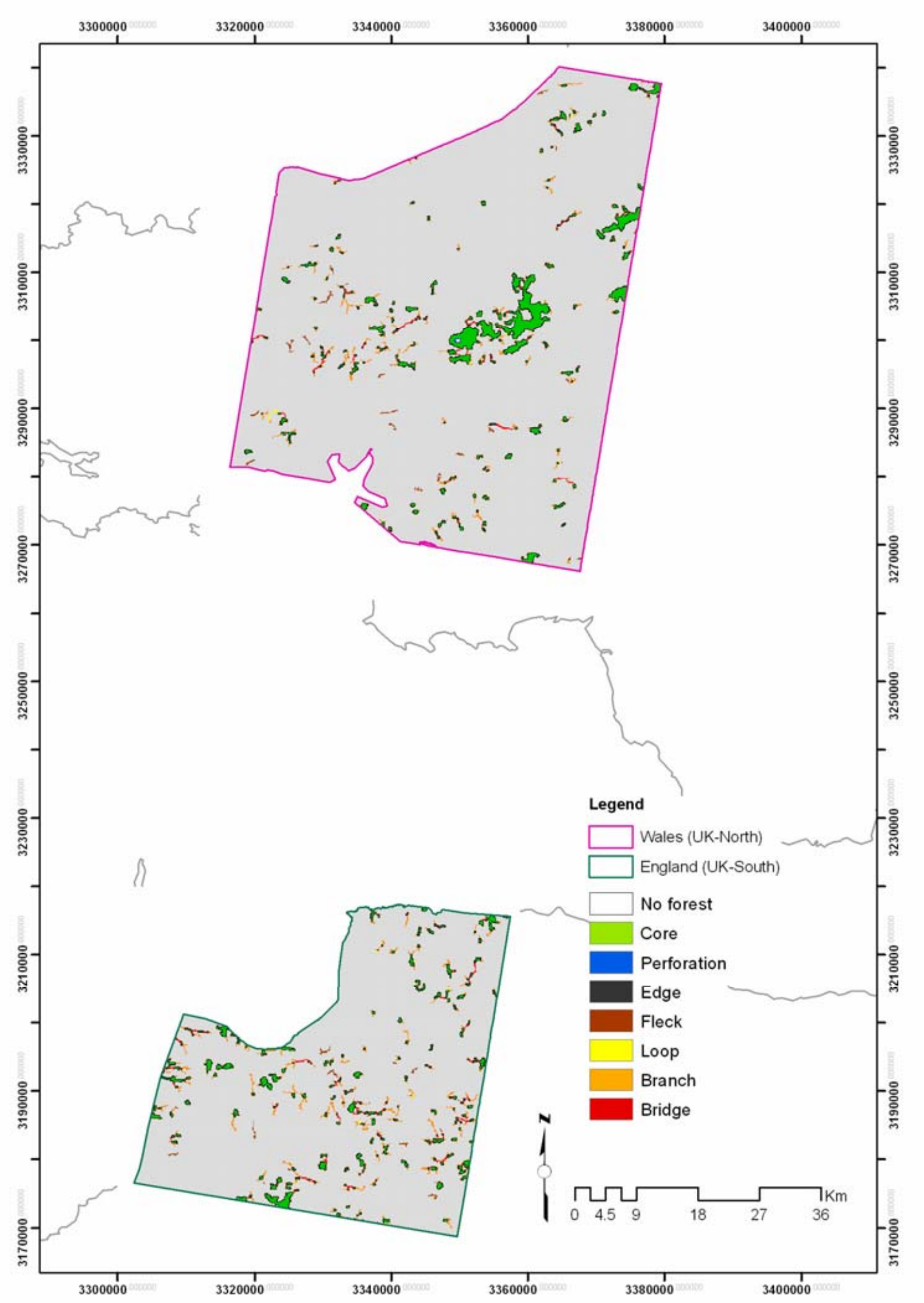


Figure 88: result of the GUIDOS analysis on multitemporal low (LR) forest map at the year 2000.

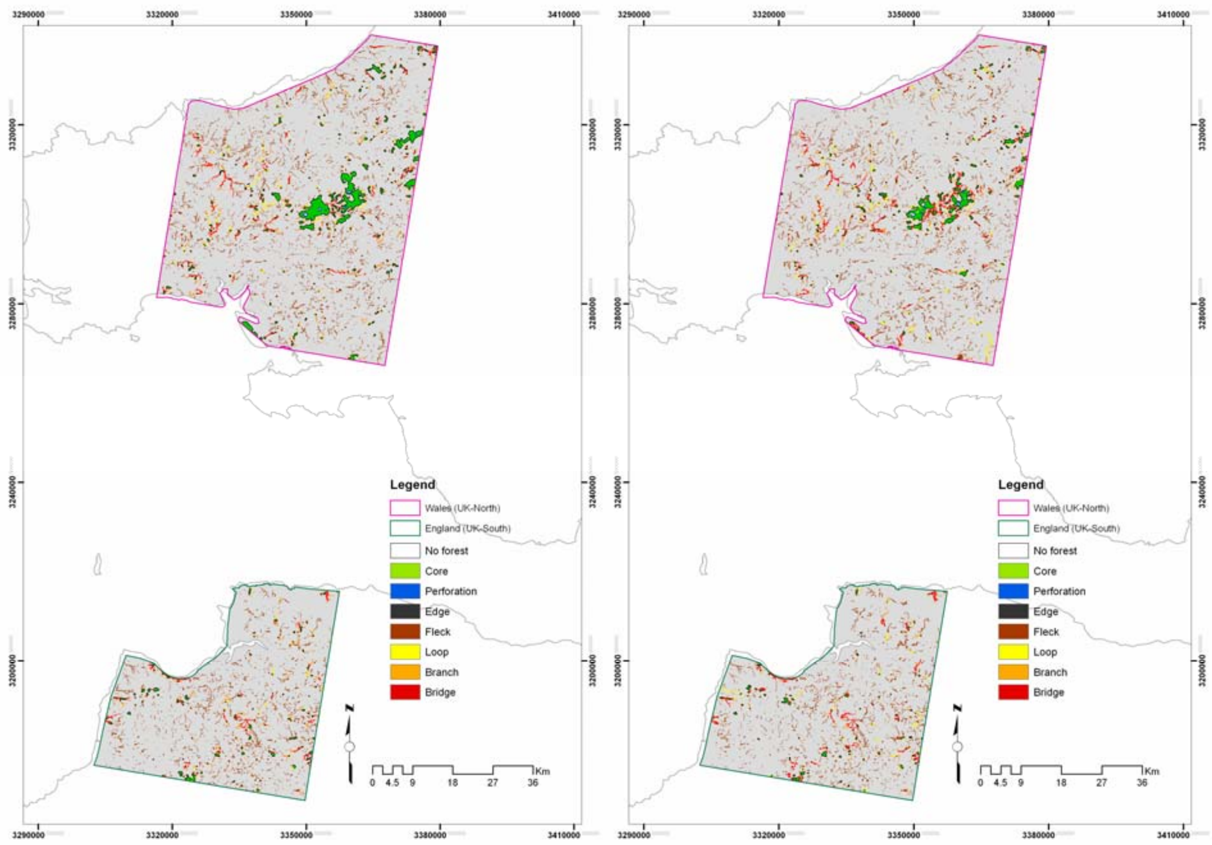


Figure 89: results of the GUIDOS analysis on multitemporal high resolution forest maps at the year 1990 and 2000.

The relative presence of core areas in both the two test sites was extremely limited.

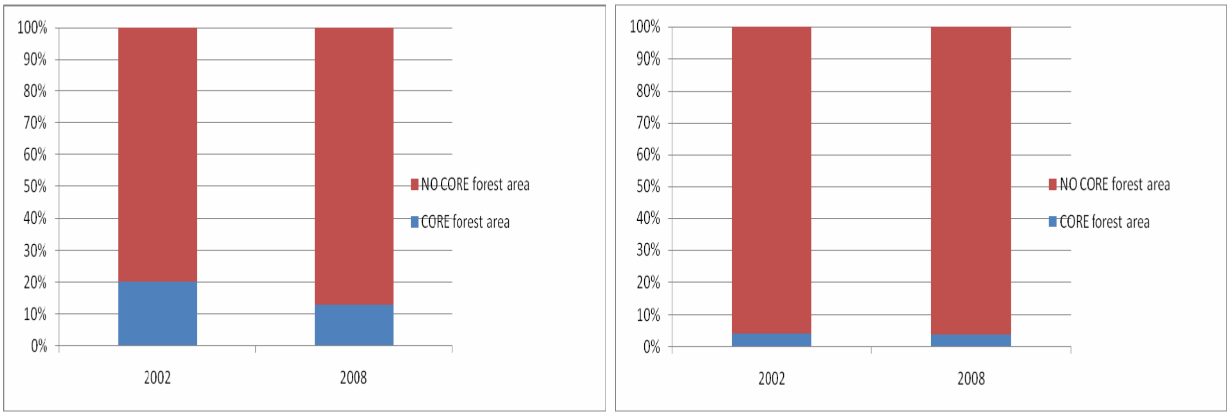


Figure 90: relative area of core and no core forest areas on high resolution maps available in the Atlantic test areas: for Wales on left and England on the right.

The Wales was interested by a limited reduction of forest area (- 1% yearly) while the area in south England was interested by a very limited increase in forest area (+ 0.1% yearly).

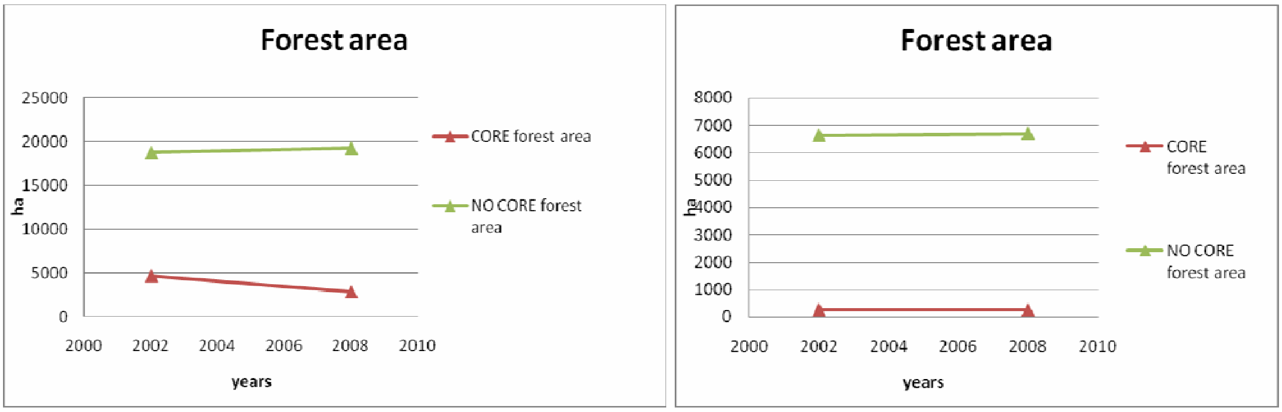


Figure 91: relative area of core and no core forest areas on high resolution maps available in the Atlantic test areas: for Wales on left and England on the right.

8.4.1 Species unspecific analysis

The species unspecific analysis was carried out on both the test sites in the Atlantic area.

In both of them, as expected, the average values of the FVRR index was higher in core areas than in no core areas in both the study years.

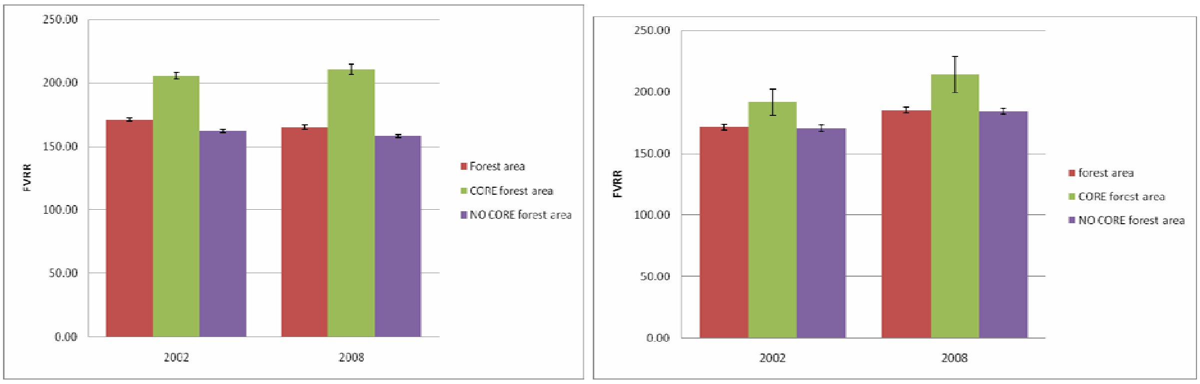
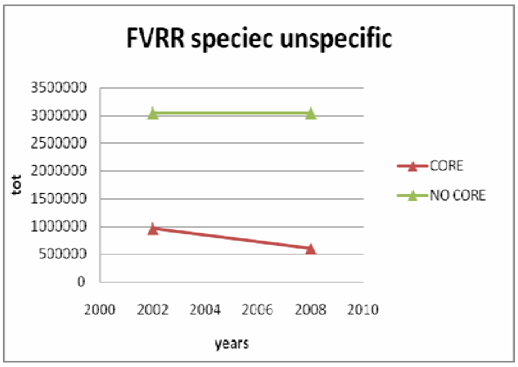


Figure 92: average values with standard error of the species unspecific index of FVRR from high resolution data for core and no core forest areas (Wales on the left and England of the right).

The temporal trends of the FVRR index in the two test sites was different. In Wales the index in core area was decreasing (- 6% yearly) while in no core areas it was stable. In England both in core and in no core area the index was increasing (+ 1.4% yearly).



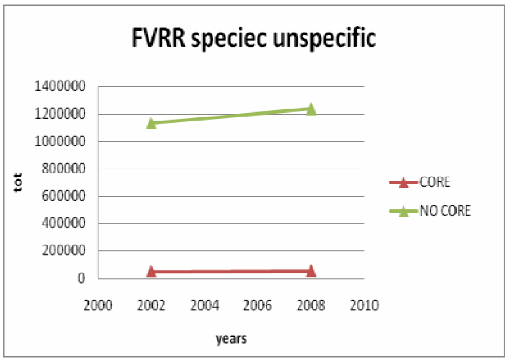


Figure 93: trends in species unspecific index of FVRR for high resolution data in forest areas. (Wales on the left and England of the right).

