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

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Abstract

Purpose – Landscape ecology represents an area of theoretical and empirical support of spatial planning, providing parameters such as heterogeneity, connectivity and fragmentation. The aim of this study was to use these parameters to evaluate the choices of a real planning tool to protect the biodiversity, to evaluate the applicability limits of concepts and methods used.

Design/methodology/approach – 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).

Findings – The study confirmed the scale's effect on the indices, unstable at the adopted resolution for extensions up to 6,000-7,000 meters. The selected indices permitted appreciation of the low effectiveness of the real planning tool in improving conservation of biodiversity. The paper suggests that empirical studies and predictive knowledge at different scales are urgent in this field. To preserve biodiversity, the choices of planning scale should primarily comply with the spatial needs of the various species.

Originality/value – Evaluates a real planning tool to protect biodiversity.

Keywords Ecology, Conservation

Paper type Research paper

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 and de Lucio, 2001; Farina, 1997; Preiss *et al.*, 1997; Jonsen and 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 now, landscape "connectivity" cannot be measured in a simple and general way (D'Eon *et al.*, 2002; Tishendorf and Fharing, 2000), but the connection rate of the "paranatural" ecosystems in a rural landscape can be an index of some of the potential

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populations (plants, birds, and small mammals) dispersal ability (Franco, 2000; Barr and 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 and Ritchie, 2002) it is linked:

- to favourable habitats size and mutual distance (e.g. Jansson and Angelstam, 1999; Whited *et al.*, 2000);
- to species dispersal capacity (Naugle *et al.*, 1999; Howel *et al.*, 2000; Delin and Andrèn, 1999); and
- to the differences within and among species (Bowers and 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 and Storch, 2002; Bowers and Dooley, 1999; MacNally *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 (PTP) of the Province of Venice, Italy) (Figure 1).

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

Materials and methods

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. 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 km^2, using the cartography of the existent and of the situation designed by the plan (element 10, Table I: features of the landscape as existent – as designed, 1:25000; paper and raster format).

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 I.

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

Indices selection

The indices used for the spatial analysis (Table II) 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 and Danielson, 1997; Forman, 1995; Fahring and Merriam, 1985; Heinen and Merriam, 1990; Merriam *et al.*, 1991; Opdam *et al.*, 2002; Söndergrath and Schröder, 2002; Vulleumier and 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 organisation (Gustavson, 1998; Tishendorf, 2001).

Heterogeneity. In order to estimate this parameter for "natural" landscapes (*B*; Table I) 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 (*M*1) summarising 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).

Connectivity. In order to estimate the landscape connectivity, related to the dispersion processes of the taxas mentioned by PTP, the connection (γ) and circuitry (α) indices (Forman and Godron, 1986) of the existing and designed "natural" or "paranatural" ecotopes network (*B, R, VP*; Table I) were selected. These indices have no ecological meanings and they do not 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 and Godron, 1986; Forman, 1995; Burel and 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 and Danielson, 1997).

Fragmentation. The fragmentation has been evaluated measuring the euclidean nearest-neighbourhood distance of paranatural ecotopes (*B, R, VP*; Table I) 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 I) on the ecotopes "naturality", using a coefficient (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 urbanised 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.

Evaluation of extension and grain

The mapped landscape covers about 83 km^2 (areatot), of which the southern surface (area 1) equals to about 40 km^2, while the northern one is about 43 km^2 (area 2). The

Table II.
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" have been considered as favourable	$\%B = \frac{\sum_{i=1}^{n} B_i}{A_t} 100$	Bi = 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 ecotope 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 and 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 and 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	$D = 1 - \sum_{i=1}^{n} \left(\frac{Ai}{At}\right)^2$	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

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

The scenarios comparison

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:

