FRANCOANGELI/Urbanistica

Planning Support Tools: Policy Analysis, Implementation and Evaluation

Proceedings of the Seventh International Conference on Informatics and Urban and Regional Planning INPUT 2012

edited by Michele Campagna, Andrea De Montis, Federica Isola, Sabrina Lai, Cheti Pira, Corrado Zoppi

Landscape fragmentation in Italy. Indices implementation to support territorial policies

by Bernardino Romano[1] and Francesco Zullo[1]

Every area has some form of biodiversity and its biopermeability may be defined as the ability to host and allow the transit of animal and plant species. Landscape biopermeability will be discussed below according to the different classes of land as agricultural areas, uncultivated and degraded areas, forest areas and pastoral lands. In areas defined as biopermeable (about 62% of the Italian territory), it is important to assess fragmentation which is an indicator of afforded ecological and functional quality. In this study we developed a new indicator to assess both current fragmentation conditions of the area studied, but also and above all the decision-making level we need to operate at in order to reduce the current fragmentation rate of landscape continuity.

Introduction

Every area has some form of biodiversity and its biopermeability may be defined as the ability to host and allow the transit of animal and plant species. Some areas, of course, have greater biopermeability than others owing to vegetation and wildlife of conservational interest (Romano, 1996; Pungetti and Romano, 2004; La Rovere, Battisti and Romano, 2006).

The aim of this paper is to investigate average levels of environmental fragmentation in Italian regions, by introducing some indicators tied to the policy and planning efforts needed to reduce discontinuity between areas of great importance for national biodiversity. The goal is to understand which regions present more critical levels of spatial disruption of areas of residual

[1] University of L’Aquila (Italy).

naturalness (biopermeability) and, as a result, the amount of design, planning and governance needed to mitigate these adverse conditions (Gustafson and Parker, 1992; Moser *et al*., 2007).

The recent current of connectivity conservation pursues the general aim of mitigating the effects of environmental fragmentation on species, communities, ecosystems and ecological processes, by adopting appropriate strategies (Vuilleumier and Prèlaz-Droux, 2002; Crooks and Sanjayan, 2006; Opdam, Steingrıver and van Rooij, 2006; Selman, 2006; Girvetz *et al*., 2008).

The geographical spaces better suited to the presence and dispersion of multiple species are, probabilistically, those that are less disturbed by human activities and settlements of varying nature (Boitani *et al*., 2007; Biondi *et al*.,2003). Biopermeability may be determined by using different territorial features (Vannuccini and Geri, 2006; Bressan and Poldini, 2006), such as phyto-sociological structure, actual vegetation or land use, even though the latter source seems to be tied more directly to degree of anthropic use detected. Moreover, land use forms are more standardised across European national territories (Corine Land Cover) on homogenous scales of detail compared to surveys of actual vegetation, for example, which are less widespread at the local level and generally less standardised. Italy is a very diversified country morphologically and climatically, with an extremely varied eco-mosaic marked by many historical cycles of landscape transformation and re-naturalisation. It is possible to highlight two levels of natural conditions starting from land use: low biopemeability, tied to more intensive use by man, and high biopermeability, tied to land scarcely or not at all transformed over the years or in conditions of natural recovery as a result of cessation of previous uses. Landscape biopermeability will be discussed below according to the different classes of land taken into consideration in its definition.

Agricultural areas

Italian agricultural areas are generally ecologically unfriendly because intensive agriculture, farmhouses, noise and illumination impoverish biodiversity. To add to this there are the barriers created by property fences, human movements and infrastructural networks (Jaarsma, 1997; Forman, Sperling and Bissonette, 2002). The level of biopermeability can be very low and consequently many of these areas are not suitable to function as ecological corridors. Despite this, a few uplands with extensive agriculture present a higher level of biodiversity and could already be used as ecological corridors. The study of these areas has shown the difficulty to assign to them a proper level of biopermeability unless a detailed study is carried out. In large areas of agricultural land the eco-connectivity quality is poor, but it varies in other areas according to cultivation typology, field dimension and productive cycles alternation. If on the one hand agricultural areas are today ecologically unfriendly, on the other hand they have the potential to become a resource for species movements, losing their attribute of barriers and gaining that of connection. Their configuration is relevant for this new attribute. Long and narrow configurations, for example, present transversally high biopermeability and facilitate the biological movement between the near areas with a low degree of disturbance.

Uncultivated and degraded areas

Uncultivated areas, with residual agricultural vegetation, present on the average a good level of biopermeability since they do not retain either occlusion to species movement or disturbance by human impact. The debate on marginal lands in Italy has been based on the equation that lack of explicit and localised environmental quality is equal to the constraining environmental impact of the planning actions. Marginal lands had little consideration in the Italian framework of environmental quality. These areas, however, retain a particular importance regarding links with the surroundings and can hence form a valuable element for an ecological network. Moreover, due to their low economic value they constitute priority sites for environmental restoration and re-naturalisation. The level of biodiversity in uncultivated and degraded areas varies from site to site and necessitates a detail examination of their physical, ecological and structural characteristics.

