

Rodents in grassland habitats: does livestock grazing matter? A comparison of two Alpine sites with different grazing histories

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ABSTRACT

Intensive livestock grazing is a major force in shaping ecosystems, especially where topography and soil features preclude other farmland practices. In subalpine landscapes, it may be the main disturbance agent. Nevertheless, information on the impact on some species, especially mammals, is lacking. Here, we investigate whether the abundance of rodents in subalpine grasslands of the Alps is related to the intensity of livestock grazing, determined according to the dominant pastoral types. Using Generalized Linear Models, we compared a site intensively grazed until recently ("formerly intensively grazed area", *Dactylis glomerata* L. pastoral type), with an area that experienced little or no grazing in past decades ("lightly grazed area", *Carex sempervirens* Vill. pastoral type). Data were collected during three trapping sessions during summer 2011, using a sampling design based on small trapping plots. Rodents (*Apodemus* spp. and *Eliomys quercinus* Linnaeus, 1766) were mainly trapped in the lightly grazed area. They were almost absent in the formerly intensively grazed area, where tall-herbs and megaforb locally prevail over other grassland species. Our results suggest that the abundance of rodents in grasslands is influenced by grazing history. Even adaptable rodents like *Apodemus* spp. are rare in the formerly intensively grazed site, despite its suitable vegetation cover. This result should encourage natural resource managers to carefully consider and act on the spatial distribution of grazing pressure.

KEY WORDS

Alps,
Apodemus,
Eliomys quercinus,
grazing intensity,
live trapping,
vegetation structure.

RÉSUMÉ

Les rongeurs dans les habitats de prairies : quel est le rôle du pâturage ? Une comparaison de deux sites alpins ayant des histoires de pâturage différentes.

Le pâturage intensif du bétail peut jouer un rôle important pour le maintien des écosystèmes, en particulier lorsque les caractéristiques topographiques et pédologiques empêchent d'autres pratiques agricoles. Dans les paysages subalpins, il peut être le principal agent de perturbation. Néanmoins, les informations sur l'impact sur certaines espèces, en particulier les mammifères, sont mal connues. Au cours de ce travail, nous testons l'hypothèse que l'abondance de rongeurs dans les prairies subalpines des Alpes peut être liée à l'intensité du pâturage, et déterminée en fonction des types pastoraux dominants. En utilisant des modèles linéaires généralisés, nous avons comparé un site intensivement pâturé jusqu'à récemment (« zone autrefois intensivement pâturée », type pastoral à *Dactylis glomerata* L.), avec un site qui a connu un pâturage faible ou pas de pâturage au cours des dernières décennies (« zone modérément pâturée », type pastoral à *Carex sempervirens* Vill.). Les données ont été recueillies en trois sessions de capture au cours de l'été 2011, par un protocole expérimental basé sur de petites parcelles d'échantillonnage. Les rongeurs (*Apodemus* spp. et *Eliomys quercinus* Linnaeus, 1766) ont été capturés surtout dans la zone modérément pâturée, et ils étaient presque absents dans la zone autrefois intensivement pâturée, où les hautes herbes ou mégaphorbiaies prévalent localement sur d'autres espèces des prairies. Nos résultats suggèrent que l'abondance de rongeurs dans les prairies est influencée par l'histoire de pâturage. Même les rongeurs adaptables comme *Apodemus* spp. sont rares dans le site autrefois intensivement pâturé, en dépit de sa couverture végétale convenable. Ce résultat devrait encourager les gestionnaires de ressources naturelles à examiner attentivement et à agir sur la répartition spatiale de la pression de pâturage.

MOTS CLÉS

Alpes,
Apodemus,
Eliomys quercinus,
intensité de pâturage,
piègeage non mortel,
structure de la végétation.

