Inventory, value and restoration of peatlands and mires:

recent contributions

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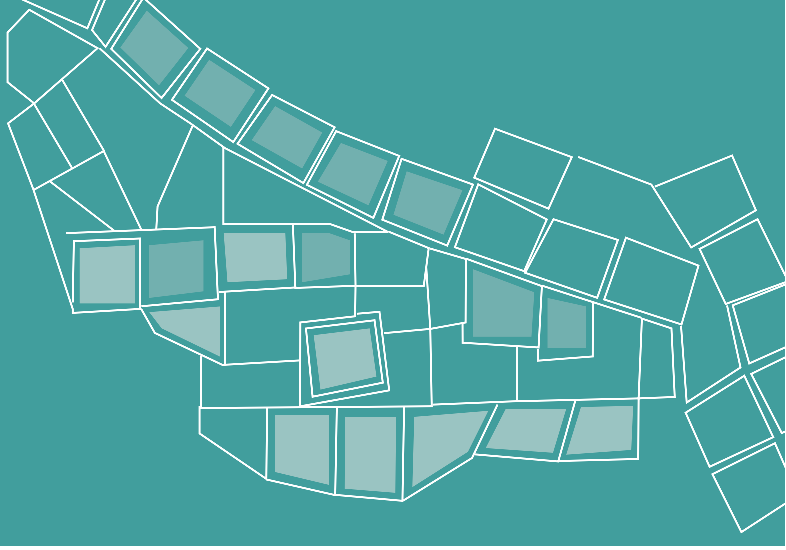

Drone photograph of the Zalama blanket bog (Bizkaia), where the cells protected by coconut fiber mesh can be seen. This helps to stabilize slopes exposed to erosion, limit the loss of peat and restore the vegetation (University of Nottingham Trent).

Cover: *Eriophorum vaginatum* (Sergio González Ahedo)

INVENTORY, VALUE AND RESTORATION OF PEATLANDS AND MIRES: RECENT CONTRIBUTIONS

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Foreword

Peatlands are among the most threatened habitats, both at global and European scales. In our continent, 60-70% of the surface of peat bogs before the 18th century is estimated to have been lost, as a consequence of an increase in the demand for new exploitable lands. The perception of peatlands, previously considered worthless and unhealthy, changed drastically, and the rate of destruction has been important since then, linked to the use of peat as fuel or to the drainage to gain soil for occupation by agriculture or forestry, especially in countries where peat bogs form landscapes. But even in Southern Europe, where the prevailing climatic and biogeographical conditions are less favourable, peat bogs and other related systems have been reclaimed and transformed.

Despite its desolate appearance to the uninformed visitor, the peat bogs that have survived harbor a disproportionately greater level of biodiversity than expected, in accordance to their extension compared with other terrestrial habitats. These ecosystems allow specialized and unique evolutionary dynamics, generated in a delicate and complex balance between elements and processes, physical and biological. But in addition, they provide ecosystem services that society is just beginning to be aware of and value, such as those linked to the storage, control and purification of water.

In the Northern Iberian Peninsula, this deterioration of peatlands across centuries is reproduced, perhaps aggravated by late inventory and scientific knowledge, which has probably prevented the implementation of protection policies. The peat bog of Zalama, in the Eastern limits of the Cantabrian Mountain range, is a paradigmatic example, since until the 1990's it did not receive the attention deserved to show up its naturalistic value, first, and to propose measures that would guarantee its conservation, later. Fortunately, the accumulation of data and the dissemination of technical information enabled its recognition and action, in a proper alliance between specialists and managers. The mitigation of pressures and the application of ecological restoration techniques at this site have been promoted by the Provincial Council of Biscay in its role as public administration, but it is true that without quality technical information, it would probably have proved difficult to make decisions, to make them correctly and to obtain resources.

In this context, LIFE + Ordunte Sostenible wanted to disseminate this type of knowledge, in the idea -credited by experience- that making these habitats visible will help to raise their profile in the political agenda. Throughout twelve chapters, recognized experts present their contributions regarding the identification, descrip-

tion, restoration and protection of peat bogs. It is fair to thank them for their participation in this volume, which also owes to the trust and exchange networks that have been created by other thematic initiatives, such as LIFE + Tremedal. As a consequence, the positive effects of the LIFE program of the European Union goes beyond particular projects, by fostering links between people and institutions, capable of mobilizing the information and putting it at the service of conservation.

The LIFE + Sustainable Ordunte team

Author: Hazi Foundation


Life


Life

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Vegetation diversity and conservation of European mires

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Abstract

European mires, including bogs and fens, are among the most threatened habitats in Europe. The distribution of these habitats have decreased in Europe more than in any other continent, especially in regions of Central and Southern Europe where they are rare and represent important refugia for flora and fauna. As a tool for describing and protecting European mires, this chapter reviews the classification of vegetation types and related habitat types in Europe. Floristic classification is based on the differentiation of ombrotrophic bogs and minerotrophic fens, which are described by four and eleven phytosociological alliances, respectively. The crosswalks between these vegetation types and the habitat types defined by EUNIS classification and the Habitats Directive legislation provides a reference for interpreting mire ecosystems. These crosswalks are however difficult to generalize at the continental scale, since in many cases the habitat concepts are broader or narrower than the phytosociological alliances. Despite this limitation, vegetation diversity is crucial for understanding mire habitats, and the development of large databases is providing a new framework to define diversity patterns across vegetation types and geographic regions. Nevertheless, effective conservation of mires can be only based on local actions, and the final interpretation of habitats must be taken at regional or local scales. To this aim, this chapter introduces a framework to improve communication between scientists, technicians and policy makers involved in the conservation of European mires.

Keywords: mires; fen; bog; peat; classification of habitats; Habitats Directive; EUNIS.

Introduction

Mires, including bogs and fens, are worldwide distributed wetlands that support a particular combination of flora, fauna, fungi and microorganisms. These wetland habitats are well differentiated by the dominance of mosses and sedges, among others specialists adapted to extreme abiotic conditions (Wheeler & Proctor, 2000). The most evident adaptations of plants for living in mires are related to the accumulation of water in *Sphagnum* species or the assimilation of insects in carnivorous plants (e.g. *Drosera*). However, most plants living in these ecosystems also have physiological adaptations to nutrient-limitation, water saturation and deficiency in oxygen (Joosten & Clarke, 2000).

Bogs and fens have been present in Europe at least since the Last Glacial Maximum, when they were mainly distributed in Western and Central Europe (Janská *et al.,* 2017). In the post-glacial period, mires colonized deglaciated areas of northern Europe and were confined to relatively small areas in Western, Central and Southern Europe. In the middle Holocene (6,000 years BP), i.e. before the explosion of human colonization and the Anthropocene, their distribution reached regions with high atmospheric humidity or sufficient precipitation to support permanent waterlogged soils. The oldest written documents referring to these habitats were provided by the romans Pliny the Elder and Tacitus, when describing large extensions of mires in sacred areas of Scandinavians. Many European toponyms are rich in names related to mires or swamps, like the Gaelic *bogach* (origin of -bog), the Proto-Germanic *fanjam* (-fen), the Old Nord *kjarr* (-carr), the old Polavian slavic *berl-/birl-* (that gave the name to the city of Berlin), the French *tourbe* (-torbiere, -turbera) or the Latin *tremere* (-tremedal).

In recent times, the distribution of mire habitats decreased in Europe more than in any other continent, with about 60% of bog and fen areas lost (Joosten, 1997). In northern Europe, bogs have been mainly affected by agriculture, forestry and peat extraction (Moen *et al.,* 2002; Vasander *et al.,* 2003). In European mountains, human-induced global change has been the main cause of local species extinctions (Topić & Stančić, 2006; Bergamini *et al.,* 2009). European mires are therefore priority habitats for conservation (Raeymaekers, 2000). In the last decades, there has been a huge effort in mire conservation, with a special role played by the International Mire Conservation Group (http://www.imcg.net/) and also by the scientific community (Van Diggelen *et al.,* 2006). Although much effort has been done to preserve mires, we still need to transform European policies into clear conservation actions at regional and local scales.

The classification of mire habitats plays a central role for their conservation since it provides a tool for communication between scientists and policy makers. Indeed, the botanical literature has been the main reference for classifying mire habitats (Ellenberg, 1988; Hajek *et al.,* 2006) and the basis for the description of European habitats. By considering recent updates in vegetation and habitat classifications, this chapter presents an outline of the classification of bogs and fens in Europe, and the major guidelines to relate major vegetation types to the classification of EUNIS and the Habitats Directive. Rather than providing a deep review of these classifications, the main objective is to provide a guide to further application of these definitions at different spatial scales.

Floristic classification of European mires

Using water and mineral supply as the main criteria, the differentiation between *ombrotrophic* bogs and *minerotrophic* fens is widely accepted in Europe (Ellenberg, 1988; Wheeler & Proctor, 2000; Hajek *et al.,* 2006). Bogs are ombrotrophic systems because the main hydrological resources are mainly (though not only) provided by atmospheric water condensation. In contrast, fens are minerotrophic systems because they depend almost exclusively of above-ground or below-ground waterflows with a certain mineralization. Bogs and fens are deficient in nitrogen and phosphorous, and in both cases water saturation and lack of oxygen makes it difficult the decomposition of organic matter. This favors the accumulation of peat, especially in *Sphagnum*-dominated bogs, which are relatively homogeneous is plant species diversity. Fens are however highly variable in species composition of both vascular plants and bryophytes, depending on soil properties, water regime and temperature, while peat accumulation is generally low or can be absent.

These differences in ecological properties and taxonomic diversity has been the basis for establishing two major phytosociological classes in Europe that largely correspond to ombrotrophic bogs and minerotrophic fens: *Oxycocco-Sphagnetea* (bogs) and *Scheuchero palustris-Caricetea fuscae* (fens). In a recent conspectus of European vegetation, Mucina *et al.* (2016) synthetized the diversity of plant communities from bogs and fens in a total of 16 phytosociological alliances (Table 1). It is important to remark that the classification of mires using only their floristic component is far from perfect, especially when topogenic or dynamic aspects are involved (Tuittila *et al.*, 2007). However, vegetation typologies offer a useful and relatively consistent classification based on the response of species diversity to environmental factors. This classification is also a good starting point for future research. For example, the European Mire Vegetation database (Peterka *et al.*, 2015) aims at storing all European vegetation surveys related to bogs and fens, in order to develop empirical studies at the continental scale (Fig. 1).

FLORISTIC CLASSIFICATION	EUNIS	HABITATS DIRECTIVE (92/43/EEC)
Ombrotrophic bogs		
GROUP 1		
Oxycocco-Empetrion hermaphroditii (Boreo-arctic dwarf-shrub and peat-moss raised bog vegetation)	D1.1 Raised bogs D3.1 Palsa mire D3.2 Aapa mire	7110* Active raised bogs 7120 Degraded raised bogs 7320 Palsa mires 7310 Aapa mires
Sphagnion medii (Dwarf-shrub and peat-moss vegetation of subcontinental and mountain raised bogs)	D1.1 Raised bogs D3.2 Aapa mire	7110* Active raised bogs 7120 Degraded raised bogs 7310 Aapa mires
GROUP 2		
Ericion tetralicis (Dwarf-shrub, sedge and peat-moss vegetation of moist peaty heath)	D1.2 Blanket bogs	7130 Blanket bogs
Oxycocco-Ericion tetralicis (Sedge and peat-moss vegetation of oligotrophic bogs)	D1.2 Blanket bogs D1.1 Raised bogs D2.1 Valley mires	7130 Blanket bogs 7110* Active raised bogs 7120 Degraded raised bogs

FLORISTIC CLASSIFICATION	EUNIS	HABITATS DIRECTIVE (92/43/EEC)
Minerotrophic fens		
GROUP 3		
Anagallido tenellae-Juncion bulbosi (Ibero-Atlantic moderately-rich fens)	D2.1 Valley mires D2.2 Fens pobres	None None
Depranocalion exannulati (Arcto-alpine intermediate non-calcareous fens)	D2.2 Poor fens	None
Caricion fuscae (Moderately to low calcium-rich slightly acidic sedge-moss fens)	D2.2 Poor fens D2.3 Transition mires D3.1 Palsa mire D3.2 Aapa mire	None 7140 Transition mires 7320 Palsa mires 7310 Aapa mires
Sphagno-caricion canescentis (Peat-moss acidic poor yet minerotrophic fens)	D2.2 Poor fens D2.3 Transition mires	None 7140 Transition mires
GROUP 4		
Scheuchzerion palustris (Bog-hollow vegetation)	D2.3 Transition mires	7150 *Rhynchosporion*
GROUP 5		
Saxifrago-Tomentypnion (Boreo-arctic sedge and brown-moss moderately calcareous fens)	D4.1 Rich fens	7230 Alkaline fens
Caricion stantis (Arctic brown-moss rich fens)	D4.2 Basic mountain flushes and stream-sides	7240 Alpine pioneer formations
Stygio-Caricion limosae (Extremely water-logged brown-moss neutral fens)	D2.3 Transition mires	7140 Transition mires
Sphagno-Tomentynion nitentis (Calcium-rich sedge-moss fens)	D4.1 Rich fens D3.1 Palsa mire D3.2 Aapa mire	7230 Alkaline fens 7320 Palsa mires 7310 Aapa mires
GROUP 6		
Caricion viridulo-trinervis (Low-sedge vegetation of subsaline dune slack fens)	D4.1 Rich fens B1.8 Moist and wet dune slacks	7230 Alkaline fens 2190 Humid dune slacks
Caricion davallianae (Sedge-moss calca-reous mineral-rich vegetation)	D4.1 Rich fens D5.2 Beds of large sedges	7230 Alkaline fens 7210 Calcareous fens with *Cladium mariscus*
Caricion atrofusco saxatilis (Low-sedge and low-productivity alpine calcareous fens)	D4.2 Basic mountain flushes and stream-sides D3.1 Palsa mire D3.2 Aapa mire	7240 Alpine pioneer formation 7320 Palsa mires 7310 Aapa mires

Table 1. Phytosociological alliances recognized for European mires by Mucina *et al.* (2016) and crosslinks with EUNIS habitat classification and the codes of the Habitat Directive 92/45/. Crosslinks are adapted from European documentation (www.eunis.eu) and the Red List of European habitats (Jensen *et al.* 2016). Since the crosslinks are not unequivocal, this list should be considered as a large-scale reference but re-defined at regional or local scale for a proper interpretation of European habitats. Groups correspond to phytosociological orders and as detailed in the main text.

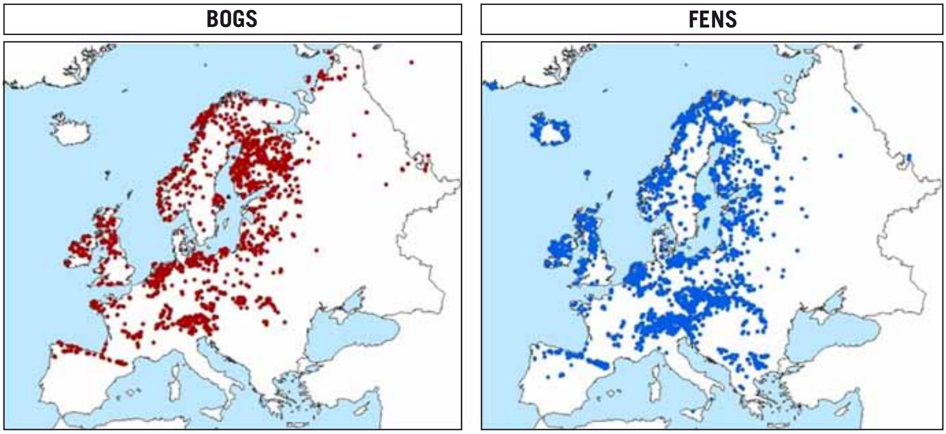

Figure 1. Distribution of vegetation plot data referred to bogs and fens in Europe, as compiled by the European Mire Vegetation Database (Peterka *et al.* 2016).

Major vegetation types in bogs

The vegetation of ombrotrophic bogs is mainly dominated by dwarf-shrubs, sedges and peat mosses; and it is classified in two major groups (phytosociological orders) and four alliances (Table 1). In general, the dominance of *Sphagnum* species and the lack of mineral-demanding species is the main character to disentangle their species composition from fens. Bogs are also characterized by extremely low values of soil and water pH (< 4) and the accumulation of big quantities of peat, sometimes several meters deep.

A first group of alliances (group 1, order *Sphagnetalia medii*) refers to bogs dominated by dwarf-shrubs and peat mosses influenced by continental or subcontinental climates. They include bog vegetation from boreal and arctic regions (*Oxycocco-Empetrion hermaphroditii*) characterized by *Sphagnum* species and cold-adapted species such as *Oxycoccus microcarpus*, *Empetrum hermaphroditum*, or *Ledum paluste*; and bogs from subcontinental and mountain regions (*Sphagnion medii*) characterized by *Sphagnum medium* or *Eriophorum vaginatum*. A second group of alliances (group 2, order *Erico-Ledetalia palustris*) is mainly related to wet heaths with atlantic and subatlantic influence, which are dominated by peat mosses and either dwarf-shrubs (*Ericion tetralicis*) or sedges (*Oxycocco-Ericion tetralicis*). In both cases, *Erica tetralix* is a characteristic species forming tussocks in combination with *Trichophorum caespitosum* or *Narteccium ossifragum*. In these mires is also common to find heath species with a physiological tolerance to poor and wet soils, like *Calluna vulgaris*. In fact, bogs usually form vegetation mosaics with wet heathland vegetation in the European Atlantic biogeographic region.

Major vegetation types in fens

The vegetation of minerotrophic fens is classified in 12 alliances (Table 1) corresponding to four major groups (phytosociological orders). The diagnostic species and geographic distribution of these vegetation types have been recently characterized by Peterka *et al.* (2016). This study supports, at the continental scale, the existence of a major gradient of pH and mineral richness (the so-called "poor-rich gradient") to explain the patterns in species composition in European fens, as it has been shown in different regions (Críıodáin & Doyle, 1997; Tahvanainen, 2004). In addition, a complementary influence of local eutrophication related to mowing or grazing determines species diversity in fens (Moen *et al.,* 2002; Hájek *et al.,* 2006).

At the poorest side of the poor-rich gradient, a group of alliances (group 3, *Caricetalia fuscae*) can resemble bog systems since they are partially dominated by *Sphagnum*; however they have a clear minerotrophic character indicated by the presence of mineral-demanding species. This group includes oceanic vegetation from the Atlantic region (*Anagallido tenellae-Juncion bulbosi*), species-poor communities from arctic regions dominated by mosses and sedges (*Depranocladion exannulati*); slightly acidic sedge-moss fens widespread in the temperate zone (*Caricion fuscae*), and *Sphagnum*-dominated acidic fens from the temperate and boreal zones (*Sphagno-Caricion canescentis*). Another group (group 4, *Scheuchzeretalia palustris*) is formed by extremely poor vegetation occupying water hollows in Central Europe and especially in flat landscapes of Northern europe (*Scheuchzerion palustris*), in many cases co-occurring with bog communities in mire vegetation complexes.

Other minerotrophic fens are characterized by brown mosses and they occupy the gradient from moderately-rich to extremely rich and calcareous soils. A first group (group 5, *Sphagno wandorsfi-Tomentynetalia*) include also calcium-tolerant *Sphagnum* species and occurs mainly in the Arctic zone and Central Europe. This group is formed by arctic-alpine moderately calcareous fens (*Saxifrago-Tomentypnion*); rich fens from Svalvard (*Caricion stantis*); and water-logged neutral fens (*Stygio-Caricion limosae*) and calcium rich fens (*Sphagno-Tomentypnion nitentis*) from boreal and temperate regions. At the richest side of the minerotrophic gradient, the last group (group 6, *Caricetalia davallianae*) is characterized by extremely mineral-rich soils, generally on calcareous bedrocks, dominanted by small sedges and brown mosses but with complete lack of *Sphagnum*.

This group includes subsaline fen vegetation from Atlantic dune slacks (*Caricion viridulo-trinervis*); widespread small-sedge vegetation from temperate Europe (*Caricion davallianae*); and an arctic-alpine variation in very low productivity soils (*Caricion atrofusco-Saxatilis*). Mucina *et al.* (2016) also recognizes three relict vegetation alliances from Southern European Mountains with an impoverished species composition, in Sierra Nevada (*Festucion frigidae*), Corsica (*Caricion intricatae*) and the Balkans (*Narthecion scardici*).

Habitat types in EUNIS and the Habitats Directive

The classification of European habitats has been largely based on the phytosociological descriptions at the level of alliance. First attempts were provided by the CORINE biotope manual (Devillers *et al.,* 1991) and the Paleartic classification (Devillers & Devillers-Terschuren, 1996). As a further development of these initiatives, EUNIS was created to provide a more clear criteria for disentangling habitat groups using a hierarchical framework and crosswalks with other classification systems (Rodwell *et al.,* 2002; Davies *et al.,* 2004). However, the correspondence between habitat types and vegetation types is not unequivocal, since for many years it was necessary to accommodate regional and national interpretations in the absence of a consistent vegetation classification.

In the last years, the European Environmental Agency (EEA) made an effort to standardize the floristic description of European alliances by using the data stored in the European Vegetation Archive (Chytrý *et al.,* 2016). Although mire vegetation has not been yet included in this review, a complementary effort to define bogs and fens can be extracted from the Red List of European Habitats (Jansen *et al.,* 2016). According to this List, the European mires with the highest threat according to IUCN criteria are raised bogs (D1.1, categorized as *Endangered*), Palsa mires (D3.1, *Critically Endangered*) and calcareous fens (D4.1, *Endangered*, but excluding quanking mires). The rest of mires are also threatened as *Vulnerable*, with the only exception of blanket bogs (D1.2) and Aapa mires (D3.2).

Red lists are useful for understanding the risk of extinction and collapse in species and ecosystems, respectively (Keith *et al.,* 2015), but they have not legal implications. Nowadays, the Habitats Directive 92/43/EEC is the only legal instrument supporting the protection of habitats and species in Europe through the Natura 2000 Network (Evans, 2010). The descriptions provided by the Habitats Directive for the so-called *Habitat types of Community Interest* are in most cases based on the phytosociological literature, but they don't correspond unequivocally with either phytosociological alliances or EUNIS types. For the countries of the Emerald network (which are expected to join Natura 2000), the Council of Europe uses EUNIS as the basis to interpret the habitats to be protected (Council of Europe, 2010), supporting the generalization of EUNIS as the reference list for habitat classification.

As a general reference for the interpretation of European mires, but assuming the limitations of using widely distributed alliances subject to regional or local variation, Table 1 shows the codes of EUNIS and the Habitats Directive that better correspond with the vegetation alliances. It is important to note that these crosswalks are difficult to establish at the continental scale, since in many cases the habitats concepts are broader or narrower than the phytosociological alliances. When this happens, the final interpretation of habitats must be taken at the regional or local scale using the detailed documentation provided in www.eunis.eu.

Correspondences between bog vegetation and European habitats

In general, the identification of ombrotrophic bogs by EUNIS and the Habitats Directive is congruent with the phytosociological system. However, bogs are complex systems that may include communities of minerotrophic fens in heterogeneous patches with water accumulation. This makes it difficult to recognize a unique crosswalk between alliances and habitat types. Nevertheless, the most typical bogs that dominate the landscape in many hemi-boreal regions of Europe (group 1 in Table 1) are generally well characterized by "raised bogs" in the description of both EUNIS and the Habitats Directive, while the oceanic and suboceanic bogs (group 2) usually correspond to "blanket bogs". However, vegetation types from subcontinental regions can be identified in both groups raised and blanket bogs.

As a rule of thumb, the majority of continental European bogs correspond to raised bogs, while those developed in gently slopes and in oceanic regions with high-precipitation (typically, but not only, in the British Islands and nearby Atlantic regions) should be considered as blanket bogs. The Habitats Directive also recognizes raised bogs that are either active or degraded which needs an evaluation of the bog development in each specific site. An additional problem for establishing direct links with phytosociological alliances is the definition of Palsa mires and Aapa mires from norther Europe (Laitinen *et al.,* 2006), which are large complex systems where different vegetation types occur, and in many cases they cannot be related to a dominant type of bog or fen.

Correspondences between fen vegetation and European habitats

In most cases, and with the exception of Palsa and Aapa mire complexes, a good description of vegetation alliances should be enough to interpret the habitats of EUNIS or the Habitats Directive. This is especially true in the crosswalks for the neutral, rich and calcareous fens (groups 4-6 in Table 1), but many poor-fen communities can be linked to various habitats. It should be noted that most poor fens do not correspond to any type of the Habitats Directive and therefore they are not protected under the EU regulation, simply because they were not considered as a conservation priority at the European scale. Further conservation legislation should be needed, when necessary, at the national or regional level.

A recurrent problem of interpretation is the habitat *Rhynchosporion*, which refers to water holes commonly found in both bogs and mires. According to the new syntaxonomical revision of Mucina *et al.* (2016) this vegetation is partially merged with the alliance *Scheuchzerion palustris* (group 4) as the result of a historical misconception of the former alliance by different authors. Similarly, transition mires are referred in EUNIS as floating mires that contain *Rhynchosporion* habitat (7150), but also the so-called Transition mires (7140). Again, the accommodation of these habitats to the European types should be based on a careful interpretation of each site and the main principles defined in the EU documentation.

European mire conservation: think globally, act locally

One interesting property of mires is that, despite the high specialization of vascular plants and bryophytes to these habitats, there is a general lack of narrow endemic plant species. Although most of European mire species only occur in mires, they are widespread at large scales and present a cosmopolite distribution. In contrast with other habitats with abundance of endemic plants (e.g. alpine grasslands), the differentiation of mire habitats is difficult to assess without the guide of species that only occur in particular regions. This is however mainly referred to vascular plants and bryophytes, whereas other organisms can be much more specific (Mitchell *et al.,* 2000). When using botanical descriptions, the combination of species can be linked to ecological conditions and related geographic variation, allowing us to differentiate a relative good number of vegetation types and habitats when good data is available. Indeed, the compilation of vegetation databases at the continental scale is increasing our understanding of regional variation, as a basis to set a global assessment of the diversity and distribution of mires in Europe (Jiménez-Alfaro *et al.,* 2014; Peterka *et al.,* 2016).

Effective mire conservation can be only based on local action. Since the first documents concerned with the conservation of mires in Europe (Joosten, 1978), there has been many advances in understanding the main factors influencing mire biodiversity and conservation. In northern Europe, peat extraction needs proper regulation and adequate policies for habitat restauration (Raeymaekers, 2000). In Eastern Europe, landscape conservation must integrate the preservation of relict bogs and fens in clear risk of collapse (Hajek *et al.,* 2010). In the mountains of Southern Europe, very small areas are covered by mires that were more abundant before, providing unique examples of mire species going into extinction (Jimenez-Alfaro *et al.,* 2016).

To address all these problems, this book presents a compilation of different case studies on the characterization and restoration of bogs and fens, supported by national and regional agencies. Many of these examples have been developed in the context of the European LIFE conservation program (e.s. http://www.lifetremedal.eu/), giving rise to some optimism about the future of mires at the continental scale. Despite the difficulties of understanding, interpreting and preserving mires, these projects are examples of a successful communication between scientists, technicians and policy makers, towards the preservation of biodiversity and the cultural heritage of European mire habitats.

References

Bergamini, A., Peintinger, M., Fakheran, S., Moradi, H., Schmid, B., Joshi, J. 2009. Loss of habitat specialists despite conservation management in fen remnants 1995–2006. *Perspectives in Plant Ecology, Evolution and Systematics* 11: 65–79.

Council of Europe. 2010. Revised annex I of Resolution 4 (1996) of the Bern Convention on endangered natural habitats types using the EUNIS habitat classification. TPVS/PA(2010)10revE, Council of Europe, Strasbourg.https://wcd.coe.int/com.instranet.InstraServlet?command=com.instranet.CmdBlobGet&InstranetImage=1763389&SecMode=1&DocId=1648180&Usage=2

Chytrý M., Hennekens S.M., Jiménez-Alfaro B., Knollová I., Dengler J., Jansen F., Landucci F. 2016. European Vegetation archive (EVA): an integrated database of European vegetation plots. *Applied Vegetation Science* 19: 173-180.

Críiodáin, C.O., Doyle, G.J. 1997. *Schoenetum nigricantis*, the *Schoenus* fen and flush vegetation of Ireland. *Biology and Environment* 97: 203–218.

Davies, C.E., Moss, D., Hill, M.O. 2004. *Eunis Habitat Classification Revised* 2004. http://eunis.eea.europa.eu/upload/EUNIS_2004_report.pdf.

Devillers, P., Devillers-Terschuren, J. 1996. *A classification of Palaearctic habitats*. Council of Europe, Strasbourg: Nature and environment, No 78.

Devillers, P., Devillers-Terschuren, J., Ledant, J.P. 1991. *CORINE biotopes manual. Vol. 2. Habitats of the European Community*. Office for Official Publications of the European Communities, Luxembourg.

Ellenberg, H. 1988. *Vegetation Ecology of Central Europe*. Cambridge Univ. Press, Cambridge.

Evans, D. 2010. Interpreting the habitats of Annex I – Past, present and future. *Acta Botanica Gallica* 157: 677- 686.

Evans, D. 2012. The EUNIS habitats classification-past, present & future. *Revista de Investigación Marina*, 19: 28-29.

Fernández-Pascual, E. 2016. Comparative seed germination traits in bog and fen mire wetlands. *Aquatic Botany* 130: 21-26.

Fernández-Pascual, E., Jiménez-Alfaro, B., Hájek, M., Díaz González, T.E., Pritchard, H. 2015. Soil thermal buffer and regeneration niche may favour calcareous fen resilience to climate change. *Folia Geobotanica* 50: 293-301.

Hájek, M., Hájkova, P., Apostolova, I., Horsák, M., Rozbrojova, Z. 2010. The insecure future of Bulgarian refugial mires: economic progress versus Natura 2000. *Oryx* 44: 539-546.

Hájek, M., Horsák, M., Hájková, P., Dítě, D. 2006. Habitat diversity of central European fens in relation to environmental gradients and an effort to standardise fen terminology in ecological studies. *Perspectives in Plant Ecology, Evolution and Systematics* 8: 97–114.

Janská, V., Jiménez-Alfaro, B., Chytrý, M., Divíšek, J., Anenkhonov, O., Korolyuk, A., Lashchinskyi, N., Culek, M. 2017. Palaeodistribution modelling of European vegetation types at the Last Glacial Maximum using modern analogues from Siberia: prospects and limitations. *Quaternary Science Reviews* 159: 103-115.

Janssen, J.A.M., Rodwell, J.S., García Criado, M., Gubbay, S., Haynes, T., Nieto, A., Sanders, N. 2016. *European Red List of Habitats. Part 2. Terrestrial and freshwater habitats.* Publications Office of the European Union. Luxembourg.

Jiménez-Alfaro, B., Hájek, M., Ejrnaes, R., Rodwell, J., Pawlikowski, P., Moen, A., Weeda, E. 2014. Biogeographic patterns of base-rich fen vegetation across Europe. *Applied Vegetation Science* 17: 367-380.

Jiménez-Alfaro, B., Garcia-Calvo, L., Garcia, P., Acebes, J.L. 2016. Anticipating extinctions of glacial relict populations in mountain refugia. *Biological Conservation* 201: 243-251.

Joosten, H. 1997. *Mires in Europe: a preliminary status report*. International Mire Conservation Group Members No. 3.

Joosten, H., Clarke, T. 2003. *Wise use or mires and peatlands*. International Mire Conservation Group and International Peat Society. Saarijärven Offset Oy, Saarijärvi, Finland.

Keith D. A., Rodríguez J. P., Brooks T. M., Burgman M. A., Barrow E. G., Bland L., Comer P. J. 2015. The IUCN red list of ecosystems: Motivations, challenges and applications. *Conservation Letters* 8: 214-226.

Laitinen, J., Kukko-Oja, K., Huttunen, A. 2008. Stability of the water regime forms a vegetation gradient in minerotrophic mire expanse vegetation of a boreal aapa mire. *Annales Botanici Fennici* 45: 342–358.

Mitchell, E.A.D., Buttler, A., Grosvernier, P., Rydin, H., Albinsson, C., Greenup, A.L., Heijmans, M.M.P.D., Hoosbeek, M.R., Saarinen, T. 2000. Relationships among testate amoebae (Protozoa), vegetation and water chemistry in five Sphagnum-dominated peatlands in Europe. *New Phytologist* 145: 95-106.

Moen, A., Lyngstad, A., Øien, D.-I. 2012. Boreal rich fen vegetation formerly used for haymaking. *Nordic Journal of Botany* 30: 226–240.

Mucina, L., Bültmann, H ., Dierssen, K. , Theurillat, J.-P. , Raus, T., Čarni, A. , Šumberová, K. 2016. Vegetation of Europe: hierarchical floristic classification system of vascular plant, bryophyte, lichen, and algal communities. *Applied Vegetation Science* 19, Suppl. 1: 3–264.

Peterka, T., Jiroušek, M., Hájek, M., Jiménez-Alfaro, B. 2015. European Mire Vegetation database: a gap-oriented database for European fens and bogs. *Phytocoenologia* 45: 291-297.

Peterka, T., Hájek, M., Jiroušek, M., Jiménez-Alfaro, B., Aunina, L., Bergamini, A., Dítě, D. 2017. Formalized classification of European fen vegetation at the alliance level. *Applied Vegetation Science* 20: 124-142.

Raeymaekers, G. 2000. *Conserving mires in the European Union. Actions co-financed by LIFE-Nature Ecosystems* Ltd. Luxembourg : Office for Official Publications of the European Communities.

Rodwell, J.S., Schamineé, J.H.J., Mucina, L., Pignatti, S., Dring, J., Moss, D. 2002. *The Diversity of European Vegetation. An overview of phytosociological alliances and their relationships to EUNIS habitats*. National Reference Centre for Agriculture, Nature and Fisheries, Wageningen.

Tahvanainen, T. 2004. Water chemistry of mires in relation to the poor–rich vegetation gradient and contrasting geochemical zones of the north-eastern Fennoscandian Shield. *Folia Geobotanica* 39: 353–370.

Topić, J., Stančić, Z. 2006. Extinction of fen and bog plants and their habitats in Croatia. *Biodiversity and Conservation* 15: 3371–3381.

Tuittila, E.-S., Väliranta, M., Laine, J., Korhola, A. 2007. Quantifying patterns and controls of mire vegetation succession in a southern boreal bog in Finland using partial ordination. *Journal of Vegetation Science* 18: 891-902.

Vasander, H., Tuittila, E. S., Lode, E., Lundin, L., Ilomets, M., Sallantaus, T., Heikkilä, R., Pitkänen, M. L., Laine, J. 2003. Status and restoration of peatlands in northern Europe. *Wetlands Ecology and Management* 11: 51-63.

Van Diggelen, R., Middleton, B. A., Bakker, J. P., Grootjans, A., Wassen, M. 2006. Fens and floodplains of the temperate zone: Present status, threats, conservation and restoration. *Applied Vegetation Science* 9: 157-162.

Wheeler, B.D. & Proctor M.C.F. 2000. Ecological gradients, subdivisions and terminology of north-west European mires. *Journal of Ecology* 88: 187-203.

02

Overview of the Zalama peat bog (Bizkaia) over the last thirty years

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Abstract

This paper compiles, chronologically, the documents that have acknowledged the value of the Zalama bog and the activities carried out by the Bizkaia Provincial Council (DFB - Diputación Foral de Bizkaia) in this unique zone, after an analysis of the activities that were putting at risk the bog`s very existence. Since 2008, the DFB has executed several actions to recover this priority natural habitat, which have been financed by LIFE+ Sustainable Ordunte. This project has allowed monitoring data to be obtained and synergies with other projects, the results of which, aim to improve the overall management of this zone.

Keywords: blanket bog; Zalama; erosion; Nature 2000; dams; enclosures; geotextile.

Acknowledgement of the value of the Zalama peat bog

According to the information available in the first management document that was drawn up for the current Ordunte Special Area of Conservation –the plan drafted in 2009 on behalf of the Department of Agriculture at the Provincial Council of Bizkaia (DFB) (Basoinsa, 2009)– the first technical references related to the Zalama bog come from the 1980s.

This Floral Catalogue of Álava, Vizcaya & Guipúzcoa (Aseginolaza *et al.* 1984) was the first document that recognized the value of the Zalama bog, owing to the existence of the only known population of *Eriophorum vaginatum* (Fig. 1) in the Basque Country (CAPV). In turn, the discoverer of this unique population, also included the vegetation in the zones around Encartaciones and Gorbea (Onaindia, 1986), in her thesis showing how little was known about this area.

In spite of this, back then the difference between the existing bog on Zalama mountain, the acidophile fens around it and the presence of rare plant species of special interest was still blurred.

In 1989, the bryologist Patxi Heras wrote a detailed study of mosses and liverworts in the three existing bogs between the mountain pass of Los Tornos and the summit of Zalama (Heras, 1990). This study is especially relevant because it was

Figure 1. Unique population of *Eriophorum vaginatum* in the Basque Country, at the Zalama peat bog.

the first time that all the habitats were not grouped together under the umbrella of wet peatlands, but because the peat bogs, with peat deposits, were differentiated from the habitats of the fens. Curiously, neither the geological charts drawn up at that time (Garrote *et al.*, 1992), nor of the vegetation distribution (Aseginolaza *et al.*, 1990) included the presence of this unique zone.

In December 1997, the Montes de Ordunte was proposed by the Basque Government as a Site of Community Importance (SCI) in Natura 2000 within the Atlantic Biogeographic Region, including the blanket bog among the habitats in the zone. Moreover, in the same year the SCI at Ordunte was also included in the "Open List of Areas of Natural Interest" in the Basque Country Land Planning Act, approved in 1997, as a zone that should be taken into consideration during territorial, sectorial and municipal planning with the aim of conserving its ecological, cultural and economic values.

The real environmental evaluation of the bog did not arrive until 2002, when Patxi Heras and Marta Infante drew up a report on behalf of the Water Management Board of the Basque Government. Thanks to the assessment of the bog expert Richard Lindsay (University of East London), it was established that the Zalama bog is blanket type, a fact of special relevance as it is the only one in the CAPV, as well as the most oriental on the Iberian Peninsula and one of the few existing in Southern Europe. This document determined the environmental values of the Zalama bog and brought recognition of it as an ombrotrophic bog, since the supply of water comes exclusively from rainfall, snow and horizontal precipitation from fog and dew. In turn, the paludification or accumulation of peat, can be demonstrated by an examination of the eroded zones, which were on a surface of loose siliceous rocks, in areas

that were previously occupied by the bog and exposed by the erosive processes of fire, wind and water.

However, at the same time that the important environmental value of the bog was revealed, in 2002 the Basque Government also approved the Sectoral Territorial Plan for Wind Energy which stated that "*the urban planning affected by the locations selected will be bound by the decisions taken by this Sectoral Territorial Plan and must be adapted to them in their first revision or modification.*" One of the locations chosen in Bizkaia was the Ordunte range, from Zalama to Kolitza mountain. Although, at the moment, it seems improbable that any plan or project not being of conservation could be executed within this zone, there is no doubt that conservation activities and others that defended this zone helped to recognize its value.

As the environmental values of the bog were being acknowledged, *E. vaginatum* only known in the CAPV from the Zalama bog, was included in the Basque Catalogue of Threatened Species on May 2003. This species is currently in the category "danger of extinction".

On December 2004, the SCI ES2130002 Ordunte was approved by the European Commission, classifying the site as Natura 2000. The Zalama bog was included as a natural habitat of priority interest and mentioned in Annex I as a "type of natural habitat of community interest whose conservation requires the classification as Special Area of Conservation" by the member states under the code 7130 "Blanket bogs". At that moment, it was thought that the bog was active, in spite of the fact that its peat generating capacity had been reduced after the aggression it had suffered.

The Sectoral Territorial Plan for Wetlands of the CAPV, approved by Decree 160/2004, set out the decisions of the Territorial Land Planning Act via an inventory and classification of wetlands, as well as the regulations of uses and activities in agreement with its capacity to cope with them. It also established a series of recommendations and general criteria for the protection of all the wetlands in the inventory. The Ordunte fens were registered in this plan in Group III with the name "Bogs in the Sierra de Ordunte" (B1B3), whilst the bog appears as "Zalama bog" (B1B5).

Moreover, in June 2006, the Provincial Council of Bizkaia approved the management plan for seven threatened species of the flora and fauna, the most noteworthy of these being *E. vaginatum* (DFB Decree 114/2006), since it was the only population of this species in the CAPV and was found in the Zalama bog. Recently, the Basque Government has named the Ordunte SCI as a Special Area of Conservation via Decree 65/2015, whose annex II includes the conservation measures related to the wet and peat-forming habitats and by extension, the Zalama blanket bog.

Causes of the deterioration of the bog

Fires

In the past, the Zalama bog lost surface area because of fires originating on its southern edges. The most important problem stemming from these were before 1965 and even 1956. Observing the aerial photographs from those years, the division between the most active zone of the bog, situated to the east, and the most western part occupied by hummocks or piles of peat without vegetation can be appreciated. The two zones are separated by a central eroded sector covered by grass, and views of the siliceous bedrock that today can be seen alongside a cut in the peat slope.

It is easy to deduce that the fires that destroyed part of the bog came from the south and that they occurred during the dry months between August and October, when the phreatic level of the bog was lower and the advance of the fire could be stoked by the dominant south wind. In August 1987, a fire worsened the deteriorating situation of the bog at the edges of the zones that had previously been affected.

Although it is true that fires have had destructive effect directly on the mass of peat and the vegetation, the most serious damage comes from the loss of water accumulated in the phreatic level during rainy or snowy periods. Drainage routes have been created on open slopes or via exposed peat surfaces lacking vegetation, altering the processes of water accumulation, making the underground store of water more fragile and shorter. One must bear in mind that once the vegetative cover is destroyed, peat is a very difficult substrate for colonization in an exposed location such as the summit of Zalama, with strong southern winds drying the land out.

It is true that peat has the capacity to absorb a large quantity of water in the porous spaces between its particles, that favours germination of seeds of all types, if the heat conditions are favourable. But when the temperature rises, the relative humidity of the air falls and there follows a process of surface drying of the peat which is faster than the speed of root growth of the germinated seeds, the latter die if they do not reach a sufficient depth and humidity within the peat that can guar-

Figure 2. Orthophoto from 1956 on the left and from 2008 on the right.

antee their survival. This is the only explanation for the lack of growth of vegetation on the hummocks, in spite of the period of time involved, 1956 to 2008, the year when the restoration of the bog began (Fig. 2).

Vehicles

The bog was crossed by a track that allowed four-wheel drive vehicles, motorbikes, mountain-bikes and quads to enter the area. This was connected to the forest tracks in the province of Burgos on the southern side of Ordunte Range. The grooves left behind by the wheels intensified the lineal erosion because of the flow of surface water. These grooves got wider and wider year after year as drivers sought to find a smooth surface easier to travel on, so the deterioration of the bog surface grew and grew and driving became more aggressive in rainy periods with the vegetation being removed and then completely disappearing from these areas.

Continuous presence of cattle

The cooler temperatures in the high zone of the *Terreros Negros*, toponym given by the locals to the Zalama bog owing to the morphology of the terrain, attracted cattle, especially in summer. They found water almost all year round when other zones of nearby pasture had dried out. The continuous treading on the slopes caused a constant erosion of the peat which was difficult to repair (Fig. 3).

Continuous erosion

On top of that, once the peat had lost its humidity after constant trampling by cattle, or it was compacted by the passing vehicles, it was transformed into smaller particles that were easily swept away by wind and water. The depth of peat in these zones lacking protection from vegetation got smaller and the eroded slopes grew little by little reducing the active surface of the bog.

Figure 3. Continuous presence of cattle on peat slopes.

Bog recovery actions in LIFE+ Sustainable Ordunte

In these circumstances, it was essential to eliminate all the processes of erosion that were affecting the bog, by carrying out activities to correct the faults.

Enclosure

In 2008, the first measure proposed to achieve bog recovery was an enclosing fence so as to impede the passage of vehicles that had created a track. In this way, a partial restoration of this sector was achieved, after a few years, by natural revegetation (Figs. 4 and 5).

The second positive impact achieved was to limit the erosion on the slopes of the bog, caused by cattle trampling. The third indirect effect that was to reduce the nitrogen input from the cattle which stayed for hours in the bog, especially in summer.

In this way, oligotrophic conditions were maintained, these favouring species adapted to nutrient-poor bogs and not others that demand greater nitrogen inputs. Accordingly, within the activities in LIFE+ Sustainable Ordunte, reinforcement work to the first wooden enclosure was carried out in 2013, and also the installation of a second enclosure on the lowest section, so as to avoid cattle rubbing and the deterioration of the fence in this sector, consolidating the efforts carried out in previous years.

Protection of the eroded zones on the slopes with coconut fibre matting

In 2010 the installation of a protective mat, made from coconut fibre, began on the exposed surfaces without vegetation on the slopes limiting the central active part of the bog so as to control the loss of water through run-off via the exposed slopes and through evaporation, as these surfaces are swept by winds from the south-east. With

Figures 4 & 5. Photograph on the left shows the area before enclosure, in 2007; and the one on the right taken in 2010, after enclosure.

the aim of giving a greater stability to this artificial covering of coconut fibre matting, it was fixed to the ground with sleepers made of Lawson trunks, creating squares 2 x 2 m, anchored with irons to the ground to prevent especially strong autumn winds from lifting the protection (Fig. 6).

In 2013, LIFE+ Sustainable Ordunte permitted the renovation of the artificial covering installed in previous years in the zones that had suffered the most notable deterioration, once again fixing the anchorings that had been lifted owing to the extreme weather conditions. Moreover, a surface area of approximately 2,000 m^2 was covered in new zones that were easily eroded as they became exposed, with the same criteria as in earlier actions.

Figure 6. Protection of the bog slopes with coconut fibre matting.

Construction of small dams in the zones with the highest surface run-off

At the same time and again using trunks of Lawson cypress, small pools were formed which intercepted the peat washed away. These were placed in the small superficial run-off streams that formed in periods of strong rains or during ice-melt, favouring the retention of peat and the revegetation around streams (Fig. 7). This work managed to avoid the loss of peat downstream and above all it added a layer of peat on the surface of mineral aggregates, whose colonization is difficult for plant species.

Figure 7. Dams constructed with Lawson Cypress trunks in small streams to promote peat retention and its revegetation.

Introduction of plants

This stabilization of the peat was accompanied by the introduction of small seedlings of *E. vaginatum*, *Eriophorum angustifolium* and *Calluna vulgaris*, trying to make sure that its root system was in contact with the deepest layers of the bog from the start, as the most stable phreatic level would allow good rooting. Of the three species, the fastest to colonize and which has started to grow new rhizomes and new shoots is *E. angustofolium*, followed by *E. vaginatum* (Fig. 8).

In agreement with the phenology observed and the practise over these years, the time best suited for the establishment of these species, which propagate by growing rhizomes, was the end of autumn when the rhizomes are still in an active state, or at the beginning of spring when, after the end of icy conditions, the tips of the rhizomes begin to develop new shoots. This same response was seen in the colonization of the grooves on the track that is now closed to traffic. In spite of the fact that the two species appeared in the zones near the track, *E. angustifolium* was the species which colonized a larger surface area spontaneously, while it was *E. vaginatum* that formed a more compact root-ball, that impedes the installation of grasses in the zones it occupies.

Accordingly, every year the DFB carries out plantation works, with the introduction of the plant *E. vaginatum*, above all, as well as other species (*E. angustifolium*, *C. vulgaris* and *Vaccinium myrtillus*) in the peat zones that are still exposed. The seeds are collected from the same bog at the best time of the year and sown in a nursery, then transplanted in spring. Around 1,000 plants of *E. vaginatum* are introduced annually (Fig. 9).

Figure 8. Rapid colonization of *Eriophorum angustifolium* species on the protected slopes against *E. vaginatum.*

Figure 9. Plantation of *Eriohporum vaginatum* in restored zones.