Forest areas

Forest areas present high ecological values for many animal species, due to the spread of refuges and hiding places and to the generally limited human disturbance. A high level of biodiversity can be found in this macro element (about 24% of entire national territory). More detailed research however is needed to identify differences as regards various parameters and indicators (*e.g.* forest texture, species diversity, past and present levels of human use, local morphological structure, disturbance factors) (Haines-

Young and Chopping, 1996). This further evaluation should present a wide spectrum of biopermeability levels, fundamental for the identification of the high level lines of environmental continuity on the ground.

Pastoral lands

Pastoral lands in Italy are a macro element with an acceptable level of biodiversity to enhance environmental continuity. The condition of vegetation diversity here is often better than in uncultivated or degraded lands, though in pastoral lands the disturbance due to human activities is higher. If occlusion to environmental continuity, *e.g.* from property boundaries, occurs in these areas, it can be easily overcome due to its deformed or uncompleted character and low impact on the territory. Detailed studies on pastoral lands can also allow verifying the different potential ecofunctions. The elements at stake are, apart from the morphological ones that condition all types of biopermeability, the phytological characters of pasture, the husbandry weight and the swath frequency.

Fig. 1 – Forestry national map.

Fig. 2 – Biopermeability national map.

In areas defined as biopermeable (about 62% of the Italian territory), it is important to assess fragmentation which is an indicator of afforded ecological and functional quality. By environmental fragmentation we mean the man-made dynamic process through which a natural area is divided up into more or less separated and progressively smaller and isolated fragments (Forman, 1995; Batistella, Brondizio and Moran, 2000; Jaeger, 2000). The effects of fragmentation may be observed at different levels, but this paper will focus on the landscape level (Bissonette, 1997). At landscape level and in areas inhabited by mankind in the course of history, there are landscape "eco-mosaics" scattered in an anthropic matrix like residual environmental fragments owing to the gradual alteration of the original pattern (Bogaert *et al.*, 2000; McGarigal and Cushman, 2002). The extent of environmental fragmentation affects ecological factors and processes at all hierarchical levels (individual, eco-system and landscape) and at different space and time levels (Schumaker, 1996; Battisti and Romano, 2007). Environmental fragmentation has become of strategic importance for the conservation of biodiversity and there is a great deal of scientific literature that stresses its significance in all countries (Fahrig, 1997, 2003; Dunning, Danielson and Pulliam, 1992; Taylor *et al*., 1993; Bennett, 1999; Battisti, 2003; Hanski, 2005; Bonnin *et al*., 2007; EEA, 2011). Its importance has been confirmed by the issue and initial application stages of the Habitat Directive and the ensuing Nature 2000 programme, as well as the Global Biodiversity Strategy and the equivalent national strategy (NBS) (World Resources Institute, 1992; Andreella and Brécciaroli, 2011). The planning undertaken by the over 8100 Italian municipalities, which has produced overall most of the territorial transformations in the country over the past 50 years (Romano *et al*., 2011), has had profound adverse effects on the environmental continuity of the territorial matrices where the major natural areas of the country are situated.

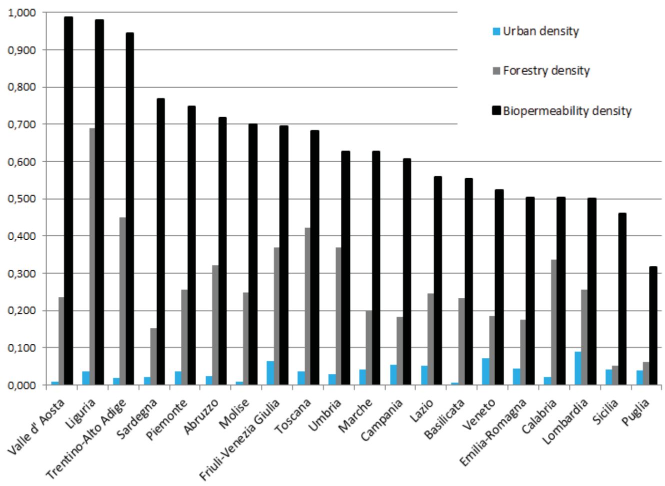

Fig. 3 – Biopermeability, urban and forestry rate in Italian region.

The spatial growth of urban settlements, especially in morphologically "weak" areas (plains and valley lines) and interruptions in continuity between natural areas caused by roads are the main causes of the high level of fragmentation (Razin and Rosentraub, 2000; Bierwagen, 2005; Jaeger *et al*., 2010).

This is leading to the swift loss of connectivity in the complementary ecosystem structure, a loss that penalizes the species present in a very different way depending on their ecology and ethology. The interference of settlements with ecosystems takes on essentially three main forms of effects on natural habitats and biocenoses present (Lindenmayer and Fisher, 2006; Jaeger *et al.*, 2007):

- spatial separation caused by linear infrastructure (road and technological networks);
- spatial disruption and suppression caused by the growth of built-up and urbanised areas;
- disturbance caused by movement, noise and lighting.

