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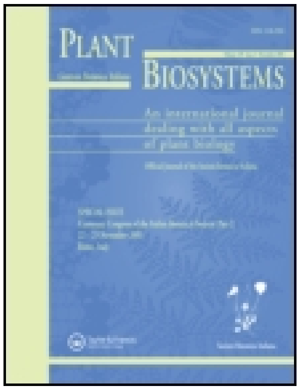

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Recovery of understory vegetation in clear-cut stone pine (*Pinus pinea* L.) plantations

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Abstract

The recovery of understory vegetation after clear-cutting of stone pine (*Pinus pinea* L.) plantations is crucial for biodiversity conservation. The development of plant cover, and the richness, diversity and composition of understory plant communities were investigated within five years from clear-cutting. After three years, plant cover ranged from 48 to 74%, and was formed by over 90% of shrubs and graminoids. Diversity decreased in cut plots and the index of similarity showed marked shifts in species composition. Richness increased and around 50 new species per plot were inventoried over three years. New species showed a high proportion of annuals with broad ecological amplitude and aliens, which changed life form and chorology spectra. Most new species had low frequency, but some were still present after five years. Extinction of resident species was low, and their frequency increased with time from clear-cutting. Most of them were perennials with sprouting ability. Common features of plant communities developing in clear-cut stone pine plantations were: (1) the linear increase of plant cover during the first three growing seasons, (2) increased species richness, (3) decreased diversity, (4) invasion by annuals, and (5) invasion by Alien and Cosmopolitan species.

Keywords: *Clear-cut plantation, diversity,* Pinus pinea*, species composition, stone pine, understory recovery*

Stone pine (*Pinus pinea* L.) is a species with uncertain origin that is widely diffused since Roman times for the consolidation of sand dunes and for the production of edible pine nuts and timber (Busotti 1997). At present, it grows, both spontaneous and cultivated, throughout the Mediterranean region and, especially along coastal areas, it forms even-aged patches of high landscape and historical value. Both seed production and plant stability decrease drastically in around 100-year old individuals, which impairs the utilization of stone pine stands for both economic and recreational purposes. Natural regeneration of this species is often limited by pathogens, predators and, in northern areas, by climate conditions adverse to seedling establishment. Thus, the most common renewal practice of mature stands is clear-cutting, followed by the artificial plantation of saplings and the natural regeneration of the understory (Bianchi et al. 2005).

The understory of stone pine stands is generally sparse and composed of typical Mediterranean trees and shrubs, and by a few forb, grass and sedge species (Tomei et al. 2004). Nevertheless, it preserves most of the floristic biodiversity of these artificial forests, and its conservation is of high concern since many plantations are part of protected nature reserves (Tárrega et al. 2012). It is, therefore, essential to identify simply determinable parameters able to measure the response of understory vegetation to disturbance and its ability to restore pre-disturbance conditions (Boring et al. 1981; Lindenmayer et al. 2000; Newmaster et al. 2007).

Direct effects of clear-cutting on understory vegetation are (1) the damage or destruction of living plants and propagules, (2) the creation of space and seedbeds for regeneration, and (iii) the change of resource availability in ways that affect competitive relationships among plant species (Haeussler et al. 2002). The ecological impact of forest clear-cutting depends on the degree to which microclimate is altered through the removal of canopy shade, the fact that the forest floor is compacted, and that the soil

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organic layer and plant propagules (vegetative, seeds or spores) are removed (Halpern 1988; Roberts 2007). After clear-cutting, a new plant community originates from the bud bank, the seed bank, and from newly dispersed exogenous seed; its final composition and distance from the pre-cut community depend on the relative contribution of the three sources. The stability of plant communities, i.e., their ability to resist or to recover after disturbance, is positively related to structural and compositional diversity (Halpern 1988), and to the ability of resident species to perennate from below-ground structures (Noble & Slatyer 1980; Jobidon et al. 2004). Since structural diversity is very low in pine plantations, their stability relies mainly on the abundance of species with perennating organs within the understory plant community. By contrast, a high contribution of exogenous seeds to understory regeneration generally decreases stability. Arrival of foreign seeds was found to increase with disturbance severity and with the proximity of sources (Haeussler et al. 2002; Newmaster et al. 2007; Dodson et al. 2008).

Structural diversity of forest stands is considered crucial for the conservation of quality habitats for numerous wildlife species, and diversity indices are often used to estimate community response to disturbance (Elliott et al. 1997; Brockway & Lewis 2003; Ciancio & Nocentini 2011). Current indices are based on species richness and evenness, but value all species equally; consequently, they do not evidence shifts in composition, and are not able to estimate the conservation of peculiarities. According to these indices, forests with frequent moderate disturbances are likely to display higher levels of diversity compared to undisturbed ones (Jobidon et al. 2004). Haeussler et al. (2002) found that species composition was more sensitive to silvicultural disturbances than diversity, and Newmaster et al. (2007) reported that clear-cutting negatively affected the distribution of ecologically marginal species, i.e., those growing at the limit or beyond the expected range of environmental conditions, while it increased the presence of non-native species and invasive ruderals. Starting from these findings, the geographic range (chorotype) and ecological distribution of component species could be effectively used to assess the degree of disturbance to plant communities.

At present, structure-based biodiversity indicators, and among these, plant species composition, are considered to be the most effective approach to monitor ecosystem response to disturbance (Chiarucci & Bonini 2005; Roberts 2007; Barbier et al. 2009). The identification and standardization of indicator species or guilds is still in progress, and the major obstacle to their diffusion is that they are mostly location-specific (Lindenmayer et al. 2000).

In the present research, the recovery of understory vegetation after clear-cutting of stone pine plantations was investigated by determining the development of plant cover, and the richness, diversity and composition of understory plant communities, with time from clear-cutting. Data were collected in the first three to five years after disturbance, and were compared with those obtained from adjacent intact pine stands having a similar history.

Material and methods

Study area

The research was conducted in even-aged stone pine (*P. pinea* L.) plantations located along the plain coast of Tuscany, Italy (43°43’ N, 10°16’ E), within a nature reserve surrounded by agricultural and urbanized landscapes. Since 2004, the area, named “Selva Pisana,” belongs to the World Network of Biosphere Reserves, because of its high biodiversity and the demonstrated balanced relationship between Man and nature.

Soil originates from unconsolidated marine sediments and alluvial deposits of the rivers Arno and Serchio, and is composed of ancient dunes running parallel to the coastline alternating with small lagoons or bogs. Climate is a hot humid Mediterranean (Csa, Köppen classification) with a cold winter and an arid summer. Mean temperatures are comprised between 6 and 9.9°C for the coldest month, and exceed 20°C during four months of the year. Annual mean temperatures range from 15 to 17°C, and annual rainfall from 800 to 950 mm, mostly concentrated in autumn and winter (Rapetti & Vittorini 1994). Photoperiod ranges from 8 h 46′ to 15 h 14′, and the growing season of the understory vegetation lasts from March to October, with a summer stop.

