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Assessment of ecological conditions of permanent meadows in the Italian Alps: loss, biodiversity and remote sensing change detection

PhD thesis

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Abstract

The monitoring of ecological condition of grasslands ecosystems in the European Alps is a main issue for mountain regions, since the abandonment of traditional and sustainable management practices has exposed grassland habitat to significant impacts in a context of global environmental change.

The present research project was focused in assessment of the state of permanent meadows in the lowlands of Valtellina Valley (80 km^2), Italian Alps, during the timeframe 1980-2000. In specific, it quantified the land use/land cover changes and identified main drivers behind permanent meadows loss; characterized the relationship between biodiversity in the meadows and the spatial-environmental conditions in the landscape and by last evaluated the use of satellite remote sensing data for fast change detection in landscape. To achieve such aims, the research project was organized in three different approaches presented in the four chapters of this thesis.

Concerning the quantification of the land use/land cover and identification of main drivers behind permanent meadows loss, the results show a strong decrease in meadows (-18.5%) in a context of agricultural land decrease and human settlements increase. This was the land cover type with highest loss and conversion rate during the study period. Meadows were converted to human settlements (urban, industrial and roads), other agriculture uses (cultivation, orchard, vineyard), bushland and uncultivated land. Meadows loss occurred mainly in soils with good land capability, low slope, exposed to south and in proximity of roads, urban settlements and bushland. Densities of urban, industrial and bushland and land capability were the only significant drivers for meadows loss, while distance to meadow edge, meadows density, distance to roads and soil degradation were the only significant drivers for meadows preservation.

Concerning the characterization of the relationship between biodiversity in the meadows and the spatial-environmental conditions in the landscape, the results evidenced that species richness and Shannon indices were best explained by regressive models including changes occurred in spatial environmental heterogeneity from 1980 to 2000. Species richness was negatively related to strong decrease in meadows habitat area and recent urban area, while Shannon index was positively related to the increase in landscape diversity. In contrast, species evenness was better explained by regressive model including recent spatial environmental heterogeneity and positively related to increase in

eastness in the study area, and negatively affected both by the area of woody and soil pH (KCl).
Concerning the evaluation of the use of satellite remotely sensing data for land cover mapping and change detection in landscape, the results show that the hybrid approach for land cover classification based of Landsat imagery was highly accurate. Image differencing is the technique which best detect changes in landscape as well as in urban, meadow and bush land. The accuracy of change detection was moderate.
This thesis concludes that the conflict by land in locations densely occupied by other land cover types with good land capability is the major threat to meadows and avoidance of fragmentation may be a good strategy for its preservation. The meadows habitat needs a well-designed landscape and farming planning, which should account the economic value of the ecosystem services provided by this habitat. In addition, to conserve plant diversity in meadows it is necessary to avoid loss of meadows habitat, maintain landscape diversity and execute a sustainable meadow management.
Remotely sensed imagery can be a reliable source of information for alps, although particular attention should be made to the image pre-processing and classification, as well as, to minimize topography effects in spectral information.

Introduction

Permanent grassland and meadow is land used permanently to grow herbaceous forage crops, through sown or naturally (self-seeded). It can be either used for grazing by livestock, or mowed for hay or silage (stocking in a silo) (Eurostat, 2010). These grasslands, which are in decline since the 1950's, occupied a larger extent of landscape in European Alps. The monitoring of this ecosystem is nowadays essential, since the spatial and temporal dynamics in structure and function affected the overall of good and services provided by this ecosystems (provisioning services, supporting services, regulating services and cultural services) and changes in this ecosystems supplied change in the ecological systems (Peters et al., 2006). Nowadays, a major issue of this monitoring is the estimation of loss in this ecosystem worldwide due to alterations in land use/land cover (LUCC). Such alteration affected status and produced severe impacts: loss of biodiversity (Niedrist, 2008), encroachment of shrubs and forest (Tasser et al., 2007), decrease of the forage production (Liu et al., 2006), altered water cycle (Mingliang et al., 2008), soil degradation (Snyman and du Preez, 2005), flood events (Flez and Lahousse, 2004) and desertification (Yong-Zhong et al., 2005). In the European Alps, loss of grasslands is a major multidimensional issue (e.g. environmental, economical and social impacts) following the decline in agricultural activities (Gellrich et al., 2007, Hersperger and Bürgi, 2009), and the Common Agricultural Policy (CAP) of European Union and World Trade Organization (WTO) negotiations, which promoted intense farming practices and liberalization of markets (Kristensen et al., 2004). These processes affected plant and animal diversity (Bolli et al., 2007, Kampmann et al., 2008, Sergio and Pedrini, 2008), increased floods (Ranzi et al., 1999) and decreased soil respiration, evapotranspiration and water use efficiency (Tappeiner and Cernusca, 1998).
In between the most threat ecosystem services in these context of loss of grasslands is biodiversity. Grasslands habitat are among the most diverse plant communities in Europe and are important habitats for a large number of rare plants and animals (Billeter et al., 2008). Biodiversity was affected by the abandonment of traditional, sustainable management practices, the decrease and fragmentation of grassland habitats and the increasing of the remaining grassland habitats during the last century (Krauss et al., 2004). Therefore, across the landscape an intriguing question for biodiversity conservation is the relationship between different plant diversity measures and the surrounding environment. In particular, it is crucial to understand the role of distribution of several elements, such as nutrients and climate, as well as the composition and

the spatial configuration, and how it influences plant species diversity (Stohlgren et al., 1998, Rosenzweig, 1995). Such evaluation would consider the relationships between plant diversity measures and the recent, past and changes in the surrounding environment.
The increasing conscious of both scientific knowledge and society for the fact that environmental and socio-economical modifications affected the ecosystem goods and services provided by agricultural practices, but also the internal regulation of ecosystems have forced the update of traditional monitoring systems. Remote sensing technology and satellite data emerged as instruments to continuously assess changes in ecological conditions in reasonable time-costs terms. They provide reliable data and methodologies at multiple spatial and temporal scales (Kerr and Ostrovsky, 2003), which can support land management policies (Boschetti et al., 2007, Fava et al., 2010). This feature is particular important for mountain regions in the European Alps, which occupied a considerable geographical extent and only a small portion being inhabited. Other important technology are the Geographical Information Systems (GIS), a useful platform of analysis due to its capacity to integrate information from different sources. They can support the quantification of changes in land use/ land cover and the development of appropriate models to indentify key drivers and to model land use/ land cover changes more dynamically (Gellrich et al., 2007, Rutherford, 2007). These models integrated characteristics from the surrounding landscape and take also into account the most influential processes (e.g. urban density) in order to model the relationships between land cover change and drivers originated from historical data (Hu and Lo, 2007, Linkie et al., 2004).

Context and aims

The present thesis was developed in the context of the project CAPPA (Agroecological characterization of permanent meadows in a Alpine district). The general goal of the CAPPA project was to characterize the permanent meadows ecosystems and to improve monitoring methodologies integrating remote sensing techniques, GIS, ecological modelling and traditional agronomical measurements.
This thesis in specific was focused in the assessment of ecological state of permanent meadows in the lowlands of Valtellina Valley, an Italian region in the Southern Alps, during the timeframe 1980-2000. The research developed aimed to: (i) quantify changes in the spatial extent of permanent meadows and

identify main factors behind this changes (ii) characterize the relationship between biodiversity in the meadows and the spatial-environmental conditions in the landscape (iii) evaluate the use of satellite remote sensing data for mapping and change detection over the landscape.

Structure of the thesis

This thesis was structured in four different chapters, which can be read separately, since each chapter was developed as single manuscripts. The study in Chapter 1 analysed the aim (i) of this research. It was focused in the loss of permanent meadows in the lowlands of Valtellina, which is the heart and remaining refugee for local farming systems after abandonment in uplands, and also the locations most susceptible to anthropogenic pressure. In specific, change detection analysis were executed to estimate meadows loss and spatial bivariate analysis and GIS-based logistic regression model analysed the spatial environmental drivers behind meadows loss in the period 1980-2000. The study in Chapter 2 analysed the aim (ii) of this research. The relationships between different plant diversity indices estimated for 34 plot surveys located in the permanent meadows and recent spatial environmental heterogeneity were analyzed, as well as change in spatial environmental heterogeneity over the landscape from year 1980 to 2000. The study in Chapter 3 analysed the aim (iii) of this research. The potential of Landsat imagery for land cover mapping and change detection techniques to discriminate changes occurred over the landscape in the lowlands of Valtellina was evaluated. An hybrid supervised classification was tested for mapping. The difference in NDVI (Normalized difference vegetation index) between two Landsat scenes (TM 31/08/1989) and (ETM+ 21/06/2001) was used as approach to assess the performance of image differencing and image regression change detection techniques. The study in Chapter 4 was a synthesis of two ecological applications developed to quantify land cover change and to monitor recent variation in permanent meadows net primary production (NPP) in the Valtellina Valley. It was based in aerial photogrammetry and MODIS satellite remote sensing data and radiation use efficiency (RUE) models.

Chapter 1

Assessment of land cover changes and spatial drivers behind loss of permanent meadows in the lowlands of Italian Alps

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Abstract

The loss of permanent meadows in the lowlands of the European Alps due to land use/ land cover changes is a major underestimated process, which affects the status of these habitats and their provision of ecosystem services. In the Italian Valtellina valley (80 km2) change detection analysis estimated meadows loss and spatial bivariate analysis and GIS-based logistic regression model analysed the spatial environmental drivers behind meadows loss in the period 1980-2000. A strong decrease in meadows (-18.5%) was found, in a context of agricultural land decrease and human settlements increase. This was the land cover type with highest loss and conversion rate during the study period. Meadows were converted to human settlements (urban, industrial and roads), other agriculture uses (cultivation, orchard, vineyard), bushland and uncultivated land. Meadows loss occurred mainly in soils with good land capability, low slope, exposed to south and in proximity of roads, urban settlements and bushland. Densities of urban, industrial and bushland and land capability were the only significant drivers for meadows loss, while distance to meadow edge, density of meadow, distance to roads and soil degradation were the only significant drivers for meadows preservation. The conflict by land in locations densely occupied by other land cover types with good land capability

is the major threat to meadows and avoidance of fragmentation may be a good strategy for its preservation. The meadows habitat needs a well-designed landscape and farming planning, which should account the economic value of the ecosystem services provided by this habitat.

Keywords: Land cover/land use changes; Meadows loss; GIS-based logistic regression; Aerial photographs; Italian Alps

1. Introduction

Loss of grasslands due to alterations in land use/land cover (LUCC) affected the status of this ecosystems worldwide, resulting in loss of biodiversity (Niedrist, 2008), encroachment of shrubs and forest (Tasser et al., 2007), decrease of the forage production (Liu et al., 2006), altered water cycle (Mingliang et al., 2008), soil degradation (Snyman and du Preez, 2005), flood events (Flez and Lahousse, 2004) and desertification (Yong-Zhong et al., 2005). In the European Alps, loss of grasslands is a major multidimensional issue (e.g. environmental, economical and social impacts) documented since the 1950s, following the decline in agricultural activities (Gellrich et al., 2007, Hersperger and Bürgi, 2009), and the Common Agricultural Policy (CAP) of European Union and World Trade Organization (WTO) negotiations, which promoted intense farming practices and liberalization of markets (Kristensen et al., 2004). This loss in grasslands threats centuries of traditional land use and also its ecologically and economically relevant values (biodiversity, water conservation, forage production, dairy products, aesthetic tourism attractiveness) in a context of global environmental changes and increasing need for food supply (Ceballos et al., 2010).

Three major LUCC processes have been shown to reduce grasslands: (1) abandonment or extensive land use in the uplands with bush encroachment and marginalisation with transformation of meadows into pastures in steep slopes (Fischer and Wipf, 2002), (2) intensification in the lowlands (agro-industry) and (3) human settlements expansion into agricultural land (Marini et al., 2007, Pimm and Raven, 2000, Sergio and Pedrini, 2008). Recent studies have shown that these processes affected plant and animal diversity (Bolli et al., 2007, Kampmann et al., 2008, Sergio and Pedrini, 2008), increased floods (Ranzi et al., 1999) and decreased soil respiration, evapotranspiration and water use efficiency (Tappeiner and Cernusca, 1998). It has also been shown that rates of land cover change vary according to landscape features (Falcucci et al., 2007, Schneeberger et al., 2007). In particular, conversions and loss in grasslands can be explained largely by topographic features (e.g. slope), soil related conditions (e.g. land capability) and local neighbourhood attributes related to proximity factors (e.g. distance to urban) (Gellrich et al., 2007, Hersperger and Bürgi, 2009, Rutherford et al., 2008).

However, while these studies documented alpine and sub-alpine regions, few studies addressed the loss in grasslands in the lowlands of mountain regions, which are the heart and last refugee for local farming systems after abandonment in uplands, and the locations most susceptible to changes due to anthropogenic pressure. In addition, in Italy, a country bounded by the Alps, these evaluations are rare and a quantification of grasslands loss is missing. This study assessed and modelled the conversion and loss of meadows in the lowlands of Valtellina, an Italian region in the Southern Alps, in order to better understand LUCC in lowlands and contribute to the creation of guidelines for the preservation and the sustainable management of this key habitat for the European Alps. In specific, for the period 1980-2000, it aims to analyse the following facets: (1) mapping of land cover for the study area (80 km2) to quantify changes and transitions in different land cover classes, focusing in meadows (2) analysis of the spatial pattern of meadows loss according to spatial environmental features (3) development of a GIS-based logistic regression model to identify significant drivers for meadows loss.

2. Material and methods

2.1. Study area

The study area (80 km2) is located in Italy, in the lowlands of middle Valtellina, Southern Alps (46.10' N, 9.50 E; Fig. 1). The elevation ranges from 250 to 750 m a.s.l. (above sea level) and slope rarely exceeds 5%. It is a U-shaped valley carved by glacial erosion during the Quaternary, with a west-east orientation, discharged by the river Adda and surrounded by high elevation mountains (> 3000 m a.s.l.). Main soil types are Eutric fluvisols, Dystric and Eutric cambisols (FAO, 1988). Owing to the proximity to Lake Como, the climate is temperate continental with mean annual temperature of 11.9°C and precipitation of 970 mm (Sondrio at 298 m a.s.l., 1973-2007). The landscape is composed by five main categories: meadow, urban settlements, vineyard, bushland and orchard. It is strongly influenced by local farming activities, including activities in uplands (Rudini, 1992). The meadows are permanent, mown 3-4 times per year for hay making (Fava et al., 2010) and subject to intensive farming practices (e.g. dung deposition, mechanical farming operations) since the increasing abandonment of grasslands at higher elevation and concentration of farming activities in lowlands from the 1960s. According to the Habitats Directive, the meadows belong to Ahrrenatherion elatioris and Polygono-Trisetion alliances (EU habitat codes 6510 and 6520). The use of these meadows still signifies an important income source for local farmers and economy, as well as an important platform for Valtellina Valley tourism.

Fig.1 Location of the study area in the Italian Valtellina valley, Southern Alps.

2.2. *Land cover mapping and detection of changes in meadows*

To determine land cover changes over time and loss in meadows at Valtellina, a visual classification with hand digitizing of two digital colour ortho-rectified aerial photographs (27/08/1980, scale 1:20000 and 01/09/2000, scale 1: 36000) was made using ArcGIS 9.1 software (ESRI, 2005). The aerial photographs were produced by the Region of Lombardia, registered and geo-referenced to the national grid system Gauss Boaga-Zone 1. The landscape was classified into 11 land cover classes (meadow, cultivation, orchard, vineyard, bushland, uncultivated, water, urban, industry, roads, hedges) adapting the European Nature Information System (EUNIS) habitat classification (http://eunis.eea.europa.eu/) for the study area. The land cover maps (vector format) were converted into grid format (10 m spatial resolution) and land cover change was quantified through the cross-tabulation algorithm, a post-classification change detection technique that performs a cross-correlation between independent classified images (Lu et al., 2004). The cross-tabulation table showed the frequencies of classes, which had remained the same (frequencies in the diagonal) or had changed (off-diagonal frequencies) (Shalaby and Tateishi, 2007). Meadows loss was located calculating a difference image (later image minus earlier image) with the differencing algorithm (Singh, 1989). The change detection operations were executed in IDRISI Andes 15.0 (Eastman, 2006).

2.3. *Spatial bivariate analysis of meadows loss according to spatial environmental features*

Spatial bivariate analysis quantified the degree of meadows loss respect to the variation of 11 auxiliary variables selected from the literature based on their expected influence in this process (Gellrich et al., 2007, Hersperger and Bürgi, 2009, Rutherford, 2007, Rutherford et al., 2008). They were grouped into 3 thematic fields: topography-related, soil-related and neighbourhood-related and are described in detail in Table 1. The variables elevation, slope and aspect measured the effect of temperature, diffuse solar radiation, landscape shape and exposure and were derived from the digital elevation model (DEM), resampled from 20 m to 10 m, to adjust spatial resolution. Soil type and land capability (likelihood for agriculture use) measured the influence of soil features, they were obtained from the geo-environmental database of Region of Lombardia (http://www.cartografia.regione.lombardia.it/geoportale/ptk), converted from vector to grid format to adjust data format. The variables measuring distance to other particular land cover type (e.g. distance to urban settlements) or landscape feature (e.g. distance to meadow edge) evaluated the effect of proximity to meadows loss. In particular, the variable distance to potential flood areas represents the distance to areas under high flood risk. These distance variables were defined using land cover data from 1980 and estimated through a two-step process: first, delimitation of edges (5 x 5 pixel moving window), and second, creation of neighbourhood distance images respect to edges, through Euclidean distance function. Spatial bivariate analysis was performed through spatial overlay (intersections) between the layer "meadows loss" and each auxiliary variable layer. The layer "meadows loss" was a binary layer (1 versus 0), where "1" indicated presence of meadows loss and "0" absence of meadows loss. All data layers were in grid format (10 m spatial resolution). This analysis describes the relationship between meadows loss and each auxiliary variable and supported the decision of which variables to integrate for logistic regression analysis in the next section.

2.4. GIS-based logistic regression modelling of meadows loss

A GIS-based logistic regression model was used to indentify the significant factors for meadows loss, since it is an effective technique for the analysis of land use/land cover transitions when the dependent variable is binary, which is the case (Luo and Wei, 2009, Rutherford, 2007, Verburg et al., 2004). The dependent variable was the binary layer meadows loss described in section 2.3. The set of independent variables was composed by 17 variable layers: the set of variables used in section 2.3 and 6 new variables integrated in the neighbourhood-related and soil-related groups of variables, as described in Table 1. The variables meadows density, urban density, industrial density and bushland density were included due to their influence in LUCC within agricultural lands (Rutherford et al., 2008) and in order to measure the effect of density features on meadows loss. These variables were defined using land cover data from 1980 and measuring the class frequency in a total of 21 kernel positions, by using a 5 x 5 pixel moving window. The variable distance to streams corresponds to the Euclidean distance to water lines and evaluated its proximity effect, an important feature for agriculture practices (e.g. crops irrigation). The variable soil degradation measured the effect of areas under soil instability (soil erosion and sediments deposition), which was expected to avoid human pressure, it was obtained from the geo-environmental database of Region of Lombardia and converted to grid format.