Maintaining current conditions of continuity, and even restoring and improving them, requires adequate territorial management and more targeted forms of planning than those currently adopted in Italy.

Methods and results

Based on our remarks in the introduction to this paper, we assigned the various levels of biopermeability specified using the categories set out in the European standard Corine Land Cover, Level 3 (Devillers *et al.*, 1991). The CORINE-Land Cover project has led to the creation of a database on land cover on a scale of 1:100.000. The legend is broken down into 44 items subdivided in 3 hierarchical units and refers to spatial units that are either homogenous or formed by elementary areas belonging to the same class which may be clearly distinguished from surrounding units. There are significant shortcomings in CLC data since the minimum mapping unit using this method is of 25 hectares (equivalent to a 2.8 mm circle or a 5 x 5 mm square on a scale of 1:100.000) and the minimum breadth of polygons is 100 m (1 mm on the nominal scale). Furthermore, for each of these areas, the prevailing and secondary land uses have been defined using photo-interpretation of cover based on digital orthophotographs from flight IT2000. Biopermeability levels have been identified by CORINE categories using the following criteria.

Low biopermeability at territorial scale: 111) continuous urban fabric; 112) discontinuous urban fabric; 121) industrial or commercial units; 122) road and rail networks and associated land; 124) airports; 131) mineral extraction sites; 142) sport and leisure facilities; 211) non-irrigated arable land; 212) permanently irrigated land; 213) rice fields; 222) fruit trees and berry plantations; 223) olive groves; 231) pastures; 241) annual crops associated with permanent crops.

High biopermeability at territorial scale: 242) complex cultivation patterns; 243) land principally occupied by agriculture, with significant areas of natural vegetation; 311) broad-leaved forest; 312) coniferous forest; 313) mixed forest; 321) natural grassland; 322) moors and heathland; 323) sclerophylous vegetation; 324) transitional woodland/shrub; 331) beaches; 332) bare rock, scree, cliffs, rock outcrops; 333) sparsely vegetated areas; 334) burnt areas; 335) glaciers and perpetual snow; 411) inland marshes; 422) salt marshes; 521) inland lagoons; 511) water courses, dunes, and sand plains canals and waterways; 512) water bodies; 522) estuaries; 521) lagoons.

The studied area are the Italian regions, while the starting material is the CLC land cover data updated at 2000. The entire procedure was carried out in the GIS environment and the territorial units considered in the study are the regions, as they present fairly homogenous forms of territorial governance. Starting from the initial fragmentation conditions in each region, indicated by a given number of biopermeable patches, external buffers are plotted at fixed and increasingly greater distances. Each time a buffer is plotted around all the patches and joined to them, the number of patches is reduced. This makes it possible to relate buffer distances and number of corresponding patches, until the extreme value of one patch is reached, when all the original patches are joined. It is therefore possible to plot curves relating buffer distances and number of patches (biopermeable area fragmentation reduction curves).

In the study application, buffers were plotted at the following distances in metres: 100, 200, 400, 800, 1200, 1600, 2400 and in some cases of very elevated fragmentation even 4800 metres. Using this data, we developed the fragmentation reduction curves for every Italian region, by plotting the buffer distance on the x-axis and the number of patches on the y-axis.

The curve shows that as buffer distances increase, biopermeable patches are compacted, thus increasing environmental continuity. The horizontal asymptote of the curve is represented by the y=1 line, a situation indicating the total interconnection between patches.

The operations described above have been applied to all biopermeable and forest (CLC codes 311, 312 and 313) patches, as these are environments of high and significant ecological value. Moreover, the fragmentation reduction curves were plotted, in this case, only at national level, even for Italian protected areas, clearly with greater buffer distances compared to regional biopermeability cases (Fig. 5).

