The evaluation of a planning tool through the landscape ecology concepts and methods

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Abstract

Landscape ecology represents a theoretical and empirical support of the spatial planning, providing parameters such as heterogeneity, connectivity and fragmentation. The aim of this study was (i) using these parameters to evaluate the choices of a real planning tool to protect the biodiversity, (ii) evaluating the applicability limits of concepts and methods used. This was achieved by analysing the selected spatial indices and their dependency scale, and by the comparison of these results with regard to spatial biotic parameters estimations (birds and mammals). The study confirmed the scale's effect on the indices, unstable at the adopted resolution for extensions up to 6-7000 meters. The selected indices permitted to appreciate the low effectiveness of the real planning tool in improving conservation of biodiversity. The paper suggests that (i) empirical studies and predictive knowledge at different scales are urgent in this fields, (ii) to preserve biodiversity, the choices of the planning scale should primarily comply with species spatial needs.

I. INTRODUCTION

Planning is an instrument for the sustainable development of landscapes (Franco, 2002; Jongman, 2002; Madsen, 2002) and in the last decades landscape ecology has supplied a support to spatial planning (Forman, 1995) providing some parameters for estimating the ecological features of landscapes. With particular reference to biodiversity, these parameters are: heterogeneity, connectivity and fragmentation.

Landscape heterogeneity variation can affect species interactions, adaptations and distribution (Dramstad et al., 2001; Manson et. al., 1999). It can modify the most vagile taxas' biodiversity (Atauri & de Lucio, 2001; Farina, 1997; Preiss, 1997; Jonsen

& Fahring, 1997; Naugle et al., 1999; Pino et al., 2000) as a function of the exploratory/perceptive levels of the considered populations. There is not a single method to estimate this parameter.

Until up today landscape “connectivity” cannot be measured in a simple and general way (D’Eon et al., 2002; Tishendorf & Fahring, 2000), but the connection rate of the “paranatural” ecosystems in a rural landscape can be an index of some of the potential populations (plants, birds, and small mammals) dispersal ability (Franco, 2000; Barr & Petit, 2001).

A landscape fragmentation process (Forman, 1995) influences its biodiversity causing a reduction of some species favourable habitats and, consequently, an increase of their energy demand for survival (Franco et al., 2002).

This relation is scale dependent and at the intermediate level (Olff & Ritchie, 2002) it is linked (a) to favourable habitats size and mutual distance (e.g. Jansson & Angelstam, 1999; Whithed et al., 2000) (b) to species dispersal capacity (Naugle et al., 1999; Howel et al., 2000; Delin & Andrèn, 1999); (c) to the differences within and among species (Bowers & Dooley, 1999; Kozakiewicz et al., 1999). The use of fragmentation as a control variable or as a comparison parameter is complicated by the non-existence of a specific accepted measure to estimate it (Tishendorf, 2001), plus the overlap between indicators used to evaluate it and the ones used to estimate heterogeneity.

Furthermore, it is not so reliable as a predictive tool (conservation management) due to secondary effects such as inter-specific relations, habitat alteration deriving from fragmentation itself and the great variability of the single species reactions (Bisonette & Storche, 2002; Bowers & Dooley, 1999; Mac Nally et al., 2000; Fauth et al., 2000).

These three parameters are spatially analysed to evaluate the choices to protect the biodiversity of a real planning tool (Provincial Territorial Plan -P.T.P.- of the Province of Venice, Italy).

The paper aims also at evaluating the application limits of the concepts and methods used.