In order to allow logistic regression analysis, the circular variable aspect, represented in degrees, was transformed to cosine aspect and sine aspect, so that it was represented as a continuous variable with output values respect to North and East (Piedallu and Gégout, 2008). The distance variables (e.g. distance to roads) were transformed to the natural log (ln) and the categorical variables (e.g. soil degradation) were transformed with evidence likelihood algorithm (Eastman, 2006), in order to attain normal distribution at different measuring scales. The set of independent variables was tested for multicollinearity (Rawlings et al., 2001), which excluded from analysis the variables elevation, soil type and distance to potential flood areas.

The GIS-based logistic regression model was estimated by maximum-likelihood algorithm. This model was recently applied in biodiversity (Gontier et al., 2010), landslides (Kamp et al., 2008) and fire regimes studies (Mermoz et al., 2005). The relationship between the dependent variable and the independent variables follows a logistic curve, where p is the dependent variable representing the probability that Y=1, x1, x2, x3,…, xn are the independent variables, b1, b2, b3,…, bn the estimated parameters coefficients, which indicate the change in Y for a one unit increase on the corresponding independent variables. The intercept is a, representing the value of Y when the values of the independent variables are zero, as shown in Eq.1. By performing a logit transformation of the equation, the model is linearised and the dependent variable of the regression is continuous in the range 0-1. Logistic regression on pixels of the variable layers resulted in a map of real scores between 0 and 1, indicating the probability meadows loss in the study area (Clark, 1986). The quality and fit of the model was assessed by Chi-square values and by ROC (Relative Operating Characteristic) (Pontius and Schneider, 2001). ROC compared the probability for meadows loss against the meadows loss layer and value ranges between 0.5 (completely random) and 1 (perfect fit).

Model spatial autocorrelation was assessed calculating Moran's I of the residuals.
Initially, a preliminary model with the entire dataset was ran, but the output presented a highly positive spatial autocorrelation in the residual image (Moran's I = 0.72), violating the logistic regression assumption of independence between observations and residuals (Hu and Lo, 2007). In order to minimise autocorrelation, data aggregation and pixel thinning functions, with a contraction factor of 10, were applied to the data layers (Hu and Lo, 2007). Subsequently, from the dependent data layer (100 m spatial resolution), 290 pixel observations were selected through stratified random sampling, with equal sampling to 1 and 0 observations. Observations for each of the independent variables were obtained through spatial overlay. All the operations for the GIS-based logistic model were executed in IDRISI Andes 15.0 (Eastman, 2006).

Eq. 1 $$\mathrm{logit}(p) = \ln(p/(1-p)) = a + b1 * x1 + b2 * x2 + b3 * x3 + \ldots + bn * xn$$

Table 1 List of variables included in the spatial bivariate analysis and GIS-based logistic regression in order to characterize meadows loss in the Vatellina valley.

Variable description				
Topography related	Unit	Proxy for	Source[a]	Task level[b]
Slope	°	Diffuse solar radiation, likelihood for meadows loss	DEM20	SBA and GIS-LR
Aspect *	°	Exposure	DEM20	SBA and GIS-LR
Elevation	m	Mean annual air temperature	DEM20	SBA and GIS-LR
Soil related				
Land capability	index	Likelihood for agriculture use	BICGA87	SBA and GIS-LR
Soil type	type	Soil quality	BICGA87	SBA and GIS-LR
Soil degradation	type	Likelihood for soil hazard	BICGA87	GIS-LR
Neighbourhood related				
Urban density	number/21	Urban structure	Lc80	GIS-LR
Industrial density	number/21	Industrial structure	Lc80	GIS-LR
Meadows density	number/21	Meadows structure	Lc80	GIS-LR
Bushland density	number/21	Bushland structure	Lc80	GIS-LR
Distance to urban settlements	m	Acessibility	Lc80	SBA and GIS-LR
Distance to roads	m	Acessibility	Lc80	SBA and GIS-LR
Distance to industrial settlements	m	Acessibility	Lc80	SBA and GIS-LR
Distance to bushland areas	m	Likelihood of abandonment	Lc80	SBA and GIS-LR
Distance to potencial flood areas	m	Likelihood of flood hazard	BICGA87	SBA and GIS-LR
Distance to meadow edge	m	Likelihood for for meadows loss	Lc80	SBA and GIS-LR
Distance to streams	m	Likelihood of flood hazard	Lc80	GIS-LR

a: DEM (digital elevation model); BICGA87 (Lombardia region-Geo-environmental cartographic database 1987); Lc80 (land cover 1980)
b: Analysis where the variable was included: SBA (spatial bivariate analysis); GIS-LR (GIS-based logistic regression)
* Variable represented by cosine aspect and sine aspect in the GIS-LR

3. Results

3.1. Land cover change in the lowlands of Valtellina valley

The land cover analysis revealed a strong decrease in meadows within the study period 1980-2000. In 1980, the landscape was largely covered by agricultural land (65.8%), mainly meadows (36.4%) and vineyards (17.4%) (Table 2). Human settlements (urban, industrial and roads) covered 17.1% and bushland 10.7%. Twenty years later (2000), the overall composition of the landscape changed, it was less agricultural addressed (-10.5%), more human settled (+6.3%) and the presence of bushland increased (+2.7%). The transition matrix evidenced that 23.5% of the study area was under LUCC with a mean velocity of change of 94.0 ha year-1. The class meadow was the most affected, from 1980 (2915.9 ha) to 2000 (2377.0 ha), the proportion of meadows to total landscape cover decreased from 36.4% to 29.7% and the total area significantly decreased 18.5 % (-538.8 ha) at a mean velocity of 26.9 ha year-1. The conversions from or to meadows represented 59.0% of the overall converted area in the lowlands. The transitions from meadows were mainly to human settlements (urban, industrial and roads; 35.6%), other agriculture uses (cultivation, orchard, vineyard; 33.7%), bushland (18.6%) and uncultivated (10.9%). Conversions from meadows to urban represented 55.4% of the total of urbanized land (+333.1 ha), conversions of meadows to industrial 64.9% of the total of new industrial settlements (+139.8 ha), conversions of meadows to bushland 43.4% of the total increase in bushland (+214.6 ha) and conversions of meadows to uncultivated land 61.6% of the total increase in uncultivated land (+145.8 ha). Conversion from meadows to other agriculture uses were more dynamic, main transitions were to cultivation (194.3 ha) and orchard (65.6 ha), however, conversion from other agriculture uses to meadows also occurred, representing 84.1% of the total area lost from meadows to other agriculture uses, a process that is in contrast with other land cover transitions and which contributes substantially to reduce the overall meadows retreat. On the contrary, 96.9% of the land converted to human settlements was definitively lost. Besides the loss of meadows, important is also the decrease in vineyards, indicating that sectors or agricultural practices with substantial work load were subjected to largest changes. Conversion from water areas into bushland (17.0 ha) and uncultivated (12.8 ha) were due to bush species colonization and hydraulic intervention on water bodies after the catastrophic landslide events in 1987.

3.2. Characterization of meadows loss according to spatial environmental features

During the study period of 20 years 824 ha of hay meadows were converted into other land cover types (Fig. 2). Spatial bivariate statistics showed that topography related variables explained to a high extent the location and the intensity of the meadows loss. Meadows disappeared when located at lower elevation (≤ 350 m; 66.7%), with a low slope gradient (≤5°; 69.8%) and mainly exposed to South (52.6%) or North (32.8%), given the west-east orientation of the valley (Fig. 3). Neighbourhood related variables evidenced that meadows loss was more pronounced in locations close to the edge of meadows (≤50 m; 89.9%), near to potential flood areas (≤200 m; 59.7%), in proximity of roads (≤200 m; 72.9%), urban settlements (≤200 m; 73.2%) and areas encroached by bushland structures (≤200 m; 66.5%) (Fig. 3). Proximity to industrial settlements had a less clear effect on meadows loss (≤300 m; 48.8%; Fig. 3). Soil related variables showed that meadows loss was higher in fluvisols (88.4%) and in soils with good land capability for agriculture use (62.7%).

Table 2 Land cover change matrix from 1980 to 2000 (in ha).

1980 / 2000→ ↓	Meadow	Cultivation	Orchard	Vineyard	Bushland	Uncultivated	Water	Urban	Industry	Roads	Hedges	Total 1980	Total 1980 (%)
Meadow	2092.3	194.3	65.6	17.5	153.1	89.8	5.6	184.5	90.7	18.1	4.3	2915.9	36.4
Cultivation	108.6	140.5	3.9	3.0	21.0	32.0	1.5	20.4	16.1	3.2	0.9	350.9	4.4
Orchard	60.8	9.8	436.9	11.1	17.0	6.3	0.0	60.7	6.2	1.8	0.1	610.8	7.6
Vineyard	63.8	18.9	41.5	1069.6	90.5	48.7	0.0	60.5	0.2	2.4	0.1	1396.2	17.4
Bushland	21.2	8.5	2.1	16.0	722.2	47.3	8.6	12.1	19.4	2.5	0.7	860.5	10.7
Uncultivated	18.1	3.6	0.6	1.7	42.5	66.4	3.5	8.8	16.1	3.1	0.1	164.4	2.1
Water	1.7	0.3	0.2	0.0	17.0	12.8	297.5	0.5	2.0	0.9	0.6	333.4	4.2
Urban	6.0	0.6	1.2	1.3	4.6	1.0	0.2	948.7	8.8	6.3	0.1	978.8	12.2
Industry	1.9	0.6	0.0	0.0	4.0	4.9	2.8	5.2	149.6	1.6	0.1	170.7	2.1
Roads	1.4	0.4	0.5	0.2	1.2	0.7	0.1	10.4	1.3	204.6	0.5	221.1	2.8
Hedges	1.4	0.6	0.2	0.0	1.9	0.4	0.0	0.0	0.0	0.0	1.8	6.4	0.1
Total 2000	2377.0	377.9	552.7	1120.2	1075.1	310.2	319.8	1311.9	310.5	244.5	9.3	8009.1	-
Total 2000 (%)	29.7	4.7	6.9	14.0	13.4	3.9	4.0	16.4	3.9	3.1	0.1	-	100

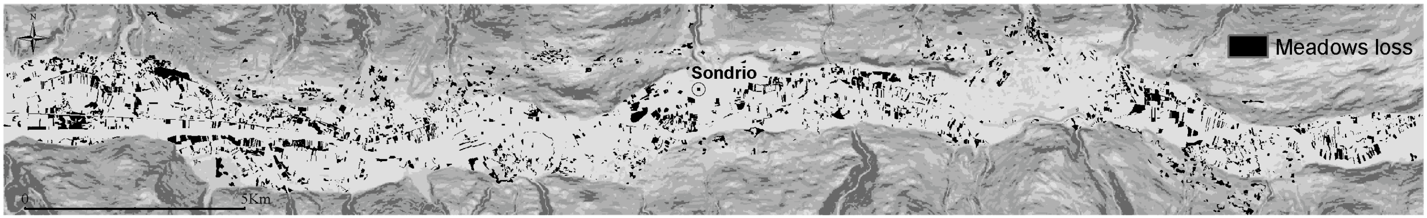

Fig.2 Map of the areas of meadows loss (824 ha).

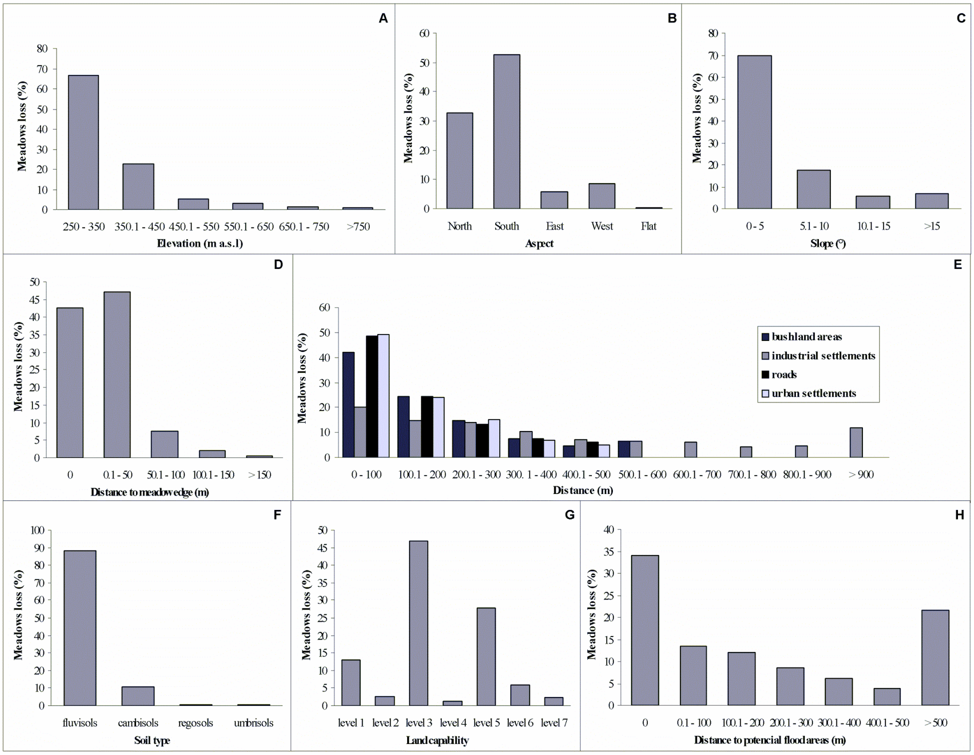

Fig. 3 Variation of the areas of meadows loss (824 ha) according to topography related variables (A, B, C), soil related variables (F, G, H) and neighbourhood related variables (D, E).

3.3. Identification of the significant drivers for meadows loss

The GIS-based logistic regression statistics covering 290 observations demonstrated that the set of independent variables were very appropriate to quantify and locate the presence of meadows loss. The model chi-square value was highly significant (95.50 Chi-square P <0.0001; Table 3) and the overall explanatory power of the model was moderately good (cf. pseudo R^2 of 0.24) (Clark, 1986). The ROC value was 0.79 and 78.6% of the estimated locations showed meadows loss in reality, while 64.1% of locations with absence of meadows loss were predicted correctly. The residuals showed weak spatial autocorrelation (Table 3). Eight independent variables significantly influenced the presence or absence of meadows loss. Neighbourhood related and soil related variables contributed most to meadows loss and topography related variables were not significantly related (Table 4). The densities of urban (3.712, P<0.0001), industrial (2.697, P<0.0001) and bushland (1.982, P<0.0001) added significantly to the presence of loss. Meadows loss increased with increasing land capability (1.011, P<0.0001). Instead, absence of meadows loss was significantly explained by the distance to meadow edge (-7.405, P<0.0001) and meadows density (-1.692, P<0.0001). Also, distance to roads (-0.338, P<0.05) reduced meadows loss, however the effect is less effective. Counter expectation, areas affected by soil degradation processes (soil accumulation or erosion) were not sites with pronounced meadows loss (-2.175, P<0.0001). Unexpected was also the non significance of most of distance variables, exception made for distance to roads and meadow edge.

Table 3 Summary statistics of the GIS-based logistic regression model (n=290 randomly selected observations).

Statistics of the GIS-based logistic regression model	Value
Number of sample pixels (presence of loss / absence of loss)	145/145
-2logL0	402.02
-2log(likelihood)	306.52
Chi-Square model (15)	95.5
Pr (>Chi-Square)	<0.0001
Pseudo R^2	0.24
ROC	0.79
Morans I	0.01

Table 4 Summary statistics of the regression coefficients of the GIS-based logistic regression model (n=290).

Independent parameter	Coeficient (β)	Standard error	Odds ratio (e^{β})	*Z*	P (> *Z*)
Intercept	29.422	0.500		58.844	0.000
Slope	0.008	6.214	1.008	0.001	0.999
Sine aspect	-0.457	0.496	0.633	-0.921	0.357
Cosine aspect	-0.374	0.709	0.688	-0.528	0.597
Land capability	1.011	0.152	2.748	6.651	0.000
Soil degradation	-2.175	0.136	0.114	-15.993	0.000
Urban density	3.712	0.152	40.936	24.421	0.000
Industrial density	2.697	0.069	14.835	38.974	0.000
Meadows density	-1.692	0.253	0.184	-6.698	0.000
Bushland density	1.982	0.130	7.257	15.246	0.000
Distance to urban settlements	0.537	0.716	1.711	0.750	0.453
Distance to roads	-0.338	0.789	0.789	2.201	0.028
Distance to industrial settlements	0.381	0.923	1.464	0.413	0.680
Distance to bushland areas	-0.288	0.838	0.750	-0.344	0.731
Distance to meadow edge	-7.405	0.314	0.001	-23.560	0.000
Distance to streams	-0.085	0.879	0.919	-0.097	0.923

4. Discussion

4.1. Landscape transformation and meadows conversion

This study found considerable land cover changes in the lowlands of Valtellina valley between 1980 and 2000 and confirmed the general trend for mountain regions in Europe, since the 1950s, of increasing human settlements and bushland and reduction in agricultural land. In particular, bushland expansion in lowlands, which was found in lowlands, is an unusual process since in the European Alps it has usually been registered at higher elevation, with a consequent reduction in agricultural activities (Gehrig-Fasel et al., 2007). This shows that the lowlands landscape in the Valtellina valley, between 1980 and 2000, were under a bidirectional development: simultaneously under intensification by anthropic processes and abandonment by decreasing of agriculture activities.

Meadows were the most affected land cover type, decreasing nearly one quarter in 20 years, a decline 2.3 times higher than the registered for grasslands at global scale from 1700 to 1980 (Dale et al., 2000). Results show that meadows were suppressed by human settlements (urban, industrial and roads), other agriculture uses (cultivation, orchards and vineyards), bushland and uncultivated land. The spatial allocation was distinct for each conversion type. In flat central locations, south-exposed (less rainfall and more sunny), the conversion of meadows to human settlements predominated, as found in France and Austria (Astrade, 2007, Tasser, 2005). In this spatial context, conversion to uncultivated land also occurred, mainly in proximity to industrial settlements or low productive lands adjacent to water streams. In medium to steep slopes, the conversion of meadows to other agriculture uses took place mainly in south-exposed locations, while the conversion to bushland prevailed mainly in north-exposed locations. In particular, conversion to orchards occurred in well drained alluvial deposits with medium slope and conversion to vineyards occurred in steep slopes, both occurring also in the South Tirol Alps region. The only exception was conversion to cultivation, which instead occurred in flat areas near water streams, due to irrigation needs.