First results

Almost spectacular results were obtained in the recovery zone of the grooves caused by wheels on the pre-existing track, resulting from the natural expansion of the rhizomes of the surrounding plants, and the seedlings introduced on the slopes show a vigorous rooting and development despite the tough climatic conditions at the summit of Zalama; exposed to strong drying winds from the south along with intense cold and ice in winter. These results allow us to think that the environmental conditions in Zalama continue to permit peat creation and that it is possible to promote the recovery of the bog.

Nevertheless, the climatic conditions and the level of rainfall, snowfalls, icy periods and fogs have changed in the last hundred years, and it is possible that a threat to the maintenance of the functionality of the Zalama blanket bog is climate change. In fact, these special conditions at the summit of Zalama contribute to the continuity of this unique spot, where rain-filled clouds are mixed with dry southern winds and where the humidity brought by the Atlantic winds is retained. When the winds begin to ascend the Pozo Negro mountainside they cool and end up condensing on the vegetation of the Zalama bog, soaking the plants and the ground with horizontal precipitation.

These very special conditions, a phenomenon known to those from the village of Karrantza as "*El bollo*", is recognisable from the off-white cloud, similar to a dense fog, and sticks to the summit of Zalama, loaded with water at noticeably low temperatures after having ascended to 1,300 m (Fig. 10). This "*bollo*", which is very frequent in winter but rare in summer, supplies the summit of the Ordunte peaks with water, especially the Zalama, and is a factor that can be associated with the permanence of the bog.

Figure 10. Presence of "bollo", typical in winter, at the summits of Ordunte.

Future challenges

Thanks to LIFE+ Sustainable Ordunte, a continuous monitoring of the hydrological state of the bog has been set up, obtaining data about the conditions of edaphic humidity via the sensors and dataloggers installed. This monitoring can be of special relevance in the clarification of the distribution of the species present in the more or less humid parts, for evaluating the progress of the functioning of the bog subjected to restoration, and to foresee the future of the conservation of this habitat.

Moreover, the dissemination that was promoted by the LIFE+ project and the efforts made in this area have allowed synergies and collaborations to be set up. An agreement was signed with the Nottingham Trent University, an experienced institution in bog research in the UK. In fact, the input from the Nottingham Trent University to the project LIFE+ Sustainable Ordunte is considered very valuable, via the use of a terrestrial laser scanner (TLS) to map the microtopography of the bog and the use of drones (UAV) to monitor the erosion, which allows the DFB to develop actions for the management of this zone.

There are new research challenges into the functioning of the bog, principally related to how the absence of grazing is going to affect the actual vegetation. It would be convenient to establish areas for monitoring in zones with a mosaic of species, inside and outside the bog enclosure, so as to observe the capacity of colonization of each of these species in the absence of livestock. If the limitations on grazing have produced some favourable effects in such a short time, it would be a good idea to be aware of any signs of change in the distribution of the species inhabiting the bog.

There is no doubt that the last 30 years have been very important for the acknowledgement of the value of the Zalama peat bog and so be able to recognise its uniqueness. In spite of a bright start to the recovery, thanks to the conservation activities undertaken by the DFB, the monitoring and observation of the results in the short and long term can provide patterns that could establish an adaptive global management of this zone.

References

Aseginolaza, C., Gómez, D., Lizaur, X., Monserrat, G., Morante, G., Salaverría, M. R., Uribe-Echebarría, P. M., Alejandre, J. A. 1984. *Catálogo florístico de Álava, Vizcaya y Guipúzcoa*. Gobierno Vasco. Vitoria-Gasteiz.

Aseginolaza, C., Gómez, D., Lizaur, X., Monserrat, G., Morante, G., Salaverría, R. M., Uribe-Echebarría, P. M. 1990. *Mapa de vegetación de la Comunidad Autónoma del País Vasco. Hoja 60-III - 85-I (Zalama)*. Gobierno Vasco. Vitoria-Gasteiz.

Basoinsa. 2009. *Plan de gestión del LIC de Ordunte*. Unpublished report. Diputación Foral de Bizkaia. Bilbao.

Decreto 28/1997, de 11 de febrero, por el que se aprueban definitivamente las Directrices de Ordenación Territorial de la Comunidad Autónoma del País Vasco.

Decreto 104/2002, de 14 de mayo por el que se aprueba definitivamente el Plan Territorial Sectorial de la Energía Eólica en la Comunidad Autónoma del País Vasco.

Decreto 160/2004, de 27 de julio, por el que se aprueba definitivamente el Plan Territorial Sectorial de Zonas Húmedas de la Comunidad Autónoma del País Vasco.

Decreto Foral de la Diputación Foral de Bizkaia 114/2006, de 19 de junio de 2006, por el que se aprueba el plan de gestión de *Eriophorum vaginatum* L., como especie en peligro de extinción y cuya protección exige medidas específicas en el Territorio Histórico de Bizkaia.

Decreto 65/2015, de 12 de mayo, por el que se designa Zona Especial de Conservación de Ordunte (ES2130002) en el Territorio Histórico de Bizkaia.

Heras, P. 1990. Estudio briológico de las turberas de Los Tornos y Zalama. *Sociedad de Estudios Vascos–Eusko Ikaskuntza. Cuadernos de Sección Ciencias Naturales* 7: 117-137.

Garrote, A., Muñoz, L., García, I. & Eguiguren, E. 1992. *Mapa geológico del País Vasco. Hoja 60-III y 85-I (Zalama)*. Gobierno Vasco. Vitoria-Gasteiz.

Onaindia, M. 1986. *Ecología vegetal de las Encartaciones y Macizo del Gorbea -Vizcaya-*. Universidad del País Vasco. Bilbao.

Heras, P., Infante M. 2002. *Determinación de los valores ambientales de la turbera del Zalama (Carranza; Bizkaia) y propuestas de actuación para su conservación*. Unpublished report. Gobierno Vasco. Vitoria-Gasteiz.

03

Restoration of ombrotrophic bogs in the East of Canada

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Abstract

The moss layer transfer method is a restoration technique for ombotrophic bogs whose peat has been exploited for horticultural use, whose aim is to restore a cover of *Sphagnum* that restarts the production of peat. This article summarizes the results of a monitoring program of 53 restoration projects using this method, in 12 bogs located in two provinces in the East of Canada, which had two fundamental objectives: 1) to identify the factors that explained the resulting vegetation after the restoration, and, if applicable, its success, and 2) to describe the evolution of such vegetation over time and its implications in the evaluation process of the success of the restoration. Analysis of 246 permanent plots after 946 observations over a series of 3 to 15 years from the end of the restoration, a redundancy analysis (RDA) and another classification analysis (*k-means*) showed that there was a generalized tendency over time to restore a cover of *Sphagnum* in most of the projects (54% of the plots). This occurred fundamentally when the network of drainage ditches within the exploited sector were effectively blocked. In 24% of the plots, the restoration was not able to establish vegetation, and were dominated by bare peat, because of the hot dry summers after the restoration and greater exploitation in neighbouring zones of this bog. Other technical decisions were decisive to explain the success of the restoration. For example, restoring in spring instead of summer or autumn gave rise to a third alternative dominated by other species of moss: *Polytrichum strictum* (22 % of the plots), which under a threshold of 30 % cover had a nurse effect for species of *Sphagnum*. These results are encouraging for implementing the restoration of these bogs in a systematic way.

Keywords: bog restoration; moss layer transfer; exploitation of peat; cutover bogs.

Introduction

In Canada, bogs occupy a surface of 114 million hectares (approximately 10 % of the territory; Canadian Sphagnum Peat Moss Association, 2016). Of these, 25,000 ha (0.02 % of the total) are being or have been exploited for horticultural purposes, predominantly in the south of the Eastern provinces: Québec and New Brunswick (Graf *et al.*, 2012). Practically all of them have been exploited using the vacuum extraction method, functioning since 1960-70, and which has typically been centred on ombrotrophic bogs dominated by *Sphagnum* spp.

Access by heavy machinery to apply this method of peat extraction requires severe draining of the bog, achieved by excavating a network of ditches that divides the bog into rectangular sectors. The vegetation is eliminated completely and the ombrotrophic peat, of better quality for horticultural use, is extracted down to the final 50 cm when the underlying minerotrophic layer, of lower quality, is exposed (Poulin *et al.*, 2005; Fig. 1). The bogs exploited in this way are rarely recolonized by vegetation (Poulin *et al.*, 2005; Triisberg *et al.*, 2011) and if they are, they tend to be dominated by a few vascular species, remaining in this state for decades if no measures are taken (Poschlod *et al.*, 2007).

Figure 1. Bog being exploited by the vacuum extraction method. In the foreground a primary drainage ditch can be seen.

According to the latest estimates from the Canadian Sphagnum Peat Moss Association, 1,800 ha of the bogs exploited had been subjected to restoration or rehabilitation processes in 2006 and more than 3,000 ha were planned for restoration in the following five years (Graf *et al.*, 2012). Although advances in the use of the vacuum method have allowed the exploitations to reach the mineral layer and be left in more "minerotrophic" conditions (cutaway type bogs, see Graf & Rochefort, 2008; Graf *et al.*, 2012), most of the restoration efforts to date have concentrated on bogs that still have residual layers of organic material (cutover type bogs).

This article concentrates on the latter, where the restoration efforts have the overriding objective of reintroducing a plant community dominated by *Sphagnum*

(Rochefort, 2000; Rochefort *et al.,* 2003; Rochefort & Lode, 2006). *Sphagnum* is the genus of bryophytes that typically dominates ombotrophic bogs (Rydin & Jeglum, 2006). *Sphagnum* are engineer species and can bring back the process of peat accumulation in the long term (Van Breemen, 1995).

So, the final objective of the restoration is to place the exploited plots on a successional trajectory which, eventually, alleviates the loss of the essential ecological function of carbon sink of the bogs (Rydin & Jeglum, 2006).

The first attempts to reintroduce *Sphagnum* in exploited and abandoned cutover bogs consisted in adapting the species introduced to the physical-chemical conditions of the residual peat, but met with little success (Wind-Mulder *et al.*, 1996). There were also unsuccessful attempts to introduce fragments of *Sphagnum* in hollows and in drainage ditches, with the hope that they would expand spontaneously to the rest of the bog (Rochefort & Lode, 2006).

The turning point of this research was to include the determining micro-climatic conditions for the establishment of *Sphagnum* in the restoration plan, instead of concentrating on reproducing the habitats of the already developed mature colonies (Rochefort & Bastien, 1998; Rochefort & Lode, 2006). Hence a method of restoration was achieved which could be applied systematically in the restoration of cutover bogs: the moss layer transfer technique (Quinty & Rochefort, 2003; Graf *et al.*, 2012).

The moss layer transfer technique is made up of four stages:

> 1) reshaping of the topography and blocking of the drainage ditches so as to improve the hydrological conditions of the bog, 2) dispersion of *Sphagnum* spores previously collected from a donor bog that would soon be exploited, 3) application of a protective layer of mulch to shield the fragments of *Sphagnum* before drying out, and 4) (occasionally) application of fertilizer with P to promote colonization of nurse species that would help to establish and develop *Sphagnum*.

Long-term monitoring of these actions to evaluate their success has shown that the method can bring about an almost complete cover of *Sphagnum* in the abandoned bogs in just one decade (Boudreau & Rochefort, 2008; Poulin *et al.*, 2012). However, the method still generates undesired successional trajectories in 40% of the cases (González *et al.*, 2013a; 2014a). A long-term monitoring plan was carried out by the Peatland Ecology Research Group from the University of Laval (Québec, Canada; GRET-PERG, 2016) from the middle of the 1990s, with the aim of determining the factors that explain why *Sphagnum* is not capable of colonizing all the restored sites with this method, and so perfect it.

This article summarizes part of the conclusions obtained after analysis of this monitoring plan, published in detail by González & Rochefort (2014). In particular there were examinations of the effects of drainage, the physical-chemical properties of the peat, the meteorology, factors of management-techniques and of the landscape on the vegetation that colonized a series of exploited ombotrophic cutover bogs which had been abandoned and restored with the moss layer transfer technique. A second objective was to describe the evolution of the vegetation over time and its implications for the evaluation of the success of the restoration.

Methods

Sites of study

This study includes a total of 53 restoration projects executed by the peat industry in collaboration with the Peatland Ecology Research Group (PERG-GRET, 2016). These 53 projects are located in the provinces of Québec and New Brunswick, specifically in 12 ombotrophic Cutover bogs distributed in three climatic regions (Environment Canada, 2012): the Atlantic region (14 sectors in 5 bogs), the Saint Lawrence River region (14 sectors in 6 bogs) and the Saint Jean Lake region (24 sectors in 1 bog; Fig. 2). Each project had a rectangular sector (limited by drainage ditches) that was restored as an independent unit, uniformly, spatially and over time, with the moss layer transfer technique. The sectors could be located at distances of up to 5 km in the same bog, or in different bogs.

A post-restoration monitoring plan documented the changes in the plant community in 246 permanent plots of 5 m^2. The number of plots per sector varied (from 1 to 32) depending on their size (from 1 to 30 ha) and the topographic heterogeneity created during the reshaping of the sector. The oldest restoration dates from 1994, the most recent from 2008 (González & Rochefort, 2014).

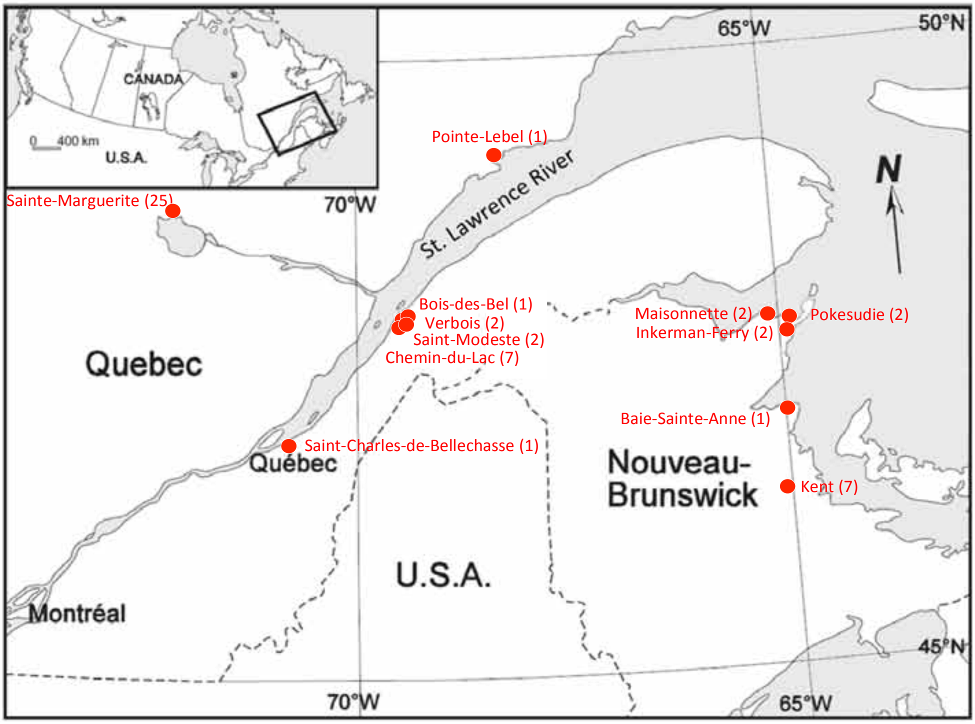

Figure 2. Location of the 53 restored sectors in 12 cutover bogs in the provinces of Quebec and New Brunswick (East of Canada). Reproduced with permission of Elsevier in Fig. 1 of González & Rochefort (2014).

Vegetation sampling

The vegetation from each plot was sampled in the autumn of the third year after the restoration and then every two years up to year 9 (on occasions 10 or 11) and every five years after that. The third year was established as the start of the monitoring because it is difficult to identify some species (especially mosses) during their early stages, and to make sure that only established individuals were recorded. The shortest series of data had only a single observation (3 years post-restoration), whilst the sectors with older restorations had a series of 6 observations up to 15 years post-restoration (González & Rochefort, 2014).

For the vascular vegetation four 1 m^2 squares were used, located systematically within the permanent plot. All vascular plants (trees, shrubs and herbs) within the squares were identified at species level, and their vertically projected cover, in addition to that of the bare peat, were estimated visually. The cover of all the bryophyte and lichen species was recorded in 20 smaller squares, of just 25 cm^2, which were also distributed systematically within the permanent plot. The samples of vegetation used in this study were made between 1997 and 2011.

Factors that may determine the establishment of the vegetation

The variables that had a potential capacity to determine a successful restoration were classified into five groups: 1) drainage, 2) physical-chemical properties of the peat, 3) meteorology, 4) management-technique and 5) landscape.

1) Drainage. The hydrological conditions in the restored sector are probably the most important factor in the recovery of the *Sphagnum* vegetation cover (Price *et al.*, 2003; Holden *et al.*, 2004). *Sphagnum* requires a high water-level (<40 cm under the surface of the peat), humidity >50 % and a substrate pressure >-100 mb (Price & Whitehead, 2001). So, the drainage in each restored sector was evaluated using three variables: effectiveness of the blocking of the primary ditches, effectiveness of the blocking of the secondary ditches, and uniformity of the flooding regime in the sector. The primary ditches are much deeper (>1 m) and mark the boundaries of the sector, whilst the secondary ditches that drain the water within the sector are shallower (<1 m). The variables were estimated visually on a semi-quantitative scale in summer of 2012 (see a more detailed explanation of the variables in González & Rochefort, 2014).

2) Physical-chemical properties of the peat. The degree of decomposition of the residual peat determines the physical properties, such as porosity and the hydraulic conductivity that control the movement of water through the peat (Schlotzhauer & Price, 1999). Other physical processes such as frost heaving can impede the recolonization of the plants on the bare peat (Groeneveld & Rochefort, 2002). The depth of the residual peat (Triisberg *et al.*, 2014) and its chemistry can also determine the resulting vegetation (Andersen *et al.*, 2011). So, the physical-chemical properties of the peat in each sector were determined from the residual peat collected next to the permanent plots. The pH and the electrical conductivity were measured in a 1:10 solution, the latter was corrected according to Sjörs (1952). The degree of decomposition was evaluated manually in the field according to the Von Post scale (Von Post & Granlund, 1926). The depth of the residual peat was measured with an iron bar.

3) Meteorology. The meteorological conditions during the first stage of growth after restoration are decisive since the fragments of *Sphagnum* do not tolerate low humidity (Sagot and Rochefort, 1996; Campeau *et al.*, 2004; Chirino *et al.*, 2006). The temperature and monthly precipitation were recorded at the nearest stations to each sector (Environment Canada, 2012) to calculate: temperature and precipitation accumulated in spring (May-June), summer (July-August) and autumn (September-October) for the first stage of growth after the restoration works (i.e., the following natural year if the restoration was made in autumn). The number of rainy days in summer were also calculated and to represent summer drought, the maximum number of days without effective rain ($\geq 2\,mm \cdot day^{-1}$). Two millimetres is the quantity of rain that is estimated that is intercepted by the mulch before it is useful for the seeds and spores that the mulch is protecting (Price *et al.*, 1998).

4) Management-technique. The success of the restoration can also be determined by the project management and other aspects of the techniques in the application of the restoration method. For example, it is not clear whether the fertilization with P is necessary in every case or if it should be limited to places that are more sensitive to frost heaving. P helps to establish the nurse species of *Sphagnum*: the bryophyte *Polytrichum strictum* (Groeneveld & Rochefort, 2002; Sottocornola *et al.*, 2007). Information was also recorded concerning: season of the restoration works (spring vs. summer/autumn), time lapsed (in years) between the end of exploitation and the restoration, size of the sector, ratio between size of the donor sector and size of the restored sector, and time lapsed (in years) between the restoration and fertilization with P.

5) Landscape. Finally, the composition of the plants in the ombotrophic bogs can be affected by human activities (agriculture, forestry uses) in the proximity (Pellerin & Lavoie, 2000; 2003; Lachance & Lavoie, 2004; González *et al.*, 2013b; 2014b). So therefore, the percentages of the land occupied in a buffer of 500 m for natural uses (mainly forest, bodies of water and bogs), bogs in exploitation, restored bogs, exploited and abandoned bogs, and other anthropological uses (mainly agriculture) were calculated by interpreting aerial photographs available on Google Earth (more details in González & Rochefort, 2014).

Processing of data and statistical analysis

The cover of all vascular and non-vascular species, and of the bare residual peat obtained from the different quadrats were averaged at the level of the permanent plot, until a vegetation matrix was generated with a row for each plot and year observed and a column for species (dimension of the matrix: 946 x 88). Although the objective of the study were the restored sectors, the experimental units were the permanent plots so that it was possible to evaluate the situation in which the vegetation that colonizes the sectors does not do so uniformly (for example, an incorrect reshaping could bring vegetation that is radically different between zones in the same sector).

Firstly, a Redundancy Analysis (RDA; Legendre & Legendre, 2012) was made using the vegetation matrix subjected to a Hellinger transformation as "Y" and a matrix with all the factors that could potentially explain the vegetal composition (drainage, physical-chemical properties of the peat, meteorology, management-technique and landscape) as "X". The RDA is a multidimensional analysis of canonical ordination, capable of summarizing the information related to the composition of the vegetation in a few axes with an ecological meaning (normally only two are interpreted) and to explain them as a function of a series of independent variables (predictors), in the same way that a multiple linear regression would be made with a single dependent variable (Legendre & Legendre, 2012). The degree of significance of the RDA was evaluated using a test with 9,999 permutations.

To aid the interpretation of the RDA, the vegetation matrix (after Hellinger) was also subjected to a partition *k-means* (Legendre & Legendre, 2012). Hence the observations (plot x year) were classified in k groups that were interpreted as a function of the success of the restoration (criteria = recovery of moss cover, preferably of *Sphagnum*). Once completed, the development of the plots was examined in the canonical space defined by the RDA, as well as the changes of such plots from a group k_i to the other, k_j over time.

All the analyses were made with the statistical software R (version 2.15.2). For further technical details of the analysis, see González & Rochefort (2014).

Results

Possible scenarios after restoration

The RDA explained 30.3 % of the variability in the vegetation matrix transformed with Hellinger (r^2 adjusted = 0.303, F = 20.566, P < 0.001). The first gradient of the RDA separated the observations where moss cover appeared - of *Sphagnum rubellum*, *S. fuscum* and *Polytrichum strictum* - from those with bare peat, where it was also frequent to find trees of the *Betula* genus along with the bryophyte *Dicranella cerviculata* (RDA1, graph of species, Fig. 3). The second axis separated the observations dominated by typical bog species, such as *Eriophorum vaginatum*, *Sphagnum rubellum* and *S. magellanicum*, from those dominated by the nurse effect moss *P. strictum* (RDA2, graph of species, Fig. 3). In agreement with the first two axes of the RDA, the partition *k-means* separated the 946 observations into three groups. These groups, as a function of the composition of the vegetation, reflected three possible scenarios after the restoration: success (dominance of *Sphagnum* spp., 402 observations; Fig. 4), failure (dominance of bare peat, 359 observations; Fig. 5); and alternative (domination of *P. strictum*, 185 observations; Fig. 6).

Factors that explain the success of the restoration

The five large groups of factors were decisive in explaining the resulting vegetation after the restoration, and in this order of importance: management-technique > meteorology > landscape > physical-chemical properties of the peat > drainage (graph of variables, Fig. 3). The dominance of bare peat was most frequent when the first sum-

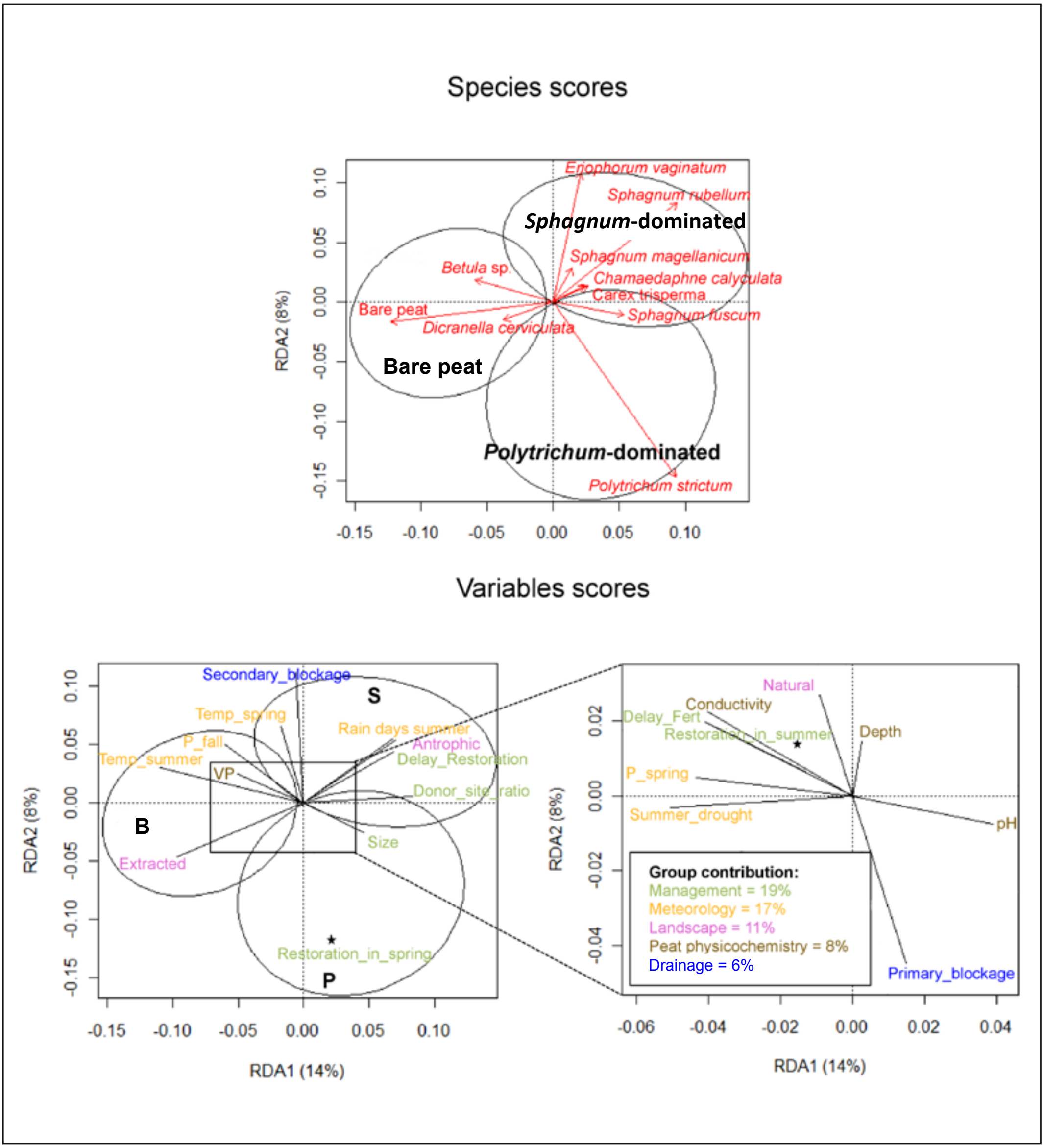

Figure 3. RDA (*scaling* = 1) for 246 permanent plots in the 53 sectors studied. The coefficients of the species and of the factors were divided by 20 and 7, respectively, so that they could be visualized. 87 species identified on bare peat, but only the 10 with the highest coefficients are represented. The groups obtained with the clustering *k-means* method are represented with ellipses that include 90 % of the plots from each category. The contribution of each group of factors was calculated via an RDA implemented for each group of factors. To improve the visualization of the graph, the factors with the highest and lowest coefficients are shown on the left and right of the two different figures. Reproduced with permission of Elsevier for Fig. 3 of González & Rochefort (2014).

Figure 4. Example of permanent plot classified as success (dominance of *Sphagnum* spp.).

Figure 5. Example of permanent plot classified as failure (dominance of bare peat).

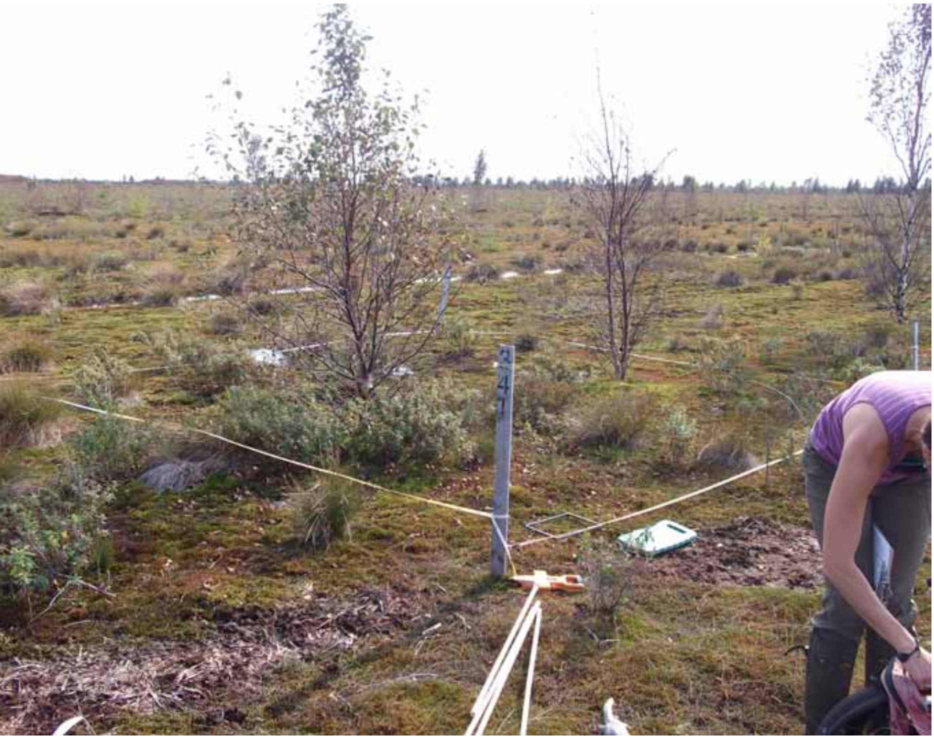

Figure 6. Example of permanent plot classified as alternative state (dominance of *Polytrichum strictum*).

mer after the restoration was hot, and when the restored sector was surrounded by sectors being exploited (left of the RDA1 in the graph of variables, Fig. 3). The greater the amount of vegetal material introduced (greater ratios donor sector:restored), the better establishment of *Sphagnum* was observed (right on the RDA1 in the graph of variables, Fig. 3). The observations dominated by *Sphagnum* were distinguished from *P. strictum* because the former had greater blockage of the secondary ditches and the flatter ones were usually restored in spring (axis RDA2 in the graph of variables, Fig. 3). And to a lesser extent, a higher level of rainfall in autumn and summer and higher temperatures in spring favoured colonization by *Sphagnum* over *P. strictum*.

Evolution of the permanent plots over time

The vegetation in the permanent plots showed a notable evolution throughout the successive observations. Most of the plots moved to the right on the RDA1 (left of Fig. 7) that is to say, the plots at the start were dominated by bare peat and then changed to one of the other two categories.

Of the 146 permanent plots classified as observation-failure three years after the restoration, 50% changed to "success" and 21% to "alternative", so only 43 were finally classified as "failure" at the end of the monitoring (right of Fig. 7). However, only two of the 75 plots that were dominated by *Sphagnum* or *Polytrichum* in the first observation ended up dominated by bare peat. In the final year of monitoring, 134 plots were classified as success, tripling those that ended as failure (54) or in an alternative state (58).

In very few sectors 100 % of the plots were classified as in the same category. Nevertheless, most sectors presented a typology of dominant permanent plot, so that the proportions of success (54 %), failure (22 %) and alternative states (24 %) were also retained, approximately, at the sector level (data not shown, see González & Rochefort, 2014 for further information).

The evolution over time of each one of the three possible scenarios after restoration can clearly be seen by observing the changes in the main groups of vascular and non-vascular plants in the 246 plots, categorizing them in agreement with the group k to which their final observation belonged (Fig. 8). In the successful plots, *Sphagnum* covered almost 50 % of the plot 9–10 years after restoration.

This, together with the vigorous development of grasses (average cover = 23%) and shrubs (22 %), left bare peat in just 13 %. In the plots dominated by *P. strictum*, other bryophytes different to *Sphagnum* (mainly *P. strictum*) occupied more than half the surface. However, owing to the fact, that the vascular plants do not develop much in this group, almost a third of the surface was still exposed 9–10 years after the end of the restoration works. On the plots classified as “failure”, no plant forms occupied more than 10 % of the surface, and the bare peat kept its majority category (~60 %) and was stable over time.

Discussion

Multifactorial control of successful restoration

The success of the restoration of cutover type bogs, measured as recovery of plant cover dominated by *Sphagnum* (González *et al.*, 2013a; 2014a, González & Rochefort, 2014), depends on a wide variety of factors. Of those included in this study, the group that best explains the greater variability in the vegetation were the questions related to the management techniques of the restoration method (contribution of groups, 19 %, Fig. 3). This result shows that there is room for improvement in the restoration process.

Simple decisions such as the amount of material applied (ratio sector donor : restored) and the season in which the restoration was carried out can be decisive for the results. The official donor : restored ratio is 1:10 (Rochefort *et al.*, 2003) and here we can observe that lowering it to 1:12 or 1:15 can mean that the restoration does not work. Restoring in spring favours *P. strictum* over *Sphagnum*.

We believe that this is due to the fact that in spring as the thaw is quite recent, the grooves left by heavy machinery are deeper as it works on softer, waterlogged terrain, causing a micro-topography in which only *P. strictum* can become established. The grooves (in the order of around a minimum depth of 20 cm) drain sufficient water to impede the establishment of *Sphagnum* (Price *et al.*, 1998).

Our results also show that the late application of P is correlated with the dominance of *P. strictum* but it is difficult to establish a causal relationship since P is usually applied after restoration when an unusually high quantity of bare peat is observed.

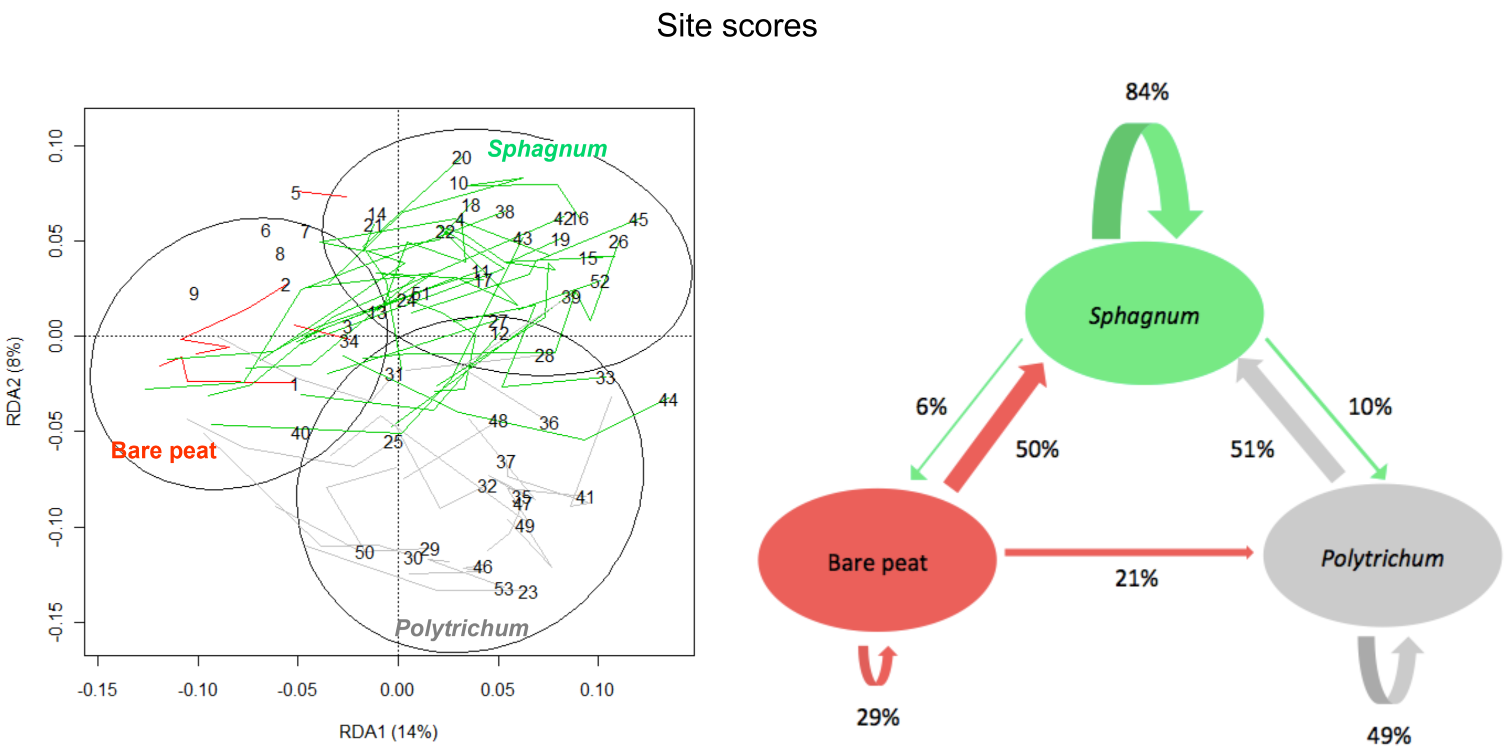

Figure 7. RDA coefficients (*scaling* = 1) of the 246 permanent plots in 53 restoration projects. To the left, trajectory of the restored sectors. For better visualization of the same, only the centroid (average of the permanent plots) in each sector was drawn, each line represents the trajectory of each sector, with the colours representing the category (success, failure, alternative state) possible from the most recent observation. The number of each sector indicates the final position in the canonical space. The groups obtained with the clustering *k-means* method were represented with ellipses that include 90 % of the plots from each category. To the right, the percentage of permanent plots that changed category from the first to final observation. The size of the arrows is proportional to the percentages, observation of a reversal of the success (green arrows) was rarely seen. Reproduced with permission of Elsevier for Fig. 4 by González & Rochefort (2014).

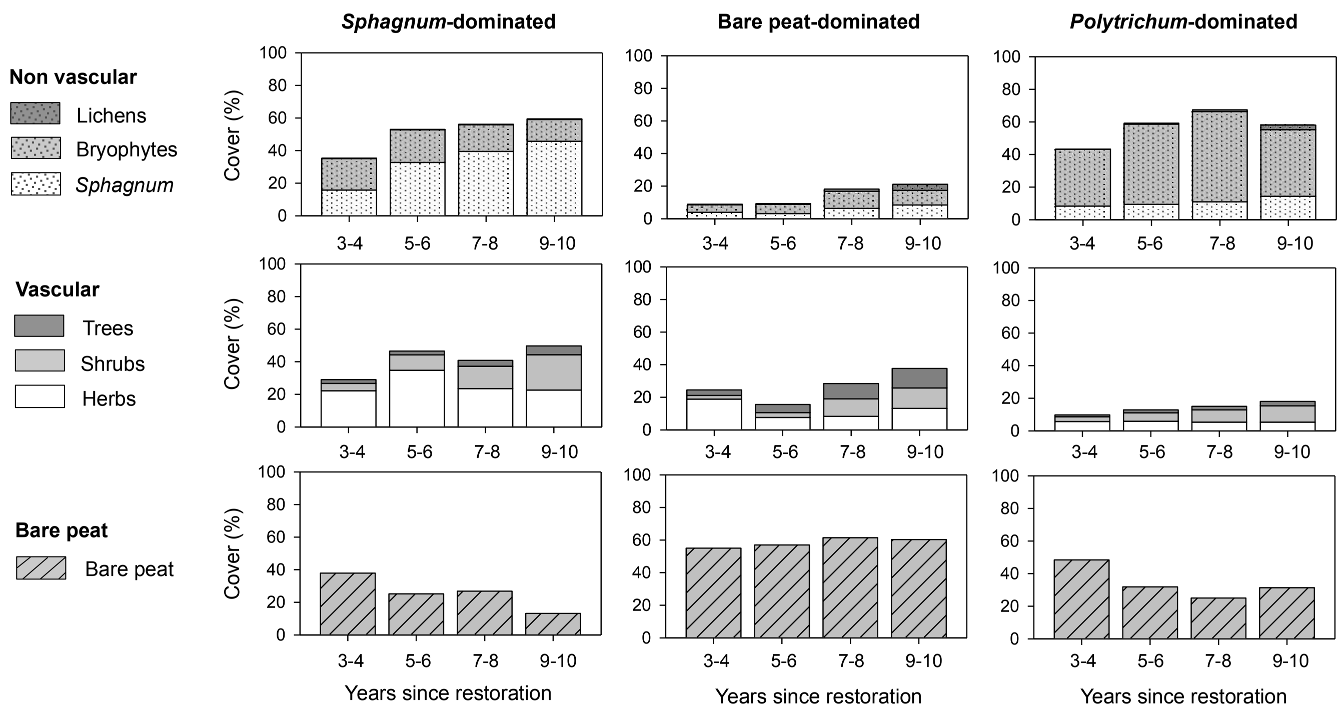

Figure 8. Changes in cover of vascular and non-vascular species, and % of bare peat, in plots dominated by *Sphagnum* (success, 134 permanent plots), bare peat (failure, 54) and by *P. strictum* (alternative, 58). For this analysis the plots were assigned to each category according to the result from the *k-means* partition in the final observation. Data over 10 years old is not shown because there were very few of this type of sectors (only 27 plots in 6 bogs). 'Bryophytes' indicates bryophytes that belong to a genus different from *Sphagnum* (mainly *P. strictum*). Reproduced with permission of Elsevier for Fig. 5 by González & Rochefort (2014).

As expected, meteorological factors played an important role (contribution of groups, 17 %, Fig. 3). High temperatures and low precipitation, above all in summer, did not favour the establishment of *Sphagnum*, as Chirino *et al.* (2006) had suggested, in experimental plots. As was also pointed out by Chirino *et al.* (2006), a uniform distribution of precipitation throughout the year is important for the establishment of introduced fragments of *Sphagnum*. This was demonstrated by a higher number of rainy days in summer and the number of consecutive days without rain ("drought" in Fig. 3) that were more significant in our RDA than the total amount of rain in summer.

Surprisingly, drainage was the group of factors that least contributed to explaining the resulting vegetation (6 %). However, the effectiveness of the blocking of the secondary ditches was the variable that best explained the second axis of the RDA, with a better blocking associated with *Sphagnum* over *P. strictum*. Although it is necessary to research into the effects of a better blocking of these secondary ditches in the hydrological balances of the restored sectors, the variable "effectiveness of the blocking of the secondary ditches" was highly correlated with the variable "uniformity of the flooding regime in the sector" (Spearman correlation = 0.71, $P < 0.001$).

Surprisingly, however, a better blocking of the primary ditches did not have any noticeable effect on the colonization of *Sphagnum*. This apparent paradox can be explained because in spite of the blocking of the primary ditches the exploitation continued in adjoining sectors nearby, drained with other primary ditches. This explanation gains more credence after the observation that the percentage of exploited adjoining sectors was positively correlated to the dominance of bare peat. In addition to the influence of the active drainage in the adjoining exploited sectors on the restored sectors, other factors related to the exploitation of the peat could have contributed to the failure of certain projects.

For example, the deposition of dry peat arriving by air (Faubert & Rochefort, 2002), or a lower deposition of seeds and spores (Poulin *et al.*, 1999), etc. For these reasons and also because of longer time lapses from the end of the restoration works, were related to the establishment and development of *Sphagnum*, a recommendation is that no restoration projects are implemented in isolated sectors but in bogs where the exploitation has already ended and which can be treated as autonomous hydrological units.

In fact, Girard *et al.* (2002), observed that sectors exploited with traditional methods (before the introduction of vacuum) of larger size saw spontaneous regrowth of vegetation more easily than other smaller ones, surrounded by active exploitations, after being abandoned. Although the correlation was weak, our results also show that largest sectors were more strongly associated with the dominance of *Sphagnum* than the smaller ones.

Evolution of the restoration over time

Although the aim of the restoration was not to completely recover the ecological state of the bogs lost through exploitation, this study showed that the changes in the plant composition of the restored bogs can be large in relatively short periods (~10 years). Most of the plots initially dominated by bare peat changed category between the first and final observations. However, the most desired final state that of dominance by *Sphagnum*, turned out to be very stable: once reached, they rarely changed back. This is a very positive result, given that restoration seems to tend, sooner or later, to reach the desired goals.

Some plots changed state more slowly than others, which highlights the problem of when is the right moment to evaluate the success, or not, of the restoration. According to Rochefort *et al.* (2013), the evaluation should not be made until at least a decade after the end of the restoration works. At that moment, the evaluation should be made by comparing the restored sector with reference type systems from the region (Poulin *et al.*, 2012; Rochefort *et al.*, 2013). Pouliot *et al.* (2011), for example, found that a minimum period of 10 to 30 years is necessary for the recovery of the typical hollow-hummock micro-topography in cutover type bogs.

However, restorers need to evaluate their restoration works a short time after their implementation (≤ 5 years in the East of Canada), either to obtain environmental certifications or to comply with legislation. As restoration consists of, essentially, accelerating natural succession (Walker *et al.*, 2007), we believe that it is legitimate to see dominance by peat (sectors classified as "failure") as definitively failed restoration experiences that must be restored again, even at risk of treating what may really be a temporary state.

So, the failure of a project should be defined as the absence of the desired vegetation after a certain amount of time. In our study, the plant composition in the plots changed rapidly during the first years of monitoring but stabilized over subsequent observations (distances between observations of the RDA, data not shown). In a study carried out in one of our sectors, the Bois-des-Bel bog, Rochefort *et al.* (2013), observed that the cover of the different species and vegetation strata did not stabilize until approximately six years post-restoration. Taking this into account, we recommend that the success of a bog restoration project should not be evaluated until at least five years after its implementation. Using the same data base, González *et al.* (2013a; 2014a), argued how to predict the success of restoration even earlier (3 years from the end of the restoration works).

The plots initially dominated by *P. strictum* also changed frequently, becoming dominated by *Sphagnum* although to a lesser extent than on bare peat. A transition from *P. strictum* to *Sphagnum* was expected knowing the first´s nurse effect on the second (Groeneveld & Rochefort, 2002; Groeneveld *et al.*, 2007).

Recent works suggest that this facilitating capacity occurs only under a threshold of *P. strictum* cover, specifically ~30 % three years after the restoration (González *et al.*, 2013a). If these thresholds are exceeded, the restored sectors seem to fall into a stable alternative state (*sensu* Beisner *et al.*, 2003). In such cases, if we take the restoration objective as reaching an abundance of *Sphagnum*, similar to that found in reference systems (ca. 70 %, Poulin *et al.*, 2005), the sectors dominated by *P. strictum* could be a second category of "failure", and should be restored again.

However, this type of vegetation complies with other objectives, such as stabilizing the peat to prevent erosion and deterioration of any bodies of water that may have been created in the sector, or as a first step in the rehabilitation of the sector for agroforestry uses.

However, the technique of moss layer transfer brought about a plant cover that was dominated by *Sphagnum*, the main objective of the restoration, in 54 % of the restored sectors. Moreover, this percentage tended to increase with the age of the restoration (over a range of 3 to 15 years). Once reached, dominance by *Sphagnum* seemed to be highly stable. A quarter of the projects gave rise to dominance by another species of moss, *P. strictum*, which could also contribute to the production of peat in the long term. If the multiple factors that explain which species were able to recolonize the restored sites are controlled, the bogs exploited for reasons of horticulture could be restored systematically.

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References

Andersen, R., Rochefort, L., Landry, J. 2011. La chimie des tourbières du Québec: une synthèse de 30 années de données. *Le Naturaliste Canadien* 135: 5-14.

Armstrong, A., Holden, J., Kay, P., Foulger, M., Gledhill, S., Mcdonald, A.T., Walker, A. 2009. Drain-blocking techniques on blanket peat: A framework for best practice. *Journal of Environmental Management* 90: 3512-3519.