Experimental plots

Two 100-year-old stone pine plantations belonging to the *artificial stone pine forest type with understory of Erica sp.* (Tomei et al. 2004) were chosen for investigations. In both plantations, cone harvesting ceased around 20 years ago allowing the development of an understory vegetation consisting of grasses, forbs, shrubs and vines, and dominated by *Erica scoparia* L. The elevation of the two plantations differed slightly, in that the former (P_D) was placed on a relict sand bank (3–4 m AMSL), whereas the latter (P_S) was at sea level. Besides, P_D distanced approximately 100 m from a wide forest clearing used for the exsiccation of pine cones and timber deposit. Cutting operations started in 2006 in P_D, and in 2007 in P_S. Narrow strips of at most 1 ha each were cut, alternated with intact stone pine stands of equal extension. The understory vegetation was completely removed leaving the forest floor bare or covered by debris of cut wood.

Five experimental plots were chosen: one strip cut in 2006 (P_D2006) and an adjacent intact stand (PC_D) in P_D; two strips, respectively cut in 2007 (P_S2007) and 2008 (P_S2008), plus one adjacent intact stand (PC_S) in P_S. In P_D2006 and P_S2008 the entire tree canopy was removed, whereas in P_S2007 ten isolated individuals of stone pine, ash (*Fraxinus angustifolia*) and holm oak (*Quercus ilex*) were left for seed dispersal. Artificial replanting was performed in all cut strips with stone pine and holm oak saplings using a 3 m × 3 m spacing. Planting occurred at different distance from clear-cut: after one year in P_S2008, after two years in P_D2006, and after four years in P_S2007.

Investigations were carried out between 2006 and 2010 and, in June 2006, five soil samples were randomly collected from the 0 to 20 cm layer of both control plots. Soil was air-dried, sieved at 2 mm diameter, and analysed for texture, organic matter content (Walkley and Black method), total nitrogen content (Kjeldahl method) and pH (aqueous solution).

Floristic inventories

Floristic inventories were conducted on the understory layer of both cut and intact plots throughout the entire period of the research. All vascular plants <2 m tall were determined following the Flora d'Italia (Pignatti 1982). Nomenclature was updated with Conti et al. (2005) and Peruzzi (2010). Species were also characterized for Raunkiaer's life form, reproductive strategy, biogeographic distribution (chorotype) and typical habitat, using data from Pignatti (1982, 2005). Finally, all species recorded during the study were compared with the floristic inventories of the area (Garbari 2000; Garbari & Borzatti von Loewenstern 2005).

Based on their presence in the understory of intact stands, species recorded in cut strips were classified as: "New," when found in cut plots and not in the corresponding control plot; "Resident," when found in the cut plot and corresponding control; "Locally extinct," when found in the control plot but not in the corresponding cut plot.

Species frequency

In May and July of all years, 25 squared sample areas of 0.25m^2 were randomly selected in all experimental plots. Within each sample area, the species present were recorded, without any reference to the number of individuals and the relative soil cover. The frequency of species was calculated separately for each experimental plot and year from clear-cutting, by dividing the number of sample areas containing the species by the total number of sample areas recorded in one year (50). Values averaged over the three cut plots were used for the distribution of new and resident species into frequency classes.

Plant cover

The evolution of plant cover was determined by the line-intercept method (Elzinga et al. 2001), which measures the vertical projection of canopy on line-transects. Measurements were performed every year in mid June (corresponding to the peak of the growing season) on five 10-m permanent line transects placed within each experimental plot. By noting along a tape all points where the canopy projection began and ended, a sequence of "covered" and "empty" segments was obtained. All vascular plants <2 m tall were recorded and vegetation cover was considered interrupted when gaps between plants or plant parts exceeded 2 cm. Percent vegetation cover was calculated for each transect as the sum of all "covered" segments divided by the total length of the transect ($\times 100$). The relative cover of individual species was calculated by dividing "coverd" segments into the projections of each species. Where several species overlapped, cover was assigned to the dominant one. Cover values were calculated for all species separately, and then summed to produce group cover estimates for graminoids (grasses + sedges), forbs, shrubs and trees.

Plant community analyzes

In order to compare the three cut strips, plant community analyzes were performed on data obtained in the first three growing seasons after clear-cutting, which corresponded to the year intervals 2006–2008 for experimental plot P_D2006, 2007–2009 for e plot P_S2007, and 2008–2010 for e plot P_L2008. Data sets collected in corresponding year intervals in the understory vegetation of the control plots PC_D and PC_S were used for comparisons.

(1) Index of similarity

Floristic inventories obtained in the five experimental plots were compared by Sørensen's index of similarity (*IS*) (Jobidon et al. 2004), calculated as:

$$IS = \left(\frac{2C}{A+B}\right) \times 100$$

In the formula, C is the number of species in common between the two plots to compare, and A and B are their richness. Species richness was obtained by counting the total number of species recorded within the first three years from clear-cutting in cut plots, and during the corresponding time span in control plots. This index equals 100 when plant lists of A and B coincide, whereas it equals 0 when they have no common species.

(2) Index of diversity
The diversity of plant communities was estimated using the Shannon–Wiener (*H*) and Simpson Reciprocal (*1/D*) diversity indices:

$$H = -\sum_{i=1}^{S} p_i \ln p_i$$

$$\frac{1}{D} = \frac{1}{\sum_{i=1}^{S} p_i}$$

In both formulas, relative cover (p) is taken to indicate the relative abundance of species and it was calculated as the proportion of the cover of a given species (i) to the total plant cover; S is the number of species. For cut strips, data of the third growing season were used, while for control plots those collected in 2008 (PC_D) and 2009 (PC_S).

(3) Life-form and chorology spectra
Non-weighted life-form spectra were calculated for cut and intact experimental plots, as the proportion of species belonging to each type within the flora. Life forms of Pignatti (2005) were clustered into five main types: (1) Phanerophytes, trees, shrubs and vines; (2) Chamaephytes, small bushes and herbs with perennial buds within 25 cm from the soil surface; (3) Hemicryptophytes, perennial herbaceous plants with buds at the soil surface; (4) Geophytes, perennial herbaceous plants with dormant parts below the soil surface, and (5) Therophytes, annual plants.

Non-weighted chorology spectra were also calculated, clustering the chorotypes of Pignatti (2005) into six groups: (1) Mediterranean (Med), including the Steno and Eurymediterranean, the Mediterranean-Montane and the Mediterranean-Turanian chorotypes; (2) European (Eu), including also the Eurasian, Paleotemperate, Eurocaucasian, Middle-European and Subatlantic chorotypes, (3) Boreal (Bor), including species with mainly Circumboreal and Eurosiberian distribution, (4) Tropical (Tr), with species of Paleosubtropical and Subtropical origin, (5) Cosmpolitans (Cos), meaning wide-range and ruderal species, also attributed to the Subcosmopolitan chorotype, and (6) Aliens (Al), which includes species indicated as exotic or adventitious in the Flora d'Italia or not recorded at all for Italy (Pignatti 1982; Conti et al. 2005).

Statistical analyzes

Independent-sample *T*-tests, with a significant threshold set at $P \leq 0.05$, were performed to test differences in soil characteristics between the control plots PC_D and PC_S. The evolution of vegetation cover was analyzed by means of ANOVA, taking the first three years after clear-cutting as the main treatment, and the five experimental plots (three cut strips and two control stands) as secondary treatments. Five permanent transects for each plot were used as sampling units (replicates). Separately, plant cover at the 3rd, 4th, and 5th vegetation season for plot P_D2006 was compared by means of a simple ANOVA. Differences between plots and year from clear-cutting were compared by Duncans's multiple range test ($P \leq 0.05$).

