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Abstract	D6.4 provides a quantitative assessment of historical changes of focal habitat and a quantitative analysis of landscape structure at the regional level in IT3 (PART 1) and anticipates the methodological outlines of the undertaken comparative habitat and landscape modelling exercise across BIO_SOS sites (PART2). These are required to support validation of the different stages outputs, of the EODHaM system.
Keywords	Quantitative landscape pattern analysis, habitat modelling, landscape modelling.

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1. Executive summary

D6.4 is a Deliverable of WP6 Task 6.2 (activity 1). It expands on the results of the pre-evaluation based on landscape pattern analysis (LPA) and on the ranked set sampling described in D6.3, and outlines the conceptual and operational framework for undertaking habitat and landscape modelling comparative exercises across BIO_SOS sites.

For the purposes of WP6 Task 6.2, activity 1, D6.4 therefore:

- PART1 is intended to report on the:

1. Assessment of focal habitat historical changes at the "regional" level for IT3;
2. Testing at the IT3 "regional" level of the framework already developed (D6.3; Mairota et al., 2012) for the comprehensive quantitative assessment of variations in human impact on the landscape, based on LPA.

This analysis is aimed at an assessment of the effectiveness of Natura 2000 for this site and at an evaluation of the current level of fragmentation of the focal habitat across this site.

- PART 2 is intended to report on:

3. State of the art on habitat and landscape modelling with reference to the use of RS data in connection to field data
4. Methodologies and protocols set up for a comparative habitat and landscape modelling exercise across BIO_SOS sites;
5. Progress to date for the application of such protocols in different BIO_SOS sites.

Habitat and landscape modelling are aimed at investigating the relations between:

a) Field data on community structure obtained from BIO_SOS field campaigns for system validation (WP4) and RS data (i.e. object features in the semantic net terminology) (habitat modelling);
b) Indices derived from LPA to both RS information and field data on community structure (landscape modelling).

The activities carried out for the purposes of D6.4 are intended to contribute to the validation of the different stages outputs of the EODHaM system which is being developed in BIO_SOS.

2. PART 1

2.1 Introduction

An exercise was carried out which can be considered as part of the assessment of the effectiveness of Natura 2000, which has been identified as one of the conservation science priorities in Europe (Pullin et al., 2009). In addition surveillance and reporting on the conservation status of habitats “of community interest” is also a legal obligation for EU Member States under articles 11 and 17 of the “Habitats” Directive.

This exercise is comprised of two parts, the first (section 2.2) is intended to assess the degree of change over two decades at the scale of the whole site; the second (section 2.3), at the same scale investigates the current level of fragmentation of the focal habitat across the site.

Section 2.2 describes the analysis carried out with the aim of assessing the trend in habitat loss within and outside the IT3 BIO_SOS site with respect to the considerable time lag occurred between the formal adoption of the “Habitats” Directive” and the different stages of transposition of the EU legislation at the national and regional levels. The implementation of Natura 2000 started in Italy with a delay of over three years compared to the provisions of the “Habitats” Directive. As a result, a first legal protection of Natura 2000 areas, introducing and obligation for appropriate assessment of land use change projects, came into force towards the end of 1997. More detailed conservation measures, including the prohibition of conversion of grassland to other land use changes, were available only towards the end of 2007.

Section 2.3 describes the analysis carried out with the aim of extending to the regional scale the quantitative landscape pattern analysis carried out (D6.3) on a sample area of the site and its surroundings coinciding with the core of the frame of the EO VHR image used (WP5) to produce the base LCCS map (100 km^2 approx.).
This, besides offering an enlarged perspective on the state of the focal habitat in this site, provides an opportunity to test on different locations the methodology based on the combination of three approaches to landscape pattern analysis experimented over a single transect lain on the sample area. Such area is also representative of the kind of interfaces of the Natura 2000 IT3 site which include two different types of interface (or buffer zone), one in which official bindings of the N2K apply but the actual legal provisions for the national park (NP) are not in force, the other with neither formal nor legal protection (Figure 2.1).

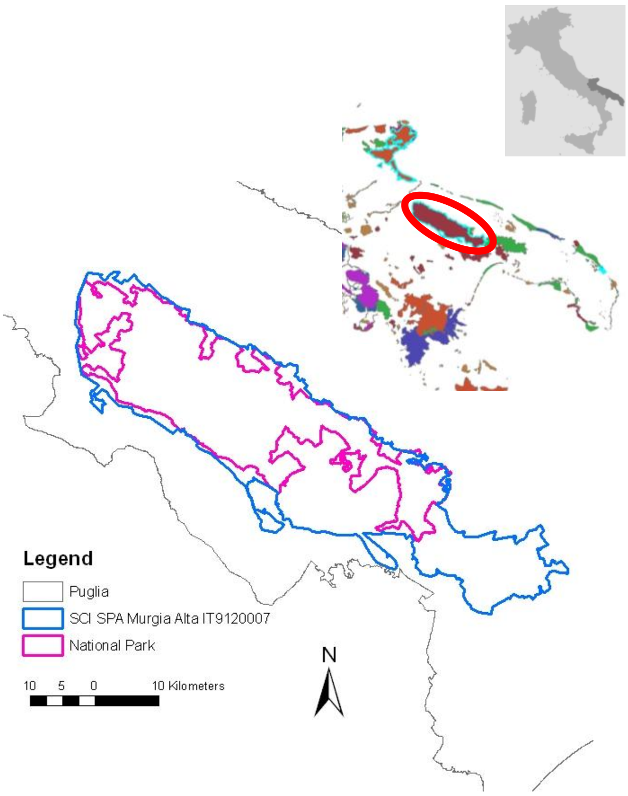


Figure 2.1 - IT3 BIO_SOS site ("Murgia Alta" IT9120007) location map.

2.2 Assessment focal habitat historical changes at the "regional" level for IT3

2.2.1 Study area and cartographic data

This analysis was carried out over an area of 2,755.22 ha defined by a fixed distance buffer (5 km buffer) within the administrative border of the Apulia Region, surrounding the IT3 BIO_SOS "Murgia Alta" site. The study area (or map extent) for the analysis was defined by means of a fixed buffer width based on the NP administrative boundary as this appeared the most appropriate method in a karst environment where surface watershed are not recognisable. A fixed buffer width of 5 km seemed appropriate to encompass different morphological terrains outside the NP boundary which might explain differences in LU/LC and management. The study area also encompasses (partly or totally) a number of other Natura 2000 sites, as well as the entire Alta Murgia National Park established within part of the IT3 in 2004, covering approximately half of the "Murgia Alta" SCI-SPA surface (Figure 2.2; Table 2.1).

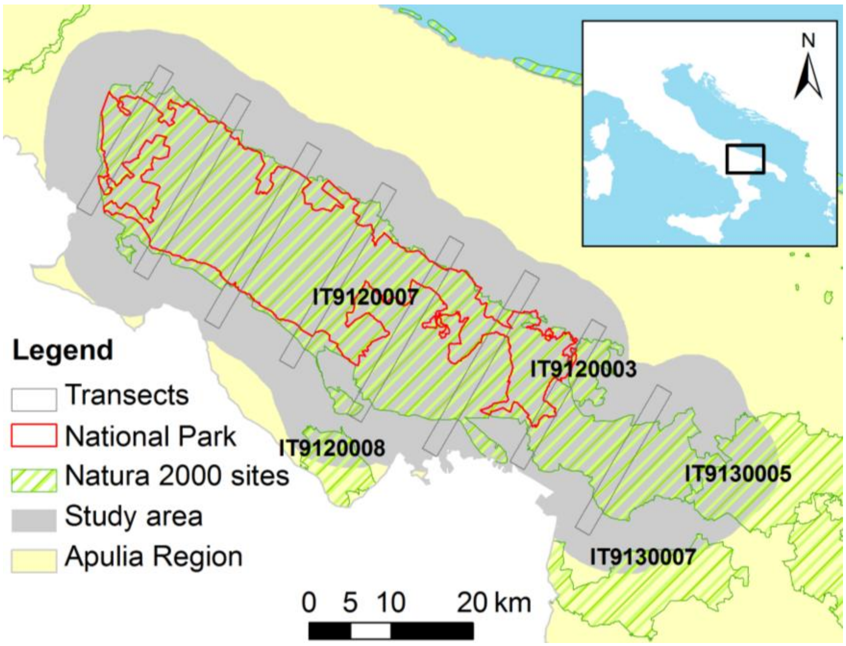


Figure 2.2 – Location map of the study area. The location of the eight transect used for the analysis in section 4 are also indicated.

Table 2.1 - List of Natura 2000 sites included in the study area and their main features.

Code	Name	Type	Total surface (ha)	Surface within study area (ha)
IT9120007	Murgia Alta	SCI, SPA	125.882	125.882
IT9130005	Murgia di Sud-Est	SCI	47.601	8.796
IT9120003	Bosco di Mesola	SCI	3.029	3.029
IT9120008	Bosco Difesa Grande	SCI	5.268	2.511
IT9130007	Area delle Gravine	SCI, SPA	26.741	2.014
			208.521	142.232

Given the limited availability of extensive and periodical land cover/land use (LC/LU) and habitat maps, the analysis of land cover/land use dynamic (in space and time), including also the calculation of focal class (habitat) area and other indicators of landscape structure and composition, was based on the available series of CORINE Land Cover maps (3rd level) related to years 1990, 1999 and 2006. For this set of maps the size of the minimum mapped unit is 25 ha, and the minimum length of polygons is 100 m (EC, 1993).
The LC/LU in the study area is predominantly agricultural, while natural grasslands and forests are present to a lesser extent (Table 2.2). Along with smaller towns and urban settlements, two relatively large towns are present (*i.e.* Gravina in Puglia and Altamura, population ~45,000 and 70,000 inhabitants respectively).

Table 2.2 Main non urban land use/land cover types in the study area (based on the Corine Land Cover 2006 map).

Land cover type	*Surface (%)*
Non-irrigated arable land	46.67
Olive groves	14.55
Natural grasslands	13.44
Complex cultivation patterns	8.29
Annual crops associated with permanent crops	5.12
Broad-leaved forest	2.84
Coniferous forest	2.56
Vineyards	2.09

Grasslands are the most representative ecosystem type in the study area. They currently cover ~37,000 ha and represent what remains of the over ~80,000 ha existing at the beginning of the 20^{th} century, due to transitions to other land uses (agriculture/urban). As for most European grasslands (EEA, 2010), these grasslands can be defined as 'semi-natural' because they have most likely developed through a mix of anthropogenic and natural processes, including long periods of grazing by domestic stock, cutting and deliberate burning regimes, and are currently modified and maintained by human activities, mainly through grazing and burning (Turbé et al., 2010). They are among the most species-rich plant communities in Europe. Most grasslands in the study area correspond to two habitat types listed in Annex I to the 92/43/EEC Directive: 62A0 "Eastern sub-Mediterranean dry grasslands (*Scorzoneratalia villosae*)" and 6220* "Pseudo-steppe with grasses and annuals of the *Thero-Brachypodietea"*.
A more detailed description of this site and of the Annex 1 habitat occurring in it is provided in both Deliverables 2.2 and 6.3.

2.2.2 Methodology

The analysis was carried out adapting to the specific case the "fragmentation geometries" procedure described in EEA and FOEN, 2011, which is useful for quantifying and visualizing fragmentation. "Fragmentation geometries" are defined as particular combinations of fragmenting elements in the landscape, both linear and areal, either natural or anthropogenic. The procedure is intended to be applicable at any scale, provided that a set of scale and problem specific fragmentation geometries are defined (EEA and FOEN, 2011). For the purposes of this analysis grasslands were considered as the focal LC/LU class, all other land cover classes were assigned to the most appropriate fragmentation geometry as suggested in EEA and FOEN, 2011.
In particular, the following fragmentation geometries (FG) were identified:

- FG_A1-A2, including all types of urban land cover and transport infrastructure;
- FG_B2a, including olive groves, fruit plantations and vineyards;
- FG_B2b, including all types of arable land;
- FG_B2c, corresponding to man-made conifer stands;
- FG_C, including natural woodland, shrubland and reeds;
- FG_D, corresponding to water bodies.

With regard to FG_A1-A2, as a consequence of the spatial resolution of CLC maps, the transport infrastructure was not detectable (width < 100 m). However, given the key role played by transport infrastructure in determining landscape fragmentation, an inclusion of this land use type in the analysis was considered necessary. The authors'

knowledge of the study area suggested that the extent and location of transport network had not changed significantly over the 1990-2006 period. Therefore, the CLC class 122 “Road and rail networks and associated land” was extracted from a 2006-2007 high-resolution LC/LU map and merged with the CLC maps. A similar procedure is described in EEA and FOEN, 2011.
In order to visualise and analyse the fragmentation pattern in space and time, the focal class patches were classified based on the patch size, in ranges defined by a geometric progression with a common ratio of 2, ranging from 25 ha (the CLC minimum mapped unit) to over 800 ha. In addition, the landscape pattern metrics “percentage of class area” (PLAND) and “effective mesh size” (MESH) were computed. The latter metric is reputed as capable of capturing the fragmentation aspects of landscape pattern (Jaeger 2000; EEA and FOEN, 2011).

2.2.3 Results

Fragmentation geometries and remaining focal class size for IT3 Murgia Alta are drawn in Figure 2.3.
Arable lands represent the main fragmentation geometry. These have greatly increased between 1990 and 1999. Focal habitat patches with a surface of over 800 ha have correspondingly decreased and smaller patches (particularly those or 400-800 ha) have increased. However, in 2006 patches over 800 ha still constitute the main range of focal habitat patch size, to 53.89% of the grassland area and to 14.60% of the landscape area in the IT3 Murgia Alta. Land use change during the 1999-2006 cannot be appreciated at the scale of CLC map (Figures 2.3 and 2.4).

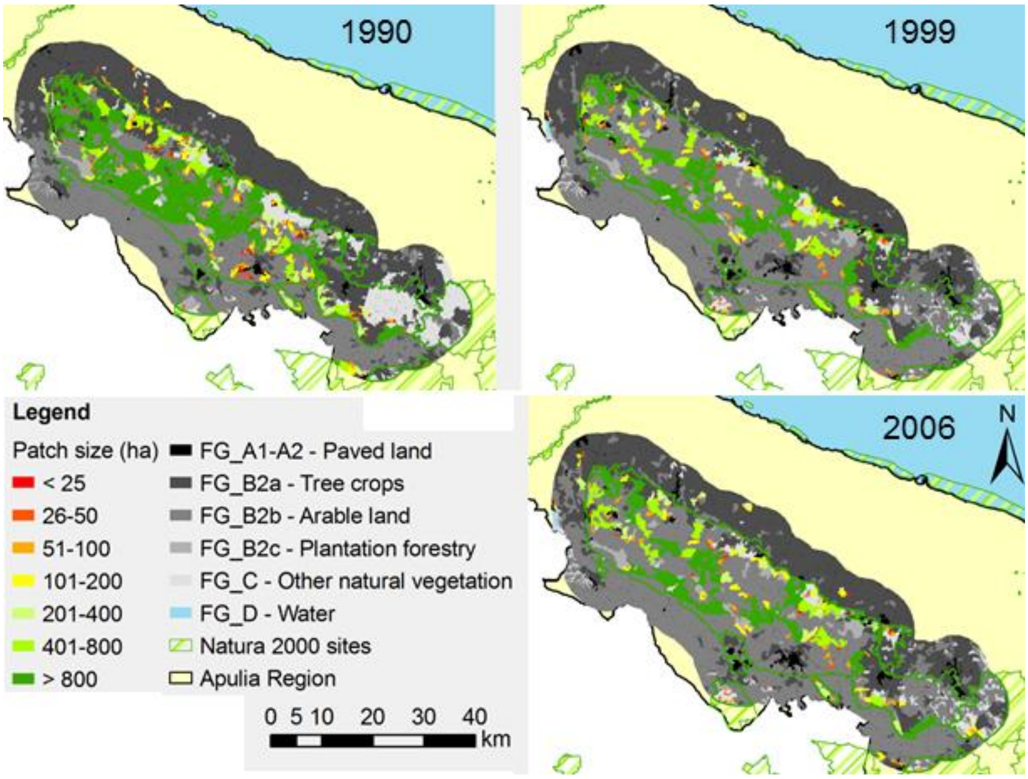


Figure 2.3 – Land use changes in terms of fragmentation geometries and focal class patch size range from 1990 to 2006 (computation based on CORINE Land Cover).

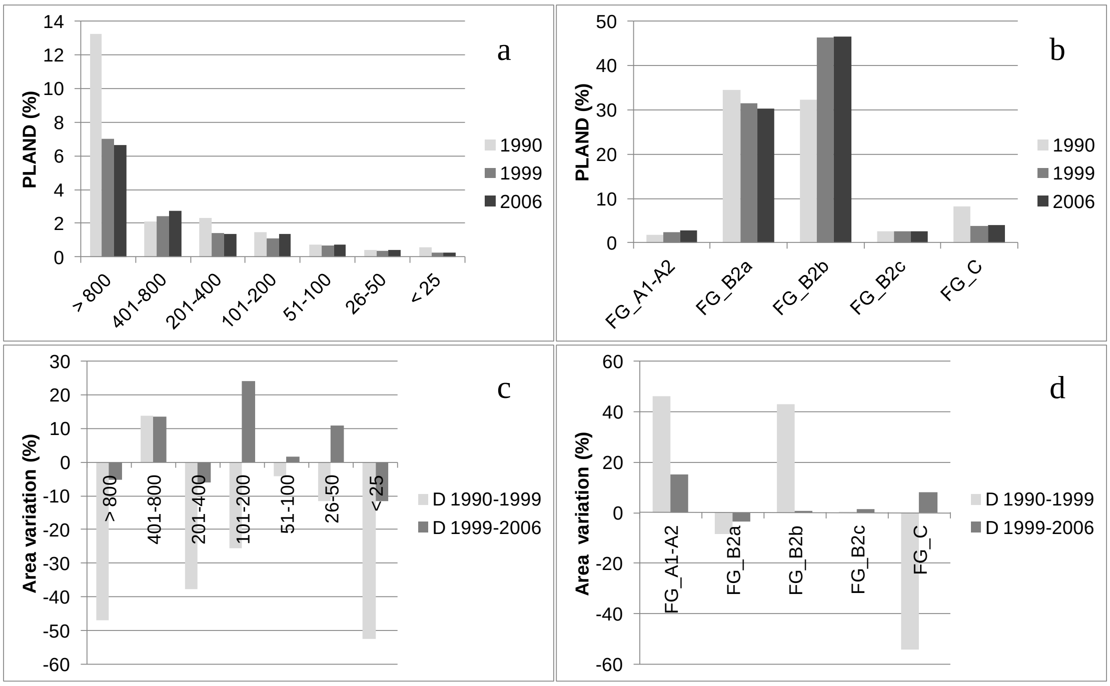


Figure 2.4 – Land use changes in terms of percentage focal class area in patch size intervals (a, absolute values and b, percentage variation) and in terms of percentage class area for fragmentation geometries (c, absolute values and d, percentage variation) from 1990 to 2006 (computation based on CORINE Land Cover).

Land use change has been much more widespread within Natura 2000 than outside Natura 2000, where the surface of focal habitat and other non-crop land covers (FG_C_other natural vegetation) was negligible during the whole study period. Focal class loss within the N2K is mainly due to conversion to arable land, while change to other artificial or natural land uses is negligible (Figure 2.4).

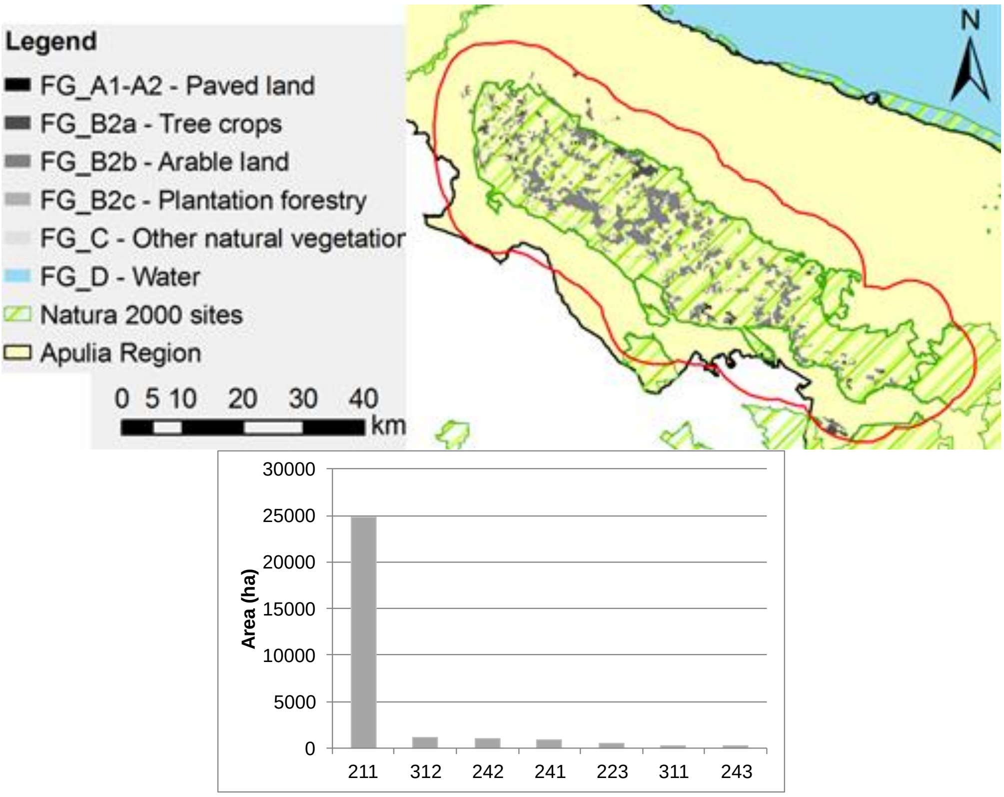


Figure 4 - Transition from grassland to other LC/LU types during the 1990-2006 period.

The trend in the MESH index also confirms that a high degree of fragmentation is associated to the first time interval in the whole area, within the set of N2K sites and within the IT3 Murgia Alta site (Figure 2.5).

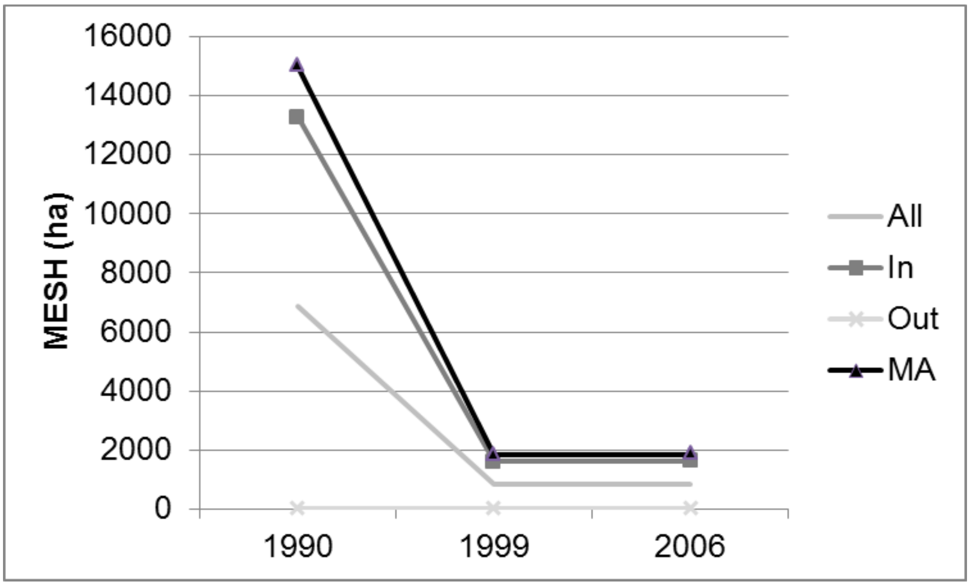


Figure 2.5 – Trend of the Effective Mesh Size Index (ha) form 1990 to 2006 (All = whole study area, Out= outside the set of N2K, In = within the set of N2K sites, MA, within the IT3 Murgia Alta site).

2.2.4 Discussion

The fragmentation geometries identified for the specific case study and observation scale and the selected LPI seem capable of providing a quantitative description and characterisation the substantial loss of grassland area evident form the visual inspection of the two CLC maps of 1990 and 2000 (Figure 2.6).

A decrease of larger grassland patches and of the effective mesh size, indicating an increase in habitat fragmentation, have occurred between 1990 and 1999, following a conversion to arable land use. These changes occurred mostly within Natura 2000 sites and were mainly caused by EU Common Agricultural Policy incentives promoting durum wheat production.

This massive land use change could not be prevented by the EEC/92/43 Directive, possibly due to the delay in its implementation at the national and regional levels. However, from the available maps it is not possible to separate the changes occurred before May 1994, the deadline established for the implementation by the Member States of the “Habitat” Directive, from the changes occurred after this deadline.

The lower order of magnitude of the changes of the fragmentation geometries and of both PLAND and MESH between 1999 and 2006, which appear as minimal at the scale of the CLC maps, might be due do the basic protection of N2K sites introduced at the national level in 1997, to significantly slowing down habitat loss and fragmentation. Therefore it is clear that such finer scale changes could be only investigated by means of intermediate and less coarse maps than CLC ones. The available set of CLC maps, neither allows an assessment of the effectiveness of the more detailed conservation measures introduced by national and regional legislation in 2007 as well as of those introduced by the National Park Authority.

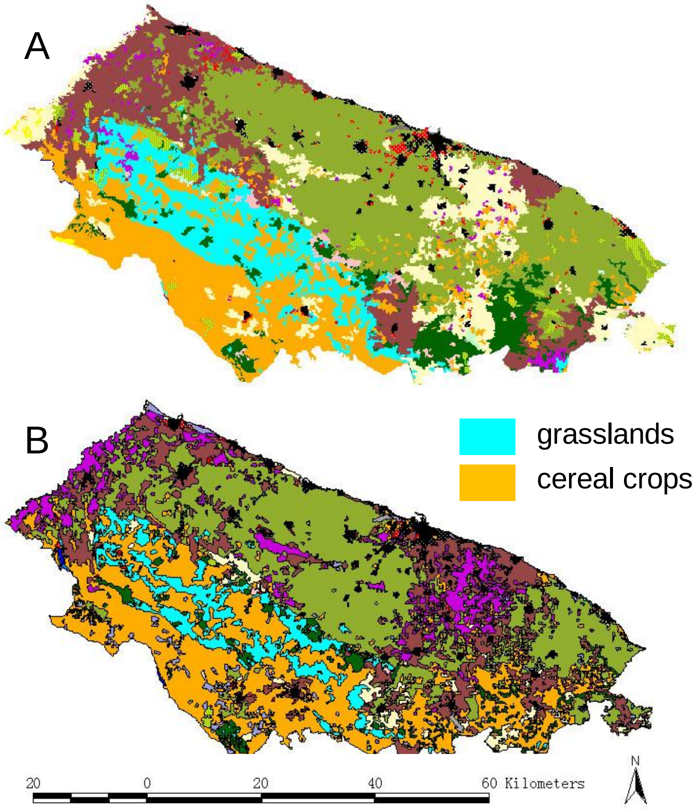


Figure 2.6 – Corine Land Cover changes (3rd level) from 1990 (A) to 2000 (B) in the former Bari province.