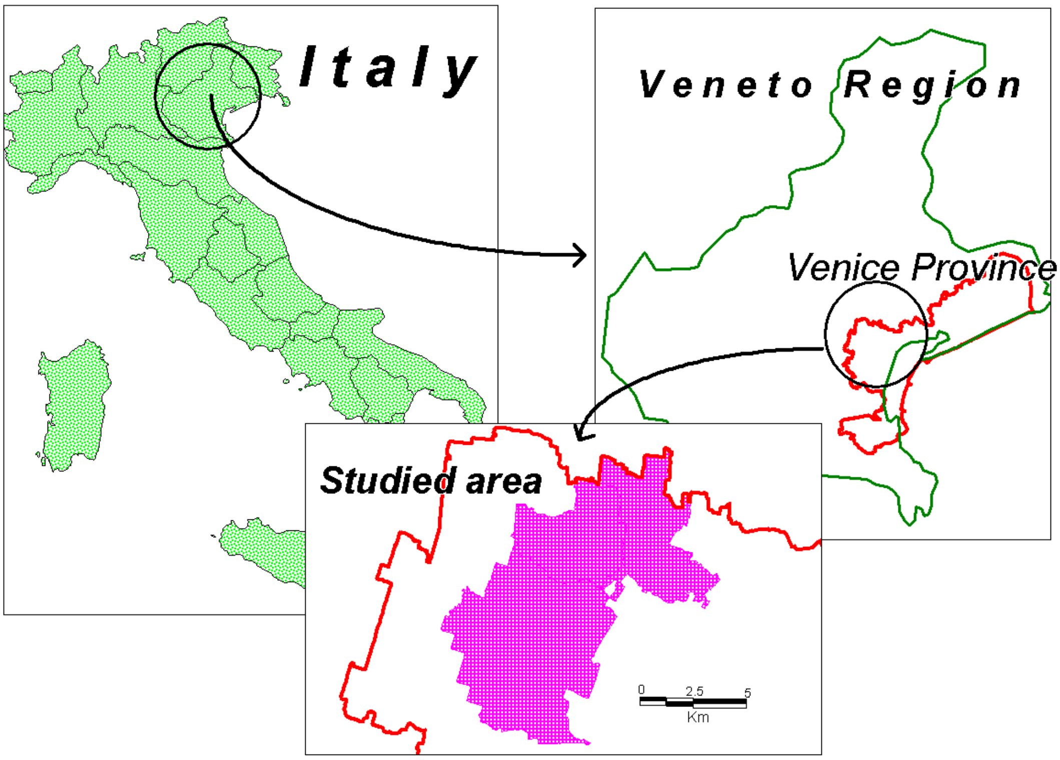
II. MATERIALS AND METHODS

II.1 Materials

The Provincial Territorial Plan adopted in 1999 for the Province of Venice bases the landscape ecological quality improvement on the creation of an ecological network, mostly correlated to birds and mammals biodiversity conservation (**Errore. L'origine riferimento non è stata trovata.**). The network design is based on the existing local protected areas and on the introduction of “re(af)forestation priority areas” and “ecological corridors” (AA.VV., 1999; AA.VV., 1994).

The analysis have been done on a portion of the Venice Province that covers 83 km2, using the cartography of the existent and of the situation designed by the plan (Element 10 table 1: features of the landscape as existent – as designed, 1:25000; paper and raster format).

Figure 1 The studied area.



The above data have been supported by the official Regional Technical Map and the mapping and classification of all the non urbanised ecotopes ("Progetto Siepi ©®; Franco, 2000).

The ecotopes of the analysed landscape have been classified as in Table 1.

The areas defined by the P.T.P. as "biotope" have been considered as "integrally" natural in both existing and planned scenario even if they agricultural areas. Moreover, all the ecosystems designed by the P.T.P. to improve the environmental area (re(af)forestation and ecological corridors, classified as integrally "natural biotopes"), have been optimistically thought as realized.

Table 1 Classification of the mapped ecotopes

DEFINITION	CODE
Urban	URB
Areas defined "biotope" by the P.T.P. ("re(af)forestation priority areas" and "ecological corridors")	B
Nursery/Plantation areas	AR
Afforestation areas	R
Public gardens and park areas	VP
Vineyard areas	V
Arable cropland	S

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Table 2 Spatial indices selected.

INDEX NAME	DESCRIPTION		
Percentage of favourable habitats (B%)	It is the favourable habitats percentage of the total studied area. In this case all the ecotopes classified as “B” has been considered as favourable	$\% B = \frac{\sum_{i=1}^{n} B_i}{A_t} 100$	B_i = area of the ecotopes classified as B; A_t = total area
Density of favourable habitats (B density)	It is the number of ecotopes classified as B by square kilometres.	$Bdensity = \frac{B}{A_t}$	B = number of B ecotopes; A_t = total area
Mean surface (Sm)	It is the estimate of the mean surface of the mapped landscape ecotope	$Sm = \frac{\sum_{k=1}^{l} A_k}{N}$	k = kind of ecotope, A_k = surface of the K ecotope, N = total number of ecotopes
Diversity (H)	It is the Shannon-Wiener index applied to the classified ecotopes.	$H = p_k \ln p_k$	p_k = surface proportion occupied by the K ecotope
Patton index (Pa)	The metric is a simple index of the ecotone density of an area (Forman, 1995; Lidicker, 1999)	$Pa = \frac{L}{2A_t\pi}$	L = total length of ecotopes margins; A_t = total area
M1 metric	The metric has been proposed (O’Neill et al., 1996) to estimate in a synthetic and robust way a landscape structural variation. The variation space is built up by three standardised spatial metrics.	$M1 = \sqrt{(H^2 + Pa^2 + Sm^2)}$	H = Shannon Wiener diversity index; Pa = Patton index; Sm= mean surface
Connectivity (γ)	The index has been used (Forman & Godron, 1986) to estimate an ecological network efficiency	$\gamma = \frac{L}{L_{\max}} = \frac{L}{3(Vo + Vi - 2)}$	L = number of links, L_{max} = maximum number of links, V_o = number of open nodes; V_i = number of intersection nodes;
Circuitry α	The index has been used (Forman & Godron, 1986) to estimate an ecological network efficiency	$\alpha = \frac{L - (Vo + Vi) + 1}{2(Vo + Vi) - 5}$	L = number of links, V_o = number of open nodes; V_i = number of intersection nodes
Mean distance (Mean Dist.)	The metric was calculated as the mean Euclidean Nearest-Neighbourhood Distance in each considered planned area, and equals the distance (km) to the nearest neighbouring patch of the same ecotopes (B) or group of ecotopes (B, R, Vp) based on shortest edge-to-edge distance.		
Maximum distance (Max. Dist.)	The metric has been calculated as the maximum Euclidean Nearest-Neighbourhood Distance in each considered planned area.		
Open nodes percentage (Vo%)	It is the percentage of nodes of the ecological network connected with only one link (Vo).	$Vo\% = \frac{Vo}{(Vo + Vi)}\%$	V_o = number of open nodes; V_i = number of intersection nodes
Rate of landscape division (D)	It has been defined (Jaeger, 2000) as the probability that two ecotopes kept by chance in a landscape are not in the same non fragmented area.	$D = 1 - \sum_{i=1}^{n} \left(\frac{A_i}{A_t}\right)^2$	A_i = ecotopes area, A_t = total area considered
Effective mesh size (M)	The metric (Jaeger, 2000) estimates the effective area where one can move without encountering a barrier.	$M = \frac{A_t}{S} = \frac{1}{A_t}\sum_{i=1}^{n} A_i^2$	A_i = i ecotope area, A_t = total area considered

