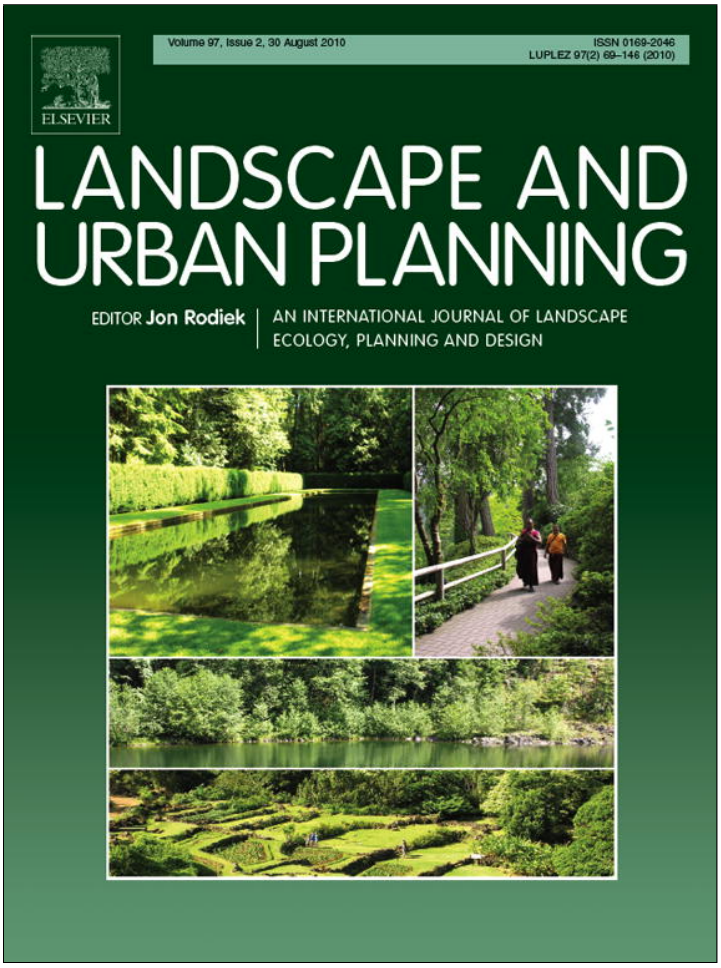
Volume 97, Issue 2, 30 August 2010
ISSN 0169-2046
LUPLEZ 97(2) 69–146 (2010)
ELSEVIER
LANDSCAPE AND URBAN PLANNING
EDITOR Jon Rodiek
AN INTERNATIONAL JOURNAL OF LANDSCAPE ECOLOGY, PLANNING AND DESIGN

Landscape and Urban Planning 97 (2010) 92–97

Contents lists available at ScienceDirect

Landscape and Urban Planning

journal homepage: www.elsevier.com/locate/landurbplan

Evaluating a vegetation-recovery plan in Mediterranean alpine ski slopes: A chronosequence-based study in Sierra Nevada (SE Spain)

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ARTICLE INFO

Article history:
Received 17 July 2009
Received in revised form 27 April 2010
Accepted 30 April 2010

Keywords:
Ski runs
Mediterranean
Plant diversity
Recovery
Evaluation
Discriminant analysis

ABSTRACT

In this paper, we assess the results found in the restoration of vegetation on ski runs in the Mediterranean high mountain, contrasting different issues widely used for evaluating recovery plans, such as cover, richness, diversity, growth, and qualitative species composition, with the aim of establishing their relative validity as well as finding a straightforward model to assess the success of the restoration of degraded areas. Ski runs were selected in Sierra Nevada ski station (SE Spain) in which hydroseeding was performed from 2002 to 2005. The sampling design was based on a chronosequence approach, using natural areas established as 'models' (i.e. target for long-term restoration) to evaluate the restoration success based on the similarity to the model. Although parameters such as growth, cover, and even richness or diversity reached similar values to the ones in the model areas after 4 years (i.e. natural perennial mountain pastures), other indicators such as composition, measured in a qualitative way as the ratio of colonizing species to total species, showed different occurrence values for the most abundant species. Moreover, when the whole pool of species was taken into account using discriminant analysis, the results differed, showing that although the process performed well, the recovery (sensu stricto) requires longer periods than the duration assessed to be fully successful. The results showed that common parameters, such as growth, cover, richness, or diversity, when used solely may lead to misinterpretation, and therefore additional methods to compare composition, such as the discriminant analysis, are strongly recommended.