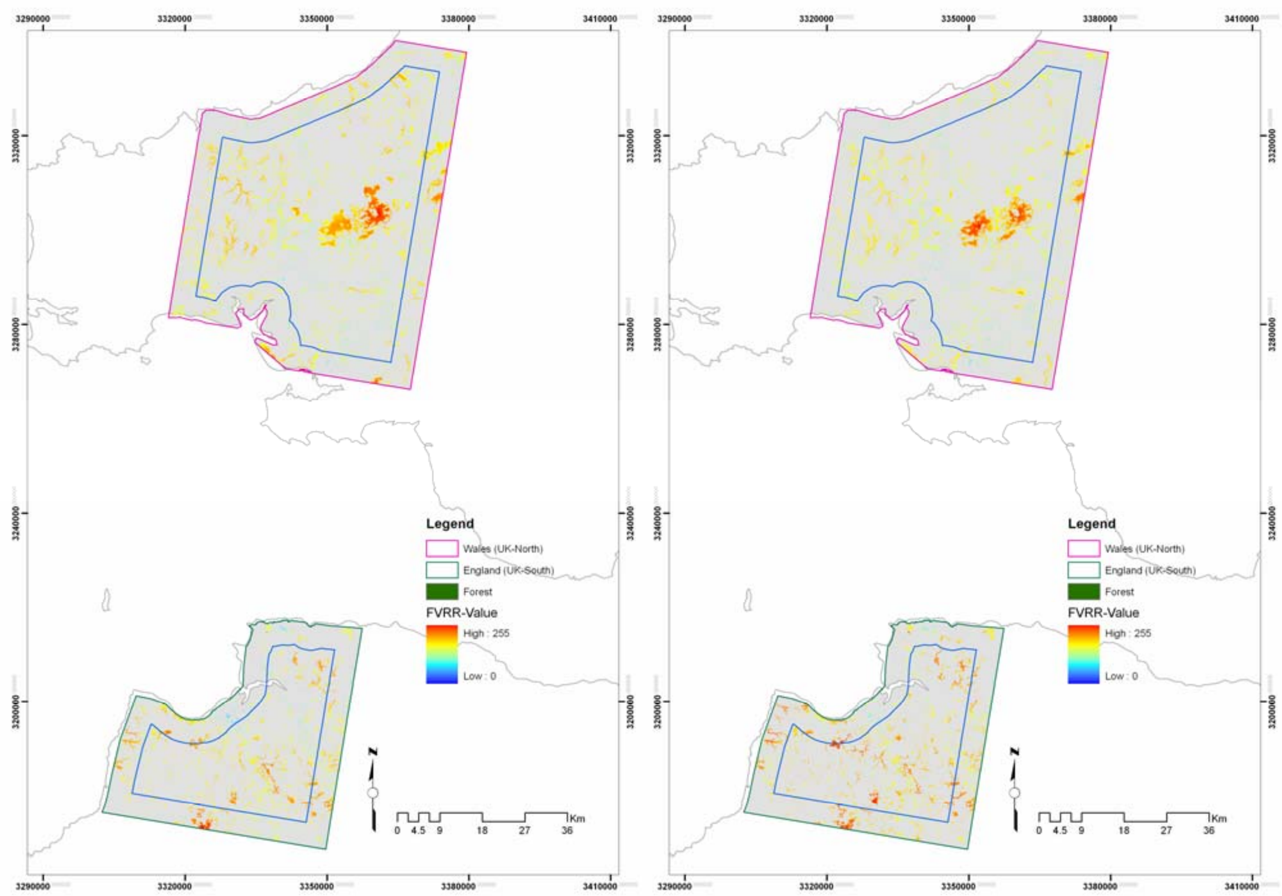


Figure 94: the index FVRR based on multitemporal high resolution forest maps from Wales (left) and England (right).

8.4.2 Species specific analysis

Here following the results from the calculation of the species specific FVRR on the basis of high resolution data from 2002 and 2008 for the two considered study sites in the Atlantic area: Wales and England.

The FVRR average values for birds are significantly higher in core areas than in no core areas. No significant differences were noticed between the two study sites.

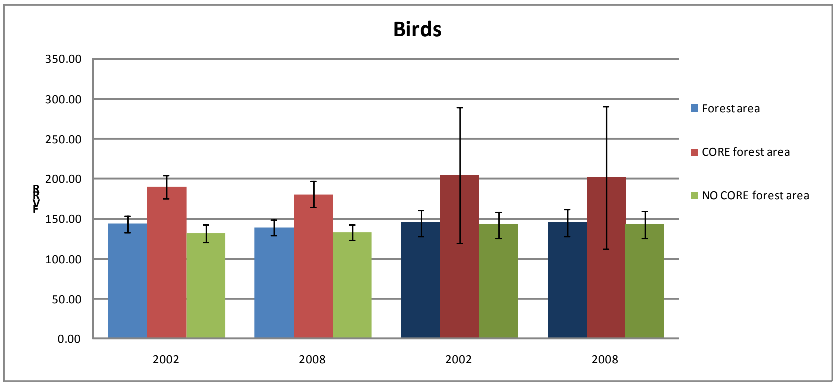


Figure 95: multitemporal average values with standard error of the birds index of FVRR on high resolution data for core and no core forest areas. On the left from Wales and on the right from England (darker colours).

The temporal trends of birds FVRR was negative in core areas (- 7% yearly in Wales, 0.7% yearly in England) but very little positive in no core areas. For this reason the general trend in forest area in Wales was -1.4% yearly and 0.2% positive in England.

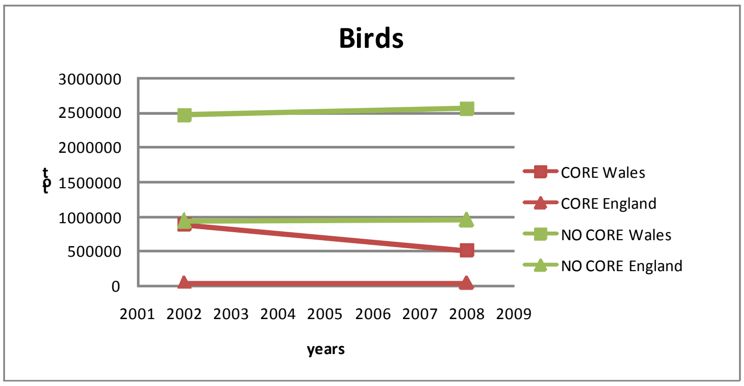


Figure 96: comparison of trends in birds index of FVRR on high resolution data for core and no core forest areas.

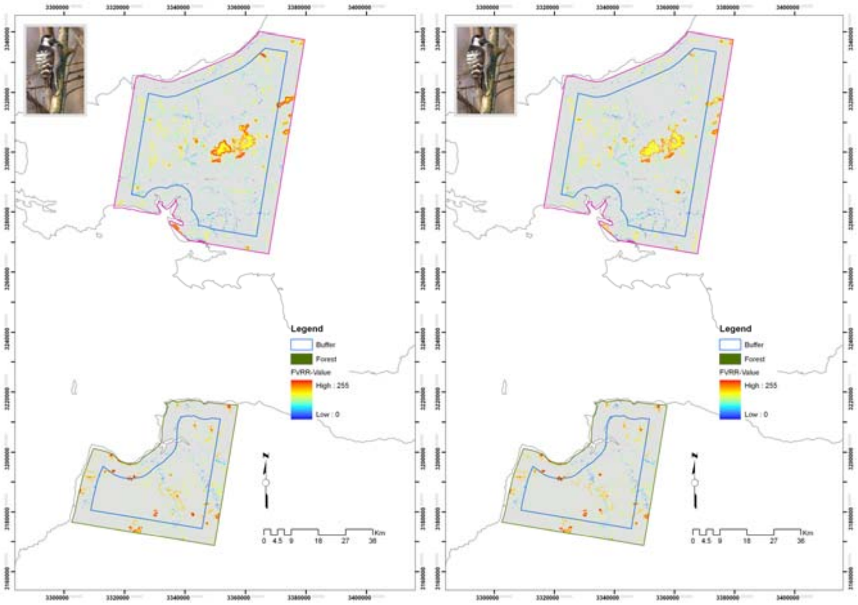


Figure 97: the index FVRR for birds umbrella species based on high resolution forest maps at the year 2002 and 2008.

For butterflies also the FVRR values are higher in core areas.

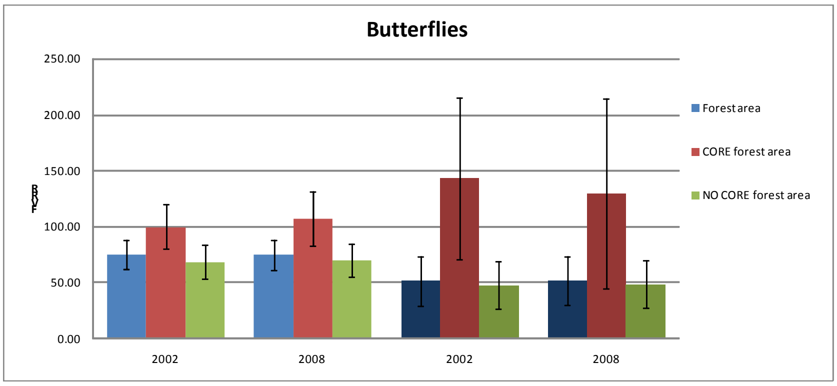


Figure 98: multitemporal average values with standard error of the butterflies index of FVRR on high resolution data for core and no core forest areas. On the left from Wales and on the right from England (darker colours).

The temporal trend figures for butterflies are the same of those ones for Birds.

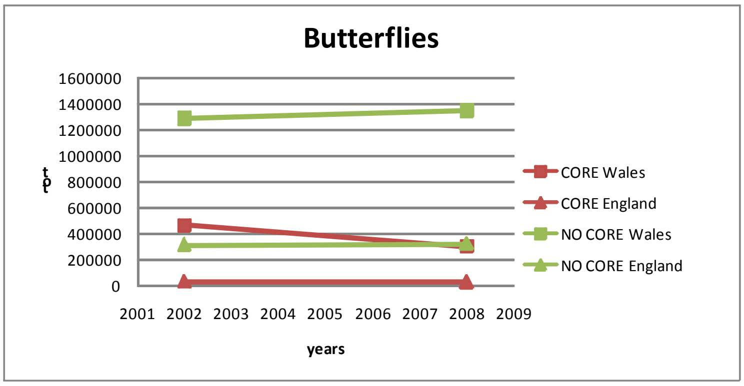


Figure 99: comparison of trends in butterflies index of FVRR on high resolution data for core and no core forest areas.

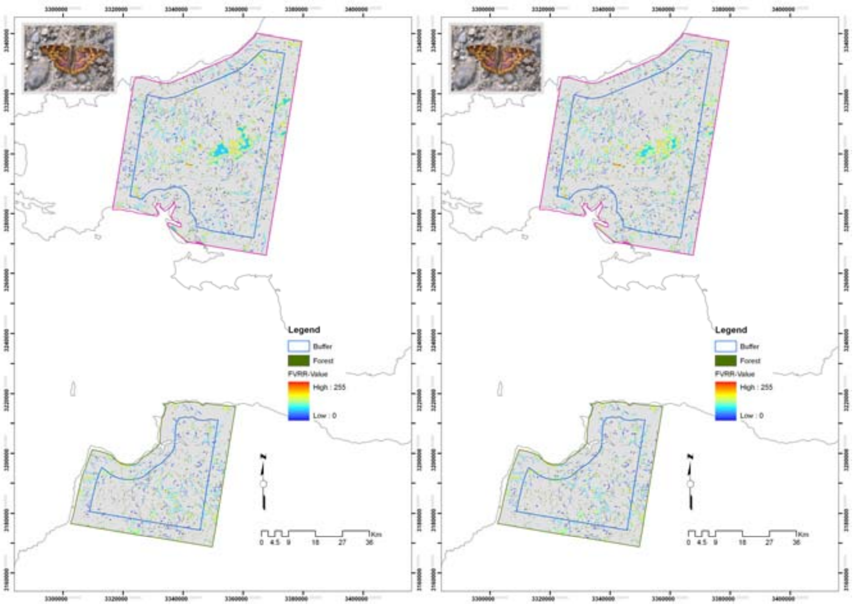


Figure 100: the index FVRR for butterflies umbrella species based on low resolution (LR) forest map at the year 2000.

Again also for marten the FVRR values are higher in core areas, even if these difference are significantly in England because of the very limited extension of core areas.

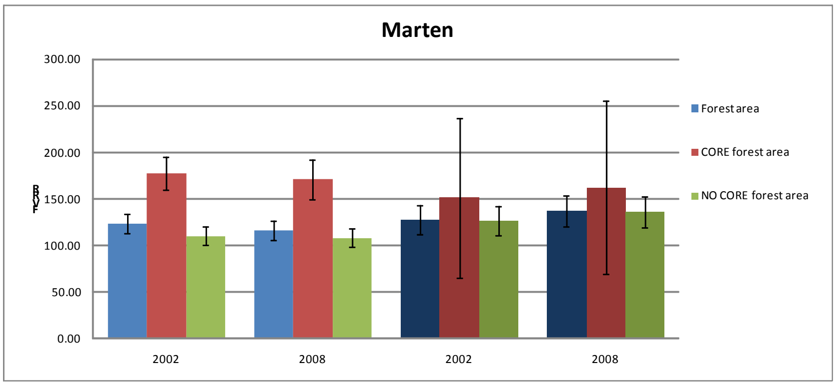


Figure 101: multitemporal average values with standard error of the marten index of FVRR on high resolution data for core and no core forest areas. On the left from Wales and on the right from England (darker colours).

For marten similar temporal figures as for birds and butterflies, in Walles forests the FVRR index decrease 1.9% yearly while it increases in England with a rate of 1.4% yearly.

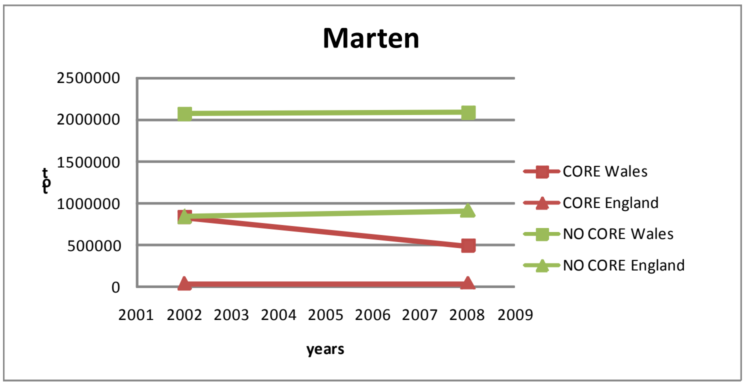


Figure 102: comparison of trends in marten index of FVRR on high resolution data for core and no core forest areas.

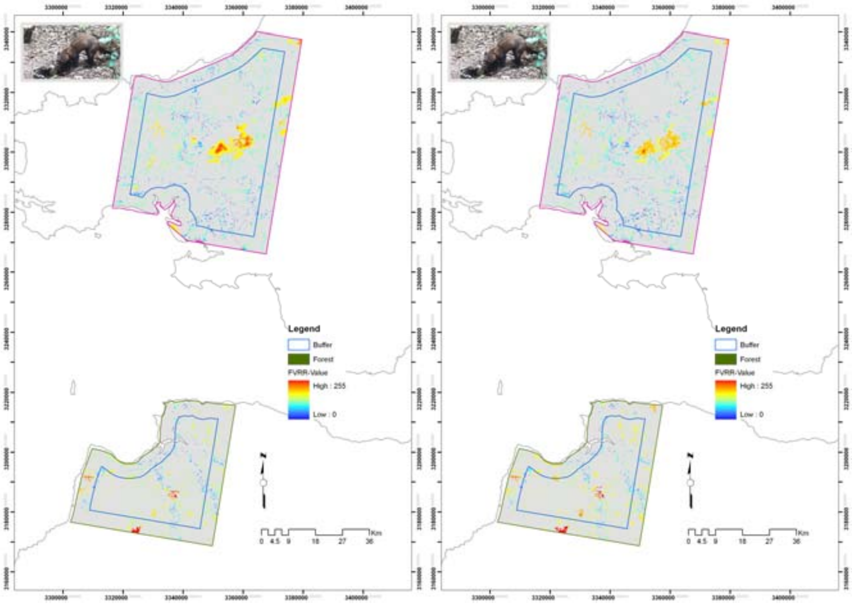


Figure 103: the index FVRR for marten umbrella species based on low resolution (LR) forest map the year 2000.