Results

Control plots

The control plots of both plantations, PC_D and PC_S, differed significantly for the chemical properties of the 0–20 cm soil layer, and for the extension, richness and composition of the understory vegetation. Compared to plot PC_D, the superficial soil of PC_S was more acidic (pH 4.6 vs. 5.5), and had higher organic matter and total nitrogen contents (5.6 vs. 3%, and 8 vs. 5.5%, respectively). Soil texture did not differ significantly between plots and was formed by 95% sand, and by approximately 3.5% and 1.5% of silt and clay, respectively.

The percent soil covered by understory vegetation did not change over the years and, when averaged over the period 2006–2010, was higher in PC_D (33% compared to 20%). The lower understory cover of PC_S was essentially due to the average larger size of *Erica* plants that often exceeded 2 m and were not recorded in the understory layer. The richness of the understory vegetation was markedly higher in PC_D than in PC_S. Over the entire period of the research, the number of recorded species was 62 in the former and 38 in the latter. The index of similarity equalled 56%, showing that, besides richness, species composition of the understory vegetation also differed sensibly between control plots. In particular, two species, *Myrtus communis* and *Viola reichenbachiana*, had a frequency close to 20% in PC_S, whereas they were never recorded in PC_D. The two plantations also differed markedly for the amount of annual species that equalled 17 in the latter and only two in the former. Accordingly, the proportion of

Therophytes in life-form spectra was 27% in PC_D and 5% in PC_S. Chorology spectra showed that, in the flora of both pine stands, the Mediterranean chorotype was the most represented, with 53% of species, followed by the European one, with 30% of species. Finally, it is worth mentioning that one Cosmopolitan and one Alien species were recorded in PC_D and none in PC_S.

Plant cover

In all cut plots, total plant cover increased almost linearly between the first and the third vegetative season after clear-cutting, and then increased only slightly (Figure 1). The rate of increase differed significantly among plots independently of the plantation of origin. After three years, the percent soil covered by vegetation ranged from 48% in P_S2007 to 74% in P_S2008. On the contrary, the partitioning of understory cover among growth forms seemed to depend on the plantation of origin: in P_S2007 and P_S2008, shrubs accounted for 52% of total cover and graminoids for 42%, whereas in P_D2006 the former represented 80% of plant cover and the latter only 13% (Figure 2a). The relative contribution of forbs and tree seedlings to plant cover was low and did not differ among cut plots; on average, it equalled 4% for the former and 2% for the latter. In all cut plots the shrub *E. scoparia* was dominant, but while in P_D2006 it accounted alone for 63% of vegetation cover, an equivalent portion of plant cover was partitioned among four species in P_S2008, and five species in P_S2007 (Figure 2b).

Plant community richness and diversity

The number of species recorded in cut strips during the first three years after clear-cutting was always markedly higher than that recorded in corresponding intact stone pine stands, and differences in richness between intact plantations were maintained in the vegetation of clear-cut plots (Table I). Accordingly, we inventoried 98 species in the plot P_D2006, compared to 83 and 78 in P_S2007 and P_S2008, respectively.

In contrast to richness, diversity was higher in both plantations for the understory vegetation of intact plots than for the plant community of cut strips, and differences appeared greater with the Simpson Reciprocal Index than with the Shannon–Wiener one (Table I). Irrespective of index, the lowest diversity was recorded in P_D2006, followed by P_S2008, whereas P_S2007 showed values that also exceeded those recorded in the control plot PC_D.

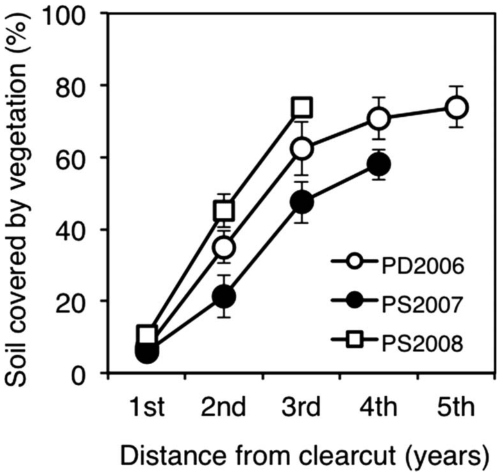

Figure 1. Development of plant cover after clear-cutting of stone pine plantations. Error bars indicate SE. $n = 5$.

Life-form and chorology spectra

In the plant community of all experimental plots, except the control PC_S, the most represented life

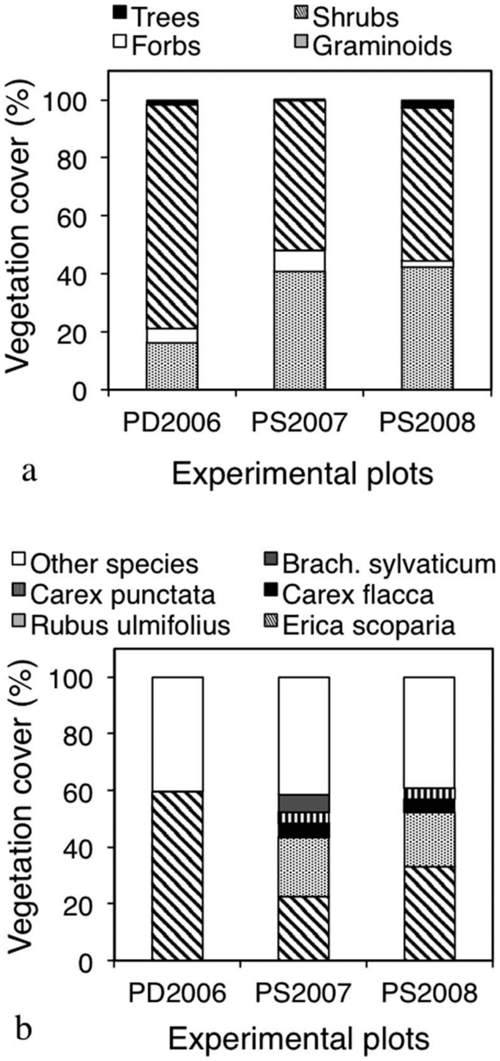

Figure 2. Partitioning of plant cover within growth forms (a) and species (b) at the third growing season after clear-cutting of stone pine plantations.

forms were Phanerophytes, Hemicryptophytes, and Therophytes (Table II). After clear-cutting, the number of Phanerophyte species did not change appreciably, whereas those belonging to the other two life forms increased dramatically. Major changes in the life-form spectra of plant communities evolved after clear-cutting were in the proportion of Phanerophytes and Therophytes (Figures 3a and 3b). The former decreased from 32 to 16% in P_D2006, and from 51 to approximately 23% in P_S2007 and P_S2008, whereas the latter increased by 13 percentage points in P_D2006 and by approximately 27 percentage points in P_S2007 and P_S2008. Changes in the proportion of Chamaephytes, Geophytes and Hemicryptophytes were always lower than five percentage points.