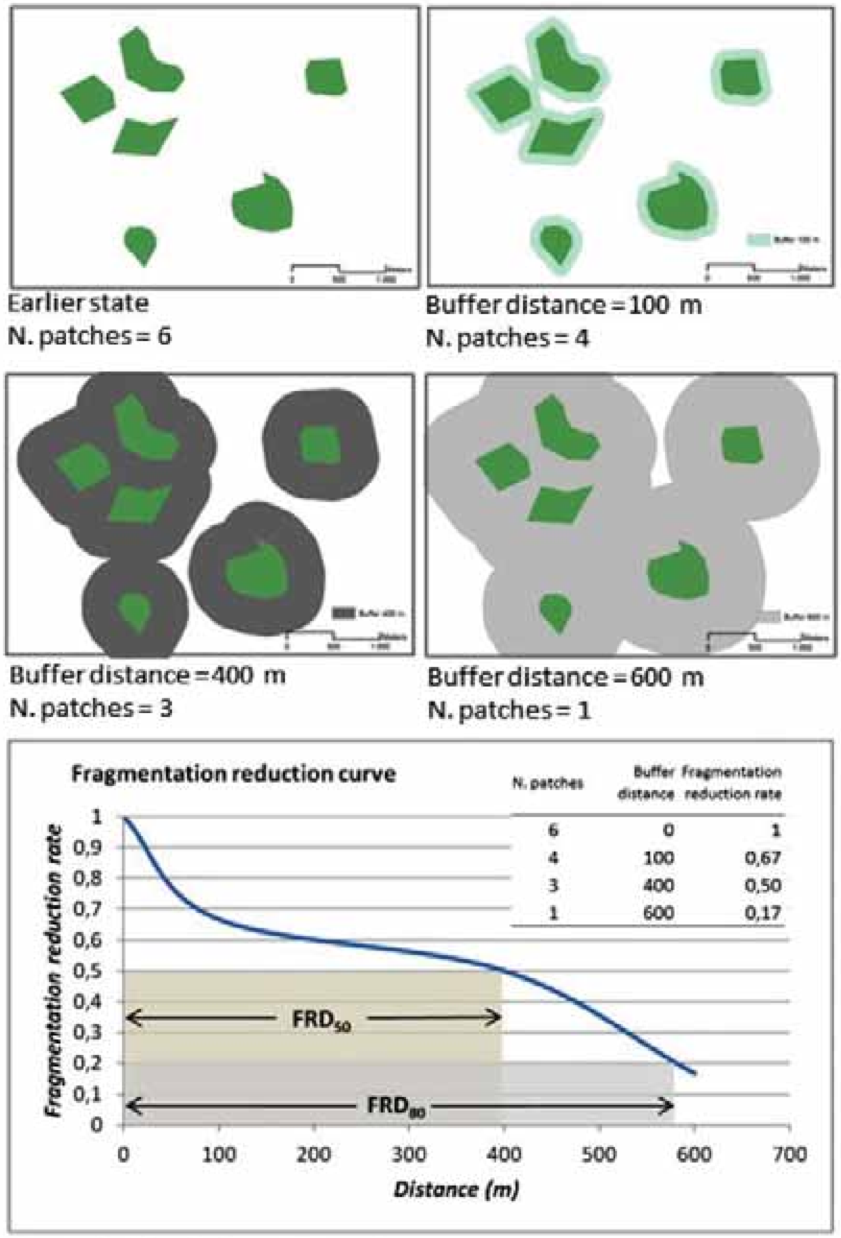

Fig. 4 – Method for calculating FRDx indices.

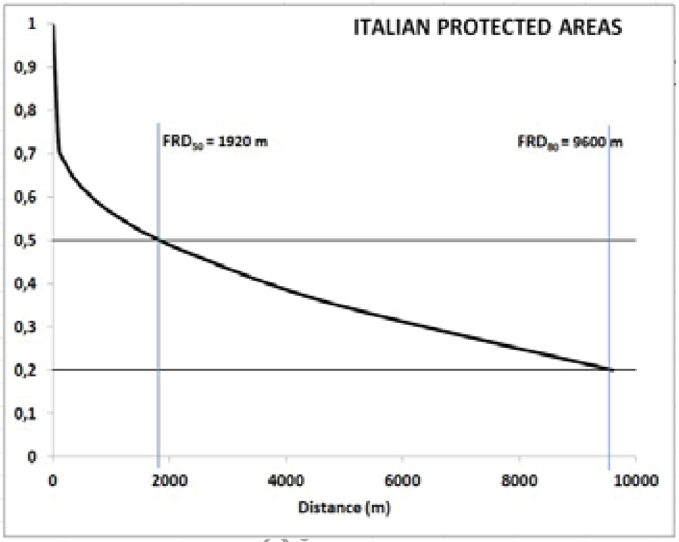

Fig. 5 – Fragmentation reduction curve and FRD indices for Italian system of protected areas.

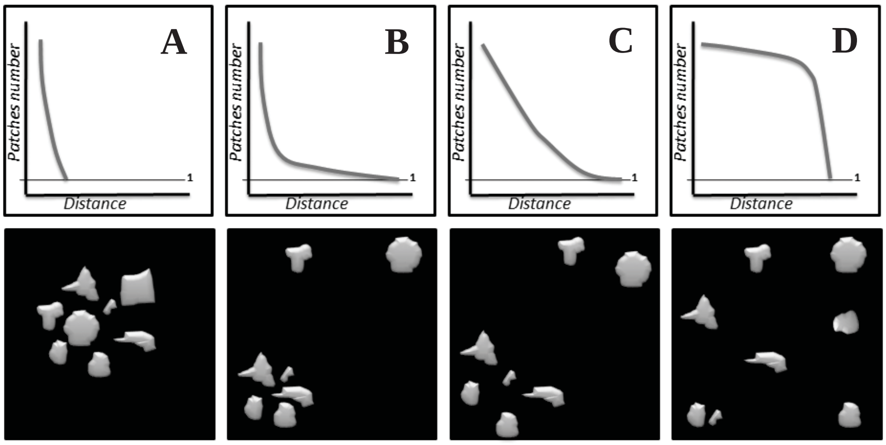

Fig. 6 – Fragmentation reduction compliance models.