1. Introduction

Since the mid-twentieth century, Mediterranean mountains have undergone a vigorous increase in tourist activities, especially those related to alpine skiing (Lasanta et al., 2007). Ski resorts yield large economic returns (Elsasser and Messerli, 2001) and provide services and infrastructure improvements for mountain areas (Snowdon et al., 2000) but on the other hand inflict heavy impact on the environment and landscape (e.g. Benthem, 1973; Pignatti, 1993; Needham and Rollins, 2005). Specifically, the construction of ski runs is one of the major causes of pastureland loss in the Alps (Watson, 1985; Urbanska, 1997). To build the ski runs, all the vegetation is eliminated, the upper soil layers are disturbed, stones are removed and the topography is modified, resulting in the complete elimination of the vegetation and the general disturbance of the ecosystem (Gros et al., 2004; Barni et al., 2007). The processes of primary succession of the vegetation are extremely slowed down (Körner, 2003) due to the adverse high-mountain conditions. Moreover, maintenance activity specific to ski resorts, such as the machine-grading of ski slopes and artificial snowing, can strongly limit the recovery of the vegetation. The machine-grading (i.e. the complete removal of topsoil and vegetation, as well as the leveled of the surface) causes severe and lasting impact on alpine vegetation, which proves difficult to mitigate even by recovery measures, particularly at higher altitudes (Wipf et al., 2005). The impact of artificial snowing depends on the current state of the vegetation, the environmental objectives of a specific ski resort, and the properties of the snow, being the sum of these usually negative, but not always (Rixen et al., 2008; Wipf et al., 2005). As a result of the management, extensive areas of soil remain bare in these environments (Tsuyuzaki, 1995; Urbanska and Fattorini, 2000). Thus, restoration is vital to prevent erosive processes as well as to restore ecosystem structure and functionality (Muller et al., 1998). Restoration should minimize resource depletion and ensure long-term ecosystem recovery, which in terms of ecological restoration is called rehabilitation (e.g. Bradshaw, 1997).

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doi:10.1016/j.landurbplan.2010.04.015

Table 1
Details of the hydroseeding seed mixture.

Species	Seed mass	% purity	g	g (m^2)	Seeds (m^2)	%
Agrostis nevadensis	0.320	0.200	30.0	0.42	8	1.6
Arenaria tetraquetra subsp. amabilis	0.193	0.610	90.0	0.02	5	1
Dactylis glomerata subsp. *juncinella*	0.280	0.470	47.5	1.11	88	17.8
Deschampsia flexuosa	0.120	0.180	25.0	3.18	119	24.1
Festuca indigesta	0.750	0.910	63.0	1.69	129	26.1
Helianthemum apenninum subsp. *apenninum*	0.670	0.950	75.0	0.25	26	5.3
Hormathophylla spinosa	0.762	0.490	86.0	0.2	11	2.2
Reseda complicata	0.041	0.398	80.0	0.12	93	18.9
Sideritis glacialis	1.218	0.420	57.0	0.12	2	0.4
Thymus serpylloides	0.700	0.210	75.0	0.47	11	2.2

Seed mass, weight per 100 seeds; % purity, percentage of purity; *g*, % of seed germination; *g* (m^2), weight of seeds applied per square meter; seeds (m^2), number of viable seed applied per square meter; %, percentage of seed belonging to one species in the mixture (data provided by technical environmental staff of CETURSA Sierra Nevada S.A.).

Some studies have pointed out the changes in plant-species composition, dynamics, biomass, etc. caused by the building of ski runs and later restoration using the different techniques (e.g. Watson, 1985; Urbanska, 1995, 1997; Tsuyuzaki, 1990, 1995, 2002; Muller et al., 1998; Urbanska and Fattorini, 2000), but all of them have focused on alpine ecosystems sensu stricto. Restoration of Mediterranean high-mountain ecosystems have not yet been studied despite that there are major differences with respect to Alpine mountains (Grabherr et al., 2003) due to climatic irregularity and the existence of two stress periods for plants: winter, owing to low temperatures; and summer, due to the typical Mediterranean summer drought (Giménez-Benavides et al., 2007). The present study was conducted in Sierra Nevada (SE Spain), a good example of a Mediterranean mountain with a ski station (the southernmost in Europe).