Beisner, B.E., Haydon, D.T., Cuddington, K. 2003. Alternative stable states in ecology. *Frontiers in Ecology and the Environment* 1:376-382.

Boudreau, S., Rochefort, L. 2008. Plant establishment in restored peatlands: 10-years monitoring of sites restored from 1995 to 2003. In. *Proceedings of the 13th International Peat Congress: After Wise Use – The Future of Peatlands, Volume 1: Oral Presentations*. C. Farrell and J. Feehan (eds.). International Peat Society, Jyväskylä, Finland.

Campbell, D.R., Lavoie, C., And Rochefort, L. 2002. Wind erosion and surface stability in abandoned milled peatlands. *Canadian Journal of Soil Science* 82: 85-95.

Campeau, S., Rochefort, L., Price, J.S. 2004. On the use of shallow basins to restore cutover peatlands: Plant establishment. *Restoration Ecology* 12: 471-482.

Canadian Sphagnum Peat Moss Association. 2016. www.peatmoss.com

Chirino, C., Campeau, S., Rochefort, L. 2006. *Sphagnum* establishment on bare peat: The importance of climatic variability and *Sphagnum* species richness. *Applied Vegetation Science* 9: 285-294.

Environment Canada. 2012. Past *Wheather*. www.ec.gc.ca

Faubert, P., Rochefort, L. 2002. Response of peatland mosses to burial by wind-dispersed peat. *The Bryologist* 105: 96-103.

Girard, M., Lavoie, C., Thériault, M. 2002. The regeneration of a highly disturbed ecosystem: a mined peatland in Southern Québec. *Ecosystems* 5: 274-288.

González, E., Rochefort, L., Boudreau, S., Hogue-Hugron, S., Poulin, M. 2013a. Can species indicators predict restoration outcomes early in the monitoring process? A case study with peatlands. *Ecological Indicators* 32: 232-238.

González, E., Rochefort, L., Poulin, M. 2013b. Trajectories of plant recovery in block-cut peatlands 35 years after peat extraction. *Applied Ecological and Environmental Research* 11: 385-406.

González, E., Rochefort, L., Boudreau, S., Poulin, M. 2014a. Combining indicator species and key environmental and management factors to predict restoration success of degraded ecosystems. *Ecological Indicators* 46: 156-166.

González, E., Henstra, S., Rochefort, L., Bradfield, G.E., Poulin, M. 2014b. Is rewetting enough to recover Sphagnum and associated peat-accumulating species in traditionally exploited bogs? *Wetlands Ecological Management* 22: 49-62.

González, E., Rochefort, L. 2014. Drivers of success in 53 cutover bogs restored by a moss layer transfer technique. *Ecological Engineering* 68: 279-290.

Graf, M.D., Rochefort, L. 2008. Techniques for restoring fen vegetation on cut-away peatlands in North America. *Applied Vegetation Science* 11: 521-528.

Graf, M., Bérubé, V., Rochefort, L. 2012. Restoration of peatlands after peat extraction, in: Vitt, D., Bhatti, J. (eds.), *Restoration and Reclamation of Boreal Ecosystems*, Cambridge University Press, pp. 259-280.

GRET-PERG (Groupe de Recherche en Écologie des Tourbières - Peatland Ecology Research Group), 2016. http://www.gret-perg.ulaval.ca

Groeneveld, E., Rochefort, L. 2002. Nursing plants in peatland restoration: on their potential use to alleviate frost heaving problems. *Suo* 53: 73–85.

Groeneveld, E.V.G., Massé, A., Rochefort, L. 2007. *Polytrichum strictum* as a nurse-plant in peatland restoration. *Restoration Ecology*. 15: 709-719.

Haapalehto, T.O., Vasander, H., Jauniainen, S., Tahvanainen, T., Kotiaho, J.A. 2011. The effects of peatland restoration on water-table depth, elemental concentrations, and vegetation: 10 years of changes. *Restoration Ecology* 19: 587-598.

Holden, J., Chapman, P.J., Labadz, J.C. 2004. Artificial drainage of peatlands: hydrological and hydrochemical process and wetland restoration. *Progress in Physical Geography* 28: 95-123.

Lachance, D., Lavoie, C. 2004. Vegetation of Sphagnum bogs in highly disturbed landscapes: relative influence of abiotic and anthropogenic factors. *Applied Vegetation Science* 7: 183-192.

Legendre, P., Legendre, L. 2012. *Numerical ecology*. Elsevier Science, Amsterdam.

Pellerin, S., Lavoie, C. 2000. Peatland fragments of southern Quebec: recent evolution of their vegetation structure. *Canadian Journal of Botany* 78: 255-265.

Pellerin, S., Lavoie, C. 2003. Reconstructing the recent dynamics of mires using a multitechnique approach. *Journal of Ecology* 91: 1008-1021.

Poschlod, P., Meindl, C., Sliva, J., Herkommer, U., Jäger, M., Schukert, U., Seemann, A., Ullmann, A., Wallner, T. 2007. Natural revegetation and restoration of drained and cut-over raised bogs in southern Germany – A comparative analysis of four long-term monitoring studies. *Global Journal of Environmental Research* 11: 205-216.

Poulin, M., Andersen, R., Rochefort, L. 2012. A new approach for tracking vegetation change after restoration: a case study with peatlands. *Restoration Ecology* 21: 363-371.

Poulin, M., Rochefort, L. Desrochers, A. 1999. Conservation of bog plant species assemblages: Assessing the role of natural remnants in mined sites. *Applied Vegetation Science* 2: 169-180.

Poulin, M., Rochefort, L., Quinty, F., Lavoie, C. 2005. Spontaneous revegetation of mined peatlands in eastern Canada. *Canadian Journal of Botany* 83: 539-557.

Pouliot, R., Rochefort, L., Karofeld, E. 2011. Initiation of microtopography in revegetated cutover peatlands. *Applied Vegetation Science* 14: 158-171.

Price, J.S., Rochefort, L. Quinty, F. 1998. Energy and moisture considerations on cutover peatlands: surface microtopography, mulch cover and *Sphagnum* regeneration. *Ecological Engineering* 10: 293-312.

Price, J.S., Whitehead, G.S. 2001. Developing hydrologic thresholds for *Sphagnum* recolonization on an abandoned cutover bog. *Wetlands* 21: 32-40.

Price, J.S., Rochefort, L., Campeau, S. 2002. Use of shallow basins to restore cutover peatlands: hydrology. *Restoration Ecology* 10: 259-266.

Price, J.S., Heathwaite, A.L., Baird, A.J. 2003. Hydrological processes in abandoned and restored peatlands: An overview of management approaches. *Wetlands Ecology and Management* 11: 65-83.

Quinty, F., Rochefort, L. 2003. *Peatland restoration guide*. Canadian Sphagnum Peat Moss Association, New Brunswick Department of Natural Resources and Energy, Quebec, Canada.

Rochefort, L., Bastien, D.F. 1998. Réintroduction de sphaignes dans une tourbière exploitée : Évaluation de divers moyens de protection contre la dessiccation. *EcoScience* 5: 117-127.

Rochefort, L. 2000. *Sphagnum* – A keystone genus in habitat restoration. *The Bryologist* 103: 503-508.

Rochefort, L., Quinty, F., Campeau, S., Johnson, K., Malterer, T. 2003. North American approach to the restoration of Sphagnum dominated peatlands. *Wetlands Ecology and Management* 11: 3-20.

Rochefort, L., Lode, E. 2006. Restoration of degraded boreal peatlands. In: *Boreal peatland ecosystems*. R.K. Wieder, and D. H. Vitt. (eds.). Springer-Verlag, Berlin, pp. 9–24.

Rochefort, L., Isselin-Nondedeu, F., Poulin, M., Boudreau, S. 2013. Plant monitoring of an industrial peatlands under restoration: comparing survey methods for tracking 8 years of vegetation changes. *Wetlands Ecology and Management* 21: 71.85.

Sagot, C., Rochefort, L. 1996. Tolérance des sphaignes à la dessiccation. Cryptogamie: *Bryologie et Lichenologie* 17: 171-183.

Schlotzhauer, S.M., Price, J.S. 1999. Soil water flow dynamics in a managed cutover peat field, Quebec: Field and laboratory investigations. *Water Resources Research* 35: 3675-3683.

Sjörs, H. 1952. On the relation between vegetation and electrolytes in north Swedish mire waters. *Oikos* 2: 241-258.

Sottocornola, M., Boudreau, S., Rochefort, L. 2007. Peat bog restoration: Effect of phosphorus on plant re-establishment. *Ecological Engineering* 31: 29-40.

Triisberg, T., Karofeld, E., Paal, J. 2011. Re-vegetation of block-cut and milled peatlands: an Estonian example. *Mires and Peat* 8: 1-14.

Triisberg, T., Karofeld, E., Liira, J., Orru, M., Ramst, R., Paal, J. 2014. Microtopography and the properties of residual peat are convenient indicators for restoration planning of abandoned extracted peatlands. *Restoration Ecology* 22: 31-39.

Van Breemen, N. 1995. How *Sphagnum* bogs down other plants. *Trends in Ecology & Evolution* 10: 270-275.

Von Post, L, Granlund, E. 1926. Södra Sveriges Torvtillgångar I. *Sveriges Geologiska Undersökning* Serie C 335: 1-127.

Walker, L.R., Walker, J., Del Moral, R. 2007. Forging a new alliance between succession and restoration. In: *Linking restoration and ecological succession* (eds.) Walker, L.R., Walker, J., Hobbs, R.J. New York. Springer. pp. 1-18.

Wind-Mulder, H.L., Rochefort, L., Vitt, D.H., 1996. Water and peat chemistry comparisons of natural and post-harvested peatlands across Canada and their relevance to peatland restoration. *Ecological Engineering* 7: 161-181.

04

Recovery of the Bernallán bog in the upper basin of the River Miera

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Abstract

The Bernallán bog is a basin or flooded plain, situated between glacial moraines and crossed by small streams, in the upper basin of the River Miera in the Site of Community Interest (SCI) Eastern Mountains of Cantabria. The geology substrate of the bog, because of its glacial origin, is made up of a deposit from the Quaternary period. Both the bog and its more rugged surroundings are used as grazing for cattle and horses during summer and autumn. The bog has a zone dominated by herbaceous plants; above all by Cyperaceae (sedges), such as *Eriophorum angustifolium*. This is accompanied by *Tricophorum cespitosum* and at least eight species of sedges (*Carex* sp.), the most notable being *Carex lepidocarpa*. In the less humid zones of the bog Cross-leaved heath is found (*Erica tetralix*) along with *Sphagnums*: *Sphagnum capillifolium* and *S. papillosum*. Moreover, there are four species of willows. On the other hand, the presence of the bryophyte *Aulacommium palustre* reflects the cattle use in the bog surroundings, and with the repeated occurrence of fire, eutrophication and erosion of the humid zone. For these reasons, the project LIFE+ Naturaleza "Conservation of the Biodiversity in the River Miera" (LIFE 13/NAT/ES/899), proposes actions to improve the Bernallán bog. Access to the bog by cattle will be limited and plots will be selected inside and outside the enclosure, which after a characterization of the flora and fauna, as well of the physical-chemical conditions, will be assessed comparatively.

Keywords: cross-leaved heather; basin of the River Miera; fires; cattle; bog.

Introduction

The Bernallán bog is situated in the upper valley of the River Miera, in San Roque de Riomiera (Cantabria - Spain) included in the interior of the SCI Eastern Mountains. It is a flooded basin or flooded plain situated between two lateral glacial moraines and crossed by three small streams in the *Barranco del Carburco*. The geology of the bog, because of its glacial origins, is made up of clay deposits, silts and gravel from the Quaternary, lying upon in the sandstone from the Cretaceous period which give way to the Pas Valley in the east.

This bog is located at an altitude of 900 m, although it is surrounded by glacial moraines at 930 m and several crests and peaks, such as El Coterón – 1,250 m that form part of the chain called Montes de Valnera, crowned by the Castro Valnera (1,717 m). Near to the bog is an area occupied by hay meadows, when the humidity is low. This is enclosed by the stone-walls typical in this area - known as *pasiego* – which is also dotted with pastures and shelters used by cattle. Both the bog and its

Figure 1. General view of the Bernallán bog.

more rugged surroundings are used for grazing by cattle and horses during the summer months and in autumn.

Bernallán is part of the SCI Eastern Mountains, a natural landscape included in the Natura 2000 network, covering the summit area in the Eastern zone of Cantabria province, as well as the source and the high stretches of the main rivers: Pas, Pisueña, Miera and Asón.

Botanical characterization

This is a plateau where water accumulates, creating a peaty zone, or bog, situated at an altitude of around 900 m in the glacial valley of the River Miera (north side of the Lunada pass). The bog is surrounded by moraine type mountainsides lying on sandstone from the Cretaceous period which give way in the East to the valley of the River Pas. A stream has waterfalls on these rocks and it is surprising that some calciphile species have colonized them – for example *Globularia vulgaris, Hypericum nummularium* and *Hieracium mixtiforme*. All of these on the mountainside close to a beech wood.

Figure 2. *Epilobium duriae*
Author: Javier Goñi

The shade of the waterfall contains megaphorbic plants and other species from this habitat are: *Adenostylis alliaria*, *Cardamine raphanifolia* (Greater cuckooflower), *Scrophularia alpina*, *Veronica ponae*, *Lamiastrum galeobdolon* (Yellow Archangel), *Chaerophyllum hirsutum* (Hairy chervil), *Valeriana montana*, *Saxifraga hirsuta* (Kidney saxifrage), *Epilobium duriaei* and *Alchemilla clatula*.

Also noteworthy is the the grass with awns *Avenula sulcata* (Bristle grass). Its relative *Helictotrichon cantabricum* lives in calcareous crags lower down. Among the ferns, it is worth mentioning *Polystichum aculeatum* (Hard shield fern).

On the slopes close to the bog are found: *Arnica montana*, *Anemone nemorosa*, *Aquilegia vulgaris*, *Thalictrum aquilegifolium*, *Hypericum humifusum, Hypericum pulchrum, Polygonatum verticillatum*, *Scilla lillio-hyacinthus* and *Scilla verna*, among others.

The Eagle fern and the Western gorse bush (*Ulex gallii*) extends into some zones, whilst on the sandstone on the plateau live Male ferns (*Dryopterix filix-max*), *Polypodium* including the Common polypody (*Polypodium vulgare*), Foxgloves (*Digitalis purpurea*) and salsifies, Viper grass (*Scorzonera humilis*).

The bog has a zone dominated by herbaceous plants whose highlights are the Ciperaceae, above all the long white hairs of the fruits of the Common cotton-grass (*Eriophorum angustifolium*), triangular at the top of the stem and rounded below. This species is accompanied by *Tricophorum cespitosum* (Deergrass - turfing reed with long pod) and at least eight species of *Carex*: *Carex lepidocarpa* (Bottle sedge - *Carex* with pendulous utricle), an indicator of neutro-basophile bogs; *Carex demissa* (Common yellow sedge), *Carex hostiana* (glumes with scaly margins), *Carex rostrata* (Beaked sedge), *Carex binervis* (with 4 cm long male spike), *Carex leporina* (sedge with similar broad spikes), *Carex echinata* (Star sedge) and *Carex pulicaris* (Flea sedge – sedge with single terminal ear).

The group of rushes, Juncaceae, is diverse and contain *Juncus squarrosus*, *J. acutiflorus*, *J. bulbosus* and *Luzula nutans*.

Figure 3. *Tricophorum caespitosum*
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Figure 4. *Eriophorum angustifolium*
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Figure 5. *Carex lepidocarpa*

In some sections the Common cotton-grass with its whitish colour contrast with the reed beds and their brownish inflorescence, mainly the Compact reeds with continuous pith (*Juncus conglomeratus* and *J. effusus*), neighbours in zones that are less flooded. The Toad rush - an annual - (*Juncus bufonius*) colonizes the sides of the nearest road.

Other species in this hydrophilous bog are *Dactylorhiza maculata*, *Gymnadenia conopsea* and *Narthecium ossifragum* (Bog asphodel), scarce in this zone, flowering in July its flattened basal leaves are coupled. The grasses include *Glyceria fluitans* (Floating sweet-grass), *Molinia caerulea* (Purple moor grass), *Anthoxantum odoratum*, *Agrostis curtisii* (Bristle bent), *Holcus lanatus*, *Cynosurus cristatus*, *Briza media* (Quaking grass), *Festuca laevigata*. Whilst Matgrass *Nardus stricta* is easier to find on the moraine slopes above the bog.

Yellow colour in the landscape is provided by *Senecio aquaticus* (Water ragwort), *Crepis paludosa*, *Caltha palustris*, *Ranunculus flammula*, *Potentilla erecta* (Common Tormentile), with its four petals, but its stem and leaves turn reddish.

Figure 6. *Caltha palustris*
Author: Javier Goñi

In this community, other colours of flowers accompany the species mentioned above: *Filipendula ulmaria* (Meadowsweet), *Astrantia major*, *Myosotis lamottiana* (Forget-me-not), *Cardamine flexuosa*, *Carum verticillatum*, *Valeriana dioica*, *Galium palustre* and *Parnassia palustris*, a late flowerer with its petiolated base leaves and its embracing leaf.

Horsetails are frequent, above all those with obtuse sporiferous cones: *Equisetum fluviatile*, with large central hollow stem, and *Equisetum palustre*, with a thin central hollow. Semi-parasitic plants such as *Pedicularis sylvatica*, are specialists in this bog, but its relative, the European rattle (*Rhinanthus major*) stays in the

Figure 7. *Potamogeton polygonifolius*
Author: Javier Goñi

Figure 8. *Primula farinosa*
Author: Javier Goñi

drier meadow-heathland. Insectivorous plants, such as Large-flowered butterwort (*Pinguicula grandiflora*), pick up the scarce nitrogen from the amino-acids of small arthropods that it traps with its sticky brilliant green leaves.

Primulaceae are in the zone, *Lysimachia nemorum* in wooded zones and the bog pimpernel (*Anagallis tenella*), the *Soldanella alpina* and the Spring primrose (*Primula integrifolia*), whilst *Primula farinosa* can be seen in seepages from sandstone rocks, in the zone of *Covalruyo*, situated at a higher altitude on the right side of the River Miera.

In the less humid zones of the bog the Cross-leaved heather stands out (*Erica tetralix*) which extends along another section, with its ciliate leaves arranged in levels. In this area its stems are no higher than 30 cm. Its relatives *Erica cinerea* and *Daboecia cantabrica* take advantage of the rockier sandstone zones whilst *Erica vagans* and the European blueberry (*Vaccinium myrtilus*) are in the shade of the waterfall and near streams.

Small alder trees can be found growing in parts of the bog and some willows, *Salix aurita* (Eared willow), *Salix atrocinerea* (Grey willow) and the ones with white hairs on the underside (*Salix caprea* and S. *cantabrica*), share the zone with some ash trees. The Bitter willow with narrow leaves (*Salix eleagnos*) is more typical in the willow woods along the main river, downstream, where there are also few Purple willows with opposed leaves (*Salix purpurea*). The bodies of water are colonized by *Potamogeton polygonifolius* and *Veronica becabunga*. The bog is surrounded by some rushes, *Juncus conglomeratus* and *J. effusus*, with Umbelliferae nearby: *Angelica major* and *Laserpitium latifolium*.

Some mosses found in the zone are *Sphagnum* species (*Sphagnum capillifolium* and *S. papillosum*), *Aulacommium palustre*, favoured by fires and grazing, *Calliergonella cuspidata*, typical in minerotrophic mires (fens) and *Brachytecium rivulare* on the edge of the streams (Infante & Heras, 2010).

Two samples of peat, which had a depth of over 50 cm, were taken and its analysis revealed two pH measurements: 5.88 and 6.37 and two conductivity measurements: of 68.3 μS/cm and 85.7 μS/cm, respectively. Based on these parameters and the flora observed, the peaty zone could be classified as 7130 (Blanket bog) and 7150 (Depressions on peat substrates of the *Rhynchosporion*) as regards to the Directive 92/43/CEE, with a zone dominated by herbaceous plants from the Cyperaceae family and another by bog heathers.

Above the bog but still in the Miera basin a relative of *Narthecium ossifragum* can be found, this is *Tofeldia canalyculata*, not located in Bernallán. Both of these, along with the carnivorous *Drosera rotundifolia*, are located in sandstone seepages in the zone of the *Covalruyo* viewpoint with the rock vegetation *Gypsophila repens* and the yellow flowered *Anthyllis vulneraria*, where the Black vanilla orchid *Nigritella gabassiana* is also seen. And in the province of Burgos, in the SCI Montes de Valnera, at an altitude of 1,400 m, the bog of *Los Cuetos* with its acid lagoons are dominated by the Cyperaceae, *E. vaginatum*, catalogued as a vulnerable species in Castilla y León region.

Problems and threats

The Bernallán bog is situated in an area of extensive grazing, where at the start of the 19th century cattle raising became the main activity in this *pasiega* district. It occupied pastures and shelters were built, and there is also high-altitude grazing. Initially the grazing was for sheep but later dairy cattle arrived and these were accompanied by horses and donkeys. Employment in the construction boom lowered the grazing pressure over the last 20 years, which gave rise to a higher level of intentional fires that were capable of progressing over a larger area. In the last few years the economic crisis has brought a return to cattle farming for young people, based mainly on beef cows. These breeds have different requirements to dairy cattle, which means a greater occupation of the natural pastures, putting even greater pressure on the peat zones and bogs located in the SCI Eastern Mountains.

The use of the area around the Bernallán bog by cattle explains to a large extent the threats that this habitat is being subjected to.

Incidence of fires

The *pasiega* zone and the Eastern Mountains are, together with the valleys of the Saja and Nansa in the Western mountains of Cantabria, the area with the highest level of fires and of burnt surface annually. These intentional fires in areas of scrubland are started with the aim of creating pasture; however, the timing of the fires was produced at the most dangerous moments of the year (anti-cyclonic conditions, presence of strong south winds and at night). These fires have affected the Bernallán bog at least once a year up to 2011, owing to the presence of pasture and heather scrubland - *Erica vagans* and *Ulex gallii*; and minimal pressure from cattle. In such a way that when fires occurred they affected the whole surface of the bog, causing the mineralization of the peat.

High cattle load

The transformation from dairy to beef cattle over the last decade brought an increase in the use of the natural grazing surrounding the Bernallán bog. This means that the occurrence of fires in the bog has dropped as there is an absence of combustible material. However, since 2011, the bog is frequented by a group of 40 to 50 cows from May to November. These cows have access to the whole of the humid zone which is connected to the neighbouring pastures in the *sierra* or with steeper slopes and plots of land with meadows and shelters. The use is not intensive, although constant, allowing the damage to the bog, such as compaction and peat erosion, to be observed; there is also the eutrophication of the zone because of the animal excrement, indicated by the presence of the bryophyte *Aulacomnium palustre*, typical in deteriorated plots.

Chemical fertilization

The increase in the cattle load in the Bernallán bog has brought the incorporation of pasture management methods whose results are damaging to the regulation of the Nitrogen cycle. So, every two years there is a chemical fertilization with N-P-K in the bog and surroundings. This increases the eutrophication of the bog even more.

Erosion of the bog

The Bernallán bog is situated on a bed of sedimentary materials deposited by a glacier in the Quaternary period. These unconsolidated materials are easily eroded, which together with the steep slopes means that small streams have formed, these penetrate the bog during the period of thaw or in heavy rains. The upper valley of the Miera, owing to its high level of deforestation of which the landscape of Bernallán is an example, possesses little capacity for water regulation. This erosion makes the main stream that cuts across the bog flow at a depth of more than one metre below the surface level, which can affect the continuance of the phreatic level of the bog and the bog erosion close to the stream.

Conservation measures

The Bernallán bog is one of the representatives of the peaty habitats (Martínez, 2009) in the SCI Eastern Mountains (ES1300002). This type of habitat accounts for just 0.35% of the habitats catalogued in the interior of the SCI. This relative scarcity together with the biodiversity it holds, leads to the proposal of conservation measures to palliate or at least reduce the threats detailed above (Jiménez-Alfaro & Díaz, 2013). To achieve this, the project LIFE+ Nature "Conservation of the Biodiversity in the River Miera" (LIFE 13/NAT/ES/899) or LIFE Miera, has put forward an action towards improving the peaty habitat in the interior of the Bernallán bog.

The River Miera basin, where the project is concentrated, has its source and upper reaches in a very rugged area, with steep slopes, where the flat zones are only on the valley floor, close to the lateral glacial moraines and the mountain passes. The few peaty zones and bogs are located in these areas and these coincide, to a large extent, with the areas used by beef cattle during the summer months. Owing to the high cattle load the bogs of the Miera are found in a state of deterioration, owing to erosion, chemical fertilization and cattle excrement. Not surprisingly, these are the only flat zones accessible in a motorized vehicle, as is the case of the Cotero Tejo bogs in the upper reaches of the River Tejuelo, between San Roque de Riomiera and Selaya. This means that these zones are much appreciated by cattle farmers and the consequent deterioration makes it difficult to implement management and conservation measures. The improvements set out within the LIFE Miera project have concentrated on the Bernallán bog, which as it is of a smaller size and does not have access by car, is used by a single farmer during the summer and autumn, and so the cattle load and the difficulties associated with reaching a management agreement are less.

The action consisted of first, carrying out a complete hydrological and botanical characterization, and so the flora and bryophytes present in the zone were catalogued. After formalizing an agreement with the owner of the cattle and the Town hall (Ayuntamiento de San Roque de Riomiera) responsible party as it is a publicly owned mountain (*Monte de Utilidad Pública*), the zone has been enclosed and the absence of cattle on 70% of the bog surface area is guaranteed, which also coincides with the parts that are most waterlogged; where *Eriophorum angustifolium*, *Dactylorhiza maculata*, *Narthecium ossifragum*, *Pinguicula grandiflora*, *Caltha palustris* and *Ranunculus flammula*; the sedges *Carex rostrata, Carex lepidocarpa* and the *Sphagnum* species *Sphagnum capillifolium* and *S. papillosum* grow and develop.

The evolution of the vegetation and of the bryophytes situated inside and outside the enclosure will be assessed throughout 2017 and 2018 with the aim of determining whether a reduction in the eutrophication of the bog is produced, as well as a reduction in the erosion and what exactly are the effects on the vegetation now that there is no grazing. To achieve this, plots inside and outside the enclosure have been selected and paired, which after a characterization of the plant species, the pH conditions, the conductivity and the depth of the peat, will allow an analysis to be carried out comparing the evolution of the different measurable parameters.

The results of this analysis will allow new conservation measures to be drawn up. These may include the transfer of specimens of interest, that is species of flora, that have a very limited distribution, and the opening of grazing to sheep within the zone closed to cattle.

The LIFE Miera aims to improve the conservation status of the habitats and the key species in the hydrographic area of the Miera within the sites of the Natura 2000 network: SCI Eastern Mountains, SAC River Miera, SAC Dunas del Puntal and Estuario del Miera, in the province of Cantabria, and SCI Montes de Valnera, in the province of Burgos. The project, that is being executed between 2014 and 2018, is coordinated by the Fundación Naturaleza y Hombre and has two associated beneficiaries: Medio Ambiente, Agua, Residuos y Energía de Cantabria, SL (MARE) and the Consejería de Universidades e Investigación, Medio Ambiente y Política Social from the Cantabrian Government, and has European Commission support, co-financing 50% of the budget. In addition to working to improve the peaty zones and bogs in the Miera basin, the natural forest cover is being recovered in the headwaters, along with the riverside vegetation habitats 91E0* (Alluvial forests with *Alnus glutinosa* and *Fraxinus excelsior*) and 92A0 (*Salix alba* and *Populus alba* galleries), eliminating species of invasive flora from the estuary, the river and the dunes in the basin. The river connectivity is also being recovered for the salmon (*Salmo salar*), the otter (*Lutra lutra*) and for the Coleoptera Stag beetle (*Lucanus cervus*) and the Odonata Damselfly (*Coenagrion mercuriale*).

References

Infante, M., Heras, P. 2010. Briófitos (musgos y hepáticas) de Cantabria: panorámica sobre su conocimiento y conservación. *Locustella* 7: 64-73.

Jiménez-Alfaro, B. Díaz, T.E. 2013. Diversidad y conservación de turberas y tremedales en la cordillera Cantábrica. *Locustella* 8.

Martínez, A. (coord). 2009. *7140 Mires de transición (tremedales). Bases ecológicas preliminares para la conservación de los tipos de hábitat de interés comunitario en España.* Madrid: Ministerio de Medio Ambiente, y Medio Rural y Marino.

05

Evaluation of the restoration of the Jauregiaroztegi wetland (Auritz/Burguete, Navarra): changes in the vegetation in the period 2011-2015

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Abstract

The Jauregiaroztegi wetland is made up of a depression mostly covered by meadows used by cattle. The vegetation is diverse and around 65% of the surface area of the 16 habitats detected is linked to the hydrological system. Outstanding, because of their interest or the area they occupy, are the waterlogged meadows (Habitat of Community Interest HCI 6410), mesohydrophile meadows, nanorush-beds of *Juncus bulbosus* (HCI 3110 - Oligotrophic waters containing very few minerals of sandy plains), communities of *Potamogeton densus* (HCI 3260 - Water courses of plain to montane levels), transition mires (HCI 7140), grasslands of *Filipendula ulmaria* (HCI 6430 - Hydrophilous tall herb fringe communities of plains and of the montane to alpine levels), in addition to several hydrophile or aquatic communities. In 2006 the damage by cattle was assessed: drainage, eutrophication, pressure on the vegetation and rarification of species because of the introduction of cattle and land consolidation. In 2011 a restoration project was developed, dams were installed in a stream, the pools and a meadow were enclosed for protection and the use by cattle was regulated. The objective was to restore the hydrological system and to avoid damage to the vegetation and flora of interest by the cattle. The changes in the vegetation were studied by mapping (scale 1:2000), monitoring of 9 permanent plots and carrying out observations and taking photographs before (2011) and after (2015) the restoration. The objectives of the restoration measures were achieved: water retention by the dams increased waterlogging, with the surrounding vegetation evolving into hydrophile or aquatic communities; the surface of HCI 3110, 3260, 6410 increased, *Glyceria* communities and reed beds of *Eleocharis palustris*. The enclosed pools favoured the development of the aquatic vegetation and eliminated the trampling damage from cattle. The peat habitats (7140) maintained their surface area and the populations of the flora of interest linked to them (*Triglochin palustris*, *Menyanthes trifoliata*) seem to be stable. The eutrophication problems detected in 2015, and the temporary erosion, were tackled in 2016 with new restoration actions that should be assessed in the future.

Keywords: mapping the current vegetation; monitoring of habitats; restoration of habitats; wetlands; conservation of biodiversity.

Introduction

According to the Red List of European habitats, peat zones show the greatest number of different types among the terrestrial and fresh-water groups of threatened habitats. The threats that most frequently affect them are modifications to the hydrological system, pollution, succession and agricultural intensification (Janssen *et al.,* 2016). Most of these habitats are distributed in the Northern (Nordic countries) and Atlantic regions (British Isles); in the north of the Iberian Peninsula they are present in a fragmentary mode, survivors of the Ice Age, and are a refuge for species of relict flora and fauna (Janssen *et al.,* 2016; Montanarella *et al.,* 2006).

Given the regression that these habitats are undergoing in Europe, numerous projects have been set up over the last two decades to restore the bogs (Raeymaekers, 1999; Schumann & Joosten, 2008).

The main bogs in the north of Navarre were studied by Heras *et al.* (2006, 2010-2011) to discover their characteristics, evaluate their conservation status and to propose management and restoration measures. One of those is the Jauregiaroztegi wetland, located in Auritz/Burguete, where the threats detected were related to its uses for agriculture and farming: excavation of channels and springs to drain sowed fields in the surroundings and meadows, eutrophication after being fertilized, damage to the vegetative cover and the introduction of rare species for pasture or from the transit of cattle, as well as land consolidation. The effect of draining affected, above all, the peat vegetation, aquatic and adapted to waterlogged ground, whilst the vegetation as a whole suffered the effect of grazing, transit and cattle excrement.

Heras *et al.* (2006) proposed several restoration measures and these were carried out in 2011 (Zaldua, 2011). With the aim of assessing their effectiveness, the situation of the habitats in the zone was diagnosed beforehand via cartography, inventories, the establishment of permanent plots and observations (León, 2004, 2011; Peralta, 2011).

The objective of this article is to evaluate the restoration actions carried out by analyzing the changes that were produced in the period between 2011-2015. The questions arising are: a) Has the surface area or the distribution of the habitats changed?; b) Has the number of species or the plant composition in the permanent plots changed?; c) Have changes in the flora of interest been observed?; d) Have the objectives of the restoration been achieved?

Materials and methods

The sections below describe the area studied, the restoration carried out and the methods used to evaluate their effectiveness. The nomenclature of the vascular plants follows the criteria of Aizpuru *et al.* (1999) and Castroviejo (1986-2015), and that of the vegetative communities of Peralta *et al.* (2013).

Study area

The Jauregiaroztegi wetland is located in the North of Navarre, in the borough of Auritz/Burguete, and is part of the Special Area of Conservation "Irati, Urrobi and Erro River Systems" in Natura 2000. It is located at an altitude of around 880 m, at montane level, with lower hyper-humid ombrotype. From the bio-geographfical point of view it is located at the Eastern extreme of the Basque-Cantabrian sector, very close to the Central Pyrennean sector, both being in the Eurosiberian region (Loidi & Báscones, 2006).

The zone is made up of a small depression of 11.6 ha occupied by various streams that flow between meadows with more or less dispersed hedges and some isolated trees. To the North it reaches fields of maize, potato or artificial meadows, and to the South there are beech woods. The geological materials are made up of deposits from the Quaternary period with gravels, sand and silt (VV.AA., 2012).

The most genuine peat vegetation is found in the east of the wetland, in an area of around 0.1 ha where peat deposits of up to 1.5 m have been recorded (Del Valle *et al.,* 2015). In this area springs up a small stream which joins another to the South, where two springs from the North discharge. Associated to these streams are aquatic and wetland vegetation. The wetland is subneutrophile because the origin of the water is minerogenous (soligenous). Its trophic state is mesotrophic to eutrophic and can be attributed to the input of fertilizer or of cattle manure (Heras *et al.,* 2006). The wetland is host to species of interest because of their rarity in Navarre, most noteworthy being *Triglochin palustris* and *Menyanthes trifoliata* (Heras *et al.,* 2011).

In spite of its small size, the vegetation is diverse and 16 types of habitat and 7 uses have been recognized (Table 1). Around 75 % of the surface area are meadows, almost all of them natural or semi-natural. The mesohydrophiles and the waterlogged meadows cover 45 % of the zone, which together with 20 % of the surface being hydrophyte or aquatic habitats, means that almost 65 % of the vegetative cover is closely linked to the hydrological system.

The habitats of community interest (HCI) according to the Habitats Directive occupy 24 % of the surface, almost all corresponding to waterlogged meadows (HCI 6410 - *Molinia* meadows on calcareous, peaty or clayey-silt-laden soils). The other four habitats of interest are the large areas of rushes *Juncus bulbosus* (HCI 3110 - Oligotrophic waters containing very few minerals of sandy plains), the community of *Potamogeton densus* (HCI 3260 - Water courses of plain to montane levels), the amphibious communities of *Potamogeton polygonifolius* and those of *J. bulbosus* and *Anagallis tenella* (HCI 7140 - Transition mires), as well as the pastures of *Filipendula ulmaria* (HCI 6430 - Hydrophilous tall herb fringe communities of plains and of the montane to alpine levels; Table 2).

Restoration measures

Heras *et al.* (2006) proposed the restoration of at least the zone with bog vegetation, where the population of *T. palustris* accompanied by *M. trifoliata* is found, as well as a monitoring plan so as to prevent further deterioration through drying or loss of species of interest. They considered necessary to stop further drainage, to regulate cattle use and to recover the natural hydrological system by increasing its surface

HABITATS AND USES	ha 2015	% 2015	% 2011	% 2015/ 2011
AL: Free water	0.04	0.3	0.1	2.5
ALa: Free water with filamentous green algae	0.007	0.1	-	-
arb: Ash plantations	0.10	0.9	0.9	1.0
Cat: Floating sweet-grass - with *Catabrosa aquatica* (*Glycerio-Catabrosetum aquaticae*)	0.38	3.3	2.3	1.4
Cc1: Mesophile meadows (*Merendero-Cynosuretum cristati*)	2.27	19.5	20.2	1.0
Cc4: Mesohygrophitic meadows (*Merendero-Cynosuretum cristati* var. hidrophile)	2.49	21.4	23.9	0.9
Ch: Herbaceous crops	0.14	1.2	-	-
Ele: Rush-beds with *Eleocharis palustres* (*Glycerio-Eleocharitetum palustris*)	0.02	0.2	0.1	1.7
Fil: Pastures of *Filipendula ulmaria* (*Ranunculo-Filipenduletum ulmariae*) [6430]	0.08	0.7	1.8	0.4
Fs: isolated beech trees	0.07	0.6	0.6	1.0
GlyA: Floating sweet-grass with *Glyceria fluitans* (*Caro-Glycerietum fluitantis*)	0.47	4.1	2.5	1.6
Iri: Floating sweet-grass with *Iris pseudacorus* (*Glycerio-Catabrosetum aquaticae* var. with *Iris pseudacorus*)	0.17	1.4	1.4	1.0
JbCs: Nanorush-bed of *Juncus bulbosus* [3110]	0.007	0.1	0.0	72.0
Jbu: Com. of *Juncus bulbosus* & *Anagallis tenella* (*Anagallido-Juncetum bulbosi*) [7140]	0.10	0.8	0.8	1.0
Jin: Nitrophile rush-bed (*Mentho-Juncetum inflexi*)	0.001	0.01	0.01	1.2
Mol: Waterlogged meadows (com. of *Deschampsia cespitosa* & *Sanguisorba officinalis*) [6410]	2.64	22.7	21.4	1.1
Pde: Com. of *Potamogeton densus* (*Ranunculo-Groenlandietum densae*) [3260]	0.005	0.04	0.01	4.2
Pot: Amphibious com. of *Potamogeton polygonifolius* (*Hyperico-Potametum oblongi*) [7140]	0.001	0.01	0.01	1.0
Pr: Artificial meadow	0.64	5.5	6.5	0.8
Rub: Thorny plants (*Rhamno-Crataegetum laevigatae*)	1.05	9.0	9.2	1.0
Sal: Bushy willows (com. of *Salix atrocinerea* & *S. lambertiana*)	0.84	7.2	7.2	1.0
Spa: Community of *Sparganium erectum*	0.12	1.0	1.2	0.9
TE: Eroded terrain	0.003	0.02	-	-
	11.63	100	100	

Table 1. Surface and percentage of habitats and uses in 2011 & 2015.

DESCRIPTION	ha 2015	% 2015	% 2011	% 2015/ 2011
0000 Habitat not included in Annex I of the Directive 92/43/CEE	8.80	75.7	76.0	1.0
3110 Oligotrophic waters with a very low content of minerals from Sandy plains (*Littorelletalia uniflorae*)	0.01	0.1	0.0	72.0
3260 Water courses of plains to montane level with vegetation *Ranunculion fluitantis* & *Callitricho-Batrachion*	0.01	0.0	0.0	4.2
6410 Molinia meadows on calcareous, peaty or clay-silt substrates (*Molinion caeruleae*)	2.64	22.7	21.4	1.06
6430 Hygrophilous tall herb fringe communities, of plains and of the montane to alpine levels	0.08	0.7	1.8	0.4
7140 Transition mires	0.10	0.8	0.8	1.0
	11.63	100	100	

Table 2. Surface and percentage of habitats of interest and priority.

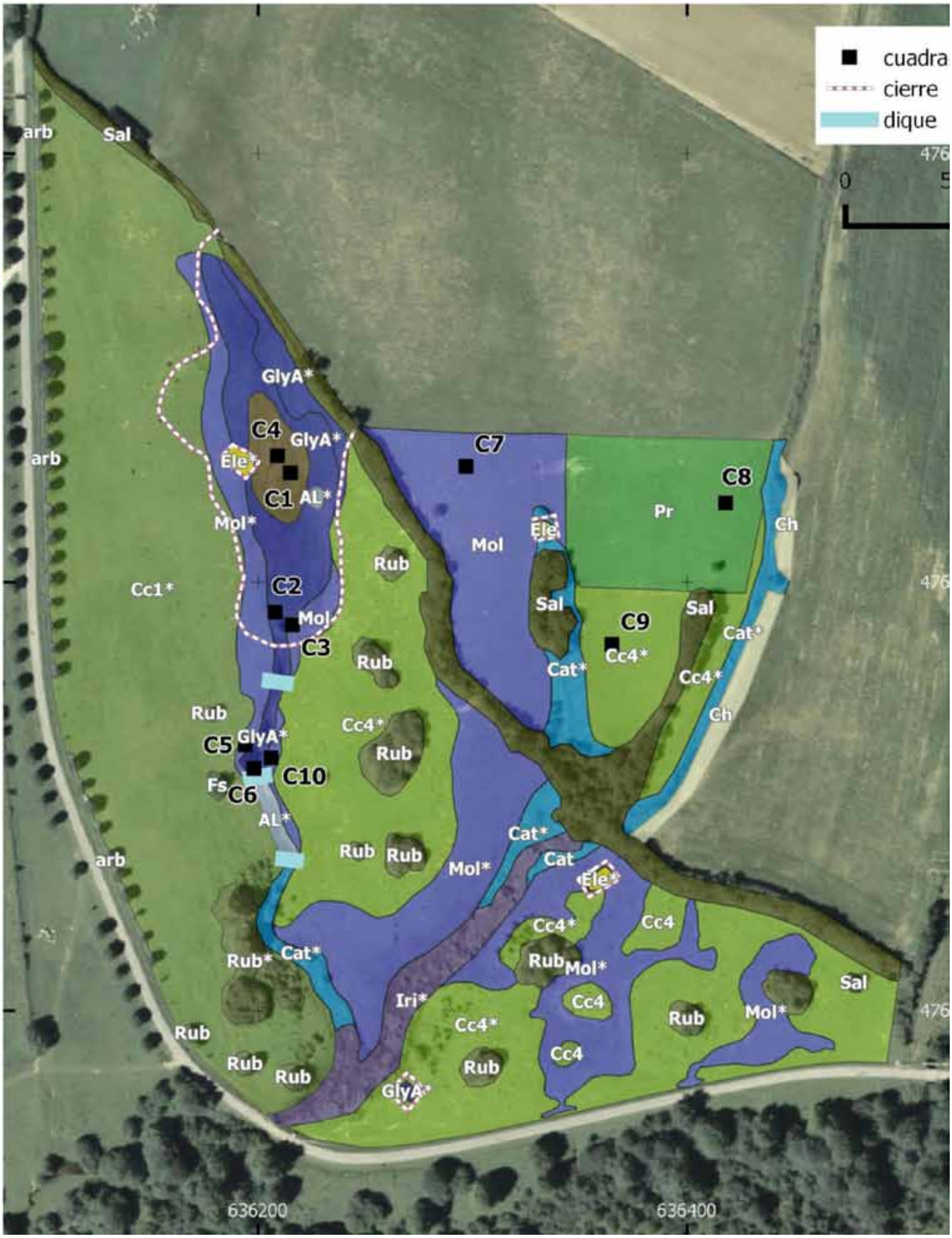

Figure 1. Map of habitats in 2015 and situation of permanent plots and dams. The habitats and uses correspond to the abbreviations in Table 1; when they are followed by an asterisk it means that there is another habitat inside in addition to the main one. Datum ED50.

area and so lengthen the duration of its waterlogging. In 2011 several actions were carried out to improve the hydrological system and to control the use of the zone by cattle (Fig. 1); at the same time there were improvements to the equipment of public use (Zaldua, 2011).

The works of hydrological correction consisted of constructing three dams that slowed down the exit of water from the wetland and which raised the phreatic level in such a way that areas with different degrees of humidity were created. The dams were made of thick tongued and grooved wooden planks (7-9 cm), placed transversally to the current; these were reinforced and fixed to the ground with wooden posts and they were covered with clay to improve their impermeability (Fig. 2). They were located in the western branches of the stream after a detailed topographical survey to assess the possible zones of flooding (León, 2011; Zaldua, 2011).

Figure 2. Lower dam.

Figure 3. Water retained by the intermediate dam.

With respect to the works related to the control of cattle, four wooden enclosures were installed around the pools (Fig. 3) and another enclosure surrounding the meadow situated to the north of the wetland. The aim of the enclosure of the pools was to improve the state of conservation of the banks, seriously damaged by the trampling of the cattle. Enclosing the meadow to the north of the hedge was put forward so as to regulate its use according to the phenology of the species of flora of interest present there, especially of *Sanguisorba officinalis*.

These actions were accompanied by a proposal for cattle management agreed with the landowner (City Council of Auritz/Burguete - Ayuntamiento de Auritz/Burguete) and the farmer renting the pastures.

Evaluation of the restoration measures

The situation in 2011 and 2015 was compared to assess the effectiveness of the restoration actions, using the following data obtained in both years: cartography of the habitats, vegetation in the ten permanent plots (Fig. 1) and observations of the flora and vegetation in the zone.

The cartography from 2015 was compared to that from 2011 to quantify the changes in the distribution and surface area of the habitats. The habitats contained in each plot, the occupation percentage of each habitat or use, the conservation status (favourable, inadequate, bad, not evaluated) and the damage observed were all identified. Both maps were made at a scale of 1:2000 on ortoPhoto (DOPTC 2011-2014) with the geographical information systems MiraMon (Pons, 2011) and QGIS (QGIS Development Team, 2011).

During the field research observations were made and photographs of the vegetation were taken in the same places/points as in 2011 to help with the cartography, as well as at other additional places. At the same time any indicator of changes in the zone was noted: signs of grazing, erosion, flooding, etc.

The following data was obtained from each plot in 2011 and 2015 so that its evolution could be studied: cover of each species and the vegetation, number of species and maximum height of the vegetation. Besides plots C1 and C4, specimens of *T. palustris* were counted and observations of the presence of *M. trifoliata* were made.

Results

In Fig. 1 the distribution of the habitats in 2015 is shown and in Figs. 4-11 the habitats that had undergone changes to their cover and distribution between 2011 and 2015. The surface area of the habitats and uses in both years and the magnitude of the change are compiled in Tables 1 and 2. The state of conservation of the habitats in 2015 is favourable in general terms, excepting some specific places where water eutrophication or erosion, was detected, both related to the transit of cattle.

The effect of the dams is evident as there was free water in the sections of the stream where they were installed, upstream from all the dams and downstream from the most Southern one (Figs. 2, 3 and 12); before the installation of the dams free water was only seen in the pools (Figs. 4 and 5). The water in many places contains

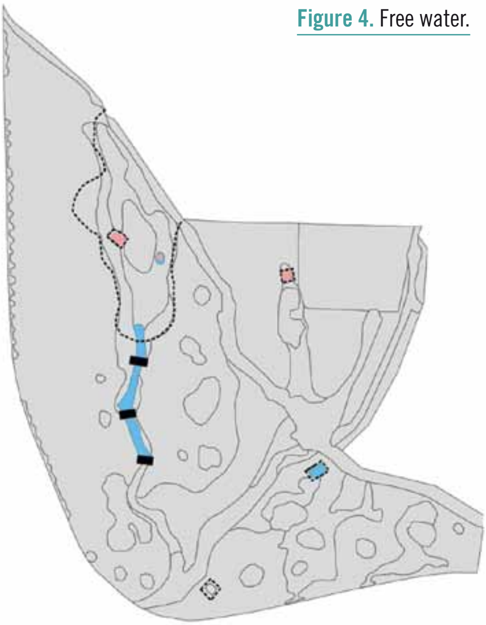

Figure 4. Free water.