In plant communities of clear-cut plots, we observed an increase in the number of species belonging to all chorotypes (Table II). These increases were not proportional to initial values so that chorology spectra also changed, with similar patterns for the two plantations (Figures 4a and 4b). In plant communities of cut plots the proportion of the most represented chorotypes, Mediterranean and European added, decreased from approximately 90% in control plots to only 70% in cut strips. By contrast, the proportion of Aliens and Cosmopolitans increased markedly, while that of Boreals only slightly. In particular, alien and cosmopolitan species accounted for 22% of the flora in P_D2006, and for 14% in P_S2007 and P_S2008, compared to only for 4 and 0% in PC_D and PC_S.

Table I. Richness and diversity of understory vegetation of intact plots (PC_D and PC_S) and of plant communities evolved on cut strips (P_D2006, P_S2007, and P_S2008). Richness was determined over the first three years from clear-cutting. The Shannon–Wiener (*H*) and Simpson Reciprocal (*1/D*) diversity indices were calculated on the basis of relative plant cover at the third growing season.

	Experimental plots				
	PC_D	P_D2006	PC_S	P_S2007	P_S2008
Richness (number of species)	56	98	37	83	78
Shannon–Wiener (*H*)	2.2	1.7	2.3	2.3	1.9
Simpson Reciprocal (*1/D*)	6.3	2.7	8.3	7.2	4.7

Table II. Number of species belonging to each life form and chorotype in the understory vegetation of intact plots (PC_D and PC_S) and in plant communities evolved in cut strips (P_D2006, P_S2007, and P_S2008). Data refer to the first three years from clear-cutting.

	Experimental plots				
	PC_D	P_D2006	PC_S	P_S2007	P_S2008
Life forms					
Phanerophytes	18	16	19	21	16
Chamaephytes	1	5	1	4	4
Hemicryptophytes	19	32	14	29	26
Geophytes	1	3	1	4	5
Therophytes	17	42	2	25	27
Chorotypes					
Mediterranean	33	46	20	38	33
European	18	23	12	21	23
Boreal	2	7	3	9	8
Tropical	1	1	2	3	3
Cosmopolitan	1	8	0	8	5
Alien	1	13	0	4	6

Similarity of plant communities

Sørensen's index of similarity calculated over the first three years following clear-cutting showed that cut plots and corresponding intact stands differed markedly in species composition (Table III). Values of similarity were higher for P_D2006 and P_S2007 (58%) than for P_S2008 (45%). Differences in species composition were also found by comparing cut strips. As expected, the index of similarity was highest (74%) between the strips of the same

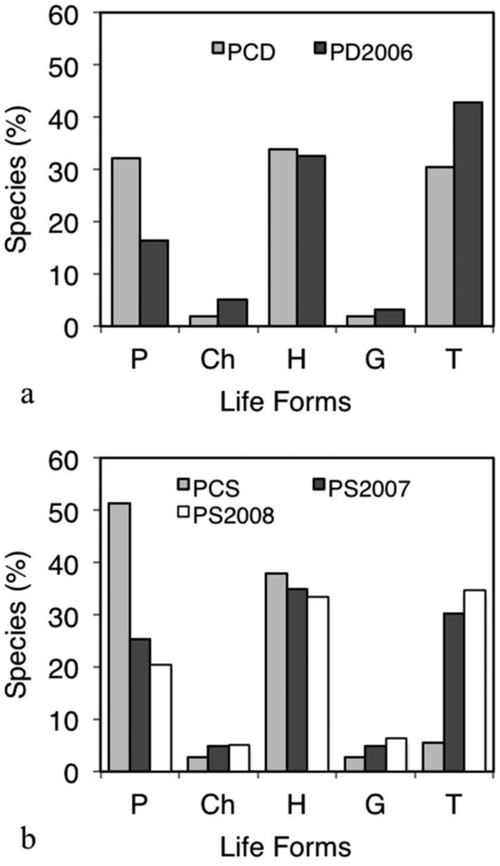

Figure 3. Life-form spectra of the understory vegetation of control plots (PC_D and PC_S) and of plant communities evolved on cut strips (P_D2006, P_S2007, and P_S2008) within three years from clear-cutting. (a) plots of plantation P_D and (b) plots of plantation P_S.

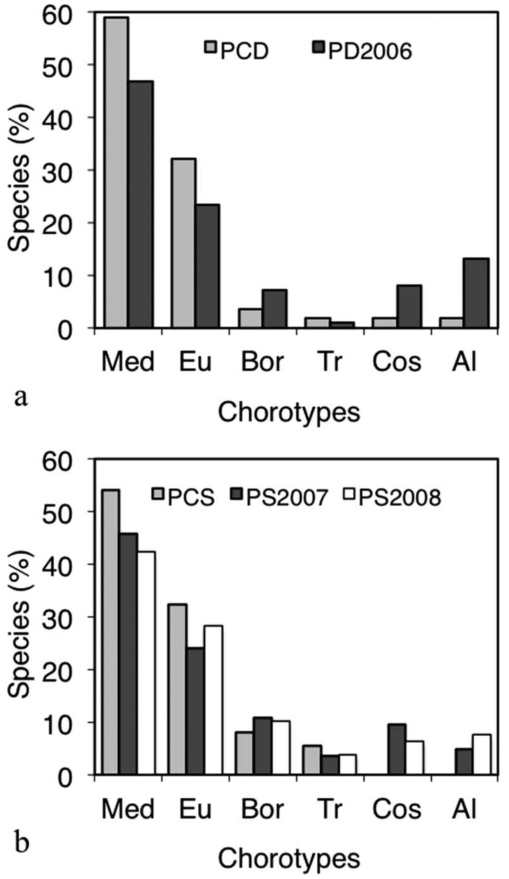

Figure 4. Chorology spectra of the understory vegetation of control plots (PC_D and PC_S) and of plant communities evolved on cut strips (P_D2006, P_S2007, and P_S2008) within three years from clear-cutting. (a) plots of plantation P_D and (b) plots of plantation P_S.

Table III. Sørensen's index of similarity calculated between plant communities of cut plots and corresponding control plots, and number of new, resident and locally extinct species recorded in cut strips within three years from clear-cutting.

	Cut experimental plots		
	P_D2006	P_S2007	P_S2008
Index of similarity (%)	58.4	58.3	45.2
New species	53	48	52
Resident species	45	35	26
Locally extinct species	11	2	11

plantation, i.e., P_S2007 and P_S2008. Lower values were recorded between P_D2006 and P_S2007 (61%), and between P_D2006 and P_S2008 (53%).

New and resident species

During the first three years after clear-cutting, we found approximately 50 species in each cut strip that were not present in the understory of the adjacent intact stand (Table III). By contrast, the number of locally extinct species differed among cut strips, and was 11 in P_D2006 and P_S2008 and only two in P_S2007. Finally, the number of resident species fell from 45 in P_D2006 to only 26 in P_S2008.