INTRODUCTION

Grazing by both domestic and wild large herbivores is known to strongly affect vegetation cover and structure in grasslands. Large herbivores may affect plant diversity in these habitats (Olf & Ritchie 1998) and their grazing may reduce the overall degree of vegetation height and cover (Schmidt *et al.* 2005; Bakker *et al.* 2009). Changes in vegetation structure produced by large herbivore grazing may also be a consequence of the modifications it induces on physical and mechanical soil properties (e.g., compaction induced by trampling) and their interactions, with a variety of effects depending on the grazing intensity (Schmidt *et al.* 2005; Zhao *et al.* 2007; Zhou *et al.* 2010; Gan *et al.* 2012). Changes in the chemical properties of soil and, especially, potential N fertilization as a consequence of grazing may also have potential effects on plant species composition and growth (Binkley *et al.* 2003; Wesche *et al.* 2010). Via these processes and because of the feeding selectivity of herbivores (Augustine & McNaughton 1998), grazing produces long-lasting changes in landscape patchiness and in vegetation structure and composition within patches (Putman *et al.* 1989; Golodets & Boeken 2006). Changes in soil properties and plant cover will in turn affect other animal populations inhabiting grazed areas, since they ultimately depend on soil and vegetation for food and cover (Gibson *et al.* 1992; Eccard *et al.* 2000; Vandenberghe *et al.* 2009).

In particular, for small terrestrial mammals (Soricomorpha and Rodentia), cover and vegetation height have been shown to be the main factors affecting population densities that are changed by livestock grazing (e.g., Bakker *et al.* 2009).

Reductions in vegetation cover and complexity may increase small mammal exposure to predation (Flowerdew & Ellwood 2001), decrease food quality and lead to exploitative competition (Keesing 1998; Vroni 2007). Trampling, causing a reduction of suitable soil for burrow systems, has also been identified as a grazing-induced factor affecting small mammal communities (Torre *et al.* 2007). Negative consequences of herbivore grazing on small mammals have been investigated and reported in several ecosystems (Eccard *et al.* 2000; Chapman & Ribic 2002; Beever & Brussard 2004; Schmidt *et al.* 2005; Torre *et al.* 2007; Wheeler 2008; Muñoz *et al.* 2009). The assessment of the consequences of livestock grazing on small mammal communities is fundamental because small mammals are key species in grazed ecosystems: they have top-down effects on plants (Manson *et al.* 2001; Bagchi *et al.* 2006; Roth *et al.* 2009; Schepher & Smit 2011) and invertebrate populations (Churchfield *et al.* 1991; Elkinton *et al.* 2004), and they are an important prey source for birds and other mammals (Hanski *et al.* 2001; Taylor 2004).

In spite of this, livestock grazing in the Italian Alps with respect to vertebrates has been studied mainly for its consequences on avian diversity (Laiolo *et al.* 2004; Rolando *et al.* 2006) and for potential competition with mountain ungulates (Mattiello *et al.* 2003; Rüttimann *et al.* 2008; La Morgia & Bassano 2009), whereas little attention has been paid to its effects in small mammals. This lack of knowledge is particularly relevant, since traditional land uses, such as sheep and cattle grazing, have deeply modified the landscape of the Alps, with the result that a large part of their biodiversity is linked to these human practices (Chemini & Rizzoli 2003).

In this paper we aim to investigate whether the current abundance of rodents in subalpine grasslands is related to disturbance and changes in vegetation due to intensive grazing in the past. We focused on a recently abandoned grazing site, with highly depleted vegetation (hereafter “formerly intensively grazed area”), and we compared it to a neighboring area that experienced little or no grazing in past decades (hereafter “lightly grazed area”). Taking into account the fact that communities of small mammals in grassland habitats may be primarily determined by structural attributes of the habitat (Grant *et al.* 1982), we investigated whether vegetation changes caused by grazing in the formerly intensively grazed area negatively affected the presence and abundance of rodents. We tested this hypothesis by trapping rodents in an experimental area in the Alpi Marittime Natural Park, south-western Italian Alps, where a detailed characterization of the vegetation (pastoral types) was already available (Cavallero *et al.* 2003, 2007).

MATERIAL AND METHODS

The present work was carried out as part of the “All Taxa Biodiversity Inventory” (ATBI) within the framework of the Transboundary Integrated Action Plan (Transboundary Cooperation Programme Italy-France Alcotra 2007-2013), funded by the European Commission from 2010 to 2012. The ATBI project aims at identifying all living species that exist within a given area, but also promotes the monitoring and study of the factors affecting biodiversity conservation (Deharveng *et al.* 2015; Villemant *et al.* 2015, this issue). The European Distributed Institute of Taxonomy (EDIT) stimulated the first European ATBI+M in the Alpi Marittime and Mercantour Natural Parks, where the critical situation of several pastures lead us to set ecological studies to evaluate the influence of grazing history on biodiversity, using also other taxa as key groups (e.g., Paschetta *et al.* 2013).