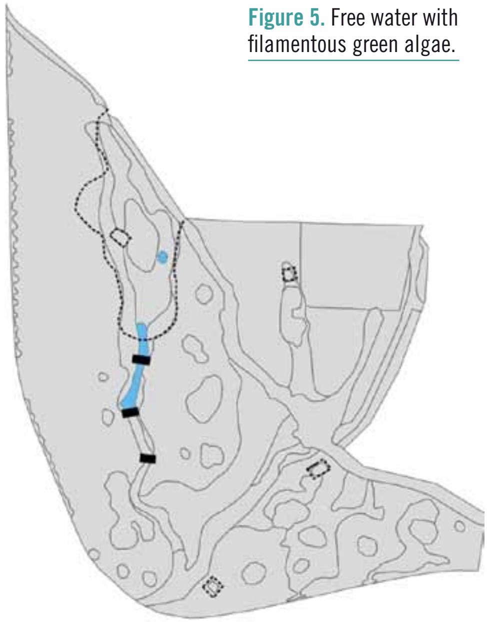

Figure 5. Free water with filamentous green algae.

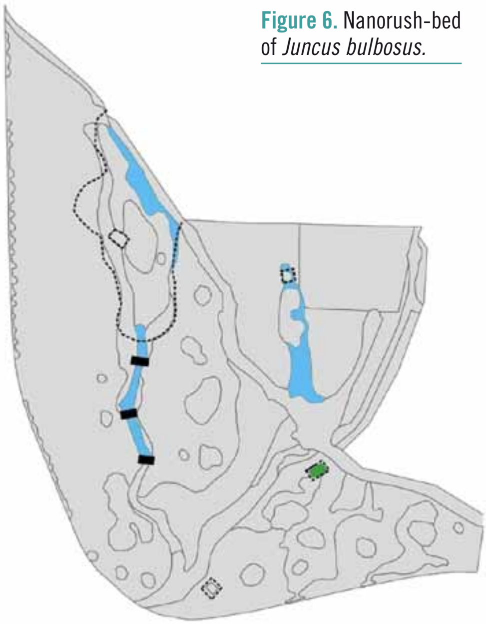

Figure 6. Nanorush-bed of *Juncus bulbosus.*

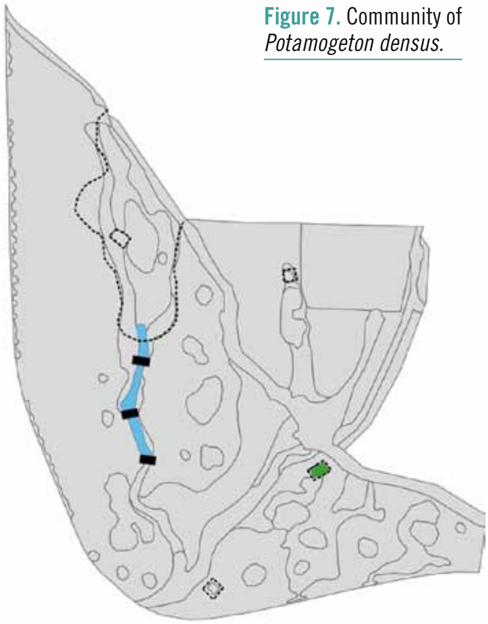

Figure 7. Community of *Potamogeton densus.*

Figures 4-7. Changes in the cover and distribution of the habitats. Restoration actions: enclosures dotted lines; dams black rectangles. Changes in cover: blue 2015 > 2011; green 2015 = 2011; pink 2015 < 2011.

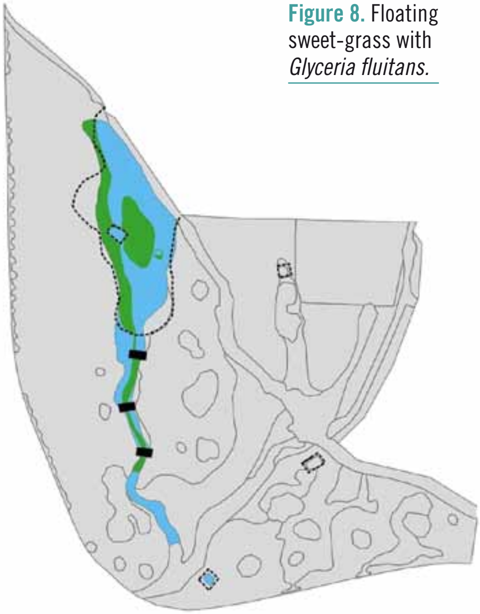

Figure 8. Floating sweet-grass with *Glyceria fluitans.*

Figure 9. Waterlogged meadows.

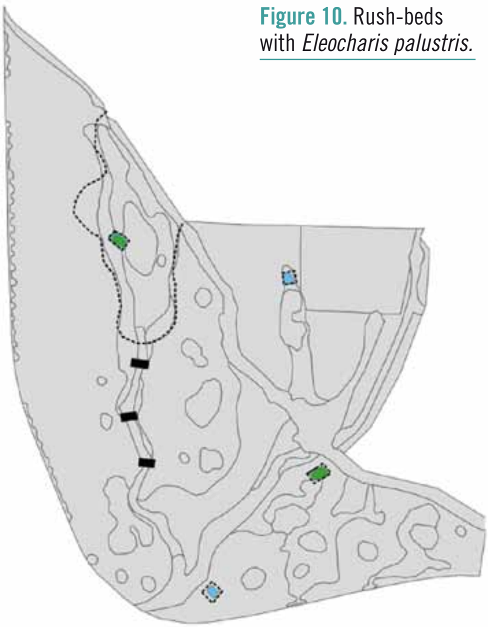

Figure 10. Rush-beds with *Eleocharis palustris.*

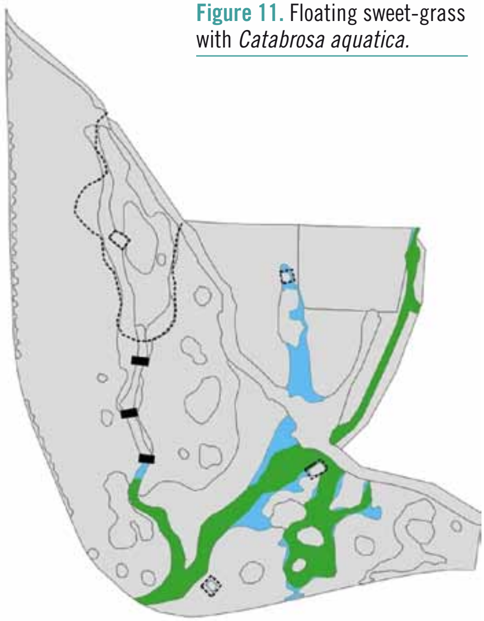

Figure 11. Floating sweet-grass with *Catabrosa aquatica.*

Figures 8-11. Changes in cover and distribution of the habitats. Restoration actions: enclosures dotted lines; dams black rectangles. Changes in cover: blue 2015 > 2011; green 2015 = 2011; pink 2015 < 2011.

green filamentous algae, which together with the presence of *Potamogeton densus* seems to indicate a certain eutrophication. In 2011 green algae were also seen in the pool with shallow water and not enclosed, to the north of the zone.

The peat habitats (HCI 7140), the amphibious community of *Potamogeton polygonifolius* and the *Juncus bulbosus* and *Anagallis tenella*, maintained their surface area. The plots situated on the spring (C1, C4) with *M. trifoliata* and *T. palustris* have not undergone any significant changes and their populations in the zone seem stable, in agreement with the observations carried out.

The nanorush-beds of *J. bulbosus* (HCI 3110) and the community of *P. densus* (HCI 3260) were the habitats that underwent the biggest increase in relative surface area compared to 2011, when they were only found in the enclosed pool in the East of the zone (Figs. 7 and 8; Tables 1 and 2); the nanorush-beds were also seen in the

Figure 12. Pool enclosure.

Figure 13. Horse grazing *Glyceria* in the tail of the lower dam.

stream that is to the north of the hedge, outside of the area of influence of the dams.

Floating sweet-grass *Glyceria fluitans* have increased its surface area around the main spring, the sides of the stream where the dams have been installed and downstream from them (Fig. 8). This situation is also reflected in the plots closest to the upper (C2, C3) and central dams (C5, C6) which showed an evolution from waterlogged meadows to floating sweet-grass, *G. fluitans* (Fig. 12 and 13). In those that are more distant (C3, C10) the change was smaller, although they also incorporated species indicative of waterlogging.

The waterlogged meadows (HCI 6410) extended their surface area to the sides of the two streams that cut across the zone. At least their extension in this area and around the dams seems to be attributable to their installation and it cannot be ruled out that the rise in the phreatic level around these dams has facilitated waterlogging more to the South (Fig. 9). Runoff from the adjacent slopes could also have an influence, in the North of the zone, depending not only on the climatic conditions but also on the land use in the agricultural terrains that occupy these slopes. In the new zones where these meadows appear, a large colony of *Juncus acutiflorus* developed, which was not observed in 2011. In most of the zones where the surface area decreased this was due to displacement by more hygrophilic communities or free water, as it can be seen in Figs. 4, 5, 8 and 11.

After the installation of the enclosures at the pools there was a large development of the vegetation, especially of the Rush *Eleocharis palustris*. These rush-beds were only found in two pools in 2011, but in 2015 they were present in the four enclosures (Fig. 10).

The increase in the surface area of the floating sweet-grass with *Catabrosa aquatica* does not seem to be linked to the actions carried out. This was produced above all in the stream that separates the meadows to the north of the hedge, where it occupied areas where previously the communities of *Sparganium erectum* and

Figure 14. Plot 6, 22/6/2011.

Figure 15. Plot 6, 19/6/2015.

the pastures of *Filipendula vulgaris* were found (Fig. 11). This zone seems to be frequented by cattle, which cross the stream to reach a meadow on the other side, and where they presumably drink. It is possible that the change in the vegetation is related to a greater use of these meadows by cattle, as is suggested by the floristic change in plots C7 and C9. In both the range of species falls, with *Sanguisorba officinalis* (C7) disappearing or decreasing in cover (C9), as well as an increase in the cover of meadow plants such as *Cynosurus cristatus* and *Agrostis capillaris*.

On the other hand, the meadow in the north-east extreme was an artificial meadow in 2011 and in that same year it was transformed into a crop of potatoes. The floristic composition of the plot C8 seems to indicate that it has been resown with herbaceous plants and its richness in species has fallen compared to 2011, possibly as a consequence of ploughing and cropping.

Another change observed in the wetland was the presence of a few very eroded zones that are frequented by cattle upstream from the upper dam and next to the central dam. In the former this is a zone crossed by cattle going to the meadows in the North of the zone. In the second the erosion seems to originate because of the passage of vehicles towards the meadow sown and cropped in 2011.

From the point of view of the conservation of the habitats of interest, their surface area increased slightly compared to 2011, going from 76 % to 78 % (Table 2), above all due to the increase in the area of the waterlogged meadows (HCI 6410).

Conclusions

Responding to the questions raised in the introduction, it can be concluded that both the surface area as well as the distribution of the habitats have changed, and that it seems that at least a part of these changes are the result of the restoration actions.

Water retention produced by the installation of the three dams increased the waterlogging around the stream in the west of the zone, giving rise to an evolution of the nearby vegetation to more aquatic and hidrophyte communities and a consequent increase in the surface area of these habitats: nanorush-beds of *Juncus bulbosus* (HCI 3110), communities of *P. densus* (HCI 3260), Floating sweet-grass with *Catabrosa aquatica* and *G. fluitans*, rush-beds with *E. palustris* and waterlogged meadows (HCI 6410), as well as free water, which on occasions contained colonies of green algae. The enclosure of the pools seem to have favoured the development of aquatic vegetation, above all communities of helophytes, and the effects of trampling on the banks have disappeared.

Regarding the changes in the permanent plots, their richness in species as well as their floristic composition changed. The plots closest to the dams (C2, C3, C5, C6) underwent a notable change owing to the waterlogging and absence of trampling by cattle, showing an evolution from waterlogged meadows to floating sweet-grass *G. fluitans*. In those a little further away (C3, C10) the change was smaller, but they also incorporated species indicative of waterlogging. The plots situated on the upwellings (C1, C4) with *M. trifoliata* and *T. palustris* did not undergo any significant changes. In two of the three plots situated on the meadows in the north of the zone (C7, C9) a higher level of pressure from cattle could be seen, with a lower height in the vegetation and reduction in the richness of species.

As for the the flora of interest, the populations of *T. palustris* and those of *M. trifoliata* seem to be stable. In addition new species in the wetland, rare in Navarre such as *Carex hostiana,* have been detected.

The objectives sought by the restoration actions have been achieved, given that the surface area has been maintained, or has increased, in the zones with habitats of community interest, especially those of 6410 and 7140. Their state of conservation is favourable and negative trends in their characteristic species were not detected. The hydrological functioning of the wetland is nearing the situation before the drainage began, as can be seen from the higher level of waterlogging. The recognition of the wetland guarantees its protection against proposals for new drainage projects.

The damage from cattle trampling on the banks of the pools has disappeared, but the pressure from cattle seems to have increased in the meadow in the northern zone. This situation must be tackled with the parties concerned, trying to adapt the cattle farming to the conservation objectives within the proposed management program.

Eutrophication problems and temporary erosion next to the stream where the dams were installed were tackled in 2016, reacting to the use of the zone by cattle. An electric enclosure was installed around the stream to control the specific period of its use and dry access for the cattle to other points of the wetland was constructed (Zaldua, 2016). These actions must be assessed and monitored in 2017. The habitats and the flora of interest must also be monitored periodically so as to know how they are evolving in relation to the actions carried out.

Acknowledgements

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References

Aizpuru, I, Aseginolaza, C., Uribe-Echebarria, P. M., Zorrakin. I. 1999. *Claves ilustradas de la flora del País Vasco y territorios limítrofes*. Gobierno Vasco. Vitoria-Gasteiz.

Castroviejo, S. (coord.). 1986-2015. *Flora Iberica. Plantas vasculares de la Península Ibérica e Islas Baleares*. Real Jardín Botánico, CSIC. Madrid.

Clavería, V., Berastegi, A. 2015. *Especies de flora amenazada y de interés presentes en los enclaves incluidos en el proyecto LIFE Tremedal (LIFE11/NA/ES/707)*. Informe técnico. Gestión Ambiental de Navarra S.A. y Gobierno de Navarra. Pamplona.

Del Valle 2013. *Resumen de las actividades desarrolladas por el negociado de suelos y climatología en su colaboración al proyecto LIFE Tremedal*. Informe Técnico. Departamento de Desarrollo Rural, Medio Ambiente y Alimentación. Gobierno de Navarra. Pamplona.

DOPTC. 2014. Ortofotomapas 1:5000, año 2014 y anteriores. Dirección General de Obras Públicas. Gobierno de Navarra. Pamplona.

Heras, F.T., Infante, M., Biurrun, I., Campos, J.A. 2006. *Cartografía y Bases Técnicas para la Gestión de Turberas*. Informe técnico. Gestión Ambiental Viveros y Repoblaciones de Navarra S.A. Pamplona.

Heras, F.T., Infante, M., Biurrun, I., Campos, J.A., Berastegi, A. 2010-2011. Tipología, vegetación y estado de conservación de los hábitats hidroturbosos del noroeste de Navarra. *Acta Botanica Barcinonensia* 53: 27-45.

Heras, F.T., Infante, M., Biurrun, I., Campos, J.A., Berastegi, A. 2011. Flora de los hábitats hidroturbosos del noroeste de Navarra. *Actes del IX Col·loqui Internacional de Botànica Pirenaico-Cantàbrica*: 191-200. Ordino, Andorra.

Janssen, J.A.M. 2016. *European Red List of habitats. Part 2. Terrestrial and freshwater habitats*. Publications Office of the European Union. Luxembourg.

León, J.M. 2004. *Evaluación del impacto ambiental y medidas de corrección del drenaje de la pradera de Jauregiaroztegi en Auritz/Burguete, Navarra*. Informe técnico. Ayuntamiento de Auritz/Burguete.

León, J.M. 2011. *Estudio hidrológico e hidrogeológico del humedal de Jauregiaroztegi, Auritz-Burguete (Navarra)*. Informe técnico. Gestión Ambiental de Navarra S.A. y Gobierno de Navarra. Pamplona.

Lorda, M. 2001. Flora del Pirineo navarro. *Guineana* 7: 1-557.

Montanarella, L., Jones, R.J.A., Hiederer, R. 2006. The distribution of peatland in Europe. *Mires and Peat*.

Peralta, J. 2011. *Diagnóstico del estadio cero de la vegetación y los hábitats en el proyec-*

to de restauración y valorización del humedal de Jauregiaroztegi (Auritz/Burguete). Informe Técnico. Gestión Ambiental, Viveros y Repoblaciones de Navarra S.A. Pamplona.

Peralta, J., Biurrun, I., García-Mijangos, I., Remón, J.L., Olano, J.L., Lorda, M., Loidi, J., Campos, J.A. 2013. *Manual de hábitats de Navarra*. Gestión Ambiental, Viveros y Repoblaciones de Navarra S.A. y Gobierno de Navarra. Pamplona.

Pons, X. 2011. SIG MiraMon v. 7. CREAF. Barcelona.

QGIS development team. 2011. QGIS Geographic Information System. Open Source Geospatial.

Schumann, M., Joosten, H. 2008. *Global peatland restoration manual*. Institute of Botany and Landscape Ecology, Greifswald University. Germany.

VV.AA. 2012. Mapa Geológico 1:25.000 de Navarra. Gobierno de Navarra. http://geologia.navarra.es/

Zaldua, A. 2011. *Proyecto de gestión restauración del humedal de Jauregiaroztegi (Auritz/Burguete). Aplicación de medidas para la conservación de los Hábitats de Interés Comunitario "7140 "Mires" de transición" y "6410 Prados con molinias sobre sustratos calcáreos, turbosos o arcillo-limónicos (Molinion caeruleae)" y de flora de interés. Proyecto Técnico. LIFE Tremedal (LIFE11 NAT/ES/707)*. Gestión Ambiental de Navarra S.A. Pamplona.

Zaldua, A. 2016. *Memoria valorada de las actuaciones de conservación de flora (Spiranthes aestivalis, Eleocharis palustris, Carex hostiana y Ranunculus lingua) y hábitats (HIC/HP 7140, 6410, 4020* y 7120*) amenazados ligados a humedales en Navarra. Convenio la Caixa-Gobierno de Navarra 2016*. Gestión Ambiental de Navarra S.A. y Gobierno de Navarra. Pamplona.

06

Study of the evolution of the topography of the Comeya bog (Picos de Europa National Park) over time with a terrestrial laser scanner

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Abstract

This article shows the work carried out to measure the changes in the topography of a bog situated in the Picos de Europa (Asturias - Spain). The analysis was made by comparing three point clouds, measured with a terrestrial laser scanner, over a period of two years in two adjacent zones, one accessible to cattle and the other inaccessible. The methodology allowed the rapid and accurate reconstruction of the geometry of the terrain. The results obtained confirm the starting hypothesis, which stated that cattle modify the natural evolution of the bog.

Keywords: bog; laser scanner; point cloud; digital model of the terrain.

Introduction

This article is part of the Europen project LIFE+ Tremedal[1], one of whose objectives was to carry out an exact study of the evolution of bog topography in the National Park and SCI ES1200001 Picos de Europa (Asturias) to determine the influence of the animals that graze there, basically cows but also horses, on these zones of special ecological interest. The starting hypothesis is that cattle affects the natural evolution of the terrain from both their trampling action, which compresses the ground, and their grazing of new shoots, which conditions the normal development of the plants.

[1] LIFE 11 NAT ES 707 Inland Wetlands in the North of the Iberian Peninsula: Management and Restoration of Mires and Wet Environments (TREMEDAL).

To carry out the monitoring of the bog topography, a terrestrial laser scanner (TLS) was used. This equipment allows high density point clouds to be obtained in a short time and with a similar accuracy to those obtained with a medium-performance topographic station. The use of this type of equipment to observe and monitor the terrain has been quite usual since the year 2000. Rosser *et al.* (2005) employed a TLS to analyze the erosion of a cliff. Alba *et al.* (2006) also made use of this equipment to measure the deformation of a dam. Prokop & Panholzer (2009) used a TLS to detect small movements on mountainsides. Lagué *et al.* (2013) used this technology to measure changes in the sediments of the canyon of the River Rangitikei. Armesto *et al.* (2013) studied the stability of a granite boulders starting with the geometric data obtained via TLS. More recently, Hayakawa *et al.* (2016) studied the deformation of the terrain in a volcano over a three-year period, and they compared their results with others obtained from levelling.

In this work we present the details of all the tasks carried out to analyze the evolution over time of the bog, including data collection in the field, the processing of the observations and the comparison of the measurements made each year. The results are shown and analyzed.

Methodology

To compare the starting hypothesis, which assumes that the topography of the bog is affected by cattle, topographical surveys were made in two adjacent zones: one bordered by a fence that excluded cattle and another that is open and accessible to cattle. Therefore, the study of the two possible scenarios could be made independently. In September 2013, 2014 and 2015, topographical surveys were made via TLS. The evolution of the topography of the terrain for both zones was carried out by comparing the heights of the points measured over those three years.

The initial idea was to use conventional measuring equipment to carry out the topographical studies, specifically, total stations or GPS receivers that allow accurate coordinates of the terrain to be obtained, with an accuracy of one to three centimetres. However, the aim was to carry out a very accurate study, measuring points on the terrain a few centimetres apart, with the aim of lowering interpolation errors as much as possible. This is the reason why conventional equipment was ruled out and why a terrestrial laser scanner was decided on. The latter get huge clouds made up of millions of points with densities that can reach a few milimetres in very short times, just a few minutes, with an accuracy that is similar to the conventional topographical equipment mentioned above.

Data collection

Measuring equipment

Data collection in the field was made with a Riegel@ LMS-Z390i (Fig. 1) terrestrial laser scanner. This is a time of flight scanner whose main technical characteristics are shown in Table 1. This equipment therefore has an accuracy that is almost com-

Figure 1. Terrestrial Laser Scanner used for data collection.

Maximum measuring distance	400 m @ Laser Class
Maximum speed of measurement	11,000 pts/sec
Field of vision	80° x 360°
Accuracy	6 mm
Repeatability	4 mm (one ray), 2 mm (averaging)
Length of laser wave	near infrared
Speed of scanning	1 scan/s to 20 scan/s
Angular resolution of measurement	0.001°

Table 1. Technical data of the Riegel LMS-Z390i laser scanner.

parable to medium-performance topographical stations or global satellite positioning systems. However, the laser scanner can get point clouds with densities that are much higher than those of other equipment, which results in surface models of the terrain that have much better detail.

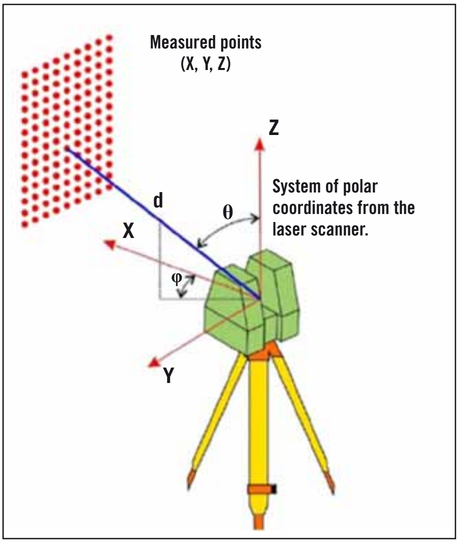


Figure 2. System of polar and cartesian coordinates from the laser scanner.

The scanner obtains the 3D coordinates for the points in polar coordinates, measuring the distance between the scanner and the points measured as well as the angles in the horizontal and vertical planes. The polar coordinates can be transformed to cartesian coordinates in a local system of axes (Fig. 2).

The measurement of the distance is made via the calculation of the time it takes an invisible laser pulse, near infra-red and emitted by a laser diode, to reach an object and be reflected back to the emitter. The frequency of the emission of the pulse can reach 11,000 points per second with this scanner.

The system of measurement for the angles used on the TLS is similar to the one on topographical stations. It is based on electro-optical techniques and has a coded glass circle with a pattern of alternating transparent and opaque areas. A beam of light passes through the glass, obtaining via a photodiode that converts the light energy into a digital signal, a numerical value of the angle in accordance with the turn made by the circle.

The terrestrial laser scanner contains a digital camera that moves at the same time as the TLS, allowing colour to be assigned to the points. The digital images can be superimposed onto triangular models of the terrain constructed based on the point clouds, hence building visual models of the objects measured with the geometrical accuracy that is conferred on them by the points observed.

Reference signals

In the first field campaign, in September 2013, before taking any measurements with the laser scanner, a set of 18 reflective signals for control were located on some of the fence-posts that prevented cattle from entering the bog (Fig. 3). These signals had to remain fixed during the whole period of study and it was posible to measure them with the scanner both inside and outside the fence, that is, there can be no objects that interfere with the laser beams that go from the scanner to the reflective signals. The objective of the signals is to obtain a common reference system for the coordinates of the points obtained with the scanner situated at different places, so that the heights of the point clouds taken from each position can be compared.

Figure 3. Laser scanner situated inside the fenced zone. The posts where the reflective signals were placed can be seen (left). Detail of the two flat circular signals placed on a post (right).

In each campaign, the scanner was placed on a point inside the fence and on another on the outside; these stationary points were slightly different every year. Therefore, the coordinates of the points observed from each of these places are in a local system whose origin and orientation are different. Every time that the scanning of the terrain began, the first thing done was to measure these reflective signals and afterwards scan the terrain. As the signals do not change position at any time they are used as a fixed-reference framework to define the common system of coordinates.

The point clouds taken from each station were adjusted to this system via a process called register of point clouds. Essentially, it consists of determining the parameters, three translations and three rotations, of a transformation of a rigid solid that takes the local system of axes in each measurement campaign to a single system, corresponding to the position of the scanner located inside the fenced zone in the first measurement campaign (2013).

The scanner is placed atop a tripod approximately 1.50 m high, this is done manually, so that the angle of incidence with the terrain (angle between the laser beam and the normal to the terrain) is as small as possible. The accuracy of the measurements is maximum for angles of incidence of 0º and the measurement decreases when this angle increases. Angles of incidence over 80º are not advisable as they produce high errors of measurement, and can even make the signal reflected by the object lose energy and it cannot return to the emitter (Kaasalainen *et al.*, 2011).

Another aspect to take into account, because of its influence on the accuracy of the transformation of the coordinates, is the relative location of the targets. It is advisable that the targets (reflective posts) are not in line or grouped, but spaced out relative to each other, so that the angles of the lines joining the centre of the targets with the centre of the scanner are not very small.

The targets can have different shapes and dimensions and be made from various materials. In this study, the work was done with flat circular signals 5 cm in diameter, supplied by the manufacturer of the laser scanner.

Scanning of the terrain

Once the signals have been measured, the scanning of the terrain from each station can commence. First, the inside of the enclosed zone was scanned with the scanner located inside, in the same position from which the targets were measured. After that, the scanner was positioned at a point located outside the fence, but close to it, so that the control targets on the posts could be measured. From this point the previous process was repeated, that is, the targets were measured first and then the points on the terrain.

Before carrying out the scan, the angular increase between points must be indicated, this determines the density of the scan. The average density of each scan was approximately 0.5 points/cm^2, although it must be taken into account that the density is greater near the scanner and gets smaller as the points get further away, so that the nearest points are at a distance in the order of two cm whilst those furthest away are at around 10 cm. With this density of points, almost impossible to obtain using classical topographical methods, a survey of every zone was completed, inside and outside, in approximately ninety minutes This included the time needed to set up the equipment, scan the signals and scan the points.

Figure 4 shows the coloured point cloud obtained with the laser scanner from one of the stations. The fenced plot, with its almost elliptical shape, can be seen clearly. The black zones correspond to areas that were not measured by the scanner, owing above all, to hidden zones or to angles of incidence that were excessively high.

Processing the point cloud

Filtering of the point cloud

After the end of the field work the next step was, for each of the campaigns, to obtain a point cloud for both the zone inside the fence as well as outside, so that they

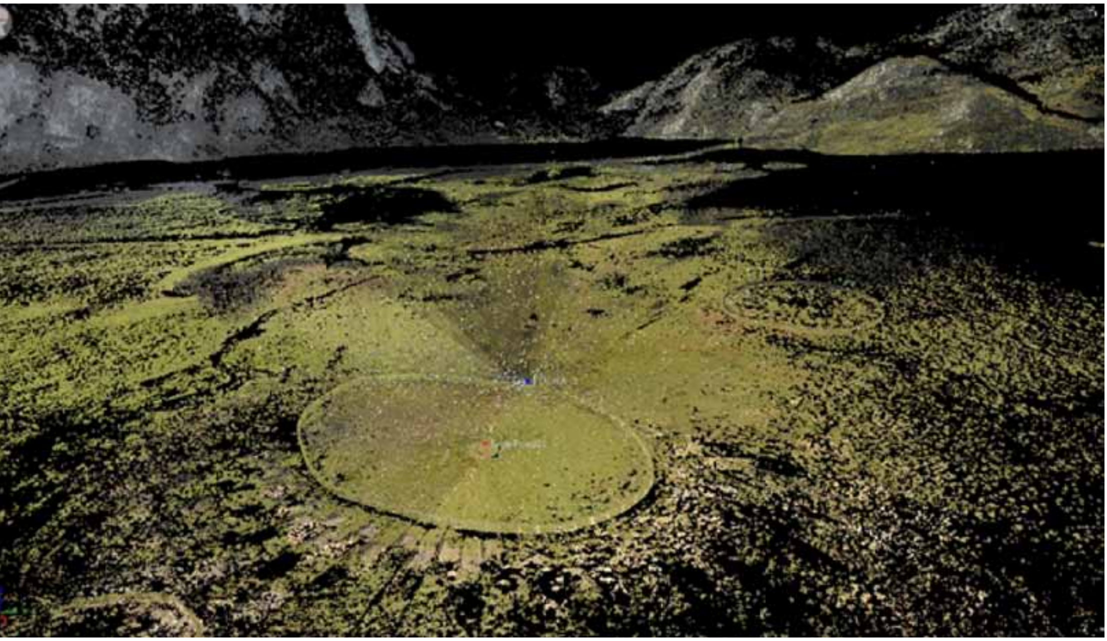


Figure 4. Point cloud in colour, measured with the laser scanner in the zone of study. In the middle of the image the fenced zone can clearly be seen.

could be compared separately. The zone inside was shown by drawing a continuous line on the point cloud itself, following the fence posts. Afterwards, the interior of this zone was cropped to eliminate the points to the exterior.

For each of the point clouds, exterior and interior, a filtering was carried out with the aim of eliminating the anomalous points, which are those that the scanner measures incorrectly, normally because of rebound effects from the edge of some objects. This is done via an algorithm that determines the distance between each point p from its closest neighbours, d_p, selecting as anomalous points those whose distance is greater than the average distance measured from each point to its closest neighbours, $\overline{d}$, plus a whole number, n, multiplied by the standard deviation σ of the distances between the closest neighbours in the whole point cloud:

$$d_p > \overline{d} + n\sigma$$

Filtering of the vegetation

The point clouds from the three years were compared with the raw data, and also with those that were processed with an algorithm included in the RiSCAN PRO software to eliminate the vegetation and other points that do not belong to the ground. This algorithm has its limitations, as it is very difficult to differentiate vegetation from the ground when a laser that can detect different pulses is not available as is generally the case with terrestrial laser scanner equipment. Nevertheless, the algo-

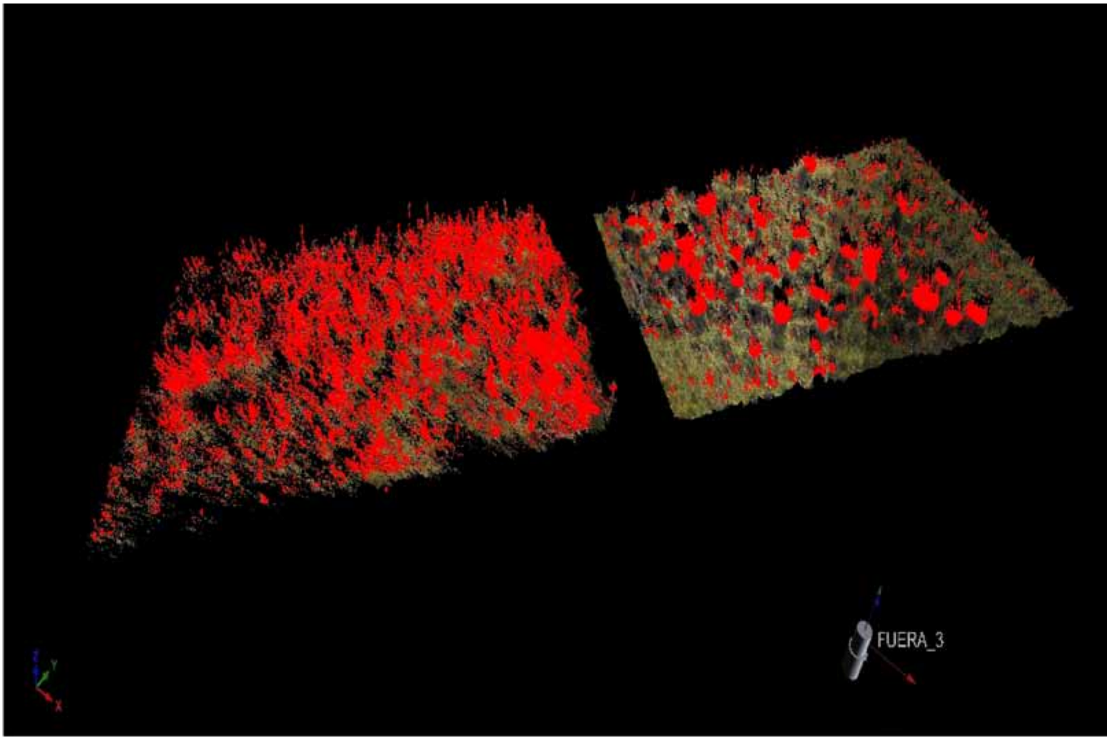


Figure 5. Result after applying the vegetation filter on a part of the zone studied. Red shows the points of the vegetation detected automatically.

rithm was applied and the time comparison made between the resulting point clouds in a way that was analogous to the non-filtered data.

The filtering of the vegetation was made hierarchically, starting with a grid of the point cloud of a certain cell size that became more and more refined with smaller cells. Firstly, a reference plane is defined, usually plane XY, and a grid is defined in this plane, initially that of the largest cell size. For each cell with at least a point inside, a representative point is selected based on a percentile defined by the user. If a percentile of 1% is defined, the representative point is selected in such a way that 99% of the points are over it and 1% under. If a percentile of 0% is defined, the representative point is the one of lowest height within the cell.

In each cell, a plan of the representative points and its neighbours was adjusted. Then in each cell two parallel planes to the previous one were constructed, each one at a distance which is defined by the user. Those points outside the region defined by the two planes were classified as not belonging to the terrain. The process was repeated with the rest of the points for the following level, constructed by dividing each cell in four, that is, each new cell had a size equal to half that of the original. After this first hierarchical filtering another was made with a lower tolerance (distance between parallel planes). In some cases, when the desired results were not achieved, it was convenient to carry out a second hierarchical filtering.

Register of the point clouds

Once the coordinates of the targets in the local reference system were known, corresponding to each position of the scanner, a transformation of coordinates was made. The aim being to determine the parameter of those transformation, translations and rotations, that allow a change from one reference system to the other. This transformation was made each time that the scanner was moved. Therefore, in each annual campaign one was made between the systems of coordinates from the scanner situated inside and outside the fence. In an analagous way, it was necessary to carry out another transformation of coordinates between each pair of consecutive years and between the first and third, both for the local system of the location point inside the fence as well as for the outside.

The year 2013 was defined as the reference for the inter-annual transformation. To achieve this, the coordinates of the retro-reflective signals corresponding to the accurate scan in 2013 were exported and used as registration points in subsequent years. The reliability of the transformations can be evalued by analyzing the residuals from the transformation from "rigid solid" among the clouds from different years.

Comparison of the point clouds

Once the three point clouds were registered and filtered, they were compared with the objective of determining how the heights of the terrain vary over time, both for the zone outside the fence as well as inside. Essentially, what is compared is the height of each point in a point cloud with that of the closest point in another observed in another year. In total six comparisons were made, three for the zone inside the fence and three for the zone outside, in the periods 2013-2014, 2014-2015 and 2013-2015. In this way the increase was determined, for each point, positive or negative, for the height over a year or the complete period of measurement, that is to say, two years.

The result of each comparison was represented on a map of height increase which helped to interpret the results. Moreover, the average values of the increases in the heights were determined as well as the standard deviation. With these results, an interpretation of the evolution of the topography of the terrain over the two years was made.

Results

Point clouds without vegetation filtering

Figures 6 to 8 show the results of the comparison between the heights of the point cloud from inside the fence, for each pair of years. They are shown as maps of colours that register the differences in the heights between each pair of clouds in absolute values. Blue with darker tones represent the distances between 0 and 3 cm, which could be attributable to errors in the scan itself. Analyzing the variations in the set of the two years (Fig. 8), it can be seen that the dominant colours are green and yellow, with a range that varies between approximately 10 to 30 cm.

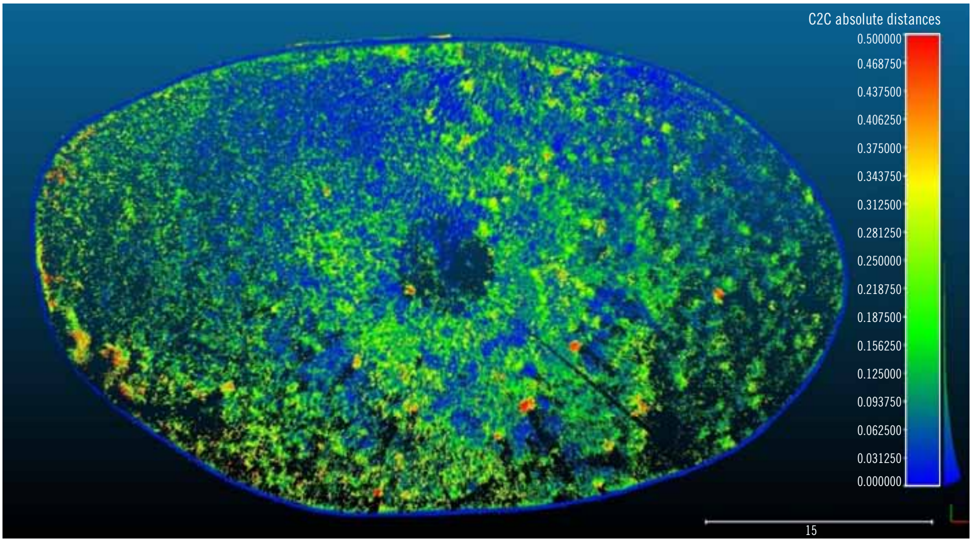


Figure 6. Comparison between two point clouds in the area not accessible by cattle, for the period 2013 – 2014. The legend corresponds to the absolute minimum distance between points from the two clouds compared.

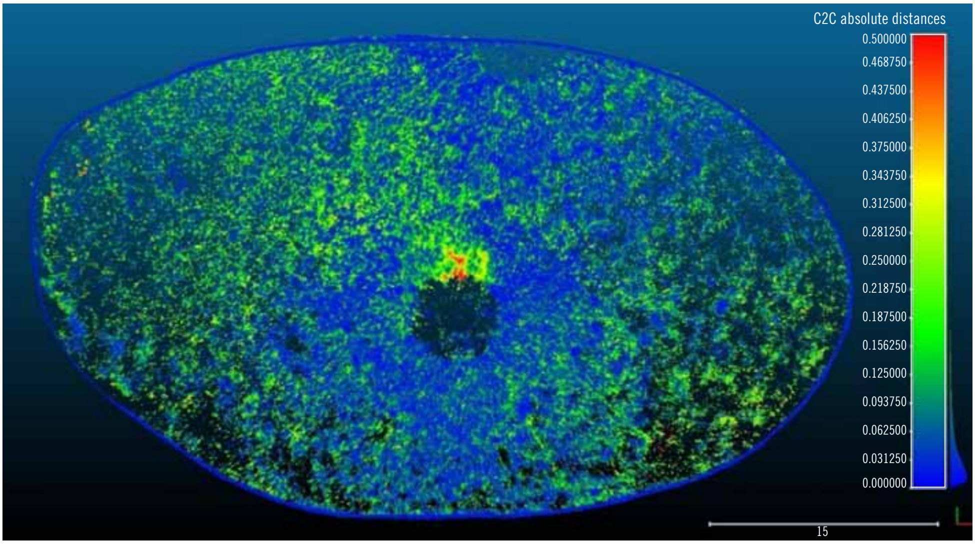


Figure 7. Result of the comparison between two point clouds in the area not accessible to cattle, for the period 2014 – 2015. Most differences are under 3 cm, in some zones the differences reach 18 cm.

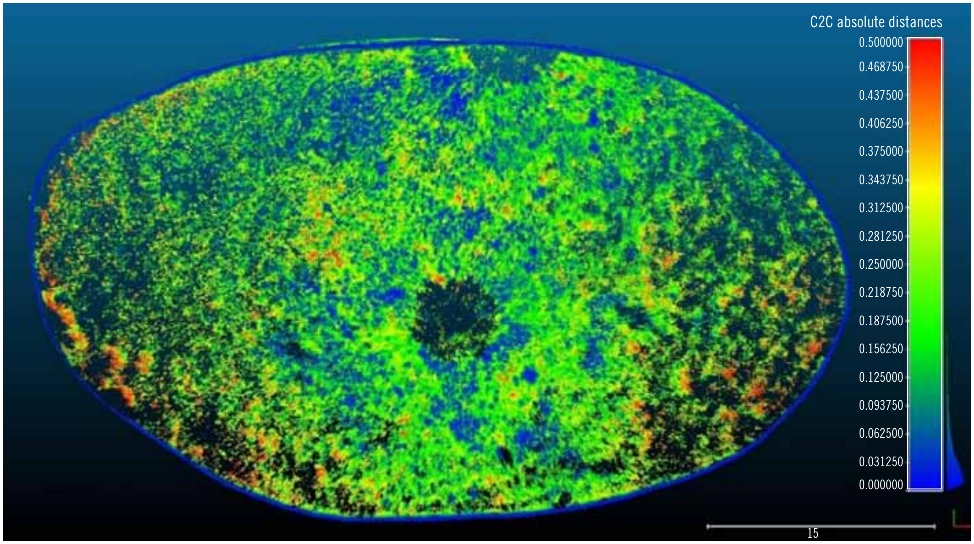
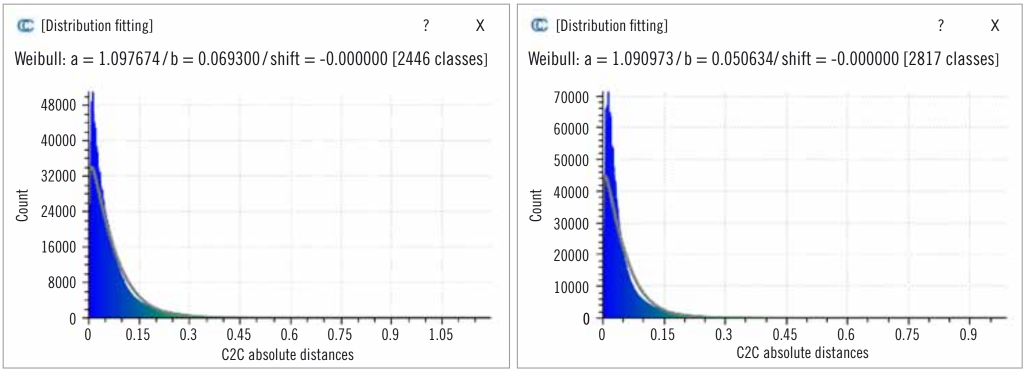


Figure 8. Result of the comparison between two point clouds in the area not accessible to cattle, period 2013 – 2015. Most differences were between 0 and 10 cm, in some zones the differences reach 35 cm.

Figure 9 shows the frequency distributions of the absolute distances corresponding to Fig. 6 to 8.



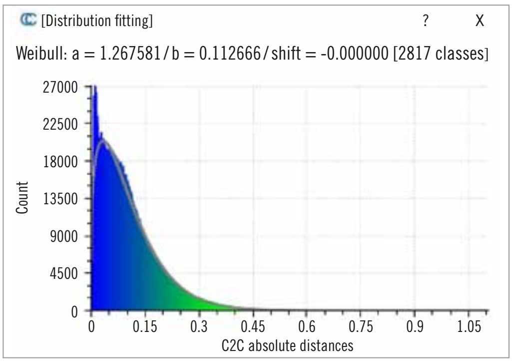


Figure 9. Clockwise from top left: frequency distributions of the minimum absolute distances between pairs of point clouds for the periods 2013-2014, 2014-2015 and 2013-2015. The continuous grey line corresponds to a Weibull density function adjusted to the frequency distributions.

Figures 10 to 12 show the results of the comparison for the zone accessible to the cattle, for the periods 2013 - 2014, 2014 – 2015 and 2013 - 2015. Dark-blue dominates, indicating that in general, there are no significant variations in the topography.

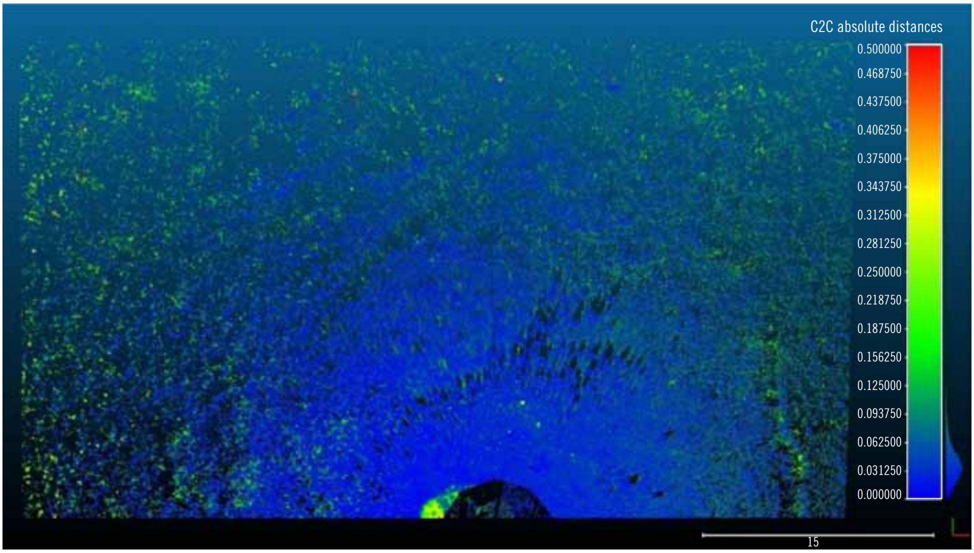


Figure 10. Result of the comparison between two point clouds in the area accessible to the cattle, period 2013 – 2014. Most differences are under 5 cm, in some zones the differences reach 15 cm.

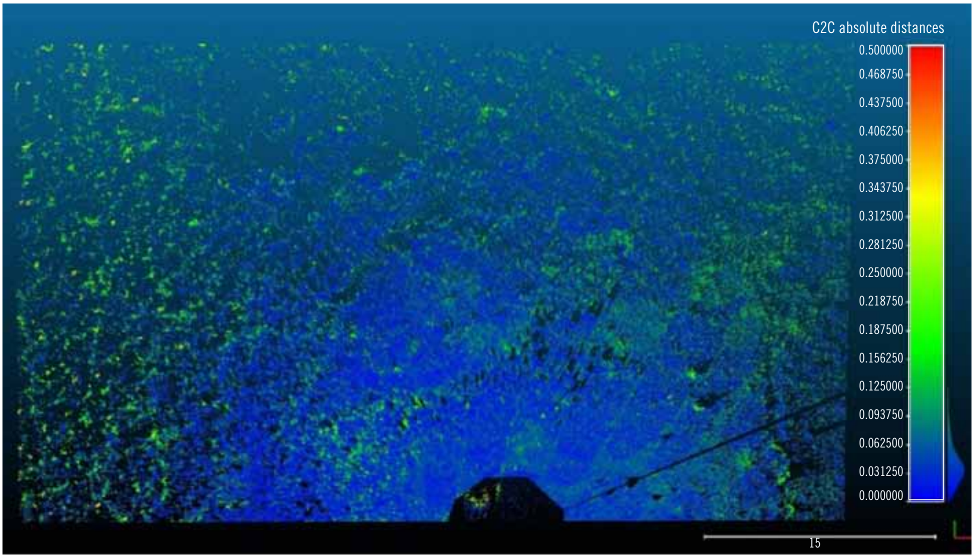


Figure 11. Result of the comparison between two point clouds in the area accessible to the cattle, period 2014 – 2015. Most differences are under 5 cm, in some zones the differences reach 15 cm.