This study assesses the results of plant restoration works on ski runs under Mediterranean conditions, comparing results from a chronosequence and evaluating different issues frequently used as indicators, such as cover, richness, diversity, growth, and species composition. The aim was to establish a methodology to assess the success of the vegetation recovery in ski runs and other disturbed environments.

2. Materials and methods

2.1. Study area

The study area is located at the ski station of Sierra Nevada (SE Spain; 37°N to 3°W), in the Sierra Nevada mountain. This mountain area covers about 2100 km^2, included in the Baetic range. Sierra Nevada is a major Mediterranean diversity hotspot (Médail and Quézel, 1997, 1999); in fact, above 2000 m asl occur nearly 100 endemic and rare taxa (Lorite et al., 2007a) as well as rare and/or endemic plant communities (Lorite et al., 2007b). Geologically, the study site is constituted of siliceous rocks, mainly micaschists (Jabaloy et al., 2008). The soils belong to the suborders Orthents and Ochrepts, the soil units being Anthropic Regosols and Humic Regosols, with sandy loam or loamy sand containing a high proportion of gravel texture (Delgado et al., 2007). In general, the climate is typically Mediterranean although with high-mountain features. The average annual rainfall of the area is 925 mm with mainly snow-based precipitations and an extended period of summer drought, while the mean annual temperature is 3 °C (coldest month: −4.3 °C, hottest month: 14 °C; lowest: −14.6 °C, highest: 26.6 °C) (for further information see Gómez, 2002). The ski station, built in 1964, is provided with 115 ski runs and 102.88 skiable kilometers, ranging between 2100 and 3300 m asl. In years with high rainfall, the average snow depth exceeds 2 m at most ski slopes during the ski season, although in some areas more than 5 m can be reached, in contrast to dry years, when the snow is almost absent. Therefore, the ski resort produces artificial snow when necessary. The first snow machines were installed in 1989, and at present day produce artificial snow for 32 km of snow runs (source: Cetursa Sierra Nevada S.A.; www.cetursa.es).

2.2. Ski-run restoration

The ski-run construction at the site dates back to the early 1960s, for which the topsoil (10–40 cm of depth, according Delgado et al., 2007) was removed and discarded, exposing the underlying mineral soil and bedrock. The first restoration works date from 2000 (Gálvez, pers. com.), although not until 2002 did the restoration program begin in strict terms. Since then the firm that manages the ski runs has undertaken sowing by hydroseeding (see further details on the technique in Merlin et al., 1999), in all cases during the autumn (throughout October) using the same mixture of autochthonous seeds every sowing period (see Table 1), harvested in the surroundings of the ski runs (see Table 1). The assessed period in this study corresponds to the hydroseedings performed over 4 years: 2002 (ca. 1.5 ha), 2003 (ca. 2.8 ha), 2004 (ca. 3.1 ha), and 2005 (ca. 1.9 ha).

2.3. Sampling design

Field samples were carried out during the summer of 2006. A chronosequence was established for this purpose (Kent and Coker, 1992; Foster and Tilman, 2000), including the areas seeded from 2002 to 2005 (H02, H03, H04, H05, hereafter) (Fig. 1). The plant community we sought to restore was a native Mediterranean-alpine perennial pasture of *Festuca indigesta.* It is also composed of others species such as *Deschampsia flexuosa*, *Koeleria crassipes*,

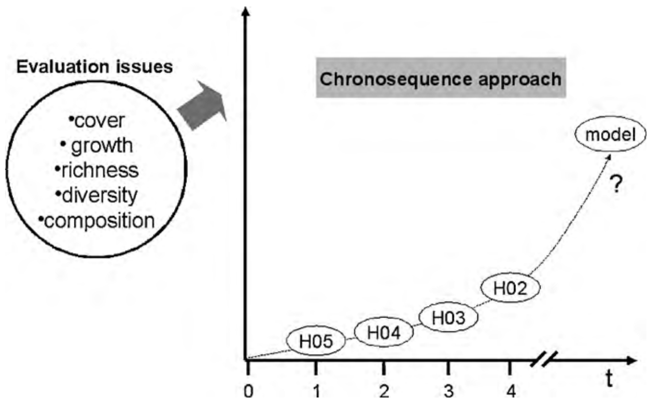

Fig. 1. Conceptual scheme established for the monitoring of the chronosequence. On the *x*-axis, time (in years) from the recovery action (note that 0 corresponds to the evaluating time in 2006). On the *y*-axis, evaluation issues settled to evaluate the recovery success (see Section 2 for further information).