The data collection was carried out at the locality of Gias della Luja, near the village of Palanfrè, Vernante municipality, Vermentina Valley (Province of Cuneo, Piedmont, Italy; 44°11'57"N, 7°29'50"E). Here, cows actively grazed pastures over much of the last century. The grazing pressure (exerted from July to October each year) was not uniformly distributed across the study area, with some areas heavily trampled and vegetation deeply modified by past livestock grazing and some areas only occasionally grazed by cattle, as documented by the Park and confirmed by informal questioning of livestock farmers.

In order to assess the vegetation composition of sites with different grazing histories within our study area, we referred to Cavallero *et al.* (2003, 2007). They determined the vegetation composition (pastoral type) by means of the Daget-Poissonet method (Daget & Poissonet 1971), i.e. via an inventory of the species along homogeneous linear transects. According to their definition, in some cases, pastoral types approximately overlap the traditional phytosociological associations that we report hereinafter.

The lightly grazed area (Fig. 1) is characterized by a *Carex sempervirens* (CS) pastoral type (Cavallero *et al.* 2003). It includes herbaceous vegetation of moderate height (10-30 cm), very limited presence of bare ground and presence of rocks or stones over less than 10% of the area. *Carex sempervirens* Vill., *Anthoxanthum alpinum* Å.Löve & D.Löve, *Festuca* gr. *violacea* and *Nardus stricta* L. are the dominant species (Cavallero *et al.* 2007). In particular, the pastoral type of our study area is characterized by the presence of both *Carex sempervirens* and *Plantago serpentina* All., whose specific contributions are 25% and 14% respectively (Cavallero *et al.* 2007). Other detected species are *Nardus stricta* (6%), *Trifolium alpinum* L. (4%), *Festuca* gr. *ovina* L. (3%), *Helianthemum oelandicum* (L.) (2%), *Avenella flexuosa* (L.) (2%), *Potentilla crantzii* (Crantz) Beck ex Fritsch (2%), *Agrostis tenuis* L. (2%) and *Festuca* gr. *violacea* (2%). This pastoral type is related to the *Caricetum sempervirentis* phytosociological association, *Caricion curvulae* (Cavallero *et al.* 2007). The soil supporting this pastoral type is a middle-altitude, gently sloping, acidified and moderately evolved soil, characterized by oligotrophic conditions.

The formerly intensively grazed site (Fig. 1), which was intensively grazed until a few (< 5) years before the study, is of the *Dactylis glomerata* L. (DG) pastoral type (Cavallero *et al.* 2003). It is characterized by herbaceous vegetation with a height of 40-80 cm, very limited presence of bare ground (less than 5%) and the absence of rocks or stones. The *Dactylis glomerata* type is typically eutrophic since it usually derives from N soil enrichment due to intense grazing activities. *Dactylis glomerata* (specific contribution: 15%) is the dominant species in this pastoral type, which is also characterized, in the Palanfrè area, by *Ranunculus acris* L. (11%) and *Trisetum flavescens* (L.) P.Beauv. (6%). Other species are *Achillea* gr. *millefolium* (5%), *Trifolium pratense* L. (4%), *Anthoxanthum odoratum* (L.) (4%), *Alchemilla* gr. *vulgaris* (4%), *Geranium sylvaticum* L. (3%), *Taraxacum officinale* F.H.Wigg (3%) and *Rumex acetosa* L. (3%). This pastoral type is often typical of resting areas near livestock folds, and a recolonization by tall-herb, megaforb vegetation (including *Urtica* and *Rumex* spp.) is also often observed. According to Cavallero *et al.* (2007), this pastoral type is directly related to the phytosociological alliances of *Arrhenatherion* Koch, 1926 and *Polygono-Trisetion* Braun-Blanq. & Tüxen ex Marshall. The soil supporting this pastoral type is a middle-lower slope, gently or very gently sloping, evolved soil, characterized by eutrophic conditions and a poor skeleton.

Both of the study areas are located at 1350-1400 m a.s.l., on a south-east facing slope, above the tree line. Beech (*Fagus sylvatica* L.) is the dominant tree species in the neighbouring woodlands, but hazels (*Corylus avellana* L.), willows (*Salix* spp.), elders (*Sambucus racemosa* L.) and golden chains (*Laburnum alpinum* (Mill.) Bercht. & Presl) are also widespread along the main streams. Wild ungulates, including ibex (*Capra ibex* Linnaeus, 1758) and chamois (*Rupicapra rupicapra* Linnaeus, 1758) (Cetartiodactyla, Bovidae) are also present in the study area.