2.3 Quantitative landscape pattern analysis at the “regional” level for IT3

2.3.1 Study area, sampling strategy and cartographic data.

The study area is the same as the one described in section 2.2. In order to overcome computation limits a sampling strategy was adopted on eight 2 km x 20-25 km NE-SW transects lain over the study area (Figure 2.1). The analyses were carried out on the same LC/LU map used in D6.3. This is the thematic LC/LU map (2006, 1:5,000 nominal scale), reclassified from the CLC classification taxonomy into the LCCS by means of site specific correspondence rules (D6.1 and D6.10).

2.3.2 Methodology

The three synthetic descriptors defined in D6.3 and in Mairota et al. 2012 were computed for the eight transects. This has implied the computation of the LPIs (i.e. the contrast weighted edge density index, CWED, and the effective mesh size, MESH) for the landscapes defined by the transects at the focal class level, MSPA main structural classes (i.e. non-core boundary, connector and islet) capable to quantify fragmentation in this landscape) at pixel level, as well as the landscape mosaic characterization, at pixel level (D6.3 and Mairota et al. 2012).

The synthetic descriptors are:

1) The diversity profiling of the landscape mosaic based on Hill’s numbers (Hill, 1973, 1997). This is useful to synthetically describe landscape composition and assess its the variability of heterogeneity degree across the different transects.
2) The Similarity Index (SI) proposed by Estreguil and Caudullo (2010). This is defined as the ratio between the share of a given morphological spatial pattern analysis (MSPA) class under a certain landscape mosaic type and the total amount of MSPA class. The SI combines information on both landscape mosaic composition and pixel configuration and thus appears very useful in characterising the of the focal habitat in the transect with regard to the context of each non-core MSPA main structural class.
3) The location of discontinuities along each the transects in terms of the mean values of both landscape pattern indices (LPIs) and MSPA non-core structural classes informing on focal habitat fragmentation, computed by means of moving split-window (MSW) performed using the square Euclidean distance (SED).

2.3.3 Results

The percentage of landscape corresponding to the focal class averages 16.06 ± 6.63 across the 8 transects. The contrast weighted edge density index (CWED) reaches its highest value in transect 1 and the lowest value in transect 8., however a gradient along the longitudinal dimension cannot be identified as a significant degree of variability is observed among transects (17.99 ± 5.57). MESH (15.77 ± 12.31) reaches the highest value (38.82 ha) in transect 1, and the lowest values in transect 7 (3.54). (Figure 2.7).

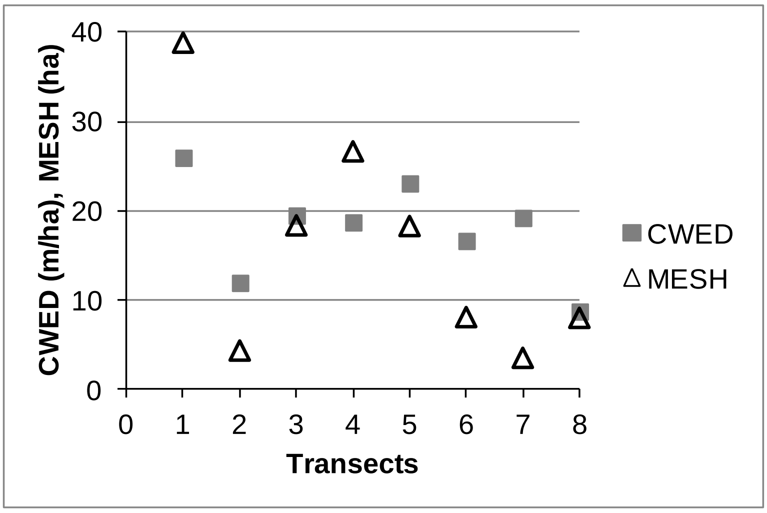


Figure 2.7 – Contrast-weighted edge density (CWED, m/ha) and effective mesh size (MESH) measured in the 8 transect.

The maps resulting from the computation of CWED and MESH with the moving window analysis (square window, 100 m side, 8-cell patch neighbourhood rule) show the variation of these metrics along the 8 transects (Figure 2.8).

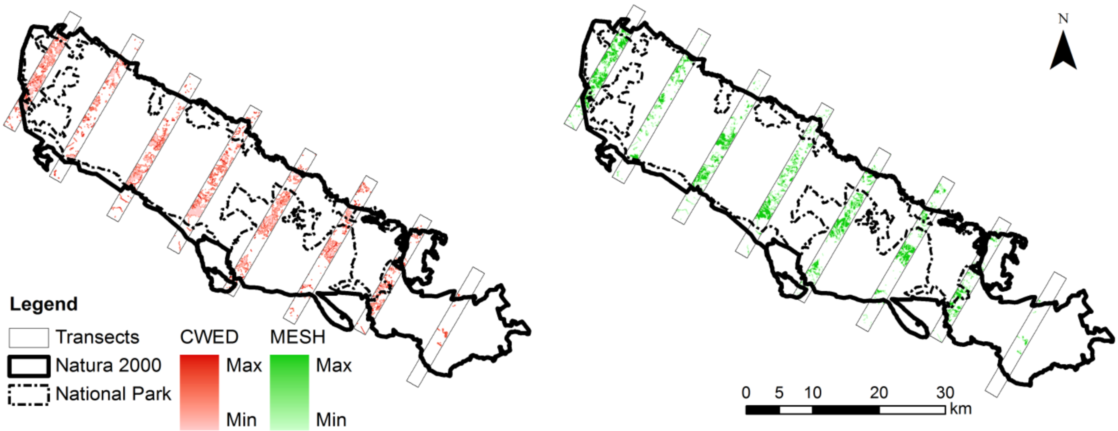


Figure 2.8 – Visual representation of the spatial variation of contrast-weighted edge density (CWED) and effective mesh size (MESH) calculated with a moving window across the 8 transects.

The distribution of the habitat area within the MSPA main structural classes: core, islet, boundary (including both edge and perforation), connector (loop, bridge and branch) is shown in Table 2.3. This grouping shows that, in transects 1 and 8, 51% of habitat area can be considered as “core”, while this proportion is lower for the remaining transects. The map resulting from the classification of the habitat area into MSPA classes indicates that most of the core habitat is found within the N2K boundary (Figure 2.9).

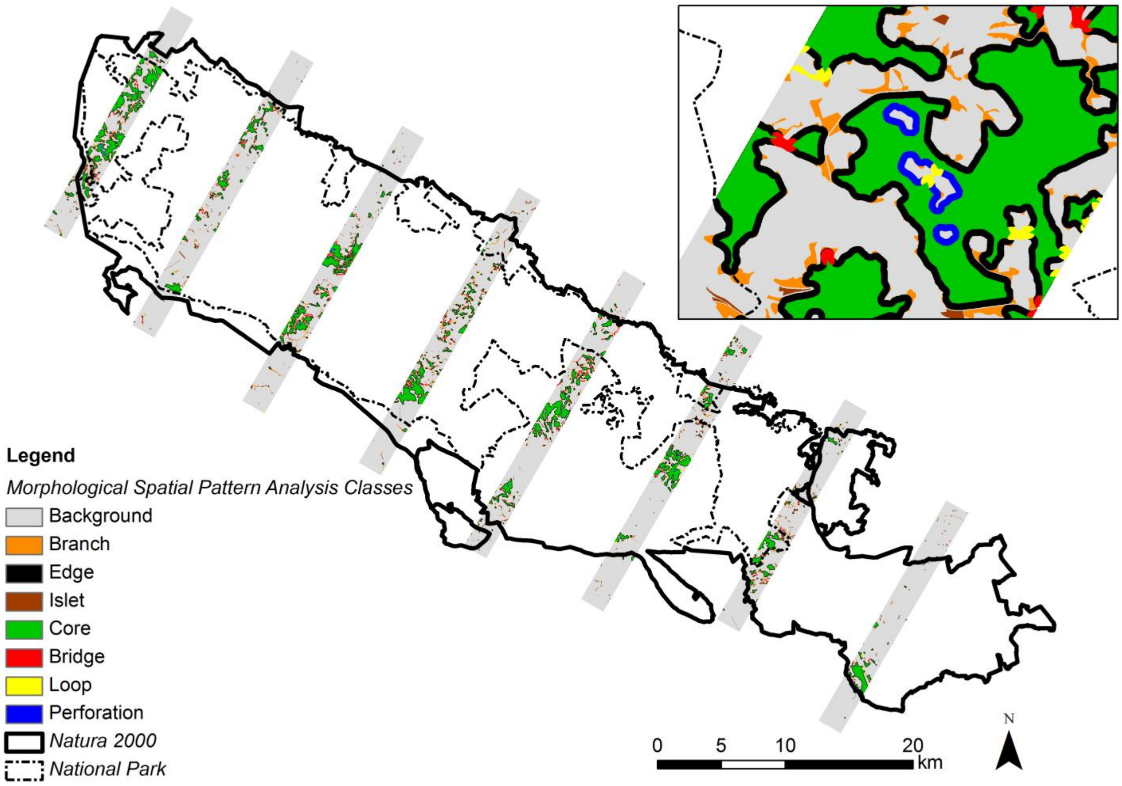


Figure 2.9 – Morphological spatial pattern analysis classes of the focal habitat within the 8 transects.

Table 2.3. Proportion of habitat corresponding to each main MSPA feature class and absolute data.

MSPA class	Transects							
	1	2	3	4	5	6	7	8
Boundary	0.33	0.36	0.33	0.34	0.35	0.33	0.32	0.27
Connector	0.14	0.18	0.18	0.20	0.16	0.14	0.22	0.10
Core	0.51	0.37	0.45	0.40	0.43	0.47	0.35	0.51
Islet	0.03	0.08	0.05	0.06	0.07	0.07	0.11	0.12

The landscape mosaic map shows the spatial variation of the landscape mosaic indicator across each transect (Figure 2.10). In the landscape mosaic indicator, capital letters indicate that the corresponding land use type (urban, agricultural or natural) covers over 60% of the land surface, with double capital letter indicating a 100% cover. Small caps letters indicate an area cover between 30% and 60%, while the letter is not reported if the corresponding land use type is absent.

Predominantly agricultural mosaic types (i.e. those with “agricultural” LC/LU ≥ 60%, corresponding to AA, A, Au, An, Aun) are the dominant mosaic types, in particular outside Natura 2000, while predominantly natural mosaic types (i.e. NN, N, Nu, Na, Nua) are found mostly within Natura 2000. Predominantly urban (i.e. UU, U, Ua, Un, Uan) land use types are instead scarcer within the transects.

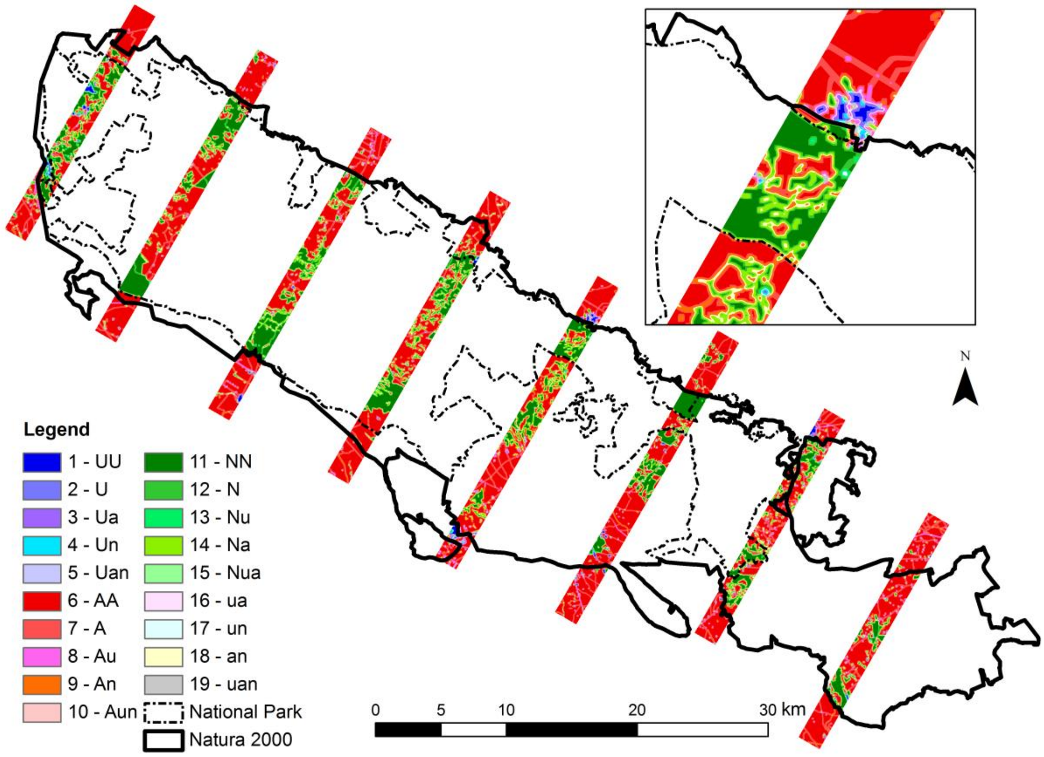


Figure 2.10 – Landscape mosaic types in the 8 transects.

AA is the predominant landscape mosaic type in all transects, followed by NN in transects 1, 2, 3, 4, and 6, and by A in transects 5, 7 and 8 (Figure 2.11). The landscape can therefore be defined as mainly agricultural in all transects, in particular towards the south-eastern part of the study area.

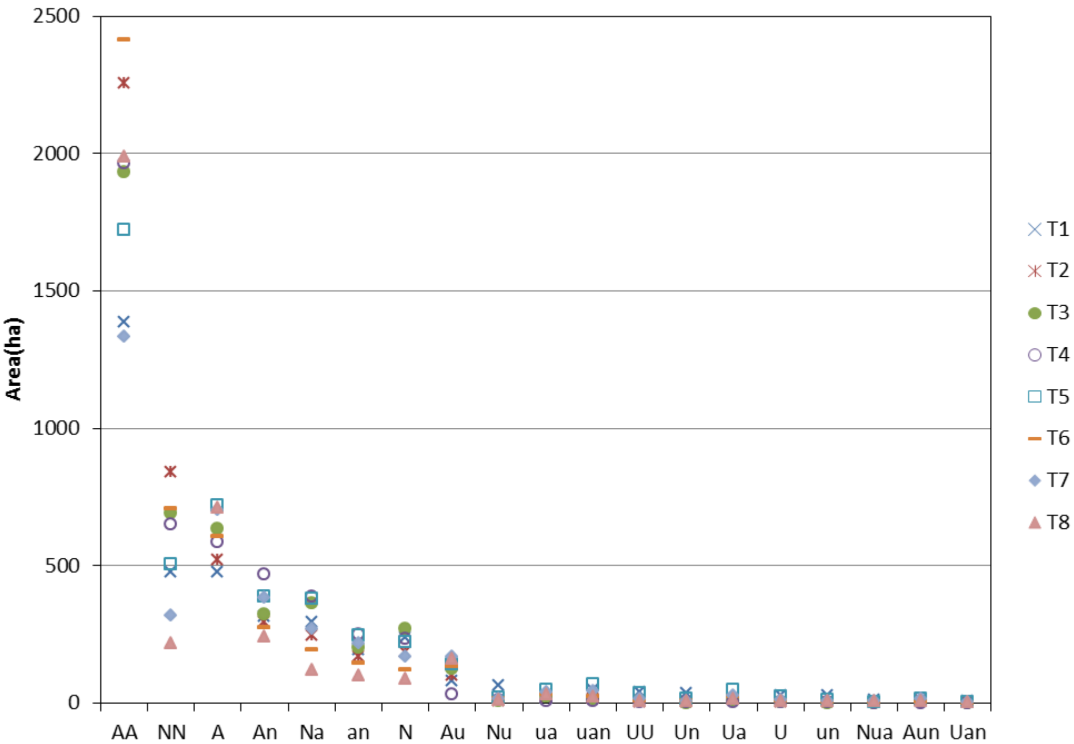


Figure 2.11 – Abundance distribution (area, ha) of the landscape mosaic types in each transect.

The landscape mosaic diversity profiling by means of the Hill's numbers shows that in all the transects the same number LC/LU classes is represented, however some differences appear with regard to their distribution. The transects show a relatively similar diversity profile, although transects 2, 6 and 8 are characterised by comparatively lower values for all the numbers (Figure 2.12).

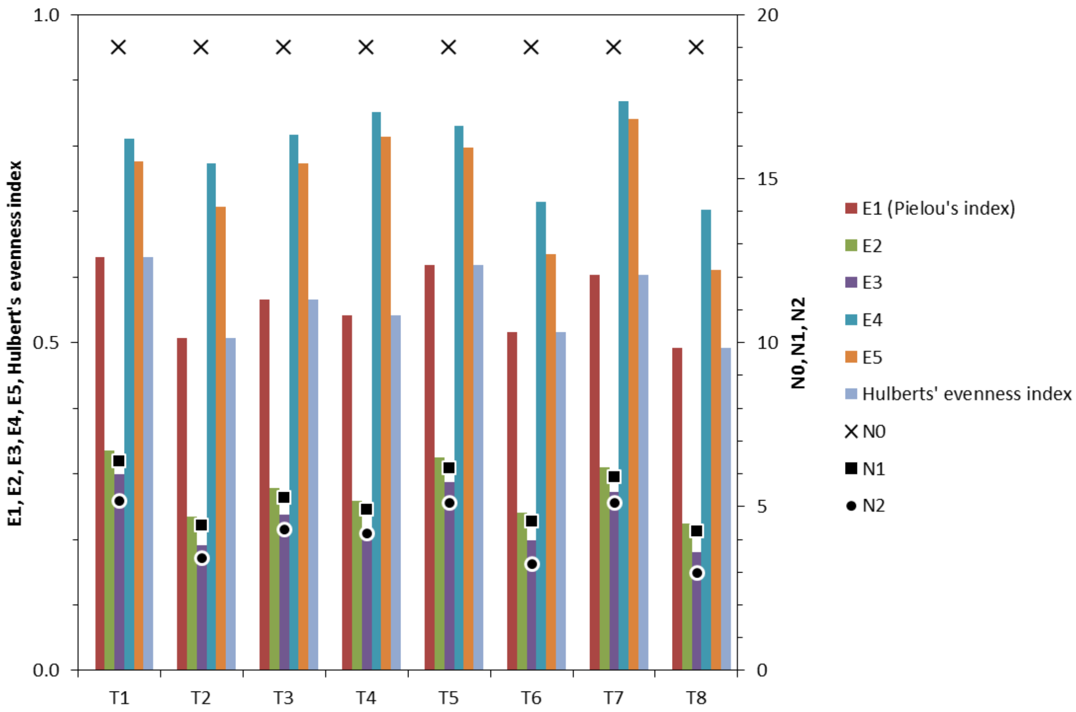


Figure 2.12 – Landscape diversity profiling by means of the Hill's numbers in the 8 transects.

Using the landscape mosaic map in combination with the MSPA output maps relevant to the main structural classes and each MSPA non-core main structural classes pixel, a landscape mosaic type is assigned, allowing for the computation of the Similarity Index (SI) for each MSPA class (Estreguil and Caudullo 2010).

SI quantifies the proportion of edges (boundary and connector) and isolated fragments in either anthropogenic or natural context. For each MSPA class, a synthetic quantitative description of landscape structure (composition and configuration) is obtained by plotting the SI of each transect against the landscape mosaic type. This provides information on the kind of potential pressures affecting each non corestructure class associated with fragmentation in across the study area. This can also be used for rapid change detection and monitoring at the landscape level (Figure 13 A (boundary), B (connector), C (islet).

Diversity profile and SI represent both quantitative and qualitative descriptors of landscape structure, and their variations for the same landscape at different time thresholds can indicate changes and specify their direction. Analogously, they allow for the comparison of different landscapes within the same region. Thus, they provide a baseline to make comparisons across space and/or time.

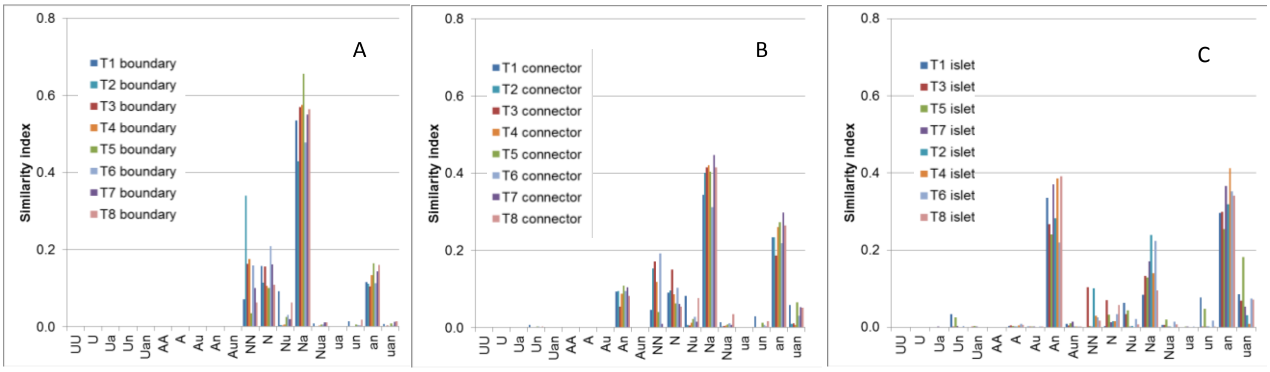


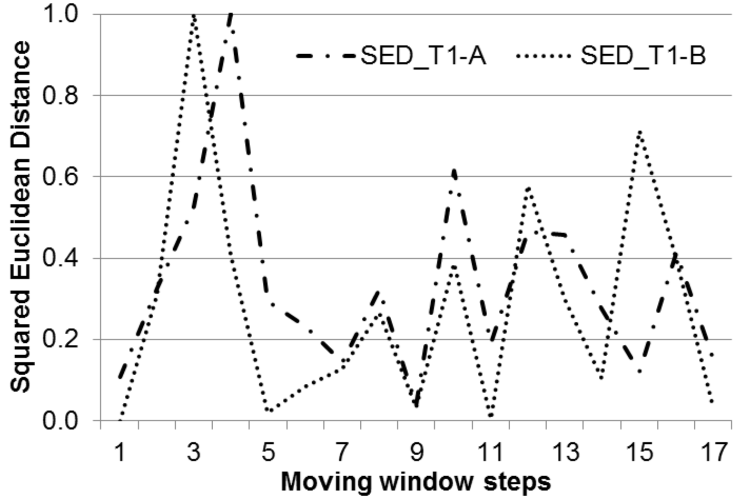
Figure 2.13 - Similarity Index, obtained combining landscape mosaic types and the boundary (A), connector (B) and islet (C) structural types derived from morphological spatial pattern analysis, for each transect.

The moving split window analysis was performed using a selection of metrics derived from the LPA, based on the results of the PCA (D6.3), CWED and MESH, and the MSPA non-core main structural classes (islet, boundary connector).

This set of metrics, for the specific landscape and for the observation scale of the study, appeared adequate to summarise the spatially explicit attributes of the focal class associated with its fragmentation.

None of these metrics when used separately for the same analysis would provide such unequivocal information.
The MSW analysis indicated the occurrence of sharp discontinuities (maxima) coinciding with the boundaries of the National Park/Natura 2000 site, or with obvious changes in landscape pattern, e.g. transition from grasslands-dominated landscapes to cropland-dominated landscapes, abrupt artificial discontinuities created by roads etc. (Figures 14, 15, 16, 17; Tables 4, 5, 6, 7). In particular, most frequent and sharp discontinuities coincide with transitions from grassland-dominated landscapes to cropland-dominated landscapes, and from less fragmented grasslands to landscapes with higher values or absence of grasslands. Also the shift between grasslands and highly contrasting LU classes (e.g. roads, tree crops) corresponds to high discontinuities.

Figure 2.14 - Graphical representation of the moving split window analysis on the two longitudinal halves (A and B) of transect 1 (above) and transect 2 (below). Peaks indicate the occurrence of discontinuities. SED = Squared Euclidean Distance.



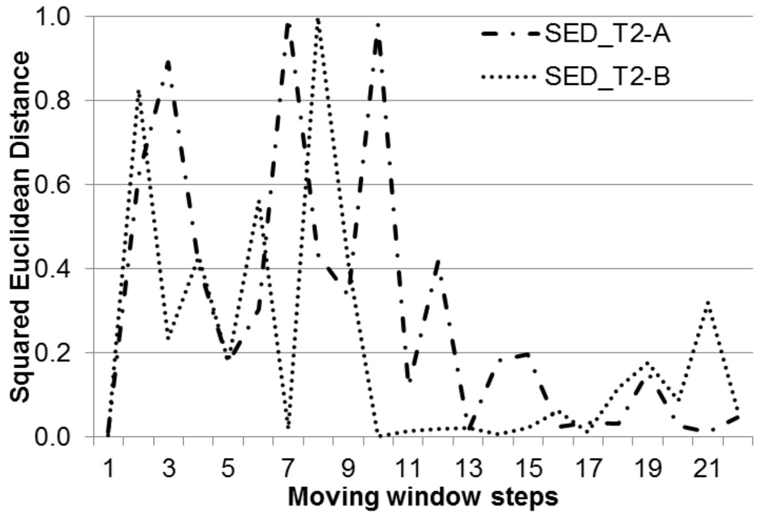


Table 2.4 – Squared Euclidean Distance maxima for transect 1 (above) and for transect 2 (below) and their possible explanation. Asterisks (*) indicate global maxima.

Transect	*Section*	*Moving window step*	*Explanation*
1	a	4*	Shift from tree crops to grasslands. Important road. Boundary of the National Park
1	a	8	Transition from less fragmented to more fragmented grasslands.
1	a	10	Shift from arable to grassland
1	a	12	Shift from grassland to urban. Road intersections. National Park boundary.
1	a	16	Shift from grassland to arable. Important road.
1	b	3*	Shift from tree crops to grasslands. Important road. National Park and Natura 2000 boundary.
1	b	8	Transition from less fragmented to more fragmented grasslands.
1	b	10	Transition from more fragmented (but with lower contrast) to less fragmented (but with higher contrast) grasslands
1	b	15	Transition from less fragmented to more fragmented grasslands. National Park and Natura 2000 boundary.

Transect	*Section*	*Moving window step*	*Explanation*
2	a	3	National Park and Natura 2000 boundary. Shift from tree crops to natural vegetation.
2	a	7*	Shift from grassland-dominated to arable-dominated landscape.
2	a	10	Shift from arable to grassland
2	a	12	No explanation.
2	a	15	No explanation.
2	a	19	No explanation.
2	b	2	Transition from less fragmented to more fragmented grasslands.
2	b	4	Transition from more fragmented (but with lower contrast) to less fragmented (but with higher contrast) grasslands
2	b	6	Transition from less fragmented to more fragmented grasslands. National Park and Natura 2000 boundary.
2	b	8*	Transition from landscape with several grassland patches to landscape with few grassland patches.
2	b	16	Roads.
2	b	19	Roads and forest edge.
2	b	21	National Park and Natura 2000 boundary.