Table 2
Results of growth estimation as in *Dactylis glomerata* (A), *Festuca indigesta* (B), and *Reseda complicata* (C).

Species		Chronosequence				Model
		H05	H04	H03	H02	Mo
(A)*Dactylis glomerata*	H	2.34 ± 0.14 c	3.38 ± 0.18 c	2.82 ± 0.16 c	5.40 ± 0.32 b	10.07 ± 0.69 a
subsp. *juncinella*	D	3.41 ± 0.25 d	8.15 ± 0.41 c	8.72 ± 0.51 c	15.79 ± 0.93 b	23.58 ± 1.48 a
	B	39.20 ± 11.70 b	266.97 ± 37.50 b	248.42 ± 37.60 b	1729.17 ± 379.40 b	7606.90 ± 1390.80 a
(B)*Festuca indigesta*	H	2.11 ± 0.12 d	3.65 ± 0.19 c	4.03 ± 0.17 c	5.44 ± 0.25 b	9.94 ± 0.48 a
	D	3.13 ± 0.19 d	8.53 ± 0.44 c	9.84 ± 0.51 c	12.57 ± 0.42 b	20.27 ± 1.19 a
	B	25.33 ± 4.12 b	328.81 ± 64.73 b	461.53 ± 67.85 b	928.49 ± 90.70 b	5167.18 ± 919.20 a
(C)*Reseda complicata*	H	2.18 ± 0.12 d	13.33 ± 0.99 c	17.96 ± 1.57 c	36.43 ± 2.06 b	56.62 ± 2.70 a
	D	1.74 ± 0.13 d	22.45 ± 1.76 c	23.99 ± 1.82 c	43.84 ± 1.99 b	69.06 ± 2.80 a
	B	9.00 ± 1.86 c	3214.66 ± 614.00 c	5196.50 ± 1593.70 c	37067.14 ± 5922.57 b	132726.69 ± 16192.70 a

H, height (cm); D, diameter (cm); B, biovolume (cm^3). Different letters in the same line indicate significant differences in *post hoc* Tukey–Cramer test at $p < 0.05$. $n = 30$ in all cases.

S. glacialis, *Avenula laevis*, *Thymus serpylloides*, *Arenaria tetraquetra* subsp. *amabilis*, and *Jasione amethystina*. This community lives in a mosaic with juniper-genista patches being dominant in higher-slope areas under the same ecological conditions of hydroseeded areas (see Lorite, 2002 for further information on this community type). This community is widely distributed in the study area and fulfills the objectives and structure (i.e. plant architecture of the community) required for ski-run maintenance and skiing. Thus, this natural plant community was established as a model (Mo, hereafter) to evaluate the restoration success, comparing this model to the chronosequence of the restored areas using hydroseeding. With this aim, a number of evaluation issues were used, such as cover, growth of key plant species, richness, diversity, and composition (see Fig. 1). A stratified sampling was made to guarantee the representation. We defined the following non-overlapping strata: all the chronosequence stages (H02–H05) and the model areas (Mo), three slope ranges: I: 0–30%, II: 30–60%, III: >60%, and two exposures: I: 0–180° and II: 180–360°. The samples were randomly placed for each stratum.

We performed 147 linear transects of 25 m with three contact points per meter, 75 contacts per transect in total, for which the number of occurrences of perennial species was recorded (chamaephytes, hemicryptophytes, and geophytes). Afterwards every species was classified by its behavior in colonizing and non-colonizing species, following Molero and Pérez Raya (1987) and Lorite et al. (2007c). To estimate the size of three key species (*F. indigesta*, *Dactylis glomerata* subsp. *juncinella*, and *Reseda complicata* (see Table 1)), we randomly collected 150 individuals (30 individuals × 5 chronosequence stages). For each individual, we measured height and diameter, after calculating the volume of the semi-spheroid form (volume of semi-spheroid = $((4/3)\pi r^2 h)/2$); where r is radius and h is height), given the cushion-like or low tussock habit of the species. The nomenclature of the species followed Blanca et al. (2009).