The geometry of previously plotted fragmentation reduction curves makes it possible to develop 4 compliance models for Italian regions (Fig. 6). Cases A and D are the two extremes: in the example in A, it is sufficient to work on short distances to link patches that are already in a pseudo-aggregate form (Valle d'Aosta and Trentino); model D shows the example of a broadly unlinked matrix that requires actions on larger distances (Puglia, Calabria, Lazio, Sicily and Sardinia). Cases B and C are the intermediate ones and apply to the majority of Italian regions. Case B shows a situation where there is a group of patches very close to one another and others situated at greater distances (Abruzzo, Campania, Emilia-Romagna, Basilicata, Liguria, Friuli Venezia Giulia and Marche). The example shown in C describes a case in which there are a group of patches not very far from one another (scarcely disrupted environmental matrix), with other more isolated residual patches (Lombardy, Piedmont, Tuscany, Umbria, Molise and Veneto).

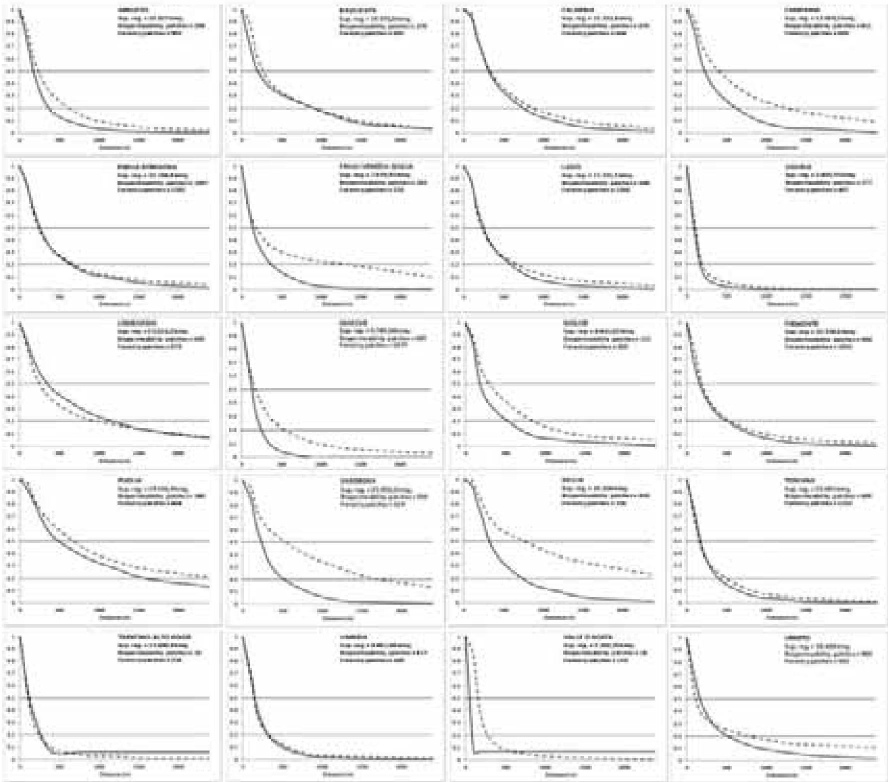

Fig. 7 – Fragmentation reduction curve of Italian regions relative to biopermeability areas (continuous line) and forestry areas (dotted line).

Using the analytical information provided by the fragmentation reduction curve, we developed an index known as Fragmentation Reduction Distance that expresses the average distance to be covered in order to reduce the existing fragmentation rate. The first of these, FRD_{50} expresses the average distance between the elements considered (biopermeable areas for

FRD_{50}, forests for FRD_{50f}) that has to be covered through spatial reconnection actions to reduce current fragmentation amounting to 50%. The FRD_{80} index (having a similar formulation to FRD_{50}) concerns the reduction of fragmentation presently amounting to 80%.

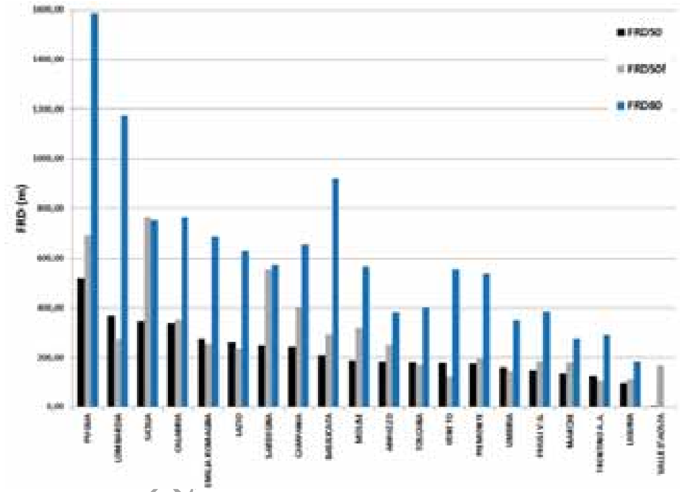

Fig. 8 – Reduction fragmentation distance for the Italian regions relative to biopermeability areas (FRD50 and FRD80) and forestry areas (FRD50f).