Combining the three cut strips, we found 75 species that were not recorded in the control pine stands of both plantations (Table IV). They belonged to 23 families, among which the most represented were Asteraceae with 13 species and Poaceae with 11, followed by Juncaceae, Fabaceae and Solanaceae, with eight, seven and six species, respectively. More than half (52%) were annuals (T), 37% herbaceous perennials (H and G), whereas small shrubs (Ch) represented only 7% and trees (P) 4%. Since 17 and 16% of species belonged to the Alien and Cosmopolitan chorotypes, respectively, about one-third of the new species had a wide geographical distribution or did not even belong to the native flora of Italy. It is worth noting that, among the new species, we inventoried only two trees, *Ficus carica* and *Ailanthus altissima*, a common ruderal and a noxious invader, respectively.

Analysing the typical habitats of the new species, it was found that only 13 are peculiar of scrubs and woodlands, and that 19 occur most commonly in natural grasslands, garigues, and marshes (Table IV). From the remaining 43 species, one belongs to the flora of sand dunes, whereas the others are typical of habitats subjected to anthropogenic disturbance, first of all ruderal sites and abandoned fields, but also human settlements, cultivated fields, gardens, pastures and trampled lawns.

The number of species recorded in cut strips and also resident in at least one of the control stands equalled 60 (Table V). They belonged to 30 families. Once again, Poaceae and Asteraceae were the most represented; Hemicryptophytes and Phanerophytes accounted for 37 and 33% of species, whereas Therophytes only for 25%. The analysis of reproductive strategies indicated that close to 90% of resident perennials showed sprouting ability. The most represented chorotypes among residents were the Mediterranean and European ones, with 53 and 32%, respectively, whereas Cosmopolitans and Aliens were represented by only one species each.

Frequency of new and resident species

The overall number of new and resident species increased up to the second year and then decreased sharply in the third (Table VI). This was essentially due to the high occurrence, in the first and second year, of species that were inventoried only once after clear-cutting (rare).

New species generally had a low frequency and none exceeded 20%. Nevertheless, the number of those with frequency higher than 1% increased from 6 to 15 with time from clear-cutting. Among these

Table IV. Occurrence, life form, chorotype and typical habitat of new species recorded in at least one cut plot (P_D2006, P_S2007, and P_S2008) and not in control plots within three years from clear-cutting.

New species	P_D 2006	P_S 2007	P_S 2008	Life form	Chorotype	Typical habitat
Amaranthaceae						
Amaranthus deflexus L.	X		X	T	Alien	Ruderal sites gardens
Chenopodium album L.	X	X		T	Cosmopolitan	Ruderal sites cultivated fields
Dyspahnia botrys (L.) Mosyakin & Clemants			X	T	Mediterranean	Ruderal sites abandoned fields
Asteraceae						
Bellis perennis L.	X			H	European	Human settlements trampled lawns
Crepis dioscoridis L.	X			T	Alien	Abandoned fields
Crepis sancta (L.) Babc.			X	T	Mediterranean	Ruderal sites pastures
Crepis suffreniana (DC.) Steud.	X				Alien	Ruderal sites abandoned fields
Dittrichia graveolens (L.) Greuter	X	X	X	H	Mediterranean	Ruderal sites abandoned fields
Erigeron canadensis L.	X	X		T	Alien	Ruderal sites abandoned fields
Helichrysum italicum (Roth) G.Don.			X	Ch	Mediterranean	Scrubs Garigues
Hypochaeris radicata L.			X	T	European	Abandoned fields pastures
Lactuca saligna L.	X			H	Mediterranean	Ruderal sites abandoned fields
Senecio sylvaticus L.			X	T	European	Coppices wood clearings
Senecio vulgaris L.	X			T	Mediterranean	human settlements rural sites
Sonchus arvensis L.	X	X		H	Boreal	Abandoned fields gardens
Sonchus asper (L.) Hill	X	X		T	European	Cultivated fileds gardens
Brassicaceae						
Cardamine hirsuta L.		X		T	Cosmopolitan	Human settlements gardens
Caryophyllaceae						
Cerastium ligusticum Viv.	X			T	Mediterranean	Ruderal sites pastures
Silene gallica L.			X	T	Mediterranean	Rural sites pastures
Cyperaceae						
Carex divulsa Stokes	X			H	Mediterranean	Coppices grasslands
Isolepis setacea (L.) R. Br.		X	X	T	Tropical	Marshes
Schoenus nigricans L.		X	X	H	Cosmopolitan	Marshes
Scirpoides holoschoenus (L.) Sojàk	X		X	G	Mediterranean	Marshes
Euphorbiaceae						
Chamaesyce maculata (L.) Small	X		X	T	Alien	Ruderal sites human settlements
Euphorbia paralias L.	X	X		Ch	Mediterranean	Sand dunes
Euphorbia peplus L.	X	X	X	T	Boreal	Ruderal sites gardens
Fabaceae						
Lotus angustissimus L.	X	X	X	T	Mediterranean	Grasslands
Lotus hispidus DC.		X		T	Mediterranean	Abandoned fields
Lotus tenuis Waldst. & Kit. ex Willd.	X			H	European	Grasslands
Trifolium campestre Schreb.	X		X	T	European	Abandoned fields
Trifolium lappaceum L.			X	T	Mediterranean	Wood clearings
Trifolium repens L.	X		X	Ch	European	Abandoned fields trampled lawns
Vicia pseudocracca Bertol.	X			T	Mediterranean	Abandoned fields
Gentianaceae						
Centaurium erythraea Hill.	X	X		H	European	Scrubs grasslands
Hypericaceae						
Hypericum androsaemum L.	X	X	X	P	Mediterranean	Woods
Hypericum humifusum L.		X	X	H	Cosmopolitan	Woods marshes
Juncaceae						
Juncus articulatus L.	X	X	X	G	Boreal	Marshes
Juncus bufonius L.		X	X	T	Cosmopolitan	Marshes
Juncus conglomeratus L-	X	X	X	H	Boreal	Marshes
Juncus effusus L.	X	X	X	H	Cosmopolitan	Marshes
Juncus inflexus L.		X	X	H	European	Marshes
Juncus cfr *squarrosus* L.			X	H	Boreal	Marshes
Juncus tenuis Willd.	X			H	Alien	Trampled lawns
Luzula forsteri (Sm.) DC.	X	X	X	H	Mediterranean	Woods

(*Continued*)

Table IV. (*Continued*).