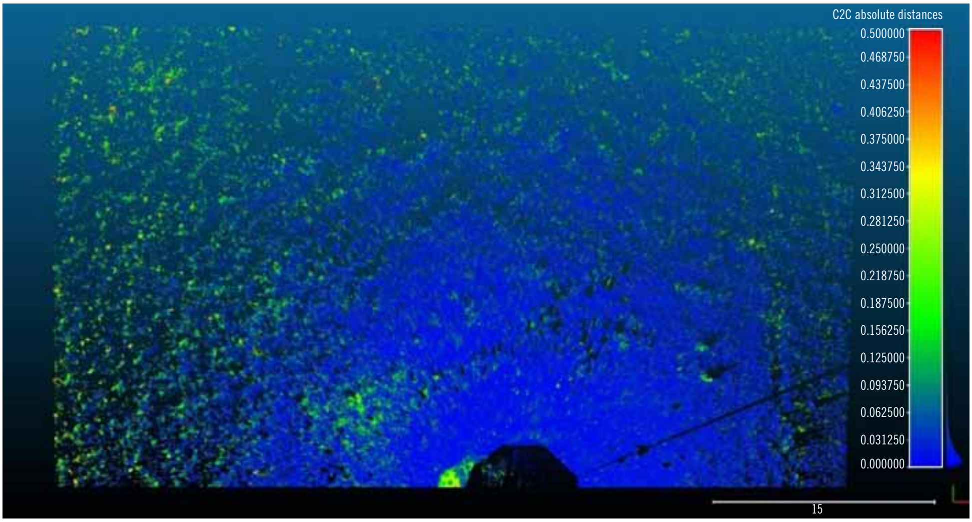


Figure 12. Result of the comparison between two point clouds in the area accessible to the cattle, period 2013 – 2015. Most differences are under 3 cm, in some zones the differences reach 15 cm.

Figure 13 shows the frequency distributions for distances between point clouds in the zone accessible to the cattle for the three periods analyzed:

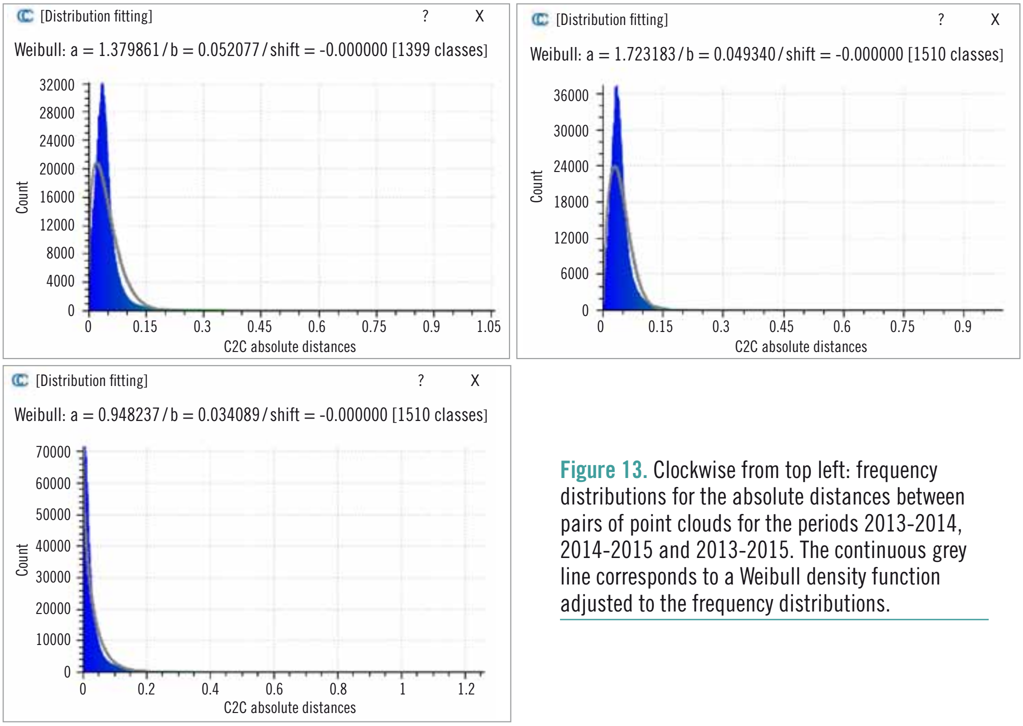


Figure 13. Clockwise from top left: frequency distributions for the absolute distances between pairs of point clouds for the periods 2013-2014, 2014-2015 and 2013-2015. The continuous grey line corresponds to a Weibull density function adjusted to the frequency distributions.

In Table 2 the means and standard deviations from the minimum distances between pairs of point clouds are shown. It can be seen that the mean variation in the two years of the study was 10 cm in the zone excluded to the cattle, with standard deviation values in the order of 8 cm, which indicates that there is a strong dispersion in the data, probably owing to the fact that the vegetation was growing freely. It can also be seen that the mean variation over the two years in the fenced zone is approximately the sum of the annual variations, as was to be expected, given that the topography in this zone has evolved freely. The variation of the zone outside the fence was stayed constant, between 4 and 5 cm, probably owing to the activities of the animals.

Comparison 2013-2014 terrain inside the fence		Comparison 2013-2015 terrain inside the fence		Comparison 2014-2015 terrain inside the fence	
Mean (m)	0.0669	Mean (m)	0.1051	Mean (m)	0.0490
Std dev (m)	0.0670	Std dev (m)	0.0848	Std dev (m)	0.0527
Comparison 2013-2014 terrain outside the fence		Comparison 2013-2015 terrain outside the fence		Comparison 2014-2015 terrain outside the fence	
Mean (m)	0.047103	Mean (m)	0.0351	Mean (m)	0.0437
Std dev (m)	0.0407	Std dev (m)	0.0477	Std dev (m)	0.0277

Table 2. Mean absolute displacements and standard deviations between point clouds.

Point clouds with vegetation filter

The results of the comparison of the point clouds, for the area not accessible to the cattle, after having filtered the vegetation, are shown in Fig. 14 to 16. The spatial distribution of the distances between points is similar to the case in which the vegetation was not filtered, although it can be seen that there is more dark-blue, which indicates that part of the rise in the terrain is really due to a growth in the vegetation, something that is obvious when simply looking at the terrain.

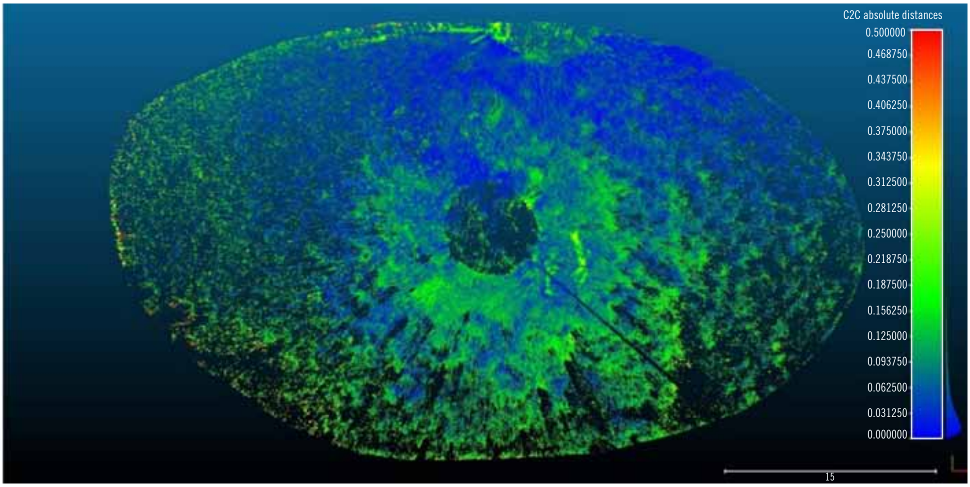


Figure 14. Comparison between two point clouds in the area not accessible to the cattle, after applying the vegetation filter for the period 2013 – 2014. Most differences are between 0 cm and 6 cm, in some zones the differences reach up to 20 cm.

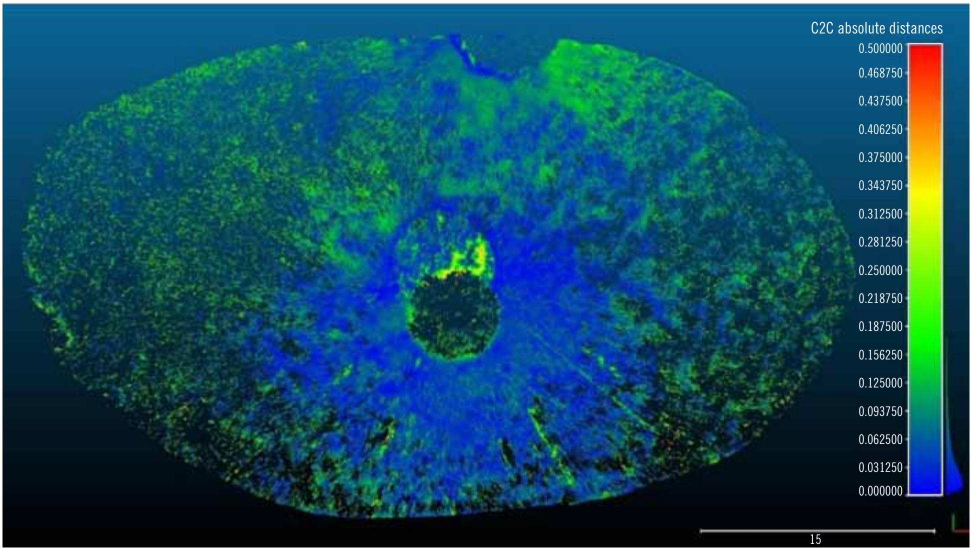


Figure 15. Comparison between two point clouds in the area excluded to the cattle, after applying the vegetation filter for the period 2014 – 2015. Most differences are between 0 cm and 3 cm, in some zones the differences reach up to 15 cm.

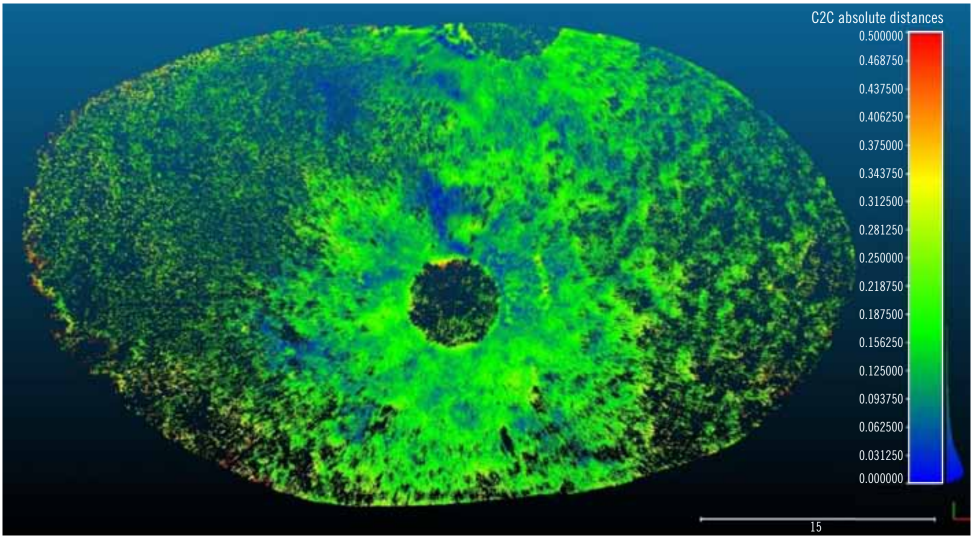


Figure 16. Comparison between two point clouds in the area excluded to the cattle, after applying the vegetation filter for the period 2013 – 2015. Most differences are between 3 cm and 15 cm, with some zones reaching differences of up to 28 cm.

In Fig. 17 the frequency distributions for distances between the point clouds in the zone accessible to the cattle are shown, after having applied the vegetation filter, for the three periods of time analyzed:

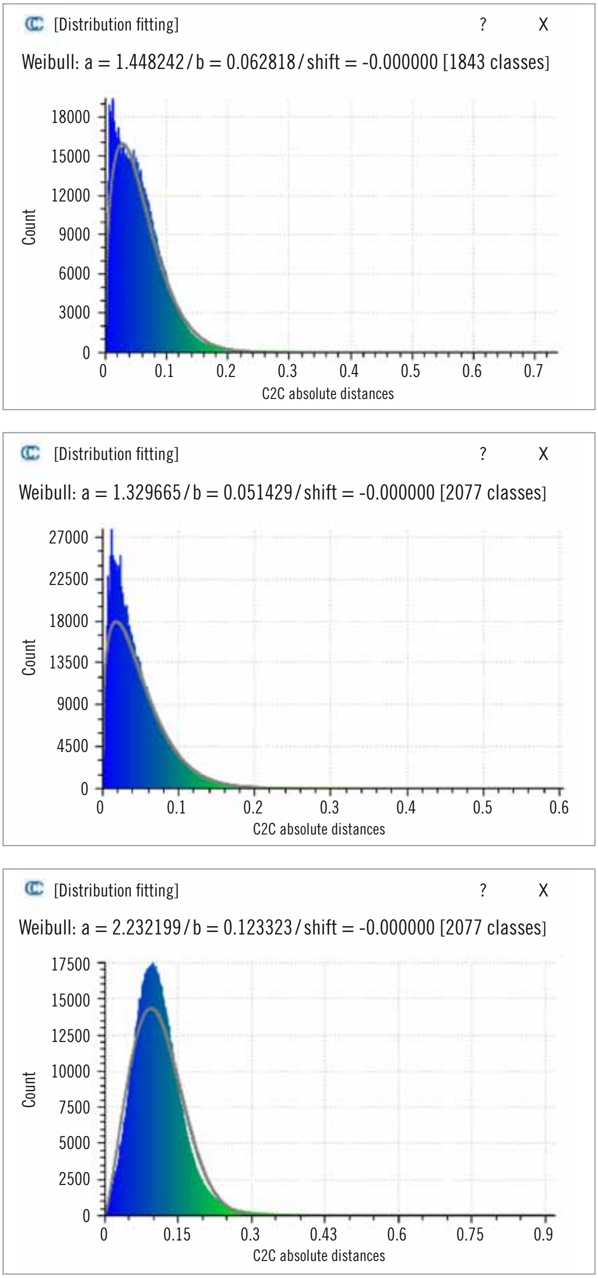


Figure 17. From top to bottom : frequency distribution of the absolute distances between pairs of point clouds for the periods 2013-2014, 2014-2015 and 2013-2015. The continuous grey line corresponds to a Weibull density function adjusted to the frequency distributions.

Figures 18 to 20 present the same results as in the four previous figures for the zone accessible to the cattle. In the whole of the two years important parts of the area under study did not show any differences over 2 cm, and cannot be considered significant given the characteristics of the scanner and the terrain, with abundant vegetation that produces reflections at different heights and noise in the point cloud.

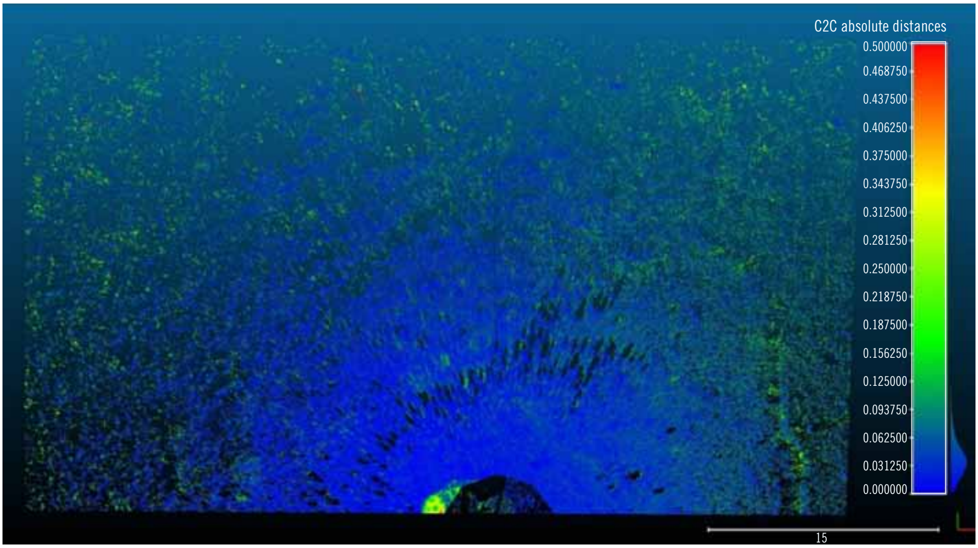


Figure 18. Result of the comparison between two point clouds in the area accessible to the cattle for the period 2013 – 2014, after applying the vegetation filter. Most differences are under 5 cm, in some zones the differences can reach 15 cm.

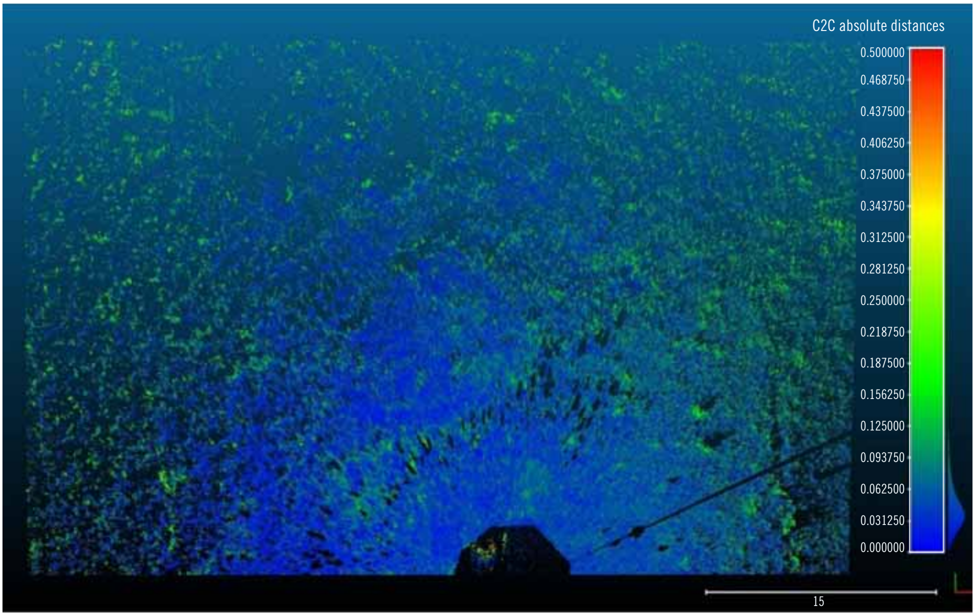


Figure 19. Result of the comparison between two point clouds in the area accessible to the cattle for the period 2014 – 2015, after applying the vegetation filter. Most differences are under 5 cm, in some zones the differences can reach 15 cm.

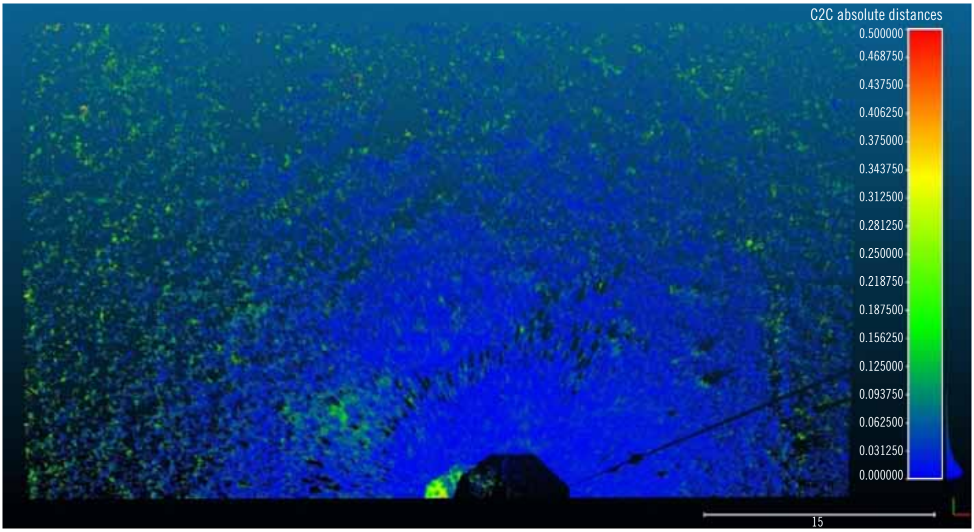


Figure 20. Result of the comparison between two point clouds in the area accessible to the cattle for the period 2013 – 2015, after applying the vegetation filter. Most differences are under 2 cm, in some zones the differences can reach 10 cm.

Figure 21 shows the frequency distributions corresponding to the comparisons for the three periods analyzed:

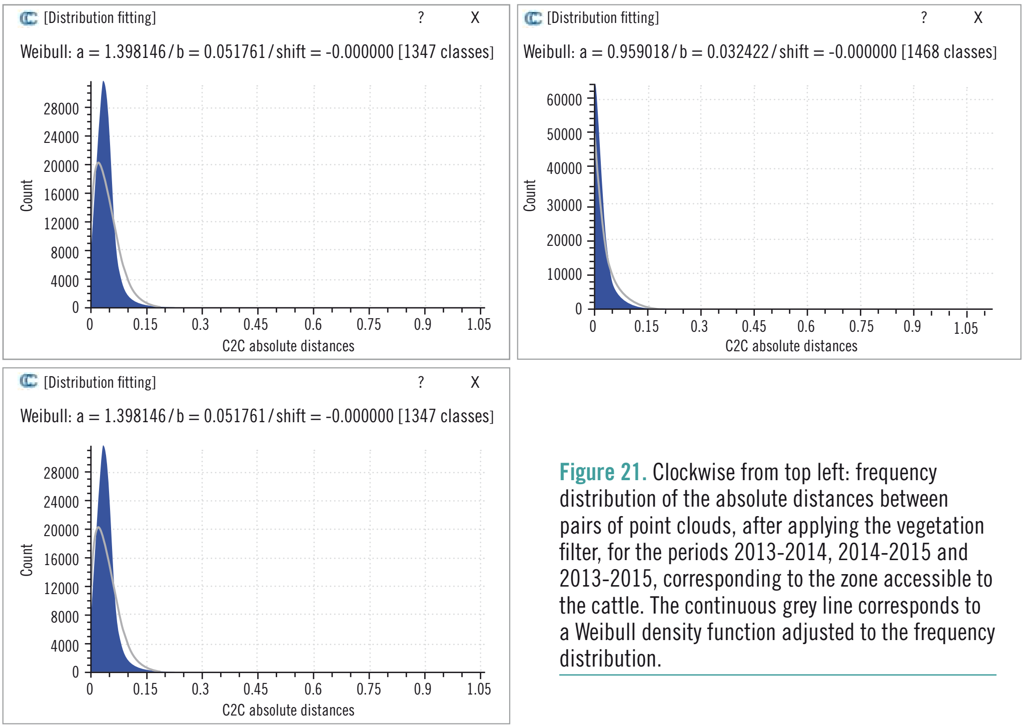


Figure 21. Clockwise from top left: frequency distribution of the absolute distances between pairs of point clouds, after applying the vegetation filter, for the periods 2013-2014, 2014-2015 and 2013-2015, corresponding to the zone accessible to the cattle. The continuous grey line corresponds to a Weibull density function adjusted to the frequency distribution.

Table 3 shows the mean values and standard deviations of the absolute distances for the point clouds after applying the vegetation filter. Compared to Table 2, it can be seen that these two statistics are similar to those obtained without the vegetation filter.

Comparison 2013-2014 terrain inside the fence		Comparison 2013-2015 terrain inside the fence		Comparison 2014-2015 terrain inside the fence	
Mean (m)	0.0570	Mean (m)	0.1093	Mean (m)	0.0470
Std dev (m)	0.0409	Std dev (m)	0.0512	Std dev (m)	0.0389
Comparison 2013-2014 terrain outside the fence		**Comparison 2013-2015 terrain outside the fence**		**Comparison 2014-2015 terrain outside the fence**	
Mean (m)	0.0468	Mean (m)	0.0332	Mean (m)	0.0451
Std dev (m)	0.0400	Std dev (m)	0.0455	Std dev (m)	0.0247

Table 3. Absolute mean displacements and standard deviations between point clouds for the clouds with vegetation filter.

Conclusions

Analysis of the point clouds measured with TLS has confirmed the hypothesis, that the action of cattle interferes with the evolution of the topography and the vegetation covering the bog. A comparison of the point clouds indicates that, in the two year period of the analysis of the terrain, the ground rose on average by 10 cm in the zone not affected by the cattle, whilst the average variation in the zone accessible to the cattle was hardly over 4 cm.

This is due to the fact that without cattle there is no compression of the ground from trampling. Inside the enclosure the *Sphagnum* mosses, which make up the majoritory of the moss cover in the bog, increase in volume as they can absorb and accumulate water and grow in height, giving rise to the generalized raising of the bog in the whole of the enclosure with respect to the zone outside.

The application of a filter to eliminate the vegetation allowed us to conclude that part of the rise is due to the growth of the herbaceous vegetation, a fact that is easy to see in the field.

The laser scanner is a very useful tool to study the evolution of the topography of this terrain at small scale, especially because of its ability to get high-density models of the terrain. However, the presence of herbaceous vegetation brings some problems: on the one hand, the laser beam emitted by the equipment can hit different heights on the plants; on the other, high plants hide parts of the terrain. These problems can be partially solved by setting up the scanner at more stations, which lowers the number of zones without points and reduces the angles of incidence, with observations more orthogonal to the terrain. Moreover, it would raise the density of the scanning, improving the adjustment to the terrain.

References

Alba, M., Prandi, F., Fregonese, L., Scaioni, M. 2006. *Structural monitoring of a large dam by terrestrial laser scanning*. www.isprs.org

Armesto, J., Ordóñez, C., Alejano, L., Arias, P. 2009. Terrestrial laser scanning used to determine the geometry of granite boulder for stability analysis purposes. *Geomorphology* 106 (3-4): 271 – 277.

Kaasalainen, S., Jaakkola, A., Kaasalinen, M., Krools, A., Kukko, A. 2011. Analysis of the incidence angle and distance effects on terrestrial laser scanner intensity: search for correction methods. *Remote Sensing* 3: 2207 – 2221.

Lagué, D., Brodu, N., Leroux, J. 2013. Accurate 3D comparison of complex topography with terrestrial laser scanner: application to the Rangitikei canyon (N-Z). *arXiv*:1302.1183

Prokop, A., Panholfer, H. 2009. Assessing the capability of terrestrial laser scanning for monitoring slow moving landslides. *Natural Hazards and Earth System Sciences* 9: 1921-1928.

Rosser, N.J.J., Petley, D.N.N., Lim, M., Dunning, S. A. A., Allison, R.J.J. 2005. Terrestrial laser scanning for monitoring the process of hard rock coastal cliff erosion. *Quarterly Journal of Engineering Geology and Hydrogeology* 38 (4): 363–375.

Hayakawa, Y., Kusumoto, S., Matta, N. 2016. Application of terrestrial laser scanning for detection of ground surface deformation in small mud volcano (Murono, Japan). *Earth, Planets and Space* 68: 114–124.

07

Program LIFE+ “Bogs of the Jura”: 60 bogs to restore in six years

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Abstract

Here, we present actions and results of the LIFE+ "Bogs of the Jura", targetting the restoration of peatlands in this French region.

Keywords: bogs, restoration, Jura, France.

Introduction

The Jura massif makes up one of the richest areas for bogs in Western-Europe and is located along the border between France and Switzerland, in the North-west of the Alps. The bogs have developed in a calcareous karstic area, and boast a very high diversity of bog types and structures that have created numerous habitats of community importance. Moreover, they are especially abundant in heritage species. However, as in other numerous European bogs, these natural zones have been subjected to many forms of damage: peat extraction, drainage channels, correction of water courses etc. These anthropic actions have caused hydrological disruptions which, in turn, have degraded the general functioning of these ecosystems linked to waterlogged ground.

Since the 1990s, several actions have been undertaken to improve the bogs around the massif. However, with the aim of consolidating the network as a whole, preserving the species and the habitats of community importance over the long term, the local managers have expressed their desire to work at a larger scale. In this respect, the LIFE Bogs of the Jura has the objective of recovering the functioning of the 60 bogs in the Jura massif Franche-Comté zone. Therefore, several actions have been implemented over the last six years, according to the problems found at each of the bogs and according to the technical and ecological possibilities at the sites.

This article gives the details of each of the large restoration techniques applied within the framework of the aformentioned project: neutralization of the drainage channels, restoration of the water courses and regeneration of the ditches dug for the extraction of peat. Finally, several examples of actions undertaken in five bogs are explained in more detail.

Bogs of the Jura

Context

The Jura is a low-altitude massif, whose highest summit reaches 1,720 metres, and is located on the border between France and Switzerland. It has a maximum width of 70 km and is 340 km long. It is characterized by its half-moon shape, the result of tectonic movements originating in The Alps to the South-east which compress the land, folding it. Although the height of 1,000 m is rapidly lost on the Swiss facing slopes, they decrease gradually towards the West in a succession of tiered plateaus.

The geological substrate is basically made up of marls and calcareous rock from the Jurassic and the Cretaceous periods. In spite of the predominance of karstic permeability related to the calcareous rock, this massif contains a relatively important number of bogs. Their presence is related to the combination of a relief that is not very rugged and the presence of marls, as well as the glacial deposits. At the same time the dominant oceanic climate, characterized by heavy rains, are relatively well-distributed throughout the year.

So, the massif holds a total of almost 500 bogs, with a combined surface total of around 5,400 ha, principally at an altitude of between 800 – 1,000 m (Moncorge *et al.*, 2016).

The bogs selected for the LIFE+ “Bogs of the Jura” are found in the Burgundy-Franche-Comté region, which makes up more than half of the massif. In this region, 65 % of the 286 bogs present are included in the Natura 2000 network.

Natural heritage

One of the peculiarities of the Jura massif in terms of bogs lies in the marked diversity of the natural spaces that can be found there. Indeed, although the geological substrate has created a group of low bogs and alkaline fens (6410, 7210, 7230), with the passing of hundreds of years, raised bogs have been created there (7110*, 7120, 91DO*), whose acid and oligotrophic nature provide a sharp contrast with the surrounding areas. Added to this there are also transition mires (7140), aquatic media (3140, 3150, 3160), pioneer depressions on peat substrates (7150), as well as low acid bogs that can develop on the edges of the raised bogs.

This variety allows a diversified flora and fauna to flourish, and among these there are species in serious danger of extinction.

In this respect, nine species are found in the Annex II of the Habitats Directive: *Saxifraga hirculus*, the orchid *Liparis loeseli* and *Hamatocaulus vernicosus* in the flora; and among the fauna, the butterflies – the Marsh Fritillery (*Euphydryas aurinia*), *Lycaena dispar*, *Maculinea nausithous*, as well as the dragonfly *Leucorrhinia pectoralis* and the land snails, *Vertigo geyeri* and *Vertigo genesii*.

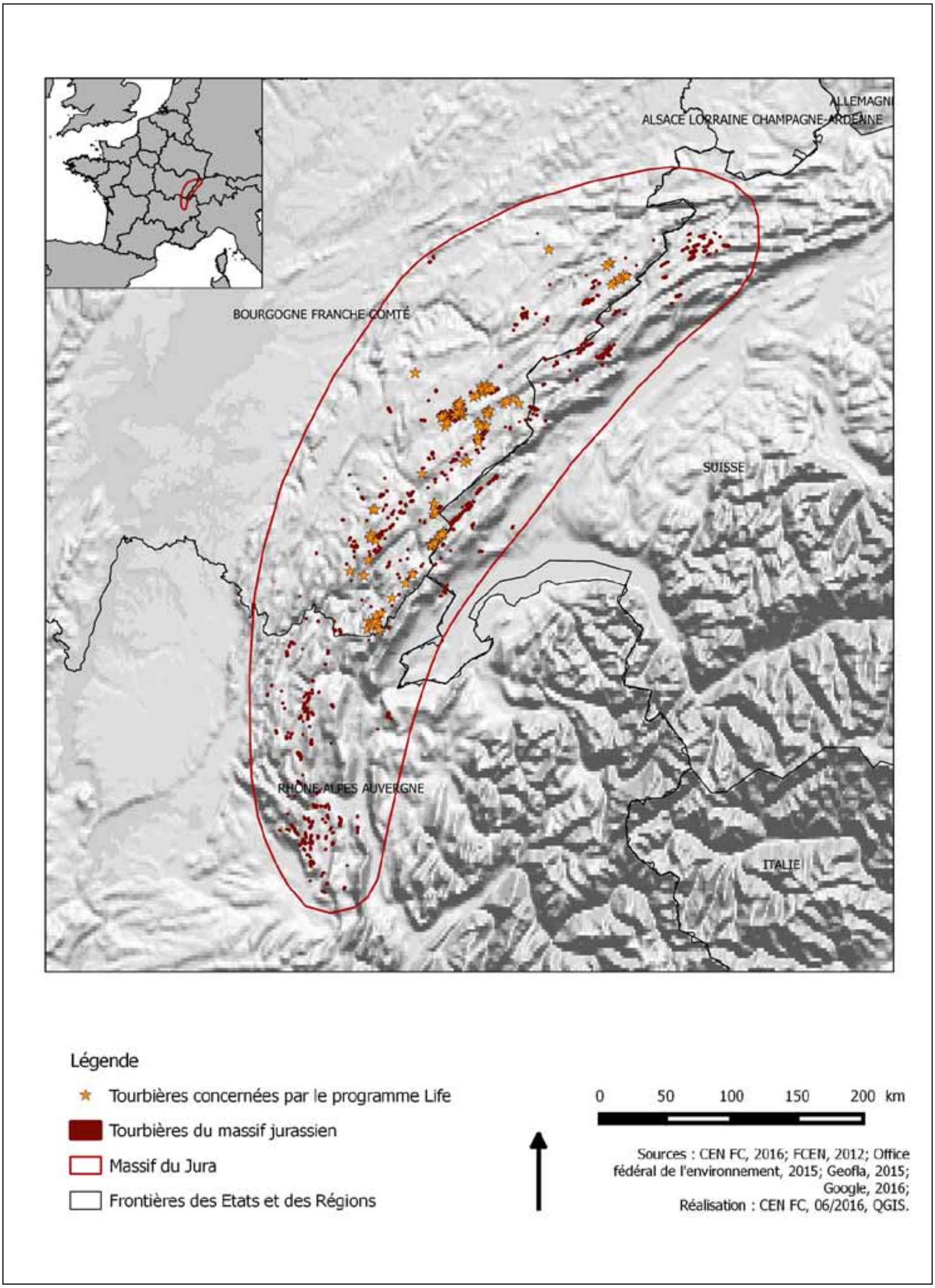

Figure 1. Bogs in the Jura massif.

Damage

The bogs of the Jura have been exploited by people for a long time, especially as zones for pasture, depending on the fluctuations in the human population of the massif.

However, its most notable use has, without a doubt, been that of peat extraction for burning, and it seems that no raised bog in the massif has been left untouched (Fig. 1). This exploitation developed between the 17th century and the beginning of the 20th century, principally to compensate the scarcity of wood. Although this activity has allowed a local diversification and regeneration of the boggy areas, often it has brought a significant long-lasting deterioration in the zone owing to the resulting drying.

Later, from the middle of the 20th century onwards there was a second large scale "offensive" on the peat areas. This time it was mechanized, since there was an attempt to convert the area, up to that time considered as useless, into zones for agriculture or forestry. Thus, drainage, correction of the water courses and plantations appeared, and once again bogs functioning was disrupted. Added to these factors in its deterioration, others can be mentioned, for example pollution, filling, water capture, creation of ponds, pools/dams, etc.

Therefore it is estimated that, approximately, 30 % of the peat surfaces in the Jura Franche-Comté have been totally destroyed (Moncorge *et al.*, 2016). Referring to the inactive surfaces, that is, those that do not accumulate peat any more, the figure is unknown but probably very considerable.

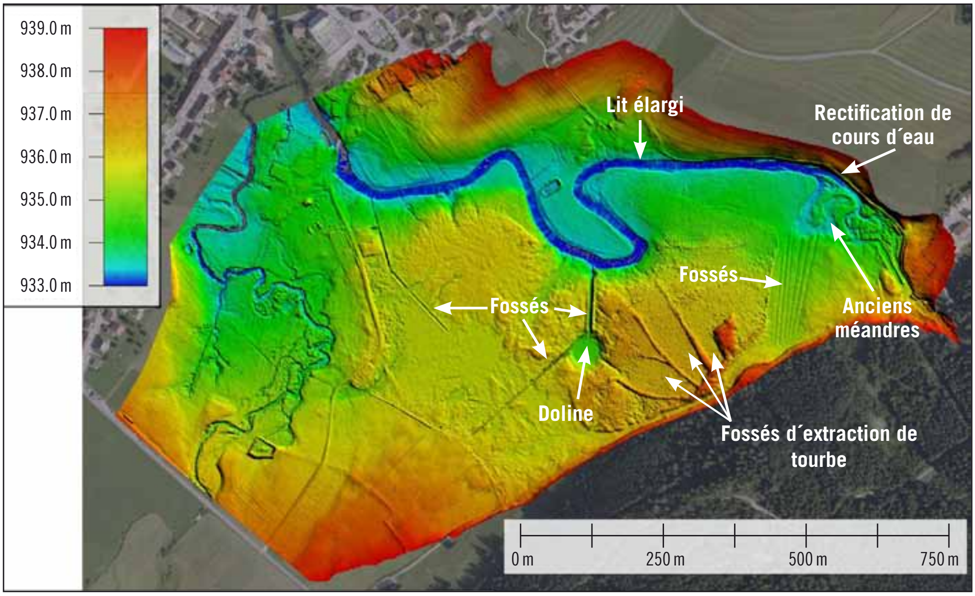

Figure 2. Digital model of the bog terrain on the Moutat en Mouthe. Peat extraction, drainage channels, harmful correction of water courses for the functioning of the zone. In the centre, a sinkhole naturally drains part of the waters towards the karst.

Moreover, the consequences of this deterioration brought a sharp reduction in the resilience of these habitats, an observation that is especially worrying in the context of global warming, and a probable increase in nitrogenated precipitation. Consequently actions that return the functional rehabilitation of these natural habitat are particularly urgent.

The LIFE Bogs of the Jura

Context and main features

In the 1990s, the first management measures undertaken in the Jura bogs concentrated, above all, on a “surface” approach, that is, the aim was to preserve habitats and species. In the 2000s, hydrology gradually became the priority for the managers, who began to take an interest, little by little, in the overall functioning of the bogs. Therefore work began on improving the hydrological aspects of these bogs. The progress made by the Swiss in this field, close neighbours, was particularly decisive.

Faced with a lack of type “classic” financing, the local managers wanted to carry out a wide-ranging program that would allow the restoration of a large number of bogs and at the same time make savings from economies of scale. After a preparatory phase of around a year, the application file for the project LIFE “Functional rehabilitation of the bogs in the Jura Franche-Comté Massif”, summarized in “Bogs of the Jura Mountains” was presented in June 2014 and was accepted by the European Commission in April 2014

Therefore, the LIFE+ Bogs of the Jura Mountains focused on the rehabilitation of the hydrological functioning of 60 bogs in the Jura massif, distributed among 16 Nature 2000 sites. The objectives, at the end of six years of LIFE were the following:

- Neutralization of drainage channels: 16,000 m;
- Recovery of meanders in the water courses: 12,000 m;
- Rewetting of the peat extraction ditches: 26 ha.

In total, 510 ha of bogs were to be restored.

Life tourbières du Jura

NAME: LIFE TOURBIERES DU JURA
DURATION: 6 years (from June 2014 to November 2020)
TOTAL BUDGET: €8,051,163
LOCATION: Jura Franco-Comtou Massif
COORDINATING BENEFICIARY: Conservatory of Natural Zones of the Franco Condado
ASSOCIATED BENEFICIARIES: Association of Aquatic Zones of the Upper-Doubs, Regional Natural Park of the Upper-Jura, Association of Friends of the Natural Reserve of Lake Remoray, Association of Planning for the Dessoubre and Evaluation of the Catchment área and Regional Environmental Management and Housing for the Franco Condado
FINANCIAL SPONSORS: European Union, Ródano-Mediterranean- Corsica Water Agency, Regional Council of the Franco Condado, Departments of Doubs and Jura

Actions undertaken

Preparatory actions

As for any ecological engineering action, the works in the program were preceded by studies that allowed a better understanding of the hydrological and ecological functioning of each bog. The diagnosis was adapted to each site and could include the study of the flows of superficial and subterranean waters (piezometry, hydrometry, physical-chemical properties, surface topography and of the underlying geological substrate), the study of the vegetation, of the geomorphology, etc. The local managers also often used LiDAR technology (Light Detection And Ranging). This topographical reading, using airborne laser, was used to generate a digital model of the terrain that gave a better understanding of the overall functioning of the site and reveal characteristics of the terrain that would not be easily detected by other means (channels, old meanders in the corrected water courses, etc.).

The ownership of the terrain is also an essential pre-requisite to carry out any action. Fortunately, most of the plots of land included in this project were common land whose management could be agreed by the beneficiaries thanks to the negotiating of agreements. The ownership has been completed over the last few years by acquisitions, rental and management agreements with the individual landowners. However, land holders are still being encouraged to participate in a process that would complete the ownership of this domain.

In agreement with the current French legislation, works in natural habitats may also need permits from some authorities. These formalities are defined according to the objective and/or the extent of the action, the presence of protecting enclosures and the legal status of the body responsible for the works.

Blocking of drainage channels

The layout of the drainage channels varies greatly from place to place. Both the width and the depth can vary between 0.5 to 4 m, and their length between 10 and several hundreds of metres. The gradients can also vary a lot. Moreover in the Jura, these channels can cross all sorts of peaty media (from bogs to forests), likewise causing disruptions of a trophic nature (eg: highly acid marsh crossed by a channel carrying calcareous waters).

Optimum actions consist of totally blocking the channel with peat or similar material. This technique, which brings a level of hydric saturation on the whole of the dried out surface, eliminates the effect of the ditches which was the priority during this program. However, it requires a large quantity of materials whose quality must be adapted to the trophic conditions of the site being restored. Naturally, they can be of peat (generally extracted close by which brings the regeneration of, for example, "*Rhynchosporion* hollows" - 7150). Another option is the use of spruce sawdust, an interesting alternative solution when the movement of the peat at the site was not possible, and which had been successfully tried at different sites in the Jura. In some cases, small diameter trees can be shredded and mixed with peat/sawdust to provide additional volume. For the major channels that are close to the mineral substrate, marly-clay can also be used (moraine clay without large stones) to replace the peat as a blocking material. In all cases, the channel was first cleared of any vegetation that was growing there, bringing better cohesion between the blocking material and the sides of the channel.

If the complete filling was impossible, an alternative technique consisted of creating dams covered with peat. The quantity of these works depended to a large extent on the gradient: the steeper they were, the greater the number and size needed. For medium-sized channels it was sufficient to place tri-fold boards. On the other hand, for larger channels, where the hydraulic pressure could be high, the creation of robust dams made with tongue and groove wooden boards anchored to the mineral substrate, below the peat, was essential. The creation of overflow systems within the dams helped to eliminate the erosion of the peat cover and in the proximity of the work, increasing the longevity. The height of this system was determined depending on the height of the phreatic layer desired upstream from the works, and could be adjusted.

Both techniques (filling and damming) could of course be combined in the same work.

As mentioned above, the whole set of works created was covered with peat with the aim of making sure that the wood used for the dams was not in direct contact with the air and the water. This technique limited the rotting of the wood, extended and maximized the longevity of the works. What is more, greening of the peat covering can be envisaged, the aim being to stabilize and avoid any erosion in the first few months/years after the works, pending the development of the natural vegetation. Padding of the peat covering was also an alternative solution to limit the erosion whilst the vegetation develops. These techniques also have the indirect consequence of reducing the visibility of the works.

Restoration of water courses

Some of the water courses that cross the bogs underwent corrections that altered their good hydrological functioning some time ago and hence, that of the bogs. In the Jura massif, the hydrological and hydrogeological composition can generate subterranean calcareous water supplies (base water) and zones of contact between alkaline and acid bogs, and means first and foremost that there must be a good understanding of their functioning to achieve good water management, and therefore the restoration of the initial flows.

In the works to restore the water courses, it was always preferred to reactivate the old meandering course whenever possible, their trajectory could be found by using, for example, LiDAR. Their altitude was verified with the aim of determining whether it was in agreement with the correct functioning of the system (the old meanders could become deeper or, on the contrary, they could need to be excavated/opened up again). When it was impossible to implement this solution, a new course was created.

In any case, the dimensions of the new bed were slightly undersized in relation to its theoretical regime. The erosion of the water course itself ended up with a size in agreement with the natural dimensions of the flow a few years later. The new trajectory must be adapted to the topography of the sector (generally, in a marsh, its length is multiplied by 1.7 to 2 regarding a rectilinear course).

Once the new flow is created, it is necessary to block the old rectilinear one, remaking the dam after clearing, and completing it with suitable materials (it may be that then the volumes are very significant). If it is impossible to regain the balance over the whole of the trajectory and if also, downstream, the system is disrupted, bottom ramps can be placed to create a stable link between restored and unrestored parts, so as to guarantee the water level and to stop any regressive erosion.

A “sedimentary reload” can also be made in the new course, to accelerate the creation of a varied substrate and restablish the erosion dynamics, a guarantee of greater morphological and ecological diversity.

In certain cases, it was not possible to totally restore the water course, and so a partial rehabilitation was started. This could consist of a narrowing of the minor bed or placing bottom ramps with the aim of raising the water level. But this solution is never an option if a more ambitious restoration can be envisaged.

Regeneration of peat extraction areas and water retention

These works have the goal of recovering the damaged areas that have been subjected to extraction, once again creating the favourable conditions for the installation of plant communities that generate peat. In this respect, the works consisted of creating one or several dams with the aim of retaining water within the channels, or more generally, of the whole area of extraction whenever possible.

The nature of the work was adapted to the dimensions of the channel and the height of the water to be retained. For small sized works (water height under 30 cm), it was enough to build a protecting dam made of peat. However, this had to be done taking into account the following precautions: good quality peat, collected in blocks, compact and protected with plant growth and/or placing of a geotextile.

Larger works (water height over 40 cm) had to be undertaken with a dam made of wooden boards which had to be able to withstand the weight of the water and the pressure. In turn, the dam had to be covered with peat, again padded or covered with vegetation.

With the aim of limiting variations in the water level and to avoid a scouring effect on the work (erosion induced leaks), an overflow system must be installed inside the dam. This could be a simple discharge (resistant to erosion), or it could be integrated into the work (a bent tube for example). In certain cases, it was useful for the overflow to be regulated so as to adjust the water level. The depth of this level depends on the zone the channel (slope, relief etc.) and the management objectives. As it favours a notable water height, a significant waterlogged zone is created which allows, according to the conditions, the creation of marshy pools (especially favourable for *Leucorrhinia pectoralis*) or reed beds. The evolution towards communities of "active" raised bog (7110*, 7150) is then relatively long. The maintenance of a fine sheet of water (emerging) allows a faster installation of communities of what are called "terrestrial" *Sphagnum* mosses. In this case, the rehabilitation towards those habitats mentioned above is much faster.