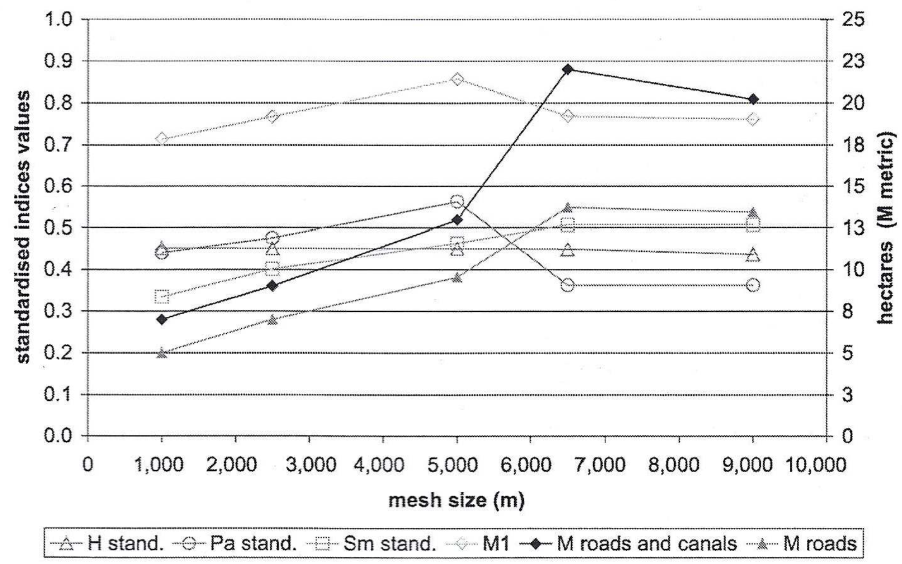
- for the total surface of the considered area (areatot);
- for the two equivalent surfaces of the considered area (area 1, area 2);
- assuming the asphalted roads as barriers (barrier 1 = *b*1); and
- assuming the roads and the canals of higher level and/or the rivers as barriers (barrier 2 = *b*2).

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.

Results

Sensitivity analysis

The extension affects the performances of the heterogeneity and the fragmentation indices (Figure 2). The only index providing stable information with the variation of



Note: 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

Figure 2.
Average results of the estimated indices for each grid (meshes of 1, 2.5, 3, 6.5, 9 km)

the scale is H (Shannon-Wiener diversity). The remaining ones have been found unstable (Delacourt and Delacourt, 1996) up to an extension of 6,000-7,000m. The values of the effective measure of the mesh (M) did not result conservative, despite the expectations (Jaeger, 2000).

Therefore, as at the resolution adopted by PTP for the landscape structures, the adoption of an area greater than 6,000 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 6,500m of mesh.

The scenarios comparison

Heterogeneity. The differences found among the existing situations evaluated by the indices did not result meaningful (Figure 3).

The highest variation (1.6 per cent) for the $M1$ metric takes place in the scenery area 1. The variations are due especially to the rising of Pa (1.7 per cent) and H (1.5 per cent) 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 per cent); on the contrary, in area 1 they tend to be negative (-0.3 per cent), because the added areas, as ecological corridors, have low surfaces. In the case of the two indices used for the "natural" ecotopes (B; Table I), the $B\%$ never exceeds 2 per cent. 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 per cent, B density habitats never attain 0.35 units for km^2 and the kind of barrier ($b1$, $b2$) never affects the obtained information.