New species	P_D 2006	P_S 2007	P_S 2008	Life form	Chorotype	Typical habitat
Lamiaceae						
Sideritis romana L.	X			T	Mediterranean	Garigues grasslands
Teucrium chamaedrys L.	X			Ch	Mediterranean	Scrubs
Moraceae						
Ficus carica L.	X			P	Mediterranean	Ruderal sites
Phytolaccaceae						
Phytolacca americana L.	X	X		G	Alien	Ruderal sites abandoned fields
Plantaginaceae						
Veronica arvensis L.		X	X	T	European	Ruderal sites cultivated fields
Poaceae						
Achnatherum bromoides (L.) P. Beauv.	X	X	X	H	Mediterranean	Scrubs
Corynephorus articulatus (Desf.) P. Beauv.	X			T	Mediterranean	Wood clearings Garigues
Holcus lanatus L.	X	X		H	Boreal	Grasslands
Lagurus ovatus L.	X	X		T	Mediterranean	Grasslands
Nasella trichotoma (Nees) Hack.			X	H	Alien	Woods (Pine stands)
Phalaris rotgesii (Husn.) Litard.		X	X	G	Boreal	Marshes
Poa compressa L.			X	H	Boreal	Abandoned fields
Setaria pumila (Poir.) Roem. & Schult.		X		T	Cosmopolitan	Rural sites
Sporobolus indicus (L.) R. Br.	X	X	X	H	Alien	Abandoned fields
Trisetaria panicea (Lam.) Maire	X			T	Mediterranean	Abandoned fields
Vulpia myuros (L.) C.C. Gmel.	X			T	Cosmopolitan	Abandoned fields pastures
Polygonaceae						
Persicaria hydropiper (L.) Delarbre	X		X	T	Boreal	Marshes
Rumex acetosella L.	X			H	Cosmopolitan	Abandoned fields
Portulacaceae						
Portulaca oleracea L.	X			T	Cosmopolitan	Rural sites
Primulaceae						
Lysimachia arvensis (L.) U. Manns & Anderb.	X	X	X	T	Mediterranean	Rural sites Garigues
Rosaceae						
Aphanes inexspectata Lippert	X	X	X	T	European	Pastures
Rubiaceae						
Galium murale (L.) All.			X	T	Mediterranean	Ruderal sites
Galium palustre L.		X	X	H	European	Marshes
Simaroubaceae						
Ailanthus altissima (Mill.) Swingle	X		X	P	Alien	Ruderal sites human settlements
Solanaceae						
Atropa bella-donna L.	X	X	X	H	European	Wood clearings coppices
Datura stramonium L.	X			T	Alien	Ruderal sites cultivated fields
Salpichroa origanifolia (Lam.) Thell.	X			Ch	Alien	Ruderal sites abandoned fields
Solanum nigrum L.	X	X	X	T	Cosmopolitan	Ruderal sites rural sites
Solanum triflorum Nutt.	X			T	Alien	Cultivated fields
Solanum villosum Mill.	X	X		T	Mediterranean	Ruderal sites rural sites
Urticaceae						
Urtica dioica L.	X			H	Cosmopolitan	Ruderal sites wood clearings

there were two Cosmopolitans, *Juncus bufonius* and *Juncus effusus*, but no Aliens, and only three (*Lysimachia arvensis*, *Dittrichia graveolens,* and *Hypochaeris radicata*) were typical of ruderal sites (data non shown).

The partitioning of resident species between low and high frequency classes changed markedly with time from clear-cutting (Table VI). The number of species with very low occurrence (r and +) decreased dramatically in the third year, whereas that of species with frequency $>1\%$ increased from 19 to 38 between the first and third year. In addition, while in the first year no species showed a frequency $>30\%$, five species exceeded this percentage in the third one. These were two Gramineae, *Agrostis castellana* and *Aira caryophyllea*, one sedge, *Carex*

Table V. Occurrence, life form, sprouting ability and chorotype of resident species recorded in cut plots (P_D2006, P_S2007, and P_S2008) and in at least one control pine stand within three years from clear-cutting.

Resident species	P_D 2006	P_S 2007	P_S 2008	Life Form	Sprouting	Chorotype
Apocynaceae						
Periploca graeca L.		X		P	+	Mediterranean
Araliaceae						
Hedera helix L.	X	X		P	+	Mediterranean
Asparagaceae						
Asparagus acutifolius L.	X	X	X	P	+	Mediterranean
Asteraceae						
Crepis leontodontoides All.	X	X	X	H	+	Mediterranean
Dittrichia viscosa (L.) Greuter	X	X	X	H	+	Mediterranean
Erigeron sumatrensis Retz.	X	X	X	T		Alien
Hieracium piloselloides Vill.	X	X	X	H		European
Hypochaeris glabra L.	X	X	X	T		Mediterranean
Leontodon leysseri (Wallr.) Beck.		X	X	T		Mediterranean
Sonchus maritimus L.	X	X	X	H	+	Boreal
Boraginaceae						
Heliotropium europaeum L.	X			T		Mediterranean
Caryophyllaceae						
Cerastium glomeratum Thuill.	X			T		Mediterranean
Cerastium semidecandrum L.	X			T		European
Cistaceae						
Cistus salviifolius L.	X	X	X	P	+	Mediterranean
Tuberaria guttata (L.) Fourr.	X	X	X	T		Mediterranean
Cyperaceae						
Carex distachya Desf.	X			H	+	Mediterranean
Carex flacca Schreb.		X	X	G	+	European
Carex punctata Gaudin		X	X	H	+	Mediterranean
Euphorbiaceae						
Euphorbia cyparissias L.	X	X		H	+	European
Ericaceae						
Erica scoparia L.	X	X	X	P	+	Mediterranean
Fabaceae						
Medicago minima L.	X	X		T		Mediterranean
Trifolium arvense L.	X	X	X	T		European
Fagaceae						
Quercus ilex L.	X	X	X	P	+	Mediterranean
Hypericaceae						
Hypericum perforatum L.	X	X	X	H	+	European
Juncaceae						
Juncus acutus L.		X	X	H	+	Mediterranean
Luzula campestris (L.) DC.	X	X	X	H	+	European
Lamiaceae						
Clinopodium nepetum (L.) O. Kuntze subsp. *nepetum*	X	X		H	+	European
Clinopodium nepetum (L.) O. Kuntze subsp. *sylvaticum* (Bromf.) Peruzzi & F. Conti	X			H	+	European
Prunella laciniata L.	X			H	+	Mediterranean
Prunella vulgaris L.	X			H	+	Boreal
Myrtaceae						
Myrtus communis L.		X	X	P	+	Mediterranean
Oleaceae						
Fraxinus angustifolia Vahl subsp. *oxycarpa* (Willd.) Franco & Rocha Afonso		X		P		Boreal
Phillyrea angustifolia L.		X	X	P	+	Mediterranean
Oxalidaceae						
Oxalis corniculata L.	X	X	X	Ch		Mediterranean

(*Continued*)

Table V. (*Continued*).

Resident species	P_D 2006	P_S 2007	P_S 2008	Life Form	Sprouting	Chorotype
Pinaceae						
Pinus pinea L.	X	X	X	P		Mediterranean
Plantaginaceae						
Veronica officinalis L.		X	X	Ch	+	European
Poaceae						
Agrostis castellana Boiss. & Reut.	X	X	X	H	+	Mediterranean
Aira caryophyllea L.	X	X	X	T		Tropical
Anthoxanthum odoratum L.	X			H	+	European
Brachypodium retusum (Pers.) P. Beauv.	X			H	+	Mediterranean
Brachypodium sylvaticum (Huds.) P. Beauv.	X	X	X	H	+	European
Bromus sterilis L.	X			T		Mediterranean
Dactylis glomerata L.	X			H	+	European
Danthonia decumbens (L.) DC. subsp. *decumbens*		X	X	H	+	European
Molinia caerulea (L.) Moench		X		H	+	Boreal
Poa annua L.	X			T	+	Cosmopolitan
Vulpia bromoides (L.) Gray	X		X	T	+	European
Primulaceae						
Lysimachia linum-stellatum L.	X	X	X	T		Mediterranean
Ranunculaceae						
Clematis flammula L.	X	X		P	+	Mediterranean
Rosaceae						
Crataegus monogyna Jacq.	X	X	X	P	+	European
Malus sylvestris (L.) Mill.	X	X	X	P		European
Rosa cfr *agrestis* Savi		X		P	+	Mediterranean
Rubus cfr *ulmifolius* Schott	X	X	X	P	+	Mediterranean
Rubiaceae						
Rubia peregrina L.	X	X		P	+	Mediterranean
Sherardia arvensis L.	X			T	+	Mediterranean
Salicaceae						
Populus alba L.	X	X	X	P		European
Smilacaceae						
Smilax aspera L.		X	X	P	+	Tropical
Thymelaeaceae						
Daphne gnidium L.	X	X	X	P	+	Mediterranean
Ulmaceae						
Ulmus minor Mill.		X	X	P	+	European
Violaceae						
Viola reichenbachiana Jord. ex Boreau		X	X	H	+	Boreal