Figure 2.15 - Graphical representation of the moving split window analysis on the two longitudinal halves (A and B) of transect 3 (above) and transect 4 (below). Peaks indicate the occurrence of discontinuities. SED = Squared Euclidean Distance.

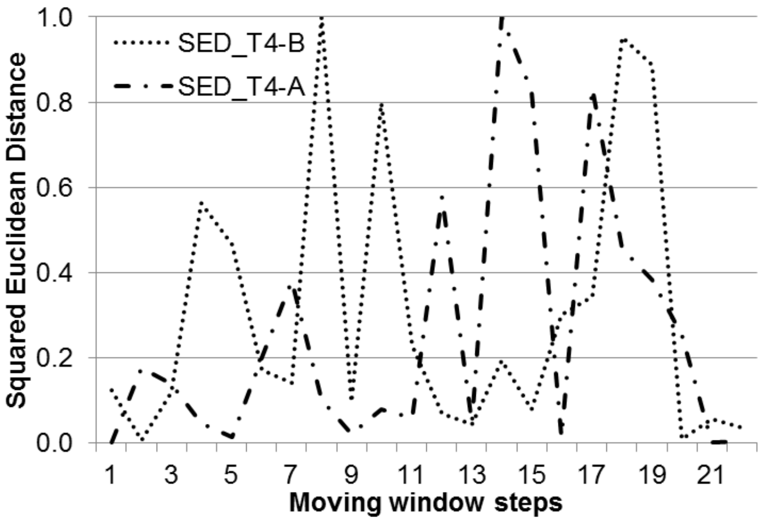
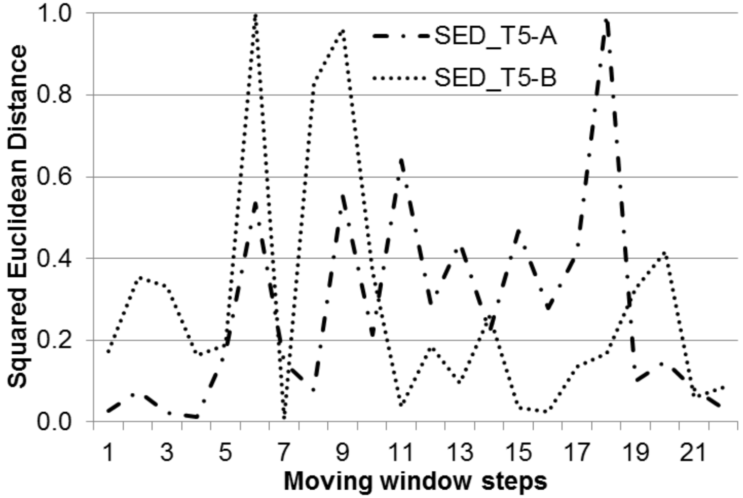


Table 2.5 – Squared Euclidean Distance maxima for transect 3 (above) and for transect 4 (below) and their possible explanation. Asterisks (*) indicate global maxima.

Transect	Section	*Moving window step*	Explanation
3	a	6	Grassland from absent to present.
3	a	9	Grasslands from more fragmented to less fragmented
3	a	13	From more fragmented and contrasted to less fragmented and contrasted grasslands.
3	a	15	Transition from arable to grassland
3	a	18*	Transition from grassland-dominated to mixed land use landscape. National Park and Natura 2000 boundary.
3	b	4	Grassland from absent to present.
3	b	6	Increase in contrast due to roads intersecting grassland areas.
3	b	9*	From more fragmented to less fragmented grasslands.
3	b	12	From less fragmented to more fragmented grasslands.
3	b	17	From grassland dominated to more diverse land use.
3	b	20	Important road intersection. Water course.

Transect	*Section*	*Moving window step*	*Explanation*
4	a	2	Natura 2000 boundary.
4	a	7	Road. Increase in grassland abundance.
4	a	10	No explanation.
4	a	12	Decrease in grassland abundance. Road,
4	a	14*	Road. Increase in grassland abundance.
4	a	17	Transition from grassland with high shape complexity to grassland with low shape complexity
4	b	4	National Park boundary. Shift from crops to natural vegetation.
4	b	8*	Transition in LC/LU pattern created by terrain morphology.
4	b	10	Road. Transition in grassland fragmentation.
4	b	14	Increase in grassland abundance.

Figure 2.16 - Graphical representation of the moving split window analysis on the two longitudinal halves (A and B) of transect 5 (above) and transect 6 (below). Peaks indicate the occurrence of discontinuities. SED = Squared Euclidean Distance.



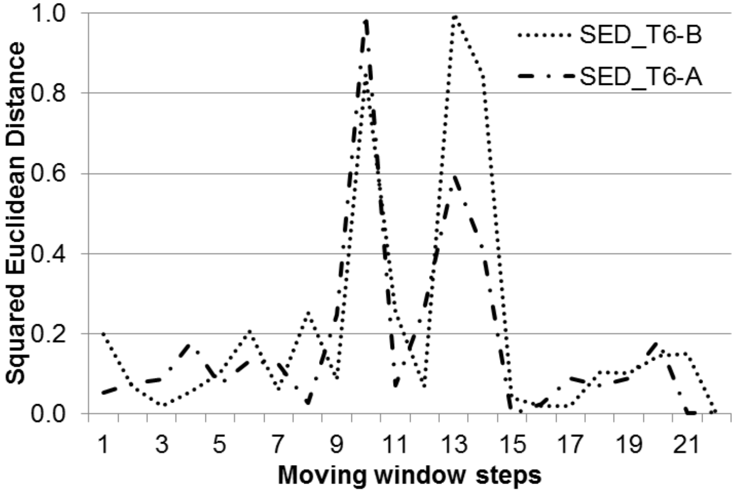
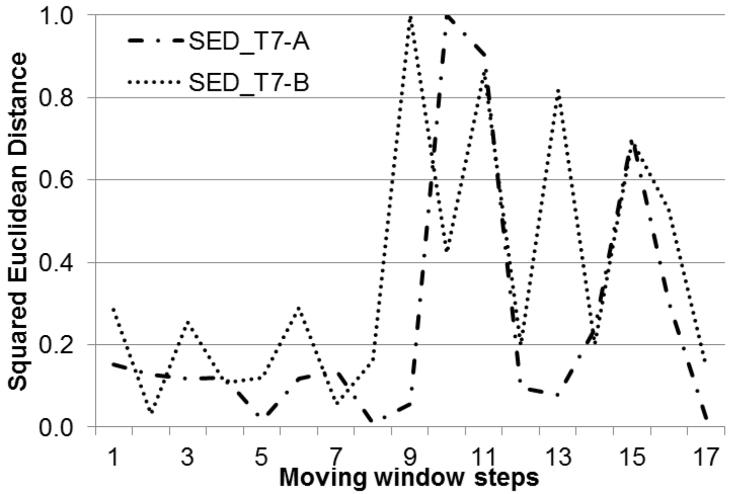


Table 2.6 – Squared Euclidean Distance maxima for transect 5 (above) and for transect 6 (below) and their possible explanation. Asterisks (*) indicate global maxima.

Transect	*Section*	*Moving window step*	*Explanation*
5	a	2	Grasslands from rare to abundant. Roads. National Park and Natura 2000 boundary.
5	a	6	Grasslands from absent to present. National Park boundary.
5	a	9	No explanation.
5	a	11	No explanation.
5	a	13	Roads.
5	a	15	Grassland from present to absent.
5	a	18*	From arable dominated to grassland dominated
5	b	2	Increase in grassland abundance. Urban settlement and roads. National Park and Natura 2000 boundary.
5	b	6*	Increase in contrast. From forest-dominated to grassland-dominated landscape. National Park boundary.
5	b	9	No explanation.
5	b	12	From grassland dominated to arable dominated. Increase in contrast. Road.
5	b	14	Decrease in grassland abundance. Roads.
5	b	20	Decrease in grassland abundance and increase in contrast. Roads.

Transect	*Section*	*Moving window step*	*Explanation*
6	a	4	Natura 2000 boundary. Shift from cropland to natural vegetation.
6	a	6	National Park boundary. Shift from natural vegetation to cropland.
6	a	10*	Shift from arable-dominated to grassland-dominated landscape.
6	a	13	Shift from grassland dominated to arable-dominated landscape. Roads.
6	a	17	Roads and rural-urban fringe.
6	a	20	Natura 2000 boundary. Roads.
6	b	6	National Park boundary. Shift in landscape fragmentation.
6	b	8	National Park boundary. Roads.
6	b	10	Shift from arable-dominated to grassland-dominated landscape. Road.
6	b	13*	Shift from grassland dominated to arable-dominated landscape. Roads.
6	b	21	Roads.

Figure 2.17 - Graphical representation of the moving split window analysis on the two longitudinal halves (A and B) of transect 7 (above) and transect 8 (below). Peaks indicate the occurrence of discontinuities. SED = Squared Euclidean Distance.



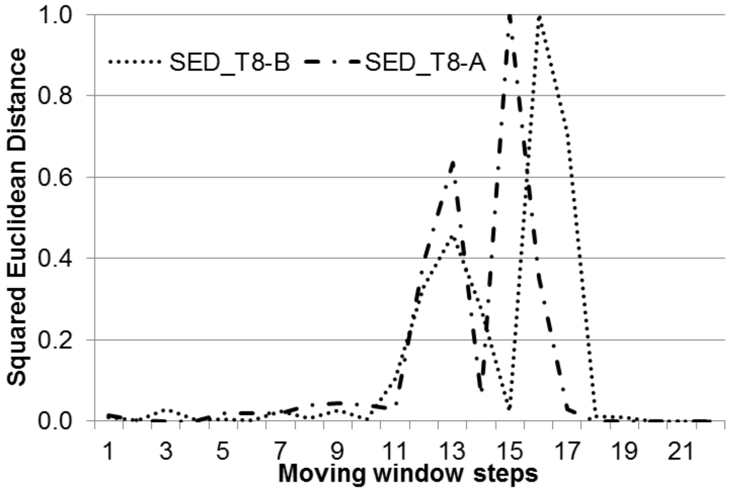


Table 2.7 – Squared Euclidean Distance maxima for transect 7 (above) and for transect 8 (below) and their possible explanation. Asterisks (*) indicate global maxima.

Transect	*Section*	*Moving window step*	*Explanation*
7	a	7	Increase in grassland abundance. National Park and Natura 2000 boundary.
7	a	10*	Increase in grassland fragmentation. Road. National Park boundary.
7	a	15	From mixed to arable-dominated landscape. Roads. Natura 2000 boundary.
7	b	3	From mixed to forest-dominated landscape. Decrease in grassland abundance. Roads.
7	b	6	Increase in grassland abundance.
7	b	9*	Roads.
7	b	11	Decrease in grassland abundance. National Park boundary. Roads.
7	b	13	Increase in grassland abundance.
7	b	15	Decrease in grassland abundance. Road.

Transect	*Section*	*Moving window step*	*Explanation*
8	a	9	Roads.
8	a	13	Shift from arable-dominated to grassland-dominated landscape cropland.
8	a	15*	Shift from grassland dominated to arable-dominated landscape. Roads. Natura 2000 boundary.
8	b	3	Roads. Natura 2000 boundary.
8	b	7	Road.
8	b	9	Roads.
8	b	13	Roads.
8	b	16*	Natura 2000 boundary. Roads. Shift from landscape with grassland to landscape without grassland.

2.3.4 Discussion

With regard to the analyses carried out on the 2006-2007 LU/LC map, these provide a validation of the comprehensive framework proposed by Mairota et al. (2012), including landscape pattern, morphological spatial pattern, landscape mosaic analysis and ecotone detection,

All the computed synthetic descriptors, besides allowing a comparison between different locations across the study area are potentially useful for a rapid evaluation of changes at the landscape level across time. Therefore they can effectively be used in characterising the landscape, monitoring landscape and focal habitat change, and identifying critical areas for habitat conservation and restoration.

In addition, the ecotone detection analysis confirms its potential to detect differences in LC/LU and management outcomes objectively informing on the most appropriate width to define a spatial buffer around a protected area when this cannot be defined adopting physiographic criteria (e.g. watersheds). In the present case discontinuities occurring in proximity of National Park/ Natura 2000 site boundaries are seem to indicate indirectly that these administrative limits have been established to coincide with LC/LC changes in turn related to geo-morphological ones and also following ownership boundaries. However management conditions (i.e. more strict rules and control), as well might have discouraged massive grassland transformation, particularly in the more recent years.

In any case, it is clear that an operative application of this approach aimed at the assessment of the effectiveness of conservation policies, within the framework of a periodic monitoring such as that required by the EU legislation, can only be undertaken if detailed and frequently updated cartographic data are available, such as those provided by means of the EODHaM system.

3. PART 2

3.1 Introduction

For the scope of the present study, habitat is defined as “an element of the land surface that can be consistently defined spatially in the field in order to define the principal environments in which organisms can live” (Bunce et al., 2008). This does not mean dismissing or diminishing other definitions of habitat that are theoretical and species centred (e.g., Hall et al. 1997). Instead, it refers to an operational definition of habitat with respect to habitat mapping. In this sense, a habitat can be seen as the tangible part of an ecosystem and thus can be considered a proxy for the landscape-level element of biodiversity (i.e., the ecosystem, Franklin and Dickson, 2001). As such, habitats represent geographical reference primitives for all ecological evaluations that proceed by direct mapping of individuals and associations or by indirect assessment of potential species distributions based on habitat requirements. For these reasons, habitats today are considered as important indicators of biodiversity, both in their own right and in connection to their relevance to plant and animal species (Bunce et al. 2012). In addition, habitat has become the reference conservation entity at the European and wider international levels (Bunce et al. 2012). Prevention of habitat loss, fragmentation and degradation is therefore crucial for biodiversity conservation.

Habitat-relevant aspects of biodiversity are those related to their spatial configuration within the landscape and to the internal properties (quality) of each patch of each habitat type. These are all highly context dependent, as is evident from the ample stream of ecological research at the landscape level, with this often aimed at investigating the implications (from ecological, planning and management perspectives) of complex ecological systems that hierarchical in nature. For some time, this has been recognised by theoretical, empirical and applied studies (e.g., Ehrenfeld and Toth, 1997; Race and Fonseca, 1996; Bedford, 1996; Bell et al., 1997; Wiens, 1999; Fischer et al 2008; Didham 2010, and cited literature). Context dependence of processes occurring at the habitat level is moreover central to the conceptualisation and modelling of movement-based functional connectivity (e.g., Schippers et al. 1996; Adriaensen *et al.* 2003; McRae *et al.* 2003; Ray, 2005). Finally, this is also endorsed by the first three focal areas of the Convention on Biological Diversity (CBD; i.e., “Status and trends of the components of biological diversity”, “Threats to biodiversity” and “Ecosystem integrity, and ecosystem goods and services”), with these bearing a strong relation to landscape structure which, according to landscape ecology, is considered to be a component of biodiversity (Jedicke 2001).

Habitat spatial configuration within the landscape is assessed by means of quantitative landscape pattern analysis (D6.2, D6.3, Mairota et al. 2012), whereas internal properties are investigated by means of habitat and landscape modelling. Both habitat and landscape modelling depend on good quality information on environmental attributes describing habitat quality which is, *per se,* not directly measurable, as well as on good quality field data.

As Earth observation (EO) techniques can be of great help in obtaining good quality information on environmental attributes (Nagendra et al. 2012), an exercise has been undertaken within the scope of the BIO_SOS project (with respect to existing studies) to explore the potential of remote sensing data to make inferences on the relations between habitat attributes and species- or functional groups-related indicators across spatial scales and biogeographical regions as well as to investigate the potential of

RS data in combination with field data to provide empirical case studies to the landscape modelling approach proposed by Didham et al. (2011).

3.2 State of the art on habitat and landscape modelling

3.2.1 Habitat modelling

By means of habitat modelling, habitat quality, assessed through measurable habitat (environmental) attributes, can be used as an estimator for certain variables representing (surrogating) biodiversity (e.g., species presence, abundance, richness, diversity, density, carrying capacity, probability of occupying a location) (see Brendan 2005 and Marcot 2006 for a review). The long theoretical and technical tradition of habitat modelling derives from the need for reliable information on biodiversity in conservation planning (Brendan 2005; Burgman and Lindenmayer 1998; Ferrier et al. 2002; Wilson et al. 2005). Habitat modelling, and particularly environmental niche modelling, is also increasingly being used to obtain information on habitat suitability with regard to both threatened and invasive species ecology and management, both in terrestrial and marine habitats (e.g. D6.7; Peterson et al. 2003; Harney, 2008,).

A few approaches have been explored in past and recent literature in order to use remote sensing methodologies and imagery to obtain extensive indirect measures of habitat estimators (Townsed et al, 2009; Rocchini et al, 2010; Feingersh et al. 2007). Two main classes of approaches can be identified: some authors start from land cover maps as obtained by applying conventional supervised classification methodologies to high-resolution satellite images; other authors use low level imagery statistics directly extracted from the images, in order to reduce the general cost of analysis by a more immediate approach. Features generally considered for the analysis in the second class are both context-based and spectral-based. All features can be detected at different spatial (focal) scales but the upscaling and downscaling approaches are still an open research issue (Wessman and Bateson 2005, He et al. 2002). Several statistical techniques based on ecological theory, used for calculating species diversity at different spatial scales, may therefore be easily translated into spectral diversity measurements. Given the difficulties associated with field-based data collection, the use of remote sensing for estimating environmental heterogeneity and hence species diversity is generally seen as a powerful tool, since it is considered as capable of providing a synoptic view of large areas with a high temporal resolution. Spectral variation hypothesis states that the greater the habitat heterogeneity, the greater the species diversity within (Palmer et al., 2002).

Little work has been explicitly undertaken to review the use of spectral heterogeneity as a proxy of species diversity. Rocchini et al. (2004) found, however, in a wetland ecosystem in Italy, that there is a scale-dependent relationship between spectral heterogeneity of a 3 m resolution multispectral Quickbird image and the number of plant species, with an increase in explanatory power as the spatial scale of analysis increased. Image spectral heterogeneity explained about 50% of the species richness variation when using vegetation plots of 1 hectare, but this explanatory power decreased to just 20% when related to vegetation data from plots of 100 m^2, indicating a strong potential for use of this approach. Also, the impact of spatial grain must be carefully considered in relation to landscape heterogeneity, which will naturally be context specific and vary from landscape to landscape.

A first aim of this review is to provide an overview of the use of spectral heterogeneity for estimating species diversity and to examine the advantages and pitfalls of this approach. In recent years some research has been conducted to use object oriented techniques for classification of different types of satellite images, by coupling context-

based information with traditional pixel-based analyses of pure spectral information. These approaches appear to be challenging when applied to very high-resolution imagery (Blaschke, 2010).

Note that all approaches in literature analysed so far are not based on a conventional classification stage as a pre-processing step to be conducted before evaluating ecological indicators. That is due to the perceived cost of conducting such a step by means of conventional supervised techniques. All of them instead extract information and extrapolate values starting from the images, not maps, as explained above.

For instance, the availability of satellite data, such as those gathered by the Landsat program, makes it feasible to study all parts of the globe with a resolution of up to 30m (Pettorelli et al., 2005). Tucker et al. (2004) present a complete description of the Global Land Cover Facility, which freely hosts this kind of data. Note that most of the results have been obtained in the past years at medium or high resolutions, while very high resolutions — as available in current commercial satellites — still need to be properly exploited for ecological studies (Nagendra and Rocchini, 2008; Rocchini et al., 2010; Nagendra et al., 2012). When medium spatial resolution imagery (corresponding to grain sizes of tens of meters) are used for ecological studies, a single pixel often encompasses a number of individual trees or plants, sometimes even crossing habitat boundaries (Small, 2004; Nagendra et al., 2010). Thus each pixel corresponds to a mixed averaged signature. Therefore the way the information aggregates signatures, and how a downscaling can be performed, are something that still need to be properly understood. For instance Stickler and Southworth, 2008 found that an ecological prediction model for a forested ecosystem in Uganda based on Landsat ETM+data outperformed the model based on QuickBird data.

Decreasing performance with increasing spatial resolution of remotely sensed data may be a limitation of the pixel-based approaches which have been criticized in several papers. Instead, object-oriented approaches have been proposed as tools for generating patches in an objective manner at a given specified scale, by maintaining the topological and hierarchical structure of the landscape over multiple scales (Hay et al., 2001; Burnett and Blaschke, 2003; Marignani et al., 2008; Blaschke, 2010). For example, Karl and Maurer (2010) found higher correlations between IKONOS imagery and field-measured cover with object-based methods than with pixel-based methods. The difference evened out with medium spatial resolution imagery such as Landsat. This suggests, therefore, that object-oriented approaches may in some cases be superior to pixel-based approaches, but also that the noise in the reflectance detected by a hyper-spatial sensor may create problems in discriminating objects and suitable scales to use for every objects.

As a rule of thumb, the higher the spectral resolution (higher number of bands), the greater the power to discriminate objects which reflect in a different manner (i.e., the power to detect the heterogeneity of an area; Gillespie et al., 2008). This could justify the use of hyperspectral sensors, but their general low availability renders them difficult to be used extensively. It seems feasible to consider adopting different criteria for analysis at high-resolutions, where spectral distance among features can appear low due to sensor limitations and geometrical properties, and where the third dimension of ground objects assumes major relevance (e.g. due to shadows and occlusions).

Note that those approaches are anyway challenging in natural contexts, and have been mainly exploited in urban areas. When pixel dimensions shrink below the size of the object studied, to a point where the pixels are smaller than the size of individual tree crowns, the variability in the signatures of pixels covering the same object, for instance an individual tree, suddenly increases. Therefore, in some cases high spatial resolution could actually confound the issue by increasing the level of intra-class variation and introducing spatial heterogeneity resulting from in-shadow pixels. This

kind of problem is only partially circumvented by object or context based approaches and could require advanced classification approaches and ontologies to deal with complex landscapes at very high resolutions.

Statistical approaches based on textural features are analysed in a few cases at different scales as predictors of ecological indicators, see for instance St.Louis et al 2006, St.Louis et al 2009, Woods et al 2012. In those works classic texture descriptors set which use second order image statistics (Grey Level Co-occurrence Matrixes) introduced originally by Haralick et al 1973, are used to characterize variation in grey level as predictors of vegetation richness and leaverage height respectively, and they are correlated at various scales with field data by using multiple size sliding windows. Texture features can be analysed potentially on any available band as well as any other derived measure, such as vegetation indexes. A similar approach is currently under our analysis and better explained in section 3.4.

When considering the use of remote sensing data, consideration needs also to be given to the ability to differentiate species, genera and/or habitats from remote sensing data and also to infer composition as well as associated fauna. From Earth observation data, information on the spatial distribution of individual species or genera can be obtained in some instances whereas in complex environments, communities may be differentiated. For example, Lucas et al. (2008) were able to distinguish species or genera of trees in Australian savannahs using reflectance spectra extracted from tree crowns delineated using airborne hyperspectral data whilst in Wales, Lucas et al., (2011) was able to differentiate habitats dominated by single species, namely bracken (*Pteridium aquilinum*), bilberry (*Vaccinium myrtilus*) and heather (*Calluna vulgaris*). These studies further highlight that the ability to discriminate individual plant species depends on the spatial resolution of the observing sensor in relation to the area covered by the plant or the community but also by the extent to which a plant dominates the community. This can be referred to as the ‘optimal’ spatial resolution and when this is too high or too low, the ability to discriminate species or genera is reduced. Where the spatial resolution is too high, and as indicated earlier, components of the plants (e.g., branches, shadow, the underlying ground surface) are often observed which complicates the process of discrimination. Where it is too low, spectral mixing becomes problematic because of variable and often poorly defined contributions from other surfaces. Hence, procedures that attempt to retrieve sub-pixel proportions (e.g., spectral mixture analysis, fuzzy-based classifications) are often used. These can, however, be successful in quantifying the composition of mixed-species communities if the spectral characteristics of contained species are sufficiently distinct. This approach differs from that of assessing spectral heterogeneity.

Whilst many studies tend to focus on the use of single-date imagery, multi-date imagery can provide information that can be used to increase discrimination between species. This is particularly the case in environments where distinct transitions occur. Examples are the forests of the northern hemisphere temperate regions, where a mix of deciduous and evergreen species occur, primarily because of seasonal variations in temperature and radiation and savannah woodlands in southern Africa where deciduous, semi-deciduous and evergreen species occur, primarily because of seasonal variations in rainfall and temperature. In tropical environments and, to a certain extent, boreal environments, changes in the reflectance characteristics of individual species and communities are comparatively small and hence the use of single-date imagery is often justified.

In terms of assessing plant species diversity, the ability to discriminate individual species either at the plant or landscape level is often overlooked. Furthermore, plant communities that are often obscured from the view of the observing sensor can be inferred from classifications of species and genera forming the upper canopy. As an

example, Lucas et al., (2002) highlighted that forests regenerating on abandoned pastures in the Brazilian Amazon were typically dominated by either the pioneer species *Cecropia* or *Vismia,* with the former favouring land that has received comparatively less intensive use prior to abandonment. Both communities could be discriminated from Landsat sensor data, particularly using the near infrared and shortwave infrared wavelength regions. Whilst many of the associated species were not observed, with these often exceeding 50 after 10 years of growth, their presence could be inferred from *in situ* data and an understanding of successional processes. Hence, potential exists for mapping the distribution of plant species based on knowledge of those that dominate or co-dominate the upper canopy and which are spectrally distinct.

In terms of faunal diversity, remote sensing has been used to directly observe animals and primarily larger mammals, including marine cetaceans but, in many cases, this cannot be achieved because of their size and behaviour. The ability to assess biodiversity in terms of fauna can, however, benefit from knowledge of the distribution of certain plant species, genera or communities mapped from remote sensing data. The issue then becomes one of scale, in terms of faunal use of the habitat and its components, temporal movements of species and the dependence of species upon others and the environment, whether through synergism (e.g., spiders and flowering plant species), behaviour (migration, breeding, response to fire), predator-prey relationships (e.g., insects, birds through to birds of prey) or ability to transform environments (e.g., elephants and woodlands). Consideration also needs to be given to the three-dimensional structure and also spatial arrangements of habitats as these components are often used by fauna. In the case of the former, an increasing amount of data are becoming available at different scales, including Synthetic Aperture Radar (SAR), LiDAR, terrestrial laser scanners and stereo imaging from Unmanned Airborne Systems (UAS). Object-based approaches can also characterise a landscape in terms of its spatial components and relate these to define a set of habitats that are suitable for a range of species. For example, water voles require the presence of open water, banks to reside in and structured vegetation (e.g., tussock grasslands or shrubs). However, whilst this information can be obtained, establishing the extent of and describing faunal diversity (e.g., in terms of abundance, richness etc.) often requires the use of species distribution or niche models.