Examples

«Mont de Voyon» alkaline fen in Granges-Narboz (Department of Doubs)

Context

- Management: Association of Aquatic Zones of the Upper-Doubs.
- Description of the site: alkaline low marsh of 6.5 ha, occupied mainly by Molinia - Moor grass - meadows on calcareous, peaty or clayey-silt-laden soils (6410) and also bogs with *Carex diandra* (7140), alkaline fen with *Carex davalliana* (7230) and population of willows - *Salix pentandra*.
- Heritage value: moderate floral wealth owing to invasion by molinia and willows, but with presence of *Carex estospitosa* and *Dianthus superbus*. Wealth of wildlife was related to the presence of *Coenonympha tullia*, *Euphydrias aurinia* and *Lyceana helle*.

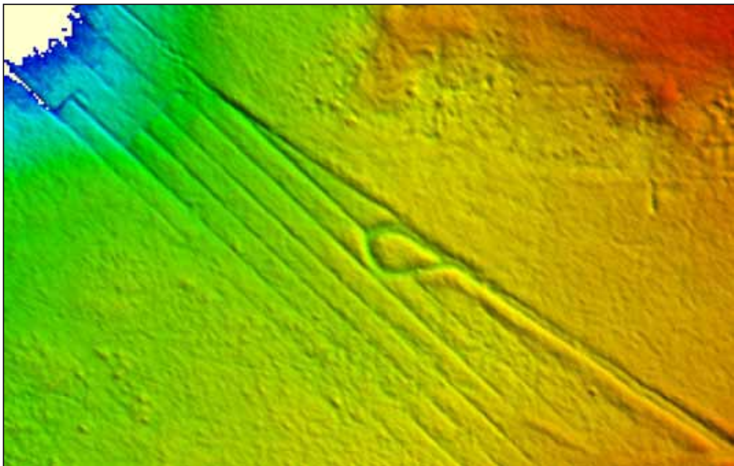

Figure 3. Mont Voyon terrain model using LiDAR.

Verdict

This alkaline fen suffered a long period of peat extraction and exploitation which from the 18th century caused, owing to desiccation, colonization by willows.

Problems pending solution

- Progressive desiccation with invasion by Willows.
- Lowering of the overall water flow.

Objectives

Slow down the flows, reload the wetland with water and in summary the phreatic layer by taking advantage of the lower flows.

Figure 4. Blocking dam and peat covering.

Figure 5. Blocking dam and sawdust covering.

Works

The channel density was significant (740 m/ha). Drained and exploited, the alkaline fen only had a thin layer of peat; no more than 2 m. The course of action taken was to completely block the channels with sawdust and wooden panels owing to the lack of peat. Downstream, where the channel is very wide, but the slope relatively gentle, wooden dams were constructed to block the flows and return them to the alkaline fen:

- Shredding of the willows so as to allow the passage of machinery (2 ha).
- Complete filling of 6 small channels, 1,270 m total length with wooden panels and sawdust.
- Neutralization of a large channel, 250 m, with 8 dams made of wooden boards (8x7 m each one, that is to say 350 boards of 1 m + 50 boards of 1.5 m).

Work carried out in July and August 2015.

Figure 6. Restoration of drainage before refilling.

Cost	Monitoring-evaluation
€62,912 taxes not included. Assembly and monitoring of the project excluded.	• Population of *Euphydrias aurinia* and *Lyceana helle* (before and after the works); • Vegetation: 6 blocks (before and after the works).

First results

The first observations showed a clear rise in the water level. The monitoring of the flora and wildlife will allow the effects to be evaluated some time in the future.

Here you can see a short video explaining the works:
http://www.life-tourbieres-jura.fr/images-travaux-page.html

«Prénovel» peat bog in Nanchez (Department of Jura)

Context

- Management: Regional Natural Park of the Upper-Jura.
- Description of the site: bog of 11 ha, occupied by a set of alkaline fens - *Molinia* meadows on calcareous, peaty or clayey-silt-laden soils (6410 and 6430), tightly interwoven active raised bogs and mires (7110*, 7120 and 7140), crossed by two water courses, the channeled sections of the Trémontagne and the Nanchez.
- Heritage value: The raised bogs continue to show an excellent diversity. In the alkaline fens and wetland pastures *Lycaena helle* and *Euphydrias aurinia* thrive.

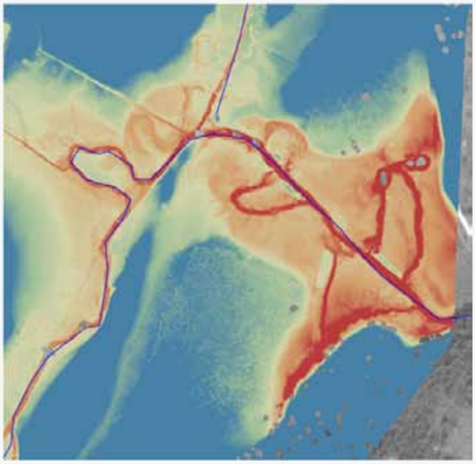

Figure 7. Terrain model using LiDAR.

Figure 8. Old and rectified river course.

Verdict

These water courses were subjected to hydraulic works in 1965. The correction of the channels brought a linear fall of 40 %, a very significant widening and a clear deepening of the riverbeds. Owing to these works a homogeneization of the aquatic habitats was produced and an increase in the speed of drying out of the surrounding surfaces. In this respect, the hydraulic functioning of the adjacent wetland areas and the vegetal communities were modified.

Problems pending solution	Objectives
• Rectilinear river. • Overwidened channel. • Waterline lowered with respect to the top/upper part. • Very uniform substrate and hydraulic conditions, unfavourable to the aquatic fauna and flora. • Excessive lowering of the phreatic layer in the dry season.	Reestablish the morphodynamic functioning of the water course to approximately a natural situation, restoring the trajectories and the height of the flows. Elimination of the draining effect of the channel in the dry season by restoring the interchanges with the phreatic level. Restore a functional phreatic level in the whole of the peaty zone by blocking the channels that cross the bogs around the stream.

Figure 9. Restoration of drainage before refilling.

Figure 10. Drainage and blocking dam.

Figure 11. Channel.

Works

A comparison of old aerial photographs and a fine topographical study of the land with the LiDAR tool showed the trajectory of the old meanders. The works were organized into the following phases:

- Felling, shredding and uprooting of the woody stumps along the corrected channel and other channels.
- Shredding of the herbaceous vegetation so as to be able to visualize correctly the microtopography during the reconsideration of the works.
- Supply of materials tn the edge of the sections to be filled, taking advantage of a snowfall so as not to damage the ground (approx. 6,000 m^3).
- "Clearing" of the old meander to be reopened. The materials extracted were used to finish off the filling of the corrected channels.
- Rescue of the fish in the refilled channel so as to conserve them.
- Filling of the corrected channels, taking care to reinforce the points of contact with the meander with spruce fascines.
- Reloading of the channel using pebbles and calcareous gravel to restore a diversity of substrates and flows (1,000 m^3).
- Elimination of 573 m of channel in the bog, via refilling and creation of pools by placing spruce boards and covering them with peat.

Work made between January and August 2016.

Cost

€ 274,584 taxes not included. Assembly and monitoring of the project excluded.

Monitoring-evaluation

- Thermal regime of the water courses.
- Piezometry of the water courses and of the neighbouring phreatic layers (raised bogs and alkaline fens).
- Fish and aquatic invertebrate population.

First results

Works too recent to have any significant results.

It seems that certain channels closed with dams have not totally recovered their water level. The surrounding peat, degraded, is perhaps too porous.

Here you can see a short video explaining the works:

http://www.life-tourbieres-jura.fr/images-travaux-page.html

«Sur les Seignes» peat bog in Frambouhans and Les Ecorces (Department of Doubs)

Context

- Management: Conservatory of Natural Zones in Franche-Comté.
- Description of the site: bog of 27 ha, mainly occupied by a mountain pine (91D0-3*).
- Heritage value: in addition to the mountain pine, presence of notable habitats such as the spruce forest with sphagnums (91D0-4*) and the active raised bog zone (7110-1*). Populations of *Leucorrhinia pectoralis*, *Lycaene helle*, etc.

Verdict

Three hectares of the bog were affected by industrial peat extraction and exploitation between 1968 and 1984 which created extraction ditches where the typical vegetation of the bogs had problems re-establishing itself. After the first phase of the works in 2003, the second began within the framework of the program in 2014.

Problems pending solution

Desiccation of the extraction ditches and colonization by what are called «degraded» groups (forests of birch trees, wetland grazing), that is to say those that do not generate peat.

Objectives

Retain the water in the channels with the aim of allowing the return of peat producing vegetation.

Works

The action was organized into the following phases:

- Creation of two protecting walls made of peat (17 m in total), as well as 3 dams of wooden boards (135 m in total) anchored to the mineral substrate, covered with peat and partially vegetated.
- Installation of overflows on all the newly created works.
- Felling of 2,000 m² of birch trees;
- Scouring of the mineralized peat.
- Creation of pools so as to get the supply of peat necessary to cover the works and to create the protecting walls.

Work carried out in August and September 2014.

Cost

€ 41,895 taxes not included.
Assembly and monitoring of the project excluded.

Monitoring-evaluation

- Photographic monitoring.
- Vegetation (before and after the works).
- Odonata population.

Figure 12. Construction of a dam.

Figure 13. Filling.

Figure 14. Wooden wall.

First results

Colonization of part of the rafts, recently waterlogged because of the scarce *Utricularia stygia*.
The works seem to be very watertight. However, the lack of vegetalization of the peat that covers the dams has created exposed sections and there are fears of progressive erosion. Different actions with the aim of limiting this phenomenon are being contemplated.

Figure 15. Six months after the restoration.

Figure 16. One year after the restoration.

«Les Douillons» peat bog in Nanchez (Department of Jura)

Context

- Management: Regional Natural Park of the Upper-Jura.
- Description of the site: bog of 23 ha, occupied mainly by *Molinia* meadows on calcareous, peaty or clayey-silt-laden soils (6410) and Hydrophilous tall herb fringe communities (6430). Moreover, there were some sectors of degraded vegetation in the active raised bogs (7120) and bog woodland (91DO).
- Heritage value: the bog is host to two species of community interest, *Euphydrias aurinia* and *Leucorrhinia pectoralis* (probably one of the most important populations in the massif).

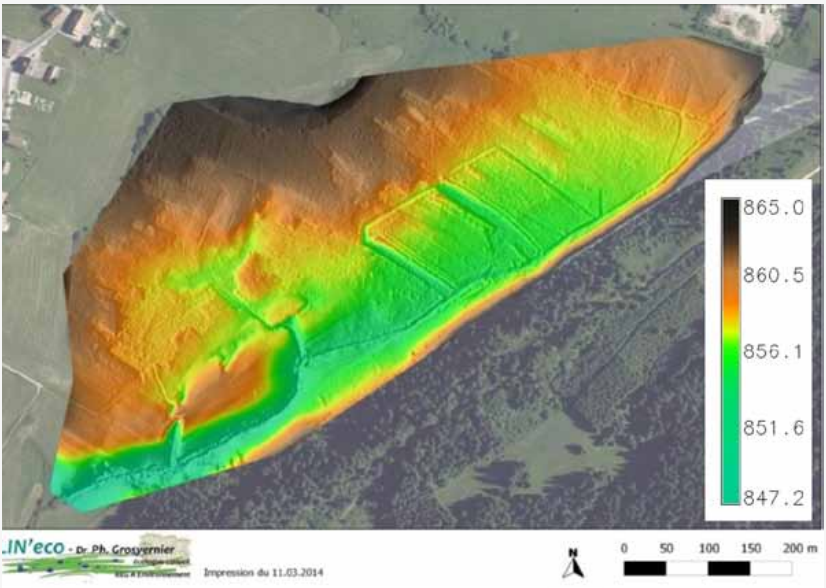

Figure 17. Terrain model using LiDAR.

Verdict

The Douillons bog is of limnigenous origin at the lower part and the remainder soligenous.

A first phase of extraction, domestic, continued until the 1950s. After that there was a large phase of drainage at the end of the 1960s, which was followed by industrial exploitation, of horticultural peat, in 1973. The latter stopped soon afterwards leaving the bog to its own ends, resulting in numerous disruptions.

Problems pending solution

Drainage channels and ditches for peat extraction that disturb the correct functioning of the bog.

Objectives

Partially restore the hydraulic functioning of the bog with the aim of allowing better water retention, as well as reactivating the peat formation dynamics and therefore carbon storage.

Works

The whole of the project and the sizing of the works owed much to the fine topography of the terrain carried out with the LiDAR tool. In this way, it was possible to accurately model the hydraulic effects and the rehumidified sectors and to give suitable dimensions to the works in fine detail. The work was organized in the following phases:

- Felling and/or shredding of the woody material from the sides of the channels that were going to be restored.
- Placing of wooden dams made of spruce boards covered with peat (45 m).
- Placing of metallic dams covered with peat (60 m): in the sectors where the height of the retained water was significant. These have a better anchoring in the mineral substrate (clay) and were installed without any previous excavation (directly anchored in the peat and the underlying layers).
- Creation of a marl and clay dam (60 m): maintaining a general humidity for the peat, perhaps also a layer of emerging water which reaches the old extraction ditches. Dam directly placed in the blue clay after digging ditch until the peat was exposed.

Figure 18. Pit covered by water in the bog.

Work carried out in two phases: reopening in January 2016 and functional rehabilitation works in July and August 2016.

Cost	Monitoring and evaluation
€ 138, 225 taxes not included. Assembly and monitoring of the project excluded.	Odonata population.

First results

Works too recent to have the first results.

«Creux au Lard» peat bog in Frasne (Department of Doubs)

Context

- Management: Association of Aquatic Zones of the Upper-Doubs.
- Description of the site: bog of 26 ha, occupied by an active raised bog (7110*), a transition mire (7140) and bog woodland with mountain pine and spruce (91DO).
- Heritage value: bog included in a natural regional reserve approved because of its wealth of flora and wildlife. Worth mentioning is the presence of: *Drosera rotundifolia*, *Scheuchzeria palustris*, *Carex limosa*, *Andromeda polifolia*, *Aeshna subarctica*, *Colias palaeno*, etc.

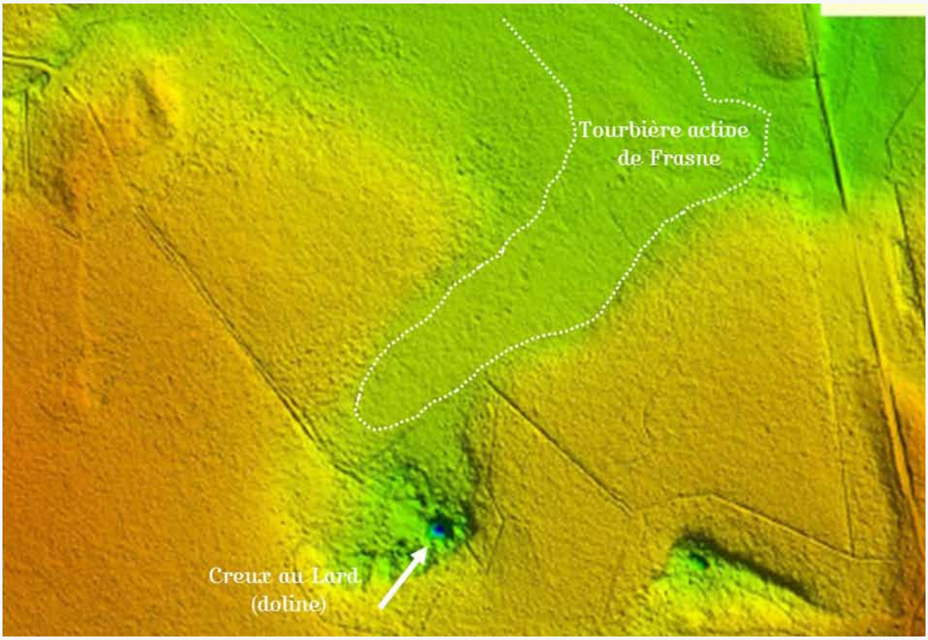

Figure 19. Terrain model using LiDAR.

Verdict

A drainage network that modifies the surface flows across the bog takes the water to a karstic sink instead of the active bog (7110*). This drainage network generates the topographical sinking of the peat which has reinforced this channeling of the flow of the surface water towards the sink. So, it has lost 8 ha of watershed slope going towards the active raised bog that fed it.

Problems pending solution

- Progressive desiccation of the bog with advance of mountain pine and spruce towards the centre;
- Lowering and greater fluctuation of the phreatic level.

Objectives

Reestablish the surface circulation of the waters and limit the drying of the bog by neutralizing the drainage network and re-establishing a topography that allows the waters to once again be returned to the active bog.

Works

As the site has a significant depth of peat, it was possible to construct enough obstacles, via dams of wooden boards and wooden panels anchored to the peat substrate, in the drainage network. In the zones where the gradient was over 1.5 %, the channel was neutralized completely by filling it with peat, and blocking it with wooden panels. When the substrate was mineral (clay, marl, moraine), the material used for the work should not go through, or into, this hermetic layer as it can create underlying leaks. Similarly, in a sector where this layer was very fine, the choice was to preferably install a metallic stockade. The work was organized as below:

- Total filling of 570 m of drainage channels with peat extracted *in situ*, with the addition of 10 wooden panels and 4 dams made of wooden boards.
- Partial neutralization of 250 m of channel by creating 11 dams of wooden boards covered with peat extracted *in situ*.
- Construction of a metal dam 90 m covered with peat, with the aim of neutralizing the topographical sinking of the peat.
- Creation of pools to collect the peat used to fill the drain and to cover the works.
- Reopening (felling of woody plants) so as to allow access to the works and compensation paid to the owners of the trees for the loss of income.

Work carried out between July and September 2015 and from May to June 2016.

Cost

€ 212,125 taxes not included, + € 28,000 taxes not included, for the felling of the trees and financial compensation. Assembly and monitoring of the project not included.

Monitoring-evaluation

- Piezometry.
- Hydrometry and hydrochemistry at the exit of the bog.
- Vegetation: 10 plots for monitoring (before and after the works).

Figure 20. Aerial view of the dams.

Figure 21. Dam.

Figure 22. Dam with peat cover.

Figure 23. Blocking dam using a metal wall.

First results

Although the works are still recent, it seems that the metal dam installed in 2015 is carrying out its role and is waterlogging the system up to the desired level. The *Sphagnum* mosses are developing in the group of areas worked in 2015, with very favourable rainfall in spring of 2016. However, it is necessary to wait for a dry period to see how effective the works are.

Here you can see a short video explaining the works:

http://www.life-tourbieres-jura.fr/images-travaux-page.html

Conclusion

Over several years the managers of the bogs in the Jura Franco-Comté massif have become aware of their role in the region for the preservation of bogs at national scale. In this respect, several European programs have been linked together and numerous restoration actions have been undertaken, often with technical and scientific support from Swiss collaborators. The LIFE+ Bogs of the Jura reflects the materialization of these dynamics. Taking advantage of the experience acquired by the whole set of agencies involved, the hydraulic functioning in the restoration has been emphasised and its fundamental importance in these zones has gradually been recognized.

At times, the Jura bogs have undergone significant modifications during their rehabilitation. However, the objective of the program was to recover the functioning of the ecosystem at the limit of the technical and ecological possibilities that each site offered. Hence, different techniques were implemented according to each habitat and the problems encountered. Some had been used successfully for several years in the Franco-Comté sites (eg: wooden dams covered with peat), and others were used for the first time in France thanks to this program: metallic dams, jointing of wooden panels. Naturally, the impact of these works will be analyzed over the period of the program thanks to a certain amount of monitoring: piezometry, water quality, fauna, flora, etc. However, in spite of the relatively long duration of the program, the evolution of the later works in the habitats and their state of conservation will be limited at the end of a six year period. There will be a wait of several years, even decades before the "definitive" results of the works in the program can be observed, especially regarding the restart of the peat making processes.

All the knowledge acquired, techniques used and the results have been compiled and analyzed in a record of experiences up to the end of the program. This compilation will allow the transfer of good practices and increase, after a period of time, the efficiency of the management in other zones with bogs that have the same problems, at regional, national and even European level.

Lastly, awareness and communication actions have been developed over the six years of the implementation, since the involvement of the public and local agencies is essential for the long-term preservation of the bogs. In this respect, a 24 minute video has been filmed, completed with seven extra themes, and in addition there is an exhibition, made up of 11 panels, several reports about the project and numerous animations.

Further information at the Internet website:
www.life-tourbieres-jura.fr

08

Landscape-scale restoration of severely eroded blanket bog in the South Pennine Moors, England

Moors for the Future Partnership[1]

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Abstract

Moors for the Future is a partnership that was created in 2003 to protect and restore the degraded landscape of the moors of the Southern Pennine mountains. Since then, it has restored more than 30 km^2 of bog in the moors of this area, with the objective of reversing the degradation processes initiated with the Industrial Revolution and achieving a good conservation status for the peat bogs. Previously, the Southern district of the Pennines was characterized by large areas of exposed peatland in process of erosion, with an extensive network of drainages and channels. The initial targets, therefore, focused on stabilizing the deposits of peat and curb erosion. Subsequently, efforts have moved to increase the diversity of species, favouring those typical of peat bogs and improving the necessary conditions in the peat for these species to recolonize.

Keywords: restoration; bog; peat; Pennine Moors; ecosystem services.

Description of the South Pennine Moors Special Area of Conservation

In England the largest expanses and deepest deposits of peat are found on the Pennine plateau, between 190 and 893 m above sea-level (Jarvis *et al.*, 1984). This includes the plateaux of Bleaklow, Black Hill and Kinder Scout in the Peak District National Park (PDNP) which, having started to form 5,000 – 7,000 years ago, are also amongst some of the oldest peatlands in the UK (Tallis, 1995).

The South Pennine Moors (SPM), a Natura 2000 site and Special Area of Conservation (SAC), include extensive blanket bog communities amongst a mosaic of moorland habitats. Lying between Sheffield, Leeds, Nottingham and Manchester these peatlands are locally important for recreational and economic opportunities as well as potentially having a significant role in flood risk mitigation for surrounding villages and towns such as Hebden Bridge, Glossop and Derby. They are nationally important in terms of their water regulation, with 70 % of UK drinking water originating from often peat dominated uplands (Bain *et al.*, 2011), and have international importance for carbon cycling (Lindsay, 2010).

Figure 1. Location of the South Pennine Moors SAC.

History of damage

Currently, only 4 % of the blanket bogs of the SPM are considered to be in 'favourable condition', while 93 % are considered to be in 'unfavourable recovering condition' (Natural England, 2016). A variety of external factors over the last 200 years have contributed to widespread loss of key peat-forming vegetation species, most notably *Sphagnum* mosses, and subsequently to severe erosion of the peat mass, with sediment yields for eroding peat catchments exceeding 100 t km^2 yr^{-1} (Rothwell *et al.*, 2008).

Industrial pollution

The initial cause of degradation of the SPM's blanket bogs is understood to have been atmospheric pollution during the industrial revolution emanating from the major industrial conurbations surrounding the PDNP (Tallis, 1964). Deposits of sulphur and other pollutants resulting from heavy industry led to losses of vegetation, in particular *Sphagnum* mosses, and a significant acidification of the peat surface, inhibiting the growth of other species. This loss of vegetation cover exposed large areas of bare peat without root networks to provide stability, leading to widespread erosion by wind and water.

Subsequently, an increasingly dense network of drainage channels formed through the erosion of the loose peat, facilitating the transport of peat away from the plateau. The increased drainage efficiency caused a lowering of the water table and a desiccation of the peat surface, accelerating erosion.

Fire damage

Fires have occurred in the SPM due to accidental wildfires resulting from recreational use of the uplands, and as part of deliberate land management practices (rotational burning) around grouse farming for shooting, where small areas of heather are burned in carefully controlled fires. Burning has also been used to create better grazing conditions for livestock, often burning much larger areas than the rotational burning, which tends to create patchworks of mixed short and long vegetation.

Peat is an effective fuel and therefore deep peat blanket bog is highly susceptible to fire damage. During combustion, the peat structure dries out, vegetation and associated root networks and seedbanks are damaged or destroyed and carbon from the peat mass is released into the atmosphere. While the most severe damage to the peat body results from uncontrolled wildfires, all burning on deep peat can contribute to peat loss (Holden, 2012).

Additionally, managed burning may lead to dominance of burning-tolerant species, such as *Calluna vulgaris*, as they may be more able to recover following burning than species which suffer more long-term damage, including blanket bog specialists such as *Sphagnum* mosses. This may increase potential fuel loads and also may increase the severity and extent of wildfires.

However, managed burning may also be used to reduce potential fuel loads by removing areas of tall, woody vegetation, and to introduce firebreaks, and therefore reduce the severity and extent of wildfires (Yallop *et al.*, 2006).

Figure 2. Bleaklow (2009). Historic damage led to large expanses of bare peat in the South Pennine Moors.

Overgrazing

The overgrazing of livestock, particularly sheep, on the SPM plateau has restricted the ability of key vegetation species to recover following pollution and erosion damage. The first notable expansion of sheep numbers was around 1550 AD (Shimwell, 1974), with further dramatic increases in stock numbers since the 1930s (Anderson *et al.*, 1997). Livestock may eat existing and young plants, and also cause disturbance to the peat surface, especially where it is bare and loose, restricting vegetation regrowth and increasing erosion.

Climate

The SPM is the UK's most south easterly incidence of blanket bog, meaning that its structure is in a more marginal climatic location than other peatlands further north, and may be affected sooner by climate change. For a large proportion of the year, temperatures fluctuate around 0°C, leading to regular formation of needle ice, which separates peat layers, causing 'frost heave', desiccation and erosion (Buckler *et al.,* 2013).

Multiple Benefits of the South Pennine Moors

The SPM provides multiple ecosystem services to humans, wildlife and the environment, both within and beyond the SAC, although its degraded status reduces these benefits, and presents potential hazards as a result.

Carbon

UK peatlands hold 40% of all national terrestrial carbon, in only 8% of its land area, representing the equivalent of 20 years of all UK CO_2 emissions (Lindsay, 2010). When in healthy condition, active blanket bog acts as a carbon sink, making a positive contribution to climate change reduction as atmospheric carbon dioxide levels are reduced.

However, the loss of vegetation in the SPM, and subsequent drying and erosion of the peat has led to elevated levels of particulate and dissolved organic carbon (POC and DOC) entering the fluvial systems. The majority of both POC and DOC has been found to 'de-gas' into CO_2 either while in the fluvial system or when re-deposited on floodplains (Evans *et al.*, 2013). These emissions have transformed the SPM into a significant source of CO_2, as the store of 'locked in' carbon is released back into the atmosphere.

Natural flood risk management

Blanket bogs are characterised by at/near-surface water tables, with low available water storage capacity as compared to other systems. As a result, catchments with a high proportion of blanket bog often exhibit a rapid runoff response, although the dense mats of mixed vegetation present on healthy blanket bogs slow down the flow of water over the peat surface. Intact systems also often have relatively sparse drainage networks, meaning that the time taken for any individual raindrop from landing on the surface to arriving in a stream channel may be relatively long.

In degraded peatlands, the bare peat surfaces generate much faster surface runoff, and the severe extent of historic erosion has led to a dense network of drainage channels, gullies and subsurface peat pipes, meaning that the raindrop's travel time to the stream channel is significantly reduced. Therefore, the 'flashiness' of peatland hydrological systems increases with the level of degradation, resulting in an increase in the severity of flood events further down the river system (Grayson *et al.*, 2010).

Biodiversity

The SPM comprise a mosaic of habitats including blanket bog, upland heathland, acid grasslands and upland woodlands and are of great importance for biodiversity. The moors support plant and bird assemblages of national, European and international importance which are protected under UK and European conservation legislation.

The historic damage to the peatlands due to industrial pollution has led to the severe decline of moorland specialist insect, mammal and bird species relying on vegetation for cover and food which, in turn, led to the decline of raptors and other predators.

Water quality

The SPM, as uplands receiving high rainfall surrounded by areas of high population density, are important sources of freshwater for humans as well as aquatic flora and fauna. The PDNP alone holds 55 reservoirs, with associated abstraction licences totalling more than 450 billion litres of raw water per year (Bonn *et al.*, 2009).

Compared with other blanket peats globally, those of the PDNP are amongst the most contaminated in the world following the industrial revolution, with pollutants including lead, zinc, nickel and titanium (Rothwell *et al.*, 2005). The peat surface represents a store of these contaminants which, if stable and vegetated, would restrict their mobility. However, the extensive areas of bare peat produce faster surface runoff, which generates higher rates of erosion and therefore higher levels of contaminants in the water flowing out of the moors.

The financial cost of decontaminating this water to make it suitable for drinking is considerable, meaning that there is a significant economic incentive for the utility companies managing the reservoirs to invest in restoration and erosion control works in the headwater catchments.

Wildfire risk

Wildfire is one of the top 25 priority risks to UK biodiversity and causes substantial environmental and economic losses in upland habitats (Sutherland *et al.*, 2009). In the SPM there are approximately 50 wildfires per year (Walker *et al.*, 2009), mostly resulting from arson or carelessness, and which are particularly frequent in highly visited areas. Healthy active blanket bogs, where water tables are high and vegetation communities are diverse, are relatively resistant to wildfire. However, in the degraded peatlands of the SPM, extensive areas of bare dry peat, combined with large areas of single age single species vegetation, constitute high fuel loads, meaning that wildfires, once established, can easily get out of control and cause widespread damage to wildlife, habitats and the peat body.

Health and wellbeing

Uplands provide health benefits to humans through the activities undertaken within them, while also providing more 'passive' benefits for mental and emotional health. Climbing and walking, for example, provide both physical and mental health benefits, as people experience reductions in stress both from the exercise and from the tranquillity and beauty of the landscape around them. The PDNP, in close proximity to several major urban conurbations, all with easy access by road and rail, receives around 22 million visitor days each year. The park, while offering benefits to health and wellbeing to a large number of people, is also impacted by this weight of visitors. If paths are not robust and well-maintained, erosion from trampling can be severe.

Overview of treatment

Moors for the Future is a partnership which formed in 2003 to protect and restore the degraded landscapes of the SPM. The work of the partnership to protect peat moorland is delivered by the Moors for the Future staff team through the Peak District National Park Authority as the lead and accountable body, with support from partners including environmental protection and conservation charities, governmental bodies, utility companies, a rural regeneration company and representatives of the moorland owner and farming community.

Since its formation, the Moors for the Future Partnership (MFFP) has undertaken restoration works on over 30km^2 of moorland, with the intention of halting the degradation initiated by the Industrial Revolution, and setting the moors on a trajectory of recovery to active, healthy blanket bog. When restoration work began, the SPM was dominated by large expanses of bare and eroding peat with deeply incised gully networks. The first priorities were therefore to stabilise the peat surface and reduce the rate of erosion. Subsequently, work has focused on increasing the diversity of vegetation species, by propagating and planting key moorland species, and improving the condition of the peat so that these species are able to re-colonise.

Restoration techniques

Bare peat stabilisation / revegetation

The stabilisation of bare peat has been achieved by applying a combination of cut heather brash and a seed mix of fast-growing amenity grasses. Over 75,000 bulk bags of brash (0.75 m^3 per bag) have been transported to the restoration sites by helicopter and then spread by hand to a depth of approximately 1 cm using gardening forks, covering a total of 500 hectares of bare peat. The amenity, or 'nurse crop', grasses (*Agrostis capillaris*, *Deschampsia* *flexuosa*, *Lolium perenne* and *Festuca ovina*) are applied by helicopter using an air drill, along with lime and fertiliser, which are necessary to reduce acidity and supplement the nutrient-poor peat surface. Further applications of lime and fertiliser are spread in the following two years, in order to sustain the nurse crop grasses in the otherwise hostile conditions.

The brash, harvested from local heather moorland sites to avoid introducing pests or diseases from outside the SAC, provides a protective layer over the peat, reducing wind and water erosion. This

Figure 3. Nurse crop grasses are seeded by helicopter, and subsequently grow up through the layer of brash, knitting the peat surface together.

stability allows the nurse crop grasses, along with other species present in the seedbank within the heather brash, to root into the peat and grow up through the brash, 'knitting' the brash and peat surface together. On steep slopes, such as gully walls, where brash will not remain in place, over 250 km of biodegradable geo-textile fabric has been used to stabilise the peat surface and allow the nurse crop grasses to establish.

Once the nurse crop grasses have established, growing conditions become significantly more favourable for more natural moorland species such as cotton grasses, mosses and moorland shrubs, which may then recolonise from the heather brash seedbank. This recolonization has been accelerated and supported by the planting of over 750,000 plug-plants of key moorland species, including *Vaccinium myrtillus*, *Empetrum nigrum*, *Rubus chamaemorus*, *Eriophorum vaginatum* and *Eriophorum angustifolium*.

In addition to the revegetation of bare peat, a further 2,500 hectares of semi-degraded moorland has been treated with lime, fertiliser, grass seed and plug plants in order to improve vegetation density and heterogeneity.

Gully blocking

The dense networks of gullies and stream channels dissecting much of the SPM create highly efficient drainage systems, resulting in significantly lowered water tables. MFFP has installed over 13,000 gully blocks along 120 km of gullies, in order to reverse this trend by trapping water and/or sediment, slowing the flow of the water and/or raising the water table.

Blocks are installed in sets along gully systems, starting at the head of a gully and working down the channel, such that the top of each dam is at least level with the base of the next upstream dam. In this way, water flowing over the top of a dam falls onto a water surface rather than onto bare peat or mineral soil, reducing the erosive impact of water flow. This means that the spacing interval has

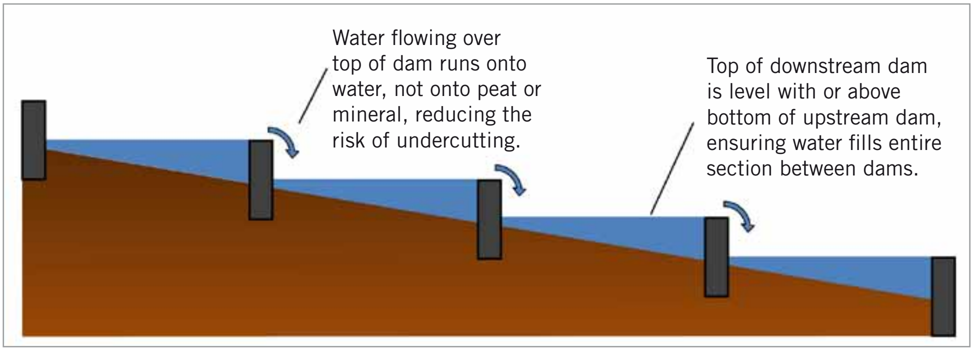

Figure 4. Dam spacing.

to be shorter as the gradient increases, with fewer dams required on gentle slopes.

Dams fall into two broad categories:

• *Impermeable dams,*

made using either plastic sheet or peat excavated from on-site, are designed to trap water to create pools (and raise the water table).

• *Permeable dams,*

made using heather bales, piles of stones or planks of timber with small gaps, are designed to trap sediment, which will build up soil on the gully floor to be re-vegetated. They also slow (but not stop) the water flow, allowing water to drain slowly down the channel during low-flow conditions. Water storage capacity is created for rainfall events, delaying the flow of stormwater to the river networks downstream and reducing flood impact.

Figure 5. Semi-permeable stone dams limit sediment transport and slow stormflow from the moors.

No one dam type is "better" than the rest for every situation; each has its place, dependent upon the ecological objectives to be achieved, the nature of the site (for example gully profile and peat depth) and their effect upon the long-term management of the site. The table below summarises the relative strengths and weaknesses of gully block methods used by MFFP, ranging from green (favourable) to red (less favourable).

With the exception of peat dams, materials are airlifted onto site and ground vehicle access is not required. Peat dams require ground access for one or more excavators, but do not require con-

DAM TYPE	Creates pools	Traps sediment	Uses natural materials	Availability of materials	Installation easy and not disruptive	Resistance to fire	Robustness (5 yrs after installation)	Cost
HEATHER BALE	yellow	green	green	yellow	yellow	green	green	yellow
PEAT	green	yellow	green	yellow	yellow	green	green	green
PLASTIC	green	yellow	red	green	green	yellow	yellow	yellow
STONE	yellow	green	green	green	green	green	green	red
TIMBER	green	green	yellow	green	yellow	yellow	yellow	yellow

Table 1. Strengths and weaknesses of gully block methods.

struction materials to be brought in; local peat is re-profiled to create these dams.

In some areas, stone dams have been found to have accumulated peat sediment up to the top of the dam, meaning that the dam ceases to prevent further transport of eroding sediment out of the catchment. Where this is the case, MFFP have started to install 'top-ups' to these dams, increasing the height of the dam and creating further sediment and storm water storage capacity.

Sphagnum mosses

Following the initial phase of revegetation and gully blocking, MFFP has started to trial various methods for re-introducing *Sphagnum* mosses to degraded areas. These mosses have an important role to play in restoration work, as they increase surface roughness, help maintain high water tables and are the key species in the formation of peat on blanket bogs. It is hoped that their reintroduction to damaged blanket bogs will help to return these habitats, which are currently sources of carbon, to functioning ecosystems that are actively sequestering carbon.

Since 2012, *Sphagnum* has been applied to 14 km^2 of sites with suitable ground conditions using the following methods: beads, slime, plug-plants and hummocks (Fig. 6).

The application of *Sphagnum* is a relatively new process and, due to the slow-growing nature of these mosses, it is not yet clear which method is the most effective/efficient. Once this is established, *Sphagnum* application will be carried out across the SPM.

Footpath improvements

Poorly maintained footpaths across sensitive areas of moorland have degraded to form wide areas of bare and disturbed peat due to the impact of the high numbers of visitors that the SPM receive. This disturbance severely inhibits the possibility of vegetation recovery and erosion control, and has been remedied by creating

Figure 6. *Sphagnum* application methods (clockwise from top left): beads, slime, plug-plants, hummocks.

Beads
Strands of moss are encapsulated within beads of liquid.

Hummocks
Clumps of moss are harvested from donor sites with abundant *Sphagnum*, transported to treatment sites and planted by hand.

Slime
Strands of moss suspended in liquid, applied by spraying.

Plug-plants
Small established plants propagated off-site, transported to site by helicopter and planted by hand.

more durable footpath surfaces which can contain the flow of walkers in a smaller footprint without causing erosion.

Over 50 km of footpaths have been improved by installing flagstones (re-cycled from disused mills, thereby transforming infrastructural materials from an industry which contributed to the degradation of the moors, into part of their restoration); pitched stone (blocks sunk into the peat to create a flat upper surface); and compacted aggregate (from locally sourced gritstone).

These stable and solid paths simultaneously improve access to the moors, reduce the footprint of visitors, and allow adjacent areas to recover/be restored.

Figure 7. Flagstone footpath installation across moorland eroding due to visitor traffic.

Monitoring techniques

MFFP's restoration works are supported by an extensive programme of research and monitoring, in order to assess and compare the efficacy of different restoration techniques, quantify conservation successes, identify future targets and contribute to the scientific community's understanding of peatlands. Monitoring focuses include peat erosion/accumulation, stream flow, water table depth, water quality, vegetation and wildlife. A standard monitoring structure of before-after-control is used across all studies, with baseline data being collected before the start of restoration works, and a control site being maintained in an unrestored condition, so that results from ongoing monitoring of the restoration sites may be compared to the untreated site, in order to account for any external factors. Where possible, 'intact' sites are also monitored in order to put restoration sites in context of the target condition, although genuinely 'intact' active blanket bog sites are rare in the SPM.

A range of monitoring methods are used, some of which are standard in this field of study, and some of which have been developed experimentally by Moors for the Future, in collaboration with the University of Manchester and other universities. Monitoring activities are carried out both by members of staff and volunteers, facilitating engagement with local communities, and making it financially sustainable to deliver monitoring programmes at fine spatial and temporal resolutions over a large number of sites.

Peat erosion/accumulation

Rates of peat erosion have been monitored using a range of methods. 'Peat anchors', constructed using lengths of M12 threaded rods and connectors, are pushed through the peat into the glacial till beneath, leaving approximately 10 cm standing proud of the bog surface. Measurements are then made from the bog surface to the top of the crowning connector. If restoration works are successful and the moors return to active blanket bog status, moorland species such as *Sphagnum* mosses may lead to peat accumulation being recorded at restoration sites, although as active blanket bogs accumulate peat mass at a rate of around 1 mm per year (Lindsay, 2010), change will only be observable in the long-term.

Figure 8. A peat anchor, used to monitor peat erosion/accumulation.

Rates of erosion are also monitored in gullies, as this is where sediment transport may be observed. In collaboration with the University of Manchester, a 'Time Integrated Mass Flux Sampler' (TIMS) unit was developed for this purpose. A 50 cm length of plastic tubing, filled with polystyrene chips and enclosed at each end with 8 mm mesh, is secured to the gully floor. Stream water entering the trap is slowed by the large surface area of the polystyrene and suspended sediment is deposited within the pipe. Sets of traps are installed across the study sites at the same time and are left *in situ* for a fixed time period, before being collected at the same time. Once removed, each trap is flushed out and cleaned using de-ionised water. This water is then passed through a filter, until only sediment is retained. This is then oven-dried, and the resulting sediment is weighed to an accuracy of 0.01 g. Sediment yields from sites with different restoration histories may then be compared against unrestored and intact control sites.

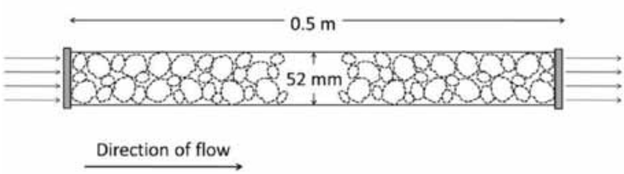

Figure 9. TIMS unit used to monitor sediment yields in stream channels.

The efficacy of gully blocks at limiting sediment transport is monitored by measuring the depth of peat immediately upstream of dams, as well as at selected points in untreated gullies. This is done immediately after gully block installation and then repeated at regular intervals. Peat accumulation, erosion and re-deposition are monitored using LiDAR (Light Detection and Ranging) data to generate precise, three-dimensional Digital Elevation Models. Using this technology, the topography of study areas may be modelled before restoration, and then re-modelled during the restoration process, revealing whether/where peat is accumulating, eroding and being deposited.

Stream flow

The potential of restoration works to reduce flood impact severity is monitored at the stream outlets of micro-catchments in river system headwaters, and also at points along the rivers further downstream. Depending on channel size, either v-notch weirs, flumes or rated sections are installed/developed, in combination with the installation of depth-sensing loggers. This enables the generation of continuous, high temporal resolution flow data, including in storm conditions, when manual data are logistically challenging to collect.

By installing these flow stations in sites which have received a range of restoration techniques, as well as in unrestored control sites and 'intact' sites, the impacts of these restoration methods on streamflow response to rainfall may be compared to each other, as well as to the scenarios in which the restoration work had not been done, and the initial damage had not been caused. The monitoring of river flow further down the fluvial system facilitates the assessment of the impact of restoration works on flood severity at the catchment scale over long-term timescales.

Water table

Water table depth is monitored using MFFP's network of over 500 dipwells. Lengths of perforated plastic tubing are pushed into the peat structure so that the tube fills with water to the level of the water table in the surrounding peat mass. The distance from the water surface to the peat surface is measured either manually or using an automatic depth-sensing data logger, as for streamflow. Water table study programmes are designed such that at each site one automatic logger is surrounded by a cluster of 15 randomly located, manually monitored dipwells.

The manual dipwells are monitored in annual campaigns, in which teams of staff and volunteers take measurements from the entire dipwell network on the same day, and repeat these measurements on a weekly basis for a 12-week period in the Autumn of each year. This facilitates the compilation of an unusually high spatial resolution dataset over a wide range of site conditions and stages of restoration, and puts the single-point but high temporal resolution data from the automated dipwells in context. In this way, MFFP is able to monitor long-term water table response to restoration works at high temporal and spatial resolutions.

Water quality

Water quality in the river networks and headwater streams is monitored by collecting water samples during field visits to restored and control sites. An auto-sampler is used to collect samples during storm conditions, which monitors stream stage and collects a series of water samples if a pre-determined stage is exceeded.

Manually and automatically collected samples are then analysed for dissolved and particulate organic carbon (DOC and POC), water colour and a range of chemical pollutants, allowing analysis of the impacts of restoration works, as well as different flow conditions, on fluvial water quality.

Vegetation

Vegetation recovery and species succession is monitored using quadrat surveys. At study sites within restoration, unrestored control and 'intact' reference areas, a series of surveying points are selected randomly, stratified by grid. These points are all surveyed prior to restoration works, by laying a fixed-size quadrat around permanently installed stakes, and recording the percentage cover of all species present within the quadrat. This process is then repeated on an annual basis, so that the revegetation of bare peat may be monitored, and the succession of nurse crop cover to more natural moorland species may be assessed. This method is used both for areas treated with brash, lime, grass seed and fertiliser, and for areas which have received targeted planting of moorland species plug-plants, or the various application methods used for *Sphagnum*.

MFFP has conducted vegetation surveys since before restoration works began in 2003, and data from surveys conducted by external organisations have been added to this dataset for analysis, meaning that vegetation change trends extending over 13 years within a large geographical extent may now be drawn, giving invaluable feedback on the comparative strengths of different restoration techniques.

A new vegetation monitoring initiative is the development of a land cover map for the whole of the SPM using aerial imagery, verified and calibrated using manual vegetation survey data. This will, in future, facilitate landscape-scale analysis of long-term vegetation response to conservation actions as well as climate change.

Results

The results presented here are a brief summary of some key findings from reports published by MFFP, the full details of which may be found at our website:

www.moorsforthefuture.org.uk/research-publications

Peat erosion/sedimentation

Erosion/sedimentation survey datasets show that the installation of gully blocks leads to the rapid accumulation of sediment, and that revegetation leads to a significant reduction in erosion rates. These studies are a recent initiative and so only show initial responses to restoration.

Gully-block sediment depth surveys show that 82 % of stone dams trap accumulated sediment within 18 months of installation, with an average of 14 cm of sediment depth accumulated at gully blocks within the first two weeks of installation, although accumulation rates slowed significantly after this (Maskill *et al.*, 2015c). LiDAR surveys support these findings, suggesting that an average of 17 cm (or 0.6 m^3) of sediment is deposited behind newly installed gully blocks (Crouch *et al.*, 2015).

Results from TIMS sediment surveys show a 99 % reduction in POC in stream water in gully-blocked and revegetated sites as compared to unrestored control sites

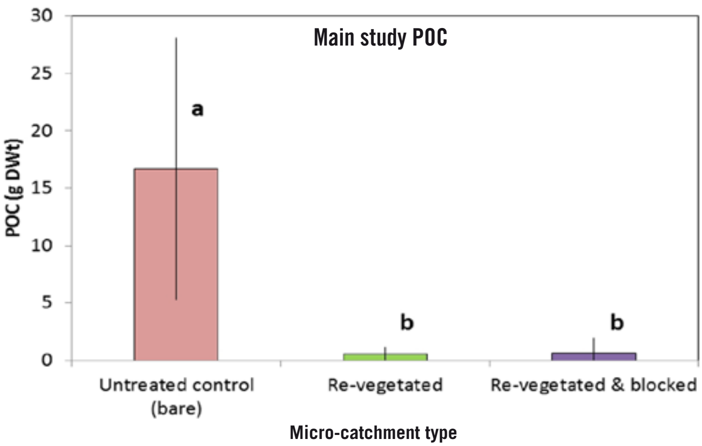

Figure 10. Mean weight of POC trapped in TIMS units, one year following restoration (Pilkington *et al.*, 2015).

within a year of restoration. A 98% reduction in POC was observed in sites which had been revegetated but where gully-blocking had not taken place (Pilkington *et al.*, 2015). This suggests that the majority of erosion reduction is as a result of revegetation as opposed to gully-blocking, as would be expected, and that these gains may be made almost immediately following revegetation works.

