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**Restoration treatments to control *Molinia arundinacea* and woody and alien species encroachment in *Calluna vulgaris* heathlands at the southern edge of their distribution**

Massimiliano Probo[a*], Davide Ascoli[b], Michele Lonati[a], Raffaella Marzano[a] and Giampiero Lombardi[a]

[a] Department of Agricultural, Forest and Food Sciences, University of Torino, Largo Braccini 2, Grugliasco, I 10095, Italy

[b] Dipartimento di Agraria, University of Naples Federico II, via Università 100, 80055 Portici, Napoli, Italy

* Corresponding author: Massimiliano Probo. e-mail: massimiliano.probo@unito.it. Phone number: +390116708790

**Abstract**

Throughout the last decades, *Calluna vulgaris* (L.) Hull heathlands have declined across Europe and nowadays their conservation is particularly challenging at the southern edge of their distribution. In the Nature Reserve of Vauda (north-western Italy), six restoration treatments were applied (extensive annual goat browsing, one-off mowing, annual mowing, one-off fire without and with annual browsing, and annual fire) and their effects on plant diversity and the cover of *C. vulgaris*, its competitor grass *Molinia arundinacea* Schrank, woody, and alien species were monitored between 2005 and 2011. In the short-term, most of the treatments changed the vegetation community, reducing *C. vulgaris* cover according to a gradient of increasing biomass removal. In the mid-term, *C. vulgaris, M. arundinacea*, woody and alien species cover followed different trajectories according to the treatment and functional group. Annual fire shifted the vegetation towards a *M. arundinacea*-dominated community, while extensive annual browsing did not affect the heathland community and resulted in the lowest increase in *M. arundinacea*, which showed a remarkable fitness in these environments. Moreover, annual burning and mowing were effective in reducing woody species encroachment ($p < 0.05$), and fire treatments triggered a peak in alien species cover (mainly *Panicum acuminatum* Swartz) in the short-term. Six years after treatment, species richness and Shannon index did not differ between treated and control sites ($p > 0.05$). In conclusion, these results highlight the need and potential benefit of integrating multiple techniques to preserve *C. vulgaris* heathlands at their southern edge.

**Keywords.** Goat browsing, mowing, *Panicum acuminatum*, plant diversity, prescribed burning.

# 1. Introduction

The conservation of threatened habitats is particularly challenging near their range edges, where populations are smaller, fragmented, and more vulnerable to environmental changes (Sexton et al., 2009). Techniques for habitat restoration, which are effective at the core of target species distribution, may have unexpected outcomes at the distribution edges. This is the case of *Calluna vulgaris* (L.) Hull heathlands, a key European cultural landscape and habitat (EU Council Directive, 1992), developed after human mediated disturbance regimes, such as grazing, burning, and mowing (Davies et al., 2016; Fagundez, 2013). Nowadays, heathlands are declining in most European countries due to different drivers of land use and environmental change. The abandonment of traditional management has often led to their conversion to woodlands (Pywell et al., 2011). Moreover, increasing atmospheric nitrogen deposition has favored the replacement of *C. vulgaris* by grasses such as *Molinia* spp. (Bobbink et al., 2010; Terry et al., 2004).

Despite the extensive knowledge on heathland conservation measures in their main oceanic distribution area (Davies et al., 2016; Littlewood et al., 2014; Pywell et al., 2011), very little work is available for their southern edges (Fagundez, 2013). Here, these ecosystems often occur under Continental rather than Atlantic climates and mineral rather than thick organic soils (Lonati et al., 2009). At these southern margins, heathlands are facing major threats because of increased heath fragmentation, minor adaptive capacity, and higher pressure by local and exotic grass, shrub and tree encroachment (Bartolome et al., 2005; Borghesio, 2014). Indeed, woody encroachment happens at faster rates (Ascoli and Bovio 2010), and *C. vulgaris* competes with vicariant and more productive grasses, such as *Molinia arundinacea* Schrank rather than *Molinia caerulea* (L.) Moench (Borghesio et al., 2014; Daněák et al., 2012). Consequently, techniques that are effective in the re-establishment of the dominance of *C. vulgaris* (e.g. browsing, prescribed burning, mowing) may not successfully achieve the target of restoring the composition of the whole plant community (Littlewood et al., 2014) and may promote competitor and alien species (Davies et al., 2016).

To address these issues, a heathland restoration experiment was established in a highly fragmented, continental dry heathland located on Po Plain lowlands, northern Italy (Ascoli et al., 2009). The study aimed to assess the long-term effects (six years after treatments) of browsing, prescribed burning, and mowing for the restoration of heathland vegetation, by answering the following questions: i) what is the effect of single restoration techniques and their combination on plant diversity and species community assemblage? ii) How does restoration affect the cover of *C. vulgaris*, *M. arundinacea* and encroaching woody species? iii) Is there any restoration treatment that triggers the invasion of alien species?

# 2. Materials and Methods

## *2.1 Study area, experimental design, and vegetation surveys*

The study area was located within the Nature Reserve of Vauda, northwest Italy (7°41’17’’E, 45°13’13’’N), at an altitude ranging from 240 to 480 m a.s.l. The climate is continental, with 81% of mean annual precipitation (1 000-1 100 mm) falling between April and November and mean annual temperature about 12°C. The Reserve lies on a fluvio-glacial terrace, characterized by ancient and leached soils with low pH (4.8), high clay content, and a thin organic layer (Borghesio, 2014). The Reserve was instituted in 1993 to maintain a relict heathland ecosystem. Despite protection policies, in the last decades the heathland has declined because of *M. arundinacea* and woody species encroachment (mainly European aspen *Populus tremula* L. and silver birch *Betula pendula* Roth) due to the abandonment of traditional management (i.e. grazing and mowing). Moreover, large and frequent pastoral uncontrolled fires during the winter dry season, when grasses dry out, threaten the heathland (Ascoli and Bovio, 2010).