The relationship of roe deer FVRR index is similar to the previous umbrella species.

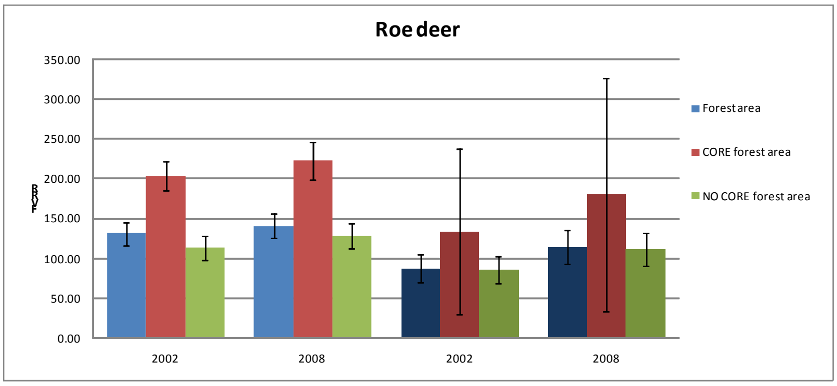


Figure 104: multitemporal average values with standard error of the roe deer index of FVRR on high resolution data for core and no core forest areas. On the left from Wales and on the right from England (darker colours).

For roe deer the temporal trend is positive in both the study sites (yearly 0.2% in Wales and 5.2% in England). Just in core areas in Wales the trend is negative: -5.5% yearly.

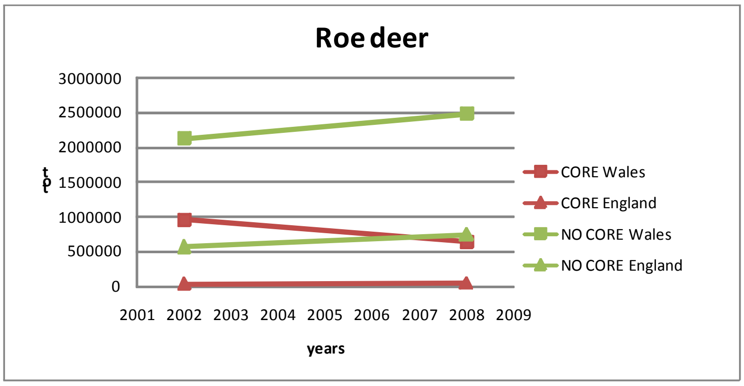


Figure 105: comparison of trends in roe deer index of FVRR on high resolution data for core and no core forest areas.

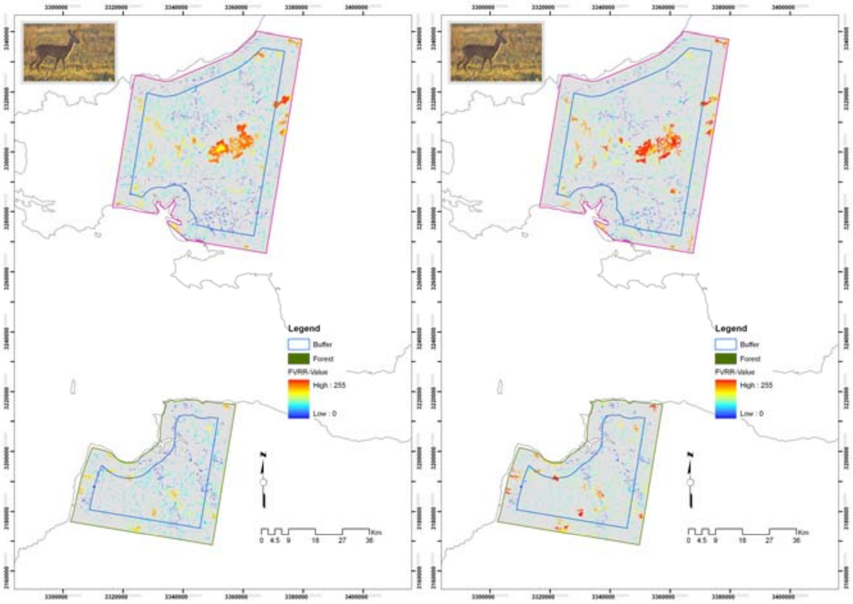


Figure 106: the index FVRR for roe deer umbrella species based on low resolution (LR) forest map at the year 2000.

The wolf the average FVRR index is significantly higher in core areas than in no core areas just in Wales.

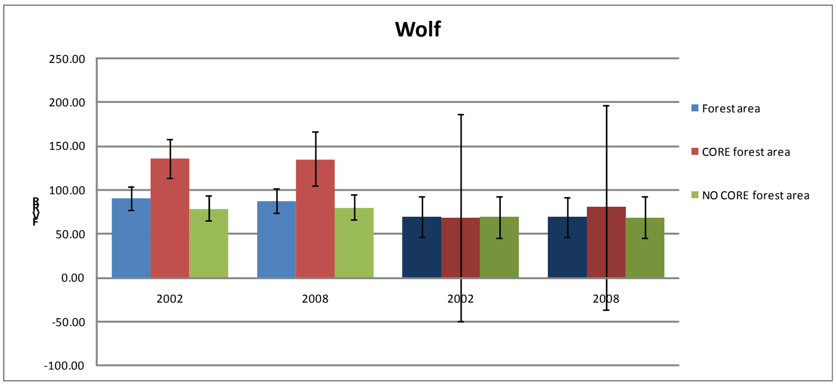


Figure 107: multitemporal average values with standard error of the wolf index of FVRR on high resolution data for core and no core forest areas. On the left from Wales and on the right from England (darker colours).

The temporal trend is negative in Wales (-1.5% yearly) but positive in England (+0.1 % yearly).

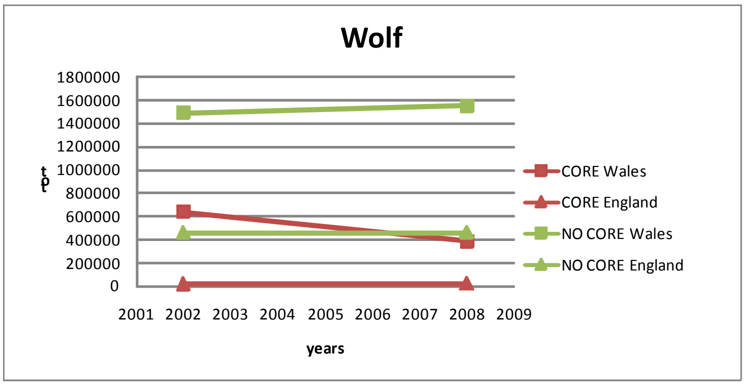


Figure 108: comparison of trends in wolf index of FVRR on high resolution data for core and no core forest areas.

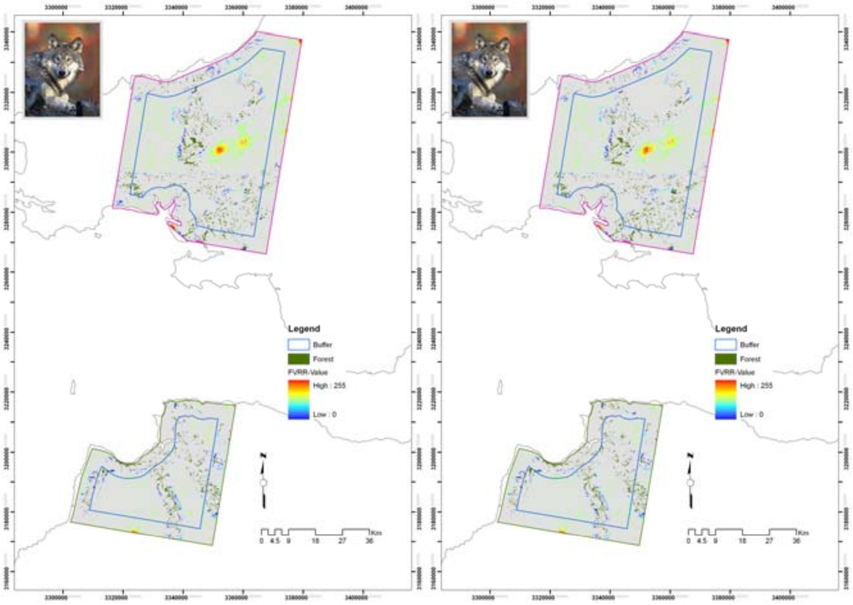


Figure 109: the index FVRR for wolf umbrella species based on multitemporal low resolution (LR) forest map at the year 2000.

The general species specific FVRR values obtained averaging the five indexes calculated for the five umbrella species are significantly higher in core areas than in no core areas just in Wales.

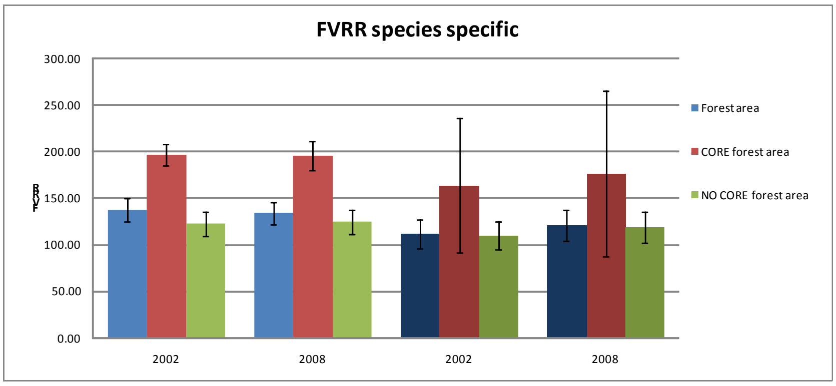


Figure 110: multitemporal average values with standard error of the aggregated index of FVRR averaged from the five considered umbrella species on high resolution data for core and no core forest areas. On the left from Wales and on the right from England (darker colours).

The general trend is negative in Wales (- 1.4 yearly) and slightly positive in England (1.5 yearly).

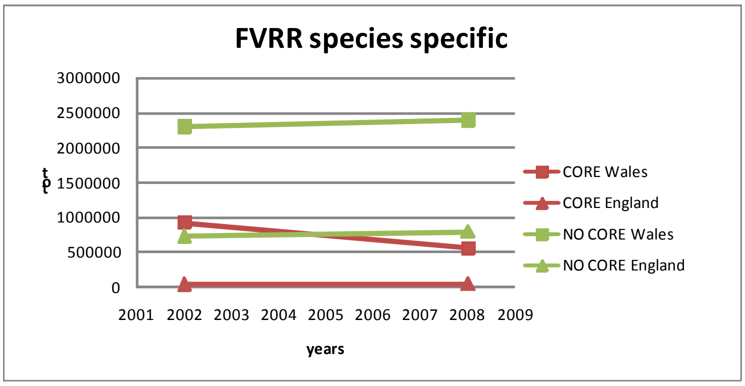


Figure 111: comparison of trends in the aggregated index of FVRR calculated averaging the five considered umbrella species on high resolution data for core and no core forest areas.

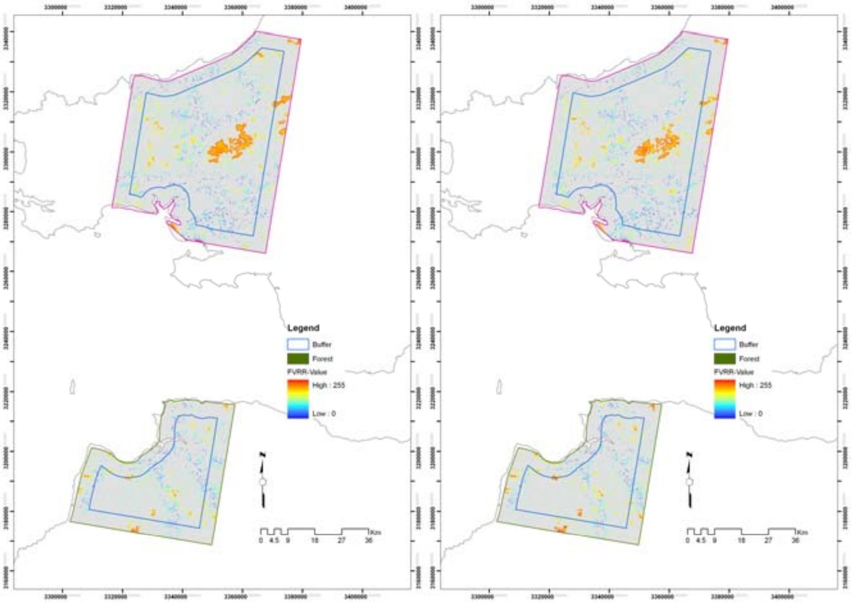


Figure 112: the index FVRR summing up all the five umbrella species based on low resolution (LR) forest map at the year 2000.

8.5 *Boreal test area*

In the Boreal area two test sites are available both in Finland one in the north part (9153 km^2 wide) and one in south (7019 km^2 wide). Here we presented the results from the northest test site. In order to make possible the comparability between high and low resolution maps the following data were extracted just from the smallest inner high resolution subarea.

In this area on the basis of high resolution maps the forest area slightly decreased during the period 1950-1990 and then it strongly decrease in the last 10 years. On the basis of low resolution data the forest area was basically unaltered during approximately the same time. This process was common for all the test areas, high resolution data are able to detect changes in forest cover that are ignored in the low resolution maps because under the minimum mapping unit of the CORINE land cover system of nomenclature (25 ha).

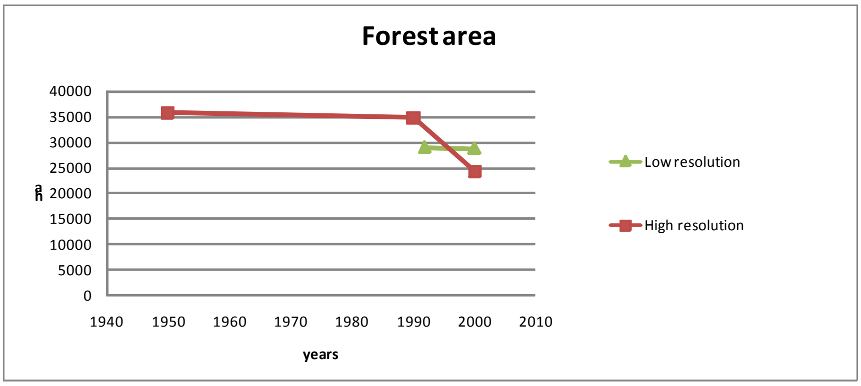


Figure 113: comparison of the trends in forest area on the basis of low and high resolution data in the northern Finnish test site.