The first peat anchors were installed in 2013 and are anticipated to provide long-term data on peat accumulation if/when sufficient peat-building vegetation species have recolonised to start forming a new peat layer. The results collected to date are seen as baseline data.

Stream flow

The key metrics when analysing stormflow hydrographs to assess flood impact are:

- Peak storm discharge (calculated by subtracting the estimated rate of baseflow from the maximum rate of total discharge recorded in a storm).
- Lag time (the time interval between maximum rainfall intensity and peak storm discharge).
- Hydrograph Shape Index (the ratio of peak storm discharge to total storm discharge).
- Percentage runoff (the percentage of total rainfall discharged as stormflow during the storm, as opposed to as baseflow after the storm).

Comparisons between pre-restoration and 'intact' reference sites show that 'intact' sites produce lower peak storm discharge, percentage runoff and HSI and longer lag time than unrestored sites. These differences all suggest a more 'flashy' hydrological regime in the unrestored sites than in the intact sites. Results from the Making

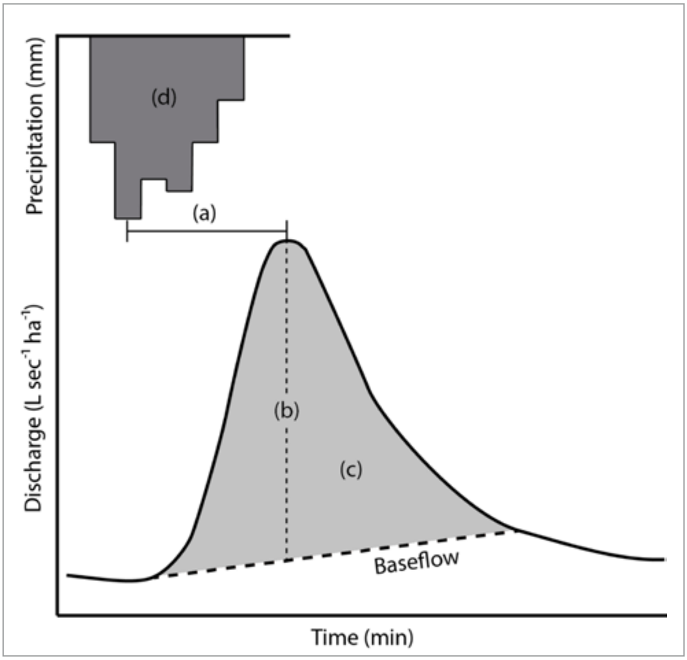

Figure 11. Typical storm hydrograph. (a) is lag time from peak rainfall to peak storm discharge; (b) is peak storm discharge; (c) is total storm runoff; (d) is total rainfall.

Space for Water (MSW) report (Pilkington *et al.*, 2015) show that restored sites 'O' (revegetated) and 'N' (revegetated and gully-blocked), which had behaved similarly to unrestored control site 'F' before treatment, made significant steps towards the behaviour of 'intact' site 'P' shortly after restoration works were completed. Over a 2-year period, peak storm discharge at O and N reduced by 8 % and 37 % respectively as compared to pre-restoration behaviour; lag times increased by 67 % and 267 %, HSI reduced by 19 % and 38 %, and there was no clear change in percentage runoff. These results clearly suggest enhanced improvement at the restoration site 'N', where gully-blocks were installed in addition to the revegetation works applied to both 'N' and 'O'. However, due to wide storm hydrograph variability, these differences were not statistically significant.

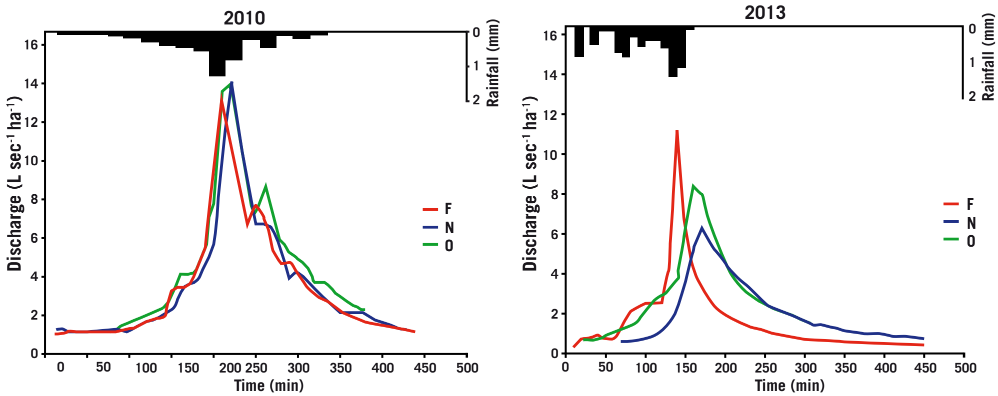

Figure 12. Example of storm hydrograph responses to typical autumn storm events at sites F (bare peat control), N (gully-blocked and revegetated) and O (revegetated only) before restoration (4/11/2010) and after restoration (16/10/2013) (Pilkington *et al.*, 2015).

Water table

Monitoring of water tables has found that water tables at 'intact' sites are approximately 250 mm shallower than at bare peat control sites (Maskill *et al.*, 2015b; Pilkington *et al.*, 2015). The effects of revegetation and/or gully-blocking on water table behaviour may be complicated by yearly climatic variability, as wetter years lead to higher water tables at both unrestored and restored sites. Therefore, the impacts of restoration are measured by comparing the differences in behaviour between control and restoration sites before and after treatment.

The MSW study found that water tables at revegetated sites had risen by 35mm relative to the bare peat control after three years, and by 100 mm after seven years. Although water tables at these late-stage restoration sites are still not comparable to those at 'intact' sites, these results suggest that they are 38 % closer to the surface than they would have been without restoration.

Cumulative frequency plots of water table depth clearly display that water tables at restored sites are more frequently closer to the surface than at unrestored sites.

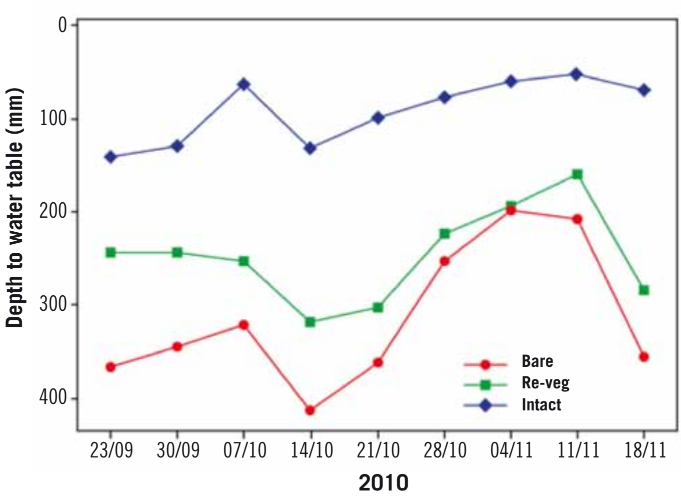

Figure 13. Water table depth at bare peat control, revegetated (7 years) and 'intact' sites, taken from manual dipwell campaign data (Pilkington *et al.*, 2015).

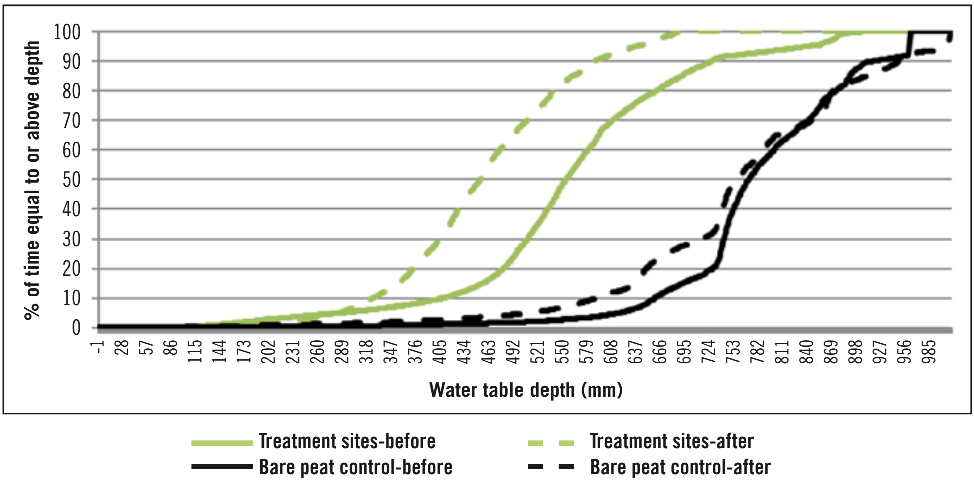

Figure 14. Cumulative frequency distribution for water table height before and after restoration, from automatic dipwell data (Maskill *et al.*, 2015b).

Water quality

Water samples have mainly been collected in low flow conditions, so do not allow analysis of the impacts of restoration on storm water quality. Data collected by MFFP to date are short-term only and suggest that water colour and chemical contaminant concentrations are affected not only by restoration works but also by weather patterns, and severity of peat degradation. Water colour has been found to decrease temporarily immediately following the application of lime (Pilkington *et al.*, 2015). Metal concentrations including copper and zinc also decrease following restoration works, but these may not represent longer-term trends. United Utilities, a local water company, observed small but significant improvements in water colour over seven years at restored sites as compared to unrestored control sites, but also noted the impact on results of year-to-year variation in climatic conditions (Ross *et al.*, 2015).

Results from water quality sampling during storm events suggest that the highest levels of DOC and POC occur during the periods of highest streamflow during storms (Maskill *et al.*, 2015b). While this monitoring has only been carried out to date on restored (and not control) sites, meaning that impacts of restoration may not be inferred, this finding does indicate the importance of increasing peat structure stability to limit in-storm erosion, and reducing rates of storm discharge.

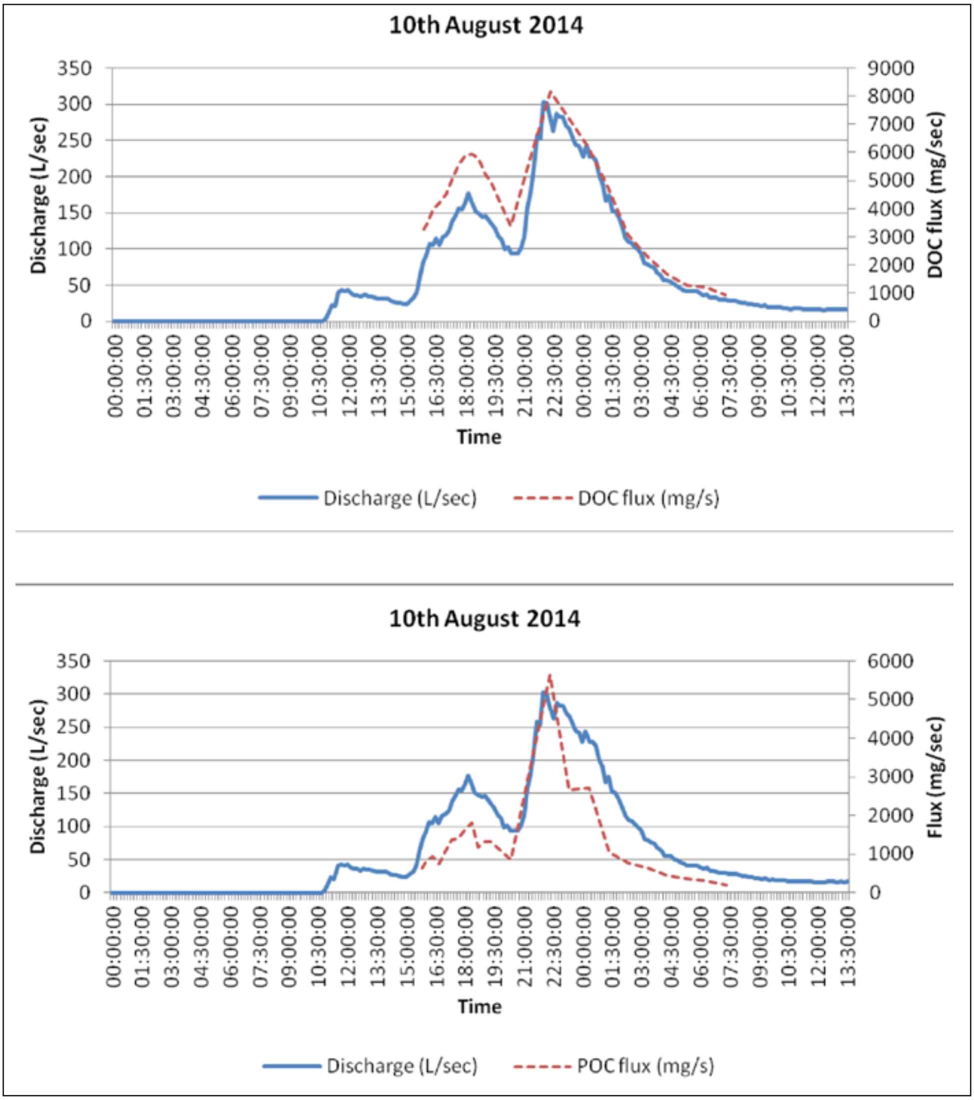

Figure 15. DOC and POC flux during a storm event (Maskill *et al.*, 2015b).

Vegetation

Results from vegetation monitoring show significant decreases in bare peat extent and increases in percentage vegetation cover, species heterogeneity and percentage cover of natural moorland species following restoration works including gully-blocking, application of heather brash, amenity grass seed, lime, fertiliser and plug plants.

Figure 16. The boundary of revegetated/untreated areas. The success of re-vegetation is demonstrated by comparison with untreated bare peat 'control' condition.

In a compilation of results from a number of long-term restoration sites, Proctor *et al.* (2013) found that bare peat extent before restoration is standardly more than 90 %, and decreases to less than 20 % after four years, and to less than 5 % after 8 years. By comparison, bare peat extent at the unrestored control site remains at more than 90 %.

The decrease in bare peat and increase in vegetation cover in the initial years after restoration is due predominantly to successful germination and development of 'nurse crop' grass species. Significant increases in nurse crop cover are clear at all restored sites, although the confidence intervals display the within-site variability of cover, due to features such as steep gully walls being present in some quadrats, where grass seed is less likely to germinate. Following the cessation of lime and fertiliser applications, approximately 3 years after the start of restoration works, nurse crop cover stops increasing at most sites.

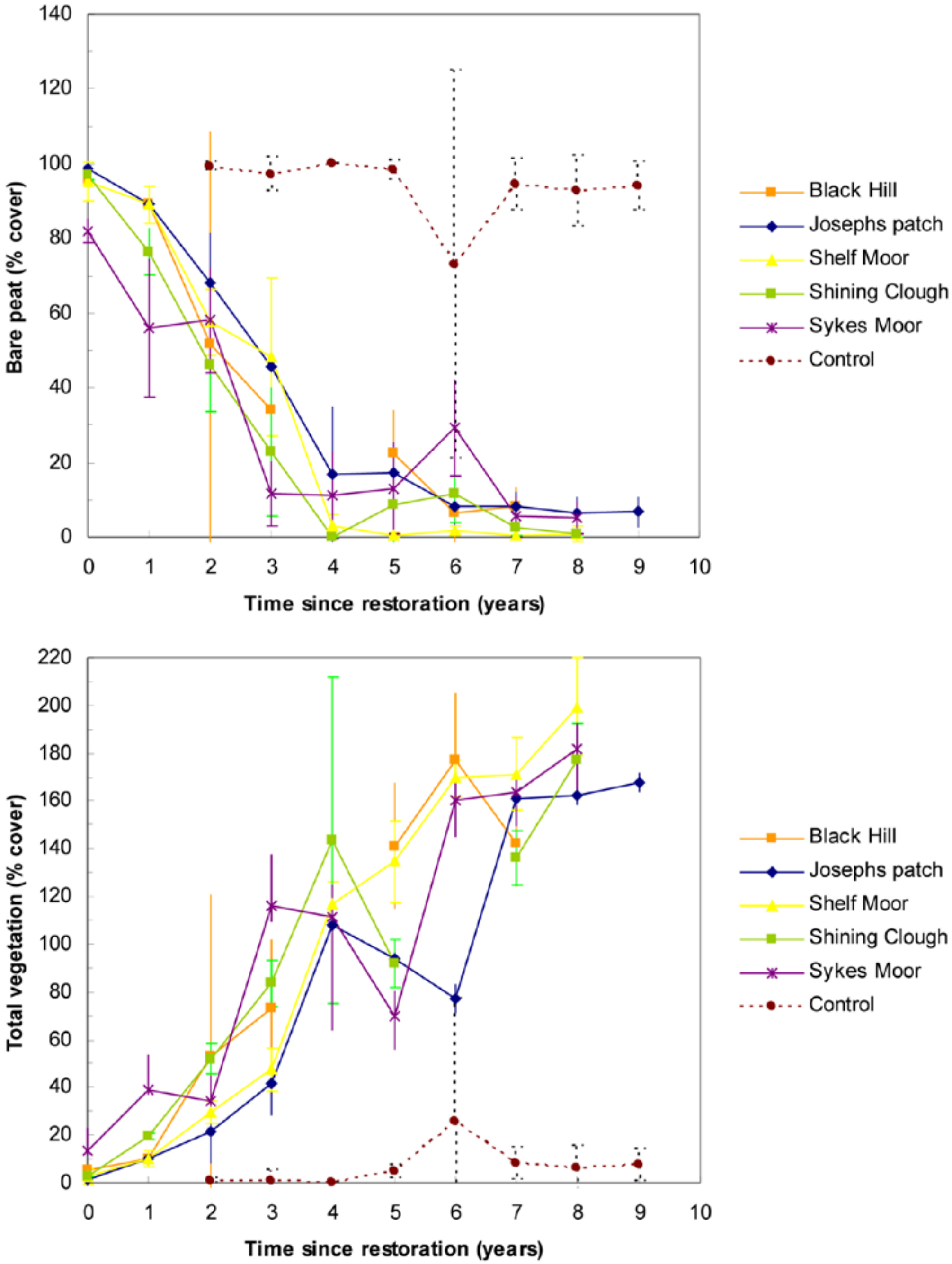

Figure 17. Bare peat % cover reduces and vegetation % cover increases at all restoration sites, but not at the unrestored control site (Proctor *et al.*, 2013).

Figure 18. Transition of a monitored quadrat at Turley Holes in the SPM from bare peat in 2010 to full vegetation cover in 2013 (clockwise from top left: 2010, 2011, 2012, 2013).

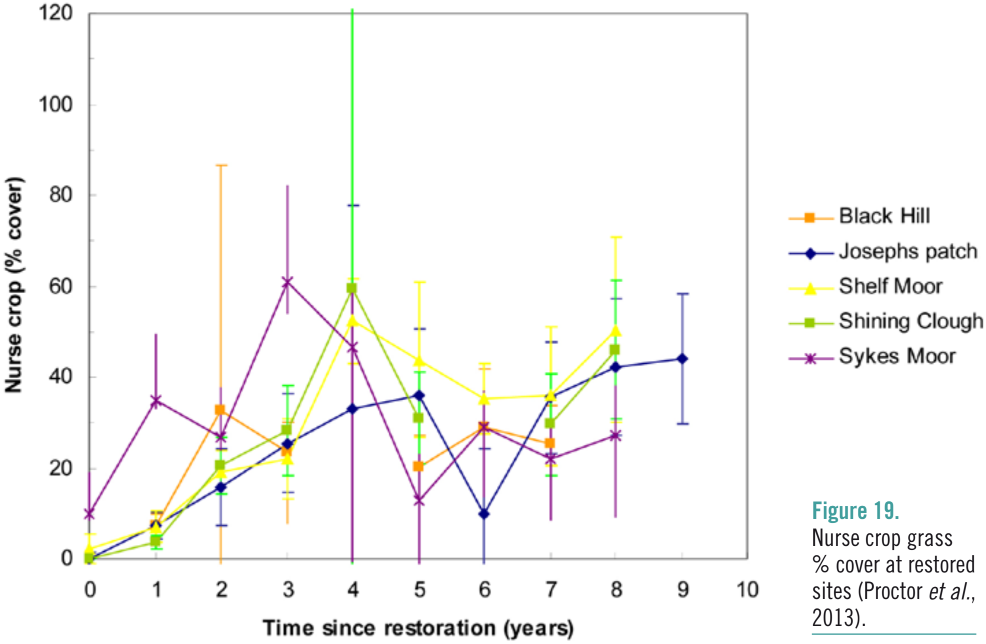

Figure 19. Nurse crop grass % cover at restored sites (Proctor *et al.*, 2013).

In the following years, percentage cover becomes dominated by a combination of *Deschampsia flexuosa*, *Calluna vulgaris* and plug plants (*V. myrtillus*, *E. nigrum*, *R. chamaemorus*, *E. vaginatum* and *E. angustifolium*). Although *Deschampsia* is included in the nurse crop seed mix, it is also considered a natural moorland species, and so is not included in nurse crop cover. The increases in *Deschampsia* and *Calluna* cover may be due to germination from the seed bank present in the heather brash. The rapid increases in plug plant cover from the fourth year onwards are due to the extensive and successful programme of plug planting carried out in the third year of restoration.

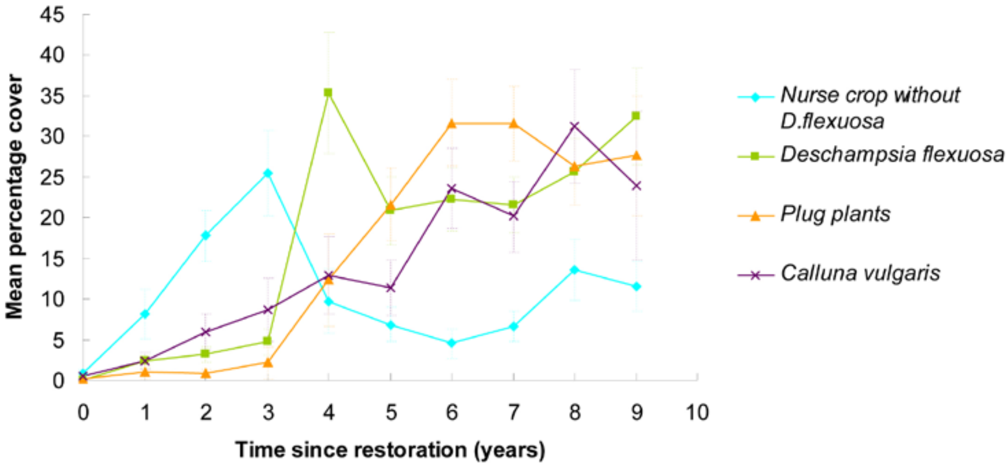

Figure 20. Change in species composition through the restoration process across revegetated sites (Proctor *et al.*, 2013).

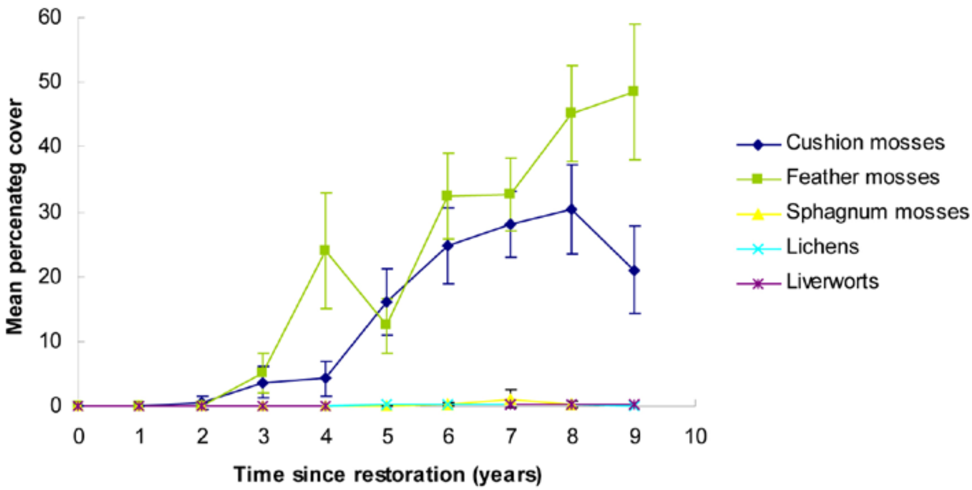

Figure 21. Mean percentage cover change of bryophytes (mosses, lichens and liverworts) across revegetated sites (Proctor *et al.*, 2013).

Percentage cover of bryophytes (mosses, lichens and liverworts) also increases once the growing medium has been stabilised and protected by the nurse crop grasses. Feather and cushion mosses are the most successful, and re-colonise from the seedbank over nine years to a combined mean cover of 75 %, without being actively planted as part of restoration works. *Sphagnum* mosses, despite increasing slightly in mean percentage cover, do not re-colonise significantly, highlighting the need for targeted re-introduction of these key blanket bog species, as well as further re-wetting of the peat surface.

The success of *Sphagnum* re-introduction is currently being monitored and, if the methods being trialled are found to be effective, this would represent significant progress towards classification of the SPM as being in 'favourable condition', under the Common Standards Monitoring guidelines.

Wildlife surveys

Studies by the Royal Society for the Protection of Birds (RSPB) suggest that restoration works are benefitting uplands birds. Repeat surveys from before and after restoration works at Dove Stone nature reserve in the PDNP show consistent increases in numbers of breeding pairs of key moorland species. From 2004 to 2014, numbers of dunlin (*Calidris alpine*), golden plover (*Pluvialis apricaria*) and curlew (*Numenius arquata*) increased by 457 %, 56 % and 56 % respectively, with population increases also recorded for raptor species including buzzard (*Buteo buteo*) and short-eared owl (*Asio flammeus*) (RSPB, 2015).

Implications of results for multiple benefits

The value of the results outlined above is clear when seen in the context of the multiple benefits of the South Pennine Moors SAC.

Carbon

Undertaking restoration works of the intensity and scale described above, especially in remote areas, is a significant logistical operation. Large quantities of materials have to be transported to restoration sites, often by helicopter, and this produces carbon emissions from the combustion of fossil fuels. The longer-term impacts of the works, however, could lead to significant reductions in carbon emissions, due to reduced erosion and the protection and development of the peatland carbon store. In order to assess the net impact of restoration works on carbon emissions, a carbon audit was undertaken by Maskill *et al.* (2015a).

This concluded that, when sources of emissions associated with restoration works both directly (transport of materials and staff to site) and indirectly (production of materials such as lime, fertiliser and stone) are included, the average total emissions is 0.56 tonnes CO_2e/ha/year over a five year restoration programme. Of this, more than 80 % is due to the high carbon intensity of production of materials, in particular lime. However, carbon emissions from eroding peatlands are estimated to be significantly higher than this. Worrall *et al.* (2011) predict that the stabilisation of Bleaklow Plateau in the SPM, transforming the eroding peatland into active blanket bog, will result in a minimum carbon benefit of 4.48 tonnes CO_2e/ha/year. This is 8 times greater than the estimated total emissions associated with the restoration works, and will continue long after the end of the works programme.

The results of TIMS and manual/LiDAR sediment accumulation studies suggest that revegetation is reducing erosion rates and that gully blocks are restricting the transport of eroded sediment from restored sites. As water tables rise, allowing key species such as *Sphagnum* to re-colonise, vegetation will diversify, further stabilising the peat surface, and leading to additional reductions in erosion and transport of sediment.

Natural flood risk prevention

Results of streamflow analysis suggest that restoration works lead to reduced contribution to downstream flood severity within three years of restoration, with average lag times increasing by up to 267 %, average peak discharge decreasing by up to 37 %, and HSI by up to 38 % (Pilkington *et al.*, 2015). These improvements, apparent in storm events of low and high intensity, indicate that the rate at which storm rainfall is entering and travelling through the drainage networks in these restored headwaters is being attenuated.

If this attenuation is sufficiently significant, delivery of stormwater from these headwaters may become 'de-coupled' from the wider catchment hydrograph, reducing peak discharge further down the river network. This may reduce flood impact in conurbations, which would have potentially significant economic and social benefits.

Flood severity is also worsened by sediment deposition in downstream river networks, which reduces river channel capacity, meaning that a lower rate of peak storm discharge results in a flood event. Reductions in sediment production and transport within and from degraded headwaters due to revegetation and gully-blocking works will reduce downstream sediment deposition, further reducing flood severity and the costs of removing sediment from river channels.

Biodiversity

In contrast to the extensive expanses of bare peat dominating the SPM fifteen years ago, the SAC is now almost entirely vegetated. 500 hectares of bare peat have been revegetated, and subsequently improved by re-introducing moorland specialist species. A further 2,500 hectares of semi-degraded (but not bare) moorland have been treated to improve vegetation cover and heterogeneity, setting the moors on a trajectory towards returning to active blanket bog. Revegetation and peat re-wetting works are restoring globally valuable habitat for a range of moorland specialist insects, mammals and birds, several of which are considered by the RSPB to be either globally threatened, or in severe decline in the UK (Eaton *et al.*, 2015). Breeding bird surveys suggest that restoration works including revegetation and re-wetting lead to significant population recovery for a range of moorland specialist birds (RSPB, 2015).

Water quality

Water quality studies have shown significant reduction in fluvial sediment almost immediately following restoration, and small but significant reductions in water colour seven years after restoration. While further work is required to assess the impacts of revegetation and gully-blocking on levels of chemical contaminants in stream-water, links between erosion of the peat surface and the release of previously deposited contaminants suggest that restricting erosion and stabilising the peat surface may serve to reduce the mobility of contaminants from the peat mass into fluvial networks (Shotbolt *et al.*, 2008).

If this is the case, restoration works will provide significant economic benefits to water companies which currently have to remove these contaminants before delivering the water in potable condition. The already-evidenced reductions in sediment content and water colour alone represent significant economic benefits for water companies.

Wildfire risk

The risk of wildfires, especially on deep blanket peat, which can be extremely damaging to the peat structure, vegetation cover and diversity, and wildlife, is reduced by the raising of water tables and the subsequent re-wetting of the peat surface. The transformation from dry, bare peat to saturated, vegetated peat increases the landscape's resilience to the ignition and spread of wildfire.

In areas of deep blanket peat, areas of extensive single-species vegetation cover associated with high fuel-loads, such as *Molinia caerulea* or *C. vulgaris*, have been broken up, with firebreaks cut and vegetation species such as *Sphagnum* re-introduced to increase diversity as well as improving resilience to wildfire by reducing the extent and severity of fire events. These improvements protect biodiversity in the SAC, as well as safeguarding the work by MFFP to reduce erosion, raise water tables, increase vegetation cover and reduce flood risk, much of which would be undermined by severe and extensive wildfire events.

Health and wellbeing

Improvements to footpaths have enabled more visitors to access the moors, allowing a higher number and diversity of people to enjoy the benefits to physical and mental health associated with exercise in upland environments. MFFP's programme of volunteering opportunities has been hugely popular, engaging large numbers of members of the local communities in restoration and monitoring activities in the SAC. This develops a greater understanding of the processes of degradation and restoration, enhancing enjoyment of the landscape and developing a sense of ownership and investment in its recovery.

Economics

Natural England have undertaken an economic assessment within the SPM SAC at Keighley Moor, for the ecosystem services provided under different land use and management interventions and comparison with the potential costs for soil carbon, water quality changes (treatment costs) and woodland carbon. Two scenarios were tested: 1) an improved scenario (habitat restoration and more sympathetic land management interventions); and 2) a decline scenario (withdrawal of public investment in land management and minimum environmental regulations). Using the National Ecosystem Assessment / DECC values, the net present value for the improved scenario is around £6.27 million (the amount society would gain where the investments in the catchment were made) and for the decline scenario it is -£6.77 million (the amount society would lose were all spending withdrawn).

So; for every £1 spent in the catchment, society benefits by £2.96. Conversely, for the decline scenario every £1 not spent in the catchment, society stands to lose an estimated £5.20 (Clarke *et al.*, 2015). Even without the inclusion of possible access and recreation benefits or changes in flood risk, the analysis appears to provide a convincing case for investment in the catchment.

Current and future plans

MoorLIFE 2020

Thanks to the successes of the extensive restoration works programme within the SPM SAC, the large expanses of bare peat have now all been revegetated. Current and future works are therefore focusing on revegetating the remaining smaller and more complicated bare peat areas, as well as consolidating, protecting and improving the large areas of young and potentially vulnerable vegetation.

The MoorLIFE 2020 project, funded by the EU LIFE programme and UK utility companies, will conserve blanket bogs by stabilising and revegetating remaining bare peat areas, halting erosion and encroachment into previously restored areas. Lime, seed, fertiliser, heather brash and plug plants will be used, according to site conditions, to increase vegetation cover and diversity. Steep gully walls are among the hardest features to stabilise and revegetate, and will be a focus for future restoration works as they remain a significant source of eroded sediment.

Further gullies will be blocked, raising water tables to improve conditions for moorland species such as *Sphagnum*, reducing the loss of dissolved and particulate organic carbon, slowing the flow of storm water, and increasing surface wetness to reduce the risk of wildfire. Where existing gully blocks have accumulated sediment to full capacity, the height of these blocks will be extended, if appropriate, to increase their ability to limit sediment transport and further raise water tables.

One of the issues that threaten the active blanket bogs of the SPM is the extensive network of subsurface peat pipes. Comparatively little is known about the behaviour of these pipes, although they may be significant conduits for stormflow, sediment transport and the development of gully networks (Holden *et al.*, 2002). Various experimental methods for blocking these pipes will be developed, trialled and monitored to evidence the efficacy and impact of peat pipe blocking on blanket bog condition and ecosystem service provision, towards informing the development of a best practice on how to address this threat.

Across the SAC there are areas which have become dominated by invasive species. Where the bog surface has dried out, due to gully formation and past burning episodes, trees such as willows and birch can invade. These are of benefit in some places and can increase the stability of the bog but can be problematic and increase drying of the bog surface. *Rhododendron ponticum* was planted historically to provide cover for pheasants and readily spread across the moors. This forms impenetrable thickets which shade out typical moorland species. Where these two different groups of plants are problematic, they will be cleared with chainsaws and herbicide or removed by hand.

Resilience of deep blanket peat to wildfire will be improved by identifying large areas of single age, single species vegetation cover with associated high fuel loads, and breaking them up by cutting and increasing diversity. Where conditions are appropriate, these areas will be inoculated with *Sphagnum* to create wet firebreaks.

Using results from the current investigations into successful and efficient methods of propagation and application, a range of *Sphagnum* mosses will be re-introduced across the SAC, where conditions have been improved sufficiently to allow them to re-establish, to increase the quantity and diversity of these key blanket bog species. If successful, this could set the blanket bogs of the SAC on a trajectory towards favourable condition as active blanket bog.

Monitoring actions

These conservation works will be accompanied by an extensive programme of monitoring, in order to assess the impacts on biodiversity and ecosystem services, focusing on vegetation, water quality, water storage and carbon. A land cover map will be created for the whole of the SAC using Earth Observation data (remote sensing data

Figure 22. *Sphagnum* mosses are key peat-building species and vital to the future of this priority habitat.

collected from sources such as satellites, planes and Unmanned Aerial Vehicles) and state of the art image classification software. This map will provide a high-resolution understanding of current land cover and habitat types, allowing strategic planning and prioritisation of land management and conservation works actions, and creating a baseline from which the delivery and impact of restoration works may be assessed in the future at the landscape scale.

Land manager engagement

MFFP works with landowners and managers to deliver conservation works on privately owned land through Higher Level Stewardship (HLS) schemes. This offers excellent opportunities to engage with local landowners around upland management, while assisting them to meet the requirements of the HLS schemes in terms of conservation targets. MFFP is able to offer beneficial services to landowners such as the clearance from their land of invasive species, as well as resources such as the Softrack access vehicle, which MFFP will purchase predominantly to allow cutting of vegetation and application of *Sphagnum* mosses but which will also be available to lend to land managers for inclusion in delivery of their rotational heather burning plans in order to allow them to assess the differences between burning and cutting.

As part of the MoorLIFE 2020 project, a communications pack will be developed, based on a series of stakeholder consultations which will include land management techniques and latest scientific evidence, and address concerns and issues raised by the landowning community, in order to deliver all possible outcomes for a site. Four demonstration sites will be established, with interpretation, and knowledge exchange events will be planned to engage and develop working solutions for the land managers and to highlight the value and importance the protection of active blanket bog.

Conclusion

At the time of the formation of the Moors for the Future Partnership, the South Pennine Moors SAC was in a severely degraded, and deteriorating condition. Following extensive restoration covering 30km^2 of moorland, including peat stabilisation, re-vegetation, gully-blocking and footpath improvement works, the decline of this important and fragile landscape has been halted. Results from MFFP's monitoring programme demonstrate that nurse crop grass cover is being succeeded by moorland vegetation, as a result of re-introduction of moorland species and re-wetting of the peat mass by gully-blocking. These improvements offer multiple benefits to society and the natural environment, which will be enhanced in the future following the current programme of planned restoration and conservation, which aim to consolidate and continue the works already completed.

References

Anderson, P., Tallis, J.H., Yalden, D.W. 1997. *Restoring Moorland: Peak District Moorland Management Project, Phase III Report*. Peak Park Joint Planning Board.

Bain, C.G. 2011. *IUCN UK Commission of Inquiry on Peatlands*. IUCN UK Peatland Programme, Edinburgh.

Buckler, M., Proctor, S., Walker, J.S., Wittram, B., Straton, P., Maskill, R.M. 2013. *Moors for the Future Partnerships restoration methods for restoring bare peat in the South Pennines SAC: evidence-based recommendations*. Moors for the Future Partnership, Edale.

Bonn, A. 2009. *Ecosystem services of peat, phase 1, project code SP0572*, Department for Environment, Food and Rural Affairs.

Clarke, S., Harlow, J., Scott, A., Phillips, M. 2015. Valuing the ecosystem service changes from catchment restoration: A practical example from upland England. *Ecosystem Services* 15: 93-102.

Crouch, T., Walker, J.S., Morley, K. 2015. *Peatland Restoration Project: Rivers Alport and Ashop Monitoring Report*. Moors for the Future Partnership, Edale.

Evans 2013. *Greenhouse gas emissions associated with non-gaseous losses of carbon from peatlands - fate of particulate and dissolved carbon.* DEFRA project sp1205 Final Report.

Eaton, M. 2015. Birds of Conservation Concern 4: The population status of birds in the UK, Channel Islands and Isle of Man. *British Birds* 108: 708-746.

Grayson, R., Holden, J., Rose, R. 2010. Long-term change in storm hydrographs in response to peatland vegetation change. *Journal of Hydrology* 389: 336-343.

Holden J., Burt T.P. 2002. Piping and pipeflow in a deep peat catchment. *Catena* 48: 163–199.

Holden J., Chapman P.J., Palmer S., Grayson R., Kay P. 2012. The impacts of prescribed moorland burning on water colour and dissolved organic carbon: a critical synthesis. *Journal of Environmental Management* 101: 92–103.

Jarvis, R.A 1984. Soils and their use in northern England. *Soil Survey of England and Wales*, Bulletin no. 10, Harpenden.

Lindsay, R. 2010. *Peatbogs and Carbon: A critical synthesis*. RSPB http://www.rspb.org.uk/Images/Peatbogs_and_carbon_tcm9-255200.pdf

Maskill, R., Sunter, K., Buckler, M., Wittram, B., King. L., Walker, J.S. 2015a. *MoorLIFE: A carbon audit of the project: Final report*. Moors for the Future Partnership, Edale.

Maskill, R., Walker, J.S, Allott, T., Evans, M. Shuttleworth, E. 2015b. *MoorLIFE: Changes to the water table and carbon budget*. Moors for the Future Report, Edale, Derbyshire.

Maskill, R., Walker, J.S., Benson, J., Allott, T., Evans, M., Shuttleworth, E. 2015c. *Woodhead Gully Block Monitoring: Final Report*. Moors for the Future Partnership, Edale.

Natural England. 2016. *Dark Peak SSSI condition summary* https://designatedsites.naturalengland.org.uk/ReportConditionSummary.aspx?SiteCode=S1003028&ReportTitle=DarkPeak SSSI

Pilkington, M., Walker, J., Maskill, R., Allott, T., Evans, M. 2015. *Restoration of Blanket bogs; flood risk reduction and other ecosystem benefits.* Final report of the Making Space for Water project. Moors for the Future Partnership, Edale.

Proctor, S., Buckler, M., Walker, J. S., Maskill, R. 2013. *Vegetation recovery on bare peat after restoration intervention: an analysis of nine years of monitoring data in the Dark Peak moorlands (2003 - 2012)*. Moors for the Future Partnership, Edale.

Ross, S. and Hammond, G. 2015. *Sustainable Catchment Management Programme.* Final Report for United Utilities.

Rothwell, J.J., Evans, M.G., Daniels, S.M., Allott, T.E.H. 2008. Peat soils as a source of lead contamination to upland fluvial systems. *Environmental Pollution* 153: 582-589.

Rothwell, J.J., Robinson, S.G., Evans, M.G., Yang, J., Allott, T.E.H. 2005. Heavy metal release by peat erosion in the Peak District, southern Pennines, UK. *Hydrological Processes* 19: 2973-2989.

RSPB. 2015. *Restoring bogs for water quality and wildlife: the positive effects on moorland birds.*

Shimwell, D. 1974. Sheep grazing intensity in Edale, 1692-1747, and its effect on blanket peat erosion. *Derbyshire Archaeological Journal*. 94: 35-40.

Shotbolt, L.A., Rothwell, J.J., Lawlor, A.J. 2008. A mass balance approach to quantifying Pb storage and fluxes in an upland catchment of the Peak District, north-central England. *Earth Surface Processes and Landforms* 33: 1721-1741.

Sutherland, W.J. 2009. The identification of 100 ecological questions of high policy relevance in the UK. *Journal of Applied Ecology* 43: 617-627.

Tallis, J.H. 1964. The pre-peat vegetation of the Southern Pennines. *New Phytologist*, 63: 363-373.

Tallis, J.H. 1995. Blanket mires in the Upland Landscape. En: Wheeler, B.D., Shaw, S.C., Fojt, W.J., Robertson, R.A. (eds.) 1995. *Restoration of temperate wetlands*. Chichester: John Wiley and Sons.

Walker, J., Hewson, W., McMorrow, J. 2009. *Spatial pattern of wildfire distribution on the moorlands of the South Pennines*. Moors for the Future report to Pennine Prospects.

Worrall, F., Rowson, J.G., Evans, M.G., Pawson, R., Daniels, S. Bonn, A. 2011. Carbon fluxes from eroding peatlands – the carbon benefit of revegetation following wildfire. *Earth Surface Processes and Landforms* 36: 1487 – 1498.

Yallop, A. R., Thacker, J., Thomas, G., Stephens, M., Clutterbuck, B., Brewer, T., Sannier, C. A. D. 2006. The extent and intensity of management burning in the English uplands. *Journal of Applied Ecology* 43: 1138-1148.

09

The potential of geospatial technology for monitoring peatland environments

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Abstract

There have been significant advances in the spectral and spatial resolution of data collected from spaceborne, airborne and terrestrial based geospatial technologies over the past 20 years. In sensitive peatland ecosystems, the non-intrusive application of these technologies offers great potential to improve vegetation monitoring and topographical mapping. This paper discusses the potential of geospatial technologies for monitoring vegetation, mapping natural erosion features and assessing artificial drainage with reference to two peatland sites in England. Earth Observation (EO) data can now provide colour imagery with spatial resolution comparable to conventional aerial photography. Enhanced spectral resolution of spaceborne sensors has also increased the accuracy of automated image classification for bog vegetation and EO data may challenge the relevance of conventional aerial photography in landscape-scale assessment. Ultra-high resolution data achievable from Unmanned Aerial Vehicle (UAV) and Terrestrial Laser Scanner (TLS) technologies are providing unprecedented levels of detail from remote sensing. UAV imagery now provides the possibility of identifying individual plants which greatly increases a researcher's ability to map vegetation from aerial imagery. UAV derived elevation data, combined with the capability of TLS, provide enhanced resolution of gully and artificial drain morphology compared to airborne LiDAR and allow a new approach for quantifying erosion. These technologies provide opportunities to extend traditional surveys over far larger areas than was previously possible and can assist both in targeting areas for future restoration and in monitoring of subsequent change. Traditional survey methods will however maintain an important role in assessing many aspects of peatlands, as they not only provide information to verify remotely sensed data, but are currently the only method that can 'see' underneath peatland vegetation.

Keywords: Earth Observation; terrestrial laser scaner; unmanned aerial vehicles; LiDAR; peatland restoration and monitoring.

Introduction

Traditional manual survey methods employed to map peatland vegetation (Hobbs, 1984) or quantify rates of erosion (Labadz *et al.*, 1991) are time-consuming and limited to small areas (Evans & Warburton, 2007; Boardman & Favis-Mortlock, 2016). Recent advances in geospatial technology include very high (<0.5 m) resolution imagery from space, <0.1 m resolution imagery from unmanned aerial vehicles (UAVs) and <0.01 m resolution 3-dimensional information from terrestrial laser scanning (TLS). In peatland environments, where many vegetation species are sensitive to trampling (Lindsay, 2010), these non-intrusive technologies offer great potential to improve ecological monitoring (Anderson *et al.*, 2010a; Knoth et al, 2013) and geomorphological mapping (Evans & Lindsay, 2010; Höfle *et al.*, 2013). Geospatial technologies also offer the opportunity to assess larger spatial extents in less time than traditional methods (Xie *et al.*, 2008).

Manual interpretation of aerial photography has facilitated landscape-scale mapping of vegetation change in upland areas (e.g. Taylor *et al.*, 2000), and archives of historical imagery allow reconstructions of peatland land cover and management over several decades (Pellerin *et al.*, 2008; Clutterbuck & Yallop, 2010; Thacker *et al.*, 2015). However, despite the very high (typically 0.25 m) spatial resolution of aerial imagery, data for the UK are only captured on 3-5 year cycles. This temporal resolution of data provision therefore constrains the potential for monitoring environments that require annual or more frequent assessment.

The far higher frequency of revisit and availability of Earth Observation (EO) data lend themselves to timely image capture, and have been used to map blanket peatlands (e.g. Brown *et al.*, 2007; Connolly *et al.*, 2009). Prior to the year 2000, the low spatial resolution of EO data has hindered the ability to map fine-scale detail, but the following decade marked the beginning of very high (<1 m) resolution image data from space. Early sensors onboard Ikonos and Quickbird platforms (now decommissioned) delivered 1 m and 0.65 m resolution panchromatic data respectively, but by 2008 advances in sensor technology increased the spatial resolution of panchromatic imagery to 0.5 m (Pleiades-1A & 1B) and 0.46 m (Geo-Eye-1; World-View-1 & 2). More recently, WorldView-3 & 4 platforms (operated by DigitalGlobe Inc.) collect 0.31 m resolution panchromatic data from an orbit height of 670 km.

Few aerial photographic surveys collect multi-spectral (MS) data, yet the majority of EO sensors collect both panchromatic and MS data. MS data predominantly comprise RGB (colour) and near-infrared (NIR) as a minimum, but these data are approximately four times lower in resolution than panchromatic data (e.g. 1.2 m for WorldView-3 & 4 sensors). Pan-sharpening algorithms can however enhance the resolution of MS data allowing the creation of colour images for manual interpretation. In addition to MS data, revisit times of satellite sensors (days) compared to the frequency of repeat aerial photographic survey highlights distinct advantages of EO data for monitoring, particularly as pan-sharpened resolutions are now comparable (0.31 m) to the resolution of conventional aerial photography (0.25 m).