II.2 Methods

The research procedure has been developed as following:

1. Selecting the spatial indices of the three parameters as a function of their explanatory strength.
2. Estimating the spatial scale effect on the indices, to define the coherence as to the extension and the resolution adopted by the P.T.P.
3. Comparing some existing and designed scenarios, differing for the extension used in the analysis and for the barriers considered (as to the potential biotic fluxes).
4. Interpreting the analysis results with regard to spatial parameters estimations (dispersion distances, home ranges) of existing potential populations.

II.2.1 Indices' selection

The indices used for the spatial analysis (Table 2) cannot quantify the ecological processes, but they can suggest ecological implication, assuming that the ecological processes interact with the landscape structures and are influenced by their configuration (Anderson & Danielson, 1997; Forman, 1995; Fahring & Merriam, 1985; Heinen & Merriam, 1990; Merriam et al., 1991; Opdam et al., 2002; Söndergrath & Schröder, 2002; Vulleumier & Prélaz-Droux, 2002).

Their effectiveness is limited by non linear relationships, ambiguous interpretations and thresholds in the process changes linked to the hierarchical nature of the landscape organization (Gustavson, 1998; Tishendorf, 2001).

II.2.1.a Heterogeneity

In order to estimate this parameter for "natural" landscapes (B; Table 1) two metrics were selected, namely the *percentage of favourable habitats* (B%) and the *number of favourable ecotopes* (B density); these have been found correlated to dispersion models (Tishendorf, 2001). For whole not urbanised landscapes the choice were*:* the *average surface* (Sm)*,* the *diversity* (H)*,* the *margins density* (Pa) and one (M1) summarizing all three (O'Neill *et al.*, 1996). This kind of spatial information correlated meaningfully to models and/or indicators of biotic processes (Miller et al., 1997).

II.2.1.b Connectivity

In order to estimate the landscape connectivity, related to the dispersion processes of the taxas mentioned by P.T.P, the *connection* (γ) and *circuitry* (α) indices (Forman & Godron, 1986) of the existing and designed "natural" or "paranatural" ecotopes network (B, R, VP; Table 1) were selected. These indices have no ecological meanings and they don't take into account the ecotopes qualities (both of structure and composition), but they can empirically give information about the functional exchanges in a landscape (see: Forman & Godron, 1986; Forman, 1995; Burel & Baudry, 1999; Franco, 2000). In order to estimate this parameter, further indices have been used: the mean and maximum distance (Mean Dist., Max. Dist.) between corridors and the percentage of "open nodes" connected by only one link (Vo%), given their impact on the connectivity effectiveness of an ecological networks (Anderson & Danielson, 1997).

II.2.1.c Fragmentation

The fragmentation has been evaluated measuring the Euclidean Nearest-Neighbourhood Distance of paranatural ecotopes (B, R, VP; Table 1) and through an index consisting of two metrics, namely the *landscape division rate* (D) and the *effective size* of the mesh (M) (Jaeger, 2000).

The cultivate ecotopes have been taken into account by weighting the anthropogenic pressure (Table 1) on the ecotopes "naturality", using a coefficient (ex Jaeger, 2000)

The coefficient values are those selected from a bibliographic analysis by one of the authors for the PLANLAND©® (Franco, 2000) procedure. The weights depress the metric value as a function of the use intensity; in the cultivated areas the minimal values (0,5) correspond to the intensive arable crops (mostly maize and soybean).

The urbanized and/or industrial patches have been assumed as completely inhospitable, and considered as a barrier. Among the corridors, in a first data set the roads (technical regional map) have been classified as barriers; while in a second data set all the roads and higher order canals have been classified assuming that the considered species reproduce only inside the unfragmented areas.