With the exception of the most recent high resolution map the ratio between core and no core forest area in the different years are similar. The reduction of the forest area in the high resolution map available for the year 2000 determined a consequent strong fragmentation effect.
The high resolution forest maps at the year 2000 for both the test sites in Finland have a geometric details completely different from the remaining maps. Probably a fragmentation process is really present in these areas but its amplitude is mainly due to the different spatial characteristics of these maps. For these reasons the following results here presented from these high resolution maps at the year 2000 have to be considered with caution.

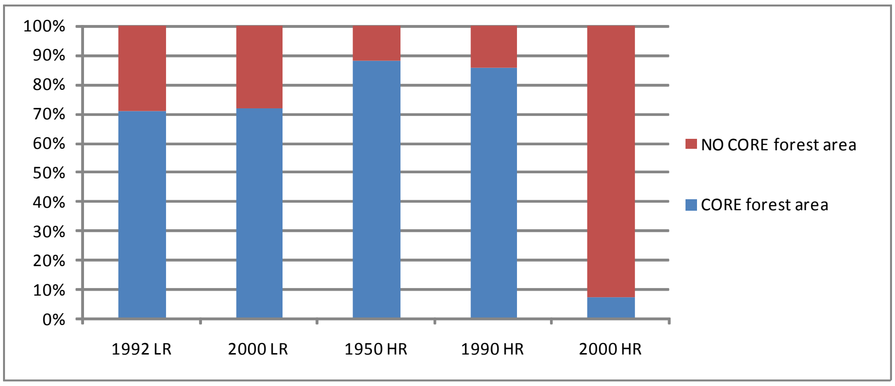
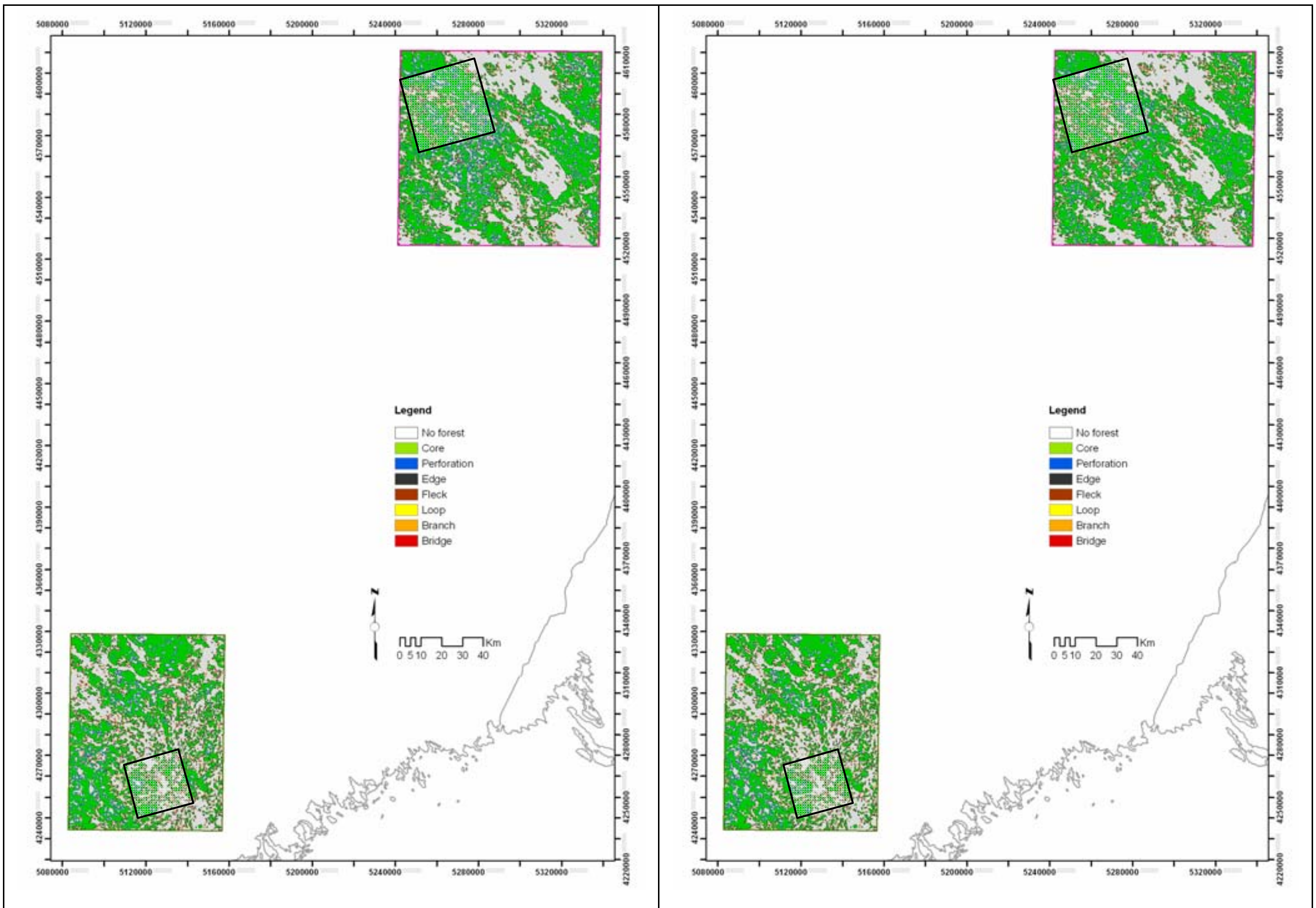


Figure 114: relative area of core and no core forest areas in high (HR) and low (LR) resolution maps available in the northern test area in Finland.



LR 1990

LR 2000

Figure 115: result of the GUIDOS analysis on low (LR) forest map at the year 2000 for the two test sites in the Boreal area.

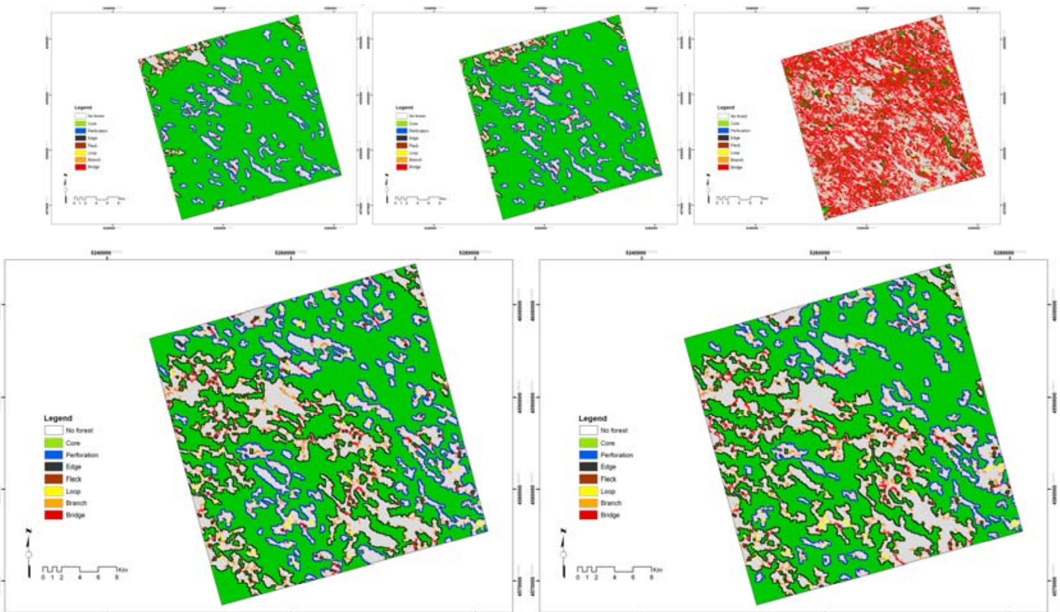


Figure 116: result of the GUIDOS analysis on multitemporal high (above, from left to right: 1950, 1990, 2000) and low (below, from left to right: 1992 and 2000) resolution forest maps from the inner subarea from the north test site in Finland.

8.5.1 Species unspecific analysis

The result of the application of the general FVRR model for the north test site in Finland showed, as expected, higher values in core areas than in no core areas, for all the different resolutions and for all the study years.

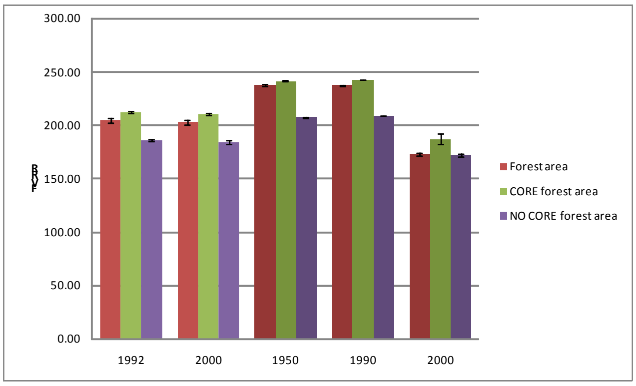


Figure 117: average values with standard errors of the species unspecific index of FVRR comparing low and high (darker colours) resolution data for core and no core areas. Data from the northern test site in Finland.

The temporal trends of the FVRR index was strongly affected by the values of the high resolution forest map at the year 2000. It is anyhow plausible a general decreasing trend in core area and an increasing trend in no core areas.

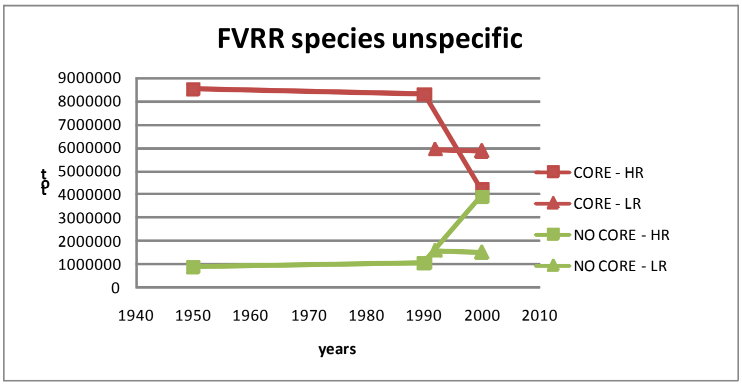


Figure 118: temporal trends of the overall values of the species unspecific index of FVRR comparing low and high resolution data for core and no core areas. Data from the northern test site in Finland.

8.5.2 Species specific analysis

The results of the FVRR for birds in this test area demonstrated no significant differences in the average values of core and no core areas for the different resolutions and study years.

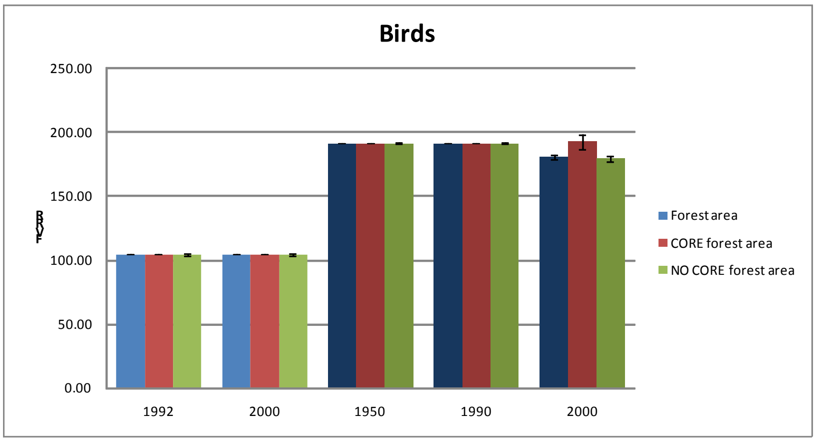


Figure 119: average values with standard error of the birds index of FVRR from low and high (darker) resolution data for core and no core forest areas.

The temporal trends of the FVRR index was generally increasing in no core areas and decreasing in core areas, at least on the basis of the high resolution data.

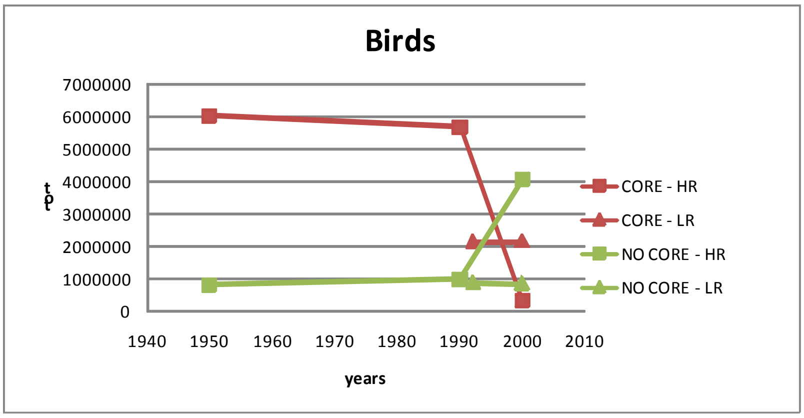
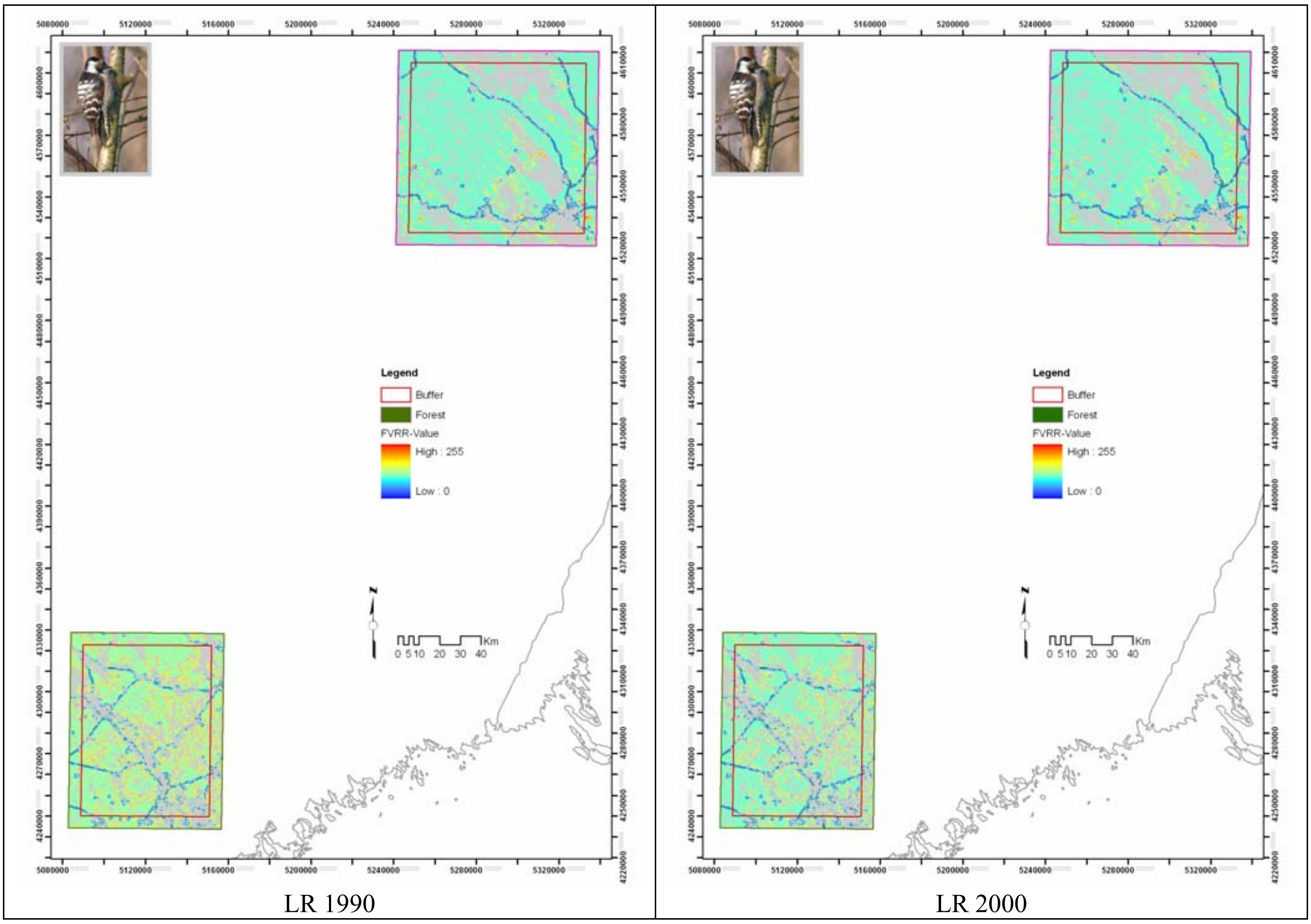


Figure 120: comparison of trends in birds index of FVRR on low and high resolution data for core and no core forest areas.