flacca, and two shrubs, *E. scoparia* and *Rubus* cfr *ulmifolius*. Four of them have sprouting ability, whereas *A. caryophyllea* is an annual.

Further evolution of plant community

Estimates on the further evolution of plant cover on clear-cut plots were performed in the oldest one, i.e., P_D2006. The overall plant cover increased from 62 to 74% between the third and the fifth year, but differences were not significant (Figure 1). Of the 53 new species inventoried in this plot within the first three years from clear-cutting, only 13 were recorded again in the subsequent two years. Four of these, *Datura stramonium*, *Erigeron canadensis*, *Phytolacca Americana,* and *Salpichroa origanifolia* were Aliens, and two, *Rumex acetosella* and *Vulpia myuros* were Cosmpolitans. However, only four new species, *D. graveolens*, *Erigeron canadensis*, *P. americana* and *R. acetosella*, seemed to be established in the cut strip after five years (data not shown).

To hypothesize what kind of forest type could evolve on cut plots in the absence of artificial replanting, we analysed the frequency of tree species three years after clear-cutting (Table VII). All recorded tree species were residents, and *P. pinea* and *Populus alba* were present in all plots. The highest tree richness was found in P_S2007, to which corresponded a low frequency of distinct species. In P_S2008 we recorded four tree species; amongst these, *P. alba*, *P. pinea,* and *Ulmus minor* showed frequencies comprised between 20 and 36%. Finally,

Table VI. Total number and distribution in frequency classes of new and resident species in the first, second and third year after clear-cutting. For each species, frequency was averaged over the three cut plots.

	Time from clear-cutting (years)		
Frequency (%)	1st	2nd	3rd
		New species (*n*)	
r	18	45	16
+	8	9	12
1–5	6	7	9
6–10	0	2	5
11–15	0	0	0
16–20	0	0	1
>20	0	0	0
Total	32	63	43
		Resident species (*n*)	
R	16	21	5
+	8	5	6
1–5	8	13	15
6–10	7	7	5
11–15	1	2	4
16–20	1	0	5
21–30	2	3	4
31–40	0	1	1
41–50	0	1	1
>50	0	1	3
Total	43	54	49

Note: r (rare), species inventoried only once in one or more plots; + (present), species with <1% frequency in sample areas.

Table VII. Relative frequency (%) of new (N) and resident (R) tree species in cut plots three years after clear-cutting. Species are reported in decreasing order of frequency.

		Cut experimental plots		
Tree Species		P_D2006	P_S2007	P_S2008
Pinus pinea	R	12	8	24
Ulmus minor	R	–	4	36
Populus alba	R	4	4	20
Crataegus monogyna	R	8	8	–
Quercus ilex	R	–	4	4
Fraxinus angustifolia	R	–	4	–
Ailanthus altissima	N	–	–	–
Ficus carica	N	–	–	–

in P_D2006 we inventoried three tree species, and only *P. pinea* had a frequency slightly higher than 10%.

Discussion

After the artificial renewal of stone pine plantations, the recovery of pre-disturbance understory vegetation is of primary importance for the conservation of their biodiversity and floristic value (Tárrega et al. 2012).

The two stone pine plantations chosen for the present study belonged to the same forest type (Tomei et al. 2004), were both around 100 years old, and had similar climatic conditions and management history. Despite this, they showed notable differences in the overall cover, richness, diversity and composition of the understory vegetation. These differences were, probably, the consequence of the lower nitrogen and organic soil content, and the less acidic and humid conditions of the superficial soil of the more elevated plantation (P_D) compared to that located at sea level (P_S). In particular, soil acidity has been reported to strongly control the floristic composition of forest understories (Brunet et al. 1997). In addition, the less developed *Erica* canopy and the proximity to a wide forest clearing could be responsible for the higher presence of annual species in P_D. Differences between plantations were maintained also in cut plots, which should be taken into account, since it can influence evolution patterns (Halpern 1988; Dodson et al. 2008).

The regrowth of vegetation proceeded linearly after clear-cutting, covering 50–75% of the soil surface within the third growing season, and then slowed down. The lowest rate of expansion was observed in cut plots where sparse trees were left for seed dispersal, and could be a consequence of shading or root competition (Connell & Slatyer 1977). Rapid regeneration of vegetation cover is positive for ecosystem stability on sandy soils like those of the present research, because plants entrap and hold nutrients avoiding their losses in stream water (Boring et al. 1981). In the succession model of Noble and Slatyer (1980), this initial pulse of recruitment is mainly due to pioneer species that were rare or absent in predisturbance vegetation. The following phase of slow cover increase is crucial for the composition of future vegetation, because pioneer and late successional species compete for establishment and dominance. Indeed, Malanson et al. (2007), reported that ecosystem resilience was less affected by the degree of initial disturbance than by further alterations to succession.

Three years after clear-cutting, the new plant cover consisted almost exclusively of shrubs and graminoids, and differed markedly from the understory vegetation of control plots in terms of richness, diversity and composition. The shrub *E. scoparia* was by far dominant in all plots, and its successful regeneration was due to both sexual and asexual reproduction. No saplings of the new tree species *F. carica* and *A. altissima* were recorded after three years, suggesting that they were not able to become established in clear-cut zones. In contrast, all resident tree species, except *Malus sylvestris*, were recorded, but their distribution and frequency differed among cut plots. Only the dominant

P. pinea and the pioneer *P. alba* were found in all plots, and it is worth noting that in P_S2007, where sparse adult trees were left for seed dispersal, we found the highest tree richness but also the lowest frequency. Present results suggest that, though slowly, natural regeneration of a stone pine forest is proceeding in all cut plots, but it is probable that final tree composition will differ among them.

In consequence of the massive invasion of new species, coupled to the moderate extinction of resident ones, richness increased markedly in all cut plots; this has been widely reported in response to low disturbance levels, and is regarded positively for maintaining high ecosystem diversity (Halpern 1988; Haeussler et al. 2002; Jobidon et al. 2004; Newmaster et al. 2007).