To collect data on rodents, we established a sampling design based on small trapping plots. As in Torre *et al.* (2007),

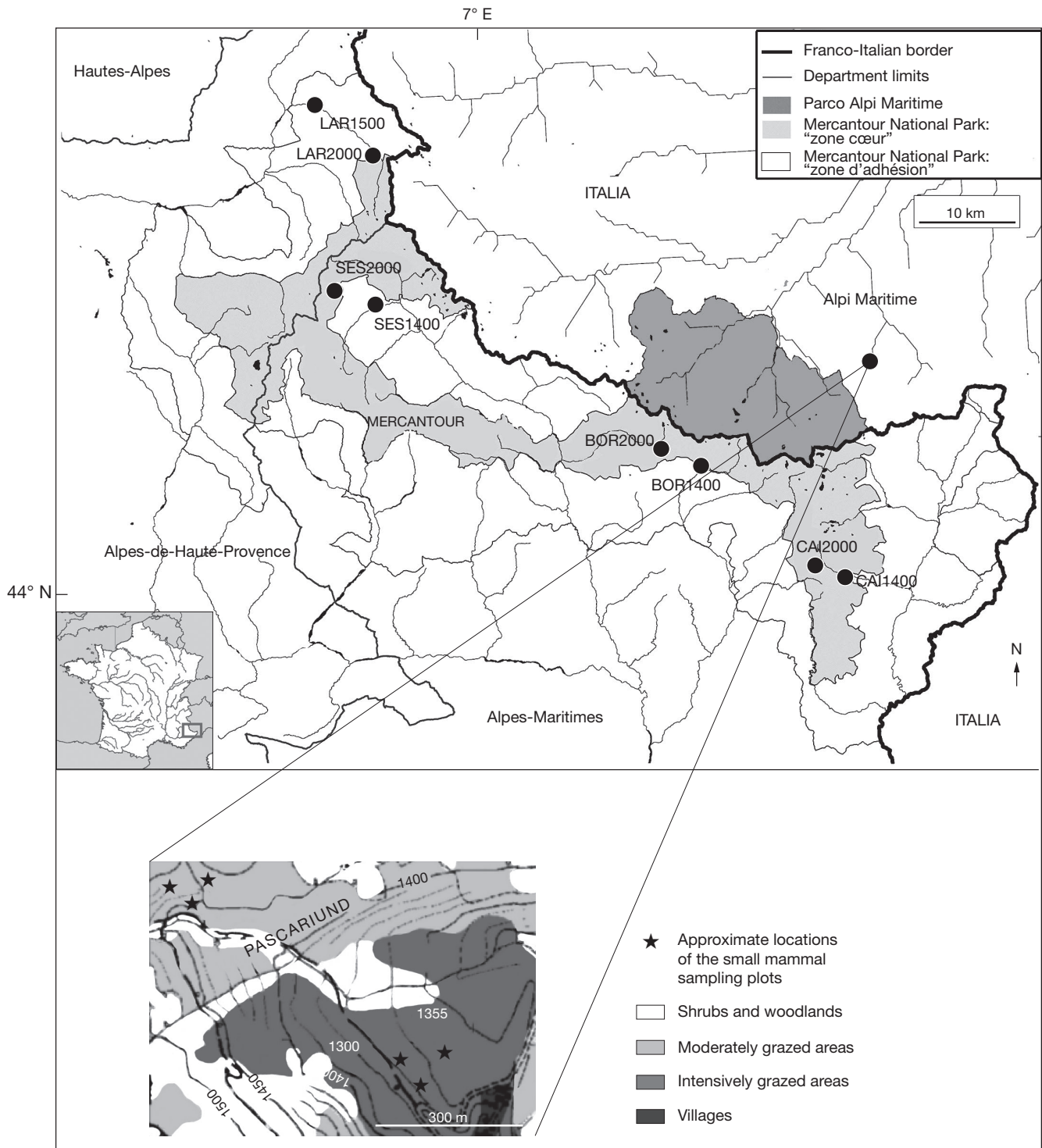


FIG. 1. — Simplified land use in the study area overlaid onto a topographic map. Sampling plots were located within the lightly grazed and in the intensively grazed areas. For the latter areas, in particular, plots were located in sites not used by cattle at present. Plot locations are approximate since we could not leave traps unattended between sessions, with the result that, for each session, plots were not exactly in the same locations as the one before.