The context of meadows conversion in the lowlands of Valtellina valley is complex. The grasslands management has been modified to make front the globalization of the sector and remain competitive (Kristensen et al., 2004). The farming model intensified the use of lowland meadows in order to enhance forage production, reduce labour costs and in some cases to allow another part-time job to raise income (Strijker, 2005). This is illustrated by the increase in livestock units (+14.3%) and the decline of upland grasslands (-31.7%) during the period of this study (Sondrio, 2004). Furthermore, in this context, strong competition for land with other economic sectors occurred, mainly with the development of the tourist industry (Martz, 1990). In addition, other agriculture types (e.g. apple orchards) become more rentable due to their higher economical value and more direct aids from CAP in the 1990s, as happened in other regions of the Italian Alps (e.g. South Tyrol). Finally, some meadows parcels became less suitable to use and were marginalized allowing successive bush encroachment, due to technical improvements in the meadows management process (e.g. mechanization), reported in other LUCC studies in the European Alps at higher elevations (Gellrich et al., 2007, Rutherford et al., 2008). The urban, industry and road sprawl in former meadows illustrates the competition for land in lowlands. These processes aimed to prevent local outmigration, increasing job opportunities (Perlik and Messerli, 2004) through the expansion of tourism and commercial infrastructures in order to attract residents from neighbour dense locations such as Milan. Although Valtellina valley experienced outmigration from 1980 to 2000, according to the Italian national statistic institute (-1871 inhabitants; ISTAT, 2001a), and despite the expansion of industrial settlements, both employment and the number of manufacturing corporations dropped by 9% and 7.5%, respectively (ISTAT, 2001b). This underpins that urban, industry and road expansion per se cannot stop outmigration. Besides, it shows the strong impact of industry for mountain landscapes and the need to improve the reuse of abandoned industrial settlements.

The expansion of uncultivated land found in former meadows is unexpected, due to the strong competition for land observed in flat zones, moreover, this transition has not been described in literature in the Alps lowlands. This study considered such transition as an intermediate step to conversion into more rentable uses (e.g. industrial), considering its spatial allocation, and also considering that licence to conversion into other uses would be easier to obtain after a period as unknown use.

4.2. Significant drivers for meadows loss according to spatial environmental features

The spatial analysis conducted revealed that meadows loss was affected by spatial patterns of environmental variables, as suggested also by recent studies in the European Alps (Hersperger and Bürgi, 2009, Schneeberger et al., 2007). Meadows loss was more frequent in flat and south-exposed locations, in proximity to urban, roads, bushland structures and in locations with good land capability and at potential flood risk. GIS-based logistic regression model revealed that densities of urban, industry, bushland and land capability were the only significant predictors for meadows loss in the study area. The structure of the land cover type has been shown previously to significantly influence landscape changes in the European Alps (Rutherford, 2007). In this study, the density of urban and industrial may influence meadows loss due to the fact that meadow parcels in urban/industrial context remain fragmented and dispersed, which difficult accessibility and increases labour costs, becoming thus less lucrative (MacDonald et al., 2000). Moreover, likelihood for urban/industrial development in mountain regions is higher in dense locations, since costs with infrastructures (e.g. accessibility) are avoided, increasing profit. This means that meadows adjacent to urban/industrial are always threaten. Also bushland density potentially contributed to meadows loss due to interference with accessibility and productivity of meadows. It has in fact been shown before that forest regrowth occurred in the Alps, in meadows near to forest edges, with limited machine access (Gellrich et al., 2007, Rutherford et al., 2008). An increase in bushland density results in the isolation of meadows parcel fragmentation respect to the farming centre. Moreover, both productivity and plant diversity are affected by the density of bushland, since it interferes with essential elements for meadows (e.g. light conditions), and meadows are subjected to intense bush colonization by fast grower species (e.g. Alnus viridis and Salix caprea) (Aavik et al., 2008). In addition, the fragmented ownership of the meadows parcels, which is common in Southern Europe (MacDonald et al., 2000), difficult bush clear and preservation of

meadows in this context. This study found that land with best likelihood for agriculture is a significant predictor to meadows loss, which is an unexpected result and in contrast to reports stating that only highly productive parcels remained managed by farmers, leading to an overexploitation of these lucrative parcels (Tasser et al., 2007). This result may threat the meadows economical sustainability, since the most productive meadows are gradually loss.
The significance of both distance to meadow edge and meadows density for the absence of meadows loss confirmed the role of fragmentation evidenced before, since locations where meadows were continuous and apart from the meadows edges were preserved and apparently agro-economically sustainable. The significance of soil degradation to meadows preservation was surprising, it might however be an indirect effect since built-up was partly banned or at least very cost effective in locations under soil instability. In Switzerland, locations susceptible to natural hazards were generally less affected by land cover changes (Rutherford, 2007).
The overall explanatory power of our GIS-based logistic regression model is in line with similar studies that applied logistic regression to characterise environmental processes, such as fires in forests (Stolle et al., 2003), urban growth dynamics (Hu and Lo, 2007) and forest expansion (Rudel et al., 2005). Logistic regression was an appropriate method to analyse spatial dimension meadows loss, confirming its potential for analysis when the dependent variable is binary (Lesschen et al., 2005). The prediction accuracy of this model was acceptable since LUCC in mountain regions express complex patterns in space and time with increasing difficulty to be reproduced by models.

5. Conclusions

This study measured the underestimated land cover changes and spatial environmental variables behind the loss of permanent meadows within the period 1980-2000 in the lowlands of Italian Valtellina valley, European Alps. Nearly one quarter of the meadows lands were definitively lost and mainly converted into human settlements, other agriculture uses, bushland and uncultivated land. The spatial analysis denoted that local spatial environmental features are important for meadows loss and avoiding fragmentation may be a good strategy for meadows preservation.

On the other hand, the conflict by land in locations densely occupied by other land cover types with good land capability is the major threat to meadows. In contrast to land use /land cover changes in other mountain regions, this study has found abandonment in the lowlands of Valtellina in addition to a massive use intensification in the remaining meadows. This questions the suitability of the current farming model for mountains regions. These results denote also that due to the powerful processes behind meadows loss, future planning in the Valtellina valley will have to account the economic value of ecosystem services provided by this habitat, to protect it from intensification and avoid extinction.

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Chapter 2

Plant diversity of permanent meadows is affected by recent and historical changes in spatial environmental heterogeneity in the Italian Alps

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Abstract

The present study analyzed the relationships between different plant diversity indices for permanent meadows and spatial environmental heterogeneity in a landscape under decline of grasslands habitat. In the lowlands of Valtellina Valley, Italian Alps, the effect of recent spatial environmental heterogeneity and change in spatial environmental heterogeneity over the landscape from year 1980 to 2000 was assessed. Spatial environmental heterogeneity was defined in four aspects: landscape, topography, soil and changes over time, combining field data survey at plot level, aerial orthophoto based mapping and spatial metrics at landscape level. Plant diversity was measured by species richness, Shannon index and species evenness indices. For each, regressive models at different spatial extents were developed (125, 250, 500m and full extent). Results evidenced that species richness and Shannon indices were best explained by regressive models including changes occurred in spatial environmental heterogeneity from 1980 to 2000. Species richness was negatively related to strong decrease in meadows habitat area and recent urban area, while Shannon index was positively related to the increase in landscape diversity. In contrast, species evenness was better explained by regressive model including recent spatial environmental heterogeneity and positively

related to increase in eastness in the study area, and negatively affected both by the area of woody and soil pH (KCl). In conclusion, plant diversity indices responded differently to components of spatial environmental heterogeneity. To avoid loss of meadows habitat, maintain landscape diversity and appropriate urban planning, together with sustainable meadow management, are crucial for conservation of plant diversity in meadows.
Keywords: Species richness, Shannon index, Evenness, Grassland, Landscape change, Landscape metrics, Stepwise regression.

1. Introduction

Grassland habitats are among the most diverse plant communities in Europe and are important habitats for a large number of rare plants and animals (Billeter et al., 2008). The abandonment of traditional, sustainable management practices during the last century has led to a massive loss of grassland habitats and the increasing fragmentation of the remaining grassland habitats represents a further major threat to biodiversity (Krauss et al., 2004).

Across the landscape, an intriguing question is the relationship between different plant diversity measures and the surrounding environment. In particular, the role of spatial environmental heterogeneity in the distribution of several elements, such as nutrients and climate, as well as the composition and the spatial configuration, and how it influences plant species diversity (Rosenzweig, 1995, Stohlgren et al., 1998). It became a main subject for biodiversity conservation issues. In general, species diversity can be described by 'richness', which represents the number of species in a given area, by the Shannon diversity index, which takes into account the abundance of each species and by 'evenness', which represents the variability in species abundance. However, recent empirical studies indicated that the relationship between species diversity and evenness is not always positive (Manier and Hobbs, 2006). Richness and evenness may respond differently to changes at the local habitat factors as well as to changes in the spatial configuration of the habitats (Kie et al., 2002, Magurran, 2004).

Spatial environmental heterogeneity can influence four main ecological processes of plants: establishment, persistence, dispersal and reproduction (Dufour et al., 2006). Recent studies found that plant species richness increased with increasing spatial environmental variability in a mountain landscape but decreased with spatial aggregation (less 'roughness') (Dufour et al., 2006) and that native, but also non-native plant species richness responded to spatial environmental heterogeneity at the landscape level; plant species richness was favoured by edge density and diversity of adjacent landscape types (Kumar et al., 2006). Other studies also found that historical connectivity of habitats over time influenced positively plant species richness (Clough et al., 2005, Lindborg and Eriksson, 2004, Müller et al., 2004). It seems therefore clear that the nature of effects of spatial environmental heterogeneity change with scale, changing the underlying mechanisms. So far, however, there is no common approach nor method to assess spatial environmental heterogeneity (Kie et al., 2002, Kumar et al., 2006). Operationally, one of the most explored approaches is the implementation of spatial environmental heterogeneity into categorical and numerical maps with the evaluation of the composition and configuration of landscape components, e.g. the spatial arrangement of patches, the number of patch types and the patch shape (Li and Reynolds, 1994).

The relationship between different plant diversity measures and spatial environmental heterogeneity in grasslands has not been studied in detail until very recently. Also, the analysis of this relationship has been relying mostly on present data and past conditions have been less considered. However, past conditions have a significant role (Ernoult et al., 2006) and empirical studies assessing the link between plant diversity and both present and past spatial environmental heterogeneity are central for landscape management, landscape planning and biodiversity conservation. In particular in Italy, studies relating plant diversity measures in grasslands to spatial environmental heterogeneity are rare, even if such studies are of crucial importance due to deep management and landscape transformations on grasslands ecosystem since the 1960's.

This study, located in the lowlands of Valtellina valley, assessed the relationships between plant diversity measures in permanent meadows (species richness, Shannon index and evenness) and both recent spatial environmental heterogeneity features (year 2000) and changes in spatial environmental heterogeneity, occurred between 1980 and 2000. These permanent meadows are subject to intense use and decreased 18.5% in 20 years at a mean velocity of 26.9ha year-1 (Monteiro, 2010), which affected its patterns and its relevant ecological and economical values (water conservation, forage production, dairy products, aesthetic tourism attractiveness) in a context of global environmental changes and increasing need for food supply (Ceballos et al., 2010).

In this study spatial environmental heterogeneity was defined by categorical maps, topographical and soil variables and the following questions were addressed: (i) How are different plant diversity measures (species richness, species diversity and evenness) related to recent spatial habitat and landscape features, topographic and soil patterns? (ii) How is plant diversity related to change in landscape during the period 1980-2000?

2. Methods

2.1 Study area

The study area comprises 120 km2 of the bottom valley of central Valtellina, a U shaped valley carved by glaciers with a west-east orientation along the River Adda in the north of Italy, southern Alps (46°10' N 9° 50' E, Fig. 1). The bottom valley is flat (slope < 5%), the elevation ranges from 250 to 750m above sea level (a.s.l.) and the most frequent soils are Eutric Fluvisols, Eutric Cambisols and Dystric Cambisols (FAO, 1988). The climate is temperate continental due to the vicinity of Lake Como: mean annual temperature 11.9°C and mean annual precipitation 970mm, measured at the climate station of Sondrio (298m a.s.l.) for the period 1973–2007. The permanent meadows covered 29% of the bottom valley (Monteiro, 2010), they are intensively used for hay production with 3-4 cuts per vegetation season and fertilized by dung addition during the winter-spring period only. According to the European Union habitats, Ahrrenatherion-Elatioris and Polygono-Trisetion represent the most abundant meadow types (EU habitat codes 6510 and 6520) (Fava et al., 2010). In the adjacent slopes, sparse grasslands are surrounded by woody trees, since abandonment of agriculture activities allowed for bush and shrub encroachment into grasslands. The landscape is also composed by urban settlements, orchards and vineyards in the southern slope at lower elevations of the valley. Besides tourism, permanent meadows represent an important source for local farming and Valtellina's economy.

2.2 Plant diversity assessment and estimation of Ellenberg's ecological indicators

For plant diversity and evenness measures 34 vegetation plots were selected with 10x10m size, randomly located in randomly selected meadows in the lowlands of Valtellina (Fig. 1). The distance between each plot was at least 500m in order to increase independence between observations, since landscape context can influence plant diversity at 500m (Marini et al., 2008). In order to reduce edge effects the sample location distanced at least 10m to meadow border. For each plot, additional data was recorded and stored in a GIS database: location and elevation (Trimble GeoXT GPS), slope, aspect and meadow manager. Vegetation surveys were carried out in May 2006 (close to peak biomass). All vascular plants were determined at species level and abundance of each species was visually estimated according to Braun-Blanquet (Braun-Blanquet, 1964). For each plot the Ellenberg's ecological indicator values (light, temperature, continentality, moisture, soil pH, nutrients, soil texture) were calculated from the species list (indicator values range 1-9). Species diversity index was calculated using Shannon diversity index ($H' = -\sum_i p_i \ln(p_i)$), where p_i is the proportional abundance of the i species, and evenness as $J' = H' / H_{max}$, where H' is the number derived from the Shannon diversity index and H_{max} the maximum value of H'.

2.3 Quantification of landscape heterogeneity

In order to measure the influence of landscape heterogeneity on plant diversity 12 spatial metrics were quantified at landscape level, which evaluated "composition", "configuration", "edge", "connectivity" and "diversity" in the landscape of the study area for the years 1980 and 2000 (Table 1). The spatial metrics selected were: urban area, meadows area, woody area, number of different land cover types (NDCT), number of patches (NP), largest patch index (LPI), edge density (ED), mean Euclidean nearest neighbour distance (ENN-MN), Euclidean nearest neighbour distance-coefficient of variation (ENN-CV), mean proximity index (PROX-MN), proximity index-coefficient of variation (PROX-CV) and Simpson's diversity index (SIDI). These were estimated individually for each one of the 34 selected plots at three spatial extents (125, 250, 500m), assuming that most of the species were influenced by the surrounding landscape features at least at a 125m spatial extent. It was assumed that the area of interaction between landscape spatial metrics and plant diversity could be described by a circle, so that the corresponding total areas are respectively 4.9ha, 19.6ha and 78.5ha. The spatial resolution was fixed (10m), in order to include the local landscape attributes as the small size of farming parcels. The metrics urban area, meadows area, woody area and NDCT area were calculated with area and query functions of ArcGIS 9.1 software (ESRI, 2005), while the estimation of NP, LPI, ED, ENN-MN, ENN-CV, PROX-MN, PROX-CV and SIDI was done with the grid implementation FRAGSTAT program (version 3.3) (McGarigal, 2002), using a 4x4 moving window. The source data for these estimations were the land cover layers from 1980 and 2000 extracted for each spatial extent of analysis. These land cover maps classified the landscape into 10 types (meadows, cultivated, orchards, vineyards, woody, uncultivated, water, urban settlements, industrial settlements, hedges) through visual interpretation of two digital colour ortho-rectified aerial photographs (1980, scale 1:20000; 2000; scale 1: 36000) and hand digitizing in ArcGIS 9.1 (ESRI, 2005). Urban settlements class includes roads network. Both aerial photographs were produced by the Lombardia Region, registered and geo-referenced to the Italian grid system Gauss Boaga- Zone 1.

2.4 Quantification of topographic and soil heterogeneity

In order to measure the influence of topographic and soil heterogeneity on plant diversity, the following variables were measured: slope, aspect, soil organic carbon (SOC), proportion of soil organic matter (SOM) and pH (KCl). Slope and aspect were derived from the digital elevation model (DEM; 20x20m) of the Lombardia region using surface analysis functions of ArcGIS 9.1 (ESRI, 2005) and estimated individually for each one of the 34 selected plots at three spatial extents (125, 250, 500m). The slope and aspect average values at each spatial extent were retained for analysis. To allow statistical analysis, the circular variable aspect was transformed into north-south gradient (northness) and east-west gradient (eastness) by using respectively the sine and cosine functions (Piedallu and Gégout, 2008). Northness varies from -1 (south-facing) to 1 (north-facing), and eastness from -1 (west-facing) to 1 (east-facing). Soil variables were obtained from soil samples at 20 cm soil depth collected for each plot during the vegetation field survey.

2.5 Statistics

Simple descriptive statistics were applied to characterize plant species diversity and Ellenberg indicator values across the 34 selected meadow plots. For the dependent variables species richness, Shannon index and species evenness, models were developed that included the independent variables spatial metrics of the year 2000, topographic and soil. For each dependent variable four models were developed, one individual model for each spatial extent (125, 250, 500m) and a full extent model including all variables from all spatial extents of analysis. In the full extent model, the independent variable aspect was the average value of the three spatial extents. In order to improve normal distribution, the dependent variable species richness was power-transformed (by 0.3).

In order to test the effects of the spatial change occurred with landscape transformation from 1980 to 2000 on plant diversity variables, a full extent model was developed, which include the difference in spatial metrics values between 2000 and 1980 and the topographic and soil heterogeneity data used in the previous models, since data for these variables for the year 1980 was not available. However, it was assumed that the topographic variables remained stable during the 20-year observation period (1980-2000). All the models were estimated using backward stepwise multiple regression, thus non-significant variables were removed from the model. Regressive models were effective in previous studies (Kumar et al., 2006, Lindborg and Eriksson, 2004). Regression analyses were conducted using the general linear model (GLM) procedure in Systat SPSS Software (version 10) (2000) and probability <0.05 was considered as significance level in all cases. All models were tested for spatial autocorrelation using the Durbin-Watson (D) statistic test, where $D=2$ indicates absence of autocorrelation, while D substantially $<$ or $>$ 2 evidences for positive or negative autocorrelation (Gujarati, 2002).