For modelling, inputs that have been outlined above can include maps of individual plant species, genera or communities that, in turn, can be combined to produce habitat maps with these needing to incorporate non-lifeform components (e.g., water, rock, soils). The three-dimensional structure of habitats and the landscape in general can be established from sensors such as LiDAR, with recognised limitations associated with resolution and accuracy, and the spatial arrangement can be defined using object-based approaches and landscape metrics (e.g., connectivity, adjacency). However, the following also have to be considered:

a) The environmental characteristics (e.g., wetness, greenness, amount of dead material, temperature, acidity).

b) Temporal variability in the life cycle of plants (in terms of flowering, phenology, growth stage as well as human-driven activities such as harvesting, sowing) but also in the fauna themselves (e.g., breeding, feeding, migration).

In the case of a) and b), such information can be obtained from remote sensing data through the use of spectral indices, that reflect the productivity of the vegetation, senescence and wetness, and seasonal observations that can track vegetation growth and dieback, leaf flush and fall, water inundation over various time scales and agricultural practices.

The challenge within BIOSOS is therefore to establish the optimal approaches to using the remote sensing observations to support the mapping and monitoring of habitats and their contained biodiversity.

3.2.2 Landscape modelling

Landscape modelling is aimed at exploring the relations between biodiversity surrogates, habitat structure and quality, on the one hand, and landscape characteristics, mainly structural ones, on the other.

A long tradition of studies carried out under the shell of landscape ecology indicate a tendency to relate particular biodiversity surrogates (e.g. species richness) or habitat selection to landscape characteristics, expressed by LPI index considered as relevant (e.g. McGarigal and McComb, 1995; Jones et al., 2000; Niemi et al., 2004, Didham, 2010) . An important stream of such studies is that related to the effects of habitat fragmentation, most of which are examined in an extensive review by Fahrig (2004). One of the issues emerged from this review is that relevant to the need for a measure of fragmentation (at habitat and landscape levels) which would be independent from habitat loss (or habitat amount). Indeed, as also noted by Didham et al. (2011), controlling for habitat amount has become a must in this kind of studies. Nevertheless, this device is not sufficient to solve the problem of underestimation of the effects of one factor over another, as the actual problem lies in the need for partitioning the indirect contributions of habitat loss (amount) and fragmentation to the intercorrelated portion of variance (Didham et al.,2011). This is due to the inherent interdependence rather than independence of factors affecting species. Such factors may belong to different hierarchical levels (e.g. patch and landscape levels), and interdependencies may occur both within and across levels. Therefore, in their article Didham et al. (2011) call on the need for and provide a conceptualization of a hierarchical causal model of interdependent habitat effects (e.g., collinearity of effects of spatial habitat configuration on their effects on species), possibly incorporating interdependence of species responses (e.g., covariance of species responses to species to habitat fragmentation). They also argue that interdependence of habitat effects and interdependence of species responses must be judged empirically on a case-by-case basis. Such an approach is advocated for the progress towards the understanding the effects of landscape change on biodiversity (Didham et al., 2011).

3.3 Habitat and landscape modelling in BIO_SOS

3.3.1 Common general methodological issues for all sites

A habitat and landscape modelling exercise has been envisaged within the BIO_SOS project, aimed at a comparative study across BIO_SOS sites for the assessment of the relations between habitat and landscape relevant aspects of biodiversity at different spatial scales. This has been assessed mainly using the available remote sensing data in combination with local scale *in situ* surveys of plant and animal community structure.

With respect to habitat modelling the question being asked is: “is local community structure related to measures of habitat quality at local and larger scales?” That is, to what an extent would managing for habitat quality be beneficial to biodiversity?

With respect to landscape modelling the question being asked is: “how are landscape heterogeneity and/or landscape configuration related to one another, and how do they (directly or indirectly) affect local community structure?”

3.3.1.1 Sampling strategy

A hierarchical nested sampling strategy was adopted for the collection of both field and RS data. From a theoretical point of view, such a choice descends from the (by now unquestionable) notion of the hierarchical organisation of ecological systems and landscapes. The hierarchy of landscape pattern corresponds to a hierarchy of decisions of faunal organisms, whose levels and objects are considered as common to all species (Orians, 1980), whilst the ranges of spatio-temporal scales relevant to such levels and objects are considered as species specific (Holling, 1992; Wiens e Milne 1989; Wiens 1997). Hence, the concept of scale is not alien from the notion of biodiversity. From an operational point of view, this strategy appears as the most appropriate one, in relation to the use of VHR RS data for all analysis, in order to avoid over-assumptions about species’ ecology while exploiting fine-grained information (Gottschalk et al. 2005, citing: both Pearson and Simons, 2002, and Morrison et al 1992). Moreover, such a strategy fits the hierarchical nature of the causal relationships across field samples within habitat patches within landscapes (Didham et al. 2011).
In particular, the analyses will be performed at different levels of the patch (near-decomposable) spatially nested hierarchy (after Wu and David 2002), identified by the spatial scale more likely to capture the characteristic scale of the ecological processes of interest.

Four levels have been recognised as relevant to the problem of interest within BIO_SOS (Table 3.1), considering that the selection of appropriate levels and extents depends on the limits of perception of the landscape of the organism for which the analysis is performed.

Table3.1 Levels of the analysis

Levels	Characteristics	What's in BIO_SOS
Regional landscape (**RL**)	Composed of different local landscapes Heterogeneous in ecosystem structure and function Characterised by the dominant biome and land use pattern in the region Climate-driven variations in land use pattern (Captured in some cases).	A sample of an EO image Possibly uniform in extent across sites (suggested extent 10x10 km)
Local landscape (**LL**)	Composed of different LC/LU and habitat types Heterogeneity of ecosystem structure and function (e.g. in IT3, characterised by the degree of fragmentation of the focal habitat and by the homogeneity/heterogeneity of landscape matrix; in PT2, characterised by small-scale mosaics of woodland and grassland patches)	Samples within the RL Not necessarily uniform in extent across sites (e.g. 1x1 km in IT3, 250x250 m in PT)
Local ecosystem 1 (**LE1**)	Relatively homogeneous vegetation-soil complex Discrete landscape elements (patches) comprised of relatively homogeneous vegetation-soil complexes (segments) (e.g., relating to human perception limits and the derived map scale 1:2,000-10,000)	LCCS classified segments (patches) within the RL (e.g. as resulting from clipping LL on to the RL (in GIS) Minimum extent in all sites depends on the common minimum mappable unit adopted within the project (i.e. 400m^2 as per the GHC methodology). Maximum extent depends on the LL extent as LE1s are by definition contained within an LL and can not be wider than this.
Local ecosystem 2 (**LE2**)	Subset of pixels within patch/segment relatively homogeneous vegetation-soil complex Will always include only one GHC and one Annex I habitat type	Sample plots within segments (suggested extent ≤ 400m^2)

3.3.1.2 Field data

Field data on community assemblages have been collected in the different BIO_SOS sites on different taxa reputed to serve as indicators of biodiversity (D6.7) mostly following the systematic sampling schemes defined in D4.3. In some cases (IN) data on the abundance of alien and invasive species were collected. In other cases (UK) already existing data were also collated deriving from more disparate, and even opportunistic, sampling schemes.

Selected organisms were, plants (PT, IN, IT3), insects (IT3, UK), birds (IT3, UK, NL).

3.3.1.3 RS data and features selection

An assessment of the potentials for using different RS information types and sources in relation to the particular problems investigated in each site has started from a reconnaissance of object features available (or shortly available) for each site and for each object type (pixel vs. segment). This represents the selection of all possible data available for each site which appear appropriate in relation to field data characteristics. In order to guide the selection of features, the feasibility to obtain data (in terms of time and computation costs) is also considered.

3.3.1.4 Scaling issues

The other aspect of scale is the *grain* (Kotliar and Wiens 1996) which, in this context, refers to the spatial resolution of the remote sensing image from which information is retrieved. In this study, contextual information at the appropriate scale is defined

using second order statistics such as Grey Level Co-occurrence Matrixes (GLCM, Haralick et al., 1973). This involves assigning a feature value to an individual pixel (object) derived from the values of the pixels in its neighbourhood. Hence, the scale will be affected by the computation even though the spatial resolution of the image remains the same.
Therefore, the definition of the values for such parameters, as much as the set of relevant extents will be selected for each site according to the “limits of perception” of the landscape of the organism for which the analysis is performed.

In ecological literature assessment of trends in land condition has traditionally involved scaling up monitoring information from small field plots to broad landscapes. Such assessments can now be done by simply applying remote sensing over large extents (i.e., by scaling down or transferring information from larger to smaller areas). Ludwig et al. (2006) do not hold this view and instead argue that remote sensing data and methods must match the scale of the problem being addressed. They propose a five step approach to establish landscape health by using remote sensing data at various scales. In most ecological studies upscaling of raster information (i.e. aggregation of raster data) is done by using a simple majority rule such as the average, or by random selection (He et al., 2002) without conducting analysis of the sensitivity to uncertainty for such collecting processes. In (Wessman and Bateson, 2006) a theoretical framework is presented that shows how aggregating information can unpredictably propagate errors, when the observed measure can be assumed as inherently non-linear. It applies for instance to NDVI, and demonstrates that upscaling is not justifiable if the underlying model is not linear and well understood.
Since the NDVI vegetation index is not a linear function of its variables (NIR and red reflectances), the NDVI of a superpixel need not equal the mean NDVI of its subpixels (i.e., the parameter-aggregated NDVI does not equal the band-aggregated NDVI). This discrepancy will introduce error in scaling up a functional relationship R between the NDVI and a ground parameter, even when R is linear - provided that the lower resolution pixels are not homogeneous. This has to be taken in consideration while dealing with NDVI (or any other index) at different scales, if computation has to be performed on band-aggregated values.

3.3.1.5 Statistical modelling issues

For the purposes of habitat modelling, correlation techniques will be exploited. These are reputed (Marcot 2006) to capture the degree of relation between the variables given an influence from different sources, not all of which may be known. In addition to traditional regression techniques, hierarchical models or multilevel models or mixed effects models (Gelman and Hill, 2007; Pinheiro and Bates, 2000) semi-parametric models (Ruppert et al 2003) will also be considered. The goodness of fit will be also evaluated by using a set of cross validation criteria (Carroll and Cressie 1996).
With regard to landscape modelling, simple causal models are thought to be incapable of handling the complexity deriving from the hierarchical interdependence among variables. As suggested by Didham et al. (2011) the possibility for the use of statistical techniques such as multi-level modelling or hierarchical models (Gelman and Hill, 2007) or semi-parametric models (Ruppert et al., 2003) will be explored.
The coupling of habitat and landscape modelling is potentially helpful to inform on the legitimacy of considering the actual spatial variables considered in the landscape models as “effects mechanisms” (sensu Didham et al. 2011) in their own right (sensu Didham et al. 2011), and of the underlying proximate abiotic and biotic factors/mechanism that are likely to trigger species or communities responses to landscape pattern.

3.3.2 Common protocols

A common, standard, scheme for arranging samples in a nested hierarchical fashion into a GIS has been agreed upon and is shown in Table 3.2.

Table 3.2 – Common scheme for arranging samples in a nested hierarchical fashion.

GIS database fields	Description	Notes
FID = 0 ÷ i		
RL_ID =	code of the BIO_SOS site	Regional landscape
LL_ID = 1 ÷ n	n= 100 if the extent of the RL is 10x10 km and the extent of LL is 1x1 km	Local landscape
LE1_ID = 1 ÷ m	m_n=number of LE1 in LL_n | $M=m_1+m_2+\ldots+m_n$	Local ecosystem 1 (= patch = segment)
LE2_ID = 1 ÷ j	j ≥ 30 number of sample plots for each focal class	Local ecosystem 2 (= aggregate of pixels found within the sample plot)
Selected	Y if the LL_n has been selected for sampling	
Impact Class	1 ÷ 5 or 1 ÷ 3 from lower to higher impact	
GHC map	Y if in the LL_n, GHC mapping on a 1/16 km (or wider) is envisaged or already carried out (in the latter case please provide relevant shp files and database)	
LCCS	LCCS code	
Focal Class	Y if in the LCCS class is the focal class for the dataset	
LE2_ID_x, y	coordinates of the centroid of the plot	

A common, standard field data summary form is proposed (Table 3.3). An individual database is meant to be to be produced for each RL (relevant to each site).

Table 3.3 – Common standard field data summary form.

Field	Taxa	Description	Notes
Latin name and Author (*)	all		indication of reference required (e.g. Pignatti S. 1982 Flora d'Italia Edagricole Bologna, or equivalent for birds and insects)
Ellenberg Code(****)	plants		
English name	all		
National name	all		
Local name	all		
Conservation status(**)	all	regional, national, EU and IUCN	indication of reference required (e.g. National Red List of vascular plants of Italy)
Annex I	all	code	

Relation to Annex I habitat/regional landscape	all	alien/invasive	
Distribution qualifier relative to Regional Landscape (RL)	all	rare/common	
	all		
Ellenberg's values	plants		indication of reference required(e.g. Pignatti 2001) to national systems
Chorology	plants		
Life Form	plants		

3.4 Progress to date

3.4.1 Habitat and landscape level modelling in IT3

Both habitat and landscape modelling will be performed in IT3 using data on passerine birds and insects (Orthoptera and Lepidoptera) as response variables and plant community as well as RS features as independent variables.

To date levels and spatial extents for the analysis have been defined (sub-section 3.4.1.1), the field sampling has been completed for plants birds and insects (sub-section 3.4.1.2) the relative databases have been populated. Data are being pre-processed. Preliminary results from data processing are presented section 3.4.1.3.

The rationale for selecting RS features (pseudo-spectral and statistical) from available sources has been delineated and RS data extraction and database population is ongoing.

3.4.1.1 Levels and spatial extents for field sampling and modelling in IT3

The analyses will be performed according to a nested hierarchical sampling design comprising four scale levels (with regard to extent) as shown in Figure 3.1, to which additional extents might be added (Figure 3.2). Samples were stratified according to the fragmentation ranking of local landscapes (D.3, Mairota et al. 2012) into fragmentation classes with regard to focal habitat type (Figure 3.1 left). The thirty LL were divided into three groups: group 1, comprising 4 landscapes, where the focal class is not fragmented (NF); group 2a and 2b, comprising 4 and 7 landscapes respectively, where the focal class is moderately fragmented and fragmented in a homogeneous matrix (MF_hom, F-hom); group 3a and 3b, comprising 5 and 10 landscapes respectively, where the focal class is moderately fragmented and fragmented in a heterogeneous matrix (MF_het, F_het).

The selection of appropriate levels and extents was done according to the limits of perception of the landscape of the organism for which the analysis is performed (Figure 3.1 right) For birds, as an example, LL and LE1 might be the most relevant levels, yet within the LL, additional extents can be identified as buffers of the point used as base for the count (Figure 3.2 left). For insect, as an example, LL, LE1, and LE2 might be the most relevant extents, yet in addition within the LE2, additional extents can be identified as buffers of the transect along which the survey was carried out (Figure 3.2 right). For plants, LL, LE1 and LE2 will likely be the relevant levels regarding community assembly and structure.

An example of the GIS database for each taxon is provided in Figure 3.3.

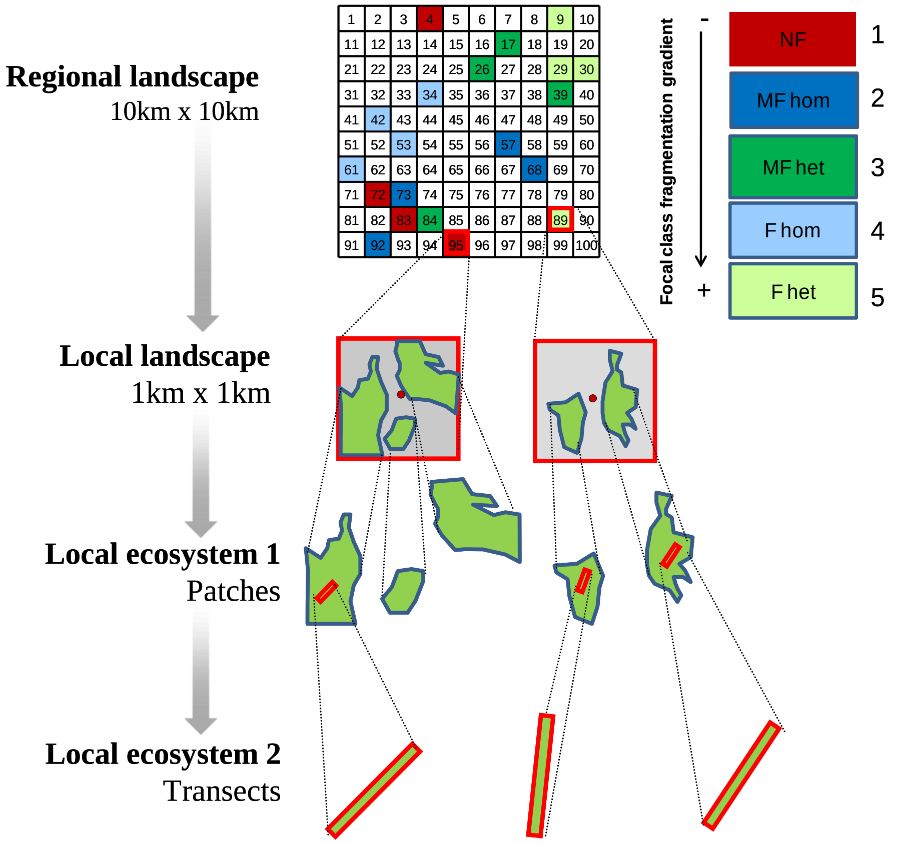


Figure 3.1 Levels, set of extents, stratification criteria and sample points/plots of the nested hierarchical sampling design for field data collection(birds and insects) in IT3 (following and Didham et al., 2011, Wu et al., 2012).

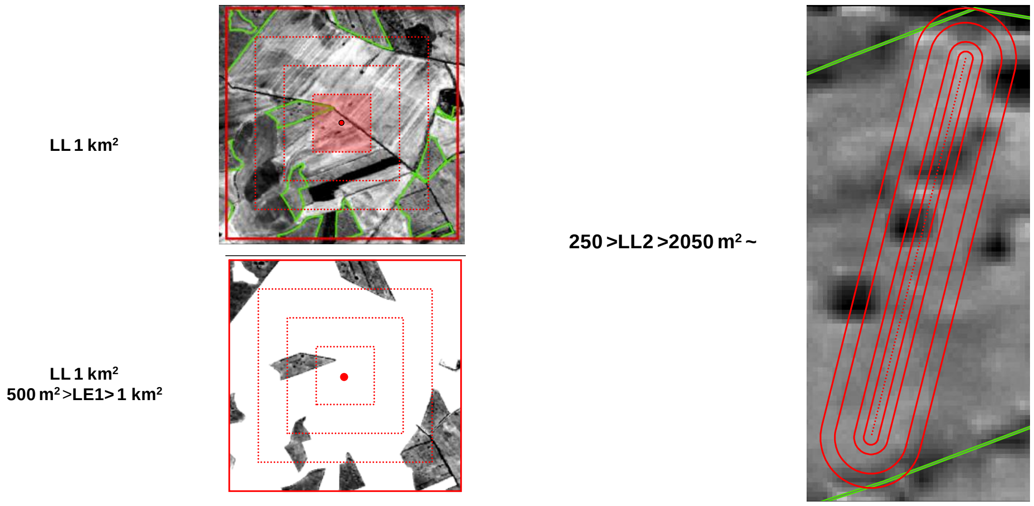
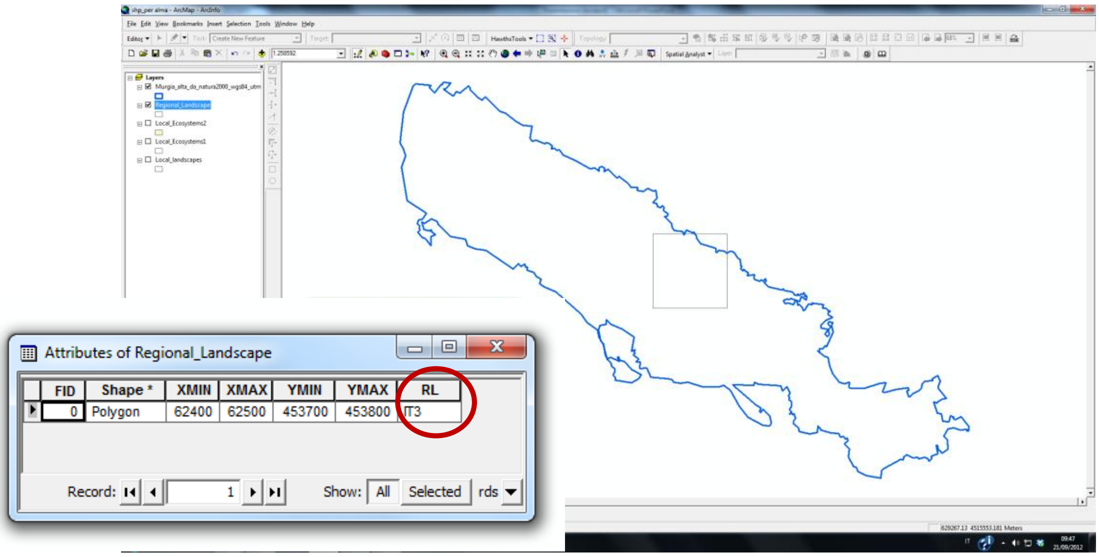
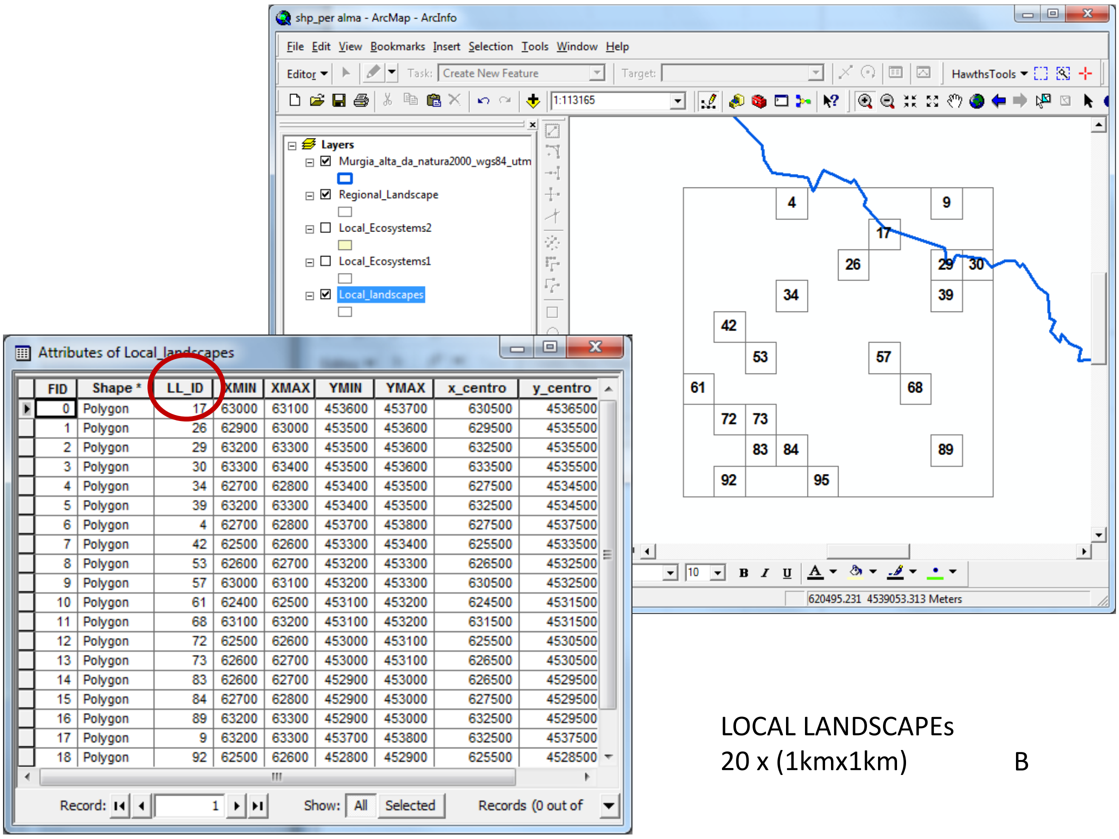


Figure 3.2 Additional set of extents for modelling for birds (left) and insects (right) in IT3.