The experimental area was composed of *C. vulgaris* stands in the building phase (*sensu* Watt, 1955) with an advanced encroachment of woody species, i.e. average (±SE) tree density and basal area were 22,722 ± 1518 stems $ha^{-1}$ and 3.1 ± 0.4 $m^2$ $ha^{-1}$, respectively (Ascoli et al., 2013). Six restoration treatments were applied from 2005 to 2011: 1) annual fire, 2) one-off fire, 3) annual mowing, 4) one-off mowing, 5) extensive annual browsing, and 6) one-off fire + extensive annual browsing. Annual fire was implemented every winter from 2005 to mimic current pastoral practices, one-off fire once under prescribed burning conditions in winter 2005 (for details see Lonati et al., 2009), annual mowing every spring, and one-off mowing once in spring 2005. Mowing was performed mechanically at 8 cm height and included biomass harvesting. Annual and one-off fire were carried out over eight 600 $m^2$ plots each, while annual and one-off mowing over eight 100 $m^2$ plots each. A herd of about 100 goats exploited annual browsing and one-off fire + annual browsing plots (which received a single winter prescribed burn in 2005) for 3.5 h and 3 h $day^{-1}$, respectively, over a period of four weeks between April and May. Annual browsing and one-off fire + annual browsing were carried out over 16 plots each (plots were 1250 $m^2$ and 1000 $m^2$, respectively), with a stocking density of about 135 Animal Units $ha^{-1}$ and a stocking rate of 0.05 AU $ha^{-1}$ $year^{-1}$ (*sensu* Allen et al., 2011). Moreover, eight untreated 300 $m^2$ plots were used as control areas. Since we expected a higher variability of vegetation cover and composition after treatment, due to the more heterogeneous effects produced by the selective feeding behavior of goats (Iussig et al., 2015), the number of plots for extensive annual browsing and one-off fire + extensive annual browsing was double compared to other treatments All 72 experimental plots were fenced and randomly selected within comparable *C. vulgaris* heathland patches, which were chosen on the basis of similar vegetation cover and composition.

In each plot, botanical composition was determined using the vertical point-quadrat method (Daget and Poissonet, 1971) along one fixed 10 m transect. In each transect, at 20 cm intervals, the species touching a steel needle were identified and recorded (i.e. 50 points of vegetation measurement). Since rare species are often missed by this method, a complete list of all other plant species included within a 1 m buffer around the transect line was also recorded (Orlandi et al., 2016).Vegetation surveys were conducted during summer 2004 (pre-treatment year), 2007, 2009, and 2011 (i.e. two, four, and six years after treatments, respectively).

### *2.2 Statistical analyses*

For each species recorded, the frequency of occurrence (number of occurrences/50 points) was calculated for each transect and converted to percentage cover (%) (Pittarello et al., 2016). In particular, the percentage cover of *C. vulgaris*, *M. arundinacea*, woody encroaching species, i.e. species classified as chamaephyte, phanerophyte, or nanophanerophyte according to Raunkiaer (1937), and alien species (Celesti-Grapow et al., 2009) was computed. Species richness and Shannon diversity index were also calculated for each survey.

A Principal Response Curve (PRC) analysis was performed to visualize the overall effect produced by treatments on the botanical composition of treated plots compared to that of control plots over time. The PRC analysis was performed using Canoco 4.5 software (Ter Braak and Šmilauer, 2009).

Generalized Linear Mixed Models (GLMMs) were used to test for differences of each treatment against control for all the vegetation variables (i.e. species richness, Shannon diversity index, *C. vulgaris* cover, *M. arundinacea* cover, woody species cover, and alien species cover) for each of the four years during which vegetation surveys were carried-out. Each treatment was considered as a fixed effect, with control used as a reference level for all the analyses. Poisson distribution was specified for count variables and Gaussian or Gamma distributions were specified for continuous data, depending if normality was met or not, respectively (normality was tested with Kolmogorov-Smirnoff test). Significance tests were performed using the Wald statistic. The GLMMs were carried out using R 3.0.3 (R Development Core Team, 2012), with the glmmADMB package (Fournier et al., 2012).

## 3. Results and Discussion

A total of 66 plant species was detected in botanical surveys (Appendix 1). Six years after treatments, species richness did not differ between treated and control sites, underlying the high stability and resistance to treatments of the floristic composition of *C. vulgaris* heathlands at their southern edge, though inter-annual fluctuation in species richness can occur among years (Figure 1a). In the short-term, most of the treatments changed the heathland community, reducing *C. vulgaris* cover in 2007 (Figures 1c) according to a gradient of increasing biomass removal: extensive annual browsing (which removed little biomass), one-off mowing, annual mowing, one-off fire without and with annual browsing, and annual fire (which removed biomass repeatedly). In the mid-term, we observed changes in *C. vulgaris, M. arundinacea,* woody and alien species cover, which followed different trajectories according to treatment and functional group (Figure 1c-f).

*Molinia arundinacea* cover increased proportionally to biomass removal, as it produced more biomass in southern edge heathlands as compared to *M. caerulea* in the Atlantic European ones (Marrs et al., 2004) (Figure 1d). Annual fire, which simulated current uncontrolled pastoral fires, repeatedly removed the heathland and shifted vegetation towards a *M. arundinacea*-dominated community. Mowing treatments and one-off fire, combined or not with annual browsing, initially reduced *C. vulgaris* cover (Figures 1c) and increased *M. arundinacea* cover in the short-term (Figure 1d), but *C. vulgaris* started recovering at all sites after the first growing season mainly by stump resprouting. However, six years later, one-off fire, combined or not with annual browsing, displayed a lower *C. vulgaris* and a higher *M. arundinacea* cover in comparison to mowing treatments. Indeed, graminoids benefited from both litter and crown biomass consumption in fire treatments. Conversely, mowing did not completely remove the crown and left the litter, which resulted in a higher *C. vulgaris* cover since 2007. In the following years, the recovery rate of *C. vulgaris* in all these treatments was similar, but graminoids maintained a higher abundance in fire treatments, in contrast to herbaceous forbs, as evidenced in Figure 2. Notably, one-off mowing did not show significant differences in both *C. vulgaris* and *M. arundinacea* cover when compared to control plots. Extensive annual browsing, the treatment with the lowest biomass removal, did not affect the heathland structure as *C. vulgaris* is barely consumed by goats (Iussig et al., 2015), and it resulted in the lowest increase of *M. arundinacea*, which was comparable to the one of the control. Since heathland vegetation was always dominated by a low number of species (namely *C. vulgaris*, *M. arundinacea*, and a few other graminoids, Figure 2 and Appendix 1), a situation comparable to that of other heathlands (Hancock and Legg, 2012; Muñoz et al., 2012), Shannon diversity index was not different between treatments and control at the end of the experiment (Figure 1b).