LR 1990 LR 2000

Figure 121: the index FVRR for birds umbrella species based on multitemporal low resolution (LR) forest map.

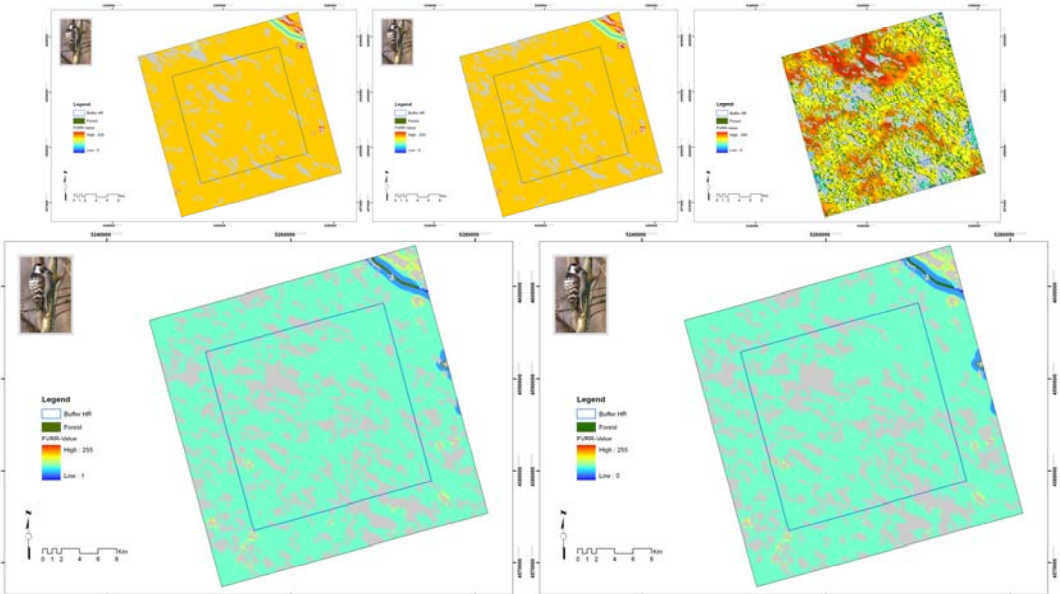


Figure 122: result of the FVRR analysis for birds on multitemporal high (above, from left to right: 1950, 1990, 2000) and low (below, from left to right: 1992 and 2000) resolution forest maps from the inner subarea from the north test site in Finland.

The results of the FVRR for butterflies in this test area demonstrated significant higher values in the average values in no core areas for the different resolutions and study years.

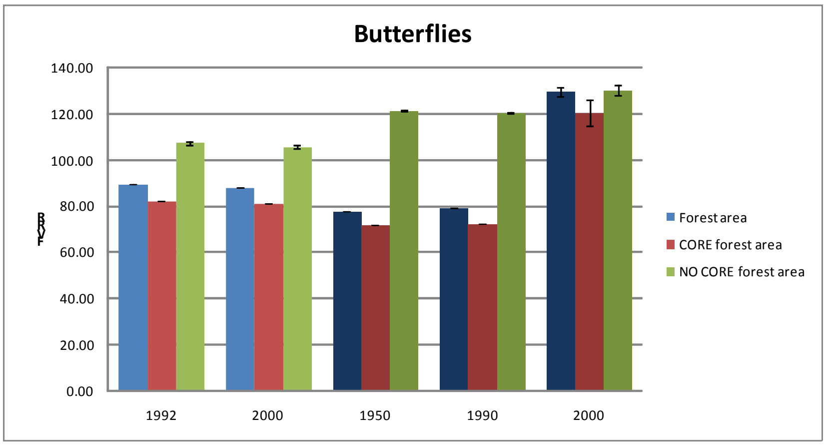


Figure 123: average values with standard error of the butterflies index of FVRR from low and high (darker) resolution data for core and no core forest areas.

The temporal trends on the basis of low resolution data seems to be decreasing in both core and no core areas and also in core areas on the basis of high resolution maps.

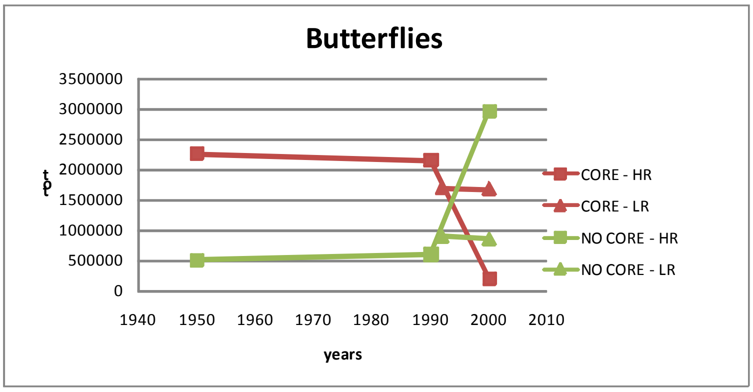
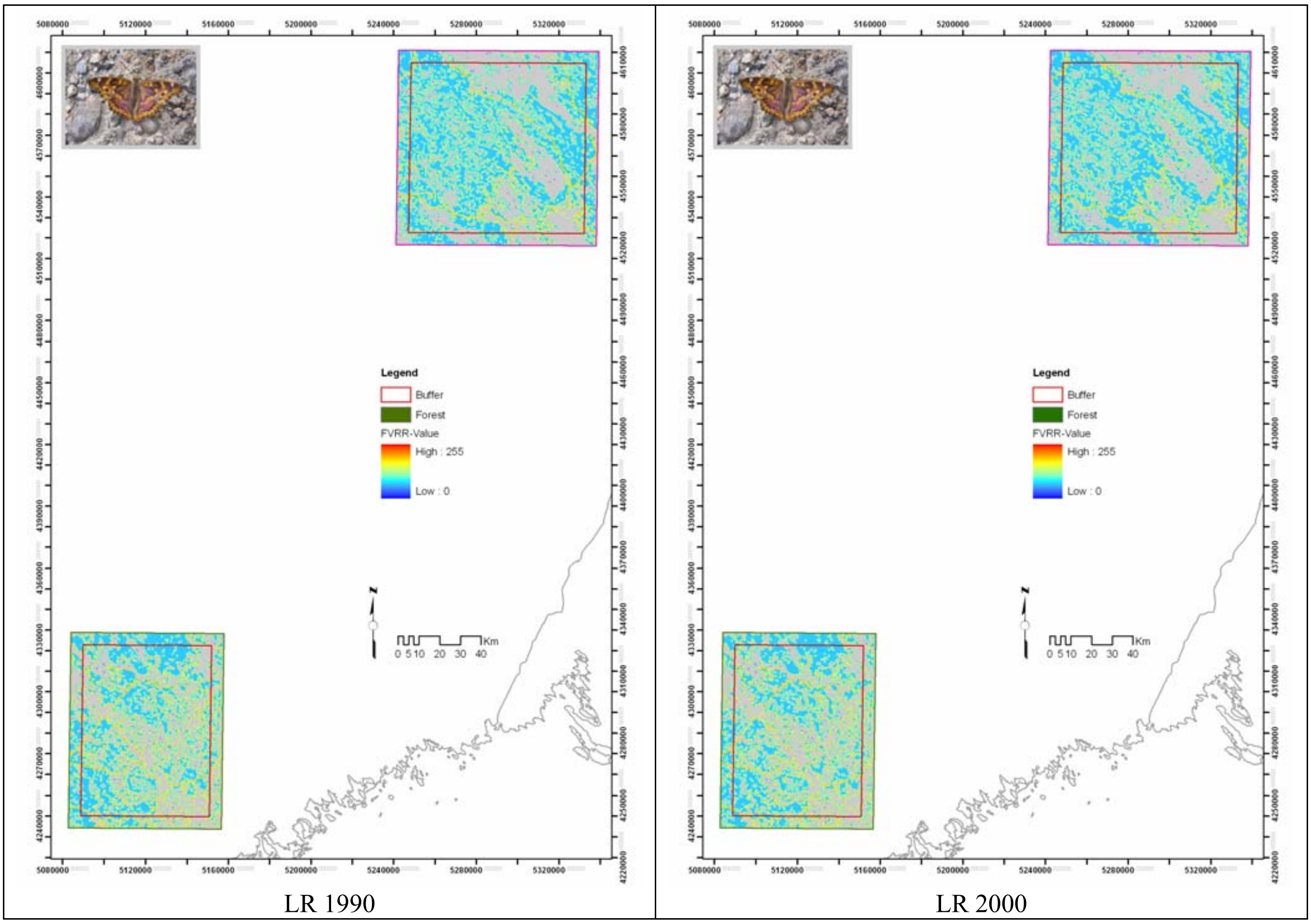


Figure 124: comparison of trends in butterflies index of FVRR on low and high resolution data for core and no core forest areas.



LR 1990 LR 2000

Figure 125: the index FVRR for butterflies umbrella species based on multitemporal low resolution (LR) forest map.

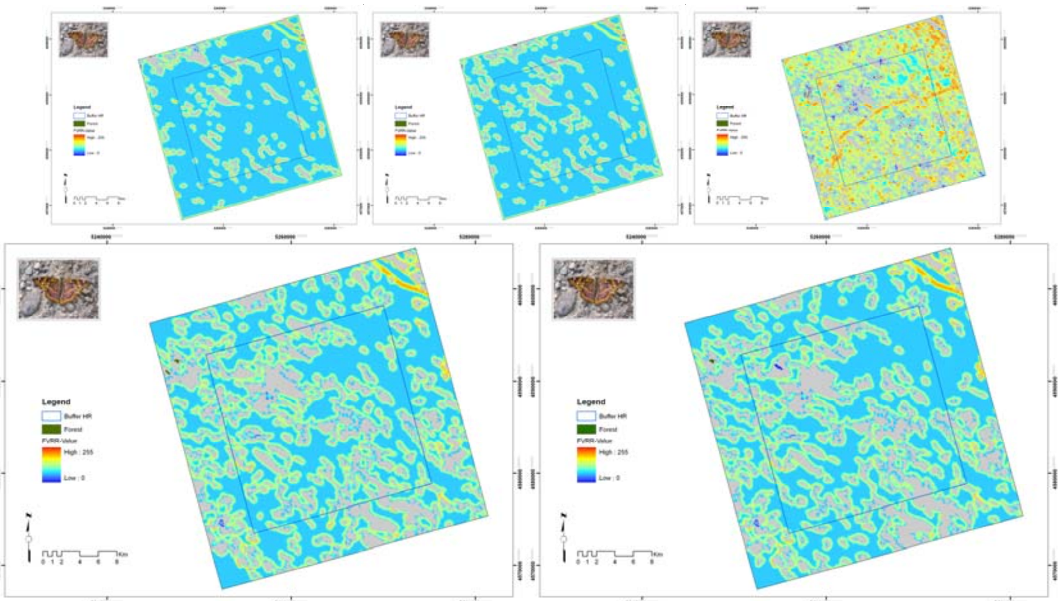


Figure 126: result of the FVRR analysis for butterflies on multitemporal high (above, from left to right: 1950, 1990, 2000) and low (below, from left to right: 1992 and 2000) resolution forest maps from the inner subarea from the north test site in Finland.

The results of the FVRR for marten in this test area demonstrated significant higher values in the average values in core areas for the different resolutions and study years.

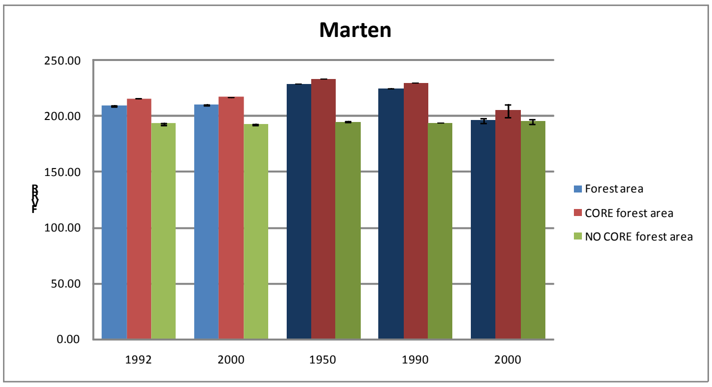


Figure 127: average values with standard error of the marten index of FVRR from low and high (darker) resolution data for core and no core forest areas.

The temporal trends in the FVRR index for marten seems to be slightly increasing in no core areas and decreasing in core areas, at least on the basis of high resolution maps.