The indices described may play a significant role in the policies to reduce ecosystem fragmentation in order to improve environmental quality and biodiversity conservation. FRD_x indices of a few dozens of metres, and up to 150-200 m, denote the possibility of opting for defragmentation on a town planning scale, with very limited and localised actions. For higher FRD_x the reduction of fragmentation has to be tackled at planning level, but on a larger scale (General Town Planning Scheme, co-ordination plans and landscape plans). We also performed a dependence analysis on the FRD_{50} and FRD_{80} indices and some physical and demographical parameters in order to understand to what extent they were interrelated. A fairly low correlation, with R^2 parameters in both cases nearing 0.25 (Fig. 9), can be found with regional demographical density, that indicate the different conditions and land policies of the Italian regions at landscape scale.

The correlation coefficients are instead fairly high in the case of dependence of morphological variables (altitude and slope gradient). In these cases (Fig. 10) the regression analysis responds well (R^2 variable from 0,46 to 0,66) considering land percentage over 600 metres above sea-level (mountain zone limit) and land percentage with slope lower than 35% (above this percentage, land is considered non cultivable). The altitude and gradient values considered are those corresponding to regional areas generally more sensitive to settlements and therefore more determinative of the fragmentation effects that the latter have on ecosystems.

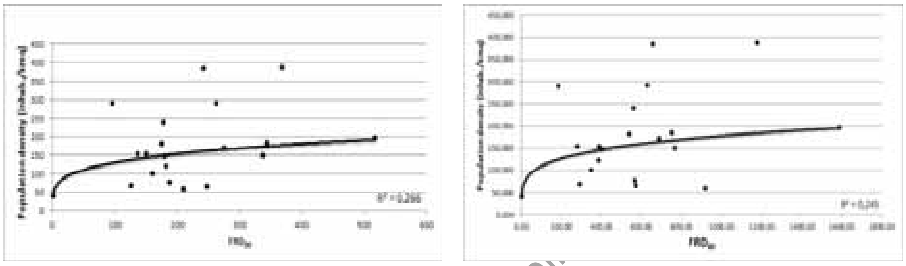

Fig. 9 – FRDx and population density: correlation analysis.

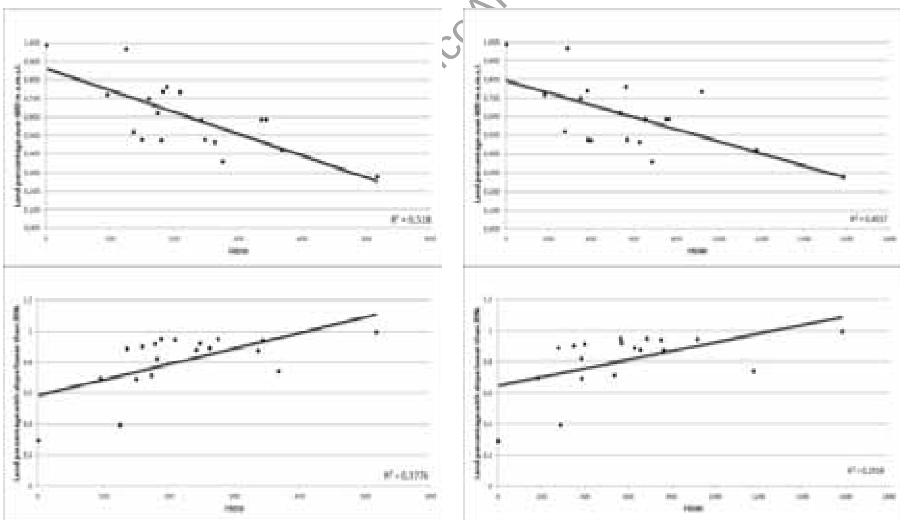

Fig. 10 – FRDx and morphological parameters: correlation analysis.

Conclusion

This is evidence of the fact that the reduction of fragmentation and biodiversity conservation require forms of governance that consider land in its entirety (urban policy approach), and not as a set of separate zones with protected natural fragments and matrices increasingly encroached upon by settlements (zoning approach) (Gambino, 1997; Romano 2011; Sargolini, 2011). This entails the conceptual overturning of conventional environmental analysis and planning techniques used until the end of the nineties. The conservation of biodiversity and the most important ecological processes (dispersion, water and nutrient cycles, carbon storage, etc.) cannot be achieved merely by safeguarding protected areas indefinitely, but rather it is essential to opt for different environmental management models which have been classified as "ecological networks" (Jongman, 1995) in the specialised literature produced in Italy starting from the mid-nineties. In the town planning tradition, uncultivated land, abandoned cultivated land, burned areas and degraded woodland have always been relegated to a position of uselessness and unavoidable pre-urbanisation. Attributing relational ecological importance to these areas today requires a thorough review of the land governance culture. The ecological network should therefore be included in the strategies related to territorial and environmental planning, conservation and ecosystem services, which are economically fundamental (Santolini, 2010). Since 2010, international year of biodiversity, various regional bills on biodiversity have been put forward, and perhaps even a national bill. These bills can certainly be used to make land protection and policies converge and maintain environmental continuity in a country like Italy which has major responsibilities towards global biodiversity.