II.2.2 Evaluation of extension and grain

The mapped landscape covers about 83 km2 (areatot), of which the Southern surface (area 1) equals to about 40 km2, while the Northern one is about 43 km2 (area 2). The whole area has been divided by a sequence of grids with steps of 1, 2,5 and 5 km. In each of the obtained meshes, the selected indices have been computed for every grid.

The area 2 is equal to a mesh of about 6.5 km., and the considered area total surface to a mesh of about 9 km. The grain has been left unchanged and every single ecotope originally mapped at higher resolution has been aggregated on the basis of the P.T.P. resolution.

II.2.3 The scenarios comparison

II.2.3.a Quantitative comparison

Founding upon the results obtained in the first phases of the procedure, 12 scenarios have been analysed, comparing the existing versus designed ones (see Material and methods) that were obtained:

- for the total surface of the considered area (areatot);
- for the two equivalent surfaces of the considered area (area 1, area 2);
- assuming the asphalted roads as barriers (barrier 1 = b1);
- assuming the roads and the canals of higher level and/or the rivers as barriers (barrier 2 = b2).

II.2.3.b Spatial data interpretation versus the comparable biotic data

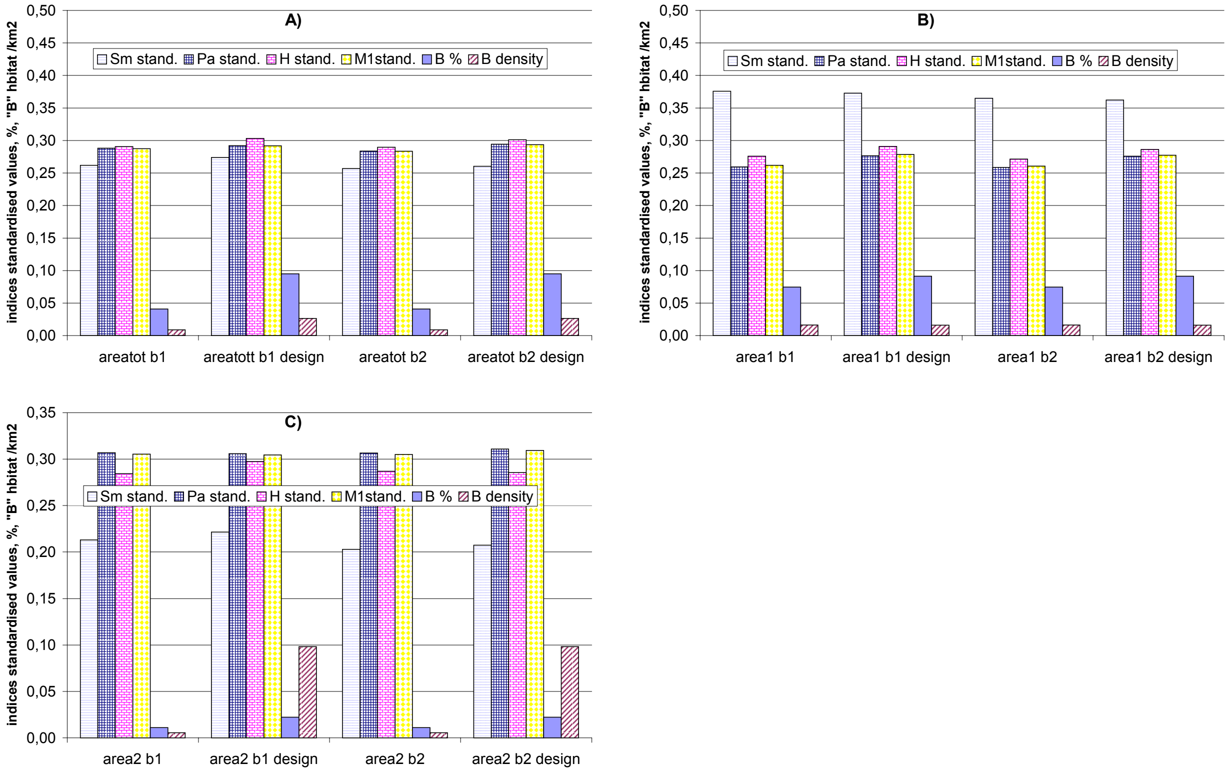
In this phase we tried to interpret the obtained spatial information versus the available spatial parameters concerning the potentially resident populations of mammals and birds.

III. RESULTS

III.1 Sensitivity analysis

The extension affects the performances of the heterogeneity and the fragmentation indices (Figure 3). The only index providing stable information with the variation of the scale is H (Shannon-Wiener diversity). The remaining ones have been found unstable (Delacourt & Delacourt, 1996) up to an extension of 6-7000 meters. The values of the effective measure of the mesh (M) did not result conservative, despite the expectations (Jaeger, 2000).

Fig. 2 Comparison among the heterogeneity indices in the real and planned scenarios :A) areatot, B) area1, C) area2. The “barrier effect” has been taken into account due to the roads



(b1) and to the roads and higher order canals (b2).