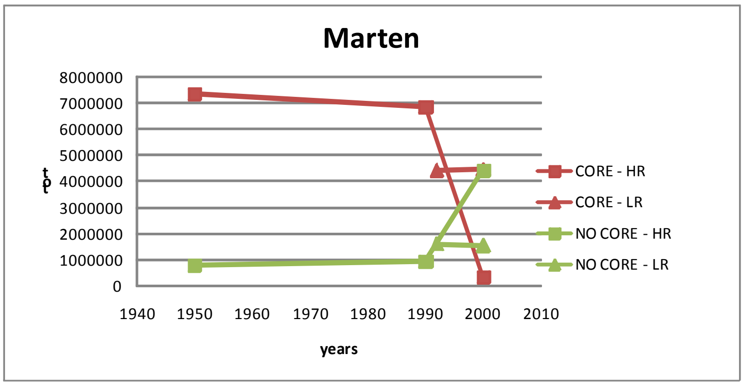
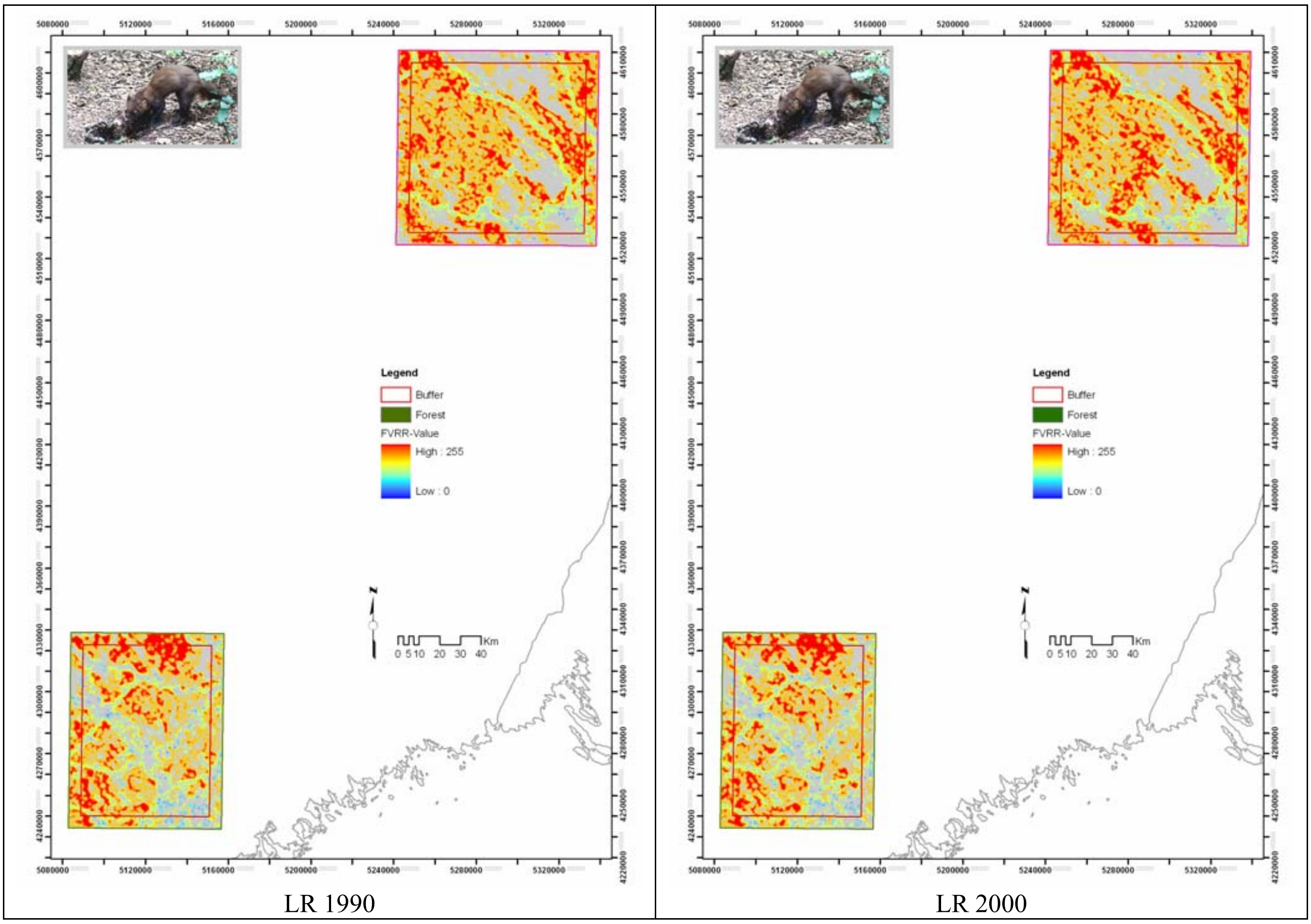


Figure 128: comparison of trends in marten index of FVRR on low and high resolution data for core and no core forest areas.



LR 1990 LR 2000

Figure 129: the index FVRR for marten umbrella species based on multitemporal low resolution (LR) forest map.

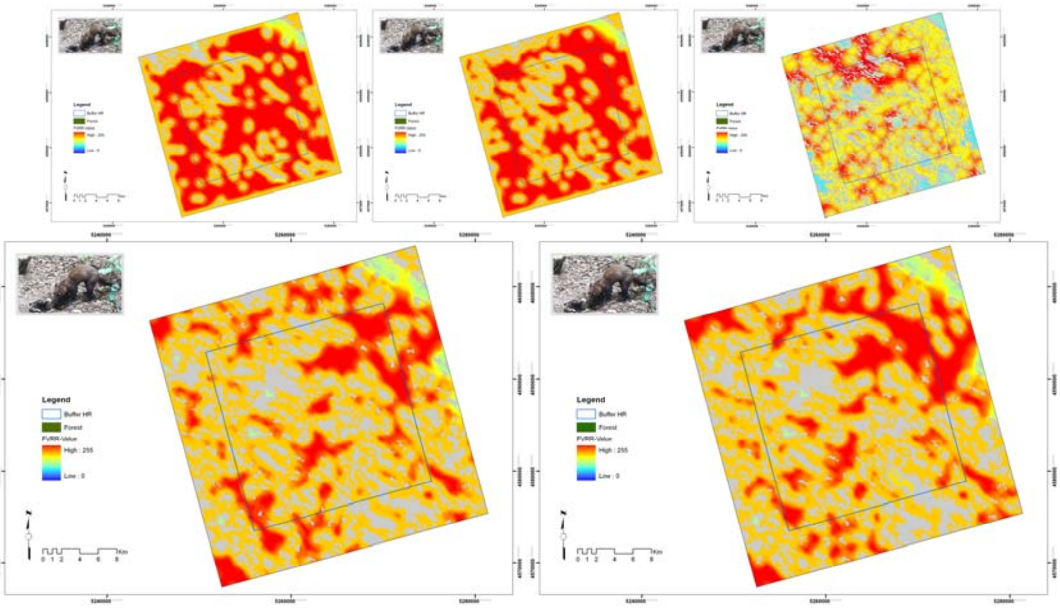


Figure 130: result of the FVRR analysis for marten on multitemporal high (above, from left to right: 1950, 1990, 2000) and low (below, from left to right: 1992 and 2000) resolution forest maps from the inner subarea from the north test site in Finland.

The results of the FVRR for roe deer in this test area demonstrated no significant differences between core and no core areas on the basis of low resolution forest maps. On high resolution maps from 1950 and 1990 it seems that the FVRR index could be higher in no core areas.

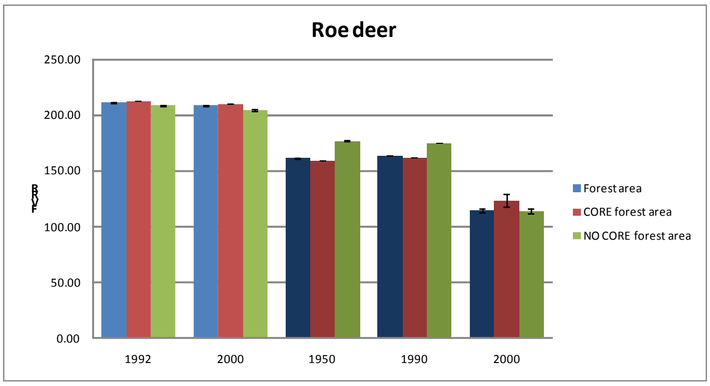


Figure 131: average values with standard error of the roe deer index of FVRR from low and high (darker) resolution data for core and no core forest areas.

The temporal trends of the FVRR index for roe deer showed little changes in low resolution data. The changes in high resolution maps in the last 10 years are instead meanly due to the different geometry of the latest forest map.

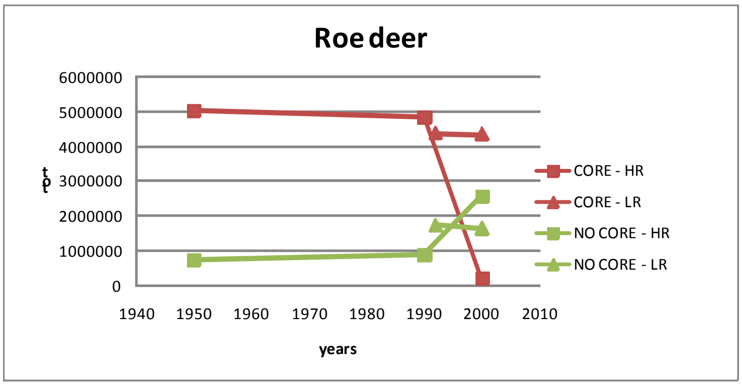
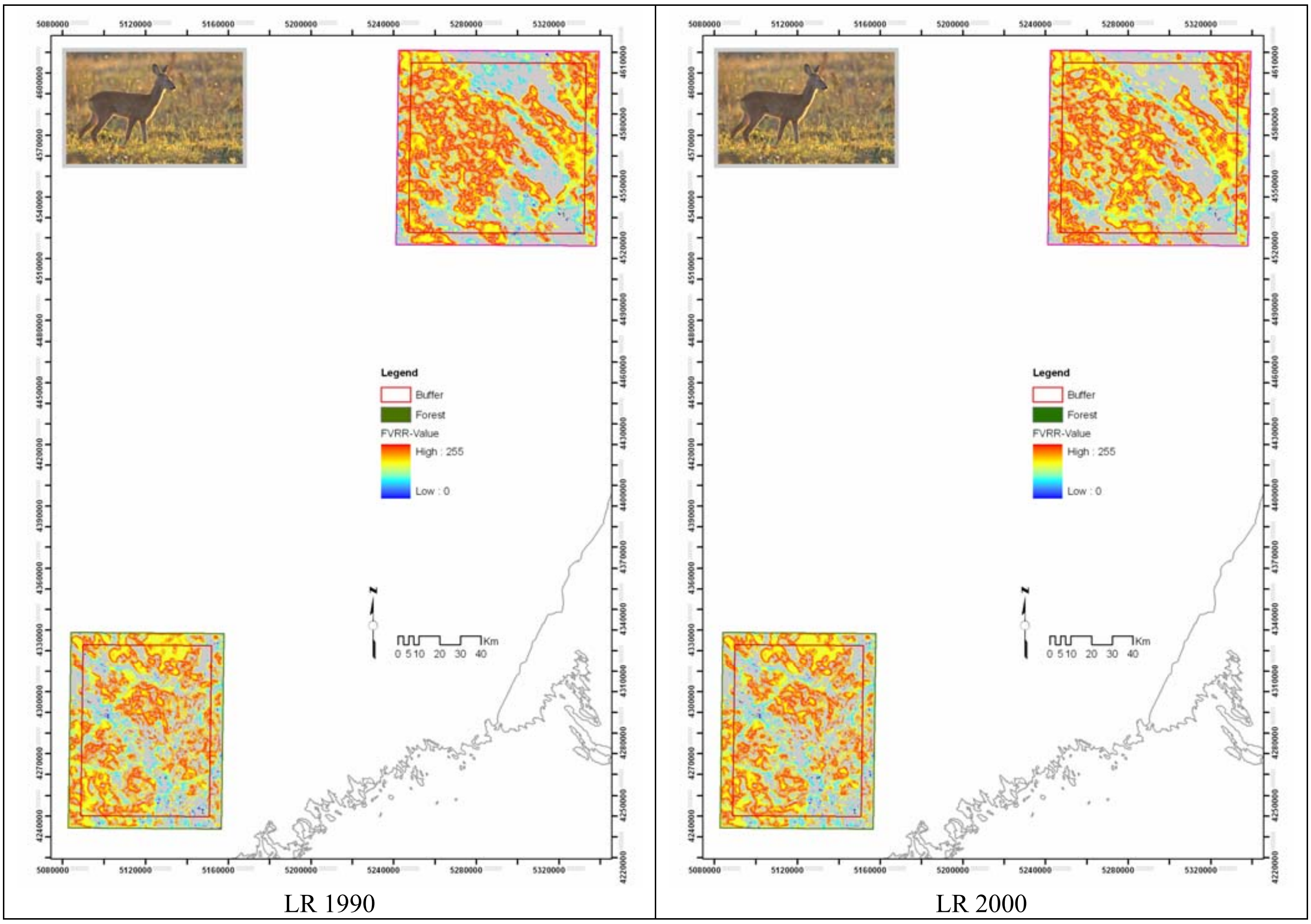


Figure 132: comparison of trends in roe deer index of FVRR on low and high resolution data for core and no core forest areas.



LR 1990 LR 2000

Figure 133: the index FVRR for roe deer umbrella species based on multitemporal low resolution (LR) forest map.

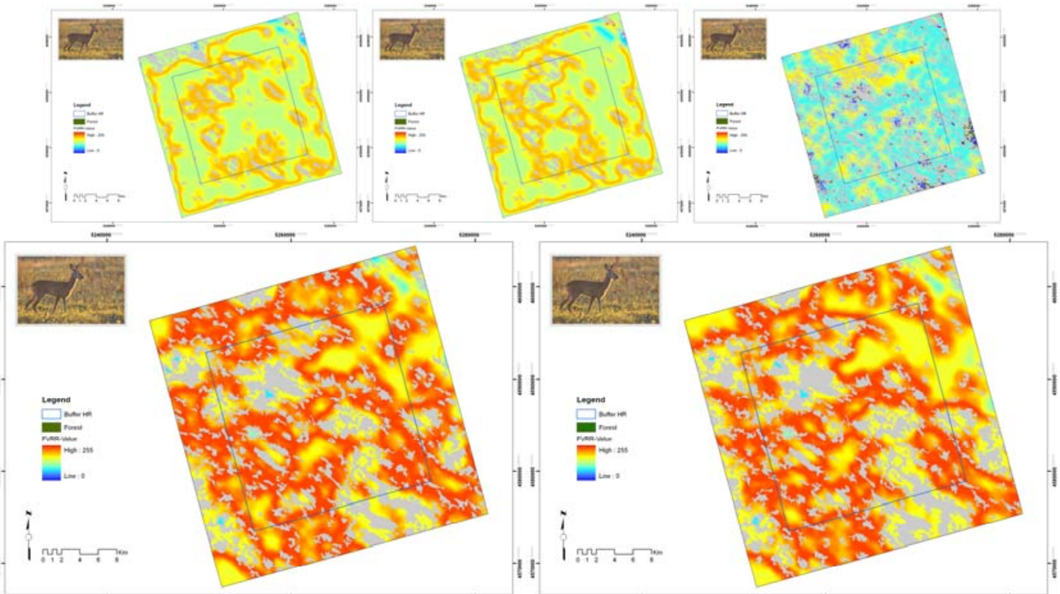


Figure 134: result of the FVRR analysis for roe deer on multitemporal high (above, from left to right: 1950, 1990, 2000) and low (below, from left to right: 1992 and 2000) resolution forest maps from the inner subarea from the north test site in Finland.

The results of the FVRR for wolf in this test area demonstrated to be higher in core than in no core areas both on the basis of low and high resolution forest maps.

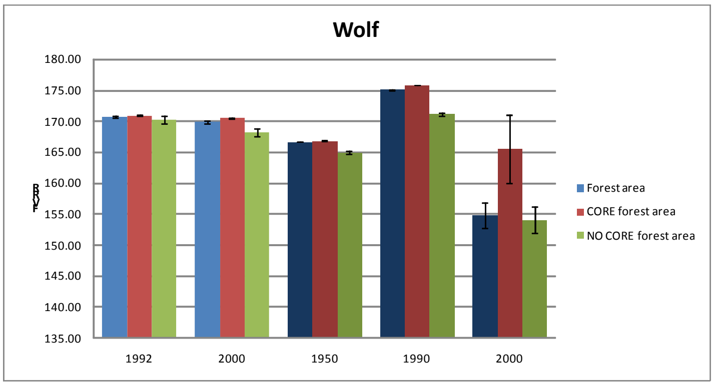


Figure 135: average values with standard error of the wolf index of FVRR from low and high (darker) resolution data for core and no core forest areas.