Woody species displayed opposite responses to treatments in comparison to graminoids (Figure 1e). Annual burning and mowing were effective in reducing woody species encroachment. One-off fire and mowing top-killed trees, but aspen and birch sprouted vigorously and only subsequent annual browsing effectively controlled shoot growth. Annual browsing had a delayed effect and woody species cover reached the same level as one-off fire + annual browsing by the end of the study. PRC analysis (Figure 2) highlighted significant differences in the botanical composition between treated and untreated plots ($p < 0.01$), with a marked increase in woody species cover in unmanaged control plots, as showed by the trend of phanerophytes (*P. tremula*, *Frangula alnus* Miller, and *B. pendula*).

Five alien species were inventoried, but only *Panicum acuminatum* Swartz reached a noticeable percentage cover (Appendix 1). Interestingly, *P. acuminatum* was triggered only by fire treatments, displayed a peak in the short-term and decreased to the level of control in 2011 (Figure 1f), remaining higher only in the annual fire treatment (Figure 2). In North America *P. acuminatum* showed a great fitness after annual fire (Walsh, 1995), a trait maintained also outside its natural distribution area (Lonati et al., 2009). However, our results confirm that this invasive species quickly declines when fire frequency is low and it does not become important in terms of density and biomass (Walsh, 1995). Under a long-term perspective, prescribed burning might have the effect in rejuvenating the *P. acuminatum* seed bank rather than considerably increasing its vegetation cover, which is however unfavorable to the control of this alien species.

## 4. Conclusions

Both frequent fires simulating current uncontrolled pastoral burns and lack of management promote heathland losses at the southern edge of their distribution, stressing the need for conservation measures. Moreover, the encroaching grass *Molinia arundinacea*, woody and alien species, such as *Populus tremula* and *Panicum acuminatum*, appear to have a high resilience to different restoration treatments. Results evidence the need and potential benefit of integrating multiple techniques to preserve southern edge fragmented heathlands. The restoration of these habitats may not be effective with just one of the tested treatments, since all the techniques involve trade-offs between undesired effects, efficacy and operational difficulties. However, six years after treatments, extensive goat browsing and annual mowing provided the best results for the maintenance of *Calluna vulgaris* and kept woody and alien species under a critical level. Likewise in Atlantic heathlands, prescribed burning may be also valuable for *Calluna* heathlands restoration at their southern range, but only when applied with long return intervals (i.e. longer than six years, but further research is needed to establish a suitable return interval). A higher caution in the use of fire is mandatory because of the presence of encroaching species with marked fire-traits adapted to a more fire-prone environment when compared to Atlantic regions, which can benefit greatly from repeated burns at the expense of *Calluna* heaths.

## 5. Acknowledgements

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## 6. References


Allen, V.G., Batello, C., Berretta, E.J., Hodgson, J., Kotchmann, M., Li, X., McIvor, J., Milne, J., Morris, C., Peeters, A., Sanderson, M., 2011. An international terminology for grazing lands and grazing animals. Grass Forage Sci. 66, 2-28.

Ascoli, D., Beghin, R., Ceccato, R., Gorlier, A., Lombardi, G., Lonati, M., Marzano, R., Bovio, G., Cavallero, A., 2009. Developing an Adaptive Management approach to prescribed burning: a long-term heathland conservation experiment in north-west Italy. Int J Wildland Fire. 18, 727-735.

Ascoli, D., Bovio, G., 2010. Tree encroachment dynamics in heathlands of north-west Italy: the fire regime hypothesis. iForest. 3, 137-143.

Ascoli, D., Lonati, M., Marzano, R., Bovio, G., Cavallero, A., Lombardi, G., 2013. Prescribed burning and browsing to control tree encroachment in southern European heathlands. Forest Ecol Manag. 289, 68-77.

Bartolome, J., Plaixats, J., Fanlo, R., Boada, M., 2005. Conservation of isolated Atlantic heathlands in the Mediterranean region: effects of land-use changes in the Montseny biosphere reserve (Spain). Biol. Conserv. 122, 81-88.

Bobbink, R., Hicks, K., Galloway, J., Spranger, T., Alkemade, R., Ashmore, M., Bustamante, M., Cinderby, S., Davidson, E., Dentener, F., Emmett, B., Erisman, J.W., Fenn, M., Gilliam, F., Nordin, A., Pardo, L., De Vries, W., 2010. Global assessment of nitrogen deposition effects on plant terrestrial biodiversity: a synthesis. Ecol. Appl. 20, 30–59.

Borghesio, L. (2014). Can fire avoid massive and rapid habitat change in Italian heathlands? J Nat Conserv. 22(1), 68-74.

Celesti-Grapow, L., Alessandrini, A., Arrigoni, P.V., Banfi, E., Bernardo, L., Bovio, M., Brundu, G., Cagiotti, M.R., Camarda, I., Carli, E., Conti, F., Fascetti, S., Galasso, G., Gubellini, L., La Valva, V., Lucchese, F., Marchiori, S., Mazzola, P., Peccenini, S., Poldini, L., Pretto, F., Prosser, F., Siniscalco, C., Villani, M.C., Viegi, L., Wilhalm, T., Blasi, C., 2009. The inventory of the non-native flora of Italy. Plant Biosyst. 143, 386–430.

Daget, P., Poissonet, J., 1971. Une métode d'analyse phytologique des prairies. Ann. Agron. 22, 5–41.

Daněák, M., Duchoslav, M., Trávníček, B., 2012. Taxonomy and cytogeography of the *Molinea caerulea* complex in central Europe. Preslia, 84, 351-374.

Davies, G. M., Kettridge, N., Stoof, C. R., Gray, A., Ascoli, D., Fernandes, P. M., Marss, R., Allen, K. A., Doerr, S. H., Clay, G. D., McMorrow, J., 2016. The role of fire in UK peatland and moorland management: the need for informed, unbiased debate. Phil. Trans. R. Soc. B. 371(1696), 20150342.

EU Council Directive, 1992. Natura 2000 Network on the Conservation of Natural Habitats and of Wild Fauna. COUNCIL DIRECTIVE 92/43/EEC (1) of 21 May 1992. The Council of the European Communities.

Fagundez, J., 2013. Heathlands confronting global change: drivers of biodiversity loss from past to future scenarios. Ann Bot-Lonon. 111, 151-172.