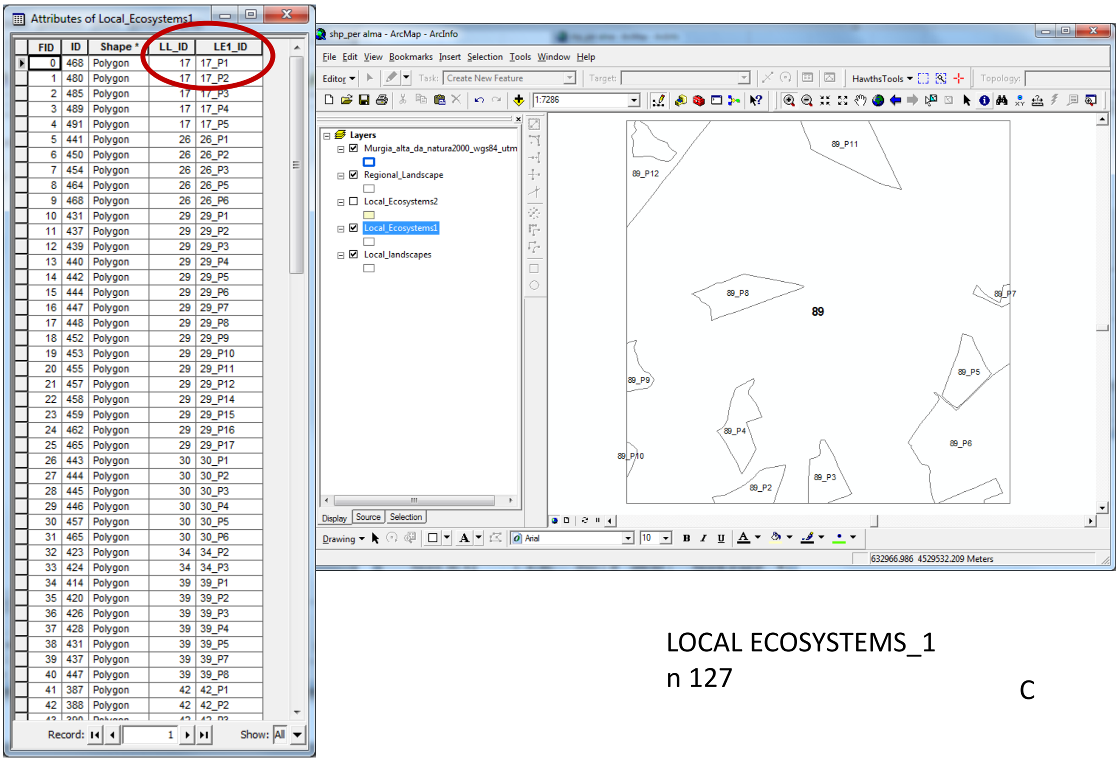


REGIONAL LANDSCAPE
10km x10km

A

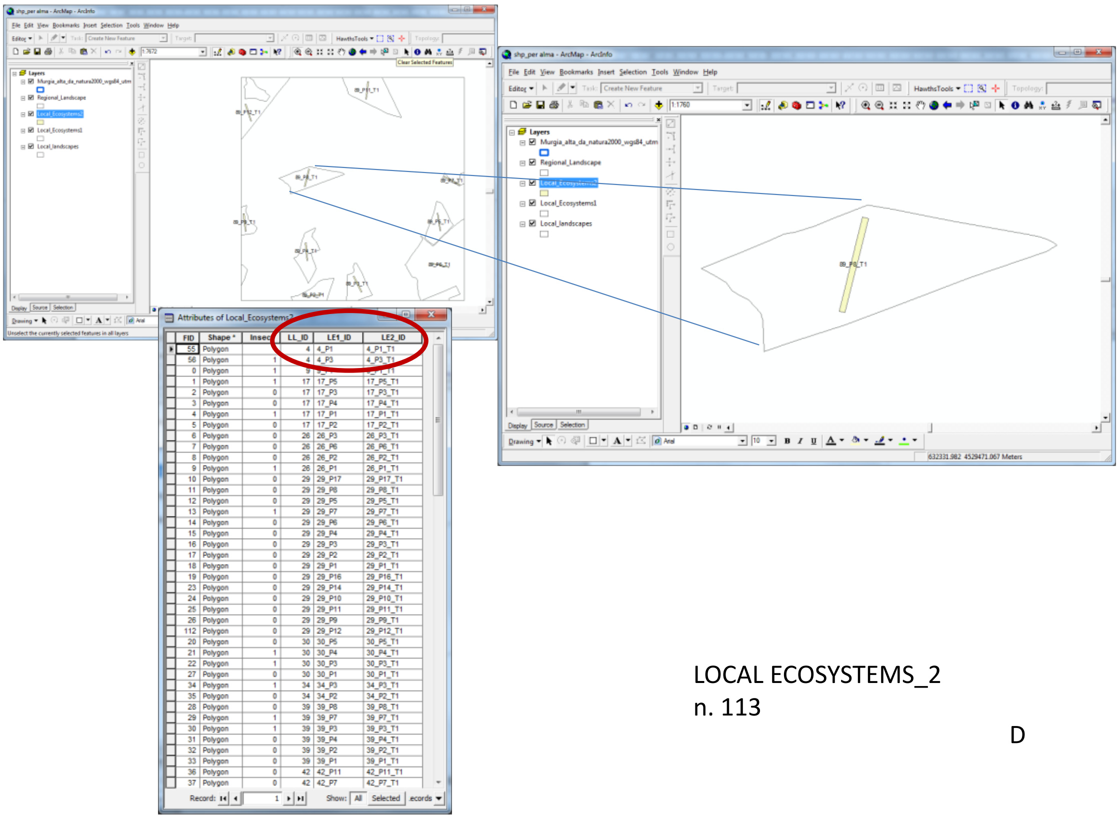


LOCAL LANDSCAPEs
20 x (1kmx1km) B



LOCAL ECOSYSTEMS_1
n 127

C



LOCAL ECOSYSTEMS_2
n. 113

D

Figure 3.3 GIS database organization for IT3 (see A, B, C, D for each level).

3.4.1.2 Methods for field sampling for plants birds and insects in IT3

The field campaign was carried out in 12 sampling sessions (24 days), from March to September 2012, repeating surveys every two weeks into each sample units.

Passerine breeding bird data were collected both in the central point of the 20 LL areas and along the 30 LE2 (transects). In all cases, each count was conducted in early morning and repeated during breeding season (May-July), recording birds heard or seen in a 15 minute period and within a distance of about 100m (Bibby et al. 1992). In addition, observations concerning non-passerine and non-breeding species were also recorded.

Adults of Lepidoptera Rhopalocera (Butterflies) and Orthoptera (Grasshoppers) were collected along the 30 LE2 linear transects, using a butterfly-net (Pollard, 1979). Samplings were repeated 12 times throughout the sampling season (March-September) in order to comply with the phenology of each species. Weather conditions, as cloudiness and wind, were also recorded for each sampling session.

Vegetation data were collected on LE2, both recording species richness along transects and carrying out vegetation relieves on smaller sample units (one or two for each transect). Plant species were collected throughout the sampling season, merely listing the species identified along sample transects. Vegetation *relevés* were carried out in late spring, when the vast majority of species is recognizable, using the phyto-sociological method (Braun-Blanquet 1932). These surveys provide further information about relative abundance (cover) of each species, together with abiotic parameters as stones/ rocks cover, soil slope and exposure. As the study mainly focuses on grassland/herbaceous communities, trees and shrubs were always recorded separately.

3.4.1.3 **Preliminary results from explorative processing of field data on plants birds and insects in IT3**

Insect and plant species were collected from March to September 2012, during 12 sampling sessions. The cumulative number of species (Figure 3.4) increased rapidly in the first half of the sampling period, then gradually settled onto a plateau, as the majority of species were discovered. Although the number of species generally varied by < 5 during the summer, the sampling effort continued until September to ensure observation of species late into the season and provide a more complete view of seasonal trends.

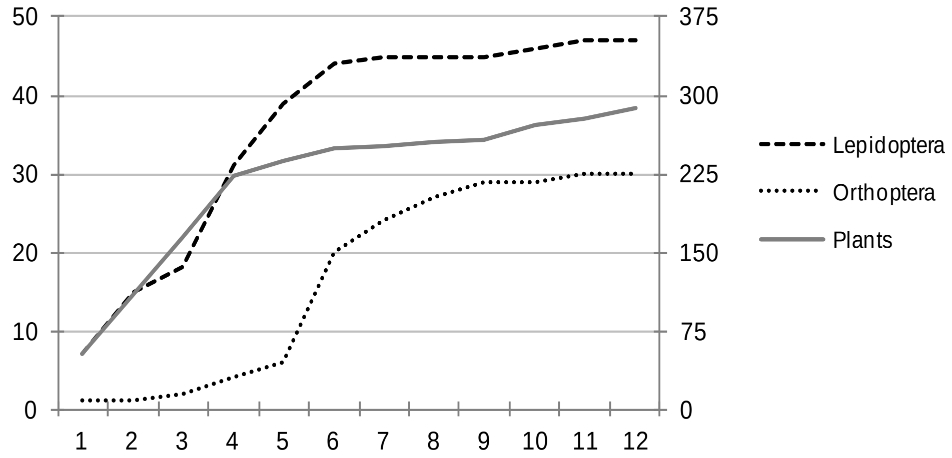


Figure 3.4 – Species discovery curves of Lepidoptera and Orthoptera (right Y axis) and Plants (left Y axis) in function of the cumulative number of sampling sessions (N = 12).

The number of species changed significantly throughout the season (Figure 3.5): the number of flowering herbaceous plants increased rapidly in the spring, reaching a peak in the first half of May; it is followed by a maximum in butterfly species number, as the presence of flowerings met the feeding habits of adult Lepidoptera. The number of Orthoptera species then increased with the coming of dry season.

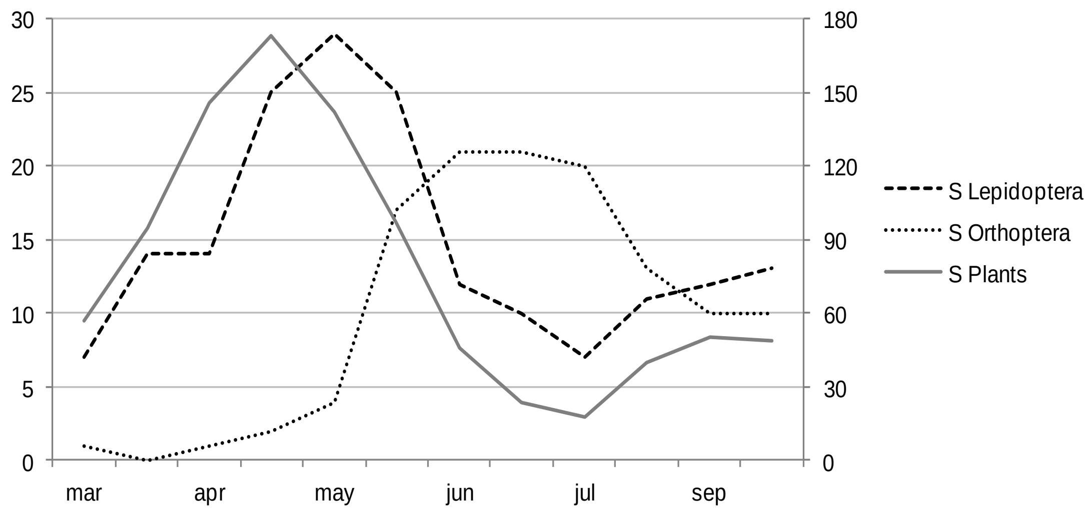


Figure 3.5 – Seasonal trend of species number of Lepidoptera and Orthoptera (right Y axis) and flowering plants (left Y axis).

298 overall plant species were recorded in the sample sites, with a mean value of 71.4 species per transect (SD = 16.6). The species composition within the herbaceous community was remarkably variable among different transects (Figure 3.6), mainly because of the different typology and degree of human pressure inside each location: extremely low values of species richness (< 40 species) were recorded in heavily perturbed areas (transects 34_P3, 42_P10), where overgrazing and continuous mowing activity seemingly reduced diversity; By contrast, very high values of species richness (> 85) were present in communities dominated by both therophytes (e.g. 30_P3, 89_P8) or hemicryptophytes (e.g. 95_P6, 84_P2).

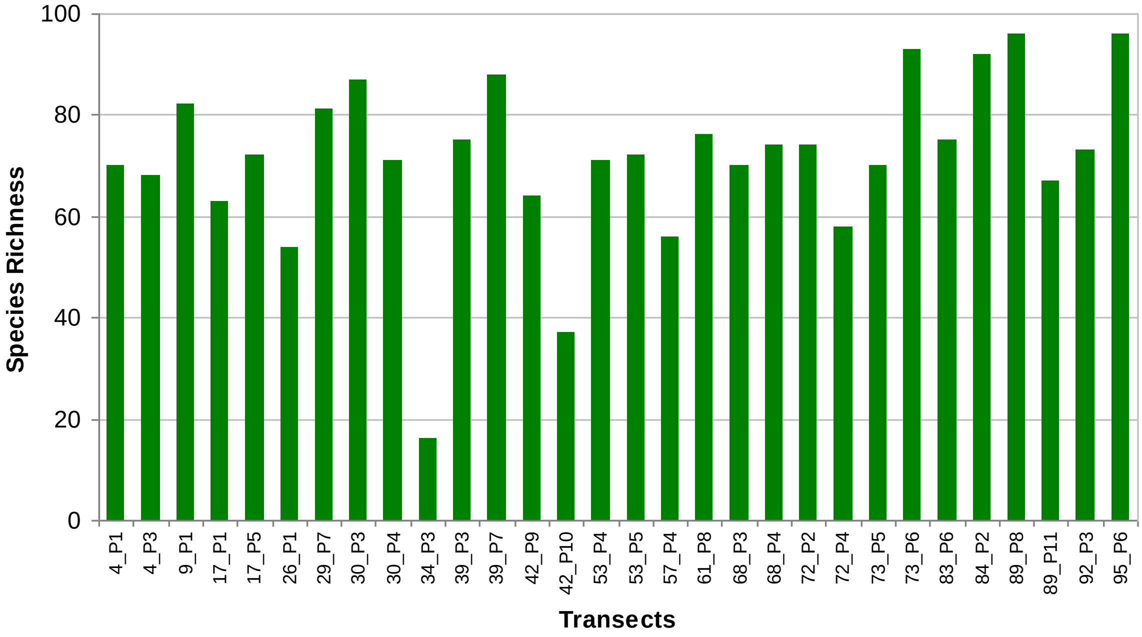


Figure 3.6 – Distribution of Plant Species Richness among LE2 transects.

The seasonal trend of flowering herbaceous species (Figure 3.7) overlapped with the peaks of living biomass in the majority of sampled species. Furthermore, sorting the species on the basis of their height, a different phenology in the flowering period (and biomass) among high (>60cm), medium (25 < 60cm) and low (<25cm) grassland layer (Figure 3.7) was evident. This was attributed to the very small plant species dominating the community towards the end of the winter, the medium-layer species rapidly grew during spring (being the majority of diversity of these grasslands) and the tallest plants mainly bloomed towards the end of spring, when other layers were beginning to dry. In autumn, there was a second peak in flowerings, which was attributed to re-flowering and autumn-flowering species; this peak was directly caused by the increased water supply and was observed earlier in burnt areas.

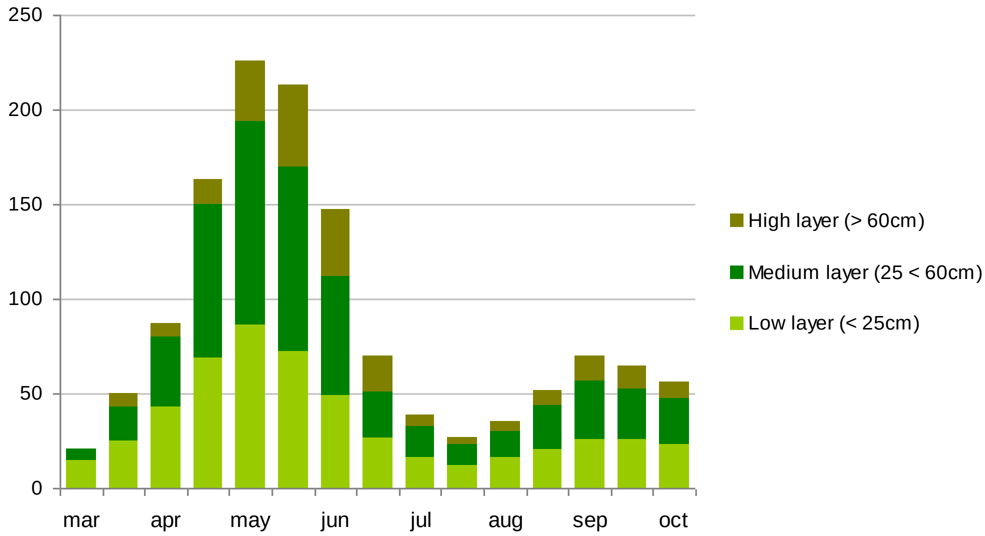


Figure 3.7 – Seasonal variation of Species Richness among three vegetation layers: high (>60cm), medium (25 < 60cm) and low (<25cm) layer.

The life-forms assortment of the whole sample (Figure 3.8) was comparable to other Mediterranean plant communities, with an high percentage of Therophytes and Hemicryptophytes. In this case study, 15 % of plants were Geophytes, with these being in the family Orchidaceae (17 species). Shrubs and trees,which are infrequent in the area, were mainly represented by old planted fruit trees (almond, fig, olive) and scattered deciduous shrubs, especially Almond-leaved Pear (*Pyrus spinosa* Forssk.) and Rock Buckthorn (*Rhamnus saxatilis* Jacq. *infectorius* (L.) P. Fourn.).

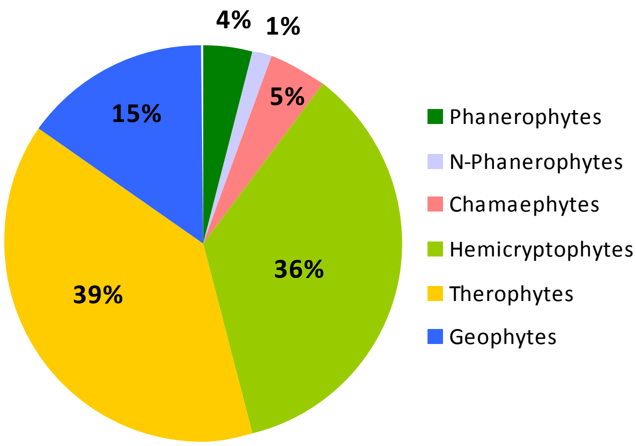


Figure 3.8 – Plant life-form spectrum of sampled grass communities.

As well as Species Richness (Figure 3.6), the combination of life-forms greatly varies among transects (Figure 3.9). In fact the increase of species number is generally well partitioned among life-forms.

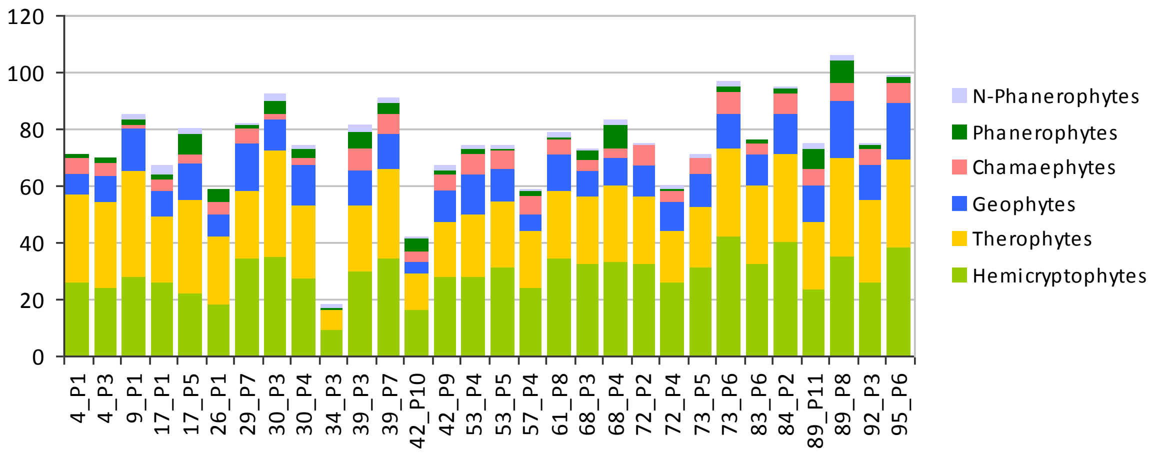


Figure 3.9 – Plant life-form spectrum within LE2 transects.

Lepidoptera Rhopalocera – Butterflies

49 butterfly species were observed in sampled areas, with an estimated total abundance of 1062 individuals. For each species, percentage values of dominance (relative abundance) and prevalence (relative occurrence in sample transects) were calculated (Table 3.4).

Table 3.4 – List of Lepidoptera species with associated number of individuals, transects of occurrence, dominance and prevalence indexes.

Species	n individuals	n transects	Dominance	Prevalence
Anthocaris cardamines	5	4	0.47	13.33
Aporia crataegi	78	28	7.34	93.33
Brenthis hecate	2	1	0.19	3.33
Brintesia circe	8	8	0.75	26.67
Callophrys rubi	1	1	0.09	3.33
Carcharodus alceae	6	5	0.56	16.67
Carcharodus boeticus	2	2	0.19	6.67
Carcharodus flocciferus	2	1	0.19	3.33
Coenonympha pamphilus	28	18	2.64	60.00
Colias alfacariensis	6	5	0.56	16.67
Colias croceus	17	14	1.60	46.67
Euchloe ausonia	11	10	1.04	33.33
Glaucopsyche alexis	1	1	0.09	3.33
Gonepteryx rhamni	5	5	0.47	16.67
Hipparchia semele	33	22	3.11	73.33
Hipparchia statilinus	44	27	4.14	90.00
Iphiclides podalirius	9	7	0.85	23.33
Issoria lathonia	2	2	0.19	6.67
Lasiommata megera	23	21	2.17	70.00
Limenitis reducta	5	5	0.47	16.67
Lycaena phlaeas	7	7	0.66	23.33
Maniola jurtina	66	21	6.21	70.00
Melanargia arge	42	20	3.95	66.67
Melanargia galathea	11	4	1.04	13.33
Melanargia russiae	314	30	29.57	100.00
Melitaea didyma	13	10	1.22	33.33
Melitaea phoebe	1	1	0.09	3.33
Muschampia proto	7	6	0.66	20.00
Papilio machaon	32	24	3.01	80.00
Pieris cfr. mannii	1	1	0.09	3.33
Pieris brassicae	5	5	0.47	16.67
Pieris rapae	21	19	1.98	63.33
Plebejus agestis	11	9	1.04	30.00
Polyommatus bellargus	22	10	2.07	33.33
Polyommatus icarus	31	21	2.92	70.00
Pontia edusa	46	27	4.33	90.00
Pseudophilotes baton	2	2	0.19	6.67
Pyrgus malvoides	3	2	0.28	6.67
Pyrgus sidae	2	1	0.19	3.33
Pyronia cecilia	27	15	2.54	50.00
Satyrium ilicis	4	2	0.38	6.67
Satyrium spini	68	22	6.40	73.33
Spialia sertorius	7	5	0.66	16.67
Thymelicus acteon	3	3	0.28	10.00
Thymelicus lineolus	13	8	1.22	26.67
Thymelicus sylvestris	10	7	0.94	23.33
Vanessa cardui	5	5	0.47	16.67
N	**1062**	**30**	**100.00**	**100.00**

Although the distribution of abundances was good overall, some species were dominant. *Melanargia russiae* was the most abundant (314 individuals) and was ubiquitous in study area; other common and abundant species were *Aporia crataegi* (78 ind.) *Satyrium spini* (68 ind.) and *Maniola jurtina* (66 ind.). All of these species were relatively uncommon elsewhere in the region, but were in the study area possibly due to better ecological conditions (e.g. host plants availability for both larvae and adults).
Within transects, an average of 15.8 species were present, with a maximum of 29 species occurring in LE2 68_P3 and only 7 - 8 species in 4_P1, 53_P5 and 72_P4 (Figure 3.10). Highest numbers were generally present where the habitat was more heterogeneous but the lowest numbers were found were human pressures (e.g., overgrazing) were high.

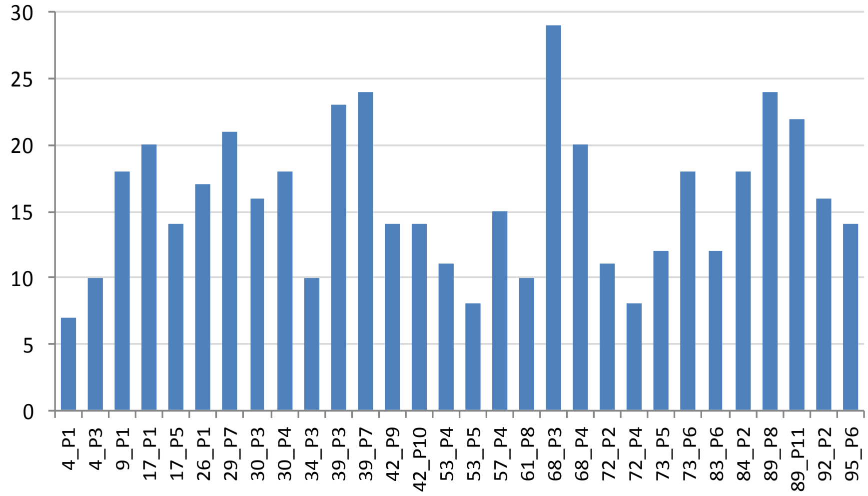


Figure 3.10 – Distribution of Lepidoptera species richness among LE2 transects.

The distribution of the Shannon's diversity value (H') is comparable to that of species richness, as they also show a good mutual correlation ($r = 0.753$, $p < 0.01$).

Orthoptera – Grasshoppers

30 Orthoptera species were observed in sampled areas, with 1390 individuals recorded. For each species, percentage values of dominance (relative abundance) and prevalence (relative occurrence in sample transects) were calculated (Table 3.5).

Table 3.5 – List of Orthoptera species with associated number of individuals, transects of occurrence, dominance and prevalence indexes.

Species	n individuals	n transects	Dominance	Prevalence
Acrida hungarica	1	1	0.07	3.33
Acrometopa italica	10	7	0.72	23.33
Anacridium aegyptium	1	1	0.07	3.33
Calliptamus italicus	365	30	26.26	100.00
Chorthippus cfr. brunneus	71	24	5.11	80.00
Decticus albifrons	26	12	1.87	40.00
Decticus loudoni	3	3	0.22	10.00

Dirshius petraeus	3	2	0.22	6.67
Dociostaurus genei	10	5	0.72	16.67
Dociostaurus maroccanus	38	6	2.73	20.00
Ephippiger apulus	20	10	1.44	33.33
Euchorthippus declivus	147	23	10.58	76.67
Locusta migratoria	5	4	0.36	13.33
Oedaleus decorus	3	1	0.22	3.33
Oedipoda caerulescens	24	10	1.73	33.33
Oedipoda germanica	50	22	3.60	73.33
Omocestus rufipes	2	1	0.14	3.33
Pezotettix giornae	35	13	2.52	43.33
Pholidoptera femorata	1	1	0.07	3.33
Platycleis affinis	2	1	0.14	3.33
Platycleis falx	17	6	1.22	20.00
Platycleis intermedia	95	27	6.83	90.00
Platycleis stricta	98	18	7.05	60.00
Platycleis tessellata	143	22	10.29	73.33
Prionotropis appula	17	15	1.22	50.00
Ramburiella turcomana	12	1	0.86	3.33
Saga pedo	4	4	0.29	13.33
Sepiana sepium	15	12	1.08	40.00
Tettigonia viridissima	4	4	0.29	13.33
Tylopsis liliifolia	168	24	12.09	80.00
N Orthoptera	**1390**	**30**	**100.00**	**100.00**

Calliptamus italicus (365 individuals) was an important dominant species, which was present in all transects. Very high values of abundances were recorded for *Tylopsis liliifolia* (168), *Euchorthippus declivus* (147) and *Platycleis tessellata* (143), with these being species that prefer sites where grass is prevalent.

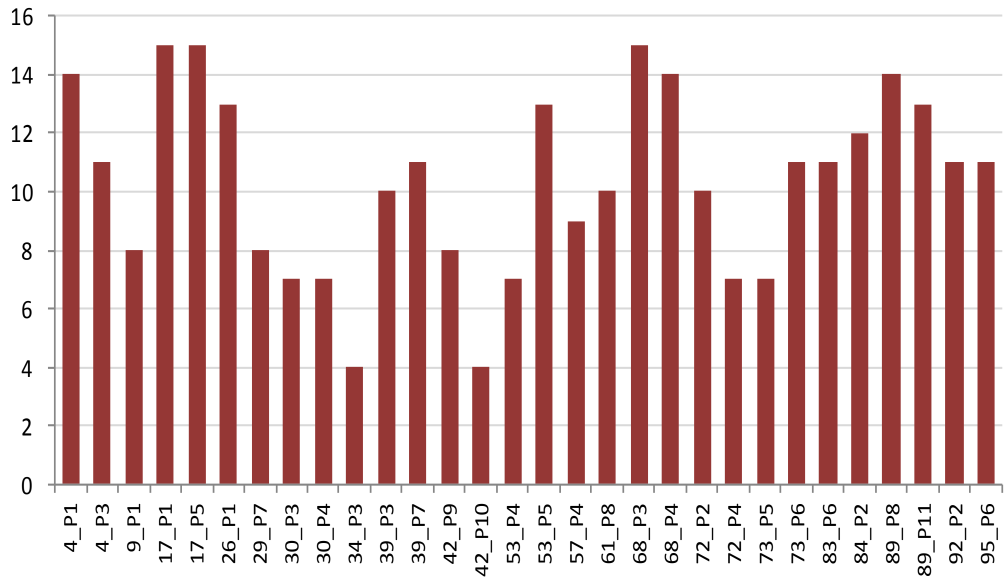


Figure 3.11 - Distribution of Orthoptera species richness among LE2 transects.