The temporal trends of the FVRR index for wolf did not demonstrated any consistent changes, with the exception of the outliers in the high resolution map of the tear 2000.

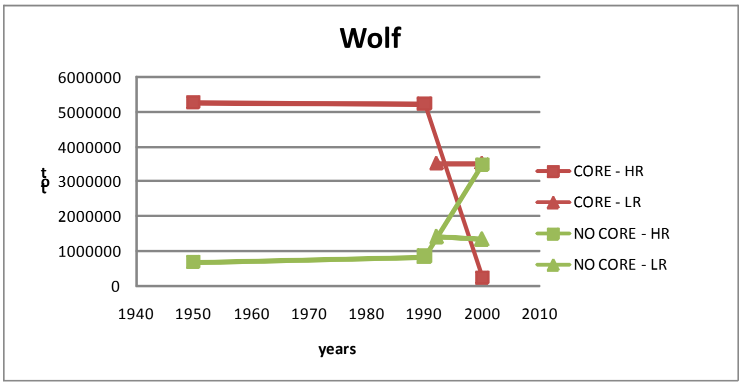
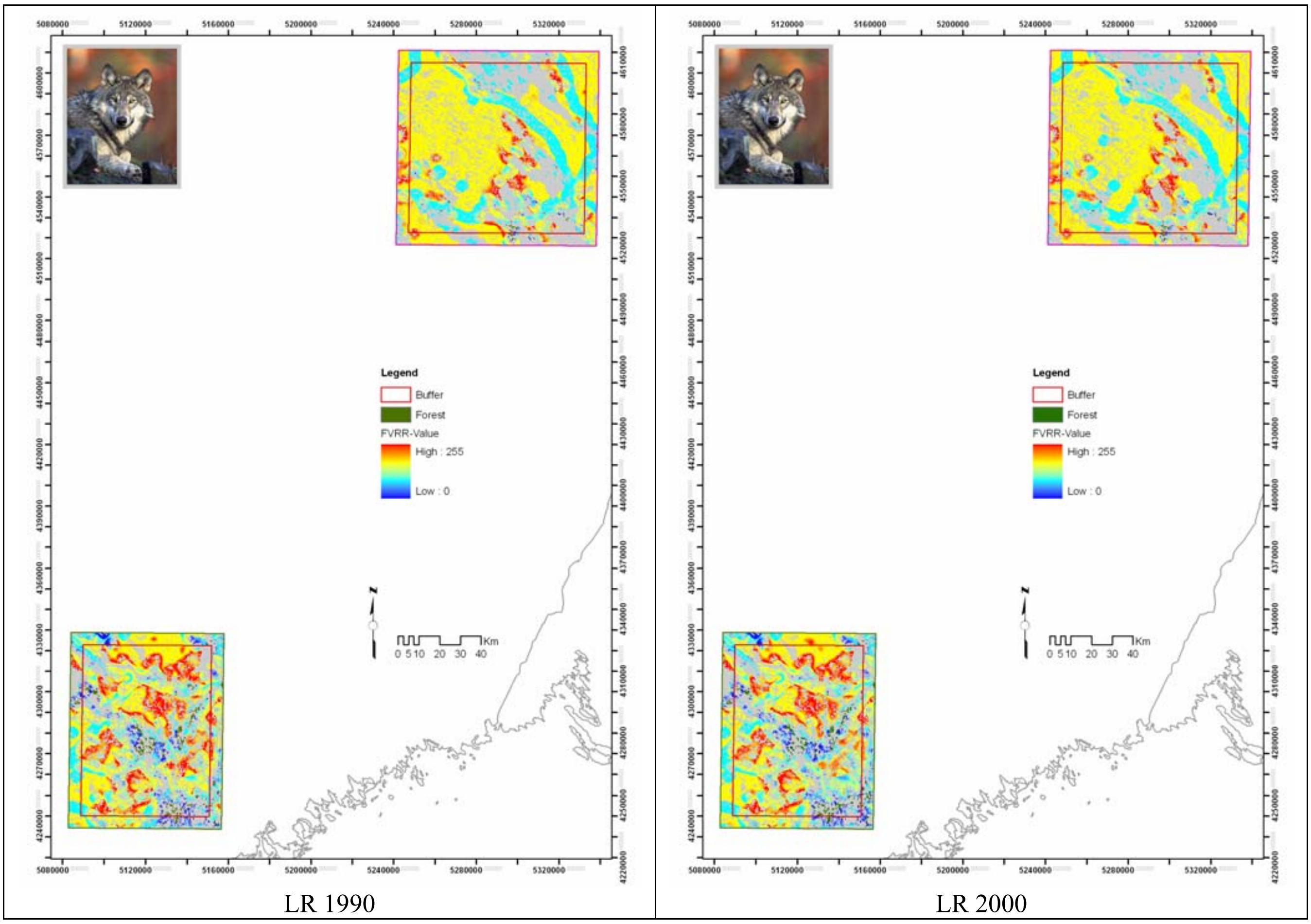


Figure 136: comparison of trends in wolf index of FVRR on low and high resolution data for core and no core forest areas.



LR 1990 LR 2000

Figure 137: the index FVRR for wolf umbrella species based on multitemporal low resolution (LR) forest map.

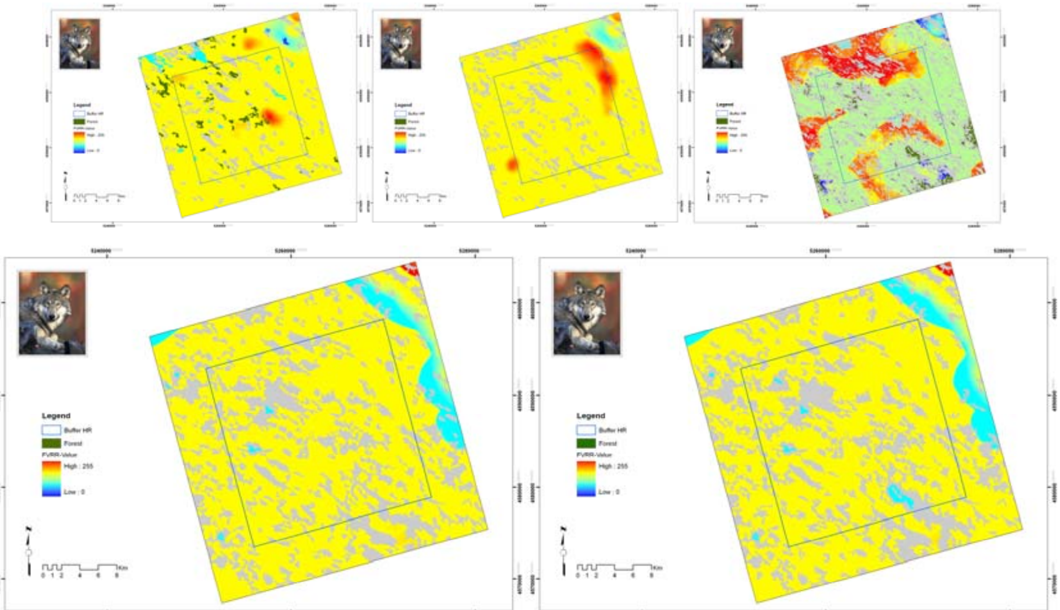


Figure 138: result of the FVRR analysis for wolf on multitemporal high (above, from left to right: 1950, 1990, 2000) and low (below, from left to right: 1992 and 2000) resolution forest maps from the inner subarea from the north test site in Finland.

Averaging the results of the FVRR index for all the five investigated umbrella species on the basis of low resolution data no significant differences were found between core and non core areas. On the basis of high resolution maps the results demonstrated higher average values in no core areas, at least from 1950 and 1990.

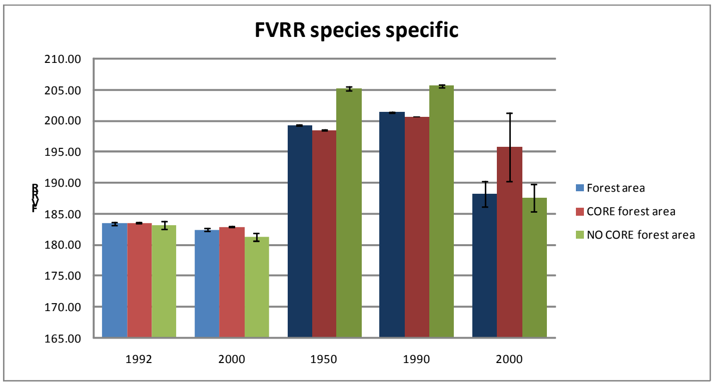


Figure 139: average values with standard error of the combined index of FVRR for all the five umbrella species from low and high (darker) resolution data for core and no core forest areas.

The temporal trends showed little changes both for core and no core and for high and low resolution data. With the exception of the outlier in year 2000 for high resolution data.

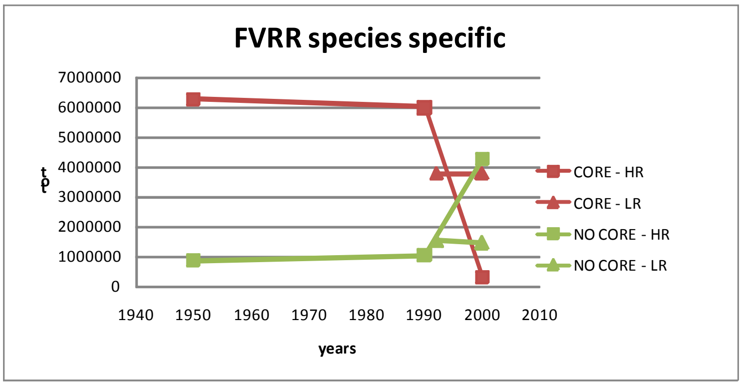
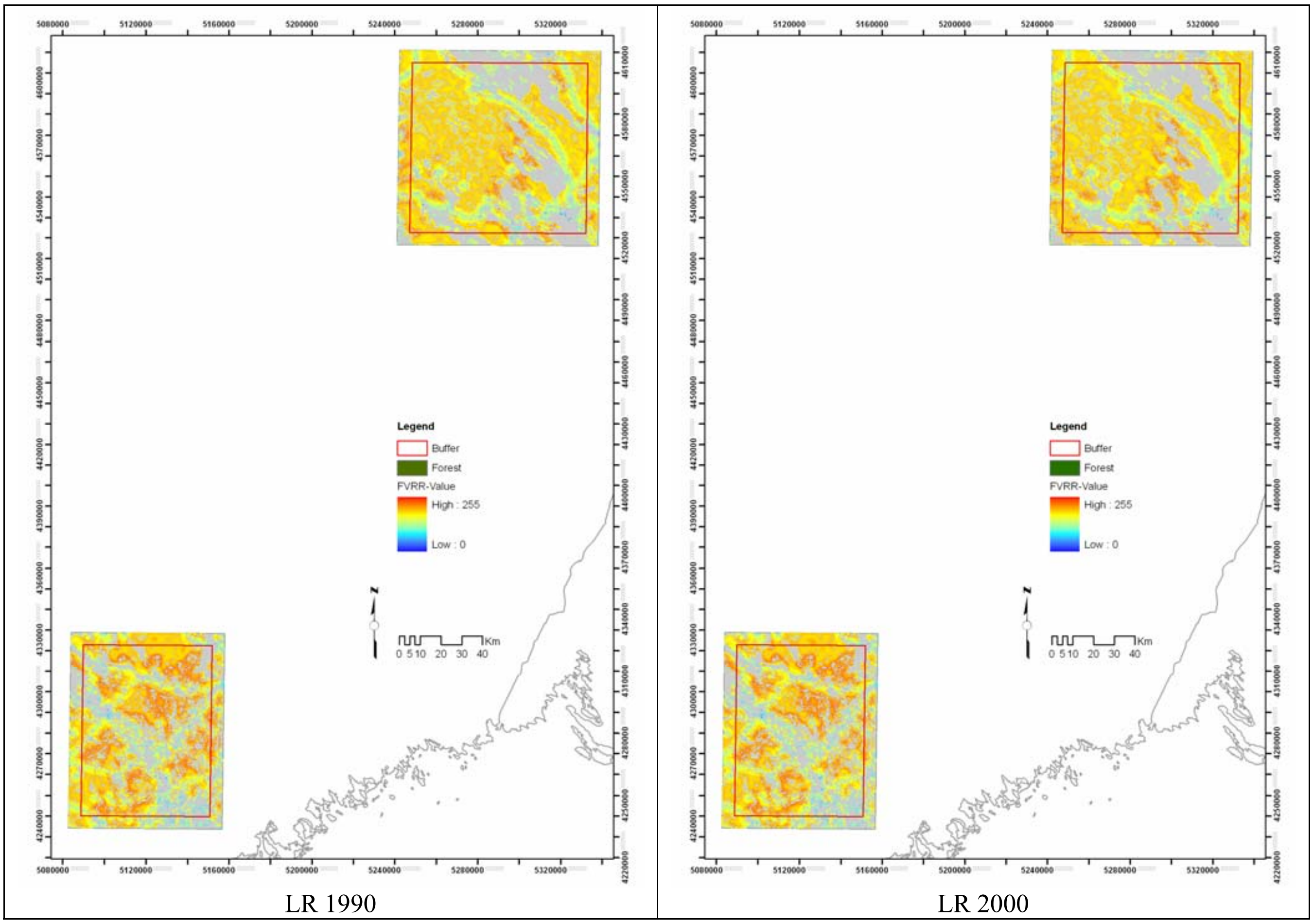


Figure 140: comparison of trends of the combined index of FVRR on low and high resolution data for core and no core forest areas.



LR 1990 LR 2000

Figure 141: the index FVRR summing up all the five umbrella species based on multitemporal low resolution (LR) forest map.

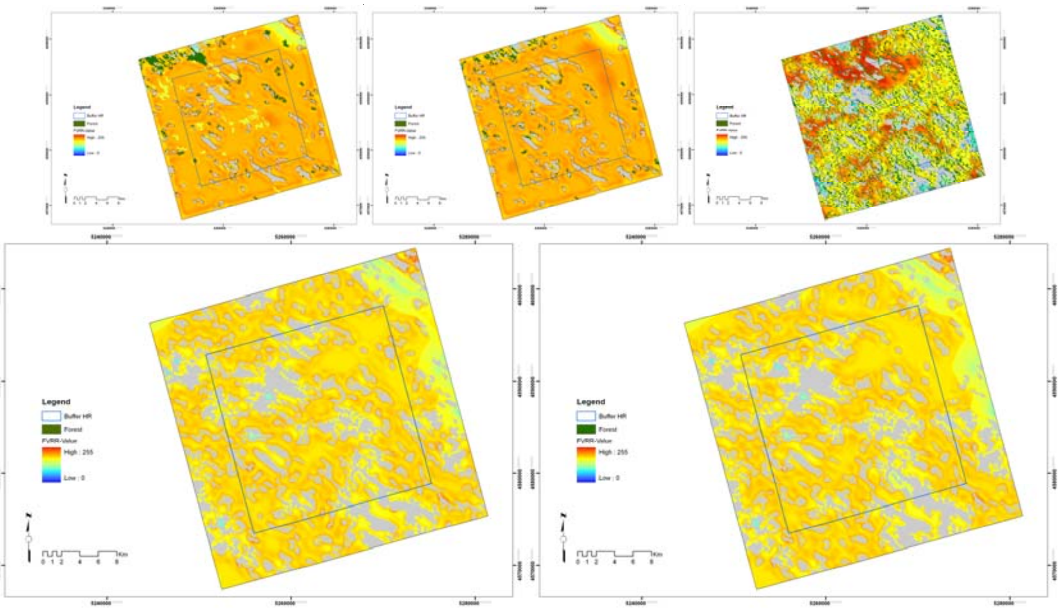


Figure 142: result of the FVRR analysis averaging the values of all the five umbrella species on multitemporal high (above, from left to right: 1950, 1990, 2000) and low (below, from left to right: 1992 and 2000) resolution forest maps from the inner subarea from the north test site in Finland.