2.4. Data analysis

The raw data matrix was used to compute the cover, size of key species, richness, diversity (Shannon–Wiener index; see Magurran, 1988) and frequency of plant colonizing species (calculated as the ratio colonizing species to total number of species encountered per transect). The data for the different variables were compared throughout the chronosequence using a one-way ANOVA (normality was checked by the Shapiro–Wilk test, and homoscedasticity by the Bartlett test). To test differences in frequency of colonizing species between the model areas (perennial Mediterranean-alpine pastures of *F. indigesta*) and the hydroseeded areas, the Wilcoxon non-parametric test was used.

In addition, to compare the floristic composition of the different samples, a discriminant analysis was used (see Hair et al., 1998). This is a powerful technique when the variable to discriminate is nominal with multiple levels (chronosequence in our case: H02, H03, H04, H05, and Mo) and the independent variables (species) are metric (number of individuals). All the statistical analyses were performed using JMP 6.0 (SAS Institute). Throughout the text, means are followed by ±SE.

3. Results

3.1. Key species growth, cover, richness, and diversity

Data concerning the size of key species in terms of biovolume (Table 2) showed that, for the three key species studied, there were significant differences between the 4 years of hydroseeding and the model areas, in height (H), diameter (D), and biovolume (B); only in the older seedings did the data approximate the model areas, although there were significant differences in all cases (see Table 2). In *Reseda complicata*, size was faster than in *Dactylis glomerata* subsp. *juncinella* or *F. indigesta*, and therefore we could expect individuals, in less time, to reach a size similar to that found in the model areas.

With regard to the total cover (Table 3), chronosequence H02 reached similar values to those of the areas Mo (39.45 ± 1.89 vs. 41.71 ± 3.07), even for the case H04 (34.00 ± 2.66), which led us to deduce that the cover was a parameter with easy recovery.

For richness, H03 and H02 showed no significant differences compared with Mo. Diversity showed even fewer significant differences than did richness, and only H05 had significantly lower diversity (1.17 ± 0.14) compared to the rest of the chronosequence (Table 3).

Table 3
Results of the one-way ANOVA comparing: cover, richness, diversity and ratio of colonizing species to total species encountered in the chronosequence.

Chronosequence	*n*	Cover	Richness	Diversity	Ratio col.:tot.
H05	13	18.87 ± 2.27 b	5.14 ± 0.61 c	1.17 ± 0.14 b	0.641304 c
H04	26	34.00 ± 2.66 ab	7.23 ± 0.35 bc	1.54 ± 0.07 ab	0.292415 b
H03	14	24.57 ± 2.73 b	7.89 ± 0.57 abc	1.70 ± 0.12 a	0.286822 b
H02	41	39.45 ± 1.89 a	8.17 ± 0.33 ab	1.64 ± 0.054 a	0.325763 b
Mo	53	41.71 ± 3.07 a	8.80 ± 0.38 a	1.71 ± 0.046 a	0.120024 a

Different letters in the same line indicate significant differences in *post hoc* Tukey–Cramer test at $p < 0.05$.

Table 4
Data on life form, distribution, and abundance of the most abundant species sampled.