The distribution of species richness among transects (Figure 3.11) shows very depleted assemblages (4 species) in highly disturbed transects (34_P3, 42_P10); maximum values of species richness (15) were observed in LE2 17_P1, 17_P5 and 68_P3. The values of species richness (S_{mean} = 10.3) and Shannon's diversity (H'_{mean} =1.89) were the most correlated (r=0.83, p<0.01). On the basis of ecological information and actual observations, Orthoptera species were grouped in 4 categories, according to the concept of life-forms (Bei-Bienko 1950, Pravdin 1978): Thamnobionta, Chortobionta, Geo-Chortobionta and Geobionta (Figure 3.12).

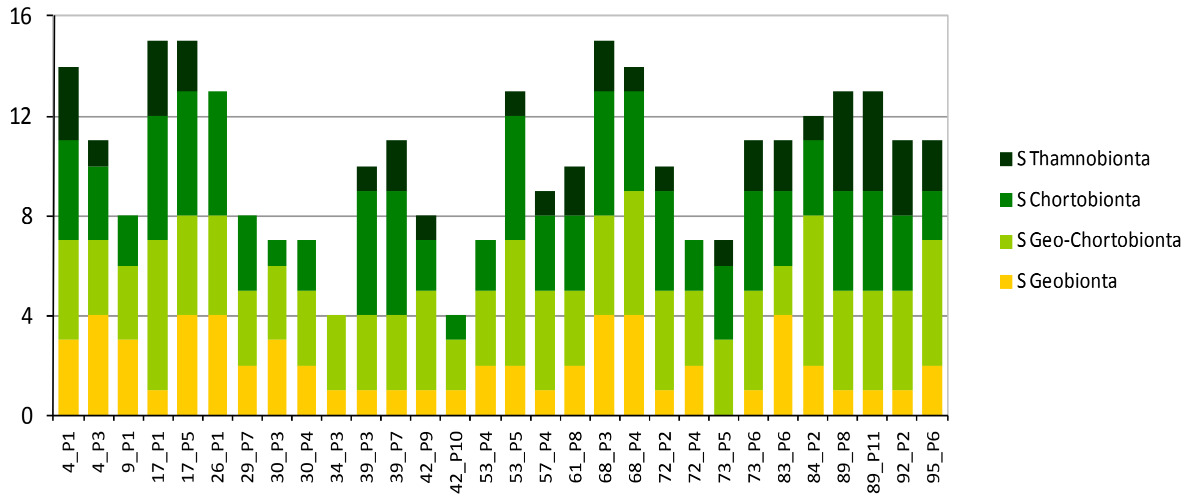


Figure 3.12 – Distribution of Orthoptera life-forms among LE2 transects.

It is interesting to note that those life-forms normally associated with tall/complex vegetation (especially Thamnobionta) were absent or under-represented in transects with poorer plant community (Figure 3.7). On the contrary, the number of Geobionta species was relatively higher in areas where extensive areas of bare soil surfaces (4_P3, 26_P1, 68_P3, 9_P1) or tracks (83_P6) occurred, with these generally formed by human activities. On this basis, life-form categorization may be a useful method for selecting functional species groups and indicators of grassland quality.

Passerine breeding bird samples were repeated during breeding season (May-July) to ensure the observation of all species. In total, 26 species were observed in sampled areas, with a total abundance of 297 individuals along the 30 transects (which are not always independent samples) and 168 in the 20 LL plots. For each species, the percentage values of dominance (relative abundance) and prevalence (relative occurrence in sample transects) are reported in Table 3.6.

Table 3.6 - List of Lepidoptera species with associated number of individuals, transects of occurrence, dominance and prevalence indexes.

English name	Species	n individuals	n transects	dominance	prevalence
Skylark	*Alauda arvensis*	8	7	2.69	23.33
Woodlark	*Lullula arborea*	13	11	4.38	36.67
Crested Lark	*Galerida cristata*	24	19	8.08	63.33
Calandra lark	*Melanocorypha calandra*	32	21	10.77	70.00
Short-ted lark	*Calandrella brachydactyla*	5	5	1.68	16.67
Tawny pipit	*Anthus campestris*	5	5	1.68	16.67
Barn Swallow	*Hirundo rustica*	24	8	8.08	26.67
Sardinian warbler	*Sylvia melanocephala*	16	12	5.39	40.00
Spectacled warbler	*Sylvia conspicillata*	4	3	1.35	10.00
Subalpine warbler	*Sylvia cantillans*	3	2	1.01	6.67
Blackcap	*Sylvia atricapilla*	3	3	1.01	10.00
Black-eared Wheather	*Oenanthe hispanica*	1	1	0.34	3.33
Stonechat	*Saxicola torquata*	4	4	1.35	13.33
Nightingale	*Luscinia megarhynchos*	2	2	0.67	6.67
Blue Tit	*Cyanistes caeruleus*	6	5	2.02	16.67
Great Tit	*Parus major*	18	12	6.06	40.00
Lesser grey shrike	*Lanius minor*	1	1	0.34	3.33
Common Chaffinch	*Fringilla coelebs*	3	3	1.01	10.00
Goldfinch	*Carduelis carduelis*	11	7	3.70	23.33
Serin	*Serinus serinus*	4	4	1.35	13.33
Italian Sparrow	*Passer domesticus italiae*	54	18	18.18	60.00
Corn Bunting	*Miliaria calandra*	13	12	4.38	40.00
Cirl Bunting	*Emberiza cirlus*	3	3	1.01	10.00
Golden Oriole	*Oriolus oriolus*	11	7	3.70	23.33
Magpie	*Pica pica*	24	18	8.08	60.00
Eurasian Jay	*Garrulus glandarius*	5	5	1.68	16.67
N tot		**297**	**30**	**100.00**	**100.00**

European Directive 147/2009, namely Woodlark, Caandra lark, Short-toed lark and Tawny pipit). In the list of Passerine birds (Table 3.6), Raven (*Corvus corax*) and Hooded crow (*Corvus corone cornix*) were not recorded because of their wide range and displacement of movement. A high diversity of species was observed, with these dominated by synanthropic species (Italian sparrow, Barn swallow), open-habitat specialists (Calandra lark, Crested lark) and wood species (Great tit). The observations of species recorded within each transect (Figure 3.13) conveyed the diversity of habitats and the species observed.

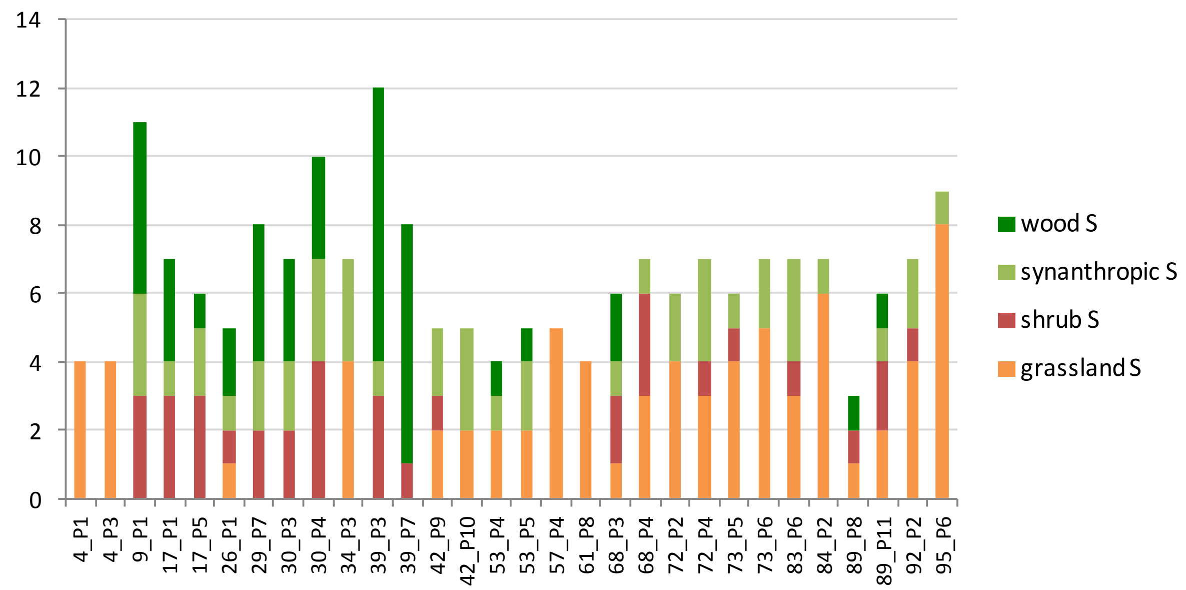


Figure 3.13 – Species richness of Passerine birds, characterized on the basis of ecological preference, among LE2 transects.

Wood and shrub specialist species were well represented in transect surrounded by oak woods (e.g. 39_P3, 39_P7, 9_P1) or pine plantations (26_P1, 29_P7). In these transects, grassland specialists were absent, which largely dominate more homogeneous habitat patches (e.g. 4_P1, 95_P6, 57_P4).

Synanthropic species were more regularly distributed and seem to be influenced only by the proximity of farmhouses and buildings, which were widespread in the sample sites. This prior classification on the basis of habitat preference will be used to improve the detection of indicators for grassland habitat quality and as response variables in future analyses.

3.4.1.4 Rationale for the preliminary selection RS features, metrics and computation parameters affecting the grain

For the appropriate selection of RS features and the relevant metrics to be used for the habitat and landscape modelling exercise in IT3 common, preliminary choices for the modelling of birds and insects communities were based on the available RS information for the site (Sub-section 3.3.1.3) as well as on existing literature (sub-Section 3.2). These basically concern the selection of the most appropriate image with respect to the date of acquisition and of the types of features to be computed.

Available images were DigitalGlobe™ Worldview-2 images (about 2 x 2m native spatial multispectral resolution) acquired at different dates and the one acquired in April 2011 was selected as this represented the closest to the conditions (Spring) in which the highest species richness values were verified for plants, birds and insects in the site during field campaigns (sub-section 3.4.1.3).

As a first instance, it was chosen to use both spectral features, such as the well–known Normalized Difference Vegetation Index (NDVI), and a statistical features, such as first and second order statistics commonly used in common texture analysis, as computed on Grey Level Co-occurence Matrixes (GLCM) (Haralick et al., 1973) because they are the most represented ones in published works applied to the applicative field.

The NDVI index is defined as follows:

$$\mathrm{NDVI} = \frac{\rho_{\mathrm{NIR}} - \rho_{\mathrm{RED}}}{\rho_{\mathrm{NIR}} + \rho_{\mathrm{RED}}} \tag{1}$$

Where ρ represent top-of-atmosphere (TOA) reflectances computed on near-infrared and red bands for the selected class of sensor.

The NDVI has been computed on all selected LLs at full image resolution, hence within a window of 1km^2 around the centroid of the LL (it results in 500x500 pixels per sample for each multispectral image).

Means and variances can be computed at different levels (i.e. LE1 and LE2) for that index within the target sample window. However, as the NDVI index is non–linear in its behaviour (Aman et al., 1992; Jiang et al., 2005; Karl and Maurer, 2010), this has been considered when correlating spatially averaged values across different scales.

In order to predict at least birds species richness and habitat preferences (St-Louis et al., 2006; Tuttle et al., 2006), the most appropriate metrics seems to be Variance/SD (in this case computed on the whole area of each LL, LE1 or LE2,) as well as GLCM second order metrics such as Information Measure of Correlation (IMC) (relevant to the degree of linear dependency of the values on those of the neighbours) and Inverse Difference Moment (IDM) or Homogeneity (Clausi, 2002). To these we choose to add the Contrast (CON), which provides measure of the amount of local variation in values among neighbours. Even though these metrics are correlated (St-Louis et al., 2006) due to their mathematical formulation, from the animal perception point of view they provide different kind of information. As an example Variance/SD informs on the relative degree of habitat homogeneity/heterogeneity. GLCM IDM values, which also informs on the same habitat characteristic, yet, by taking into an account (as all second order metrics) the spatial orientation and direction of pixels, informs on the relative aggregation degree within the co-occurrence matrix.
GLCM second order metrics are mathematically defined as follows (Haralick et al., 1973):

Contrast (CON)

$$\sum_{n=0}^{N-1} n^2 \left\{ \sum_{i=1}^{N} \sum_{j=1}^{N} p(i,j) \right\} \tag{2}$$

where N is the grey levels quantization considered, and $p(i, j)$ is the (i,j)th entry in a normalized grey level co-occurrence matrix.

Correlation (IMC)

$$\frac{\sum_i \sum_j (ij)\rho(i,j) - \mu_x \mu_y}{\sigma_x \sigma_y} \tag{3}$$

with μ_x, μ_y and σ_x, σ_y the means and the standard deviations of p_x and p_y, and p_x and p_y are the *i*th entries in the marginal-probability matrix $p(i, j)$ by summing the rows and columns of respectively.

Homogeneity (IDM)

$$\sum_i \sum_j \frac{1}{1+(i-j)^2} p(i,j) \qquad (4)$$

Besides common choices, distinct choices were taken for the two groups as for scale-related issues relevant to the differences of spatial perception limits between birds and insects.

The first differential choice is related to the selection of the most appropriate spatial resolution of the data (which should represent the organisms' lower limit of perception, i.e., the grain). For birds, it was assumed that the image native resolution (2m x 2m) could be appropriate but for insects, it was assumed that the most appropriate scale could be that derived by pansharpening (1m x 1m) the image. These assumptions, however, have to be intended in relative terms as, for instance, it is known that their lower limit of perception is generally of a sub-metric order of magnitude.

Other differential choices were those relevant to the actual feature to be used for the computation of the texture metrics, as well as those relevant to the setting of some parameters within the algorithms which are likely to more likely to affect the scale via indirect influence on the grain (sub-section 3.3.1.4). These are critical choices in relation to the differences in the scale and the mode of perception of the environment between insects and birds. In contrast to birds, insects cannot be resolved as "objects". Rather their perception and interpretation of the environment is based on physical as well as chemical clues such as shadow and temperature at very small scales (even smaller than 1m), particularly as they respond to environmental variables exhibited by the microhabitat level (e.g. soil moisture, soil pH, vegetation cover). Pan-sharpened NDVI data were therefore considered to be the most appropriate for insects, despite this potentially compromising the computation of textural metrics. The parameters used for setting the computation algorithms included window size, quantization, shift and direction.

Window sizes

In the case of birds, for which habitat and landscape modelling will be carried out with reference to both the landscape mosaic as a whole (heterogeneous) and the focal habitat (quasi homogeneous), the literature examined (St-Louis et al., 2006; Tuttle et al., 2006) concerns studies carried out both in homogeneous and heterogeneous environments and with both VHR (1m) and HR (15 m), suggests that windows of similar sizes yield similar results. Therefore it appears that rather distant window sizes should be selected (e.g. 3x3 and 31x31). As there is no ecological criterion to decide between these two grains (and hence scales), it is sensible to recall that hierarchical theory of patchiness, suggests to consider at least three levels of scale, one below and one above the scale of interest. We assumed that the scale of interest is that defined, for any given extent, by an intermediate window size between the two extremes, we should compute textural metrics using 3x3, 15x15 31x31 (in pixels) kernels. These correspond to 6x6, 30x30, 62x62 windows in meters which most probably reflect the limits of perception/home range of different passerine species and in addition allow for statistical analysis.

In the case of insects, for which the modelling will be carried out mainly with reference to the focal habitat (quasi homogeneous environment), to the best of our knowledge there are no studies on the relations between RS features and the composition of

insects communities. Our sample areas are often very small (80x5 m) and therefore small window sizes should be selected to avoid the influence of the context of the sample (within and outside the patch). This might also help the matching of the scale and the mode of perception of the insects. However when selecting one or another of the two possible extreme sizes (3x3 and 9x9) we should consider that each of these sizes is more appropriate to one or the other among two, possible guilds. These, however broadly defined, might emerge from differences based on dispersal abilities during breeding season, e.g. poor dispersers (not winged) vs. good dispersers (winged). Therefore it was chosen to work at both (3mx3m and 9mx9m) as well as at an intermediate window size (5mx5m).

Quantization

Quantization should be consisted with the window size (in order to provide a sample size good enough for the second order distribution modelling), therefore it was chosen to use 32 levels of quantization for birds and 16 levels for insects.

Shift

Information about shift size to be used are generally not provided by the published works examined. Therefore it was chosen to make reference to the only published indication found in Tuttle et al. (2006), i.e. a shift of one pixel.

Direction

GLC matrices can be computed for any direction and distance. It is assumed that all textural information is contained in the spectral value spatial-dependency matrices for canonical angles of 0, 45, 90, and 135 degrees.

We are aware that computation of all these metrics for many window sizes may be extremely costly, therefore an evaluation of the computation costs will also be carried out.

3.4.1.5 Rationale for the selection of landscape metrics.

Indices related to the main independent variables at both at landscape and patch were identified and computed in the previous quantitative landscape pattern analysis carried out within BIO_SOS (D6.3, Mairota et al. 2012).

Namely, relevant to landscape level, independent variables for “habitat quantity” “matrix quality” and “habitat isolation” were respectively identified in the percent of LL occupied by the focal habitat (PLAND), the contrast-weighted edge density (CWED), the effective mesh size (MESH), the contagion index (CONTAG), the Shannon’s diversity index (SHDI), the incidence of focal habitat classified as the non-core MSPA main structural class” islet”.

Patch level independent variables for “patch area”, “patch shape” and “edge effects” were respectively identified in the patch area index (PA), the fractal index (FRAC) and the edge contrast index (ECON).

3.4.2 Habitat modelling for species distribution in the an UK site

The BIOSOS project has focused primarily on the Cors Fochno Natura 2000 site in mid Wales. *Ad hoc* faunal and floral data have been collated from the Countryside Council for Wales (CCW) over a range of years (Table 3.7). Whilst these data were mostly not collected to support the BIO_SOS project, they have been collated and are available to support the objectives. The surveys encompass vegetation, insects, reptiles, birds and mammals and have been collated over variable periods and often on an annual basis.

Extensive surveys of vegetation have been carried out within the active bog but also the surrounding landscape. For BIO_SOS, General Habitat Category (GHC) surveys were carried out in 2011 for 15 1 x 1 km squares within the Dyfi catchment, including within the primary site of Cors Fochno. The catchment is considered as being the ‘boundary’ of the primary area where human activities will affect the conservation status of the estuarine raised bog, saltmarshes and sand dunes. GHC point (2 x 2 m), linear plot and area (10 x 10 m) surveys were also conducted for sites distributed within and in close proximity to the active raised bog. The primary habitats of interest are the active and degraded raised bog as well as the saltmarsh and sand dune complexes (Table 3.8) although other habitats within the catchment include grasslands, forests and heaths, as reported in WP5 deliverables.

A number of key faunal species have been surveyed which can be used in landscape pattern analysis and habitat suitability modelling, with these being small red damselfly, rosy marsh moth, large heath butterfly and Welsh vernal mining bee (insects), otter and water vole (mammals), lapwing (birds) and grass snake, slow worm and adder (reptiles). The habitat preferences for these species are listed in the following sections and outlined in Table 3.

Otters (*Lutra lutra*) are generalized feeders and are common in many river estuaries where food is abundant. However, the active bog provides freshwater, which is essential for the animals to clean their coats of salt and retain the insulation function. The bog area is used for resting and breeding being a large, relatively undisturbed wetland. Both flowing and standing water are used even though the ditches and pools do not supply much food. As otters are omnivores, they are able to supplement the fish and crustacean food supply of the estuary with invertebrates, amphibians and eels, which occur along the kilometres of flooded ditches on the bog.

Water voles (*Arvicola amphibious*) feed on aquatic vegetation that is either floating or emergent and includes young reed stems. The nest site is often a burrow in a river or ditch bank but can also be found in dense undergrowth close to ditches or streams. The species prefers still or slow flowing water, whether fresh or brackish.

Table 3.7 - Flora and fauna surveys undertaken at Cors Fochno.

Species	Code	Dates recorded	Description	Method of assessment	Recorded
Vegetation	AberleriNVC_01	1993	NVC vegetation survey	NVC quadrats + NVC map	NVC community type, species DOMIN values
Vegetation	AberleriNVC_02	Sep-02	NVC vegetation survey	NVC 4x4m + 7x5m quadrats + NVC map	NVC community type, species DOMIN values
Vegetation	Cors Fochno_01	2007	NVC vegetation survey	NVC quadrats +NVC map	DOMIN values, PFT, PFT heights, litter%
Vegetation	Cors Fochno_FX_01	2003	Rhynchosporion monitoring plots	Fixed plot monitoring, % cover of selected species	Sphagnum and Rhynchospora % cover of selected species
Vegetation	Cors Fochno_FX_01	2003	Degraded bog monitoring plots	Fixed plot monitoring, % cover of selected species	Sphagnum and Rhynchospora % cover of selected species
Vegetation	Cors Fochno_FX_02	2008-2009	Rhynchosporion monitoring plots	Fixed plot monitoring, % cover of selected species	Sphagnum and Rhynchospora % cover of selected species
Vegetation	Cors Fochno_FX_02	2008-2009	Degraded bog monitoring plots	Fixed plot monitoring, % cover of selected species	Sphagnum and Rhynchospora % cover of selected species
Vegetation	SW_Cors_Fochno_01	2002		NVC 2x2, 4x4, 1x4m quadrats +NVC map	NVC community type, species DOMIN values
Vegetation	Ty_Mawr_Cors_Fochno_01	1993	NVC vegetation survey	NVC quadrats	NVC community type, species DOMIN values
Vegetation	Ty_Mawr_Cors_Fochno_02	2011	NVC vegetation survey	NVC 2x2m quadrats + NVC map	NVC community type, species DOMIN values
Vegetation	Cors_Fochno_+	2009- 2011	GHC vegetation survey	GHC methods 10x10m,, 10x1m and 2x2quadrats	GHCs and species%
Birds	Cors_Fochno	2007-2011	BBS-RSPB transect survey	Transects yearly	Count of species spotted at time of transect recorded
Insects	Small red damselfy	1995-2011	Transect	Tranasect same time every year	Count of males, pairs, immature
Insects	Rosy marsh moth	1988 - 2011	Quadrats (half in burnt 1988 and half unburnt 1988)	Fixed Quadrats - (along boardwalk)	No. of larva, species found on, height of vegetation
Insects	Pitfalls	2009-2010	Pitfall traps	Pitfall traps taken monthly from the same site (along boardwalk)	No. of individual species in each.

Table 3.7 (continued) - Flora and fauna surveys undertaken at Cors Fochno.

Species	Code	Dates recorded	Description	Method of assessment	Recorded
Mammal	Watervole_a	2010-2011	Raft monitoring	Random recordings of activity on raft	Signs of presence or absence, main vegetation and water feature present recorded
	Watervole_b	2010-2011	Watervole signs	Random recording of watervole signs	GPS of watervole latrine, droppings, burrows, feed remains
Mammal	Otter_a	1996-2010	Otter spraints	Random recording of otter spraints	GPS of otter spraint
	Otter_b	2010 - 2011	Otter signs	Random recording of otter signs	GPS of otter path, prints, slide, spraint
Reptile	Grass snake	2011	Under tin	Random spotting and under tin	Count
Reptile	Adder	2011	Under tin	Random spotting and under tin	Count
Reptile	Slow worm	2011	Under tin	Random spotting and under tin	Count

Table 3.8 - Flora and fauna surveys undertaken at Cors Fochno.

Type	Species/category	Main feature	Remote sensing instrument	Main features
Vegetation	Active bog	Pool/hummock sequences	LiDAR and multi-temporal WV-2	Surface height descriptors (local minima and maxima) Classifications of bog pool species (e.g., Narthesium, Sphagnum)
	Sand dunes	Proportions of vegetation by type and open sand	LiDAR and multi-temporal WV-2	Digital Terrain Model Classifications of sand and vegetation
	Saltmarsh	Varying tidal regimes	LiDAR	Digital Terrain Model and varying proportions of mud and open water.
Birds	Lapwing (*Vanellus vanellus)*	Open, low grasslands, edges of pools.	Multi-temporal WV-2	Classifications of grasslands/water/aquatic vegetation; retrieved biophysical properties (e.g., sward height).
Insects	Small red damselfly *(Ceriagrion tenellum)*	Sheltered acidic pools with emergent or submerged sphagnum.	Multi-temporal WV-2	Classifications of open water and submerged/emergent vegetation
	Rosy marsh moth (*Ceonophilla subrosea*)	Presence of Bog myrtle (*Myrica gale*) growing in specific conditions.	Multi-temporal WV-2	Classifications of bog vegetation (including *Myrica gale*)
	Large Heath Butterfly (*Coenonympha tullia*)	Open, wet heathland and bog with Hare's-tail Cottongrass (*Eriophorum vaginatum*).	Multi-temporal WV-2	Classifications of bog vegetation (including *Eriophorum* species).
	Welsh Vernal Mining Bee (*Andrena hattorfiana*)	Sand dunes with Creeping Willow (*Salix repens*), semi-stabilized and south facing.	LiDAR and multi-temporal WV-2	Surface aspect, classifications of shrub vegetation and different proportions of sand and vegetation.
Mammals	Otters (*Lutra lutra*)	Fresh/brackish water, vegetation.	Multi-temporal WV-2	Classification of habitats, including open water and tidal/non-tidal areas.
	Water voles (*Arvicola amphibious*)	Fresh water/banks/vegetation cover	LiDAR and multi-temporal WV-2	Classification of open water and associated habitats, slope from LiDAR and standing/still water.
Reptiles	Grass Snake (*Natrix natrix*)	Heterogeneous habitats on margins of bog; proximity to water	Multi-temporal WV-2	Classifications of habitats and assessments of diversity and proximity to others. Proximity to fresh water (e.g., as defined using Water Band Index)
	Slow Worm (*Anguis fragilis*)	Moderate to dry conditions; lives in soil and vegetation.	Multi-temporal WV-2	Classification of aquatic and terrestrial vegetation and associated habitats.
	Adder (*Vipera berus*)	Grassy swards, open spaces with structured vegetation in proximity. Largely absent from disturbed habitats.	Multi-temporal WV-2	Classifications of grasslands; use of near infrared and red edge bands to indicate grass amount and biomass.

The **Lapwing** (*Vanellus vanellus*) requires different habitat structures for nesting (bare ground or very short turf) and chick rearing (meadow or marshy grassland with less dense patches for feeding, but cover for avoiding predators; edges of water). The suitability of such habitat can change rapidly (within days or weeks). For instance, fertilization can make the sward too lush and dense for chick survival. The species prefers to feed in an open diverse sward with abundant invertebrate food and struggles to survive in lush rapidly growing ‘improved’ swards where chicks cannot easily dry out and so succumb to hypothermia.

Reptiles include the **Grass Snake** (*Natrix natrix*), which prefers the more hetereogeneous habitats on the outer margin of the bog. The bog is too acidic for most amphibians (e.g., frogs), although palmate newt is plentiful in the ditches and small mammals are abundant in areas with abundant Purple Moor Grass (*Molinia caerulea*).