Fournier, D.A., Skaug, H.J., Ancheta, J., Ianelli, J., Magnusson, A., Maunder, M.N., Nielsen, A., Sibert, T.J. 2012. AD Model Builder: using automatic differentiation for statistical inference of highly parameterized complex nonlinear models. Optim Method Softw. 27, 233–249.

Hancock, M. H., Legg, C. L., 2012. Diversity and stability of ericaceous shrub cover during two disturbance experiments:one on heathland and one in forest. Plant Ecol. Diversity. 5, 275-287.

Iussig, G., Lonati, M., Probo, M., Lombardi, G., 2015. Plant species selection by goats foraging on montane semi-natural grasslands and grazable forestlands in the Italian alps. Ital J Anim Sci. 14, 484-494.

Littlewood, N.A., Greenwood, S., Quin, S.L.O., Pakeman, R.J., Woodin, S.J., 2014. Long-term trends in restored moorland vegetation assemblages. Community Ecol. 15, 104-112.
Lonati, M., Gorlier, A., Ascoli, D., Marzano, R., Lombardi, G., 2009. Response of the alien species *Panicum acuminatum* to disturbance in an Italian lowland heathland. Bot Helv. 119, 105-111.
Marrs, R. H., Philipps, J. D. P., Todd P. A., Ghorbani J., Le Duc, M. G. 2004. Control of *Molinia caerulea* on upland moors. J. Appl. Ecol. 41, 398-411.
Muñoz, A., García-Duro, J., Álvarez, R., Pesqueira, X. M., Reyes, O., Casal, M., 2012. Structure and diversity of *Erica ciliaris* and *Erica tetralis* heathlands at different successional stages after cutting. J Environ Manage. 94, 34-40.
Orlandi, S., Probo, M., Sitzia, T., Trentanovi, G., Garbarino, M., Lombardi, G., Lonati, M., 2016. Environmental and land use determinants of grassland patch diversity in the western and eastern Alps under agro-pastoral abandonment. Biodivers. Conserv. 25(2), 275-293.
Pittarello, M., Probo, M., Lonati, M., Lombardi, G., 2016. Restoration of sub-alpine shrub-encroached grasslands through pastoral practices: effects on vegetation structure and botanical composition. Appl Veg Sci. 19(3), 381-390.
Pywell, R.F., Meek, W.R., Webb, N.R., Putwain, P.D., Bullock, J.M., 2011. Long-term heathland restoration on former grassland: The results of a 17-year experiment. Biol. Conserv. 144, 1602-1609.
R Development Core Team, 2012. R Development Core Team, Vienna, Austria.
Raunkiaer, C., 1937. Plant Life Forms. The Clarendon Press, Oxford.
Sexton, J.P., McIntyre, P.J., Angert, A.L., Rice, K.J. 2009. Evolution and ecology of species range limits. Annu Rev Ecol Syst. 40, 415-436.
Ter Braak, C.J.F., Šmilauer, P., 2009. CANOCO 4.5. Biometrics - Plant Research International, Wageningen, the Netherlands.
Terry, A.C., Ashmore, M.R., Power, S.A., Allchin, E.A., Heil, G.W., 2004. Modelling the impacts of atmospheric nitrogen deposition on Calluna-dominated ecosystems in the UK. J. Appl. Ecol. 41, 897-909.
Walsh, R. A., 1995. *Dichanthelium acuminatum*. In: Fire Effects Information System, [Online].
Watt, A. S., 1955. Bracken versus heather, a study in plant sociology. J Ecol. 43(2), 490-506.