height traps were used in each plot, according to a 4 × 2 trapping grid with traps spaced 10 m apart. We set up six plots: three in the lightly grazed area and three in the formerly intensively grazed area. Replicated plots for each grazing level were set up to avoid a confounding effect of plot location on collected data. We used Sherman traps (Large Folding

Aluminum Traps, 7.5 × 9 × 23 cm) whose locations were marked with bamboo stakes and individually labeled. Minimum distance between plots was set to about 100 m (Negro *et al.* 2009) to ensure data independence. Trapped animals were also fur marked (Gurnell & Flowerdew 2006) in order to detect any movement between plots during trapping

sessions. Three trapping sessions, consisting of five consecutive nights, were carried out at the beginning of July, August and September 2011, relocating trapping plots using a GPS device. Traps contained dry hay for bedding and were baited with a mixture of rolled oats, chocolate and anchovy paste, and a piece of apple. They were camouflaged and thermally insulated using herbs or small rocks. They were checked daily at dawn and at dusk. Captured animals were identified, fur marked and then released at the point of capture.

We tested grazing effects on rodents in a Generalized Linear Model framework (McCullagh & Nelder 1989). In particular, we tested the affect of grazing on the Trap Night Index (TNI), i.e. the number of individuals captured per 100 trap-nights (Gurnell & Flowerdew 2006), calculated by pooling the data for all species. The Trapping Success (TS) for each species, calculated as the number of individuals captured over the number of trap-nights (see Simonetti 1986), was not considered to be a suitable variable because for some species no captures were made in the overgrazed areas (see Results section below). For the statistical analysis, the TNI response variable was modeled as a Poisson variable, as opposed to a proportion, because the capture probability was very small relative to the number of trials. The confounding effect of trapping sessions (July, August, September) was tested by including the month of capture as fixed factor in GLMs (as in Torre *et al.* 2007). Taking into account both the explanatory variables relating to the potential effect of grazing on rodents and that relating to the trapping session, we thus developed the four candidate regression models (Burnham & Anderson 2002) shown in Table 2. We selected the best model according to the AIC values (Akaike 1974). Once the final model structure was selected, we fitted a quasi-Poisson and a zero inflated model to test whether the potential overdispersion of count data and the large number of zeros in our dataset affected the results of the analysis (Zeileis *et al.* 2008).

In order to quantify the difference in vegetation composition of the lightly and formerly intensively grazed areas, we considered the data on the frequency of detection of plant species as provided in the work of Cavallero *et al.* (2003, 2007). From these data, we calculated the Jaccard (1901) and Sørensen (1948) asymmetric binary similarity indices. To take full advantage of the quantitative information concerning pastoral types, we also applied the Kulczynski (1928) and Rudjichka (Goodall 1978) asymmetric coefficients. Finally, to quantify and compare the diversity and heterogeneity of each pastoral type, we calculated the Shannon-Weaver (Shannon 1948) and Pielou (1966) indices and displayed abundance data via rank/abundance plots (Magurran 2004).

RESULTS

We caught 34 rodents during the study period, for a total effort of 720 trap-nights. Most individuals trapped (29) belonged to the genus *Apodemus* Kaup, 1829 (Rodentia Bowdich, 1821, Muridae Illiger, 1811), while 5 of them were garden dormice (*Eliomys quercinus* Linnaeus, 1766) (Gliridae). Among the *Apodemus* spp., one individual could be classified as yellow-necked

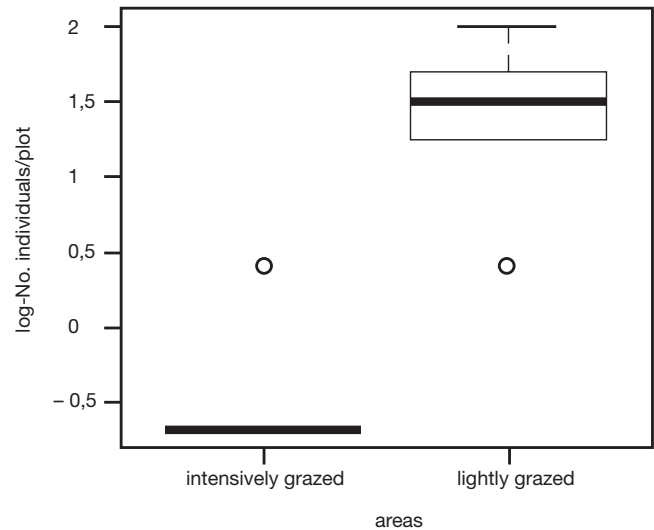


Fig. 2. — Box and whisker plots showing the median, first and third quartile and the minimum and maximum values for the log of the captured animals in each plot. Outliers are plotted as open circles.