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

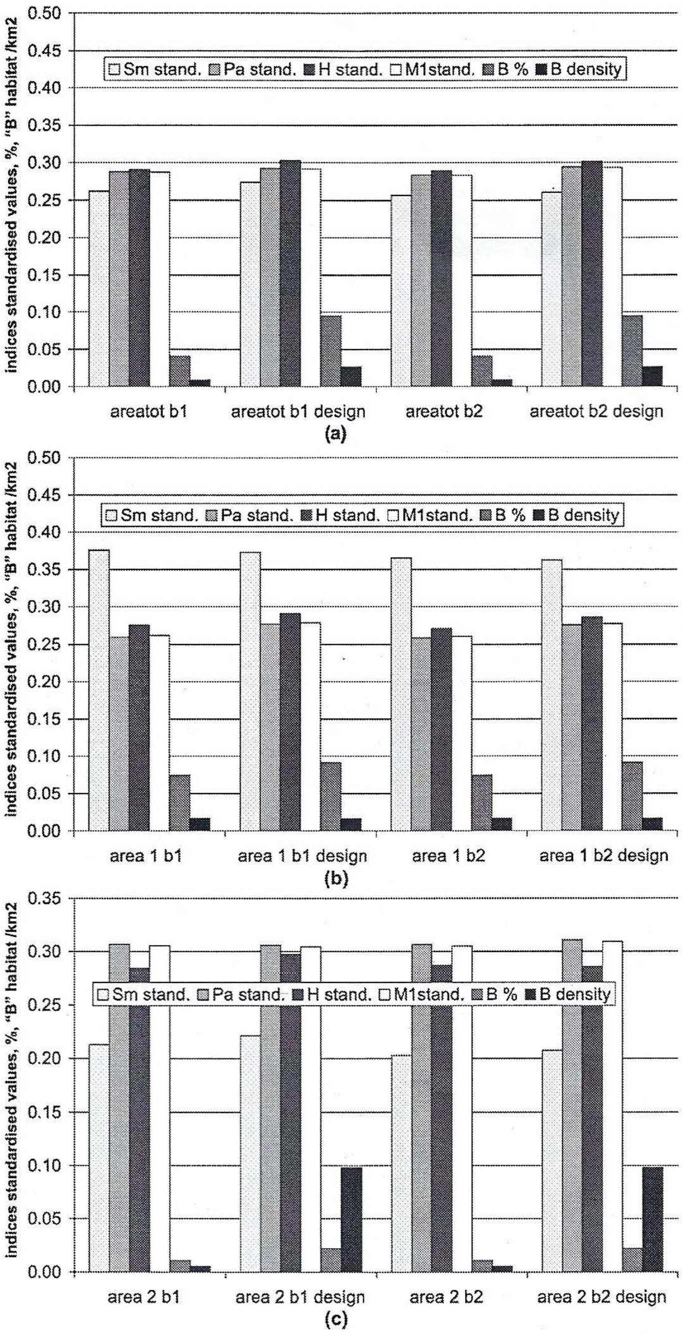
In the designed scenarios the values of the γ index increase at values between 20 and 23 per cent, while the α one between -21 per cent and -22.4 per cent. $Vo\%$ lies between 68 per cent and 80 per cent (Figure 4). Finally, the Mean Dist. among "favourable ecotopes" (B, R, VP; Table I) changes from values close to 2 km to values around 1 km, while the Max. Dist. changes from 4 km to values close to 2 km (Figures 4 and 5).

Fragmentation. The increase of M is included between 0.5 per cent (area 1) and 4 per cent (area 2). It also shows the highest difference between the scenarios marked by the kind of barrier ($b1 = 2$ per cent; $b2 = 4$ per cent) (Figure 5). 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 I), then the rises are poor in area 1 (0.1 hectares) and in high percentage (97 per cent) in area 2, where the surfaces increase from negligible values to about three hectares. Areatot balances the spatial differences, showing effective percentage variations between those found in area 1 and in area 2 (52 per cent, 1.6 hectares). The D index keeps unchanged, having values next to 90 per cent.

Discussion

Quantitative comparison

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



Note: The "barrier effect" has been taken into account due to the roads (b1) and to the roads and higher order canals (b2)

Figure 3.
Comparison among the heterogeneity indices in the real and planned scenarios: A) areatot, B) area 1, C) area 2

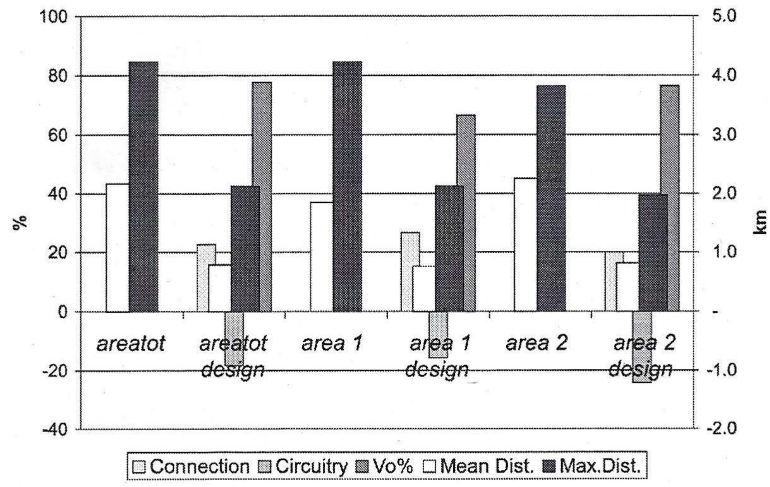


Figure 4.
Comparison of the connectivity indices (γ,α, *Vo*%, Mean Dist., Max. Dist.) among the area tot, area 1 and area 2 existing and planned scenarios

- 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 (Figure 5) 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.
- 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.
- Differences of the parameters, even if small, have been found according to their position (area 1, area 2) 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, M*1), and for the "natural" habitats corresponding to the B% and B density values (Figure 3).