## 7. Figures

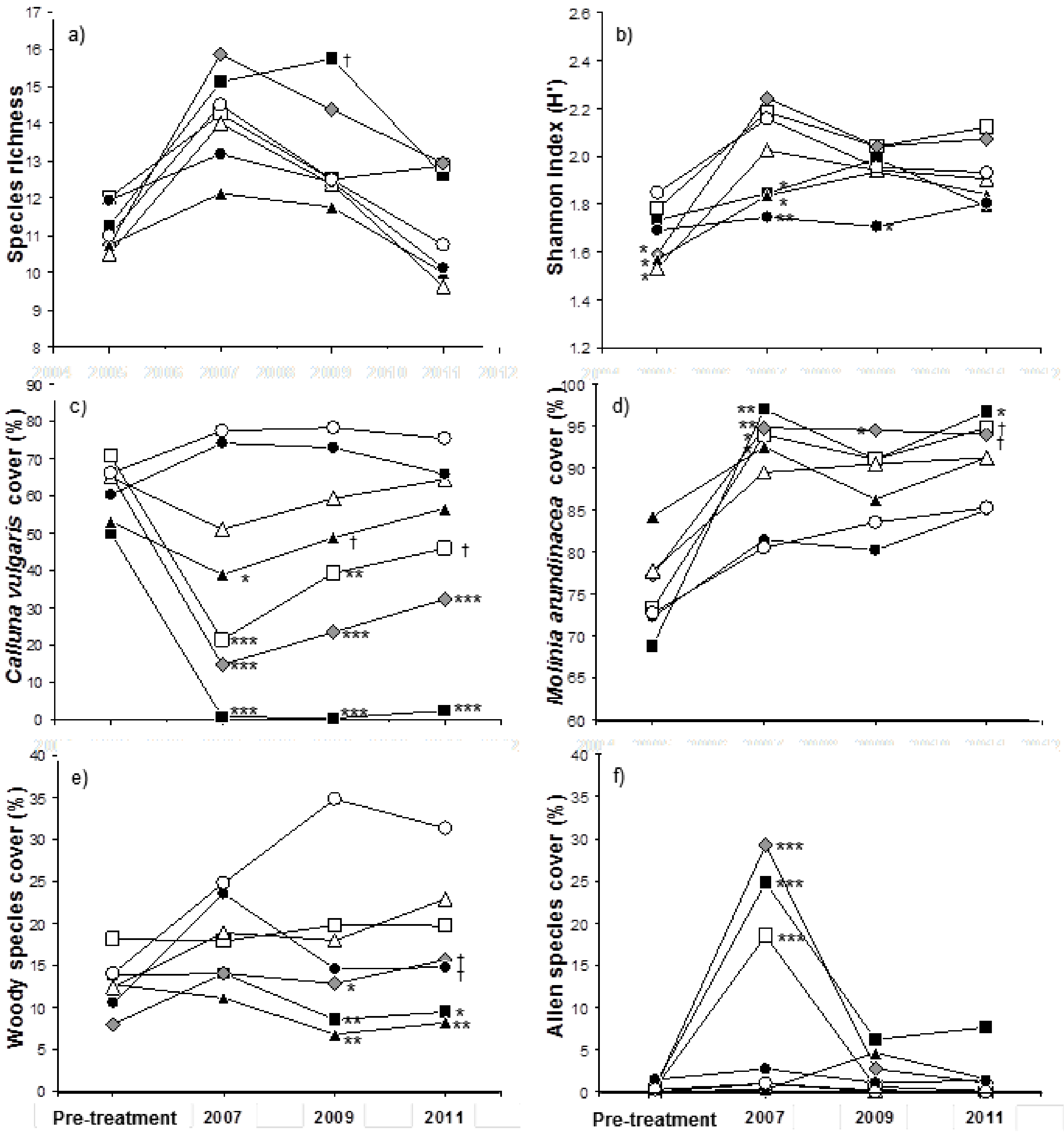


**Fig. 1.** a) Species richness, b) Shannon diversity index, c) *Calluna vulgaris* cover (%), d) *Molinia arundinacea* cover (%), e) woody species cover (%), and f) alien species cover (%) of *C. vulgaris* heathlands, untreated (i.e. control plots) or subjected to six restoration treatments. † = $p < 0.1$; * = $p < 0.05$; ** = $p < 0.01$; *** = $p < 0.001$. Statistical significance values are related to differences of each treatment against control. ■ = annual fire; □ = one-off fire; ◆ = one-off fire + annual goat browsing; ▲ = annual mowing; △ = one-off mowing; ● = annual goat browsing; ○ = untreated.

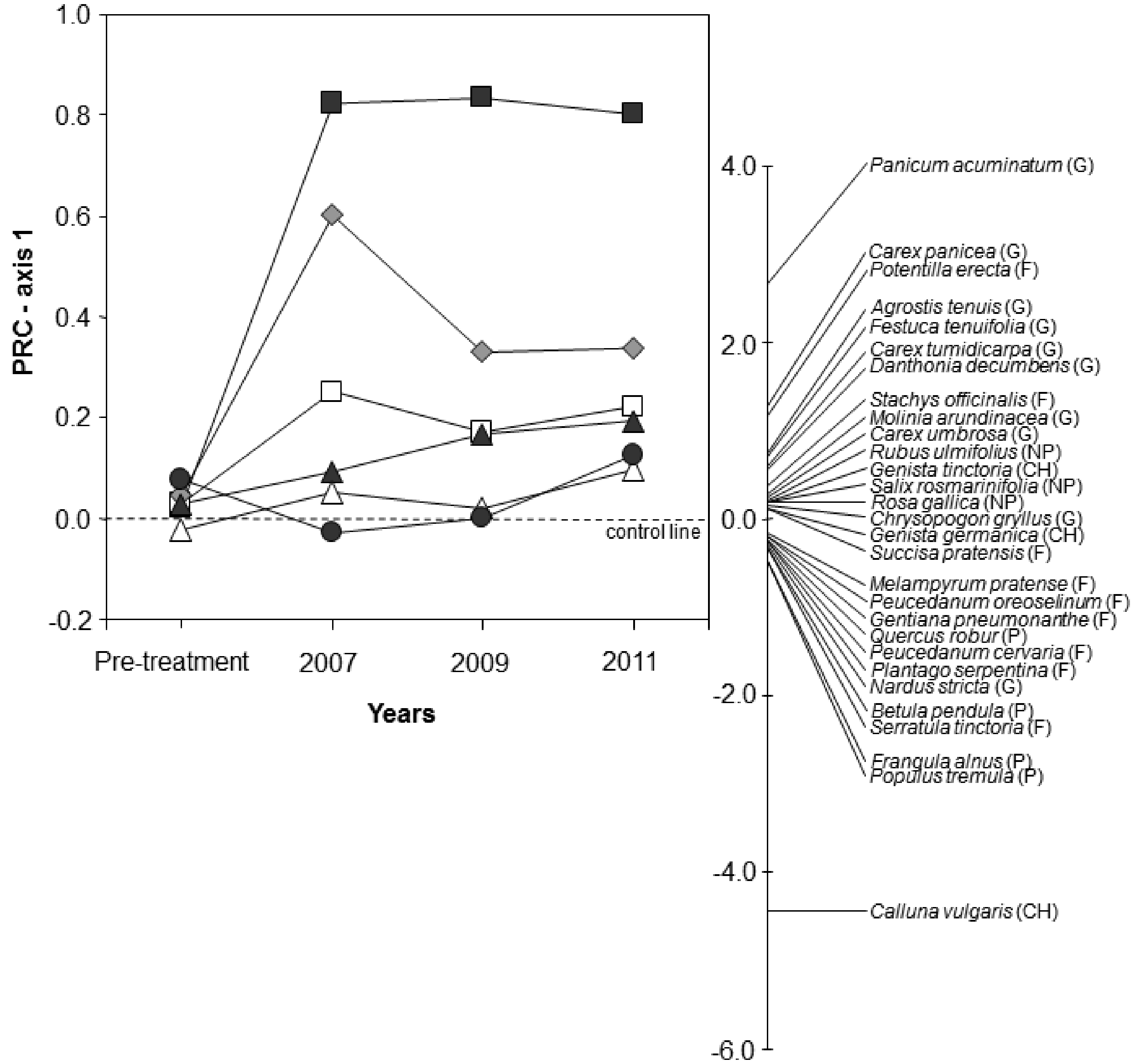


**Fig. 2.** PRC (axis 1) showing effects on botanical composition of *C. vulgaris* heathlands produced by the implementation of six restoration treatments with respect to untreated control (control line) from pre-treatment stage (2004) to 2011. The scores of the most frequent species (namely, species present in at least the 14 plots, i.e. the 5% of total plots) are shown on the right side of the graph: positive values represent species whose canopy cover increased after treatments, whereas negative values represent species whose canopy cover decreased over time. Letters within brackets indicate: G = graminoid, F = herbaceous forb, NP = nanophanerophyte, CH = chamaephyte, P = phanerophyte. ■ = annual fire; □ = one-off fire; ◇ = one-off fire + annual goat browsing; ▲ = annual mowing; △ = one-off mowing; ● = annual goat browsing; ○ = untreated.