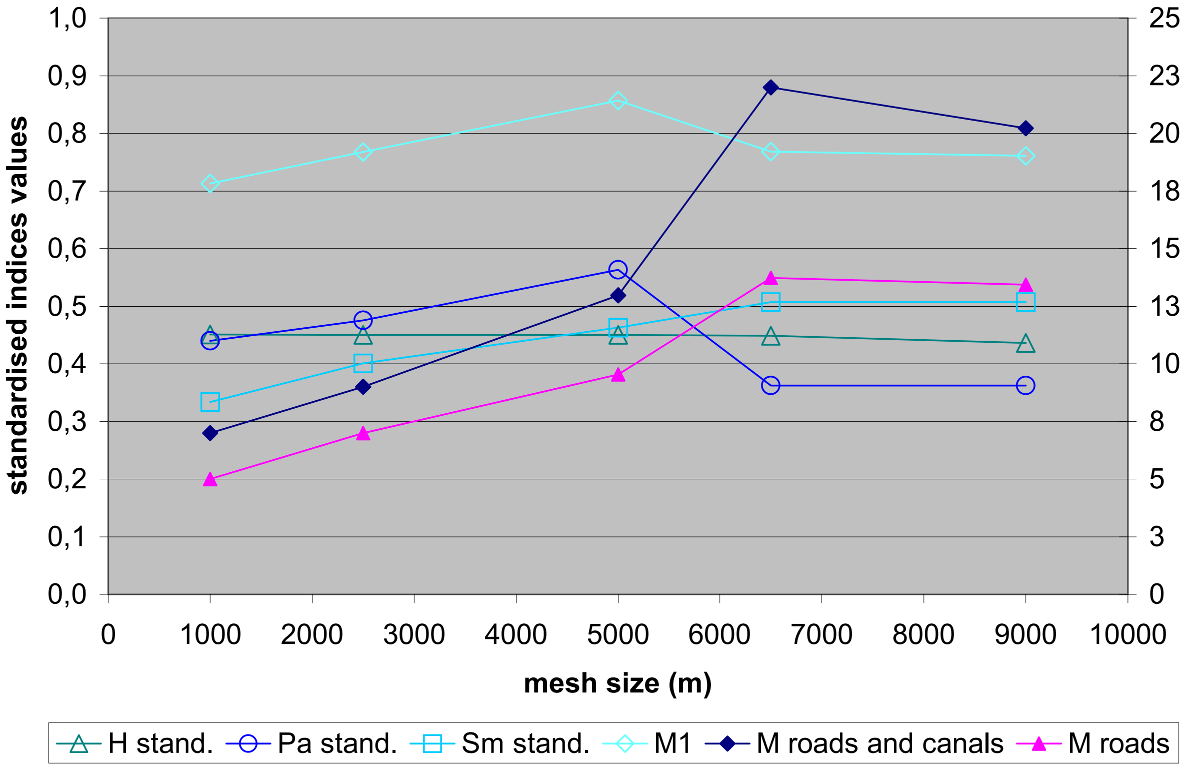
Therefore, as at the resolution adopted by P.T.P. for the landscape structures, the adoption of an area greater than 6000 meters of mesh does not result misleading in the use of the indices. This outcome has justified the comparison between the scenarios corresponding to the total investigated area (*areatot*) and between the two sub–areas (*area 1* and *area 2*) having more than 6500 meters of mesh.

III.2 The scenarios comparison

III.2.1 Heterogeneity

The differences found among the existing situations evaluated by the indices did not result meaningful (Fig. 2).

Figure 3 Average results of the estimated indices for each grid (meshes of 1,2.5,3,6.5,9 km). The values of diversity (H), margins' density (Pa), average ecotope surface (Sm) have been standardized for a comparison among them and with the M1 metric.



The highest variation (1.6%) for the M1 metric takes place in the scenery *area 1*. The variations are due especially to the rising of Pa (1.7%) and H (1.5%) between the existing and the planned situation. Substantial differences are observed for the Sm: the increments in the planned situation are found wholly greater in the *areatot* (1.2%); on the contrary, in the *area 1* they tend to be negative (-0.3%), because the added areas, as ecological corridors, have low surfaces. In the case of the two indices (see page 7) used for the "natural" ecotopes (B; Table 1), the B% never exceeds 2%. Furthermore, the B density habitats never attain 0.35 units for km^2. The kind of barrier (*b1*, *b2*), never affects the obtained information.