The **Adder** (*Vipera berus*) is prefers grassy swards with open spaces but where the structure is quite diverse for both hunting and concealment. The species is also often found on warm banks and on the edge of scrub or woodlands but generally avoids dense shade. They are largely absent from disturbed habitats and farmland, particularly where grazing and mowing occur. They feed on a wide range of small animals, including amphibians and nestling birds. Adder, and the other reptiles present at Cors Fochno use artificial refugia (corrugated tins), put in place to facilitate reptile recording.

The **Slow Worm** (*Anguis fragilis*) lives in the upper surface of soils and prefers relatively drier conditions and so is restricted to drier habitat patches within the wetland ecosystem.

The **Rosy Marsh Moth** (*Coenophila subrosea*) is rare and very local but occurs in the Cors Fochno mire where its principal food plant is Bog Myrtle (*Myrica gale*). The nocturnally active caterpillar may require the foodplant to be growing in specific conditions where the ground layer affords a suitable structure and micro-climate for diurnal resting. However, the caterpillar is also thought to use an alternative food plant during its early stages of growth before *Myrica* buds and foliage are available. To add to the complexity Rosy marsh moth, is able to survive at some locations where *Myrica* is absent, using an alternative food plant. This is the case at Cors Caron (c40km from Cors Fochno), where it feeds mainly principally on crowberry *Empetrum nigrum.*

The **Small Red Damselfly** (*Ceriagrion tenellum*)is confined largely to acidic pools with abundant submerged or emergent sphagnum. The insect is a relatively weak flier and so prefers sheltered conditions. The species is nationally scarce.

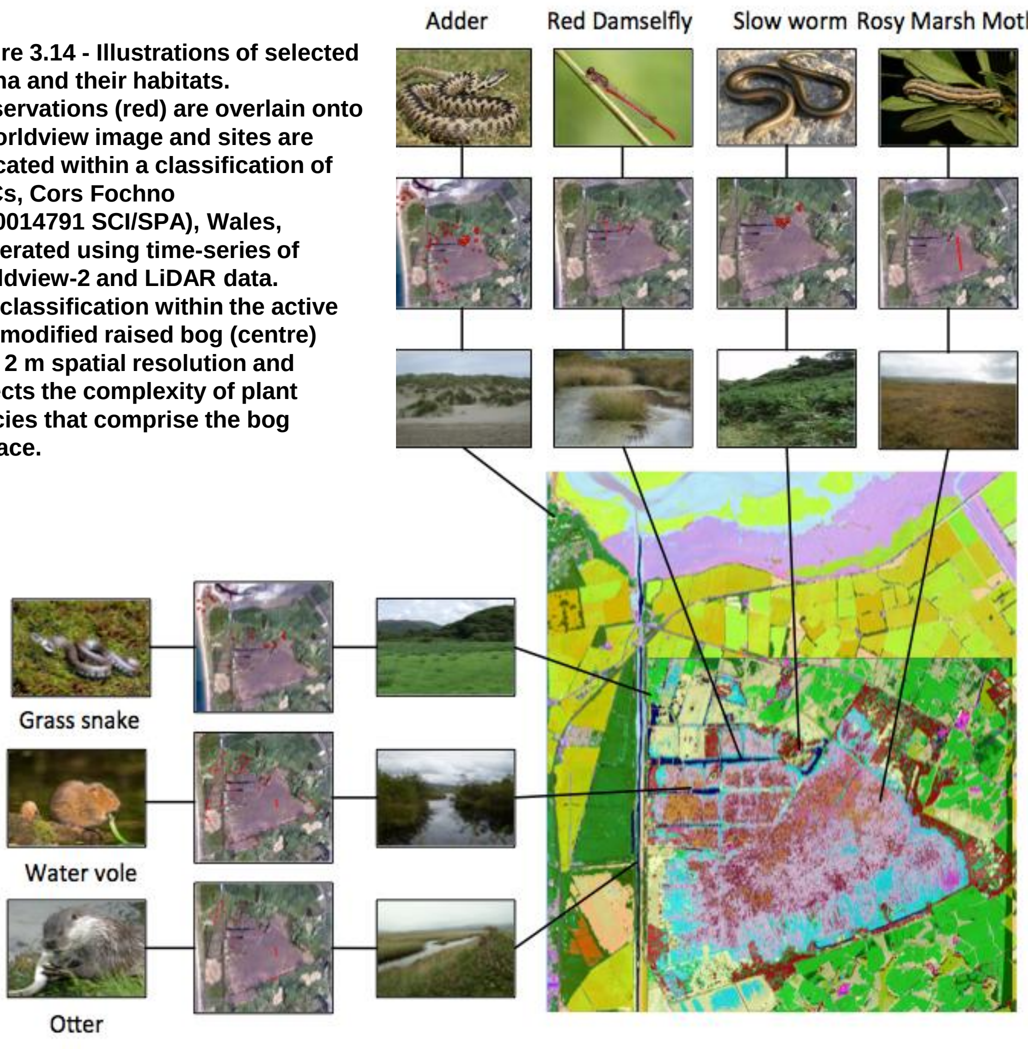
The **Large Heath Butterfly** (*Coenonympha tullia*) is found on open wet heathland and bog and, at Cors Fochno, it is at the southernmost end of its range. The species prefers relatively cool conditions with the adults flying mainly during sunny periods. The larvae feed on **Hare's-tail Cottongrass** (*Eriophorum vaginatum*).

The Welsh vernal **mining Bee** (Colletes cunicularius) is found on the sand dunes. This species collects pollen from **Creeping Willow** (*Salix repens*), which flowers in early spring and is also confined largely to sand dunes. The bee prefers to burrow in sandy banks that are semi-stablilised and south facing. The pollen collected is stored in the burrows and used to feed the larvae.

Maps showing the observed distribution of several of the species listed above are provided in Figure 3.14 together with illustrations of their habitats and a classification of GHCs. The information able to be extracted from remote sensing and used for landscape pattern analysis is also indicated in Table 3.8. In the majority of cases,

multi-temporal Worldview-2 data are required to generate the classifications of key habitats required for the species listed above. In the detailed GHC classification for Cors Fochno from Worldview and LiDAR, the central bog supports a mix of shrubby chaemophytes (SCH; e.g., *Myrica* (purple) and *Calluna*-dominated (dark pink), grass species (CHE; e.g., *Eriophorum*-dominated (light pink) and *Molinea/Phragmites* dominated (blue) as well as wetter pools (with *Sphagnum* (HEL; white) and *Cladonia* (CRY; orange) species). LiDAR data are required primarily where information on the terrain surface is required, as in the case of saltmarshes or the sand dunes used by mining bees. Derived indices from these data are also important in the classification of habitats but also for describing these in terms of, for example, relative wetness (e.g., using water band indices), productivity (based on the NDVI) and sward height (red edge), which are variables of importance for many flora and fauna species. Other indices that may prove useful include those that measure the amount of senescent or dead material (e.g., the Plant Senescence Reflectance Index or PSRI). In addition, the spatial arrangement of habitats and the heterogeneity (or otherwise) of the landscape can also be considered. Hence, measures such as proximity, adjacency, relative area and enclosure (to differentiate flowing and standing water for example) become important. A summary of the main features that can be used for understanding and determining the distribution of faunal species is given in Table 16.

Figure 3.14 - Illustrations of selected fauna and their habitats. Observations (red) are overlain onto A worldview image and sites are indicated within a classification of GHCs, Cors Fochno (UK0014791 SCI/SPA), Wales, generated using time-series of Worldview-2 and LiDAR data. The classification within the active and modified raised bog (centre) is at 2 m spatial resolution and reflects the complexity of plant species that comprise the bog surface.



With consideration given to the modelling of fauna and flora distributions, diversity and abundance within Cors Fochno, remote sensing is, in summary, considered to provide information on the extent of habitat and vegetation class patches, the structure and heterogeneity of vegetation, connectivity of patches and the linear extent of patch boundaries and transitions between these. Slope, aspect and other topographic/geographic features can also be quantified at fine spatial resolution. Some consideration does, however, need to be given to the following:

a) High species diversity does not necessarily equate with high habitat quality. Some habitats (e.g., raised bogs) are naturally species poor (although a high percentage of species may be rare or uncommon). Degradation of raised bogs (and other habitats) may indeed increase species diversity by allowing generalist species to move in.

b) Knowledge of species autecology is crucial in attempting to link any animal population with vegetation categories. It is also important to recognize that key aspects of a species behavioural ecology may differ across their range.

c) Many species exist as metapopulations with interdependent colonies distributed across a landscape. Thus, individual and collective habitat patch extent and their juxtaposition are important for survival.

d) Many species require close juxtaposition of more than one habitat. The rationale behind landscape modelling (a step forward habitat modelling) is that of trying to investigate the relations between the target group species and the landscape attributes in relation to “mosaic habitat” requirements.

e) A very wide range of physical and ecological factors often act in combination to determine habitat suitability for any given animal species.

f) A lack of records for a species in a habitat that appears suitable does not mean the species is absent. However, through ecological niche modelling, it may be possible to obtain information on the existence of conditions that might indicate a relative likelihood of presence. Actual presence does, however, require ground measurement. Many invertebrates and some vertebrates are very difficult to find due to their habits and/or low population density. Expert knowledge and sampling skill is required to find many scarce species. This is often not available or too costly.

g) A species can be absent from an area that perfectly fits its habitat requirements for geographic or other reasons.

In support of the BIOSOS outcomes, additional biodiversity datasets are being collected during 2013 across Cors Fochno. Focus will be on those species that are associated with particular habitats (e.g., indicators) and which are likely to change in distribution with changes in the habitat. Preliminary analysis undertaken in late 2012 have focused on the active raised bog and closed perennial grasslands dominated by *Molinea* and *Phragmites* species, which are encroaching in some areas. In the case of the raised bog, a higher diversity of fauna (particularly spiders) were observed in pitfall traps compared to the closed grasslands, although the latter favoured small mammals because of the tussocky nature of the habitat. This example illustrated how the VHR maps generated from the Worldview and LiDAR data could be used as input to species distribution models and landscape pattern analyses.

3.4.3 Assessing habitat quality for native oak forests from plant community structure in the PT2 site

3.4.3.1 Focal process and habitat of interest

In the PT2-1 Area of Interest (AoI) ("Castro Laboreiro"; Figure 3.15), the focal habitat type (i.e. LE type) will be native oak forests, corresponding to Annex I habitat 9230, and the process/pressure to address will be **land use dynamics** and the corresponding changes in forest habitat quality. Farmland and pastureland in mountains and other marginal areas have been suffering abandonment in recent decades across large areas of Southern Europe. This has resulted in gradual (though sometimes dramatic) changes in landscape structure, due to scrub and forest encroachment, as well as increased fire risk (Lomba et al. 2012; Proença et al. 2012). Such changes may be expected to induce important changes in habitat quantity, quality and distribution.

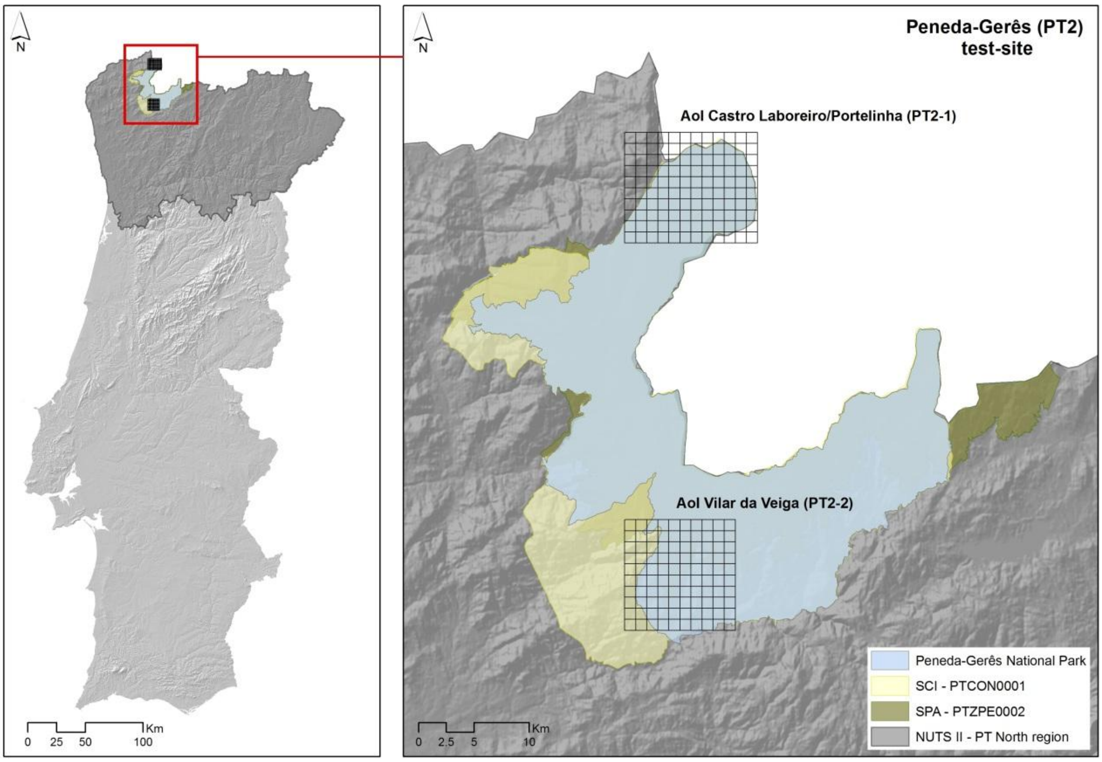


Figure 3.15 - The two AoI's in the PT2 test site of BIO_SOS ("Peneda-Gerês"), with the PT2-1 AoI at the northernmost end of the site.

In the case of **native oak forests**, the underlying rationale is that an increase of patch area, resulting from farmland abandonment and the coalescence of isolated forest patches, will promote habitat quality, translated e.g. in more complex vegetation structure and in the expansion of forest specialist species. Conversely, fragmentation of forest areas by local expansion of farming or pastoral use will promote modifications of vegetation structure as well as the expansion of generalist species, thereby decreasing habitat quality.

In the wider context of the habitat modelling framework described in this report, this PT2 case study will address the following **research questions**: (1) How is plant

community structure (here a proxy for habitat quality) influenced by patch and matrix features? Are the effects of landscape context on community structure (and on habitat quality) more evident in smaller patches of the focal habitat type? and (2) How is community structure modelling and detection influenced by extent and grain size? Is there a significant added-value for community level modelling in the use of VHR habitat maps?

3.4.3.2 Background and rationale – plant community structure as a proxy of habitat quality

In the PT2-1 Area of Interest (AoI), the focal taxonomic group will be **vascular plants**. Among the many forest patch characteristics, the quality of habitat patches (e.g. expressed as area or isolation) has been highlighted as an important determinant of species distribution in the landscape (e.g. Dupré and Ehrlén, 2002; Fleishman et al., 2000). In such a framework, the ecological rationale underlying the study is that measures of plant diversity (structural and functional) are potential proxies for estimation of "habitat quality" (Liira and Sepp, 2009). In fact, several features of forest structure and species composition have been reported in previous research as expressing forest habitat quality (e.g. see Liira and Sepp, 2009; Lomba et al., 2011).

Here, the workflow will include the **collection of floristic data** from forest patches during field surveys, and then the use of functional and ecological classifications to qualify the recorded plant species. Additionally, characteristics expressing structure of forest vegetation will be collected (e.g. species-area relationships, or SARs; Average value of tree diameter at breast height, DBH; Tree density). Ultimately, plant species and floristic composition data from surveyed patches will be related to forest habitat quality. The general proposed workflow is represented in Figure 3.16.

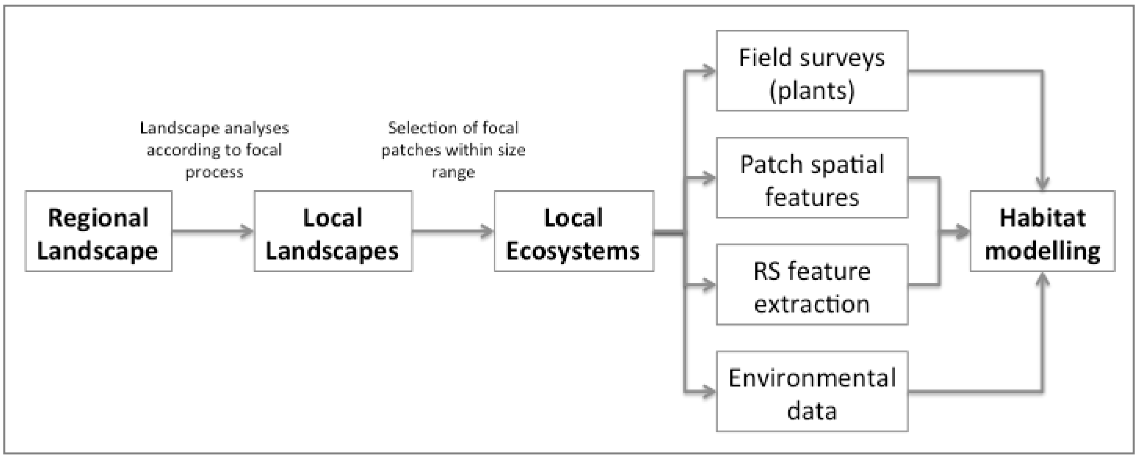


Figure 3.16 - Proposed workflow for habitat quality modelling in the PT2 site of BIO_SOS. See text for a detailed description of each stage.

Plant community structure will be assessed from standard species diversity measures as well as species-area relationships (SARs) (e.g. see Pereira and Daily, 2006; Wang et al., 2011; Lomba et al., accepted). The structural taxonomic approach to express diversity and community assembly will be complemented with distinct species classifications (functional and ecological) as described in BIO_SOS D4.3 and elsewhere (e.g. Lomba et al., 2011; Scheiner, 2012). Emphasis will also be put on the distribution an abundance of plant species of conservation interest that have forests and/or forest edges as their primary habitats.
Patch-level diversity is known to have a close relationship with the composition and structure of the surrounding landscape, but also indirectly through spatiotemporal

patterns of land use. Geological, geomorphological and climatic variables have also been successfully used in regional scale biodiversity research; however they are often not sufficient for understanding local scale diversity patterns. Other factors have been considered as determinants of the species pool colonising forest fragments, such as the stand structural properties (e.g. forest structure, canopy cover, Estevan et al., 2007), disturbance regime and management, and other spatial attributes (e.g. patch area, Guirado et al., 2008; Yamaura et al., 2008; shape, Saura and Cabral, 2004). **Structural complexity of vegetation** can also be used to address levels of plant diversity in the case of forests (Staudhammer and LeMay, 2001; Neumann and Starlinger, 2001; Estevan et al., 2007; Lomba et al., 2011). Here, stand structure indices (e.g. density and diameter of canopy trees; diameter at breast height, DBH) will be related to plant diversity levels, as they express distinct management regimes. Such indices will be assessed through standard forest inventory methods (e.g. Staudhammer and LeMay, 2001; Neumann and Starlinger, 2001; Lomba et al. 2011) and by using plant life forms to define vegetation strata (e.g. Bunce et al. 2008).

In summary, the following general features of plant community structure and of vegetation structure will be addressed as **proxies of habitat quality**:

(1) total species richness and species-area relationships;
(2) species richness and diversity for selected functional groups (e.g. herbs vs. shrubs, ruderals vs. stress-tolerants);
(3) species richness and diversity for selected ecological groups (e.g. forest specialists);
(4) distribution and abundance of endangered and/or protected species;
(5) density and diameter of canopy trees; and
(6) height and cover of vegetation strata.

3.4.3.3 Levels and spatial extents for field sampling and modelling in the PT2-1 AoI

In the PT2-1 AoI, a 100km^2 area considered as the **Regional Landscape** (RL) for habitat quality modelling, two complementary types of landscape mosaics can be found: farmland in concave areas, and extensive grazing land in convex areas. Forest encroachment due to abandonment may occur in both types of landscapes, however this study of habitat quality will only focus on the former. Therefore, Local Landscapes (LLs) in this AoI will consist of mountain farmland-woodland mosaics, a patterned landscape mixture of croplands, meadows, woodland and tall scrub, in the neighbourhood of rural villages (Figure 3.17).

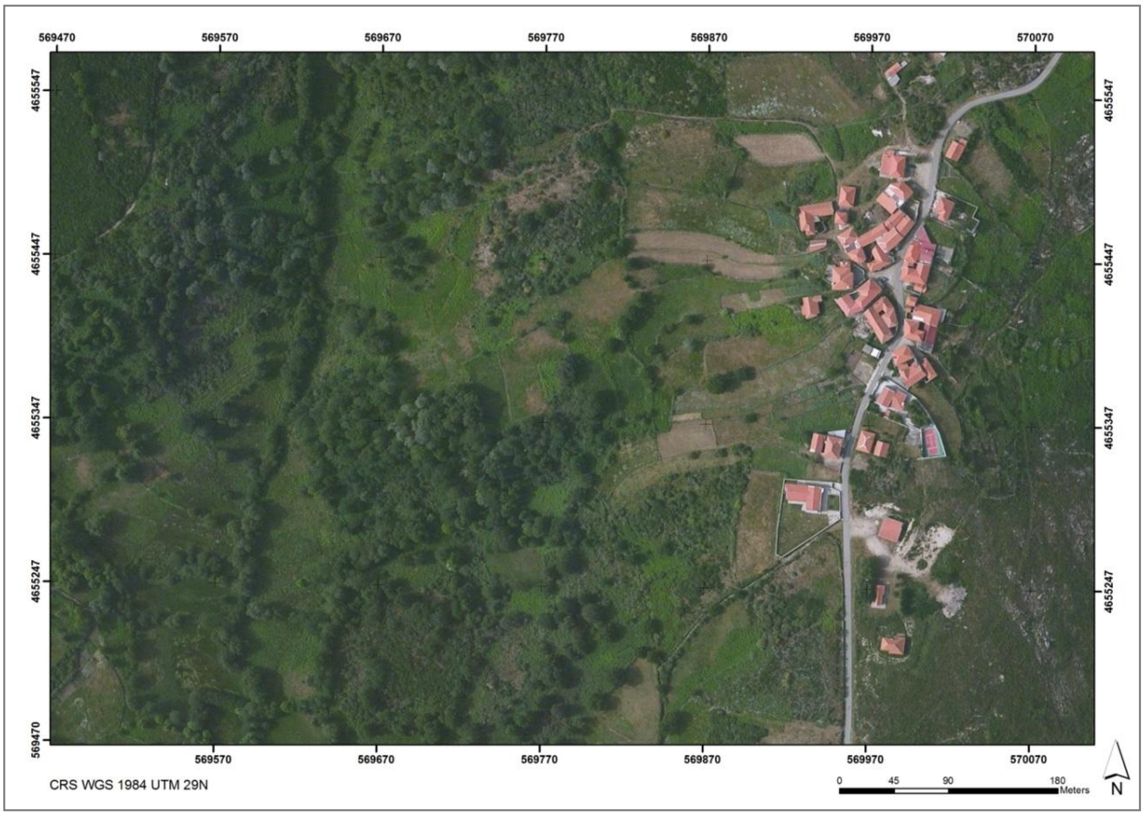


Figure 3.17 - Aerial photograph of landscape mosaic in the PT2-1 area of interest, suitable for habitat quality modelling in the PT2 test site. A meadow, woodland and scrub mosaic can be seen towards the left side of the landscape, whereas crop fields, roads and other ruderal areas are more common around the village.

The selection of suitable landscape mosaics for habitat quality modelling will be based on a recent land cover map with a scale of 1:10 000 (Figure 3.18). Eligible **Local Landscapes** will be identified based on the following general criteria: (i) predominance of farmland areas or farmland/woodland mosaic areas; and (ii) presence of a minimum of two patches of the habitat type of interest within the focal size range (see below).

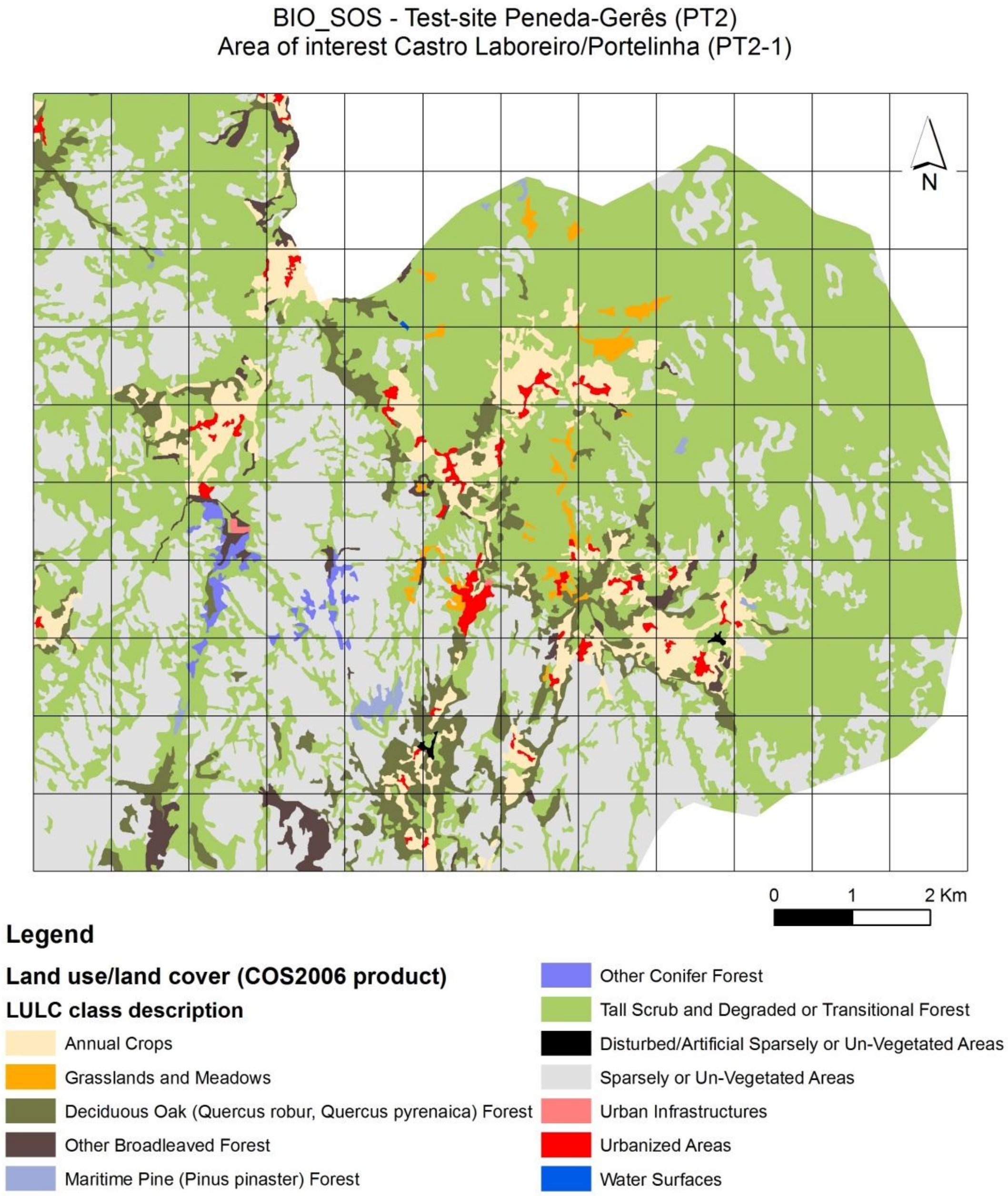


Figure 3.18 - Land cover (1:10 000 scale) for the RL in the PT2-1 AoI.