fieldmouse *A. flavicollis* (Melchior, 1834), but its discrimination from *A. sylvaticus* (Linnaeus, 1758) and *A. alpicola* (Heinrich, 1952) on the basis of external characters was not possible and none of morphometric, biochemical or molecular analyses needed for a reliable identification (Michaux *et al.* 2001; Barčiová & Macholán 2006; Capizzi & Santini 2007; Gornung *et al.* 2009) was performed. Only two *Apodemus* mice were caught in the formerly intensively grazed area and all the garden dormice were trapped in the lightly grazed area (Fig. 2). We recorded four recapture events involving *Apodemus*, but no movement between plots was detected. The overall TNI, expressed as number of captures per 100 trap-nights, was 5.3 and we did not detect the presence of rodents in most of the plots located in the abandoned grazing area (Table 1).

All candidate models supported the hypothesis of a strong effect of grazing level on the abundance of rodents. Indeed, the model selected according to AIC values was the one that included only grazing level as an explanatory variable (Table 2), while the one attempting to explain the number of captured rodents on the basis of the month only had the biggest AIC value. According to the selected model, the estimated regression coefficient for lightly grazed areas vs. formerly intensively grazed ones was 2.77 ± 0.73 SE. The results of the corresponding quasi-Poisson model and of the zero-inflated regression were consistent with those obtained via the simple Poisson model.

The values of indices calculated to quantify the similarity of the *Carex sempervirens* (CS) and *Dactylis glomerata* (DG) types are reported in Table 3. The diversity and evenness of the two pastoral types were also similar, at least according to the values of the Shannon-Weaver diversity index (5.06 for the CS type, 5.10 for the DG type) and according to the Pielou rank/abundance plots (Fig. 3). Although the relative abundance of the dominant species was higher in the CS type than in the DG one, both curves can be attributed to log normal or even broken stick models, with shallow slopes.