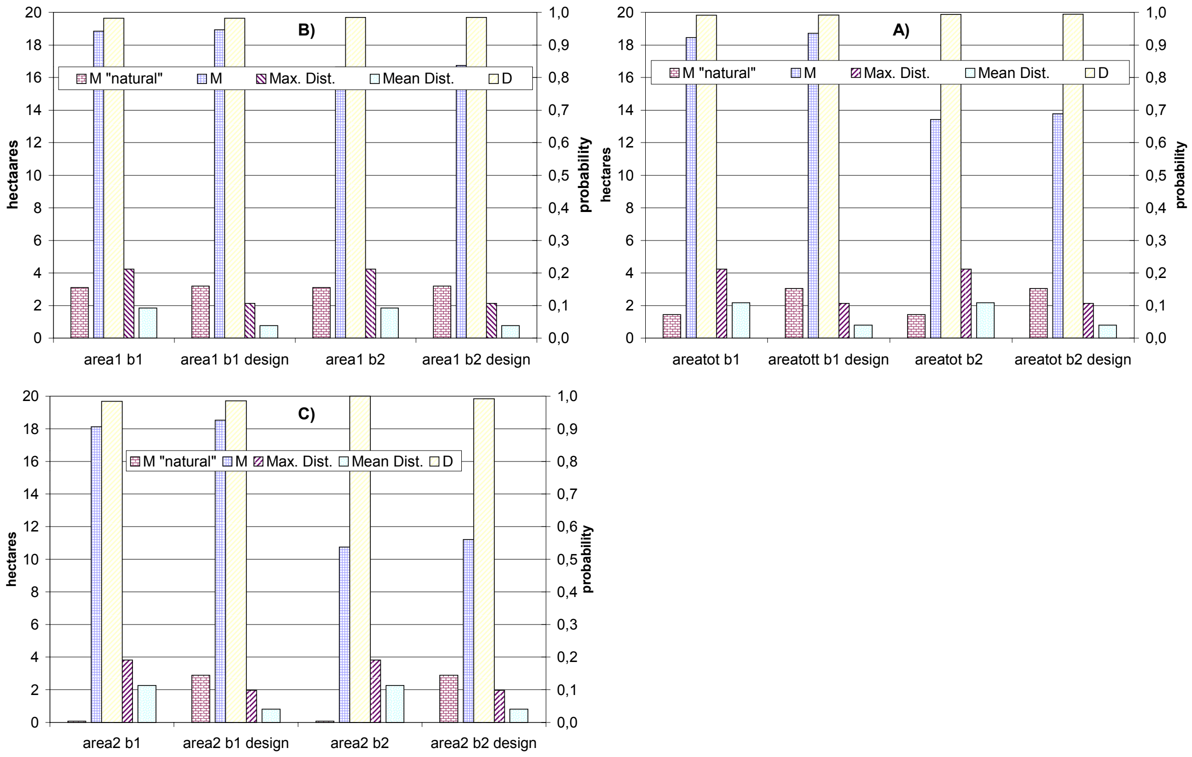
The highest variation detected for the *areatot* scenario is equal to 0.2 ha; B% never overcomes 2%, B density habitats never attain 0.35 units for km^2 and the kind of barrier (*b1*, *b2*) never affects the obtained information.

III.2.2 Connectivity

In the existing scenario the indices of connectivity and circuitry were not considered because the ecological corridors aren't defined at the adopted resolution.

In the designed scenarios the values of the γ index increase at values between 20 and 23%, while the α one between –21% and –22.4%. Vo% lies between 68% and 80% (**Errore. L'origine riferimento non è stata trovata.**). Finally, the Mean Dist. among “favourable ecotopes” (B, R, VP; Table 1) changes from values close to 2 km to values around 1 km, while the Max. Dist. changes from 4 km to values close to 2 km (**Errore. L'origine riferimento non è stata trovata.**, Fig. 4).

Fig. 4 Comparison of the fragmentation indices among the A) area tot, B) area 1 and C) area 2 existing and planned scenarios, taking into account the barrier effects of roads (b1) and the roads and the higher order canals (b2).



III.2.3 Fragmentation

The increase of M is included between 0.5% (*area1*) and 4% (*area2*). It also shows the highest difference between the scenarios marked by the kind of barrier (*b1*=2%; *b2*=4%) (Fig. 4). The percentage variations are referred to change below 0.5 ha. If in the metric’s computation we insert only the “(para)natural” ecotopes (B, R, VP; Table 1), then the rises are poor in *area 1* (0.1 hectares) and in high percentage (97%) in *area 2*, where the surfaces increase from negligible values to about 3 hectares. *Areatot* balances the spatial differences, showing effective percentage variations between those found in *area 1* and in *area 2* (52%,1,6 hectares). The D index keeps unchanged, having values next to 90%.

IV. DISCUSSION

IV.1 Quantitative comparison

The sensitivity analysis highlights that, at the resolution adopted by the P.T.P. and to the extensions of the analysis (meshes of 6.5 and 9 km), the indices effectively allowed to appreciate variations of the considered spatial parameters because:

1. They have been able to find out the different effects of the transformations expected in each parameters. In the case of heterogeneity, for instance, area 1 changes quite differently from area 2 (Fig. 4) even if over a limited variation range. In fact, the "natural" areas designed are smaller than the average size of the ecotopes. This increases both variety and ecotone conditions, while decreases the average surface of the existing ecotopes.
2. Thanks to the performed analysis, there are some differences in the gained information concerning the analysed parameters: the metrics relating to heterogeneity are sensible neither to the kind nor to the magnitude of the barriers, in contrast with the metrics relating to fragmentation.
3. Differences of the parameters, even if small, have been found according to their position (area1, area2) and to the change of the adopted scale.