# 8. Appendix

**Appendix 1.** Species frequency, name, growth form, life form, origin, mean cover under different restoration treatments and years (2004, i.e. pre-treatment year, 2007, 2009, and 2011).

| Species frequency (%) | Species name | Growth forms (Raunkiaer, 1937) | Life form | Origin | Mean species cover (%) Annual fire | | | | One-off fire | | | | Fire + browsing | | | | Annual mowing | | | | One-off mowing | | | | Annual browsing | | | | Control | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | 2004 | 2007 | 2009 | 2011 | 2004 | 2007 | 2009 | 2011 | 2004 | 2007 | 2009 | 2011 | 2004 | 2007 | 2009 | 2011 | 2004 | 2007 | 2009 | 2011 | 2004 | 2007 | 2009 | 2011 | 2004 | 2007 | 2009 | 2011 |
| 100.0 | *Molinia arundinacea* | Hemicryptophyte | Graminoid | Native | 68.8 | 97.0 | 91.0 | 96.8 | 73.3 | 94.0 | 91.0 | 94.8 | 77.3 | 94.8 | 94.5 | 94.0 | 84.3 | 92.5 | 86.3 | 91.3 | 77.8 | 89.5 | 90.5 | 91.3 | 72.3 | 81.4 | 80.3 | 85.0 | 72.8 | 80.5 | 83.5 | 85.3 |
| 96.5 | *Calluna vulgaris* | Chamaephyte | Forb | Native | 50.0 | 0.6 | 0.2 | 2.2 | 70.5 | 21.3 | 39.3 | 45.8 | 65.8 | 14.8 | 23.4 | 32.3 | 53.0 | 38.8 | 48.8 | 56.5 | 65.0 | 51.0 | 59.3 | 64.5 | 60.3 | 74.1 | 72.9 | 65.8 | 66.0 | 77.5 | 78.3 | 75.3 |
| 93.1 | *Potentilla erecta* | Hemicryptophyte | Forb | Native | 1.4 | 3.6 | 5.5 | 4.0 | 0.9 | 4.3 | 4.1 | 7.3 | 1.8 | 8.1 | 5.5 | 6.7 | 1.0 | 2.4 | 1.7 | 1.4 | 0.5 | 3.0 | 1.8 | 2.8 | 2.2 | 2.0 | 1.7 | 2.1 | 1.8 | 3.8 | 2.4 | 2.1 |
| 92.7 | *Frangula alnus* | Phanerophyte | Forb | Native | 0.5 | 1.1 | 0.6 | 1.2 | 0.9 | 1.2 | 0.9 | 0.9 | 2.8 | 2.3 | 1.5 | 1.7 | 2.4 | 0.3 | 0.3 | 0.3 | 2.9 | 1.9 | 0.9 | 0.5 | 1.0 | 2.7 | 3.0 | 2.6 | 1.9 | 2.6 | 4.9 | 4.4 |
| 92.4 | *Carex panicea* | Geophyte | Graminoid | Native | 2.8 | 11.8 | 18.8 | 43.8 | 2.5 | 16.3 | 9.8 | 23.0 | 3.3 | 17.9 | 17.6 | 38.9 | 3.6 | 22.0 | 30.3 | 39.3 | 2.8 | 15.8 | 7.5 | 17.3 | 4.4 | 7.6 | 8.1 | 23.9 | 2.8 | 14.5 | 7.0 | 10.5 |
| 86.5 | *Populus tremula* | Phanerophyte | Forb | Native | 4.6 | 9.5 | 5.0 | 7.0 | 4.3 | 8.5 | 9.0 | 11.0 | 6.4 | 8.3 | 9.1 | 10.5 | 3.3 | 5.8 | 3.5 | 6.3 | 3.4 | 11.0 | 12.0 | 16.5 | 7.0 | 18.2 | 10.1 | 9.6 | 8.8 | 17.3 | 24.0 | 21.5 |
| 58.0 | *Serratula tinctoria* | Hemicryptophyte | Forb | Native | 0.2 | 0.2 | 0.2 | 0.4 | 0.2 | 0.4 | 0.3 | 0.2 | 0.2 | 0.5 | 0.2 | 0.2 | 0.2 | 0.6 | 0.4 | 0.2 | 0.2 | 0.7 | 0.4 | 0.1 | 0.6 | 0.4 | 0.5 | 0.3 | 0.2 | 1.6 | 0.4 | 0.3 |
| 55.6 | *Salix rosmarinifolia* | Nanophanerophyte | Forb | Native | 1.4 | 1.7 | 1.6 | 0.6 | 0.6 | 7.8 | 8.8 | 6.3 | 2.3 | 1.8 | 1.7 | 3.0 | 2.1 | 1.8 | 2.3 | 1.0 | 4.0 | 3.5 | 3.8 | 4.0 | 0.9 | 1.0 | 0.7 | 1.0 | 2.3 | 3.8 | 2.5 | 2.8 |
| 54.5 | *Panicum acuminatum* | Therophyte | Graminoid | Alien | 0.5 | 24.3 | 6.1 | 7.4 | 0.5 | 18.5 | 0.6 | 0.1 | 0.9 | 29.3 | 2.8 | 1.1 | - | 0.3 | 4.6 | 1.5 | - | 1.0 | 0.2 | - | 1.4 | 2.7 | 1.1 | 0.9 | 0.3 | 1.0 | - | - |
| 53.1 | *Danthonia decumbens* | Hemicryptophyte | Graminoid | Native | 0.1 | 0.1 | 6.1 | 1.0 | 1.0 | 0.1 | 3.6 | 1.8 | 0.7 | 1.1 | 6.7 | 3.3 | 0.4 | - | 1.2 | 0.9 | - | 0.6 | 2.2 | 1.3 | 1.1 | 0.6 | 0.5 | 0.5 | 0.3 | 0.5 | 1.3 | 0.3 |
| 51.7 | *Betula pendula* | Phanerophyte | Forb | Native | 2.7 | 0.8 | 0.4 | 0.3 | 0.3 | 0.2 | 0.4 | 0.4 | 1.1 | 0.7 | 0.3 | 0.2 | 4.0 | 2.8 | 0.4 | 0.1 | 1.9 | 2.2 | 1.1 | 1.8 | 0.4 | 0.7 | 0.1 | 1.0 | - | 0.7 | 1.3 | 0.2 |
| 37.2 | *Carex tumidicarpa* | Hemicryptophyte | Graminoid | Native | 1.5 | 1.0 | 7.5 | - | - | 3.8 | 7.8 | 0.3 | 0.2 | 1.5 | 3.4 | - | 0.3 | 1.0 | 2.8 | - | 0.3 | 0.9 | 5.0 | 0.8 | 0.1 | 0.2 | 1.2 | 0.1 | 1.3 | 0.5 | 0.1 | 0.3 |
| 33.7 | *Peucedanum cervaria* | Hemicryptophyte | Forb | Native | 0.3 | - | - | - | 0.9 | 0.8 | 0.3 | - | 0.9 | 0.6 | 0.5 | 0.3 | 0.9 | 0.8 | 1.0 | 0.4 | 0.8 | 0.6 | 1.1 | 0.1 | 0.4 | 0.2 | 0.2 | - | 1.3 | 0.2 | 0.6 | - |
| 32.6 | *Quercus robur* | Phanerophyte | Forb | Native | 0.8 | 0.2 | 0.2 | 0.2 | 0.1 | 0.2 | 0.6 | 0.6 | 0.4 | 0.1 | 0.1 | 0.1 | 0.8 | 0.3 | 0.3 | 0.6 | 0.1 | 0.1 | 0.2 | 0.1 | 0.5 | 0.2 | 0.1 | - | 0.8 | 0.3 | 1.9 | 2.0 |
| 31.9 | *Melampyrum pratense* | Therophyte | Forb | Native | 0.1 | 0.6 | 1.5 | - | 0.1 | 0.2 | - | - | 0.1 | 1.6 | 0.1 | - | 0.1 | 0.2 | - | - | 0.1 | 4.8 | 0.6 | - | 0.1 | 2.0 | 0.1 | 0.2 | - | 0.9 | 0.3 | - |