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 (Figure 5).

Regarding connectivity, the connections increase must be evaluated considering the low circuitry of the network and high percentage of "open nodes" (Figure 4). Excluding the "ecological quality" of the corridors, these characteristics indicate a bad spatial organisation of the network for the metapopulations that perceive these structures as corridors (Anderson and Danielson, 1997). Finally the average and maximum distances among "natural" ecotopes decrease, even if they are still remarkable.

If the indices used can evaluate the examined landscape parameters, and if these parameters are related both theoretically and empirically with the metapopulation

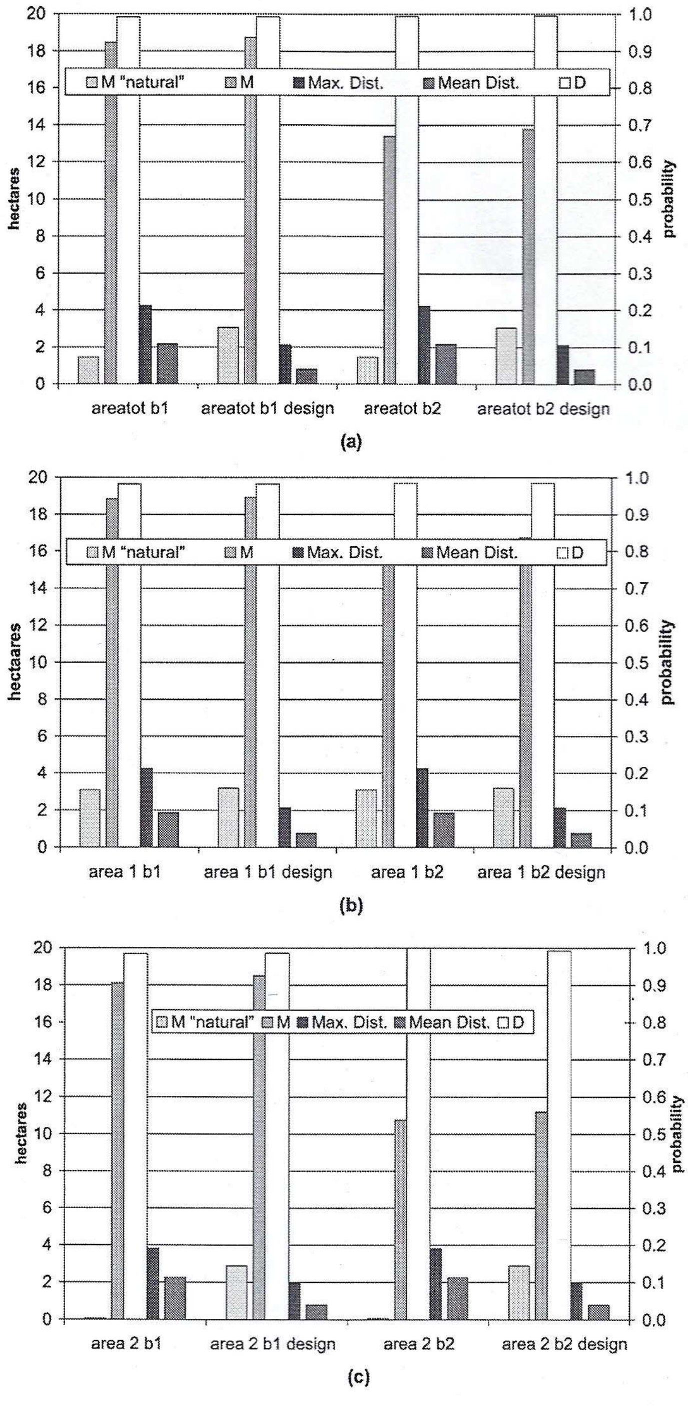


Figure 5.
Comparison of the fragmentation indices among the (a) area tot, (b) area 1 and (c) area 2 existing and planned scenarios, taking into account the barrier effects of roads (*b*1) and the roads and the higher order canals (*b*2)

dynamics, it is 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.

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 and Lee, 2000), in relation with ecosystems' structure and landscape configuration (Brotons and 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 and Herrando, 2001; Opdam, 1991).