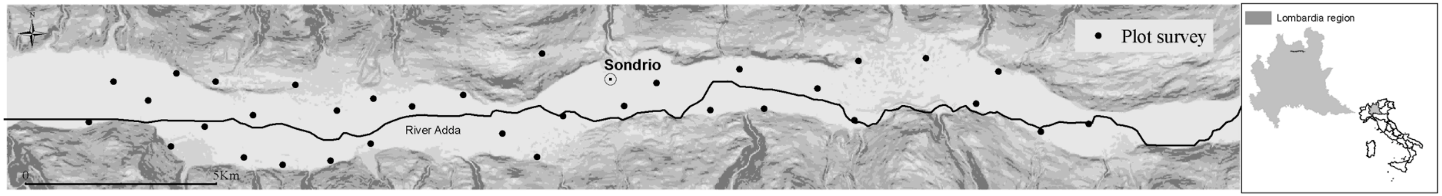

Fig.1 Location of the study area in the Valtellina Valley, northern Italy.

Table 1 List of spatial metrics estimated at three spatial extents (radii of 125, 250, 500 m) to quantify landscape heterogeneity for the 34 vegetation plots in the lowlands of Valtellina Valley.

Spatial metrics	Acronym and units	Measurement scale	Spatial extent of analysis		
Urban area	Urban area (ha)	landscape level	125m	250m	500m
Meadows area	Meadows area (ha)				
Woody area	Woody area (ha)				
Number of different land cover types	NDCT				
Number of patches	NP *				
Largest patch index	LPI (%) *				
Edge density	ED (m/ha) *				
Mean proximity	PROX-MN *				
Proximity- coefficient of variation	Prox-CV *				
Mean Euclidean nearest neighbour	ENN-Mn (m) *				
Euclidean nearest neighbour- coefficient of variation	ENN-CV (%) *				
Simpson's diversity index	SIDI *				

* These spatial metrics were described in (http://www.umass.edu/landeco/research/fragstats/documents/Metrics/Metrics%20TOC.htm).

3. Results

3.1 Plant diversity and Ellenberg's values for the hay meadows of low lands of Valtellina

The vegetation surveys were carried out in May 2006, thus six years after the recent land cover layer (2000). Across the 34 selected meadow plots, a total number of 86 vascular plant species was recorded (Appendice I), indicating a high species richness despite the rather intensive land management. The mean value of species richness was 22 ± 4.8 (Table 2). The mean value of Shannon–Wiener index of diversity was 3.26 ± 0.46 and 70% of overall values ranged between 2.90 and 3.80. The mean value of species evenness was 0.71 ± 0.09, representing a not very pronounced variation in species abundance between selected meadows (Table 2). Species richness was only weakly associated with Shannon index (R^2=0.26), whereas Shannon index was closer and positively associated to evenness (R^2=0.68). Lowest means of Ellenberg indicator values were calculated for soil moisture (2.90±0.19) and for pH (KCl) (2.97±0.19), indicating a rather dry, acidic soil. Highest mean values were observed for texture (4.01±0.08) and light conditions (3.57±0.17), slightly lower means were calculated for nutrients (3.38±0.28) and humus indicators (3.26±0.19). Mean value of temperature indicators was 3.13±0.22, and for continentality indicator was 3.04±0.07, indicating a not very pronounced continental climate at the study site (Table 2).

3.2 Regression models for plant diversity measures for recent spatial environmental heterogeneity

Individual regression models were developed for the dependent variables species richness, Shannon index and species evenness at three spatial extents and one full extent model, covering the recent spatial environmental heterogeneity (year 2000) as explanatory variables (see Table 3). For the dependent variable 'species richness', each one of the models at 3 different spatial extents (125, 250 and 500m) was statistically significant, but the regression coefficients were relatively low (R^2=0.28, P=0.006, 125m spatial extent; R^2=0.30, P=0.004, 250m spatial extent; R^2=0.33, P=0.002, 500m spatial extent). Variation in species richness was mainly explained by cosine aspect and soil organic carbon (SOC). Species richness was higher in the south-facing slopes and in soils rich in organic carbon. The full extent model yielded a closer relationship between species richness and explaining variables (R^2=0.73, P<0.0001): increase in slope (at 125m spatial extent) enlarged species richness, whereas spatial metrics such as urban area (at 250m spatial extent), number of patches (at 125m spatial extent), largest patch index (at 500m spatial extent), were negatively correlated to species richness. In line with the single spatial extent models, increasing mean cosine aspect (i.e. northness) reduced species richness.

Regression models for the dependent variable 'Shannon index' were also statistically significant, however, overall regression coefficients were even lower compared to the models for species richness and coefficients varied with spatial extent (R^2=0.13, P=0.04, 125m spatial extent; R^2=0.45, P=0.008, 250m spatial extent; R^2=0.13, P=0.041, 500m spatial extent). The model fit at 250m spatial extent was the most accurate and spatial metric variables such as woody area, NP, ED, PROX-CV and soil pH (KCl) were negatively associated to Shannon index. Increasing sine aspect (eastness) led to higher Shannon index values. In the full extent model (R^2=0.28, P<0.008), the variable woody area (at 500m spatial extent) was negatively related to Shannon index and connectivity (PROX-MN at 250m spatial extent) was positively correlated.

Higher regression coefficients were observed for the dependent variable species evenness in comparison with the other dependent variables. The model at 250m spatial extent presented the higher regression coefficient (R^2=0.71, P<0.0001). Species evenness at this spatial extent was related positively to sine aspect and number of patches (NP), but negatively to woody area, ED, PROX_CV and soil pH (KCl). At 500m spatial extent (R^2=0.41, P=0.001), woody area again was negatively related to species evenness, whereas connectivity (PROX_MN) was positively related. At 125m spatial extent (R^2=0.42, P=0.001), species evenness was negatively related to the spatial metrics LPI, NDCT and soil organic carbon (SOC). The full extent model for species evenness was highly significant (R^2=0.67, P<0.0001) and species evenness was positively related to connectivity at 500m (PROX_MN) and negatively related to largest patch at 125m (LPI), slope (at 500m) and proportion of soil organic matter at plot level.

Table 2 Summary statistics of plant diversity indices and Ellenberg indicators for permanent meadows in the study area (n=34, 100m2 plots).

				Ellenberg indicators							
	Species richness	Shannon index	Species eveness	Light	Temperature	Continentality	Moisture	pH	Texture	Nutrients	Humus
Mean	22.4	3.3	0.7	3.57	3.13	3.05	2.91	2.97	4.02	3.38	3.26
SD	4.9	0.5	0.1	0.17	0.23	0.08	0.19	0.11	0.09	0.28	0.19
Min	14.0	1.7	0.4	3.17	2.60	2.88	2.27	2.77	3.83	2.69	2.96
Max	33.0	4.1	0.9	3.83	3.58	3.35	3.38	3.33	4.19	3.85	3.71

Table 3 Relationships between plant diversity indices and spatial environmental heterogeneity. Regression models developed at three spatial extents (radii of 125, 250, 500 m) and at full extent. Relationships were significant at alpha= 0.05 (Landscape level, n=34, 100m^2 plots).

					Regression model				
Dependent	Spatial extent (m)	Variable	Parameter	*P*	R^2	df	*F*	*P*	*D*
Landscape level									
Species Richness	125	Intercept	2.361	0.0001	0.28	2, 31	6.141	0.006	1.92
		Cosine aspect	-0.113	0.006					
		SOC	0.005	0.012					
	250	Intercept	2.361	0.0001	0.30	2, 31	6.777	0.0004	1.89
		cosine aspect	-0.124	0.004					
		SOC	0.005	0.01					
	500	Intercept	2.360	0.0001	0.33	2, 31	7.719	0.002	1.92
		cosine aspect	-0.137	0.002					
		SOC	0.005	0.009					
	Full extent	Intercept	2.692	0.0001	0.73	5, 26	13.856	0.0001	2.25
		Urban area (250 m)	-0.025	0.002					
		NP (125 m)	-0.009	0.002					
		LPI (125 m)	-0.005	0.013					
		Slope (125 m)	0.030	0.0001					
		Mean cosine aspect	-0.118	0.0001					
Shannon index	125	Intercept	3.343	0.0001	0.13	1, 30	4.632	0.04	2.35
		Sine aspect	0.276	0.004					
	250	Intercept	7.352	0.0001	0.45	6, 27	3.716	0.008	2.29
		Woody area	-0.066	0.017					
		NP	0.029	0.013					
		ED	-0.007	0.016					
		PROX-CV	-0.006	0.002					
		Sine aspect	0.569	0.008					
		pH (KCl)	-0.196	0.024					
	500	Intercept	3.500	0.0001	0.13	1, 30	4.58	0.041	2.33
		Woody area	-0.010	0.041					
	Full extent	Intercept	3.34	0.0001	0.28	2, 29	5.649	0.008	2.33
		Woody area (500 m)	-0.012	0.011					
		PROX-MN (250 m)	0.005	0.021					
Species eveness	125	Intercept	1.1	0.0001	0.42	3, 30	7.362	0.001	2.12
		LPI	-0.003	0.002					
		NDCT	-0.035	0.005					
		SOC	-0.003	0.005					
	250	Intercept	1.69	0.0001	0.71	6, 26	10.705	0.0001	2.03
		Woody area	-0.016	0.0001					
		NP	0.008	0.0001					
		ED	-0.002	0.0001					
		PROX-CV	-0.002	0.0001					
		Sine aspect	0.161	0.0001					
		pH (KCl)	-0.034	0.009					
	500	Intercept	0.731	0.0001	0.41	2, 28	9.731	0.001	1.99
		Woody area	-0.004	0.001					
		PROX-MN	0.001	0.002					
	Full extent	Intercept	0.896	0.0001	0.67	4, 28	14.163	0.0001	2.21
		LPI (125 m)	-0.003	0.0001					
		PROX-MN (500m)	0.002	0.0001					
		Slope (500m)	-0.006	0.006					
		SOM	-0.002	0.0001					

R^2 is the coefficient of determination; P p-value; d.f. degrees of freedom; F test statistic; D Durbin-watson statistic.

3.3 Changes in spatial environmental heterogeneity

In order to test the effects of changes in spatial environmental heterogeneity on species richness measures, we included the differences in spatial metrics between the year 2000 and 1980 in our regression models. Spatial metrics showed that landscape features changed from years 1980 to 2000 at all spatial extents of analysis (Table 4). Landscape composition around the vegetation plots substantially changed, both urban and woody area become more frequent at all spatial extents, the first in particular at 250m (+38.6%), while woody area more at 125m (+20.0%). By opposite, meadows area was significantly reduced, loss ranged from 1.1% at 125m to 20.1% at 500m. Configuration of landscape was also gradually altered, the decline in NP, ED and in particular in LPI values from 1980 to 2000 show a decrease in patches, shape complexity and its size. Connectivity in landscape reflects such changes in configuration. The dispersion of the patches of each land cover category was considerably reduced, since PROX-MN values increased from 8.4% at 125m to 16.9% at 500m. Despite of this, the increase of ENN-MN values illustrates that some land cover types were most favoured than others, since the running distances from one patch to its nearest neighbour of the same class have increased, moreover ENN_CV shows that asymmetry in the distribution of running distances increased, mainly at 125m (16.4%). Also, PROX-CV shows a great asymmetry in the distribution of PROX-MN over the landscape, illustrating that effects of landscape change varied greatly according land cover type. In fact, landscape is recently more diverse, since proportional distribution of area among patch types becomes more equitable as denoted by the increase in SIDI values mainly at 500m spatial extent (+7.6%). It suggests a fragmentation of patches into smaller size, particular damaging for meadows habitat area, which decreased strongly during this period (1980-2000) (Table 4).

The regression coefficients for the 3 different dependent variables ranged from $R^2=0.39$ ($P<0.0001$) for species evenness to $R^2=0.76$ ($P<0.0001$) for species richness (Table 5). Species richness was negatively related to meadows loss between 1980 and 2000 at the 250m spatial extent and to increase in number of patches (NP) at 125m spatial extent. Species richness was lower in north-facing areas (mean cosine aspect) and negatively related to elevation at a 500m spatial extent. Regression coefficient for Shannon index was with R^2=0.62 (P<0.0001) lower than for species richness. Shannon index was positively related to an increase in landscape diversity at 500m (SIDI) and to lesser extent also to increase in connectivity (PROX-MN; 125m spatial extent) from 1980 to 2000. The variable proportion of soil organic matter, which only contained recent values, added to the overall explanation of this regression model. Shannon index was negatively related to the decrease in number of patches (NP) at 125m spatial extent and meadow area from 1980 to 2000. The species evenness regression model was not significantly affect by changes in spatial environmental metrics (R^2=0.39, P<0.0001).

Table 4 Summary statistics of spatial metrics estimated at three spatial extents (radii of 125, 250, 500 m) to quantify change in landscape heterogeneity for the 34 vegetation plots in the lowlands of Valtellina Valley.

Spatial metrics	Year	Spatial extent					
		125m		250m		500m	
		Mean	SD	Mean	SD	Mean	SD
Urban area (ha)	2000	0.64	0.72	2.55	2.16	11.38	1.06
	1980	0.55	0.58	1.84	1.33	8.27	0.00
Meadows area (ha)	2000	2.90	1.04	8.84	3.60	25.85	3.33
	1980	3.05	1.07	10.24	3.90	32.30	5.30
Woody area (ha)	2000	0.48	0.57	3.11	2.82	16.55	0.00
	1980	0.40	0.52	2.61	2.26	15.20	0.00
NP	2000	13.06	6.00	34.38	13.69	86.24	33.00
	1980	13.88	6.96	38.44	16.56	103.21	31.00
LPI (%)	2000	42.30	16.92	30.72	11.70	23.47	7.52
	1980	44.55	17.81	33.95	13.82	26.20	8.25
ED (m/ha)	2000	473.65	75.85	334.60	50.99	231.60	151.20
	1980	474.26	80.71	335.84	58.85	243.46	144.30
PROX-MN	2000	13.13	12.18	38.34	27.07	80.04	18.52
	1980	12.11	12.49	33.93	22.04	68.46	5.42
Prox-CV (%)	2000	155.66	63.63	212.23	69.09	254.30	118.23
	1980	146.27	64.29	230.21	88.11	288.94	184.59
ENN-Mn (m)	2000	26.70	12.94	35.91	16.39	45.10	23.67
	1980	23.96	12.22	31.00	10.63	34.85	21.02
ENN-CV (%)	2000	53.65	37.97	103.26	45.34	140.20	81.80
	1980	46.10	33.66	98.58	42.32	149.37	49.49
SIDI	2000	0.50	0.19	0.62	0.15	0.71	0.41
	1980	0.47	0.22	0.59	0.17	0.66	0.15
NDCT	2000	4.50	1.24	6.21	1.25	7.26	4.00
	1980	5.03	1.66	6.41	1.44	6.82	4.00

Table 5 Relationships between plant diversity indices and change in spatial environmental heterogeneity. Regression models developed at full extent. Relationships were significant at alpha= 0.05 (Landscape level, n=34, 100m² plots).

					Regression model				
Dependent	Spatial extent (m)	Variable	Parameter	*P*	R^2	df	*F*	*P*	*D*
Landscape level									
Species richness	Full extent	Intercept	2.475	0.0001	0.76	7, 26	11.869	0.0001	1.97
		Woody area (250 m)	0.031	0.012					
		NP (125 m)	-0.022	0.0001					
		PROX-MN (125 m)	0.009	0.0001					
		Meadows loss (250 m)	-0.137	0.0001					
		Meadows loss (125 m)	0.014	0.0001					
		Elevation (500 m)	-0.122	0.017					
		Mean cosine aspect	-0.064	0.040					
Shannon index	Full extent	Intercept	4.34	0.0001	0.62	5, 25	8.049	0.0001	2.13
		Meadows area	-0.012	0.006					
		NP (125 m)	-0.036	0.002					
		PROX-MN (125 m)	0.013	0.001					
		SIDI (500 m)	4.97	0.001					
		SOM	-0.13	0.019					
Species evenness	Full extent	Intercept	0.915	0.0001	0.39	1, 29	18.412	0.0001	2.44
		Meadows area	-0.003	0.0001					

R^2 is the coefficient of determination; P p-value; d.f. degrees of freedom; F test statistic; D Durbin-watson statistic.

4. Discussion

4.1 Ellenberg's indicators

The vegetation surveys indicated that plant species richness patterns were typical for hay meadows under rather intensive agricultural use. Species diversity indices were relatively high when compared with indices of other studies (Fischer and Wipf, 2002, Niedrist, 2008). The low values of Ellenberg's indicators for continentality emphasizes the oceanic climate at the study site due to the proximity of Lake Como. However, the low moisture is not easy to explain, taking the intermediate soil texture values into account, we assume that plants were not heavily subjected to water stress condition or drought. This is also supported by the large standing fresh biomass in the range of 19.57t ha-1 and the mean fuel moisture content (76.9%), observed mainly in the meadows at the lowest elevation (cf. standing biomass had been assessed just after the plant surveys of the present study (Fava et al., 2010)). In meadows that were transformed into urban or industrial areas, soils often showed reduced soil moisture content due to hampered water infiltration and studies identified decrease in Ellenberg moisture indicator with increasing urbanization (Pysek et al., 2004). Urbanisation and industrialisation represents one main reason for the loss of permanent meadows in the lowlands of Valtellina. For instance, in the observation period from 1980 to 2000, 35% of the loss registered by permanent meadows was created by urban and industrial areas (Monteiro, 2010).

4.2 Plant diversity and spatial heterogeneity

The results indicated that the three plant diversity measures were associated to different spatial environmental heterogeneity variables and even more importantly, changes over time in spatial environmental heterogeneity affected these three plant diversity measures differently. Several studies emphasized that today's plant diversity is still affected by past landscape conditions (Fischer, 2008, Lindborg and Eriksson, 2004). In Sweden, Lindborg and Eriksson found that plant diversity responded to change in landscape configuration and was positively affected by past habitat connectivity (Lindborg and Eriksson, 2004). In Switzerland, the differences in the cultural traditions (Romanic, Germanic and Walser) turned out to still affect landscape diversity through land use diversity and therefore present-day plant diversity at plot level (Fischer, 2008). Reitalu and co-worker suggest that landscape history should whenever possible be incorporated in spatial heterogeneity analysis (Reitalu et al., 2009). In their study on semi-natural grassland in Sweden, they observed that plant evenness and richness, which were both associated with historical land cover maps, responded differently to habitat fragmentation. Evenness was much less affected by grassland isolation than species richness.