9 Comments and conclusions

The development of the project requested the acquisition of a huge amount of spatial data that needed to be harmonised and integrated.

The development of the FVRR modelling approach based both on a general species unspecific and on five different species specific models was mainly based on a wide bibliographic review carried out during the first part of the project.

Once that all the input raw dataset were prepared and that the models were fully developed an intensive raster GIS based processing phase bring to the production of the different FVRR maps: multitemporal, multiscale, based on different ecoprofiles (five umbrella species + one general model) and replicated in seven test areas in five biogeographical areas in Europe.

Through the application of the MSPA software (GUIDOS), it was possible to calculate the different trends of the FVRR indexes in core and no core forest areas demonstrating meaningful differences in average values and in temporal trends.

The full analysis and discussion of the results produced is intended to be completed also after the official end of the project in order to finalise one or more scientific papers.

The innovative approach developed seems to be able to monitor the resistance, resilience and vulnerability of forest areas on the basis of their proximity to the potential ecological *optimum*.

The general model is easy to be applied and much of the information produced in the species specific models is redundant. At least the models developed for birds, marten and roe deer have a consistent level of autocorrelation. While the models developed for butterflies and for the wolf are more specific and less redundant.

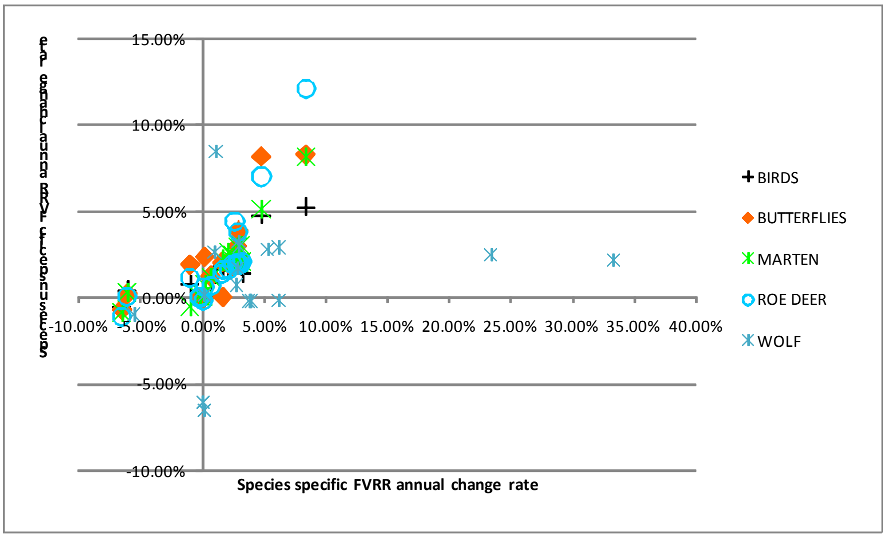


Figure 143: relationship between yearly change rates of species specific and species unspecific FVRR indexes for the five umbrella species (data from all the test sites and resolutions).

The correlation between the sum of the species specific models and the general species unspecific model is consistent. For this reason the application of the general model could be replicated on large areas and could be proposed as an integrated index for monitoring forest conditions at landscape level.

The relationship between low and high resolution data is also interesting since the temporal trends between the two adopted resolution are frequently but not always consistent. The models based on high resolution data are anyhow able to track in a more detailed way the temporal trends of forest areas, especially when the two observations are very close (10 years or less).

The information acquired through the application of the MSPA software, at least for differentiating core vs. no core areas, was interesting since all the FVRR indexes frequently demonstrated meaningful differences in average values and in temporal trends.

The relationship between spatial pattern classes (core/no core) and the species specific and unspecific FVRR values change depending upon the considered species and of the local conditions.

Study area	Model	FVRR index					
		Forest		Core forest		No core forest	
		Average	Yearly trend of total	Average	Yearly trend of total	Average	Yearly trend of total
Mediterranean	Birds	147.18	2.00%	170.7	1.20%	131.76	3.24%
	Butterflies	119.2	2.20%	111.8	1.10%	126.96	3.02%
	Marten	116.04	2.30%	135.1	1.40%	101.95	3.45%
	Roe deer	139.33	2.60%	156.2	1.30%	127.68	4.30%
	Wolf	38.41	8.80%	53.4	1.80%	27.53	3.17%
	Aggregated species specific	137.73	2.30%	152.0	0.80%	124.98	2.91%
	Species unspecific	157.23	2.40%	178.36	1.50%	142.38	3.74%
Alpine	Birds	140.75	-0.44%	141.58	-0.85%	135.10	0.26%
	Butterflies	140.75	0.08%	141.58	-0.35%	135.10	0.42%
	Marten	157.32	-0.16%	159.47	-0.51%	124.09	0.62%
	Roe deer	157.96	-0.21%	162.85	-0.70%	150.21	0.69%
	Wolf	120.31	0.25%	130.85	-0.40%	104.10	1.78%
	Aggregated species specific	161.01	-0.11%	165.75	-0.67%	152.53	0.84%
	Species unspecific	170.06	-0.15%	182.49	-1.94%	136.58	-0.32%
Continental	Birds	117.79	0.01%	124.45	0.01%	114.23	0.01%
	Butterflies	121.37	0.01%	104.10	0.01%	138.49	0.01%
	Marten	145.42	0.01%	159.68	0.01%	129.91	0.01%
	Roe deer	148.25	0.01%	158.53	0.01%	138.91	0.01%
	Wolf	51.46	1.25%	59.55	1.18%	41.51	1.22%
	Aggregated species specific	141.82	0.17%	147.15	0.18%	136.55	0.18%
	Species unspecific	172.58	-0.19%	191.22	-0.18%	156.02	-0.19%
Atlantic	Birds	143.25	-0.61%	194.41	-3.86%	137.65	0.42%
	Butterflies	63.22	-0.37%	120.01	-3.91%	58.79	0.64%
	Marten	126.21	-0.24%	165.50	-3.08%	120.33	0.78%
	Roe deer	118.60	2.69%	185.20	-0.21%	109.92	3.99%
	Wolf	79.29	-0.70%	105.11	-2.07%	74.56	0.34%
	Aggregated species specific	126.18	0.06%	183.17	-2.87%	119.12	1.12%
	Species unspecific	174.01	-0.01%	205.72	-2.43%	171.09	0.75%
Boreal	Birds	136.35	-1.34%	137.75	-2.46%	138.60	4.19%
	Butterflies	102.42	-0.01%	90.65	-2.50%	129.21	6.95%
	Marten	192.84	-1.11%	201.43	-2.23%	174.55	5.30%
	Roe deer	168.05	-1.26%	171.96	-2.08%	165.28	2.71%
	Wolf	161.04	-1.04%	163.15	-2.36%	156.55	5.12%
	Aggregated species specific	180.55	-0.94%	181.63	-2.23%	181.32	4.94%
	Species unspecific	198.91	-1.17%	210.29	-2.14%	177.24	4.29%

Table 5: aggregated results of the study for the different biogeographical areas. Average values and yearly temporal trends of the total of the FVRR index are reported.

This is an important results because it confirms the capacity of this approach to highlight different environmental conditions. With a good sensibility to local phenomena, potentially induced by socio economics or climate change trends in the time.

As expected the values of the FVRR index increased with the increasing of the forest area over the time for both general and species specific models and for both high and low resolutions datasets.

On the basis of the test here presented it is strongly recommended the possible future application of the method on wide areas on the basis of high resolution maps (25 m) but with a simplified version of the species unspecific model.

For an operational application of the proposed method several preparatory activities are needed. Probably the most relevant is the need for a pan-European high resolution (25 m) forest categories raster map based on real forest vegetation (accordingly to FAO forest standard definition) classified accordingly to the European Forest Types system of nomenclature developed by EEA (2006). Such a map should be developed also for the past from historical remotely sensed data, just with a simplified scheme based on forest non-forest basis.

The information acquired from multitemporal forest non-forest historical maps can be combined with recent forest types maps to develop several potentially relevant analysis. For example on forest persistency. Overlaying several multitemporal maps it is possible to develop a forest persistency maps showing, the number of years the forest patch remained spatially unaltered within the local landscape and within the ecological network. The length of the persistency of a forest area may be related to its potential biodiversity and environmental value.

An example of this kind of map is here reported calculated for the Mediterranean test site (Regione Molise) where the longest temporal frame was covered by the multitemporal high resolution maps: 1936, 1954, 1992, 2005.

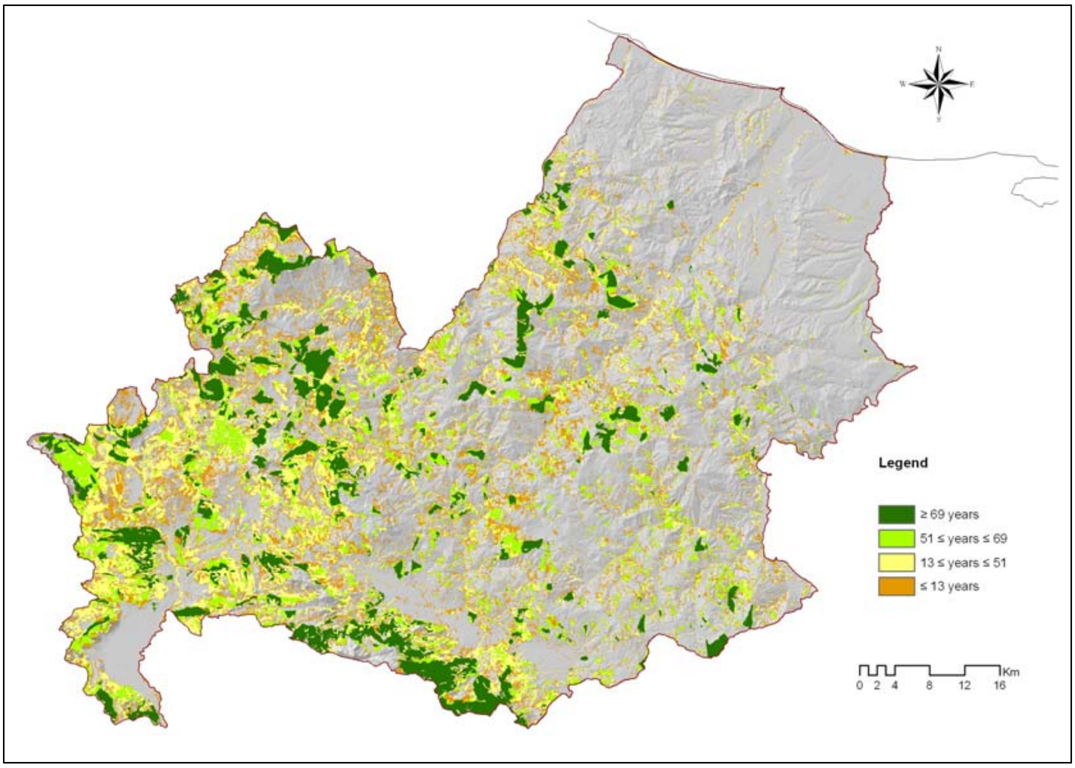


Figure 144: forest persistency map at the year 2005 of the Mediterranean test site (Regione Molise, Italy). Spatial resolution of 25 m.

On the basis of the MSPA analysis carried out on the 2005 forest map the most frequent spatial pattern class in the oldest class of the forest persistency map (≥ 69 years, long persistent, potentially oldest forest) was *core* forest. The youngest class (≤ 13 years, less persistent, new forest area recently created) is instead mostly related to the *bridge* spatial class.

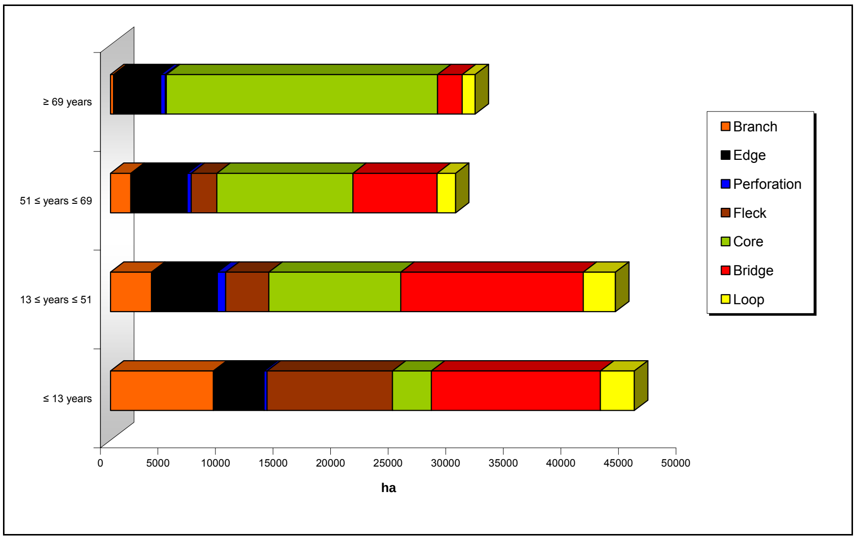


Figure 145: relationship between forest persistency classes and spatial pattern classes from the GUIDOS system. Year 2005, Mediterranean test site (Regione Molise, Italy), spatial resolution of 25 m.

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Abstract

The project “Linking and harmonizing the forests spatial pattern analyses at European, national and regional scales for a better characterization of the forests vulnerability and resilience” (JRC contract 382391 F1SC) covers one of the seven topics to be studied in the frame of the Regulation (EC) 2152/2003 on the monitoring of forest and environmental interactions, the so-called "Forest Focus" Regulation. It first demonstrates the application of the mathematical morphology based forest spatial pattern analysis tool developed at the Joint Research Centre (GUIDOS); it then develops one index related to potential forest vulnerability, resistance and resilience (FVRR index) on the basis of multi-criteria fuzzy modelling technics. Its final aim is to address linkages between forest spatial pattern and forest ecological functionality with emphasis on forest vulnerability, resistance, resilience.

This study was conducted by a European consortium coordinated by the University of Molise (Italy) and included partners from the University of Hamburg (Germany), the European Forest Institute (Finland) and the Forest Research (United Kingdom). The overall supervision of the project and the processing of forest spatial pattern were done by the Joint Research Centre.

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