Acknowledgements

The study was conducted within the framework of the Environmental Sciences Doctorate (25^{th} cycle) of L'Aquila University and the 2011 programme of the Observatory for Biodiversity, Rural Landscape and Sustainable Planning of Umbria Region.

References

Andreella M. and Brécciaroli B., eds. (2011), *Breve guida alla strategia nazionale per la biodiversità*, Ministero dell'Ambiente e della Tutela del Territorio e del Mare, Rome, Italy.

Batistella M., Brondizio E.S. and Moran E.F. (2000), "Comparative analysis of landscape fragmentation in Rondônia, Brazilian Amazon", *International Archives of Photogrammetry and Remote Sensing*, XXXIII, B7: 148-155.

Battisti C. (2003), "Habitat fragmentation, fauna and ecological network planning: toward a theoretical conceptual framework", *Italian Journal of Zoology*, 70, 3: 241-247.

Battisti C. and Romano B. (2007), *Frammentazione e connettività: dall'analisi ecologica alla pianificazione ambientale*, Città Studi, Milan, Italy.

Bennett A.F. (1999), *Linkages in the landscapes. The role of corridors and connectivity in wildlife conservation*, IUCN, Gland, Switzerland, and Cambridge, United Kingdom.

Bierwagen B.G. (2005), "Predicting ecological connectivity in urbanizing landscapes", *Environmental Planning B: Planning and Design*, 32, 5: 763-776.

Biondi M., Corridore G., Romano B., Tamburini P., Tetè P. (2003), "Evaluation and planning control of the ecosystem fragmentation due to urban development", *ERSA 2003 Congress*, August, Jyvaskyla, Finland.

Bissonette J.A. (1997), *Scale sensitive ecological properties: historical context, current meaning*, in Bissonette J.A., ed., *Wildlife and landscape ecology: effects of pattern and scale*, Springer-Verlag, New York, NY, United States pp. 3-15.

Bogaert J., Van Hecke P., Salvador-Van Eysenrode D., Impens I. (2000), "Landscape fragmentation assessment using a single measure", *Wildlife Society Bulletin*, 28, 4: 875-881.

Boitani L., Falcucci A., Maiorano L., Rondinini C. (2007), "Ecological networks as conceptual frameworks or operational tools in conservation", *Conservation Biology*, 21, 6: 1414-1422.

Bonnin M., Bruszik A., Delbaere B., Lethier H., Richard D., Rientjes S., van Uden G., Terry A. (2007), *The Pan-European ecological network: taking stock*, Council of Europe, Strasbourg, France.

Bressan E. and Poldini L. (2006), "Proposta di reti ecologiche sulla base della biopermeabilità vegetazionale: il caso del Friuli Venezia Giulia", *Biogeographia*, XXVII: 173-186.

Crooks K.R. and Sanjayan M. (2006), *Connectivity conservation, conservation biology*, Cambridge University Press, Cambridge, United Kingdom.

Devillers P., Devillers-Terschuren J., Ledant J.P., eds. (1991), *CORINE Biotopes manual, habitats of the European Community. A method to identify and describe consistently sites of major importance for nature conservation*, Office for Official Publications of the European Communities, Luxembourg.

Dunning J.B., Danielson B.J. and Pulliam H.R. (1992), "Ecological processes that affect populations in complex landscapes", *Oikos*, 65, 1: 169-175.

EEA (2011), *Landscape fragmentation in Europe, Joint EEA-FOEN report, EEA Report no. 2/2011*, EEA, Copenhagen, Denmark.

Fahrig L. (1997), "Relative effects of habitat loss and fragmentation on population extinction", *Journal of Wildlife Management*, 61, 3: 603-610.

Fahrig L. (2003), "Effects of habitat fragmentation on biodiversity", *Annual Review of Ecology, Evolution, and Systematics*, 34: 487-515.

Forman R.T. (1995), *Land mosaics. The ecology of landscapes and regions*, Cambridge University Press, New York, NY, United States.

Forman R.T.T., Sperling D. and Bissonette J.A. (2002), *Road ecology. Science and solutions*, Island Press, Washington DC, United States.

Gambino R. (1997), *Conservare, innovare*, UTET, Turin, Italy.

Girvetz E.H., Thorne J.H., Berry A.M., Jaeger J.A.G. (2008), "Integration of landscape fragmentation analysis into regional planning: a statewide multi-scale case study from California, USA", *Landscape and Urban Planning*, 86, 3-4: 205-218.