In the present study, the best regressive models evidenced that evenness was not related to changes over time in spatial environmental heterogeneity, whereas species richness was negatively associated to a substantially meadow loss (12.5%) at 250m spatial extent and decrease in NP at 125m spatial extent during the observation period. Shannon index on the other hand was positively related to increase in the landscape diversity (SIDI) at 500m spatial extent (7.6%). These relations underpin the negative impact of habitat loss and change in landscape composition to species richness, while those changes that increased diversity of adjacent landscape had a positive effect on Shannon index. The negative relation of species richness to habitat loss follows the island biogeography theory (MacArthur and Wilson, 2001), since decrease in habitat area can affect the establishment of new plant individuals and populations, especially in our study area where meadows habitat was mainly converted to urban and woody areas with meadows patches becoming more isolated. However, in a context of meadows decline, the remaining meadows are subjected to intense use and the effect on species richness decrease cannot be excluded it. Other studies have shown that past habitat area explained present species richness in Estonian and Swedish grasslands (Cousins et al., 2007, Helm et al., 2006), while in Germany, Finland and Belgium species richness in grasslands was better explained by present habitat area (Adriaens et al., 2006, Krauss et al., 2004, Raatikainen et al., 2009). Shannon index, in contrast to species richness, was positively affect by the change in landscape which increased landscape diversity at a 500m spatial extent, a spatial extent which registered a mean decrease of 20.1% in meadows habitat from 1980 to 2000. This finding follows the niche theory since increasing heterogeneity enhances the potential number of species that may exist in a given area. A previous study showed that landscape diversity significantly influences species diversity in Europe (Araujo et al., 2001) and grasslands surrounded by forest presented higher abundance of species than grasslands surrounded by other grasslands in Sweden (Cousins, 2008). However, species diversity was not linked with past conditions neither with change in spatial environmental conditions in both studies.

Species evenness was better explained by regressive model including recent spatial environmental heterogeneity. It was positively influenced by the increase in eastness in the study area, as well as negatively affected by the area of woody and pH (KCl). The effect of eastness is probably due to the fact that anthropogenic pressure is reduced with this gradient, while woody area can interfere with dispersal of grassland plants and in addition supply colonization of open meadows by competitive and fast growers species (e.g. Alnus viridis and Salix caprea). This negative effect over plant diversity was also found in Estonia in calcareous wooded meadow (Aavik et al., 2008).

Even if species richness in this study was better explained by model including change in spatial environmental heterogeneity, a moderate response of species

richness to recent spatial environmental heterogeneity was also found. Species richness was negatively related to increase in urban area at 250m spatial extent (e.g. urban, roads), which again evidenced the role of landscape composition. This spatial extent registered the stronger increase in non-habitat area (38.6%), which decreased meadows habitat area affecting both establishment and dispersal of new plant species. As found in German grasslands in a low mountain range region (Dauber et al., 2003), also in this study species richness was lower with northness gradient, while increased with slope due to differences in meadows management respect to flat locations, as observed for managed grasslands in other location of the Italian alps (Marini et al., 2009).
In the overall of this study, best regression models were always composed by spatial metrics, topographic and soil variables, which illustrates the importance of spatial environmental heterogeneity in landscape to plant diversity.

5. Conclusions

In the present study, plant diversity indices of permanent meadows responded differently to components of spatial environmental heterogeneity and were both related to recent and change in spatial environmental heterogeneity in the lowlands of Valtellina Valley. Species richness and Shannon index were best explained by models including change occurred in spatial environmental heterogeneity, while species evenness was best explained by models including recent spatial heterogeneity variables. Species richness respond negatively to changes in habitat area, while Shannon index is higher with increasing landscape diversity, which result from landscape fragmentation and lead to a conflict between species richness and Shannon index requests. Species evenness was most explained by aspect, woody area and soil features. For plant diversity conservation issues in the permanent meadows of Valtellina Valley, a sustainable meadow management is crucial to avoid loss of meadows habitat and maintain landscape diversity through appropriate landscape planning, in particular for urban and bushland areas.

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Appendice I

Appendice I List of species recorded across the 34 selected meadow plots during the vegetation surveys in the lowlands of Valtellina Valley.

Achillea millefolium
Agropyron repens
Agrostis tenuis
Ajuga reptans
Allium sp.
Alopecurus utriculatus
Anthoxanthum odoratum
Arabidopsis thaliana
Arrhenatherum elatius
Artemisia vulgaris
Avenula pubescens
Bellis perennis
Bromus hordeaceus
Bromus sterilis
Capsella bursa-pastoris
Cardamine hirsuta
Cardaminopsis halleri
Carex pairaei
Carum carvi
Centaurea nigrescens
Cerastium glomeratum
Cerastium holosteoides
Chenopodium album
Clinopodium vulgare
Convolvulus arvensis
Crepis biennis
Dactylis glomerata
Daucus carota
Erigeron annuus
Erodium cicutarium
Festuca arundinacea
Festuca pratensis
Festuca rubra aggr.
Galium mollugo aggr.
Geranium molle
Glechoma hederacea
Heracleum sphondylium
Holcus lanatus
Lamium album
Lamium maculatum
Lamium purpureum
Leontodon hispidus
Leucanthemum vulgare
Lolium multiflorum
Lolium perenne
Lotus corniculatus
Lychnis flos-cuculi
Medicago sativa
Myosotis ramosissima
Ornithogalum umbellatum
Pastinaca sativa
Phleum pratense
Pimpinella major
Plantago lanceolata
Plantago major
Poa pratensis
Poa trivialis
Polygonum aviculare
Potentilla reptans
Ranunculus acris
Ranunculus bulbosus
Ranunculus ficaria
Ranunculus nemorosus
Ranunculus repens
Rumex acetosa
Rumex obtusifolius
Salvia pratensis
Secale cereale
Silene dioica
Silene vulgaris
Stellaria graminea
Stellaria media
Taraxacum officinale
Thalictrum minus
Trifolium pratense
Trifolium repens
Trisetum flavescens
Urtica dioica
Valerianella locusta
Veronica arvensis
Veronica chamaedrys
Veronica filiformis
Veronica persica
Vicia cracca
Vicia sativa
Viola tricolor

Chapter 3

Evaluation of land cover classification and change detection techniques based in Landsat imagery data in the Italian alps

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Abstract

This study aimed to evaluate the application of Landsat data for land cover mapping and to assess the effectiveness of change detection techniques in the Valtellina Valley. Two Landsat Tm/ETM+ scenes recorded on 31 August 1989 and 21 June 2001 were used as dataset. Images were geometrically, atmospherically and radiometrically corrected. An hybrid supervised classification was executed to map land cover for both dates. Image differencing and image regression change detection techniques were applied and the accuracy assessment was based in the land cover layers generated. The results show that hybrid approach for land cover classification based in Landsat imagery was highly accurate, while image differencing was detected moderatelly changes occurred in landscape, as well as in urban, meadow and bush land. In conclusion, remotely sensed imagery can be a reliable source of information for alps, although particular attention should be required for image pre-processing and classification, as well as, to the minimize of topography influences for the spectral variability.

Keywords: European Alps; Valtellina Valley; Image differencing; Image regression; Hybrid classification; Remote sensing.

1. Introduction

Satellite-based remote sensing is nowadays a key technology to monitor land cover changes in terrestrial ecosystems (Foody and Mathur, 2004) due to its

capacity to provide information over large geographical extent at reasonable costs (Lunetta, 2006). At a regional scale, Landsat TM/ETM+ are the most frequently used data (Lu, 2007) and for accurate results is necessary to define adequate hierarchical levels for mapping, discrete land-cover units discernible and select representative training sites (Cingolani et al., 2004).
In image classification, supervised approaches were most efficient in map land cover respect to unsupervised classification, but they were highly dependent in the a-priori knowledge of the study area and good training data (Cihlar et al., 1998). However, collect good training data in the field is difficult in mountain regions and the assumption of normal spectral distribution is weak in complex landscapes (Lu, 2007). In such context have emerged the hybrid approaches combining unsupervised and supervised classification methods to discriminate spectral homogeneous areas for optimized training and ground data collection (McCaffrey and Franklin, 1993) to subsequent supervised classifications (Lo and Choi, 2004). In addition, the integration of decision trees in the classification process including e.g. DEM-related variables can also improve the classification accuracy (Franklin et al., 2001).
For change detection issues or to state the differences in a landscape between different times (Singh, 1989), several change detection techniques were developed, although to discriminate changes in a particular location such techniques should compared in order to select the most useful for the specific application (Lu et al., 2004). In particular, four main change detection techniques were used at larger extent: image differencing, image rationing, image regression and change vector analysis (CVA) (Berberoglu and Akin, 2009). All this techniques involve the selecting a threshold for change.
In the European alps, satellite data have been used to map glaciers retreat (Paul et al., 2007) and recently to map land cover (Waser and Schwarz, 2006). However, the applying of remote sensing for mapping and change detection, mostly due the complex landscape.
In this study for the Valtellina Valley our specific aims were:
i) to evaluate the applying of multitemporal Landsat TM/Etm+ for land cover mapping
ii) to test the hybrid classification approach to overcome difficulties in delineating appropriate training samples for complex mountain study areas

iii) to evaluate the image differencing and image regression change detection techniques

2. Material and methods

2.1. Study area

The study area (190 km2) is located in Italy, in the middle Valtellina, Southern Alps (46.10' N, 9.50 E, Fig. 1). The elevation ranges from 250 to 1620 m a.s.l. (above sea level). It is a U-shaped valley carved by glacial erosion during the Quaternary, with a west-east orientation, discharged by the river Adda. Main soil types are Eutric fluvisols, Dystric and Eutric cambisols (FAO, 1988). Owing to the proximity to Lake Como, the climate is temperate continental with mean annual temperature of 11.9°C and precipitation of 970 mm (Sondrio at 298 m a.s.l., 1973-2007). The landscape is composed by five main categories: meadow, urban settlements, vineyard, bushland and orchard. It is strongly influenced by local farming activities, including activities in uplands (Rudini, 1992).

2.2 Data Collection

In this study, two Landsat scenes provided by the Global Land Cover Facility (http://glcf.umiacs.umd.edu/index.shtml) were utilized: Landsat TM 31-08-1989 and Landsat ETM+ 21-06- 2001. These are multispectral images composed respectively by 7 (TM) and 8 (ETM+). These bands are centered in different regions of the electromagnetic spectrum: blue (b1) 0.45-0.515 µm, green (b2) 0.525-0.605 µm, (red) b3 0.63-0.69 µm, near-infrared (b4) 0.75-0.90 µm, mid-infrared (b5) 1.55-1.75 µm, thermal-infrared (b6) 10.4-12.5 µm, mid-infrared (b7) 2.09-2.35 µm, panchromatic (b8) 0.52-0.9 µm. Both scenes were stored in Geotiff file format and the spatial resolution was 28.5 m in all bands with exception for the thermal-infrared (60m) and panchromatic band (14.25 m) for Landsat 2001. In addition, the digital elevation model (DEM 20 m pixel resolution) and two land cover dataset originated from hand digitizing process over orthorectified aerial photograph for 1980 and 2000 were used as auxiliary data.

2.3 Landsat imagery pre-processing

Both images were orthorectified and georeferenced to the Universal Transverse Mercator- UTM, WGS 84 - Zone 32 N. Next, a image-to-image co-registration using 23 ground control points well distributed through the images was applied. A root mean square error (RMS) of 0.06 after resampling with the nearest neighbour algorithm was obtained, which means an accuracy within one pixel in the image. The co-registration procedure is crucial for change detection accuracy. The images were also atmospherically corrected using the dark object subtraction technique (DOS), a relative atmospheric correction effective in change detection and classification studies with landsat data (Song et al., 2001). This correction minimizes the effect of distinct levels of haze and dust in atmosphere in images acquired in different times. Further, a band-by-band radiometric normalization of images was executed through the brightness normalization technique, meaning that if a pixel is unchanged between the different times, the digital number (DN) difference will be zero (Cem et al., 2005). For mountainous locations, radiometric normalization of multi-temporal data is a crucial step, since spatial and temporal differences in illumination and radiance can induce similar surfaces to reflect differently, which can influence the classification results (Kuemmerle et al., 2006). Both, atmospheric and radiometric corrections were effective in this study (Fig. 2).

2.4 Image classification

Landscape was grouped into six land cover types for the classification (agriculture, meadow, urban, coniferous forest, bush and water). The classification process was divided into four phases: image data fusion, training sites and ground truths definition, supervised classification, post-classification. Image data fusion was executed only for the Landsat 2001, using the principal components (PC) sharpening technique. This operation enhanced the spatial resolution from 28.5 m to 14.25 m merging the low spatial resolution bands (28.5 m) with the high spatial resolution panchromatic band (14.25 m) (Liu et al., 2004). Next, training sites and ground truths were defined for Landsat 1989 and 2001. Training data was defined running the iterative self-organizing data analysis (ISODATA) algorithm over the images (Jensen, 1996). This unsupervised classification method clustered each image into 40 spectral homogeneous clusters, which is a higher number respect to the number of land cover types to map, although it can be useful since the total number of spectral classes in the image is unknown (Kuemmerle et al., 2006). Next, the clusters were empirically merged into a unique cluster through visual discriminant association between each cluster and the respective land cover class. Six major clusters with known land cover type and spectrally homogeneous resulted from this operation. Despite, this operation was done only for clusters which unambiguously belong to a unique land cover type, information which was provided by the land cover layers and aerial image dated to year 2000 and 1980. From these major clusters 370 and 332 training sites for year 2000 and 1989 were collected. Each training site was defined at least by 6 pixels. Further, each pair of training sites was tested for the spectral using the Jeffries-Matusita distance measure (Richards and Jia, 2005) and all pairs of training sites presented good spectral separability since the values were higher than 1.9. The only exception was the pair agriculture – bush, which presented 1.88 coefficient. This is probably due to the broad range of land cover types included in the agriculture class, in particular the 'orchards' which present spectral similarity with 'bush'class.

A supervised classification using the training areas and applying the maximum likelihood algorithm (MLH) was executed (Richards and Jia, 2005) and in a post-classification phase, decision trees were used to separate the forest areas from areas affected by shadow and misclassification of meadows into agriculture in the slopes.

This hybrid classification approach integrated the benefits of supervised and unsupervised classification (Kuemmerle et al., 2006). All the procedures were executed in ENVI 4.4 software environment.

2.5 Change detection analysis

Since the aim was to identify change in land cover and the study area is dealing both with bush expansion and anthropic development, the differences between two dates in NDVI vegetation index was followed as approach. Vegetation indices are among the most common and efficient methods in land cover change detection. NDVI was estimated for both Landsat images through the ratio: (near-infrared – red) / (near-infrared + red).

Image differencing and image regression were the techniques used to detect change in NDVI. The first technique is the subtraction between the two NDVI images and the pixels with small variation in reflectance are located around the mean, while pixels with higher change in reflectance are located in the tails of the distribution (Singh, 1989). The second technique considered the NDVI in the late time as a linear function NDVI in time one. It considerer the differences in mean and variance between pixel values for different dates in order to reduce the temporal differences in atmosphere condition or sun angle, which can affect change detection accuracy. In this study, a linear relationship was established between the images the (independent variable) NDVI 1989 and the NDVI 2001 (dependent variable). The parameters of this linear regression function were applied to NDVI image of 1989 and predicted a NDVI image for 1989. Next, the predicted image and subtracted NDVI image for 2001 were subtracted.
For the definition of change threshold for image differencing and image regression techniques, different standard deviations intervals against the ground data were tested in order to identify the suitable limiting value. Later, the threshold was applied for each change detection technique and '0' classified no change areas and '1' changed areas.

2.6 Accuracy assessment of image classification and change detection procedures

The accuracy of the image classification and change detection techniques applied in this study was estimated with an error matrix and kappa analysis, which are statistics common discrete multivariate statistics applied in remote sensing (Congalton, 1991). For the image classification were created ground truths and the classification was compared against ground truths. For change techniques, the process was identical, however the ground data used to make the comparisons was derived from the classification of the landsat images for 1989 and 2001.

Fig. 1 Study area located in the Valtellina valley, Southern Alps.

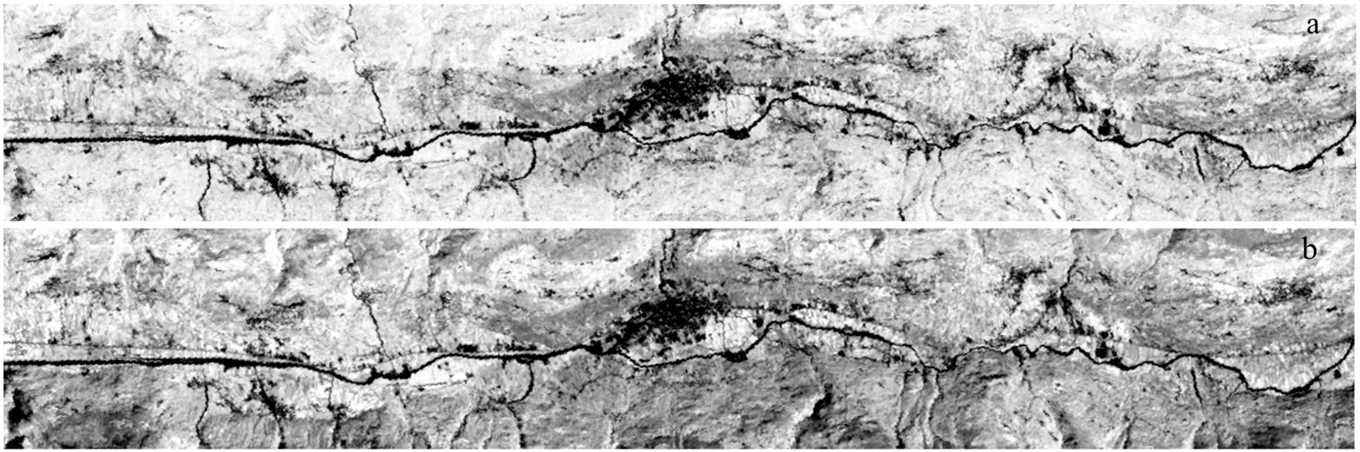

Fig. 2 NDVI image from 1989-08-31 before (a) and after (b) radiometric normalization.

3. Results and discussion

3.1 Land cover classification

The land cover classification showed that Valtellina valley was mostly cover by agriculture, bush and meadows land in 1989 (Fig. 4), while in 2001 bushland dominated landscape and both agriculture and meadows decreased considerably. From 1989 to 2001, the development of antropic uses was observed, urban sprawl in landscape was registered.