Species[a]	col.[b]	biot.[c]	distrib.[d]	n cont.[e]	Mo[f] (n = 53)	H[g] (n = 94)	p-Value[h]
Dactylis glomerata subsp. *juncinella* (Bory) Stebbins & Zohary	0	Hc	N s.l.	779	1.58 ± 0.54	6.83 ± 0.62	<.0001
Festuca indigesta Boiss.	0	Hc	N-A	608	4.64 ± 1.13	3.74 ± 0.39	0.0057
Spergularia rubra (L.) J. Presl & C. Presl	1	Th/Hc	W	295	0.90 ± 0.23	2.45 ± 0.51	0.2597
Reseda complicata Bory	0	Ch	N s.l.	202	0.77 ± 0.29	1.60 ± 0.24	0.0068
Thymus serpylloides Bory	0	Ch	N s.l.	183	2.85 ± 0.71	0.45 ± 0.10	0.0042
Agrostis nevadensis Boiss.	0	Hc	N-A	142	1.44 ± 0.47	0.71 ± 0.10	0.2608
Jasione amethystina Lag. & Rodr.	0	Ch	N s.l.	124	1.98 ± 0.36	0.28 ± 0.11	<.0001
Artemisia absinthium L.	1	Ch	W	121	0.00 ± 0.0	1.17 ± 0.27	0.0002
Polygonum aviculare L.	1	Th/Hc	W	106	0.083 ± 0.05	0.99 ± 0.22	0.0008
Carex nigra (L.) Reichard	0	Hc	W	88	1.82 ± 0.93	0.00 ± 0.00	0.0031
Arenaria tetraquetra subsp. *amabilis* (Bory) H. Lindb. fil.	0	Ch	N s.l.	83	1.52 ± 0.32	0.10 ± 0.04	<.0001
Pilosella pseudopilosella (Ten.) Soják	1	Hc	W	78	0.40 ± 0.16	0.57 ± 0.14	0.5031
Carduus carlinoides subsp. *hispanicus* (Kazmi) Franco	1	Hc	N s.l.	76	0.33 ± 0.12	0.58 ± 0.16	0.3470
Herniaria boissieri Gay	1	Ch	N s.l.	64	0.44 ± 0.13	0.42 ± 0.10	0.4517
Genista versicolor Boiss.	0	Ch	N s.l.	61	0.83 ± 0.42	0.20 ± 0.06	0.3895
Paronychia polygonifolia (Vill.) DC.	1	Hc	WM	48	0.42 ± 0.23	0.27 ± 0.06	0.2783
Linaria aeruginea subsp. *nevadensis* (Boiss.) Malag.	0	Hc	N s.l.	46	0.19 ± 0.09	0.36 ± 0.07	0.0385
Festuca pseudoeskia Boiss.	0	Hc	N s.l.	45	0.87 ± 0.3	0.03 ± 0.02	<.0001
Rumex angiocarpus Murb.	1	Hc	WM	44	0.10 ± 0.09	0.38 ± 0.1	0.0149
Lotus corniculatus subsp. *glacialis* (Boiss.) Valdés	0	Hc	B	44	0.81 ± 0.26	0.05 ± 0.03	<.0001
Festuca clementei Boiss.	0	Hc	SN	42	0.08 ± 0.07	0.37 ± 0.12	0.1018
Eryngium glaciale Boiss.	1	Hc	N-A	40	0.50 ± 0.18	0.16 ± 0.05	0.0776
Plantago holosteum Scop.	1	Ch	B	32	0.27 ± 0.12	0.18 ± 0.06	0.9281
Sideritis glacialis Boiss.	0	Ch	B	29	0.52 ± 0.22	0.04 ± 0.02	0.0149
Arenaria armerina Bory	0	Ch	WM	29	0.54 ± 0.19	0.03 ± 0.02	0.0002
Dianthus brachyanthus Boiss.	0	Ch	WM	28	0.52 ± 0.23	0.03 ± 0.02	0.0007
Anthyllis vulneraria subsp. *pseudoarundana* H. Lindb.	0	Hc	N s.l.	26	0.42 ± 0.14	0.06 ± 0.02	0.0005
Trisetum glaciale (Bory) Boiss.	0	Hc	SN	25	0.52 ± 0.17	0.00 ± 0.00	<.0001

[a] Species with more than 25 contacts in all transects.
[b] Behavior of the species (1, colonizing; 0, non-colonizing).
[c] Biotype (Ph, Phanerophyte; Ch, Chamaephyte; Hc, Hemicryptophyte; Th, Therophyte).
[d] Distribution of the species (SN, Sierra Nevada; N s.l., Sierra Nevada, Sierra de Baza and Sierra de los Filabres; B, Baetic Mountains (Sierra Nevada and other nearby mountains); N-A, Nevadense and North Africa; WM, Western Mediterranean; W, wide).
[e] Number of contacts per species in 147 transects.
[f] Average number of contacts per transects of the species in model areas.
[g] Average number of contacts per transects of the species in hydroseeded areas.
[h] p-Value from non-parametric Wilcoxon tests (values of $p < 0.05$ in bold).