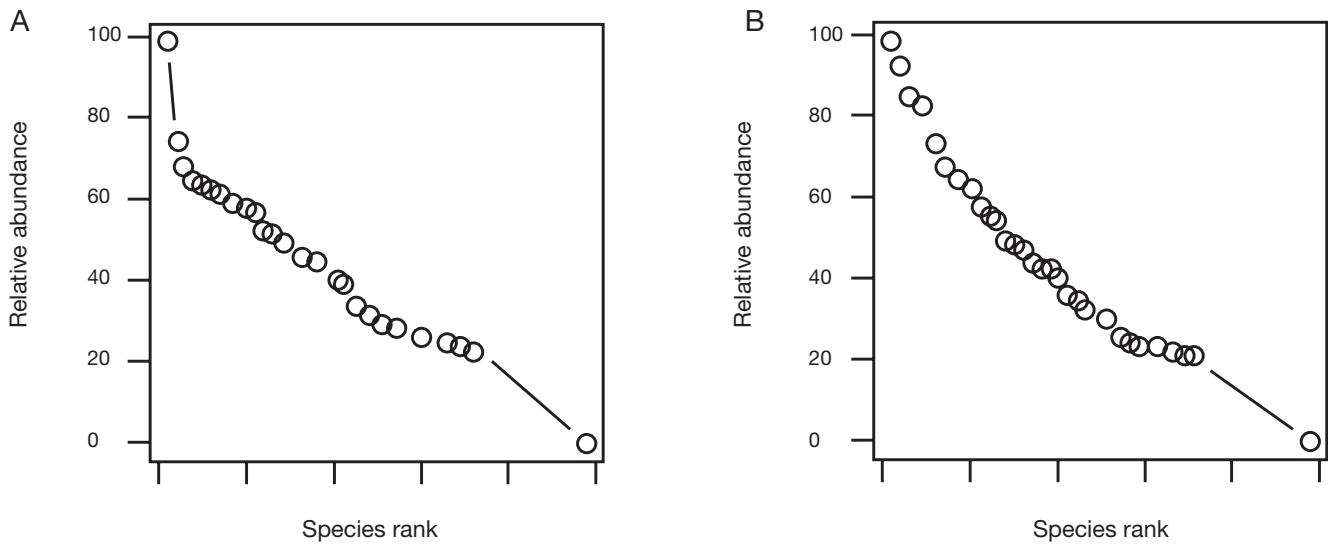


FIG. 3. — Rank/abundance plots for the *Carex sempervirens* (CS) and *Dactylis glomerata* (DG) pastoral types. Plots are based on data published by Cavallero *et al.* (2003, 2007) and refer to: **A**, the pastoral type of the lightly grazed areas (CS type); **B**, the abandoned, formerly intensively grazed areas (DG type).

TABLE 1. — Number of animals captured per 100 trap-nights for each trapping session and for the six different plots (three in intensively grazed and three in lightly grazed areas; survey effort was 40 trap-nights for each plot and session). Recapture events are not included.

Grazing level	Trapping session		
	July	August	September
intensively grazed	2.5	0	0
	0	2.5	0
	0	0	0
lightly grazed	7.5	12.5	10.0
	12.5	7.5	17.5
	0	2.5	10.0

TABLE 2. — Model structures and AIC values of GLMs (Poisson error and log link) hypothesized to explain the number of rodents caught in sampling plots. The first and second models consider both explanatory variables, but the first one also includes their interaction term.

Model structure	AIC
Grazing level * Month	56.72
Grazing level + Month	55.15
Grazing level	52.91
Month	85.08

TABLE 3. — Values of indices calculated to quantify the similarity of the *Carex sempervirens* and *Dactylis glomerata* types.

	Jaccard	Sørensen	Kulczynski	Rudjichka
Index value	0.16	0.28	0.25	0.14

DISCUSSION

As a disturbance agent in ecosystems, grazing by domestic herbivores is likely to induce changes in the vegetation patterns of the landscape and alter ecological processes (Fleischner 1994). In the case of small mammals, the effects of livestock presence have been documented in a wide range of ecosystems and mostly attributed to changes in habitat structure (Tabeni & Ojeda 2003). In spite of this, in Alpine environments and in the Alps in particular, livestock grazing has only been studied with reference to its potential impact on birds and mountain ungulates (Mattiello *et al.* 2003; Laiolo *et al.* 2004; Rolando *et al.* 2006; Rüttimann *et al.* 2008; La Morgia & Bassano 2009). Limited attention has been directed to other vertebrate taxa, which may be strongly affected by grazing, even at small spatial scales, because of their ecology and behavior, and especially because of their limited dispersal abilities. As a consequence, the drawing-up of well-informed management policies is hampered by this lack of knowledge.

In the geographical context of the present study, the lack of data on livestock effects on small mammals may also be due to the usually low trapping success recorded in alpine, subalpine and montane belts (as defined by Pedrotti 2013). In our study, we caught 34 animals and recorded four recapture events (out of 720 trap-nights). Similar values have been recorded in other studies (Hadley & Wilson 2004; Negro *et al.* 2009; Novara *et al.* 2013). In the Western Italian Alps, Patriarca & Debernardi (1997) captured 27 individuals (out of 1502 trap-nights) in habitat types similar to those sampled in our study (ecotones, shrubwoods and open habitats). Moreover, based on the literature data we did not expect high species diversity because of the relatively high elevation of our study area (Lomolino 2001; McCain 2005). But in this respect, we should also mention that in similar habitats Patriarca & Debernardi (1997) also caught bank voles, *Myodes glareolus* (Schreber, 1780) (Rodent-



FIG. 4. — *Apodemus* spp. caught during the study.

tia, Cricetidae), snow voles, *Chionomys nivalis* (Martins, 1842) (Cricetidae), common shrews, *Sorex areneus* (Linnaeus, 1758), and pigmy shrews, *S. minutus* (Linnaeus, 1766) (Eulipotyphla, Soricidae). In our study, we did not record the presence of either voles or shrews. These negative results may be attributable to the limited sampling effort and the use of a single trap type. Indeed, the use of pitfall traps could have increased the probability of capturing shrews (Gurnell & Flowerdew 2006), although with the same Large Folding Sherman traps (7.5 × 9 × 23 cm) and similar sampling efforts we were able to capture *Crocidura suaveolens* Pallas, 1811, *Crocidura leucodon* Hermann, 1780 and *Sorex alpinus* Schinz, 1837, as well as voles, in other sampling areas (Balbo 2008; Novara *et al.* 2013).