The hybrid classification applied for 1989 and 2001 images show a good performance, which resulted in two consistent land cover maps for Valtellina Valley (Fig. 3). An overall classification accuracy of 80.3% for 1989 and 88.6% for 2001 was estimated, with kappa coefficients of 0.73 and 0.84, respectively. However, the land cover classification for 2001 presented best accuracies than 1989.

The users and producers accuracy varied between classifications and between each land cover class. For 1989, meadow, water and coniferous forest registered users and producers accuracy up to 80% (Table 1), while for 2001, urban, bush and water classes were up to such accuracy limit (Table 2). The land cover classes urban, bush and agriculture for 1989 and agriculture, coniferous forest and meadow for 2001 presented accuracies less than 80% (Table 1).

This is explainable by the difficult in classify areas with mixed classes as occurred with meadow and agriculture or bush land and agriculture land, in our study area. This classes presented spectral similarities that difficult its spectral separability, which suggest some conflict between our thematic class and its spectral separability for mapping. In fact, other studies illustrated the difficulty in classify correctly bush land due the overlap with grassland (Kuemmerle et al., 2006). The high spectral heterogeneity of class composition, as for agriculture class, which included orchards, vineyards and crops, can also explain less accuracy. The unsupervised clustering executed before the maximum likelihood classification was particularly important, since it helped to identify homogeneous spectral classes and reduce bias in the training data.

Accuracy assessment was made through ground truth defined based in aerial images and automatic random sampling, since to obtain data for all areas and for 1989 image was impossible. However, the ground truths were cautiously constructed to be independent from the training data and to cover the physiographic range of the study area.

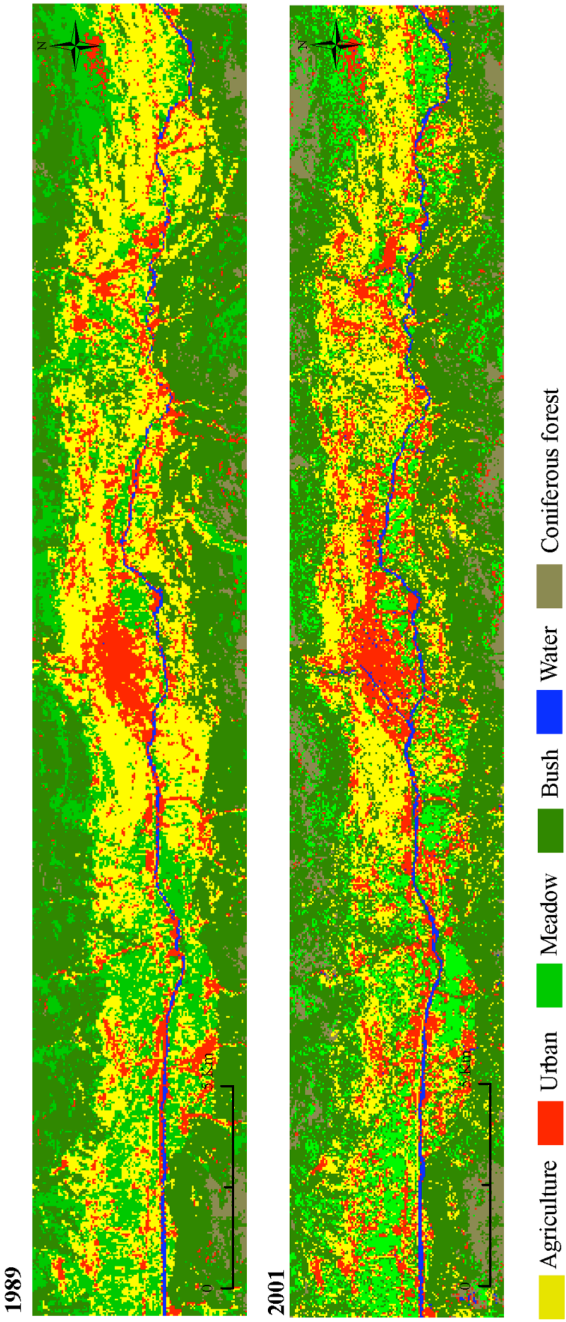

Fig. 3 Land cover maps for the Valtellina valley.

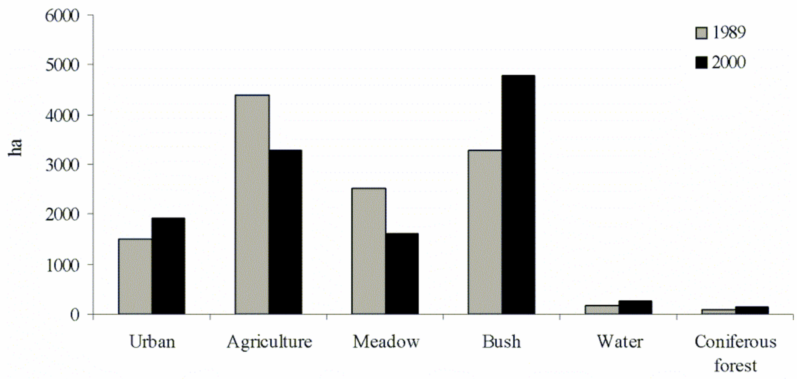

Fig.4 Land cover composition for 1989-08-31 and 2001-06-21.

3.2 Accuracy of change detection techniques

The change detection techniques applied in this study were commonly applied in land cover change studies. The threshold applied for NDVI to identify change was set to 0.7 standard deviation, since it was the most adequate respect to the ground data after testing other standard deviation intervals: 2s, 1.5s, 1.s, 0.8.s, 0.5s (Fig. 5). The change detection results for each technique were cross-tabulated against the ground data 2001-1989.

In this study, the image differencing technique provided best results than image regression and it was the most accurate technique to detect land cover variation in the landsat images (Table 3). The performance of the techniques can be considered moderate, since overall kappa coefficient was in the range 0.40–0.80 (Congalton, 1991). The techniques show similar performances in detect change in urban, meadow and bush, while image differencing was definitively better in estimate the overall changes in the Valtellina Valley. Changes in urban were most accurately estimated than changes for meadow and bush. The level of accuracy found in this study were in line with other studies (Berberoglu and Akin, 2009).

In the overall, the accuracy of remote sensing change detection techniques in mountain regions can be limited by several aspects: differences in moisture and atmospheric conditions in the satellite data per se, physiographic attributes, differences in vegetation composition, phenology. Such aspects enhance the spectral variation between scenes and in addition its necessary to account with similar reflectance of major land covers which limited the spectral separation (e.g. urban vs vineyard, crops vs meadow). By last, the image processing methods can influence also the results. All this factors are consistent for this study, since we used as reference data for change an image resulting from post-classification comparison of the land cover scenes for 1989 and 2001, which were not completely accurate. In fact, the accuracy of the post-classification comparison highly depends on the accuracy of the initial classification. Misclassification errors in the original images always can make the results obtained using post-classification comparison less accurate and per se influence the estimate for the other change detection techniques (Teng et al., 2008).

Table 1 Confusion matrix for the hybrid classification of Landsat TM 1989-08-31.

		Reference data							
	Classes	Urban	Agriculture	Meadow	Bush	Water	Coniferous forest	total	User's accuracy (%)
Classified data	Urban	999	239	0	89	2	12	1341	74.5
	Agriculture	580	2861	262	858	0	0	4561	62.7
	Meadow	97	306	2762	292	0	0	3457	79.9
	Bush	0	0	67	5876	0	404	6347	62.73
	Water	0	0	0	0	172	2	174	98.9
	Coniferous forest	0	0	0	2	0	455	457	99.56
	total	1676	3406	3091	7117	174	873	16337	
	Producer's accuracy (%)	59.6	84	89.4	82.6	98.9	52.1		
				Overall accuracy= 80.3%			Kappa coefficient= 0.73		

Table 2 Confusion matrix for the hybrid classification of Landsat ETM+ 2001-06-21.

		Reference data							
	Classes	Urban	Agriculture	Meadow	Bush	Water	Coniferous forest	total	User's accuracy (%)
Classified data	Urban	5365	910	3	3	122	78	6481	82.8
	Agriculture	476	8168	7	890	0	0	9541	85.6
	Meadow	23	1101	6580	928	0	0	8632	76.2
	Bush	2	251	1275	29449	0	392	31369	93.9
	Water	40	0	0	0	833	95	968	86.1
	Coniferous forest	0	2	0	118	0	1672	1792	93.3
	total	5906	10432	7865	31388	955	2237	58783	
	Producer's accuracy (%)	90.8	78.3	83.7	93.8	87.2	74.8		
				Overall accuracy= 88.6%			Kappa coefficient= 0.84		

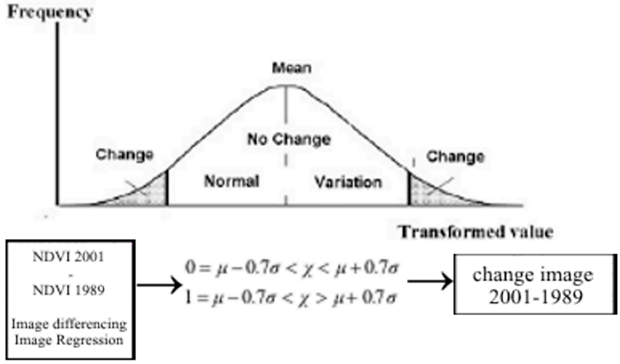

Fig. 5 Treshold applied for change.

Table 3 Comparison of the accuracy of the change detection techniques.

Change detection techniques	Change level 1989 - 2001			
	Urban	Meadow	Bush	All changes
Image differencing	0.72	0.55	0.5	0.63
Image regression	0.71	0.52	0.47	0.47
	Overall Kappa coefficient			

4. Conclusions

This study aimed to evaluate the application of Landsat data for land cover classification and to assess the performance of common change detection techniques in the Valtellina Valley. The focus was to contribute to elucidate about the potential application for the Alps mountains. It was shown that hybrid approach for land cover classification based in Landsat imagery was highly accurate, while image differencing change detection technique detected moderately changes occurred in landscape, as well as in urban, meadow and bush land. We suggest that remotely sensed imagery can be a reliable source of information for alps, however particular attention is required for image pre-processing and classification.

Chapter 4

Agro-ecological dynamics of mountain agricultural systems in Lombardia: the case study of Valtellina Valley

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Abstract

Aerial photogrammetry, satellite remote sensing, and ecological models were used to quantify land cover changes during the period 1980-2000 and to monitor recent variation in permanent hay meadow net primary production (NPP) in the Valtellina Valley (Southern Alps). The results showed a significant reduction in agricultural land in favour to human settlements, mainly associated to the loss of permanent hay meadows. NPP mapping between 2002 and 2007 years showed that less productive meadows are commonly located in proximity to areas affected by meadow loss in the last 30 years. The combining of historical landscape analysis with post-change characterizing of physical-environmental attributes of ecosystems may be useful to develop effective concepts for future ecosystem management and

1. Introduction

Agricultural land in the European Alps is facing a continuous trend of abandonment since the Second World War in connection with the marginalization of traditional farming systems (Batzing, 1994). The industrialization of agriculture and the implementation of the Common Agricultural Policy (CAP) have lead to a progressive intensification of agriculture on more fertile and accessible areas, usually located in the low lands, at the expenses of marginal and less productive lands, typically situated at higher elevation and with reduced accessibility (MacDonald et al., 2000). Such changes over the territory introduced several environmental and socio-economical modifications, which ultimately affected not only the ecosystem goods and services provided by traditional agricultural practices, but also the ecosystem "functioning", or the internal regulation of ecosystems. In this context, the study of land use and land cover changes can be extremely useful to understand past conditions and current trends, and to formulate future scenarios for agricultural land development. In the European Alps, the integration between the comprehension of long term processes and the

implementation of specific strategies to monitor the current dynamics of agro-ecosystems can provide an effective instrument to support land management policies (Boschetti et al., 2007, Fava et al., 2010). Remote sensing data, from aerial photograph to satellite images, together with agro-ecological models, can give a significant contribution for both purposes (Kerr and Ostrovsky, 2003), since they provide a synoptic, multi-scale and recurrent source of information about land cover and surface biophysical properties.
Here, we report the case study of the Valtellina Valley (Italy, Southern Alps), as a representative and relevant example in the European Alps, of a mountain region affected by agricultural abandonment/intensification dynamics in connection with traditional farming systems decreasing and human settlements expansion. We used aerial colour photographs of the years 1980 and 2000 to characterize land cover changes over the study area, and to individuate the most important dynamic process occurred in this period. According to the results of the previous analysis, we focused on permanent hay meadows and we tested the use of MODIS satellite remote sensing data and radiation use efficiency (RUE) models for monitoring recent meadow net primary productivity (NPP) dynamics.

2. Methods

2.1. Study area

The study area covers about 120 km2 and is located in the bottom valley of medium Valtellina (46.10’ N, 9.50’ E), northern Italy. The bottom valley is mainly flat (slope <5%) with an east-west orientation and its elevation ranges from 250 to 380 m a.s.l. Geologically, its “U” shape features resulted from erosion and deposition processes during the Quaternary with sediment accumulation in the less steep areas. The main soil types are Eutric Fluvisols, Dystric and Eutric cambisols (FAO, 1988). The climate is temperate-continental with mean annual air temperature values of 11.9°C and mean annual precipitation of 970 mm, registered at climate station of Sondrio (1973-2007) (46.10’ N, 9.50’ E; 298 m a.s.l.). The features of Valtellina Valley have origins in the traditional Romanic culture, characterized by a self-sufficient mixed farming system practiced from the villages in the low lands, in the subalpine pastures and in the summer settlements at higher elevations during the transhumance (Maurer et al., 2006). The land structure is fragmented in small-size parcels due to the prevalent hereditary partitioning systems of parcels (Batzing, 1994). This strongly affected the shape of the landscape until the beginning of the 20th century, when socio-economic changes determine a progressive abandonment of these systems.

2.2. Land cover change analysis

Land cover change analyses were derived from the combination of image photogrammetric processing, geographical information systems and spatial statistical analysis. We defined 11 land cover classes (permanent meadows, orchards, water, urban settlements, industrial settlements, vineyards, hedges, uncultivated, cultivation, roads, woody), adapting the European Nature Information System (EUNIS) habitat classification (http://eunis.eea.europa.eu/habitats.jsp) for the study area. Hand-digitizing processing was performed over two digital ortho-rectified aerial photos for years 2000 (scale 1:36 000) and 1980 (scale 1:20 000) in ArcGIS 9.1 (ESRI, 2005). The aerial photos (1m resolution), acquired by the Region of Lombardia, were registered and geo-referenced to the national grid system Gauss Boaga- Zone 1. The imagery processing resulted in two land cover maps for 1980 and 2000 years in vector format. In order to quantify land cover transition during the period of analysis, both vector maps were transformed into raster format in ArcGIS 9.1 (cell size 10m) and exported to Idrisi Andes GIS environment (Clark, 1986). Subsequently, a post-classification change detection technique was applied, based on a cross-tabulation algorithm (Lu et al., 2004), which executed a cross-correlation between the two independent classified images (1980/2000). The resulting cross-tabulation table identified the frequencies of classes that remained the same (frequencies in the diagonal) or have changed (off-diagonal frequencies).

2.3. Net primary productivity analysis

In order to monitor the dynamics of net primary productivity of hay meadows in the study area, we integrated the MODIS (Moderate Resolution Imaging Spectroradiometer) satellite data and the radiation use efficiency (RUE) model

MOD-17 (Running, 1999). The algorithm used for generating the MODIS NPP standard product was modified to increase the spatial resolution of the NPP estimates from 1 km to 250 m, which is affected by the complex topography and fragmentation of the study region. It follows a brief description of the most important methodological steps, while an exhaustive description can be found in (Colombo et al., 2009). The basic formulation of the model is the following:

[Eq. 1] $$GPP(t) = [\varepsilon_{max} \cdot f(T_{min}).f(VPD)] \cdot fAPAR(t) \cdot PAR(t)$$

where GPP [gC day-1] is the gross primary productivity, ε_{max}[gC MJ-1] is the maximum radiation use efficiency coefficient, $f(T_{min})$ and f(VPD) are scalar functions [0,1] of the minimum daily temperature (T_{min},°C) and vapour pressure deficit (VPD, Pa), introduced to keep in account reduction in ε_{max} due to non-optimal growth conditions. From daily GPP values, net photosynthesis (PSNnet, gC day-1) is obtained subtracting the maintenance respiration. NPP is estimated annually as the cumulative sum of PSNnet less the cost of annual maintenance and growth respirations. The model requires several eco-physiological parameters specific for grasslands. The detailed list of parameter values used in this study can be found in (Colombo et al., 2009). According to Eq. 1 the model input variables are the fraction of photosynthetically active radiation absorbed by plants (fAPAR), the incident PAR [MJ m-2 day-1], and the minimum, maximum and mean temperature values, used to derive the scalar functions $f(T_{min})$ and f(VPD). Daily PAR values were obtained from incident global radiation (GRAD) measurements at the Sondrio meteorological station, applying an empirical relation (PAR=GRAD*0.48, Tsubo and Walker, 2005). Temperature values were obtained from the same station. Finally, fAPAR values were obtained from Normalized Difference Vegetation Index (NDVI), derived from MODIS satellite data, through the empirical relationship proposed by Sellers (Sellers et al., 1994). NDVI time series were generated from the standard 16-day 250m spatial resolution MODIS vegetation indices product (MOD13Q1), downloaded for the period 2001-2007 from the NASA Warehouse Inventory Search Tool (WIST https://wist.echo.nasa.gov). To exclude low quality data and reduce noise in the NDVI time series, the filtering algorithm proposed by Chen and colleagues was used (Chen et al., 2004). Finally, a linear relationship between consecutive observations was assumed to obtain daily NDVI data for the whole study period. The final outputs of this procedure are hay meadows daily GPP and PSNnet, and annual NPP maps, over the whole study area.