In order to capture areas where agricultural use is dominant or at least well represented in the landscape, the **percentage cover of arable land** per 1km^2 Local Landscape (i.e. grid cell; see Lomba et al., 2010) was determined (Figure 3.18). More specifically, for each per grid cell, the percentages of annual crops and other agricultural land uses were estimated from the land cover map (Figure 3.19) and merged into a single value. Other metrics expressing composition (e.g. class richness, PR; Leitão et al., 2006) and configuration (e.g. mean patch size, AREA_MN; shape, SHAPE; Leitão et al., 2006) will be determined for complementary analyses. This will result in the selection of the set of eligible Local Landscapes in the context of the PT2-1 Regional Landscape.

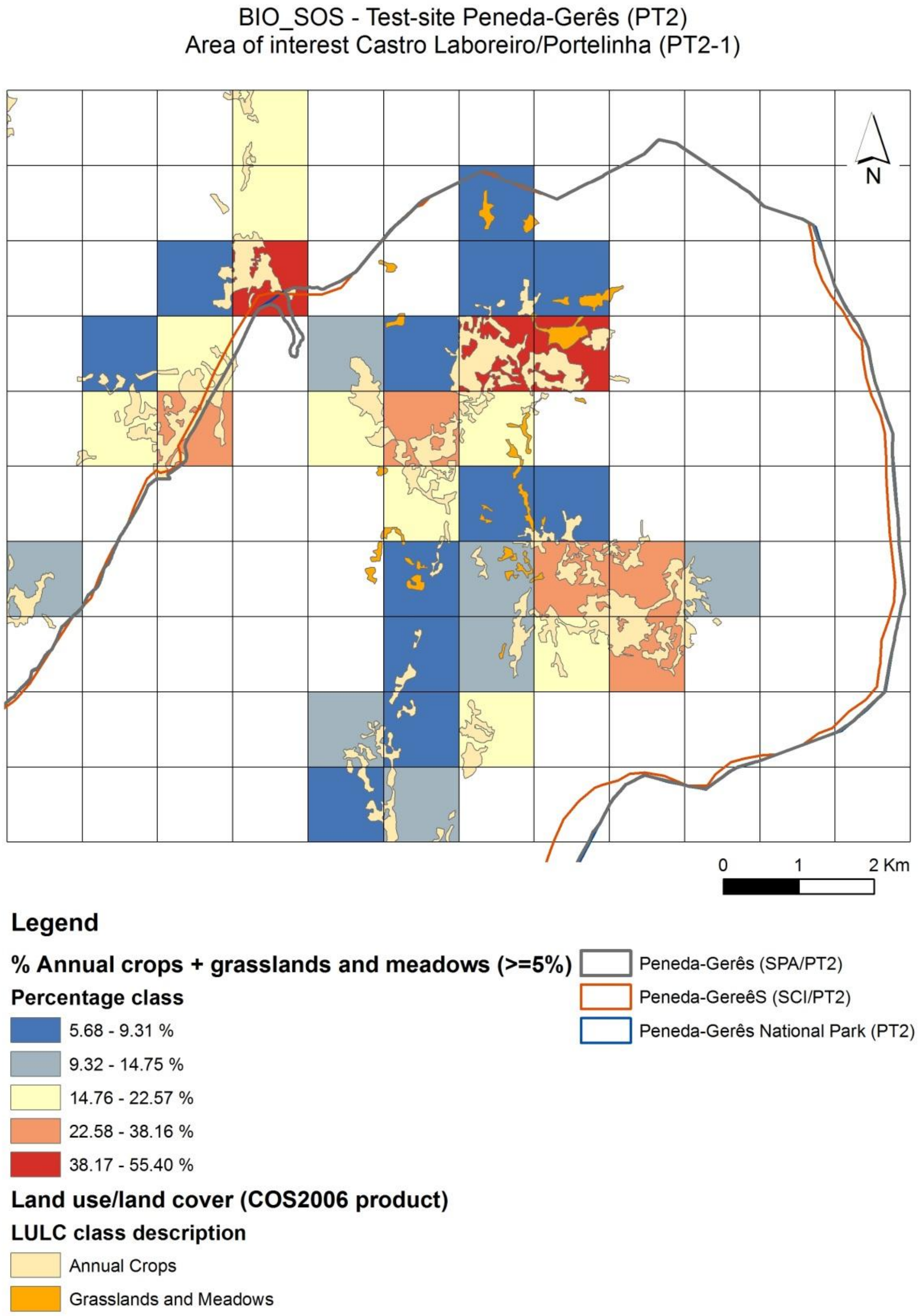


Figure 3.19-. Percentage cover of areas with agricultural use per Local Landscape in the PT2-1 Regional Landscape.

3.4.3.4 Landscape analysis and sampling strategy

As mentioned before, the Regional Landscape (RL) will be the AoI of Castro Laboreiro in the PT2 test site (PT2-1), with a total extent of 100 km^2. Eligible Local Landscapes (LL) will be farmland-woodland mosaics with 1km^2 total area. The **final set of Local Landscapes** will be selected from the eligible set of LLs in the RL grid as described below.

A set of **landscape pattern metrics** at the LL-level (considering both composition and spatial configuration) will be computed based on a preliminary selection supported by the relevant literature (e.g., Yu and Ng 2006, Gellrich et al. 2007, Geri et al. 2010, Azevedo et al. 2011) as well as by BIO_SOS previous work and guidelines. These metrics should be able to translate the effects of agricultural land abandonment and associated land use dynamics in several ecologically relevant aspects of these changing rural landscapes, such as contagion/diversity, dominance, fragmentation/connectivity, interspersion, or patch size variability (Cushman et al. 2008) (Figure 3.20). Change analysis, using transition rates between two different dates, will be considered to map alterations in the selected landscape pattern metrics (Pôças et al. 2011), when adequate land cover data and satellite imagery will become available for this AoI. Geomorphological and topographic heterogeneity indices may also be considered to segment different landscape mosaic types.

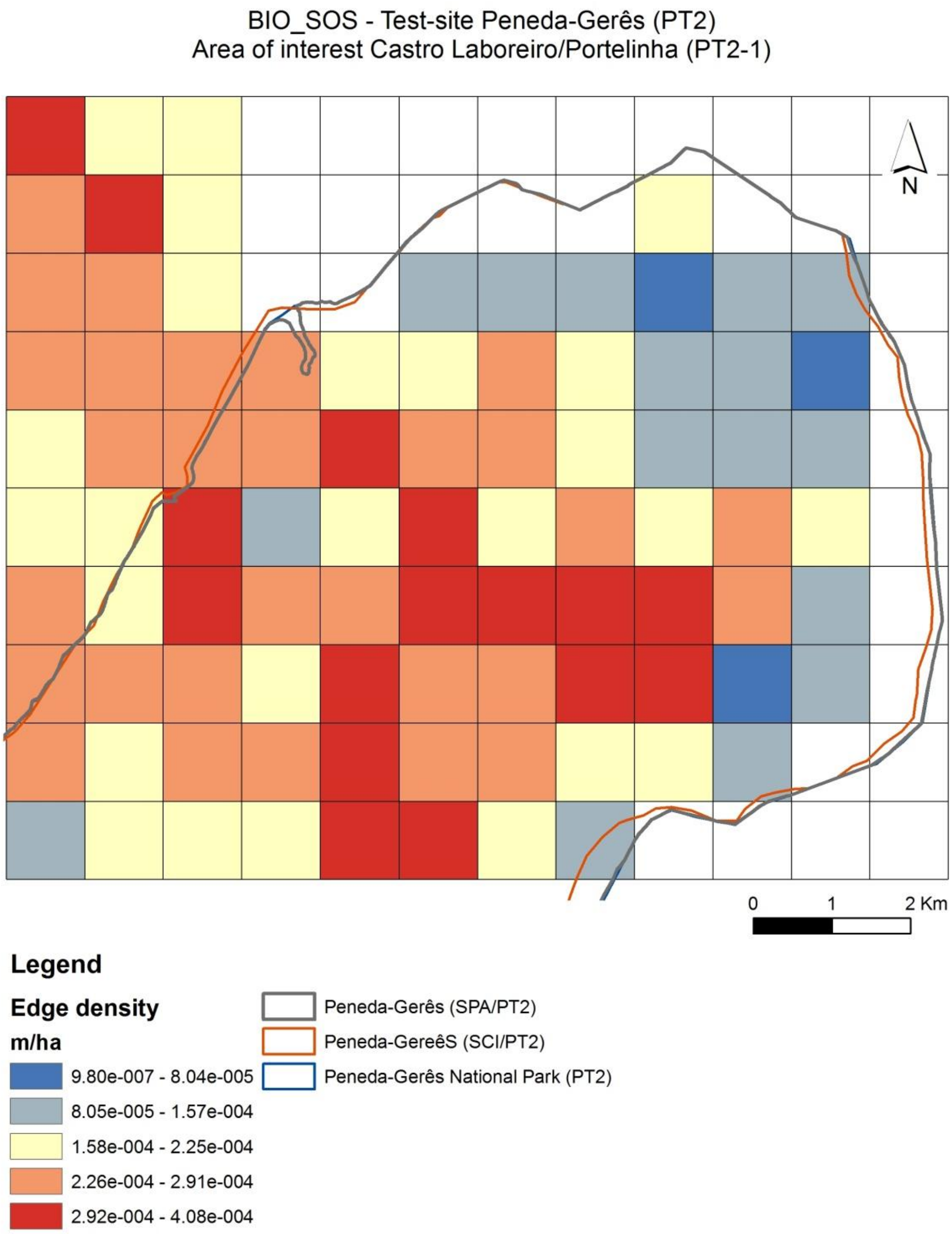


Figure 3.20 -Edge density, an example of a landscape metric related to land use dynamics, here computed for each LL of the RL.

Based on this preliminary analysis of landscape structure at the LL-level, cluster analysis techniques will be used to produce a stratification sensitive to land use dynamics (related to farmland abandonment) gradients. **Stratified random sampling** (Gruijter et al. 2006), with proportional-to-area allocation, will then be used to select the final set of Local Landscapes.

3.4.3.5 Field methodology and data processing

Two to three forest patches (i.e. **Local Ecosystems**) will be randomly selected within each of the selected Local Landscapes. Selected patches should range from 0.3 to 3 hectares in surface area. This will allow considering variation in patch size while ensuring that continuous forest areas are not considered at this stage of habitat quality modelling. In case of total impossibility for surveying any of the randomly selected forest patches (e.g. in the case of clearing, land use conversion), the nearest forest patch within the defined area range will be surveyed.

Field surveys will consist in the collection of data on community composition and structure for vascular plants. In each forest patch, the presence and abundance of all vascular plant species will be determined for the whole patch. To do so, in each patch exhaustive surveys will be performed starting with a $10m^2$ square plot at the centroid of the patch, followed by gradual nested increases of plot area (10^2m, 10^3m and eventually 10^4m) until the whole extent of the patch is reached, assuring a standardised approach across patches and a complete survey of each individual patch (see Lomba et al., 2011, for more details).

The abundance of individual plant species will be estimated in each plot area using the **Domin cover scale**, by visually determining the area of the ground per patch occupied by the aboveground parts of the plants when seen from above (Kent and Coker, 1992). As the two first classes of Domin cover scale express number of individuals, they will be aggregated, according to the following simplified 9-class scale: (1) <1%, (2) 1–4%, (3) 4–10%, (4) 11–25%, (5) 26–33%, (6) 34–50%, (7) 51–75%, (8) 76–90%, and (9) 91–100%. For statistical analyses, the mean value of each class will be considered and used. Species nomenclature to be followed will be that of “Flora Iberica” (Castroviejo et al., 1986–2010).

Data on **vegetation structure** will be collected according to standard methodologies (see BIO_SOS D4.3 and Lomba et al., 2011). For the whole forest patch, the percentage cover of each vegetation stratum will be determined. Five strata will be considered: dwarf herbaceous (height < 0.5 m), tall herbaceous (0.5–1 m), dwarf shrubs (between 0.5 and 2 m height), tall shrubs (2–8 m), and trees (>8 m). Values for the percentage cover of each vegetation stratum will be determined by visual estimation. The total number of trees and the diameters of all individual plants belonging to tree species (diameter at breast height, DBH) (Ferreira et al., 2005), and the number of trees within the $100m^2$ plot, will also be determined to estimate tree size and density.

Local Ecosystem qualifiers will also be recorded from each surveyed patch following the field guidelines described by Bunce et al. (2011). Environmental qualifiers related to key topographic and soils attributes, and management qualifiers, will be among those to be collected.

Field surveys will be performed in **late spring**, which corresponds to the local vegetation flush or peak of plant activity.

3.4.3.6 RS data, feature selection, and scale

In order to generate **patch-level attributes** as covariates for developing correlative models and/or other statistical analysis techniques, two different types of features will be considered, namely: (i) remote sensing derived features, and (ii) patch-level pattern metrics. These analyses will use RS imagery and GHC/LCCS maps produced within BIO_SOS. Imagery from at least two sensors will be used: Worldview-2 (pixel size 2m) and Landsat (pixel size 30m) in order to test the effects of grain or LC mapping and or habitat quality modelling. Images will be acquired for the peak of season (flush), i.e. late spring, and for an earlier transition date (i.e. late autumn or early spring).

Remote sensing features will include vegetation indices such as NDVI or EVI (considering different aggregation statistics at patch-level, e.g., mean, standard-deviation) (Pettorelli et al., 2005, Levanoni et al. 2011) as well as texture related features that are able to capture different heterogeneity dimensions. This may include textural features such as histogram, absolute-gradient, run-length and co-occurrence matrices (Szczypiński et al., 2009).

Patch-level landscape pattern analyses will be conducted for each individual surveyed patch (LE1), including its spatial context. For LE1 contextual analyses, a set of concentric buffers will be established at 10^4, 10^5 and 10^6 m^2. Standard landscape metrics will be used and complemented with focal patch metrics (Torres et al. 2010, suppl. mat.; Figure 3.21) for LE1 analyses. Focal patch metrics will be computed through a new R package named Landscape Pattern Analysis Tools (LSpAT) (Gonçalves, 2010), which uses the Object Relational Database Management System, PostgreSQL, and its module PostGIS (which adds support for geographic objects), to implement spatial analyses procedures and calculations related to focal patch metrics based on vector land use/cover data. Internal heterogeneity analyses (LE2) will also be conducted for each surveyed patch based on RS features derived from VHR imagery.

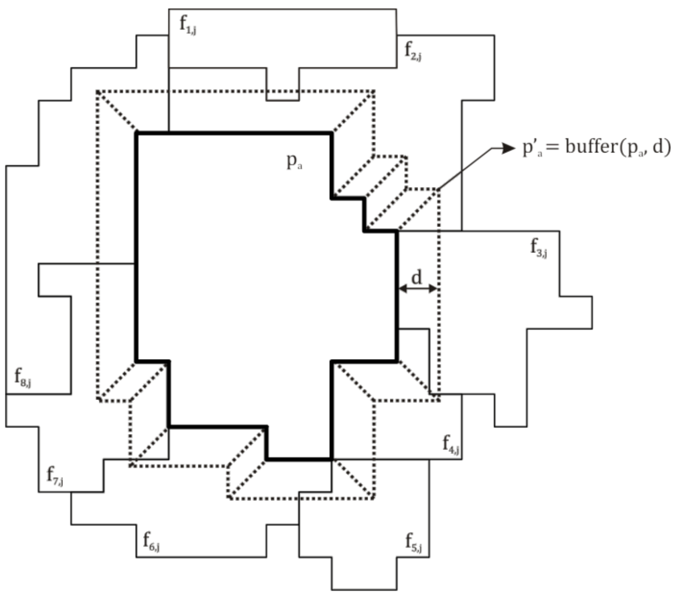


Figure 3.21 -The immediate landscape context of a given focal patch, which is analysed to compute focal patch metrics (adapted from Torres et al. 2010, suppl. mat.)

3.4.3.7 Habitat quality modelling

Variations in forest spatial attributes, vegetation structure and management, as well as **plant community structure** will first be examined using univariate statistics. Total species richness, native species richness, alien species richness, woody species richness, herbaceous species richness, and forest specialist species richness (see Lomba et al., 2011) will be computed from species lists. Similar measures will be obtained for functional groups or life strategies. Species-area relationships will also be obtained for each patch from the nested sampling of species richness. Statistical analyses of community structure data against landscape metrics and environmental qualifiers will be made under the common modelling framework described in previous sections of this deliverable.

3.4.4 Habitat modelling for invasive species in India

In the Indian sites IN1 and IN2, invasive species represent a major anthropic pressure on native habitats. Within the broader framework of habitat modelling provided by this deliverable, specific factors important for invasive species to establish themselves include favourable microclimate, availability of resources such as light, water, and soil nutrients, ecological factors such as propagule pressure, and existence of surrounding habitat types that are conducive to their growth

Site IN1, the Biligiri Rangaswamy Temple Wildlife Sanctuary is an area of high biodiversity lying in the Western Ghat hill ranges of the southern Indian peninsula. This 540 km^2 sanctuary is unique in its heterogeneity of physiographic forms, the climatic regime it experiences, as well as the diverse endemic flora and fauna it supports. In the past decade, studies have reported the occurrence (Murali and Setty, 2001), and subsequently a ten-fold increase (Sundaram, 2011), of invasive alien plant species, with special emphasis on *Lantana camara* (L.). While these studies provide important baseline information about invasion of *L. camara* in this landscape, no detailed study has undertaken work on understanding the underlying mechanisms which may be encouraging the growth and spread of *L. camara*. Habitat modelling can be explored to assess its potential to predict areas where factors are more conducive to its spread and identify potential future occurrences.

Consequently, field data collection was initiated in IN1 during February and March 2012, with further sampling planned in later months, to understand the relationship between the presence of *L. camara* with its environmental and ecological parameters.

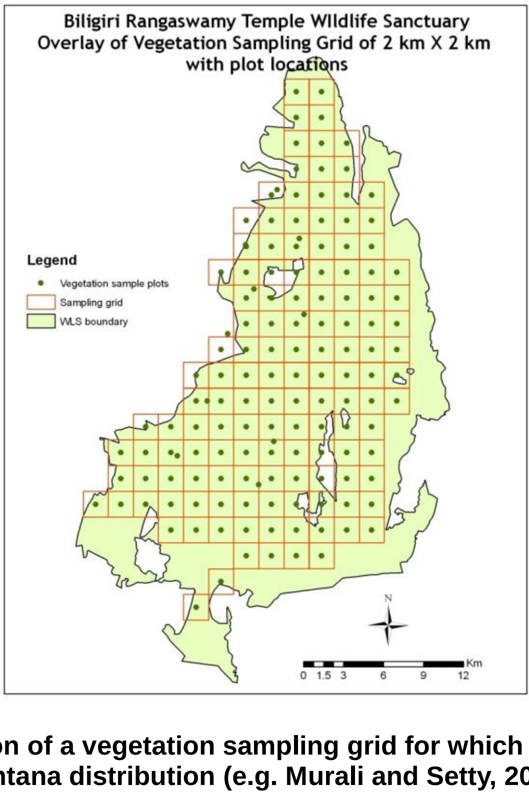


Figure 3.22. The location of a vegetation sampling grid for which previous long term data is available on Lantana distribution (e.g. Murali and Setty, 2001; Sundaram, 2011).

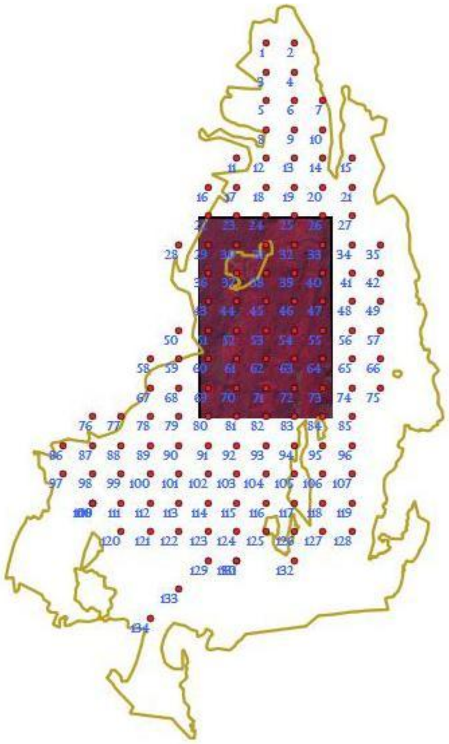


Figure 3.23. The same site and grid, with the location of a GeoEye image acquired by BIO_SOS overlaid.

As can be seen from Figure 3.22, the site IN1 is very large, but for the purpose of prospective landscape modelling, we focused on the area shown in red in Figure 3.23, for which we have coverage provided by a VHR Quickbird image of 7th January 2011.

Within this region, for a set of about 35 plots, we collected data including identification and enumeration of all plant individuals > 1cm gbh; measurements of canopy closure at 5 locations within plot using spherical densitometer; height of invasive species *L.camara* and *Chromolaena odorata* ((L.) King and H.E.Robins) and the closest tree species by ocular estimation; as well as information on vegetation cover type and habitat details. This dataset is being processed for analysis. In order to get comprehensive information on vegetation habitat characteristics from the EO images, it is necessary to acquire a second VHR image from the leaf-off season, to compare with the image of 7th January when the deciduous trees in this location have trees. This will be very important as the Lantana leaf density also changes considerably across seasons, and multi-temporal imagery within the same year can be very important for relating to the field data. Unfortunately, imagery for the previous years have too much cloud cover to be of use for our analysis, and we are therefore waiting for images acquired in 2012 to assess their use for this analysis.

Site IN2 lies in the Netravali Wildlife Sanctuary in the Indian state of Goa. Within this site, we have created a 2 km grid for vegetation data collection similar to that developed for site IN1, as depicted in Figure 3.24.

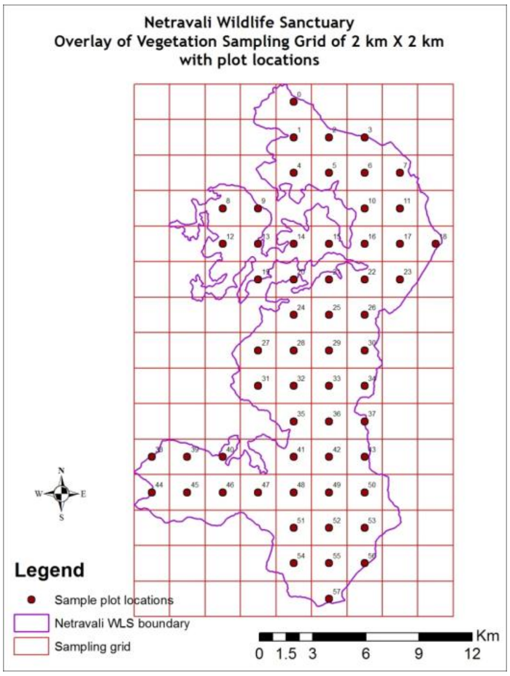


Figure 3.24 The location of a vegetation sampling grid for IN2 site.

Sampling for field data collection has just begun, and is ongoing at the time of writing this deliverable. Vegetation parameters will be collected similar to those collected for site IN1, including information on invasive species, canopy closure and habitat information.

3.5 Concluding remarks

Ultimately, habitat and landscape modelling studies in BIO_SOS are largely grounded on the possibility of **extracting estimates of indicators** on species, biodiversity, habitat quality and landscape features related to pressures from the future EODHaM system. Therefore, the study design as well as the analyses will be conducted as to address this possibility and identify possible caveats.

Besides contributing to further clarify landscape-biodiversity and community structure-habitat quality relations, the modelling framework described and instantiated in this deliverable is expected to demonstrate that RS data and habitat maps generated from EODHaM can be used to **derive robust estimates of key habitat and biodiversity indicators**. Together with the core LC/LU and habitat mapping modules, estimates are intended from EODHaM for indicators related to: (1) range and area of focal habitat types, (2) connectivity/fragmentation of those habitat types, (3) quality/integrity of habitat patches, and (4) distribution and abundance of rare endangered species having the focal habitat type as their primary habitat.

Such estimates can be most helpful to support end-users in meeting their commitments under **international reporting** initiatives, e.g. SEBI or Article 17 of the EU Habitats Directive. In addition these would allow for **consistent multi-annual monitoring** of protected areas and their surroundings in the Mediterranean, Northern Europe and other regions including India, with these sites located within different climate zones of the world.

The IT3 analysis are expected to provide an empirical case study for testing interdependence of habitat effects in order to identify key causal effects of habitat fragmentation which can then be addressed by means of management measures.

The PT2-1 case study is mainly expected to contribute for robust assessment of regional habitat extent for several Annex I habitat types as well as of habitat quality for habitat types 9230 (native oak forests).

The UK investigation is expected to demonstrate the VHR maps generated from the Worldview and LiDAR data could be used to obtain spatial information on the structure, and functioning of landscapes to be used as input to species distribution models and landscape pattern analyses.

The IN case study is expected to assess the potential of RS data to investigate the underlying (environmental or ecological) mechanisms which may be encouraging the growth and the spread spread of invasive species and identify potential future occurrences to put anticipatory controlling measures.

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Abbreviations and Acronyms

BIO_SOS = Biodiversity Multisource Monitoring System: from Space TO Species
CBD-SEBI = Convention on Biological Diversity - Streamlining European Biodiversity Indicators
CLC= CORINE Land Cover
CON = Contrast
CORINE = Coordinate Information on the Environment
D=Deliverable
EEA = European Environmental Agency
EO = Earth Observation
EODHaM = EO Data for Habitat Monitoring
EVI = Enhanced Vegetation Index
FG = Fragmentation Geometry
FOEN = (Swiss) Federal Office for the Environment
GHC = General Habitat Category
GLCM = Grey Level Co-occurence Matrix
IDM = Inverse Difference Moment
IMC = Information Measure of Correlation
LANDSAT = Land Satellite
LC/LU = Land Cover / Land Use
LCCS = Land Cover Classification System
LiDAR = Light Detection And Ranging
LPA = Landscape pattern analysis
LPI = Landscape pattern indices
MESH = Effective mesh size
MSPA= Morphological spatial pattern analysis
MSW = Moving Split Window
N2K = Natura 2000
NDVI = Normalised Vegetation Index
NIR = Near Infra Red
NP = National Park
PCA = Principal Component Analysis
PLAND = Percentage of landscape
RS = Remote Sensing
SAR = Synthetic Aperture Radar
SCI=Site of Community Interest
SD = Standard Deviation
SED = Squared Euclidean Distance
SPA= Special Protected Area
VHR= Very High Resolution
WP = Work Package