| 29.9 | *Agrostis tenuis* | Hemicryptophyte | Graminoid | Native | - | 1.7 | 0.7 | 1.1 | - | 1.8 | 0.4 | 0.4 | - | 2.1 | 0.2 | 0.5 | - | - | 0.3 | 0.8 | - | 0.1 | 0.5 | - | - | - | 0.2 | - | 0.1 | 0.3 | - | - |
| 28.8 | *Festuca tenuifolia* | Hemicryptophyte | Graminoid | Native | - | 1.8 | 2.8 | 3.8 | - | 1.9 | 1.8 | 1.8 | 0.1 | 2.1 | 0.1 | 2.4 | - | 0.5 | 0.8 | 1.8 | - | 1.3 | 2.3 | 2.0 | - | 0.2 | 0.4 | 1.4 | 0.3 | 1.3 | 0.3 | 1.0 |
| 25.7 | *Rosa gallica* | Nanophanerophyte | Forb | Native | 0.1 | 0.4 | 0.4 | 0.2 | 0.1 | - | - | 0.3 | 0.3 | 0.6 | 0.1 | 0.2 | - | - | - | - | - | - | - | - | 0.3 | 0.6 | 0.4 | 0.7 | 0.1 | 0.1 | 0.1 | 0.3 |
| 22.9 | *Genista tinctoria* | Chamaephyte | Forb | Native | - | 0.5 | 0.9 | - | 1.0 | 1.3 | 0.8 | 0.1 | 0.6 | 1.3 | 0.3 | 0.1 | 0.5 | 0.3 | 0.3 | - | 0.3 | 1.0 | 0.1 | 0.1 | 0.1 | - | 0.3 | 0.2 | 1.8 | 2.5 | 0.3 | 0.3 |
| 22.2 | *Stachys officinalis* | Hemicryptophyte | Forb | Native | - | 0.9 | 0.1 | 0.3 | - | 0.1 | - | - | - | 0.6 | 0.1 | 0.1 | - | 0.6 | 0.3 | - | 0.1 | - | 0.1 | - | - | 0.2 | - | - | - | - | - | - |
| 21.5 | *Genista germanica* | Chamaephyte | Forb | Native | 0.1 | 0.3 | 0.1 | - | 0.5 | 0.1 | 0.1 | 0.8 | 0.1 | 0.5 | 0.2 | 0.1 | 0.3 | 0.1 | - | - | 0.1 | 0.6 | 0.3 | - | 0.2 | 0.1 | - | 0.1 | 0.5 | 1.0 | 0.3 | 0.3 |
| 12.2 | *Nardus stricta* | Hemicryptophyte | Graminoid | Native | 2.3 | 0.3 | 1.0 | - | 8.5 | - | - | - | 0.1 | - | 0.1 | - | 0.3 | 0.3 | 0.1 | - | - | 0.1 | - | - | 4.5 | 2.3 | 1.3 | 0.5 | 2.5 | 1.0 | - | - |
| 10.8 | *Rubus ulmifolius* | Nanophanerophyte | Forb | Native | - | 0.3 | 0.3 | 0.1 | - | - | - | - | - | 0.1 | - | 0.1 | - | - | - | - | - | - | - | - | 0.1 | 0.1 | 0.1 | - | - | - | - | - |
| 10.4 | *Peucedanum oreoselinum* | Hemicryptophyte | Forb | Native | - | 0.1 | 0.3 | - | 1.3 | - | - | - | 0.1 | 0.3 | - | - | - | - | - | - | - | - | - | - | - | 0.2 | - | - | 1.5 | 1.8 | 0.3 | 0.5 |
| 9.0 | *Plantago serpentina* | Hemicryptophyte | Forb | Native | - | - | 0.1 | - | 0.8 | - | - | - | 0.3 | - | - | - | - | - | - | - | - | - | - | - | 0.6 | 0.3 | 0.1 | - | - | 0.8 | 0.3 | - |
| 8.3 | *Carex umbrosa* | Hemicryptophyte | Graminoid | Native | - | - | - | 1.5 | - | - | - | 3.3 | - | - | - | 1.3 | - | - | - | - | - | - | - | 0.8 | 0.1 | - | - | 0.5 | - | - | 1.0 | - |
| 8.3 | *Succisa pratensis* | Hemicryptophyte | Forb | Native | - | 0.1 | 0.1 | - | - | 0.1 | - | 0.3 | 0.1 | - | - | - | - | 0.1 | 0.1 | - | - | - | - | - | 0.1 | 0.1 | - | - | - | - | - | - |
| 6.6 | *Gentiana pneumonanthes* | Hemicryptophyte | Forb | Native | - | 0.1 | - | - | 0.1 | - | - | - | - | - | - | - | - | - | - | - | - | - | - | 0.3 | - | 0.1 | - | - | - | - | 0.5 | - |
| 5.2 | *Chrysopogon gryllus* | Hemicryptophyte | Graminoid | Native | - | - | - | - | - | - | - | - | - | 0.1 | 0.4 | 0.3 | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - |
| 4.5 | *Aristida gracilis* | Therophyte | Graminoid | Alien | - | 0.5 | - | 0.3 | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | 0.1 | - | 0.4 | - | - | - | - |
| 4.2 | *Lotus corniculatus* | Hemicryptophyte | Forb | Native | - | - | - | - | - | - | - | - | - | 0.1 | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - |
| 3.8 | *Crataegus monogyna* | Phanerophyte | Forb | Native | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | 0.3 | - | - | - | - | - | - | - |
| 3.8 | *Gladiolus palustris* | Geophyte | Forb | Native | - | 0.1 | - | - | - | - | - | - | - | 0.1 | - | - | - | 0.3 | - | - | - | - | - | - | - | - | - | - | - | - | - | - |
| 3.8 | *Inula hirta* | Hemicryptophyte | Forb | Native | - | - | - | - | - | - | - | - | 0.1 | - | - | - | - | - | 0.3 | - | - | - | - | - | - | - | - | - | 0.3 | 0.1 | - | - |
| 3.1 | *Salix caprea* | Phanerophyte | Forb | Native | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - |
| 2.8 | *Juncus conglomeratus* | Hemicryptophyte | Graminoid | Native | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | 0.1 | - | - | - | - | - | - | - | - | - |
| 2.1 | *Leontodon hispidus* | Hemicryptophyte | Forb | Native | - | - | - | - | - | - | - | - | 0.4 | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | 0.8 | 0.3 | - | 0.8 |
| 1.7 | *Carex acutiformis* | Geophyte | Graminoid | Native | - | - | - | - | - | - | - | - | 0.5 | - | - | - | - | - | - | - | - | - | - | - | 0.5 | - | - | 1.5 | - | - | - | - |