3. Results and discussion

3.1. Land cover change 1980-2000

The land cover analysis revealed an important decrease in agricultural addressed land in favour of land cover types connected with the secondary and tertiary economical sectors. According to Table 1, in the earlier date (1980) the landscape of Valtellina was largely covered by agricultural land (66%), mainly hay meadows (36%) and vineyards (17%). Human settlements (urban, industrial, roads) covered approximately 17% of the area. In the late date (2000), the overall composition of landscape did not change radically: agricultural land was still the main cover type (55%) and human settlements were the second most represented land use class (23%). Nevertheless, during the period of analysis, the ratio between agricultural land and human settlements has changed from 4/1 to nearly 2/1, suggesting significant urbanization processes and loss in agricultural lands. In fact, the land cover change analysis evinced a net land cover conversion in 22,8% of the bottom valley, with a mean velocity of change of 88,7 ha year-1. Looking into the nature of this change, results indicate a strong tendency to the reduction of the permanent meadow lands (-539 ha), which represents about 30% of the overall converted area. Moreover, we observed the decrease in vineyards (-276 ha), the increase of the urbanized surfaces (+333 ha), industrial settlements (+140 ha), uncultivated land (+146 ha), and woody land (+215 ha) (Table 1). Considering the overall of agricultural land, we estimated a loss of 846 ha, contrasting with the gain of 496 ha from human settlements. These results identify a clear dichotomous trend in land cover: the retreat of agriculture addressed land and the increase of human settlements, which have transformed the Valtellina landscape during the period of analysis. These tendencies of urban expansion and conversion of traditional agricultural systems affected mainly hay meadow cover, which account for the 54% of urban and for the 63% of industry growth (Table 2 and Figure 1), as already reported in the Alps (Peter et al., 2008). Another remarkable result is the net conversion of agricultural land to woody (234 ha) and uncultivated (152 ha) areas (Table 2). Recent studies in mountain regions related this process to agricultural abandonment in marginal high elevation or slope areas (Rutherford et al., 2008). However, in this study case the same tendency was observed in the bottom valley. If confirmed from other studies, this result would evidence a relatively new tendency of agricultural loss even in the more fertile and favourable areas.

Overall, the fast decline of permanent hay meadow cover is strictly associated to both processes, and represents the most important agro-ecological change impacting the area. Meadow loss and decline is a particular damaging agro-ecological process, because of the well-recognized multifunctional role of grassland ecosystems in the European Alps (e.g. high quality local forage provisioning, biodiversity maintenance, carbon sequestration, soil erosion protection, touristic and recreational attractiveness) (Becker et al., 2007, Peter et al., 2008).

Table 1. Land cover type area, proportion, and changing rate between 1980 and 2000.

Land cover type	1980		2000		Changing rate
	area (ha)	proportion (%)	area (ha)	proportion (%)	area (ha $year^{-1}$)
Per. Meadow	2915.9	36.4	2377.0	29.7	-26.9
Cultivation	350.9	4.4	377.9	4.7	1.4
Orchard	610.8	7.6	552.7	6.9	-2.9
Vineyard	1396.2	17.4	1120.2	14.0	-13.8
Woody	860.5	10.7	1075.1	13.4	10.7
Uncultivated	164.4	2.1	310.2	3.9	7.3
Water	333.4	4.2	319.8	4.0	-0.7
Urban	978.8	12.2	1311.9	16.4	16.7
Industry	170.7	2.1	310.5	3.9	7.0
Roads	221.1	2.8	244.5	3.1	1.2
Hedge row	6.4	0.1	9.3	0.1	0.1
Total	8009.1	100.0	8009.1	100.0	-

3.2. NPP monitoring 2002-2007

Productivity can be considered as reliable indicator of ecosystem conditions and it is directly linked to the economic value of a specific crop (Assessment, 2005). Indeed, a cost-effective methodology for monitoring hay meadow NPP at landscape scale in Valtellina was implemented, as a tool to provide updated information about the status of hay meadows in the area. An example of measured NDVI and modelled GPP curves for a single 6,25 ha image pixel and a single year is shown in Figure 2. The seasonal profile of GPP has a multiple peak distribution, likely related to the effect of meadows cut. From daily GPP data, annual NPP maps were generated, accounting for maintenance and growth respiration. Estimated mean annual NPP values in the study area ranged from 8,5 t ha-1 in 2004 to about 10 t ha-1 in 2001, with maximum values reaching 14,4 t ha-1. Even if a quantitative validation of these results was not possible, obtained values are comparable with those presented by Riedo in the Swiss Alps (Riedo et al., 2000). NPP temporal trends were not observed in the study period. However, the temporal variability of the mean NPP was significantly correlated with cumulate precipitation between the start of February and the end of August ($r=0,92$, $p<0,001$), indicating the key role of rainfall regime in the study area. Spatial anomaly maps of 5 years (2003-2007) averaged NPP were generated to evince areas with low productivity, likely abandoned or in poor conditions (Figure 3). A strong spatial variability was observed, but generally negative anomalies resulted in isolated and fragmented meadows, suggesting a potential connection between meadow fragmentation and declining condition in the area. Moreover, the comparison of the meadow loss (Figure 1) and NPP (Figure 3) maps evidenced that less productive meadows are generally located in proximity to areas characterized by meadows loss in the years 1980-2000.

Although the methodology proposed needs validation and suffers from the limited spatial resolution of MODIS data, which may not be suitable in highly complex Alpine landscapes, the results obtained are promising and open interesting perspectives for future operational applications.

Table 2. Land cover change matrix from 1980 to 2000

1980 / 2000	Per. Meadow	Cultivation	Orchard	Vineyard	Woody	Uncultivated	Water	Urban	Industry	Roads	Hedge row	Total 1980
Per. Meadow	2092.32	194.25	65.61	17.45	153.14	89.79	5.64	184.49	90.7	18.13	4.34	2915.86
Cultivation	108.55	140.47	3.92	2.97	21.02	31.95	1.45	20.38	16.13	3.17	0.88	350.89
Orchard	60.82	9.79	436.91	11.08	17.04	6.31	0.02	60.73	6.17	1.78	0.14	610.79
Vineyard	63.76	18.93	41.53	1069.62	90.5	48.69	00.03	60.52	0.21	2.43	0.05	1396.24
Woody	21.17	8.48	2.06	15.95	722.23	47.32	8.6	12.12	19.39	2.51	0.66	860.49
Uncultivated	18.1	3.6	0.6	1.67	42.49	66.37	3.47	8.8	16.14	3.07	0.07	164.38
Water	1.65	0.27	0.17	0	17	12.77	297.5	0.47	2	0.94	0.63	333.43
Urban	5.95	0.55	1.24	1.32	4.64	1.01	0.21	948.73	8.79	6.29	0.09	978.82
Industry	1.87	0.62	0	0	3.97	4.9	2.77	5.21	149.64	1.58	0.12	170.68
Roads	1.41	0.35	0.47	0.15	1.21	0.65	0.07	10.39	1.34	204.6	0.49	221.10
hedge row	1.42	0.6	0.19	0	1.89	0.43	0	0.04	0.01	0.01	1.83	6.42
Total 2000	2377.02	377.91	552.7	1120.21	1075.13	310.19	319.8	1311.9	310.52	244.5	9.3	8009.10

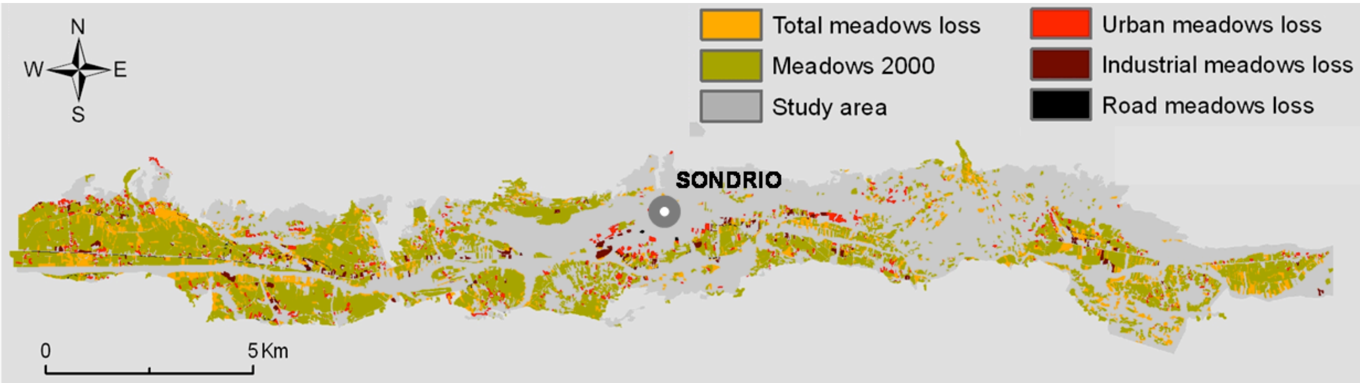

Figure 1. Hay meadow conversion to urban, industry and roads between 1980 and 2000

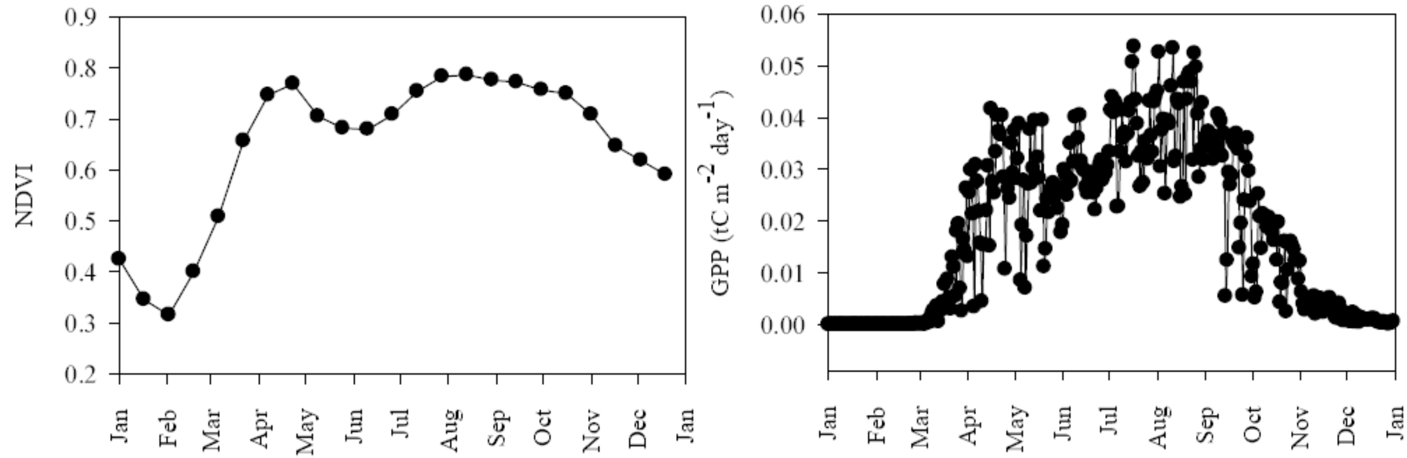

Figure 2. measured MODIS NDVI (left) and modeled GPP for a single image pixel in 2006

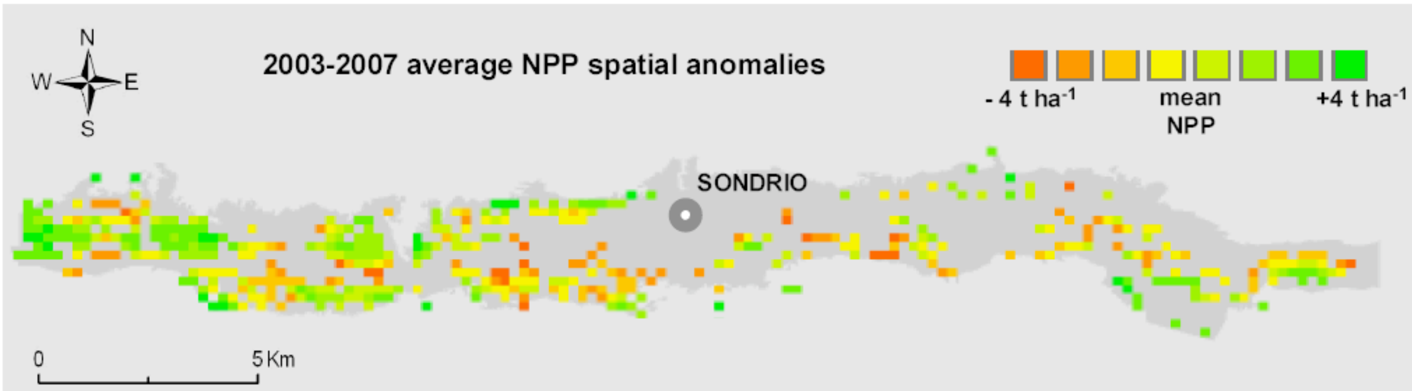

Figure 3. 2003-2007 average NPP anomalies

4. Conclusions

Agricultural land cover trends in the Valtellina bottom valley illustrate that the set of agricultural policies implemented according to the EU PAC are probably not the most suitable for the local reality. Agricultural land has being seriously reduced between 1980 and 2000, in connection with a strong expansion of human settlements. Permanent hay meadows were particularly affected by this process, despite the relevant ecosystem services they provide, denoting reduced interest in conservation of agriculture addressed lands.

The combination of land cover analysis with remote-sensing derived NPP data proved to be effective in relating land cover trends and ecosystem status, which is an essential task for ecosystem monitoring and for implementing effective conservation policies in the European Alps.

Appendice

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Fine-scale assessment of hay meadow productivity and plant diversity in the European Alps using field spectrometric data

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ABSTRACT

The potential of field hyperspectral remote sensing data for non-destructive assessment of hay meadow biomass and vascular plant diversity has been investigated. Spectrometric and agronomic data were acquired at peak biomass over 34 sites distributed at diverse elevation and slopes over an area of 220 km^2 in the Central Alps (Valtellina, Northern Italy). Different modelling approaches were tested to evaluate the predictive performance of spectral measurements: (i) the use of two band ratios of reflectance as input in ordinary least square regression models and (ii) the use of all reflectance bands as input in multivariate partial least square regression models. Each model was subjected to leave-one-out cross-validation and evaluated using the cross-validated coefficient of determination and the root mean square error. Fresh biomass and fuel moisture content were predicted with an average error of <20%, while Shannon Diversity Index and plant species richness were predicted with an average error of <15%, with no relevant differences between the two modelling approaches. Best models for plant diversity indicator prediction were based on chlorophyll/nitrogen sensible bands in the blue and red spectral regions. This observation together with the apparent negative correlation between hay meadow plant diversity and canopy chlorophyll/nitrogen content in the study area suggests a potential connection between reflectance and plant diversity indicators based on meadow biochemical properties.

1. Introduction

Permanent hay meadows are the source of a wide range of public goods and services, including high quality forage, protection from natural hazards, recreational and tourism opportunities (Peter et al., 2008), and greenhouse gases sequestration (Gilmanov et al., 2007). Nowadays, European grasslands are disappearing at an alarming rate (12.8% from 1990 to 2003, according to FAO, 2006) and are among Europe's most threatened ecosystems. As a consequence, permanent hay meadows are protected in the European Natura 2000 network, which comprises Sites of Community Importance (SCI), designated under the Habitats Directive (92/43/EEC of 21 May 1992). Developing new methods for spatial characterization and monitoring of hay meadow biophysical and ecological properties is thus fundamental for the sustainable management, valorisation and conservation of their productive and ecological functions.

Several studies have shown that narrow-band vegetation indices derived from hyperspectral remote sensing data provide essential information for assessing biophysical (Thenkabail et al., 2000; Mutanga and Skidmore, 2004) and biochemical characteristics of vegetation (Fava et al., 2009). Nevertheless, vegetation indices are generally calculated using few spectral bands, underutilizing the information provided by hyperspectral sensors, which acquire spectral data in tens to hundreds of bands. Indeed, recent studies have focused on multivariate statistical models, like partial least square regression (PLSR), to exploit all the information available from hyperspectral data and to use several bands in model development (Schmidtlein and Sassin, 2004). Yet a limited number of studies have investigated the use of PLSR models for grassland biophysical and biochemical properties assessment (Cho et al., 2007; Darvishzadeh et al., 2008), and research is needed to understand if these statistic models can improve the predictive potential of hyperspectral remote sensing data compared to traditionally used vegetation indices.

Parallel to the research of methods for estimating grassland biophysical and biochemical properties, in the last few years

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growing interest has been addressed toward the use of remote sensing for biodiversity assessment (Gillespie et al., 2008). Most studies have evaluated biodiversity through indirect approaches based on habitat classification mapping (Nagendra, 2001), which have limited applications for fine-scale diversity assessment established through quantitative indicators, such as plant species richness (PSR) or Shannon Diversity Index (SDI). Hyperspectral sensors, characterized by high spectral and spatial resolution, may offer new potential for overcoming these limitations and for developing biophysically-based approaches for fine-scale biodiversity assessment (Carlson et al., 2007; Nagendra and Rocchini, 2008), based on a direct connection between remotely-sensed data and diversity indicators. Recent studies found significant relationships between hyperspectral indices, PSR and SDI in forest ecosystems (Carlson et al., 2007; Kalacska et al., 2007), in American tall-herb grasslands (Carter et al., 2005), and wet meadows (Lucas and Carter, 2008), using airborne hyperspectral sensors.

Based on these considerations we planned a field experiment with the objective to evaluate the potential of field hyperspectral spectrometric data for assessing at the same time hay meadow biomass (fresh biomass weight and fuel moisture content) and plant diversity indicators (PSR, SDI). To address these issues we compared the performance of two modelling approaches: (i) linear regression models, with narrow-band indices (i.e. two band reflectance ratios) as independent predictors and (ii) multivariate PLSR models, using all reflectance bands (400–1000 nm range, 1 nm interval) as independent predictors.

2. Materials and methods

The study area comprises the central portion of Valtellina valley and its main northern lateral valley, Val Malenco (Northern Italy). The two valleys have an east–west and south–north orientation and cover some 220 km^2. Their climate is continental, with mean annual temperature of 11–12 °C and rainfall of 970 mm (Sondrio, 307 m a.s.l.), the growing season spanning from April to September. Hay meadows represent the main land agricultural type of the region. According to Gusmeroli et al. (2008), four main hay meadow communities can be identified in the study area: lowland mesophilous meadows (*Pastinaco-Arrhenatheretum* Passarge 1964), submontane slightly thermophilous meadows (*Ranuncolo bulbosi-Arrhenatheretum Ellmauer in Ellmauer et Mucina* 1993), mountain meadows (*Trisetetum flavescentis* Rübel 1911), and transition meadows in the montane belt, ascribed to *Agrostio-Festucion* Puscaru et al. 1956. The presence of high representative and widespread mountain and lowland hay meadows belonging to *Arrhenatherion elatioris* and *Polygono-Trisetion* alliances (EU habitats 6510 and 6520, according to Habitats Directive) was also documented. Lowland hay meadows are generally mown 3–4 times per year, submontane and montane meadows 3 and 2–1 times per year, respectively. In the bottom valley meadows are generally subjected to more intensive agricultural use (i.e. fertilization treatments, mechanical farming operations), while in the slopes they are gradually abandoned, leaving space to woody species encroachment.