Therefore, analysing the indices values for the different scenarios one can effectively infer that the improvements produced by planning are negligible.

As for heterogeneity, this is true both for the landscape as a whole, composed of "paranatural" and agricultural habitats (Pa, Sm, H, M1), and for the "natural" habitats corresponding to the B% and B density values (Fig. 2).

Even the landscape fragmentation seems to be scarcely influenced by the planning, with values that sometimes seem to be of high percentage, while actually concern very low surface values (Fig. 4).

Regarding connectivity, the connections increase must be evaluated considering the low circuitry of the network and high percentage of "open nodes" (**Errore. L'origine riferimento non è stata trovata.**). Excluding the "ecological quality" of the corridors, these characteristics indicate a bad spatial organization of the network for the metapopulations that perceive these structures as corridors (Anderson & Danielson, 1997). Finally the average and maximum distances among "natural" ecotopes decrease, even if they are still remarkable (cfr. pag. 14).

If the indices used can evaluate the examined landscape parameters, and if these parameters are related both theoretically and empirically with the metapopulation dynamics, it's reasonable to suppose the existence of a link between this limited or negative variation of parameters and biodiversity. This is true for area 1, area 2 and for areatot (the sum of both areas) with very small differences.

IV.2 Interpretation of the spatial analysis as regards the biotic data

The *richness* and the *diversity* of birds species are sensible to the extension of the forested patches (Forman, 1995; Park & Lee, 2000), in relation with ecosystems' structure and landscape configuration (Brotons & Herrando; 2001; Fauth et al. 2000;

Naugle et al. 1999), and with the autoecological characteristics of each species (e.g. Howel et al., 2000; Brotons & Herrando, 2001; Opdam, 1991).

Fig. 5 are reported the number of paranatural biotopes for each extension class in the real and planned scenarios.

Only the empirical relationship between the species richness and the dimension of the "natural" areas examined was considered (forested or wetlands, e.g. Jansson & Angelstam, 1999). Using the same surface extension categories of a study carried out in a periurban landscape (Park & Lee, 2000: <1 ha, 1-9.9 ha, 10-100 ha, >100 ha) in our case we found an increase of the 1-9,9 and 10-100 areas (**Errore. L'origine riferimento non è stata trovata.**). It would be possible that this increase influences positively the number of species that perceive the landscape without barriers (Brotons & Herrando, 2001) and have a mean dispersion capacity of 1 km and a maximum capacity higher than 2 or 3 km.

These characteristics are really critical for the protected poplations potentially present in the area (e.g. *Passeridae, Fringillidae* and *Paridae*) and not critical for the species more adaptable in this rural landscape (e.g. *Alaudidae, Corvidae, Sturnidae, Columbidae*) (Peterson et al,. 1983; Brichetti et al., 1996). Even small mammals most adaptable to the agricultural landscape (for ex. *Apodemus* spp.) do not cover these dispersal distances, which could instead be favourable to more vagile species, such as *Rattus* spp (Corbett & Harris, 1991; Grassè & Dekeyser, 1955; Kozakiewicz et al., 1993, 2000; Santini, 1983). The limited availability of favourable habitat does not change the nowdays status of populations having higher dispersal ability (e.g. among birds, *Accipitridae, Falconidae, Stigidae* and *Tytonidae*) (Fig. 2, Fig. 4).

The increases of the effective mesh size deriving from the plan are limited for all kinds of birds and small mammals more adapted to the agricultural environment (*A. agrarius, R. rattus*; Santini,1983). On the other hand these variations are uninfluential for adaptable and with low dispersal range micro mammals (*S. araneus, S minutus, C. suaveolens, C. russula*, etc).

The increase of the effective mesh (Fig. 5) seems too small for the species more linked to "natural" environments (in this case AR, R, B, Table 1) and sensible to the barriers (roads/canals) that might be present (e.g. *Parus spp., T. troglodytes, Regulus spp, Phyllosocopus Bonelii, Sylvia atricalylla, Serinus serinus, Emberiza spp., Columba Palumbus*, *Mulstela nivalis*, *A. flavicollis, A. sylvestris, C. glareolus, M. agrestis, M. arvalis*; Bélisle & Desrochers, 2002; Brichetti et al., 1996; St.Clair et al., 1998; Corbet & Harris, 1991; Corbet & Ovenden, 1985; Spagnesi & Morselli, 1996).

The effect of the design on connectivity (γ, **Errore. L'origine riferimento non è stata trovata.**) might result low due to the reduced extension of the hospitable habitats (Fig. 2), and to the poor impact on the dynamics of the small mammals pry populations linked to the dynamics of genus as *Stigidae* and *Tytonida*.