3.2. Composition-based parameters

The ratio of colonizing species to total species significantly differed among all the hydroseedings performed (H05, H04, H03, and H02) compared with Mo (Table 3). Furthermore, we found that H04, H03, and H02 did not significantly differ, while the ratio decreased the first year (H05) after the hydroseeding and later remained constant (H04, H03, and H02).

When the areas Mo (n = 53) and the hydroseeding areas (n = 94) were analyzed as two groups, with respect to the most abundant species (n total of contacts >25), significant differences were detected (see Table 4). Three groups of species were established, according to their performance: Group 1: significantly more abundant species in the model area, such as *F. indigesta*, *Thymus serpylloides*, *Jasione amethystina*, *Carex nigra*, *Arenaria tetraquetra* subsp. *amabilis*, *F. pseudoeskia*, *Lotus corniculatus* subsp. *glacialis*, *Sideritis glacialis*, *Arenaria armerina* subsp. *armerina*, *Dianthus brachyanthus*, *Anthyllis vulneraria* subsp. *pseudoarundana*, and *Trisetum glaciale*. Group 2: the most abundant species on the ski runs. The cases of *Artemisia absithium* or *Polygonum aviculare* were notable for their abundance on the ski-run area (1.17 ± 0.27 and 0.99 ± 0.22 contacts per transect, respectively) but absent or scarce in the model area. Group 3: species not showing significant differences between the hydroseeding areas and the models. This group included species that frequently appear in pastures but seem to adapt successfully in order to colonize open and altered areas, such as *Eryngium glaciale*, *Pilosella pseudopilosella* or *Spergularia rubra*.

Lastly, results from discriminant analysis, show significant differences between the three groups of samples formed (see Fig. 2): (1) H05, (2) Mo, and (3) H02, H03, and H04, this last group con-

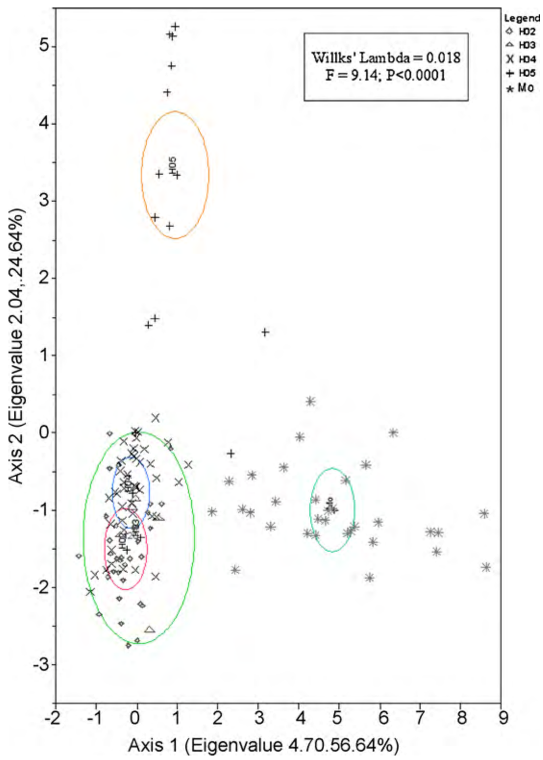

Fig. 2. Biplot showing the results from discriminant analysis, with ellipsoids of probability of 95% superimposed. Mo, H02, H03, H04 and H05 are the stages of the chronosequence (see Section 2 for further information).

sists of three quite overlapping choronosequences (see probability ellipsoids of 95% in Fig. 2). These results show that frequency and abundance of plant species significantly differed between Group 3 (H02, H03 and H04) and both Mo areas as well as for those more recently seeded (H05).

4. Discussion and conclusions

Results from the hydroseeding were quite good, showing the importance of sowing-adapted species, as pointed out by other authors (e.g. Quezel, 1977; Urbanska, 1997; Muller et al., 1998). The cover of the pastures as well as the one of the restored area was relatively low compared with other alpine pastures of temperate areas (i.e. 50–60% in Japan; Tsuyuzaki, 1995, 2002). This illustrates the comparatively harsher conditions of the Mediterranean mountain due to the additional effect of the summer drought (Giménez-Benavides et al., 2007).