2.1. Field measurements

Plant species composition and diversity indicators were derived from an extensive phytosociological study conducted by Gusmeroli et al. (2008) in 2005 on 210 hay meadows in the study area, using the Braun-Blanquet methodology (Braun-Blanquet, 1965). All vascular plants were detected in 10 m $\times$ 10 m plots and the percentage cover was visually estimated. Plot location was selected randomly in each meadow, with a minimum distance of 10 m from the meadow border, in order to reduce edge effects. From these data, plant species richness (PSR) was measured as the sum of all taxa identified in each plot, while Shannon Diversity Index (SDI) was measured as [Eq. (1)]:

$$\mathrm{SDI} = \sum_i p_i \ln(p_i) \tag{1}$$

where p_i is the proportional abundance of the ith species. Each plot was geo-located with a Differential GPS (Trimble GeoXT). In 2006, among the 210 relevés, 34 sampling sites representative of the variability of hay meadow species composition in the region were selected for the experimental work. The exact location of the previous year phytosociological survey plots was individuated by Differential GPS, and it was verified by an expeditive analysis that no change in hay meadow botanical composition occurred.

Plant biomass was sampled at meadow first biomass peak, just before the first hay cut. This was made in order to characterize meadow production at the same growth stage, independently from the site elevation and exposure. To meet this task, four field surveys were performed in 2006, in May 15th, June 5th, June 15th, and July 5th. Fresh biomass (FB, t ha^{-1}) was clipped along a linear transect 7 m long and 1 m wide and weighted immediately after cutting. A sub-sample of the total biomass was used for biomass dry weight (DB, t ha^{-1}) determination after drying in ventilated oven at 70 °C until constant weight. Finally, the biomass water content, expressed as fuel moisture content (FMC, %), was calculated as the difference between FB and DB divided by FB.

Spectrometric data were collected just before biomass sampling using an ASD Fieldspec HandHeld spectroradiometer. This instrument measures the target radiance in the visible and near-infrared region (325–1075 nm) with a Full Width Half Maximum (FWHM) of 3.5 nm and a spectral resolution of 1 nm. A cosine diffusor foreoptic was mounted on the instrument probe and reflectance was calculated as the ratio between vegetation irradiance (sensor pointing nadir) and sky irradiance (sensor pointing zenith). The instrument was placed on a tripod with a rotating horizontal arm in order to minimize operator influence on the incident radiance. The foreoptic was positioned 70 cm above canopy height. Since the radiance upwelling from the target is "weighted" by cosine law in the measured upwelling irradiance, 90% of the signal came from a circular area having a radius of 1.92 m. Spectra were collected in clear sky conditions around solar noon every 1.4 m along each transect (five measurement spots per transect). Each spectrum was calculated as the average of minimum 10 readings. Transect mean reflectance and coefficient of variation (CV) were finally calculated.

2.2. Data analysis

A correlation analysis between transect mean reflectance in the 400–1000 nm range and all the investigated variables was first performed. For diversity indicators the correlation analysis was extended also to the transect CV of reflectance. Statistical significance of the correlation was evaluated with the Student's t-test.

An ordinary linear least square regression (OLSR) analysis was performed between narrow-band reflectance ratios and the investigated variables. Narrow-band ratios (Ri/Rj, where R is reflectance and i and j are sensor bands) were calculated using all the possible two band combinations from 400 to 1000 nm. In order to evaluate vegetation index saturation an exponential fit was also tested.

Finally, a partial least square regression (PLSR) (Wold, 1966) analysis between spectral reflectance and the investigated variables was carried out. In order to avoid model over-fitting, the optimal components of the PLSR were selected minimizing the cross-validated root mean square error ($\mathrm{RMSE_{CV}}$) and adding an extra component only if this corresponds to a reduction of $RMSE_{CV}$ over 2% (Cho et al., 2007; Darvishzadeh et al., 2008). PLSR was run in two ways: the first using as input variables the full reflectance spectrum. Secondly, since PLSR can be enhanced by eliminating irrelevant input variables (Kubinyi, 1996; Martens and Martens, 2000), a subset of spectral bands was selected. Optimal input bands were individuated by a two step procedure: a first PLSR was conducted and well spaced bands with emerging weighted regression coefficients or high significance of the Marten's uncertainty test (Martens and Martens, 2000) were selected. Subsequently PLSR analysis was repeated using only the selected bands for final model development, according to Schmidtlein (2005). Cross-validation was computed with the leave-one-out method (LOO), where each sample is excluded in turn and the model is calculated with all the remnants samples and used to predict that sample. Model predictive performances were evaluated using the $RMSE_{CV}$, and the cross-validated coefficient of determination (R^2_{CV}).

3. Results

3.1. Correlation analysis

FB, FMC, SDI, and PSR had large variability in the measured data set, reflecting the differences in elevation and slope among the experimental sites (Table 1).

According to the correlation analysis, FB and FMC decreased with increasing elevation and slope, while an opposite behaviour was observed for plant diversity indicators (Table 2). Inter-correlation among FB and FMC was high, as for diversity indicators. SDI and PSR were significantly correlated with FB and FMC, but correlations were lower compared to inter-correlations within the two variable groups.

Large variations were also observed in the reflectance spectra collected across experimental sites (Fig. 1). While standard deviation (SD) resulted greater in the near-infrared (NIR) region, the coefficient of variation (CV) had two peaks in the visible region, near 400 nm and between 600 and 680 nm.

The correlation analysis between FB, FMC and reflectance measured at each wavelength indicated a significant positive correlation in the NIR (760–1000 nm) and red-edge (720–760 nm) spectral regions, while a negative correlation was observed in the red (600–700 nm) (Fig. 2). PSR was significantly and positively correlated with bands in the red and negatively correlated with NIR bands. SDI showed a significant correlation only in the NIR. Both variables had no correlation with transect CV of reflectance. For this reason transect CV was omitted in further data analysis.

Table 1
Descriptive statistics of FB, FMC, SDI, PSR, elevation, and slope in the field sampling sites ($n = 34$).

	FB ($t\,ha^{-1}$)	FMC (%)	SDI	PSR (No)	Elevation (m a.s.l.)	Slope (%)
Mean	19.57	76.59	3.52	25.97	685	14
SD	9.18	7.56	0.66	7.52	460	15
Min	3.54	57.00	2.00	11.00	260	0
Max	38.42	84.00	4.56	39.00	1645	58

Table 2
Correlation matrix between FB, FMC, SDI, PSR, elevation, and slope in the field sampling sites ($n = 34$). Highlighted values indicate Student's t-test statistical significance ($p < 0.001$).

	FB ($t\,ha^{-1}$)	FMC (%)	SDI	PSR (No)	Elevation (m a.s.l.)	Slope (%)
FB	1.00	–	–	–	–	–
FMC	**0.84**	1.00	–	–	–	–
SDI	−0.43	**−0.58**	1.00	–	–	–
PSR	**−0.58**	**−0.69**	**0.86**	1.00	–	–
Elevation	**−0.74**	**−0.83**	**0.60**	**0.65**	1.00	–
Slope	**−0.63**	**−0.78**	**0.68**	**0.78**	**0.69**	1.00

3.2. Hyperspectral vegetation indices

The coefficient of determination (R^2) patterns obtained for FB and FMC were similar (Fig. 3a and c), as expected from high inter-correlation between variables. Two regions with the highest R^2 values were evidenced. The first region is located where NIR bands between 750 and 900 nm coincided with bands in the red-edge between 720 and 740 nm. The second region corresponds to band ratios between bands in the NIR shoulder (850–930 nm) and in the absorption feature located between 940 and 1000 nm. Diversity indicators showed a different pattern (Fig. 3b and d). Highest R^2 was obtained with ratios between NIR and red bands, in ratios between red and blue (400–500 nm) bands, and in ratios calculated with two bands in the blue region of the spectrum.

For each variable the most efficient index was selected and cross-validated statistics were computed (Table 3). The best indices for FB and FMC were located in the NIR region between 930 and 970 nm. Absolute best performances were obtained for FMC. Exponential models had slightly better fitting performance only for FB, indicating weak index saturation. Most efficient narrow-band

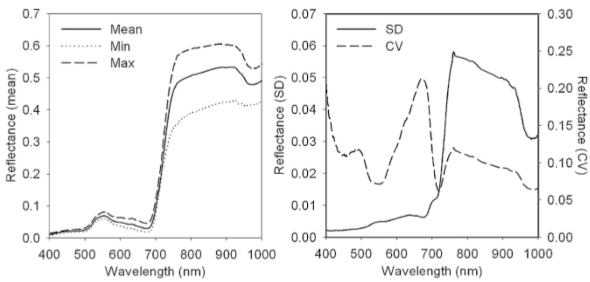

Fig. 1. Summary statistics of reflectance in the range 400–1000 nm measured in the field ($n = 34$). Reported statistics are: mean, maximum and minimum reflectance values (left), SD and CV (right).

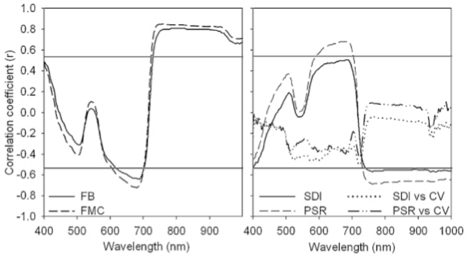

Fig. 2. Correlogram of transect mean reflectance versus FB, and FMC (left). And correlogram of transect mean reflectance and CV versus SDI, and PSR (right). Horizontal lines indicate upper and lower correlation coefficient thresholds of Student's *t*-test statistical significance ($p < 0.001$).

indices for SDI and PSR estimation were located in the blue region between 450 and 500 nm. PSR was predicted with better accuracy than SDI.

3.3. PLS regression

The PLSR models obtained for all the variables investigated using (i) the entire reflectance spectra and (ii) a subset of bands selected by high weighted regression coefficient or significant Marten's uncertainty test are reported in Table 4.

Table 3
Best OLSR and exponential models and related accuracy parameters for estimating FB, FMC, SDI, and PSR using band ratios of reflectance.

	Index *Ri*, *Rj*	OLSR model			Exponential fit
		R^2	R^2_{CV}	$RMSE_{CV}$	R^2
FB ($t\,ha^{-1}$)	933,947	0.80	0.77	4.32	0.86
FMC (%)	933,969	0.82	0.78	3.33	0.81
SDI	473,494	0.65	0.61	0.41	0.63
PSR (No)	454,471	0.69	0.66	4.31	0.67

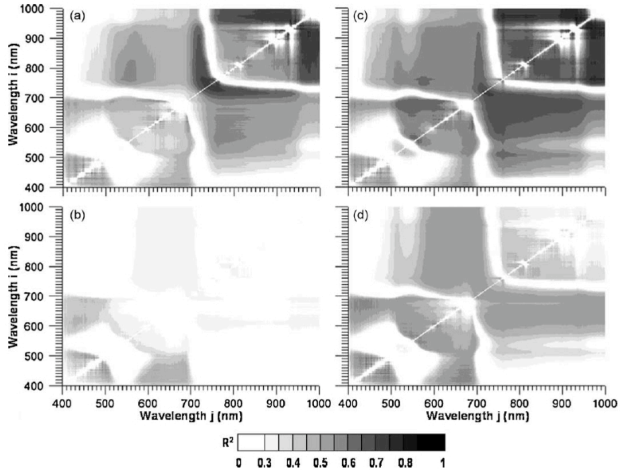

Fig. 3. Two-dimensional contour plots representing the coefficients of determination (R^2) of the OLSR models between band ratios of reflectance calculated using all the possible two band *i* and *j* in the range 400–1000 nm and FB (a), FMC (c), SDI (b), and PSR (d). Areas with statistically significant relationships ($p < 0.001$) are plotted in grey scale.

Table 4
Best PLSR models and related accuracy parameters for estimating FB, FMC, SDI, and PSR from reflectance data. The number of PC and the input bands used in the simplified models are also indicated.

	All bands PLSR model				Simplified PLSR model				
	PC	R^2	R^2_{CV}	RMSEcv	Bands	PC	R^2	R^2_{CV}	$RMSE_{CV}$
FB ($t\,ha^{-1}$)	4	0.85	0.80	4.21	672–715–896–913–951–974	4	0.89	0.84	3.69
FMC (%)	3	0.78	0.73	3.98	541–706–934–969–1000	4	0.86	0.81	3.33
SDI	1	0.31	0.27	0.57	464–509–548–599–644	4	0.73	0.64	0.40
PSR (No)	2	0.54	0.46	5.62	429–438–455–482–496	4	0.79	0.72	4.02

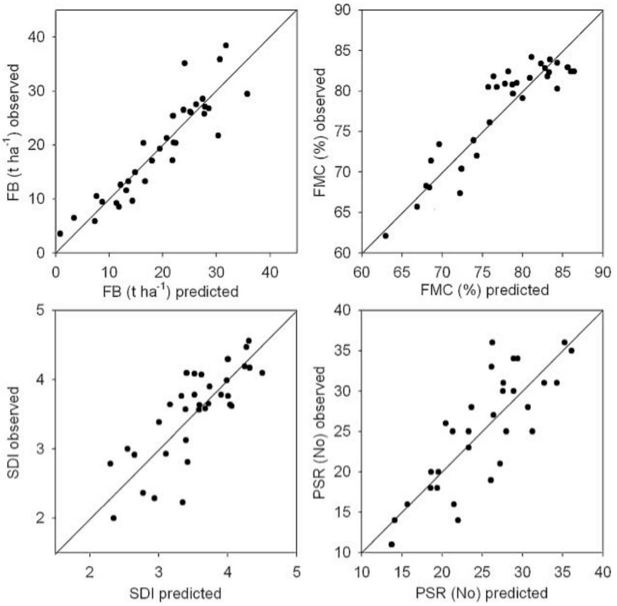

Fig. 4. Scatterograms of observed versus predicted FB, FMC, SDI and PSR values using best PLSR models. The input bands used in each model are those reported in Table 4.

For FB estimation PLSR models performed always better than band ratios of reflectance and OLSR models. The optimal number of selected principal components (PCs) was 4. The reduction of input data dimensionality further increased model accuracy. Best PLSR models increased R^2_{CV} values by 0.07. Results obtained for FMC using the full spectrum as input for PLSR were slightly worse than those obtained with band ratios, while a better accuracy was reached selecting a variable subset, with a R^2_{CV} increase of 0.03. Similar results were obtained for plant diversity indicators: full spectrum PLSR models had worse performances than OLSR models based on band ratios, while better results were obtained with PLSR after variables subset selection, with an increase in R^2_{CV} compared to OLSR models of 0.03 for SDI and 0.06 for PSR. The predictive performances of the best PLSR models are illustrated in Fig. 4.

4. Discussion

More productive meadows were located in the bottom valley, while diversity indicators had an opposite behaviour, indicating the important ecological role of mountain hay meadows, characterized by a less intensive agricultural use (Peter et al., 2008). The higher productivity of bottom valley meadows can instead be related to several factors, including nitrogen fertilization and the higher fertility of alluvial soils, which promoted and selected more demanding and productive species (e.g. *Lolium perenne*, *Dactylis glomerata*, *Trifolium* sp. pl., *Festuca pratensis*). Inter-correlation between FB and FMC was high, as usually observed, and this hampers their independent prediction via spectral data. The same consideration can be extended to SDI and PSR. On the contrary, plant diversity indicators and productive parameters showed generally weaker inter-correlations, suggesting that the application of different techniques for their independent assessment may be useful.

Results of the correlation analysis between reflectance and FB, FMC were consistent with other studies conducted on hay meadows in the Alps (Gianelle and Guastella, 2007; Gianelle and Vescovo, 2007), confirming the well established correlation with NIR and red reflectance (i.e. increasing green biomass corresponds to increasing pigment absorption in red and to increasing reflection in NIR due to multiple scattering effects) (e.g. Hoffer, 1978). Concerning band ratio of reflectance, several studies have already reported the importance of the red-edge region for accurate assessment of biomass in mixed grassland ecosystems using hyperspectral band ratio indices (Darvishzadeh et al., 2008; Fava et al., 2009), likely due to the more linear response of this region compared to traditionally used red bands, which saturate for high leaf area index (>3) and biomass values (Boschetti et al., 2007). The absorption feature between 940 and 1000 nm has been also previously reported as important for crop biophysical variable assessment (Thenkabail et al., 2000), being strongly influenced by canopy dry matter and moisture content (Bowyer and Danson, 2004). The absolute best OLSR models for FB and FMC were all based on indices located in this region, with minor saturation effects even for the highest biomass values. The use of PLSR models allowed increasing the accuracy of all variable prediction, and relative errors using this approach resulted always below 20%, demonstrating the reliability of this methodology for spatially distributed assessment of hay meadow production.

Both OLSR and PLSR models allow predicting SDI and PSR with better accuracy than previous works (Carter et al., 2005; Lucas and Carter, 2008), likely due to the higher precision and spatial resolution of field spectrometric data compared to airborne data. However, absolute best results were obtained with PLSR models after the selection of an optimal subset of bands, with a relative error of 11% and 15% for SDI and PSR, respectively. These results further confirm the validity of PLSR for hyperspectral data analysis (Schmidtlein and Sassin, 2004; Cho et al., 2007; Darvishzadeh et al., 2008), thanks to the possibility to use a greater amount of the information carried out by hyperspectral data compared to vegetation indices. Nevertheless, the improvements obtained using PLSR models were moderate (maximum R^2 increase of 0.07), despite the higher number of predictor variables used. Indeed, while PLSR may provide slightly higher modelling accuracy, OLSR models based on two band vegetation indices are more parsimonious and likely more replicable.

The biophysical relationship between reflectance and diversity indicators is not straightforward: previous authors have explained this correlation in grasslands by the degree of soil exposure and the linkage between plant species richness and animal disturbance (Carter et al., 2005; Lucas and Carter, 2008). Hay meadows at peak production have full ground cover and are not affected by animal disturbance, and the best performing band ratio indices and PLSR models were all based on bands located in the red and blue visible regions, dominated by the influence of chlorophyll a + b absorption peaks (Taiz and Zeiger, 2002), which in turn are related to canopy nitrogen content (Everitt et al., 1985). These observations lead to the hypothesis that the biophysical base of the observed relationship between hay meadow canopy reflectance and plant diversity could be driven by variations in canopy chlorophyll and nitrogen content. Despite the lack of quantitative data in this study to confirm this hypothesis, further support was derived by plant species composition analysis in our field sites, where we observed a negative correlation between plant diversity indicators and the presence of species adapted to soil with elevated nitrogen content (e.g. *Capsella bursa-pastoris*, *Veronica arvensis*, *Trifolium repens*), as already reported by Marini et al. (2008) in the same ecosystem.

The results reported in this study prove the potential of field spectrometric data for assessing hay meadow biomass and plant diversity indicators at canopy level. This is seen as a first step for further studies at landscape scale by means of satellite and aerial observations, aimed at monitoring the spatial and temporal variability of hay meadow biophysical and ecological properties across the European Alps.

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