V. CONCLUSIONS

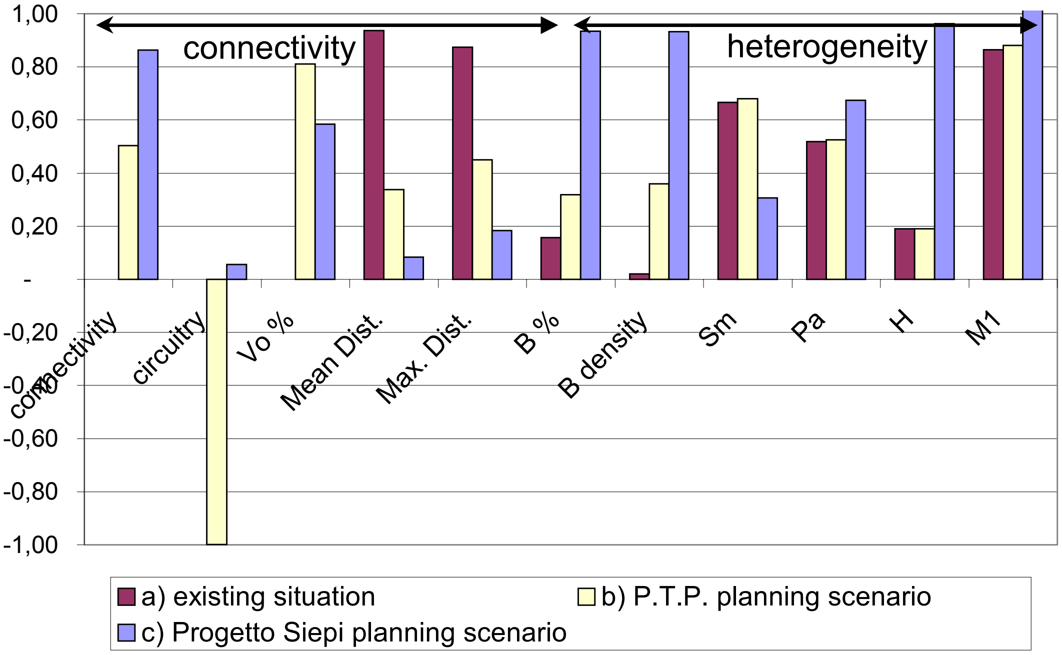
The study of the scale effect on the indicators used for fragmentation and heterogeneity confirmed the high influence of the extension on their informative content.

All the indicators selected, but the diversity index (H), came out to be unstable for meshes up to 6-7000 meters. Using these values as significance field threshold, it was possible to evaluate the indicators efficiency in estimating the considered parameters.

It was possible, joining several non redundant indicators, to appreciate the spatial alterations caused by planning on fragmentation, heterogeneity and connectivity of the rural landscape.

The analysis outlined that the plan would probably lead to a little effect on these three landscape parameters and, as a consequence, to a secondary effect on the biodiversity.

Fig.6 Comparison among the metrics of the area 2 considering A) the existing condition, B) the analysed planning solutions and C) the results of another plan in the same area designed at lower resolution. In this case the data come from a rural landscape amelioration plan by means of the agroforestry network optimisation (Progetto Siepi, see Franco, 2000).



Even if this result is useful to evaluate the planning potential effects, its interpretation met several difficulties, such as the interpretation of indicators without any theoretical limits. In fact, it is unclear how much the measured variations may be meaningful referring to metapopulation's dynamics, even if the use of several simple indices to describe a single parameter and the comparison between percentage and dimensional variations was useful to detect ambiguous information. Moreover, once a first qualitative relationship has been established comparing the landscape spatial data with the spatial biotic parameters of the potentially relevant populations (i.e.: no variation will imply no effect), it's hard to transform this indication in predictable effects on the species present; however, the comparative analysis highlighted that the impact of transformation is likely to be of little influence. The landscape structure on the basis of the P.T.P. resolution is not significant for most of the species that could live there, characterised by a multiple use of resources, limited dispersal capacity, and influenced by rural and paranatural structures not foreseen by the plan. Furthermore, the species of a highest interest among the chiropters, reptiles, insects and the flora as a whole, are all widely influenced by the landscape structures at this resolution level.

Including in the spatial evaluation the landscape structure with a resolution lower than the P.T.P. one (agroforestry linear systems; Franco 2000), all the spatial indicators will undergo considerable changes, affecting direct influences on all the population parameters, (dispersal distances and home ranges) previously not scratched by P.T.P. (**Errore. L'origine riferimento non è stata trovata.**).

This work highlighted also some limits:

- Absence of uniformity in the use of indicators and/or uncontrolled practice with indicators without experimental or theoretical validation.
- Absence of reference framework about the links among the landscape's structures, biodiversity and landscape functions (e.g. Dramstad et al., 2001) at different scales.
- Lack of accessible and co-ordinated information about the spatial parameters of the reference species or guilds.

Finally, the work takes us to two conclusions:

- In order to achieve biodiversity conservation, planning have to be based on the planned landscape species spatial needs. The choice of the plan scale (extension and resolution), should primarily comply with these needs, and only secondarily with the administrative ones.
- Studies at different scales are urgent. Without reference methodologies and empirical and predictive knowledge, the biodiversity management by means of planning could dangerously turn from a motor of the sustainable landscape development (Franco, 2002) into a simple bureaucratic device.

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