Concerning the captured species, our study sites evidently contained suitable habitat for both *Apodemus* (Fig. 4) and the garden dormouse. The latter is less arboreal than some other dormice, often being found on the ground (Bertolino *et al.* 2003). It has been recorded from sea level (Capizzi & Santini 2007) to about 2200 m (Patriarca & Debernardi 1997). The genus *Apodemus* is ecologically flexible. *Apodemus sylvaticus* and *A. flavicollis* are closely related species occurring from plains to

about 2000 m a.s.l. (Capizzi & Santini 2007). They are usually found in forested areas, but they can be more associated with forest edges than other woodland species (Hille & Mortelliti 2010). The presence of the alpine mouse is linked to bushy forests, shrublands, alpine meadows (grazed and rocky), rivers and permanent snow (Reutter *et al.* 2003).

Our results do not provide an unbiased estimate of species composition in our study area because of the use of only one trap type (large Sherman) and because the sampling effort was not calibrated to produce a stable estimate of species richness (Conard *et al.* 2008). Our study should thus be regarded as a preliminary attempt to investigate whether grazing activities influence the presence of rodents in subalpine grasslands. Given the simple comparison of two sites, we are not able to exclude the affects of other factors, such as soil type, on the observed differences in vegetation and, subsequently, on the abundance of rodents. In order to generalize our findings, further investigation and replication are needed.

Nevertheless, evidence of the role of cattle grazing has been provided by Cavallero *et al.* (2003, 2007). These authors showed that the *Dactylis glomerata* pastoral type, character-

istic of our intensively grazed site, with a high abundance of zoogenic and anthropogenic species (*Festuca gr. rubra*, *Agrostis tenuis*, *Phleum alpinum* L., *Trifolium alpinum* and *Rumex* spp.) unquestionably results from the impact of livestock grazing. The differences in vegetation observed between the intensively grazed and the lightly grazed sites cannot be explained without reference to intensive grazing. The latter probably interacted with other site characteristics, such as soil type, to determine the final outcome in terms of both vegetation structure and composition, and hence the presence and abundance of small mammals.

As expected, for the *Dactylis glomerata* pastoral type, we observed a significantly lower abundance of rodents than in lightly grazed areas, as is evident from data shown in Table 1. All candidate regression models confirmed a strong effect of grazing level on the abundance of rodents. In our analyses, we also tested for the month effect because many small mammal populations exhibit cycles, with peaks in densities (Crawley 1970; Gurnell 1978), which may also occur within the same season (e.g., Stradiotto *et al.* 2009). In our study, we recorded a slight increase in the TNI from July to September but, because of the low number of (re)captures, we could not estimate the population density. We therefore limited our consideration to the increase in the number of different animals captured (Mt+1), another abundance index for small mammals that is likely to perform as well as mark-recapture estimates when the trapping protocols remain constant (Slade & Blair 2000). Although we initially hypothesized a relationship between the increase in the number of animals and the progress of the breeding season, this trend was not significant.

In conclusion, our results suggest that the presence and abundance of rodents could be related to vegetation differences in the sampling sites, which can in turn be related to grazing history, excluding a significant effect of the month on the number of captured individuals. Although the limited sampling effort and the lack of replication do not allow a generalization of these results, our findings are consistent with literature documenting the effect of livestock grazing on small mammals due to changes in vegetation structure. Even though Bakker *et al.* (2009) suggested a positive connection between vegetation height and small mammal presence, here we observed that rodents were almost absent in formerly intensively grazed areas, despite the increase in tall herbs. Till now, no evidence of this grazing impact had been given for the grassland habitats of the Alps. Also, it has been recognized that small mammal species can respond quite differently to grazing, depending on habitat type and on species ecology (Tabeni & Ojeda 2005; Tabeni *et al.* 2007; Buesching *et al.* 2011). Because our abundance data mainly refer to *Apodemus* spp., our results point out that even ecologically flexible species suffer from the deep vegetation changes observed after intensive grazing. Given that negative consequences of large herbivore grazing on small mammal populations may occur only above particular threshold ungulate densities (Steen *et al.* 2005), this result should encourage natural resource managers to carefully consider and act on the spatial distribution of grazing pressure, especially within protected areas.

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