| 1.7 | *Galium verum* | Therophyte | Forb | Native | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - |
| 1.7 | *Hieracium umbellatum* | Hemicryptophyte | Forb | Native | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | 0.1 | - | - | - | - | - | - |
| 1.7 | *Prunella grandiflora* | Hemicryptophyte | Forb | Native | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - |
| 1.7 | *Viburnum opulus* | Phanerophyte | Forb | Native | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - |
| 1.4 | *Carex pallescens* | Hemicryptophyte | Graminoid | Native | - | - | - | - | 0.3 | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | 0.3 | - | - | - |
| 1.4 | *Corylus avellana* | Phanerophyte | Forb | Native | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - |
| 1.4 | *Eleocharis carniolica* | Hemicryptophyte | Graminoid | Native | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | 0.3 | - | - | - | - | - | - | - | - | - | - |
| 1.4 | *Euphorbia flavicoma* | Chamaephyte | Forb | Native | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - |
| 1.4 | *Gratiola officinalis* | Hemicryptophyte | Forb | Native | 1.0 | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | 1.0 | - | - | - |
| 1.4 | *Vincetoxicum hirundinaria* | Hemicryptophyte | Forb | Native | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - |
| 1.0 | *Juncus effusus* | Hemicryptophyte | Graminoid | Native | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | 0.1 | - | - | - | - | - | - | - | - | - | - |
| 1.0 | *Lysimachia vulgaris* | Hemicryptophyte | Forb | Native | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | 0.8 | - | - | - | - | - | - | - | 0.1 | - | - |
| 0.7 | *Carex caryophyllea* | Hemicryptophyte | Graminoid | Native | 0.3 | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | 0.1 | - | - | - | - | - | - | - |
| 0.7 | *Fraxinus excelsior* | Phanerophyte | Forb | Native | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - |
| 0.7 | *Galium lucidum* | Hemicryptophyte | Forb | Native | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - |
| 0.7 | *Hypochoeris radicata* | Hemicryptophyte | Forb | Native | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - |
| 0.7 | *Juncus bulbosus* | Hydrophyte | Graminoid | Native | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - |
| 0.7 | *Juncus tenuis* | Hemicryptophyte | Graminoid | Alien | - | - | 0.1 | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - |
| 0.7 | *Polygala vulgaris* | Hemicryptophyte | Forb | Native | - | - | - | - | - | - | - | - | 0.3 | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | 0.5 | - | - | - |
| 0.7 | *Solidago gigantea* | Hemicryptophyte | Forb | Alien | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - |
| 0.7 | *Thymus serpyllum* s.l. | Chamaephyte | Forb | Native | - | - | - | - | - | - | - | - | 0.1 | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | 0.3 | - | - | - |
| 0.3 | *Centaurea bracteata* | Hemicryptophyte | Forb | Native | - | - | 0.3 | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - |
| 0.3 | *Fraxinus ornus* | Phanerophyte | Forb | Native | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - |
| 0.3 | *Holcus lanatus* | Hemicryptophyte | Graminoid | Native | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - |
| 0.3 | *Hypericum perforatum* | Hemicryptophyte | Forb | Native | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - |
| 0.3 | *Lythrum salicaria* | Hemicryptophyte | Forb | Native | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | 0.1 | - | - | - | - | - |
| 0.3 | *Quercus rubra* | Phanerophyte | Forb | Alien | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - |
| 0.3 | *Viola riviniana* | Hemicryptophyte | Forb | Native | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - | - |

## 9. Highlights

1. Six restoration treatments were applied in southern *Calluna vulgaris* heathlands
2. Annual fire shifted the vegetation towards a *Molinia arundinacea*-dominated community
3. In the short-term, fire treatments triggered a peak in alien species cover
4. Six years after treatments, plant diversity did not differ between treated and control sites
5. Six years after treatments, goat browsing and annual mowing provided the best results