According to most authors (e.g. Gómez-Campo and Malato-Beliz, 1985; Sainz Ollero and Hernández Bermejo, 1985; Blanca and Molero, 1990; Heywood, 1996; Blanca et al., 1998), the main objective for the restoration of areas with high levels of diversity, such as the one studied, is to encourage the recovery of the most natural and original-like ecosystem. In most of the studies dealing with ski-run restoration (e.g. Tsuyuzaki, 1990, 1995; Urbanska, 1997; Grismer et al., 2008), the most frequently used parameters to assess success are: cover, richness, diversity, and size, while only a few papers include other methods to evaluate species composition (e.g. Sluis, 2002). In our study, the cover, richness, and diversity results reflect that 3–4 years after the restoration, or, as in the size case in a 5–6 years term, the recovery of the area can be achieved. The results found by other authors in studies comparing ski runs and native communities indicate the opposite, with degraded areas showing low values of cover, richness, and diversity (Wipf et al., 2005), although the areas surveyed, in most cases, have not been subjected to a recovery plan. According to our results, if we examine other parameters that appraise composition, using either a straightforward approach such as the ratio of colonizing species to total species or comparisons between the most abundant species, or else using a more complex method of counting all the species present and using a discriminant analysis, we found a completely different outcome. Although the recovery appeared to be performing well, the process was far from being concluded, with a larger richness and abundance of colonizing species than in natural pasturelands. This finding matches the common idea that colonization and succession processes in the high mountain are very slow (see Körner, 2003 for a revision), especially where the uppermost soil layer has been removed (Urbanska, 1995; Muller et al., 1998; Gros et al., 2004). Consequently, our study site should be considered rehabilitated rather than restored (sensu Bradshaw, 1997).

Therefore, we conclude that the parameters for evaluating the success of this kind of restoration, such as cover, richness, diversity, etc., are useful only for assessing the ability of a recovery plan to slow down erosive processes and improve soil structure and composition (e.g. Grismer et al., 2008). Although these parameters, particularly species richness and diversity, have been suggested as an indicator of a perceived community or system quality (Bowles and Jones, 1999; Woodward et al., 1999), the assemblage of species composition may be equally or more important in judging restoration and management success (Henderson, 1999), taking into account that the main target of recovery plans should be to restore the native-plant-community structure and composition. As some authors have pointed out (e.g. Sluis, 2002), the maintenance of species richness and mechanisms causing species patterns are not sufficient to allow the re-creation of patterns of species found in remnant grassland communities. Our study aims in the same direction, and thus we conclude that the above-mentioned parameters should never be used solely to assess the recovery stage of the vegetation but jointly with others that take into consideration the assemblage of species in comparison with a selected recovery model. In this sense, the discriminant-analysis technique, which, as far as we know has not previously been used in similar studies, provides a useful tool for fairly straightforward and powerful analysis while enabling the user to check whether a group significantly differs from another/others. This is defined as different states of a label-type variable, the difference being based on the composition (species appearing) and frequency of each taxon.

Although some authors have pointed out the originality of the different ecological processes in the Mediterranean mountains (e.g. Grabherr et al., 2003; Giménez-Benavides et al., 2005, 2007), studies remain scarce, and the work presented here represents the first analysis of the restoration of degraded areas in Mediterranean mountains. For this reason, together with the absence of nearby areas with similar features, our results cannot be compared with others, highlighting the need for further studies in other Mediterranean mountains, in order to establish specific patterns and methodologies for these specific environments.

We conclude that under current management-planning frameworks, which include concepts such as Limits of Acceptable Change (LAC), Visitor Impact Management (VIM) or Visitor Experience, and Resource Protection (VERP) (see Needham and Rollins, 2005), it is important to develop indicators or tools to evaluate the success of restoration measures. In this context the methods used here; chronosequence, restoration model, and a multivariate approach, such as discriminant analysis to evaluate the composition, all together may constitute a useful and reliable way to evaluate the relative success of vegetation-recovery plans, not only on ski slopes but also in other degraded areas.

Acknowledgements

This research has been financed by Cetursa Sierra Nevada S.A. The authors appreciate the contribution of Lea Gálvez, from the technical staff of Cetursa, who promoted the research and provided us the information regarding the recovery plan and the management of the ski resorts. We also would like to thank David Nesbitt for linguistic advice. Three anonymous referees improved significantly the style, as well as the content of the manuscript.

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