Only the empirical relationship between the species richness and the dimension of the "natural" areas examined was considered (forested or wetlands, e.g. Jansson and Angelstam, 1999). Using the same surface extension categories of a study carried out in a periurban landscape (Park and 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 (Figure 6). It would be possible that this increase influences positively the number of species that perceive the landscape without barriers (Brotons and 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 populations 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 example, *Apodemus* spp.) do not cover these dispersal distances, which could instead be favourable to more vagile species, such as *Rattus* spp (Corbet and Harris, 1991; Grassé and Dekeyser, 1955; Kozakiewicz *et al.*, 1993, 1999; Santini, 1983). The limited availability of favourable habitat does not change the nowadays status of populations having higher dispersal ability (e.g. among birds, *Accipitridae, Falconidae, Stigidae* and *Tytonidae*) (Figures 3 and 5).

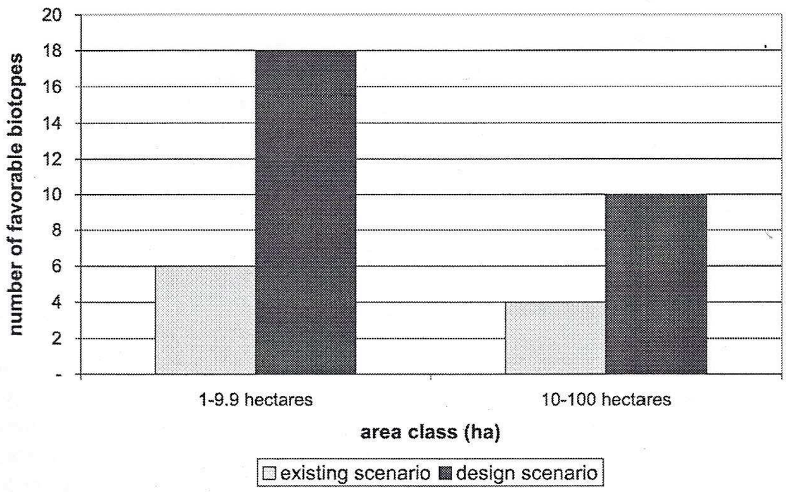


Figure 6.
Area reported the number of paranatural biotopes for each extension class in the real and planned scenarios

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 (Figure 5) seems too small for the species more linked to "natural" environments (in this case *AR, R, B*, Table I) 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 and Desrochers, 2002; Brichetti *et al.*, 1996; St. Clair *et al.*, 1998; Corbet and Harris, 1991; Corbet and Ovenden, 1985).

The effect of the design on connectivity (γ, Figure 4) might result low due to the reduced extension of the hospitable habitats (Figure 3), and to the poor impact on the dynamics of the small mammals pry populations linked to the dynamics of genus as *Stigidae* and *Tytonida*.

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 6,000-7,000 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.

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 is 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 PTP 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 PTP 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 PTP (Figure 7).

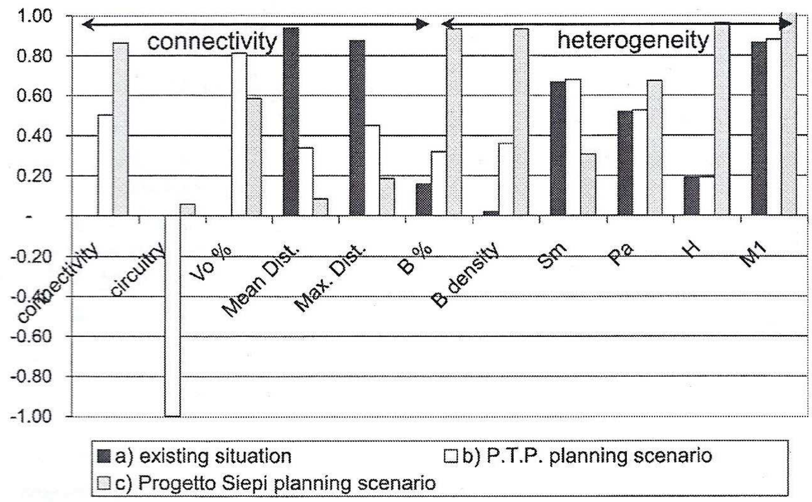
This work also highlighted some limits:

- 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; and
- 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:

(1) 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.

(2) 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.



Note: In this case the data come from a rural landscape amelioration plan by means of the agroforestry network optimisation (Progetto Siepi, see Franco, 2000)

Figure 7.
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

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