In the present research, however, diversity was lower in plant communities of cut plots than in the understory vegetation of intact stands, as confirmed by cover and frequency data indicating that only few species were widely diffused and dominant. These findings reveal that clear-cutting severely affected the plant community of stone pine plantations, despite the rapid revegetation of bare soil (Ito et al. 2006). Decreased diversity following clear-cutting was also reported by Elliott et al. (1997) for temperate mixed forests, and by Haeussler et al. (2002) for pine-dominated forests on poor soil, whose conditions are similar to those of our plantations. The higher diversity of the cut plot P_S2007, combined with lower extinction of species and lower regrowth rate, was, perhaps, related to the slightly less severe clear-cutting operations performed on this plot (Brockway & Lewis 2003).

The index of similarity revealed marked shifts in the floristic composition of plant communities after clear-cutting. Colonization by new species started immediately during the first growing season and peaked at the second one, so that it is highly probable that their seeds were already present in the seed bank of intact stone pine plantations (Hopfensperger 2007). In support, common traits within new species were annual living cycle and herbaceous habitus that are both related to the production of persistent seed. More then half of the new species belonged to the typical flora of human-disturbed ruderal sites, and one-third were species with broad ecological amplitude or non-native. These traits are consistent with those reported for invaders of boreal and temperate conifer and mixed forests (Elliott et al. 1997; Dodson et al. 2008), and are considered typical for early successional colonizers (Connell & Slatyer 1977). Asteraceae was the most represented family, and its species are often regarded to be among the most efficient invaders, because of the abundant production of wind-dispersed seed (Boring et al. 1981; Halpern 1988; Swope & Parker 2010).

Most resident species survived the clear-cutting operations, probably thanks to their ability to sprout from underground organs or from other residuals, and this can be considered a trait of good resistance to disturbance of stone pine plantations (Noble & Slatyer 1980; Whitford et al. 1999; Szabo Kraft et al. 2004; Newmaster et al. 2007). According to Ito et al. (2006), the adopted management practice, i.e., clear-cutting on narrow strips, could also have contributed to low plant extinction. Other authors report that the degree of disturbance to soil, litter and plant roots also play an important role in regeneration patterns (Boring et al. 1981; Newmaster et al. 2007; Roberts 2007). Consistently, in the present research, a significantly lower extinction was observed in the cut plot P_S2007, where slightly less severe clear-cutting operations were performed and artificial replanting occurred after the time span of the present research.

Besides the index of similarity, life-form and chorology spectra also indicated that plant composition changed markedly in response to clear-cutting. The proportion of herbaceous annuals (Therophytes) increased at the expense of trees and tall shrubs (Phanerophytes), which is in line with the higher seed production and persistence in the seed bank of the former (Bossuyt & Hermy 2001). Chorology spectra evidenced a marked increase in Boreal and, especially, Cosmopolitan and Alien species and, in turn, a decrease in the proportion of species with Mediterranean and European distribution. The higher presence of Boreal species in cut plots contrasts with the known sensitivity of marginal species to disturbance (Haeussler et al. 2002). It could be explained with the establishment, in cut plots, of a more suitable microclimate, characterized by higher water and temperature excursions than under the intact stone pine canopy (Boring et al. 1981). The increased proportion of species with broad ecological and geographical amplitude or of non-native ones is a common response to severe disturbance and is a major threat in the conservation of natural habitats (Halpern 1988; Rose & Hermanutz 2004; Newmaster et al. 2007; Dodson et al. 2008). Orlandi and Arduini (2010) found that several new species were never recorded previously in the study area (Garbari 2000; Garbari & Borzatti von Loewenstern 2005); thus, their establishment and diffusion in the understory vegetation could markedly reduce the floristic value of these stone pine plantations.

The number of new species was quite similar in all cut plots, indicating that they were equally susceptible to invasion, which had to be expected since they are all located within an agricultural and urbanized area (Newmaster et al. 2007; Dodson et al. 2008). However, the higher proportion of annuals, Cosmopolitans and Aliens in P_D2006, suggests that the floristic integrity of the P_D plantation is more threatened, probably because of its proximity to a disturbed open space that could be a source of invader seeds (Rose & Hermanutz 2004; Hopfensperger 2007). This is confirmed by the fact that several annual species and even one Alien, *Erigeron sumatrensis*, and one Cosmopolitan, *Poa annua*, were also inventoried in the control plot of this plantation (PC_D).

By combining data from the three plots, it was observed that the overall number of new species was higher than that of residents only during the second vegetative season, which is consistent with the hypothesis that the seed bank was the major source of new species, and that most of them had only a transient occurrence. Indeed, while the frequency of most residents tended to increase over time, only one-fifth of new species had a frequency higher than 1% after three years, and, among these, no Aliens were recorded. A rather low ability of new species to colonize cut strips and a small proportion of alien species in terms of plant cover was also reported by Newmaster et al. (2007) and by Dodson et al. (2008), and is considered an indicator of good system resilience (Halpern 1988; Haeussler et al. 2002). Nevertheless, it must not be ignored that several new species were still present in the cut strip P_D2006 after five years, among which Aliens and species that were never recorded previously in the study area (Garbari 2000; Orlandi & Arduini 2010).

Based on the above results, we suggest that the performance with time from clear-cutting of both Aliens and Asteraceae could be used to measure resilience of plant communities. The use of these guilds could be a more suitable approach compared to that of individual indicator species, whose response is mostly location specific (Lindenmayer et al. 2000). Our findings also suggest that determining cover and/or richness alone are not enough to describe the response of plant communities to disturbance (Lindenmayer et al. 2000; Haeussler et al. 2002), but that it is necessary to supplement these measures with estimates of diversity and with the analysis of community composition (Szabo Kraft et al. 2004). Life-form and chorology spectra, that are widely used by botanists to evidence floristic peculiarities and to evaluate changes in composition over time and space, could be also used effectively to study responses to disturbance (Garbari 2000; Chiarucci & Bonini 2005; Poldini et al. 2011).

Conclusions

The increase in richness, the decrease in diversity, and the colonization by annuals and by species with broad ecological amplitude or non-native, were observed in all cut strips and can be considered a common response of the understory vegetation of stone pine plantations to clear-cut operations. Microclimate conditions and local environment affected substantially understory richness and composition of the two plantations of origin and these traits were maintained in the new plant communities. Regrowth rate, extinction of resident specie and conservation of diversity seemed to be also influenced by slight differences in management practices.

Three years after clear-cut the new plant community showed rather good recovery, which was mainly due to the high proportion of species with sprouting ability within residents. The moderate local extinction of resident species and the general low frequency of invasive and Aliens could be interpreted as traits of resistance.

The presence of the shrub *E. scoparia* in the understory of stone pine plantations should be regarded as positive, since its fast growth reduces both risks of erosion and the diffusion of invader species. Nevertheless, it also causes a decrease in diversity and, later on, it could hinder the establishment and growth of tree seedlings.

Before drawing definitive conclusions on the ability of stone pine understory vegetation to restore pre-disturbance conditions, it is essential to monitor, at a greater distance from clear-cut, changes in the frequency and relative cover of indicator species, such as the dominant shrub *E. scoparia*, Aliens and invasive Asteraceae. In order to avoid a new flush of invasion and to reduce the interference with competition dynamics within the new plant community, we recommend planning replanting of tree saplings at a short distance from clear-cut, during the phase of rapid expansion of plant cover.

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