Gustafson E.J. and Parker G.R. (1992), "Relationships between landcover proportion and indices of landscape spatial pattern", *Landscape Ecology*, 7, 1: 101-110.

Haines-Young R. and Chopping M. (1996), "Quantifying landscape structure: a review of landscape indices and their application to forested landscapes", *Progress in Physical Geography*, 20, 4: 418-445.

Hanski I. (2005), "Landscape fragmentation, biodiversity loss and the societal response", *EMBO reports*, 6, 5: 388-392.

Jaarsma C.F. (1997), "Approaches for the planning of rural road networks according to sustainable land use planning", *Landscape and Urban Planning*, 39, 1: 47-54.

Jaeger J.A.G. (2000), "Landscape division, splitting index, and effective mesh size: New measures of landscape fragmentation", *Landscape Ecology*, 15, 2: 115-130.

Jaeger J.A.G., Schwarz-von Raumer H.G., Esswein H., Müller M., Schmidt-Lüttmann M. (2007), "Time series of landscape fragmentation caused by transportation infrastructure and urban development: a case study from Baden-Württemberg, Germany", *Ecology and Society*, 12, 1: 22.

Jaeger J.A.G., Bertiller R., Schwick C., Cavens D., Kienast F. (2010), "Urban permeation of landscapes and sprawl per capita: New measures of urban sprawl", *Ecological Indicators*, 10, 2: 427-441.

Jongman R.H.G. (1995), "Nature conservation planning in Europe, developing ecological networks", *Landscape and Urban Planning*, 32, 3: 169-183.

La Rovere M., Battisti C. and Romano B. (2006), "Integrazione dei parametri eco-biogeografici negli strumenti di pianificazione territoriale", *AISRE, Atti XXVII Conferenza Italiana di Scienze Regionali*, October 12-14, Pisa, Italy.

Lindenmayer D.B. and Fisher J. (2006), *Habitat fragmentation and landscape change. An ecological and conservation synthesis*, Island Press, Washington DC, United States.

McGarigal K. and Cushman S.A. (2002), "Comparative evaluation of experimental approaches to the study of habitat fragmentation studies", *Ecological Applications*, 12, 2: 335-345.

Moser B., Jaeger J.A.G., Tappeiner U., Tasser E., Eiselt B. (2007), "Modification of the effective mesh size for measuring landscape fragmentation to solve the boundary problem", *Landscape Ecology*, 22, 3: 447-459.

Opdam P., Steingrıver E. and van Rooij S. (2006), "Ecological networks: a spatial concept for multi-actor planning of sustainable landscapes", *Landscape and Urban Planning*, 75, 3-4: 322-332.

Pungetti G. and Romano B. (2004), *Planning the future landscapes between nature* *and culture*, in Jongman R. and Pungetti G., eds., *Ecological networks and greenways*, Cambridge University Press, Cambridge, United Kingdom, pp. 107-127.

Razin E. and Rosentraub M. (2000), "Are fragmentation and sprawl interlinked? North American evidence", *Urban Affairs Review*, 35, 6: 821-836.

Romano B. (1996), *Oltre i parchi, la rete verde regionale*, Andromeda, Teramo, Italy.

Romano B. (2011), "Le reti ecologiche tra "zoning approach" e "urban policy approach"", *Territorio*, 58: 27-35.

Romano B., Zullo F., Cargini M., Febo D., Iezzi C., Mazzola M., Rollo P. (2011), "Gli stati e le dinamiche dei processi insediativi e infrastrutturali di trasformazione dei suoli in Italia", *Ri-Vista ricerche per la progettazione del paesaggio*, 15-16: 65-77.

Santolini R. (2010), "Servizi ecosistemici e sostenibilità", *Ecoscienza*, 3: 20-23.

Sargolini M. (2011), "Scomparsa dell'agricoltura e sprawl insediativo: la ricerca di nuovi equilibri naturali", *PRISMA Economia – Società – Lavoro*, III, 1: 50-54.

Schumaker N.H. (1996), "Using landscape indices to predict habitat connectivity", *Ecology*, 77, 4: 1210-1225.

Selman P. (2006), *Planning at the landscape scale*, Routledge, Abingdon, United Kingdom.

Taylor P.D., Fahrig L., Henein K., Merriam G. (1993), "Connectivity is a vital element of landscape structure", *Oikos*, 68, 3: 571-573.

Vannuccini M. and Geri F. (2006), "Valutazione di connettività ambientale attraverso analisi di cost distance su specie target", *Atti XVI Congresso Nazionale SITE, Società Italiana di Ecologia*, pp. 164-165.

Vuilleumier S. and Prélaz-Droux R. (2002), "Map of ecological networks for landscape planning", *Landscape Urban Planning*, 58, 2-4: 157-170.

World Resources Institute, IUCN, UNEP (1992), *Global biodiversity strategy*, WRI, Washington DC, United States.