Low-level aerial surveys employing platforms such as kites (Lorenz & Scheidt, 2014), hot-air balloons (Marzolff & Poesen, 2008) and small petrol-engine remote controlled aircraft (Hodson *et al.*, 2007) to mount cameras have enabled targeted image capture. This approach has been revolutionised by the increased accessibility of battery operated UAVs which are now being employed across a wide range of environments from mountain (Tonkin *et al.*, 2014) and glacial (Tonkin *et al.*, 2016), through agricultural (Gago *et al.*, 2015; Santesteban *et al.*, 2016) to coastal habitats (Mancini *et al.*, 2013). The accessibility of UAV technology allows greater flexibility of image acquisition over EO data and aerial photography. Owing to the height of the UAV platform above the ground, spatial resolution of derived imagery is also far higher and predominantly <0.1 m, although <0.01 m is achievable (Mancini *et al.*, 2013).

Imagery derived from UAVs is beginning to see application in peatland environments (Kalacksa *et al.*, 2013; Knoth *et al.*, 2013; Lehmann *et al.*, 2016), and may provide unprecedented levels of detail for identifying target areas for restoration and enhancing subsequent monitoring. Environmental consideration (e.g. Joys & Crick, 2004) and aviation regulation (e.g. CAA, 2016) can, however, restrict the timing of surveys and the extent of individual flights.

Laser scanning technology (LiDAR; light detection and ranging) determines the distance of objects from a sensor by measuring the time taken for a laser beam to reach a target and return (Wehr & Lohr, 1999). In airborne survey, LiDAR is often referred to as laser altimetry producing a 3-dimensional data point cloud that can be processed to produce a pixel-based digital elevation model (DEM). Where all data are retained, the resultant digital surface model (DSM) includes objects and vegetation. The data can be filtered to remove surface objects to produce a digital terrain model (DTM) and to allow assessment of, for example, tree canopy height (Dubayah & Drake, 2000). Airborne LiDAR data enable landscape-scale mapping of features, and in peatland environments DTM data allow assessment of erosion features such as natural drainage gullies (Evans & Lindsay, 2010) and vehicular tracks and footpaths (Kincey & Challis, 2010). DSM data are more suited to assessment of vegetation structure (Anderson *et al.*, 2010a; Li *et al.*, 2011), and have also been combined with EO data to improve the accuracy of automated image classification of bog vegetation (Anderson *et al.*, 2010b).

Although spatial resolution of airborne LiDAR data for urban areas may be as high as 0.25 m, over rural areas in the UK (including UK peatlands), spatial resolution is typically 1-2 m. Features smaller than the point spacing will not be extracted and this is an area where TLS (sometimes referred to as ground-based LiDAR) is finding increased application in peatland research (Grayson *et al.*, 2012; Höfle *et al.*, 2013; Luscombe *et al.*, 2014). Ground-based laser scanner technology has advanced rapidly over the past 15 years, with scanners now capable of recording over 1 million points per second and recording <0.002 m resolution data with accuracies <0.001 m at 10-25 m from the scanner (Idrees & Pradhan, 2016).

In light of these recent technological advances, this paper discusses the potential of geospatial technologies for monitoring vegetation, mapping natural erosion features and assessing artificial drainage with reference to two peatland sites in England.

Study areas and methods

The study sites are located approximately 9 km distant from each other in the Peak District National Park (PDNP; South Pennines), which lies between three key cities: Manchester, Sheffield and Leeds (Fig. 1)

Howden Moors

This study site is part of the Howden Moors blanket mire complex, and is located between Stainery Clough and Rocking Stones (Fig. 1a). The accumulation of peat is relatively shallow with a maximum depth of 70 cm (measured by the authors in 2016), although a number of artificial drains are present. Vegetation is dominated by heather (*Calluna vulgaris*) with bryophytes (*Sphagnum* spp. and *Polytrichum commune*) and rush (*Juncus* spp.) appearing in wet flushes. Bilberry (*Vaccinium myrtillus*) and bracken (*Pteridium aquilinum*) are more dominant on the steep slopes on the east side up to Rocking Stones, approximately 510 metres above sea level (Fig. 2a).

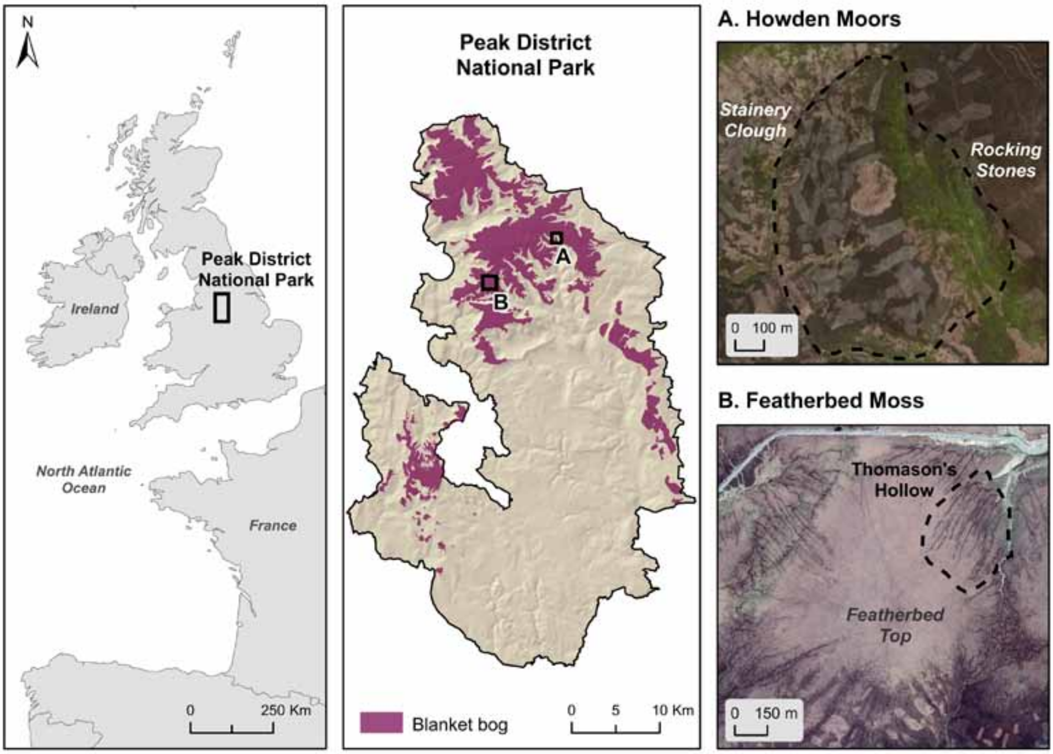

Figure 1. Location of the Peak District National Park, areas designated as blanket bog and study sites (blanket bog BAP priority habitat inventory for England version 2.1; Natural England, 2008). A) Aerial photograph of Howden Moors (© 2016 Source basemap: ESRI); B) Pan-sharpened WorldView-2 image of Featherbed Moss (© 2016 DigitalGlobe, Inc. – provided by European Space Imaging).

The area was affected by a wildfire on 9TH May 2016, but the most recent aerial photograph available for the study area was captured in 2012 (available in ArcGIS), precluding the ability to assess the extent of the fire damage or provide baseline data for monitoring vegetation recovery remotely. UAV technology offers an alternate approach to undertaking aerial survey of the burn scar, however the wildfire occurred during the ground bird nesting season (1st March – 31st July; Joys & Crick, 2004) and survey using a UAV could not be conducted until August 2016.

A satellite image covering the study site was captured less than four weeks following the fire by sensors onboard the WorldView-3 platform. These data enabled precise assessment of the extent of the fire, which burned an area of approximately 42 ha (Fig. 2b). The very high resolution (0.31 m) also enables manual interpretation of dominant vegetation composition, comparable to detail visible in 0.25 m resolution aerial photography (Figs. 2a & b). NIR data were used to assist mapping of bare peat (fire extent) and vegetation that was not consumed in the fire (Fig. 3).

The satellite data significantly improved the ability to plan and undertake field survey of the burn scar in August 2016 using both UAV and traditional methods. Ground control points (GCPs) to orthocorrect the UAV imagery were positioned on a

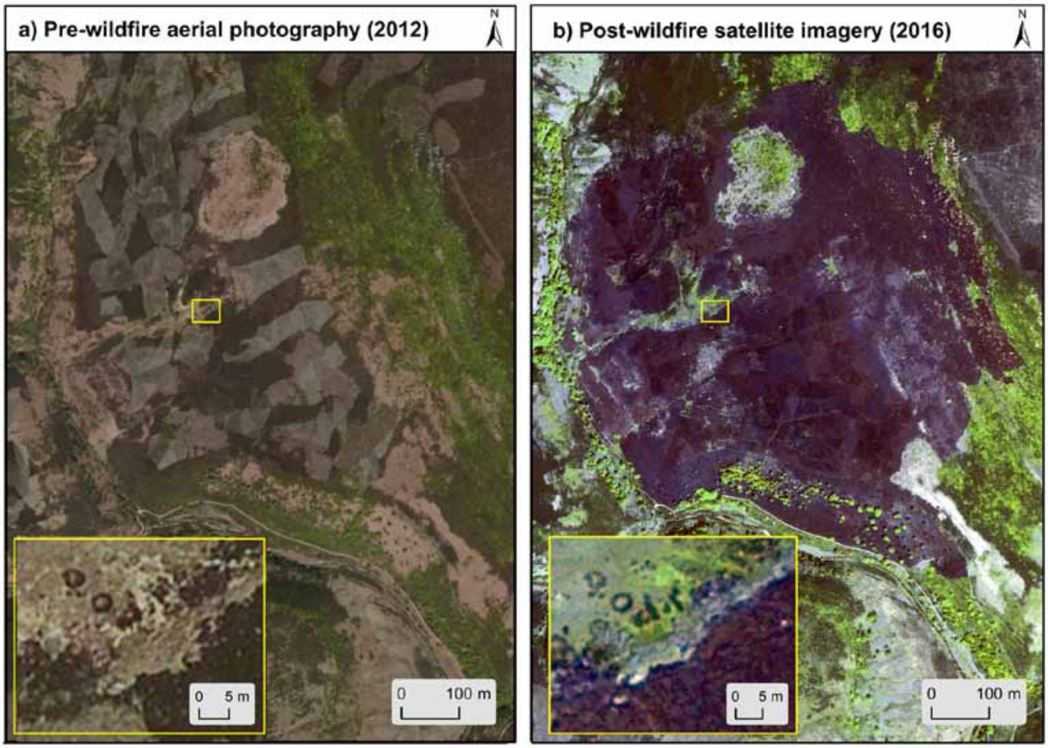

Figure 2. a) Most recent (2012) aerial photograph (0.25 m resolution) of study site (© 2016 Source basemap: ESRI); b) Pan-sharpened WorldView-3 image (0.31 m resolution) captured on 5th June 2016 (© 2016 DigitalGlobe, Inc. – provided by European Space Imaging).

100 m triangular grid across the study area (following guidance by Tonkin & Midgley, 2016), and 3D coordinates for the centre of each target (0.5 m square black and white targets) were measured using Trimble Geo 7X DGNSS units (± 0.01 m horizontal and ± 0.015 m vertical post-processed accuracy). An area located 2 m to the south of each GCP was surveyed using 0.5 m quadrats, and where present, percent vegetation cover was recorded to species level. Peat depth was measured in the centre of each quadrat using 1 m sections of threaded metal rod (0.006 m in diameter).

A fixed wing UAV (Sensefly eBee) was used to survey the burn scar from a height of 60 m above the ground. Two flights were undertaken to collect colour (RGB) imagery and near infra-red (NIR) data independently using Canon S110 RGB and S110 NIR cameras. Each flight was conducted in less than 30 minutes and covered an area of 34 ha. The data were processed using Pix4Dmapper to generate a mosaic of orthophotos and a DSM for the survey area from each flight.

Featherbed Moss (Thomason's Hollow)

Featherbed Moss comprises a dome of ombrotrophic blanket bog with a far greater accumulation of peat compared to Howden Moors (typically ranging from 2-4 m (Tal-

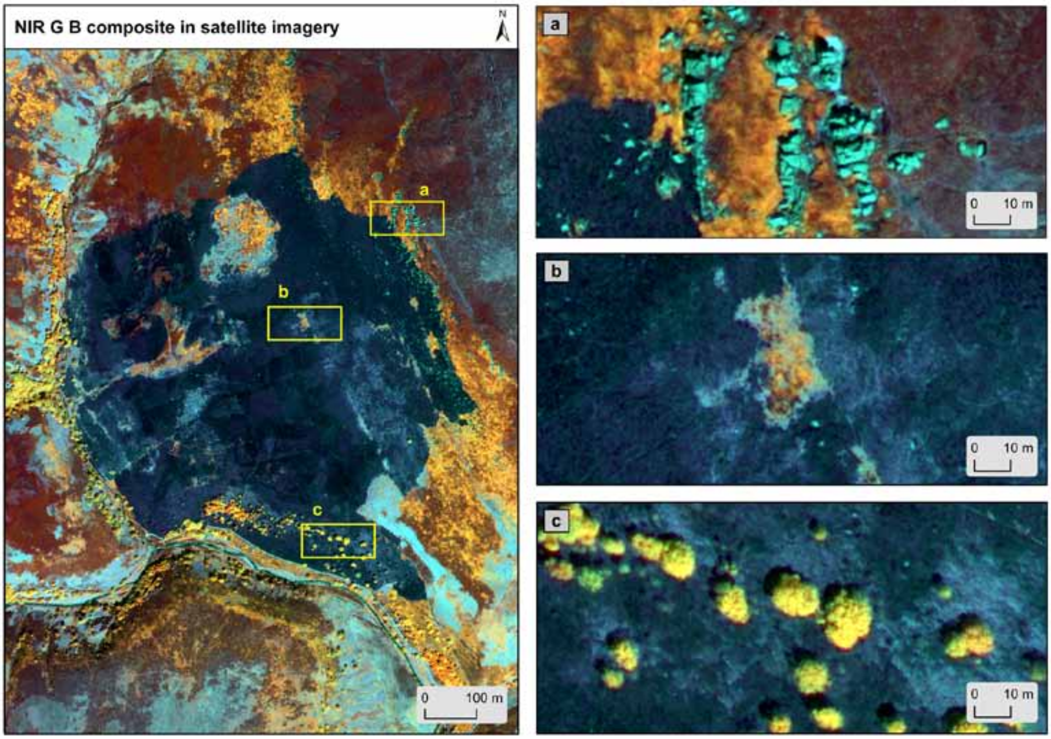

Figure 3. NIR G B composite of pan-sharpened WorldView-3 image (© 2016 DigitalGlobe, Inc. – provided by European Space Imaging), highlighting the potential to assist mapping of bare peat (fire extent) and a) dominant vegetation and rock outcrops, b) areas of vegetation not consumed in the fire and c) trees.

lis 1973; data collected by the authors in 2015). The bog is visibly degraded, with numerous natural drainage gullies cutting into the dome (Fig. 1b). These gullies are undergoing restoration by the land owner (National Trust) using gully-blocking techniques (Labadz *et al.*, 2015).

Vegetation across the Moss is dominated by cotton grass (*Eriophorum vaginatum* and E. *angustifolium*) with increasing abundance of other species including heather (*Calluna vulgaris*), bilberry (*Vaccinium myrtillus*) and cloudberry (*Rubus chamaemorus*) around the edges of the gullies (Tallis, 1973). *Sphagnum* spp. are scarce (Carroll *et al.*, 2009), but re-introduction is being trialled on the south side of the Moss (Hinde *et al.*, 2010). The altitude of Featherbed Top is approximately 544 masl.

The network of gullies on the north-east side of the Moss in Thomason's Hollow (Fig. 1b) was surveyed using a Sensefly eBee with a Canon S110 RGB in September 2016. GCPs were positioned on a 100 m triangular grid across the study area (Tonkin & Midgley, 2016), and 3D coordinates for the centre of each target were measured using Trimble Geo 7X DGNSS units. The UAV was flown at a height of 50 m above the ground covering an area of 23 ha in under 30 minutes. The data were processed using Pix4Dmapper to generate a mosaic of orthophotos and a DSM for the survey area.

Mapping and monitoring vegetation

EO data have clear utility for mapping landscape-scale features including wildfires (Millin-Chalabi *et al.*, 2014). For Howden Moors, EO data provided the most contemporary record of the burn scar within four weeks of the fire occurring. Whilst traditional survey to record the extent of the fire using handheld GPS could be undertaken within days, the spatial and spectral resolution of these EO data provide exceptional level of detail for the burn extent, far beyond the capability of manual survey, including areas that did not burn (Fig. 3). Dominant vegetation type can be mapped visually using these data, which may challenge the relevance of aerial photography in the near future. Although captured three months after the fire, the superior spatial resolution of imagery captured from the UAV survey (0.028 m) allows identification of individual plants (Figs. 4c-d, 5 & 6a).

Inclusion of elevation data (DSM) derived from the imagery enables construction of a near first-person view of the ground, providing additional information on plant height and morphology (Fig. 5). It is possible to visually identify rush, heather, several moss species, bilberry, bracken, bare peat, rocks and even sheep with confidence in these data (Figs. 3-5a). Traditional vegetation survey does still have the advantage of being able to see underneath vegetation canopies, but geospatial technology now provides an unprecedented ability to map the top layers of vegetation and can be used to extend conventional vegetation mapping data to far greater areas post field survey.

EO data are also suited to automated image classification (pixel- or object-based) that enables mapping of large areas far quicker than manual interpretation. This is primarily due to the availability of MS data, and sensors on the WorldView platforms since WorldView-2 (launched 2009; 1.8 m resolution) collect eight bands of data (coastal, blue, green, yellow, red, red-edge and two NIR). The WorldView-3 sensor (launched in 2014) has a spatial resolution of 1.2 m. Previous land cover mapping from classification of satellite data for the UK (Land Cover Map 2007; LCM2007) report an overall mapping accuracy of 83 % (Morton *et al.*, 2011). However, relatively low accuracies (43 %) were reported over areas of bog. Satellite data used to derive LCM2007 had far lower spatial resolution of 20-30 m and this may explain some of the spectral confusion arising between bog and other land cover classes. Object-based classification of fen habitat from pan-sharpened imagery from GeoEye-1 (launched 2008; 1.8 m MS resolution) reported accuracy of 82 % (Dribault *et al.*, 2012), and the wildfire on Howden Moor and new (post-2014) satellite capability provides the opportunity to assess the applicability of higher spatial and spectral resolution EO data to monitor the growth of vegetation across a currently bare surface.

Automated classification of EO imagery here could be supplemented with data from contemporary UAV surveys. In addition to improving 'training' data for satellite image classification, classification of NIR data captured by UAVs are already report-

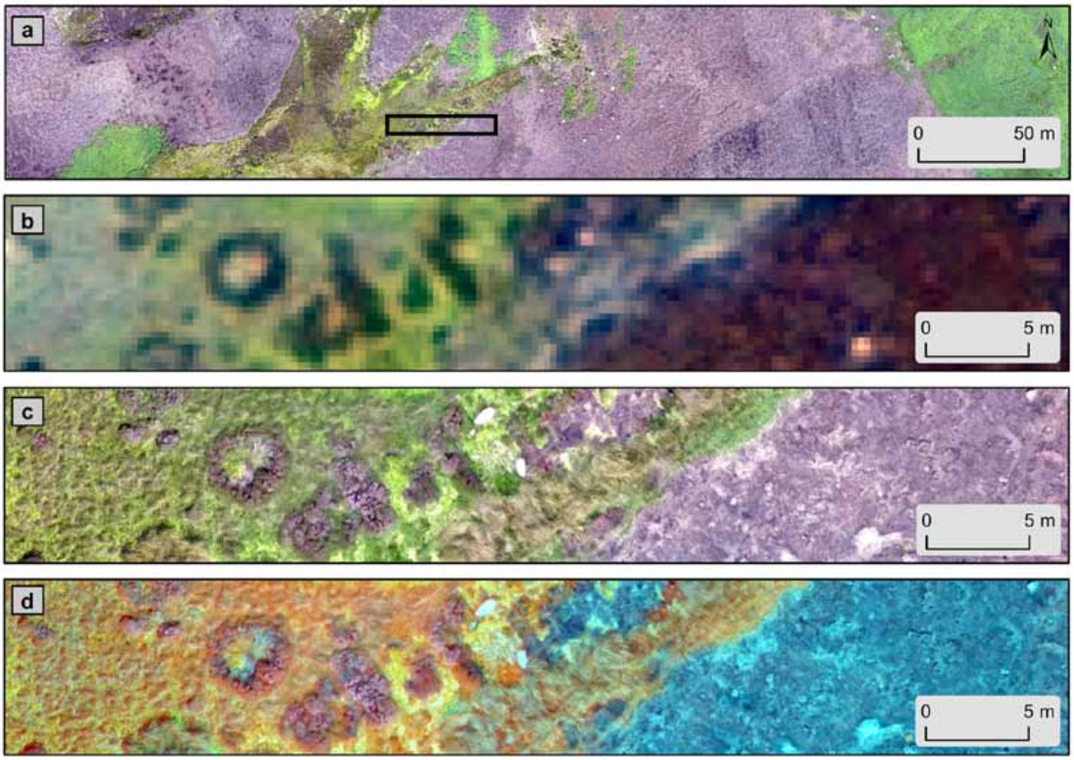

Figure 4. Capability of 0.028 m resolution UAV derived imagery for mapping vegetation; a) Section of UAV survey; b) WorldView-3 data (0.31 m resolution; © 2016 DigitalGlobe, Inc. – provided by European Space Imaging) c) RGB and d) NIR, G, B image composite captured using Sensefly eBee.

ing exceptionally high accuracy (91 %) for bog vegetation (Knoth *et al.*, 2013). The error of orthocorrection of the RGB and NIR UAV image data collected in this survey (± 0.015-0.018 m RMSE) allowed highly accurate alignment and production of a 4-band MS image stack (R, G, B, NIR; Fig. 4d).

The NIR data clearly allow visual separation of bare peat and burned heather from unburned heather compared to RGB imagery (Fig. 4d), and also provide 0.028 m resolution baseline data to enable very high resolution monitoring of regrowth in the future. The addition of thermal IR sensors to UAVs may further assist in separating vegetation spectral signatures for classification (Berni *et al.*, 2009) and also allow measurement of additional characteristics of peatlands such as near-surface hydrology (Luscombe *et al.*, 2015).

Despite the higher spatial resolution of UAV data compared to recent EO data, automated image classification presents a number of new challenges. At such high resolution, individual plants have distinct variation in spectral response, for example, 0.028 m (or smaller) dimension pixels containing heather solely comprise either stick, leaf or flower (Figs. 4 & 5). Additionally, a satellite image is captured in a far shorter time than a UAV survey which may comprise up to hundreds of individual

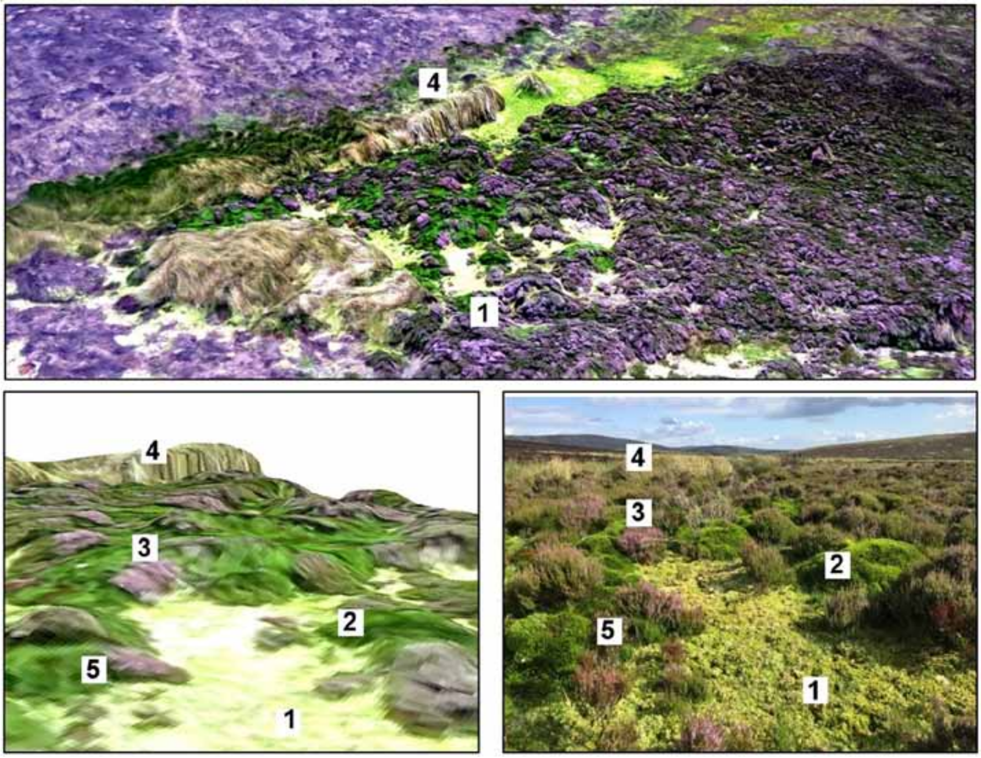

Figure 5. Morphology of plant species visible from imagery draped over DSM derived from Sensefly eBee (0.028 m resolution) compared with first person photograph (bottom right):1) *Sphagnum* spp. carpet; 2) *Polytrichum commune* hummock; 3) flowering *Calluna vulgaris*; 4) *Juncus* spp.; 5) mixed sward of *Calluna vulgaris* and *Polytrichum commune* hummocks.

photographs. During a 30 minute (or longer) survey, particularly in peatland environments, atmospheric conditions or cloud cover may change and affect the spectral response of vegetation between individual photographs across a study area.

Mapping natural erosion features and artificial drainage

In addition to providing the ability to visually identify individual plants in UAV imagery, both natural erosion features (Fig. 6b) and artificial drainage (Fig. 6c) appear exceptionally well-defined. Some of the key issues associated with drainage gullies and artificial drains in degraded peatland areas relate to erosion and drawdown of water-tables (Holden *et al.*, 2004; Allott *et al.*, 2009;). This can promote decomposition of peat in the upper layers and impact on drainage water quality (Armstrong *et al.*, 2010) and also on adjacent vegetation composition, including peat forming species (e.g. *Sphagnum* spp.; Lindsay *et al.*, 2014).

These geomorphological features are therefore of key interest in peatland restoration, so the availability of very high-resolution mapping of morphology may assist in targeting restoration efforts. Airborne LiDAR data are effective for mapping erosion gully form and depth (Evans & Lindsay, 2010), but the authors note that 2 m resolution data reduce the ability to map gully width and depth where gullies are less than 8 m in width. At Thomason's Hollow on Featherbed Moss, some erosion gul-

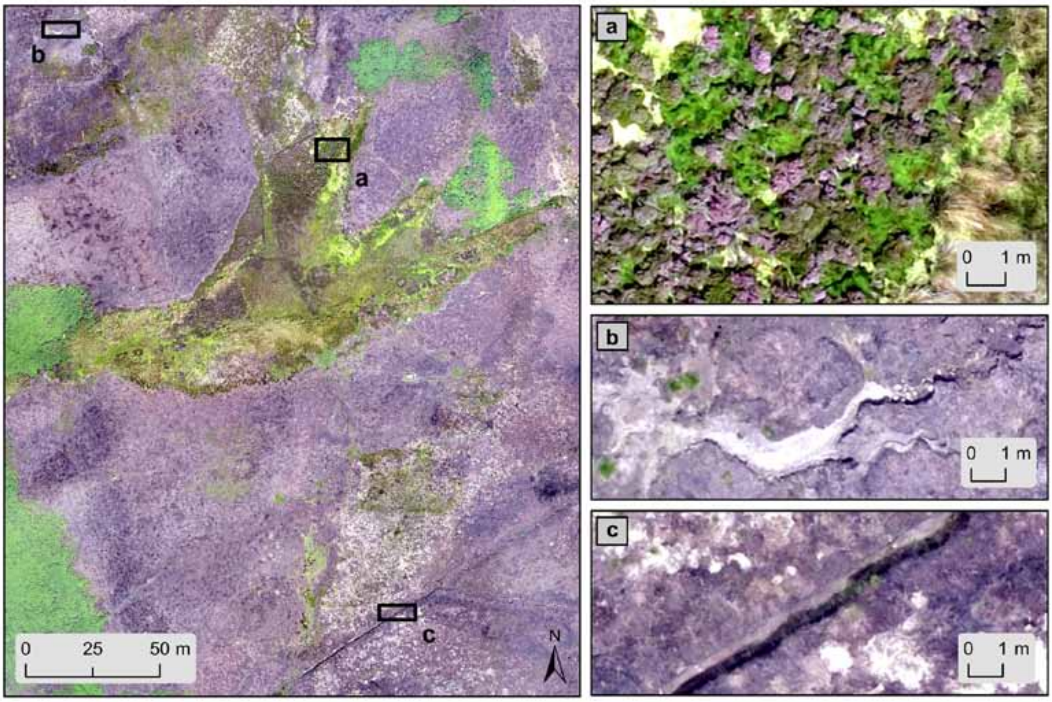

Figure 6. Features visible in 0.028 m resolution RGB imagery captured using Sensefly eBee; a) vegetation; b) natural drainage gully; c) artificial drain.

lies are less than 2 m distant from the next, and the data from Howden Moor (Fig. 6b) highlight the potential for UAV derived DSM data in mapping finer-scale detail.

The UAV data collected for Featherbed Moss have a spatial resolution of 0.02 m, and although the derived DSM data contain vegetation, the morphology of individual drainage gullies is exceptionally clear (Fig. 7a-b). It is also possible to identify features that appear to be new gullies in the early stages of development, possibly from collapse of peat pipes as outlined by Holden (2005). In some of these gullies, significant changes in morphology are evident on the ground as a result of peat slumping taking peat blocks (sometimes up to 0.5-0.6 m wide) and vegetated 'slabs' (Fig. 7b) into the gully. These features can be identified in such high resolution data and it is likely that UAV derived DSM data will become a key tool for monitoring peatland erosion in gully networks.

However, Evans & Warburton (2007) highlight that in some peatland sites in the UK, rates of erosion recorded using traditional methods are less than 0.01 m yr^{-1}. Mapping this scale of erosion from UAV data is less certain, and is an area where far higher (<0.01 m) resolution data captured by TLS may see increased application. The superior spatial extent and resolution of TLS data compared with erosion pins on peatland has been noted (Grayson *et al.*, 2012). However, aligning multi-tempo-

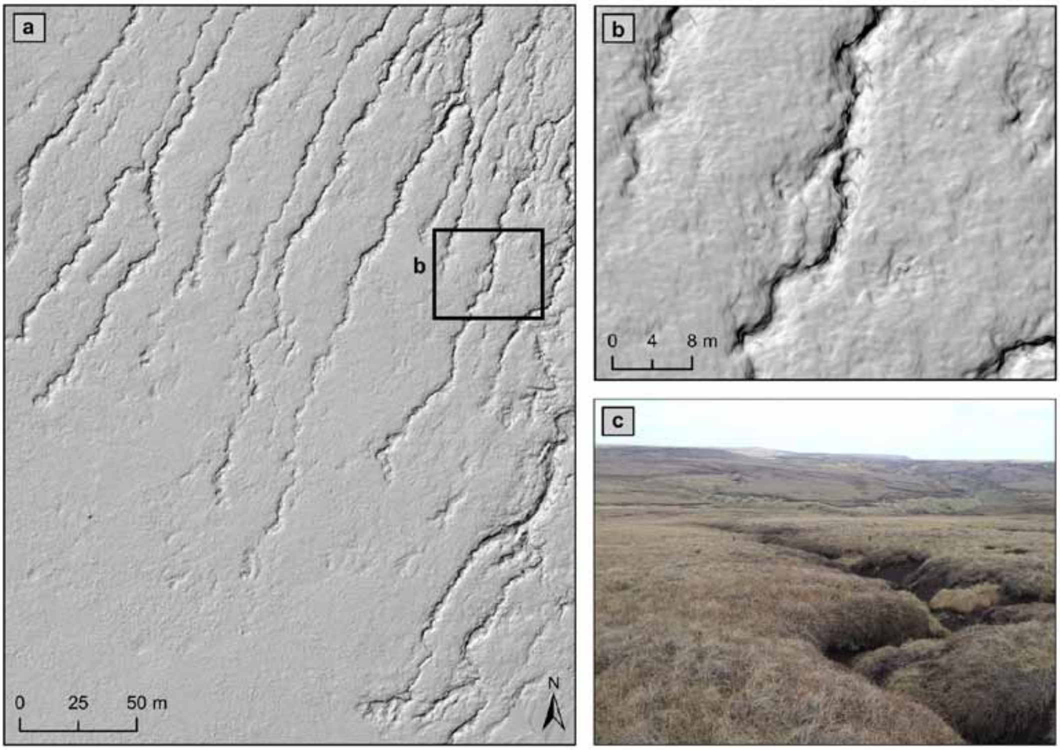

Figure 7. Natural drainage gullies in Thomason's Hollow; a) section of survey data showing DSM derived from 0.02 m resolution RGB imagery; b) detail visible in 0.02 m resolution DSM; c) scale of morphological change in some areas, here vegetation slumping into gully.

ral scan data with sufficient accuracy to quantify such small change between time periods remains a key challenge. In vertosol soils, accuracy of alignment <0.01 m has been achieved (Goodwin *et al.*, 2016), but in peatlands, the nature of the 'soil' and other factors including 'mire-breathing' can have a significant influence on the ability to align multi-temporal data (Grayson *et al.*, 2012).

Additionally, other features impacting on gully morphology, such as overhanging vegetation, highlights the fact that multiple scans collected in each survey may still not provide full coverage of gullies (Höfle et al, 2013). Despite covering far smaller areas, the role of traditional methods will still exist, as these data allow validation of rates of erosion estimated from geospatial technologies.

With regard to restoration efforts, focussing on areas in which to intervene is a key stage of planning. The location of gullies and drains can be mapped using EO data or aerial photography, but mapping of gully dimensions will help target the largest features. The data from Howden Moor demonstrates the potential for extracting very high resolution measurements of artificial drains from UAV derived imagery (Fig. 8) and in conjunction with very high resolution vegetation mapping, this further highlights the potential for geospatial technologies to become part of the standard toolbox for peatland mapping, restoration and monitoring.

Conclusion

There have been significant advances in the resolution of geospatial data. EO data provide the most reliable method for capturing sudden events in peatland environ-

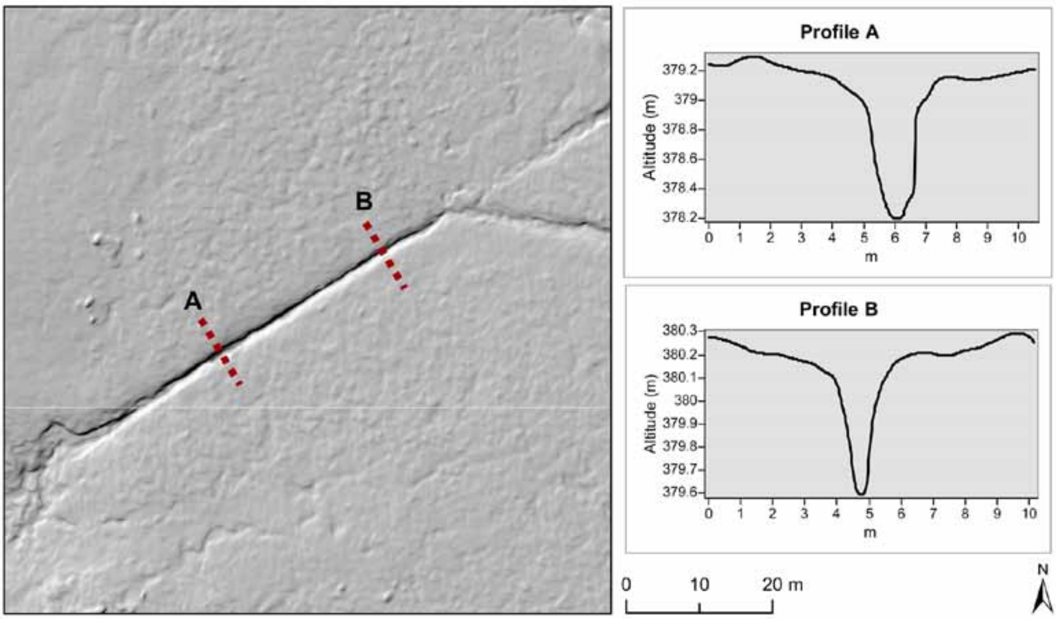

Figure 8. Artificial drain as visible in 0.028 m resolution DSM derived from UAV imagery for Howden Moor and cross-sectional profiles at two locations (note exaggeration in vertical scale).

ments and the increased spatial resolution of recent (post-2014) sensors allows mapping comparable to that enabled by conventional aerial photography. This advance in technology may be fundamental to future mapping, and increased spectral resolution may also improve the accuracy of automated image classification of peatland vegetation.

Ultra-high resolution data achievable from UAV and TLS technology are providing unprecedented levels of detail from remote sensing. UAV imagery now provides the possibility of identifying individual plants which greatly increases a researcher's ability to extend traditional vegetation surveys over far larger areas. UAV-derived elevation data combined with the capability of TLS offer enhanced resolution of gully and drain morphology compared to airborne LiDAR and provide a new approach for quantifying erosion.

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References

Allott, T.E.H., Evans, M.G., Lindsay, J.L., Agnew, C.T., Freer, J.E., Jones, A., Parnell, M. 2009. *Water tables in Peak District blanket peatlands*. Report to Moors for the Future, United Kingdom.

Anderson, K., Bennie, J., Wetherelt, A. 2010a. Laser scanning of fine scale pattern along a hydrological gradient in a peatland ecosystem. *Landscape Ecology* 25(3): 477-492.

Anderson, K., Bennie, J.J., Milton, E.J., Hughes, P.D.M., Lindsay, R., Meade, R. 2010b. Combining LiDAR and IKONOS data for eco-hydrological classification of an ombrotrophic peatland. *Journal of Environmental Quality* 39: 1-14.

Armstrong, A., Holden, J., Kay, P., Francis, B., Foulger, M., Gledhill, S., McDonald, A.T., Walker, A. 2010. The impact of peatland drain-blocking on dissolved organic carbon loss and discolouration of water; results from a national survey. *Journal of Hydrology* 381: 112–120.

Berni, J.A.J., Zarco-Tejeda, P.J., Suarez, L., Fereres, E. 2009. Thermal and narrowband multispectral remote sensing for vegetation monitoring from an unmanned aerial vehicle. *IEEE Transactions on geoscience and remote sensing* 47(3): 722-738.

Boardman, J., Favis-Mortlock, D. 2016. The use of erosion pins in geomorphology. *British Society for Geomorphology* 3(5.3): 1-10.

Brown E., Aitkenhead, M.A., Wright, R., Aalders, I. 2007. Mapping and classification of peatland on the Isle of Lewis using Landsat ETM+. *Scottish Geographical Journal* 138(3): 173-192.

Civil Aviation Authority (CAA), 2016. The Air Navigation Order 2016 and Regulations. CAP393. United Kingdom.

Carroll, J., Anderson, P., Caporn, S., Eades, P., O'Reilly, C., Bonn, A. 2009. *Sphagnum in the Peak District Current Status and Potential for Restoration*. Moors for the Future Report No 16, United Kingdom.

Clutterbuck, B., Yallop, A.R. 2010. Land management as a factor controlling dissolved organic carbon release from upland peat soils 2: Changes in DOC productivity over four decades. *Science of the Total Environment* 408: 6179–6191.

Connolly, J., Holden, N.M., Seaquist, J.W., Ward, S.M. 2009. Detecting recent disturbance on Montane blanket bogs in the Wicklow Mountains, Ireland using the MODIS enhanced vegetation index. *International Journal of Re-*

mote Sensing 32(9): 2377-2393.

Dribault, Y., Chokmani, K., Bernier, M. 2012. Monitoring seasonal hydrological dynamics of minerotrophic peatlands using multi-date GeoEye-1 very high resolution imagery and object-based classification. *Remote Sensing* 4: 1887-1912.

Dubayah, R.O., Drake, J.B. 2000. LiDAR remote sensing for Forestry. *Journal of Forestry* 98(6): 44-46.

Evans, M., Lindsay, J. 2010. High resolution quantification of gully erosion in upland peatlands at the landscape scale. *Earth Surface Processes and Landforms* 35(8), 876-886.

Evans, M., Warburton, J. 2007. *The Geomorphology of Upland Peat: Pattern, Process, Form*. Blackwell, Oxford, United Kingdom.

Gago, J., Douthe, C., Coopman, R.E., Gallego, P.P., Ribas-Carbo, M. Flexas, J., Escalona, J., Medrano, H. 2015. UAVs challenge to assess water stress for sustainable agriculture. *Agricultural Water Management* 153: 9-19.

Goodwin, N.R., Armston, J., Stiller, I., Muir, J. 2016. Assessing the repeatability of terrestrial laser scanning for monitoring gully topography: A case study from Aratula, Queensland, Australia. *Geomorphology* 262: 24-36.

Grayson, R., Holden, J., Jones, R.R., Carle, J.A., Lloyd, A.R. 2012. Improving particulate carbon loss estimates in eroding peatlands through the use of terrestrial laser scanner. *Geomorphology* 179: 240-248.

Hinde, S., Rosenburgh, A., Wright, N. Buckler, M., Capron, S. 2010. *Sphagnum re-introduction project: A report on research into the re-introduction of Sphagnum mosses to degraded moorland*. Moor for the Future Research Report 18, United Kingdom.

Hobbs, R.J. 1984. Length of burning rotation and community composition in high-level *Calluna-Eriophorum* bog in N England. *Vegetatio* 57(2/3): 129-136.

Hodson, A., Anesio, A.M., Ng, F., Watson, R., Quirk, J., Irvine-Fynn, T., Dye, A., Clark, C., McCloy, P., Kohler, J., Sattler, B. 2007. A glacier respires: Quantifying the distribution and respiration CO_2 flux of cryoconite across an entire Arctic supraglacial ecosystem. *Journal of Geophysical Research* 112(G4): 2156-2202.

Holden, J. 2005. Controls of soil pipe frequency in upland blanket peat. *Journal of Geophysical Research* 110(F01002): 1-11.

Höfle, B., Griesbaum, L., Forbriger, M. 2013. GIS-based detection of gullies in terrestrial LiDAR data of the Cerro Llamoca peatland (Peru). *Remote Sensing* 5: 5851-5870.

Idrees, O. M., Pradhan B. 2016. A decade of modern cave surveying with terrestrial laser scanning: A review of sensors, method and application development. *International Journal of Speleology* 45(1): 71-88.

Kalacksa, M., Arroyo-Mora, J.P., de Gea, J., Snirer, E., Herzog, C., Moore, T.R. 2013. Videographic analysis of *Eriophorum vaginatum* spatial coverage in an ombotrophic bog. *Remote Sensing* 5: 6501-6512.

Kincey, M., Challis, K. 2010. Monitoring fragile upland landscapes: The application of airborne LiDAR. *Journal for Nature Conservation* 18: 126-134.

Knoth, C., Klein, B., Prinz, T., Kleinebecker, T. 2013. Unmanned aerial vehicles as innovative remote sensing platforms for high-resolution infrared imagery to support restoration monitoring in cut-over bogs. *Applied Vegetation Science* 16: 509-517.

Labadz, J.C., Burt, T.P., Potter, A.W.R. 1991. Sediment yield and delivery in the blanket peat moorlands of the Southern Pennines. *Earth Surface Processes and Landforms* 16 (3): 255-271.

Labadz, J.C., Clutterbuck, B., Butcher, B., Hart, R., Topliss, D., Midgley, N. 2015. *Featherbed Moss Monitoring Programme.* Progress Report to National Trust, United Kingdom.

Lehmann, J.R.K., Münchberger, W., Knoth, C., Blodau, C., Nieberding, F., Prinz, T., Pancotto, V.A., Kleinebecker, T. 2016. High-Resolution classification of South Patagonian peat bog microforms reveals potential gaps in Up-Scaled CH4 fluxes by use of Unmanned Aerial System (UAS) and CIR imagery. *Remote Sensing* 8: 173.

Lindsay, R. 2010. *Peatbogs and carbon: a critical synthesis to inform policy development in oceanic peat bog conservation and restoration in the context of climate change*. Technical Report. University of East London, Environmental Research Group, London, United Kingdom.

Lindsay, R., Birnie, R., Clough, J. 2014. Peat bog ecosystems: structure, form, state and condition. *IUCN UK Committee Peatland Programme* 2: 1-7.

Lorenz, R.D., Scheidt, S.P. 2014. Compact and inexpensive kite apparatus for geomorphological field aerial photography, with some remarks on operations. *GeoRes*J 3-4: 1-8.

Li, S., MacMillan, R.A., Lobb, D.A., McConkey, B.G., Moulin, A.P., Fraser, W. 2011. LiDAR DEM error analyses and topographic depression identification in a hummocky landscape in the prairie region of Canada. *Geomorphology* 129(3-4): 263-275.

Luscombe, D.J., Anderson, K., Gatis, N., Wetherelt, A., Grand-Clement, E., Braizer, R.E.

2014. What does airborne LiDAR really measure in upland ecosystems? *Ecohydrology* 8(4): 584-594.

Luscombe D. J., Anderson K., Gatis N., Grand-Clement E., Brazier R. E. 2015. Using airborne thermal imaging data to measure near-surface hydrology in upland ecosystems. *Hydrological Processes* 29(6): 1656-1668.

Mancini, F., Dubbini, M., Gattelli, M., Stecchi, F., Fabbri, S., Gabbianelli, G. 2013. Using unmanned aerial vehicles (uav) for high-resolution reconstruction of topography: the structure from motion approach on coastal environments. *Remote Sensing* 5: 6880-6898.

Marzolff, I., Poesen, J. 2009. The potential of 3D gully monitoring with GIS using high-resolution aerial photography and a digital photogrammetry system. *Geomorphology* 111: 48-60.

Millin-Chalabi, G., Mcmorrow, J., Agnew, C. 2014. Detecting a moorland wildfire scar in the Peak District, UK, using Synthetic Aperture Radar (SAR) from ERS-2 and Envisat ASAR. *International Journal of Remote Sensing* 35(1): 54-69

Morton, D., Rowland, C., Wood, C., Meek, L., Marston, C., Smith, G., Wadsworth, R., Simpson, I.C. 2011. *Final Report for LCM2007 - the new UK Land Cover Map*. Countryside Survey Technical Report No. 11/07 NERC/Centre for Ecology & Hydrology (CEH Project Number: C03259).

Joys, A.C., Qrick, H.Q.P. 2004. *Breeding periods for selected bird species in England*. BTO Research Report 352, United Kingdom.

Pellerin, S., Mercure, M., Desaulniers, A.S., Lavoie, C. 2008. Changes in plant communities over three decades on two disturbed bogs in southeastern Quebec. *Applied Vegetation Science* 12: (107-118).

Santesteban, L.G., Di Gennaro, S.F., Herrero-Langreo, A., Miranda, C., Royo, J.B., Matese, A. 2016. High-resolution UAV-based thermal imaging to estimate the instantaneous and seasonal variability of plant water status within a vineyard. *Agricultural Water Management* 183: 49-59.

Tallis, J.H. 1973. Studies on Southern Pennine Peats: V. Direct Observations on Peat Erosion and Peat Hydrology at Featherbed Moss, Derbyshire. *Journal of Ecology* 61(1): 1-22.

Taylor, J.C., Brewer, T.R., Bird, A.C. 2000. Monitoring landscape change in the National Parks of England and Wales using aerial photograph interpretation and GIS. *International Journal of Remote Sensing* 21: 2737-2752.

Thacker, J.I., Yallop, A.R., Clutterbuck, B. 2015. IPENS 055. *Burning in the English Uplands. A Review, Reconciliation and Comparison of Results of Natural England's Burn Monitoring: 2005-2014*. Natural England Report, United Kingdom.

Tonkin, T.N., Midgley, N.G. 2016. Ground-control networks for image based surface reconstruction: An investigation of optimum survey designs using UAV derived imagery and Structure-from-Motion photogrammetry. *Remote Sensing* 8(9): 786.

Tonkin, T.N., Midgley, N.G., Cook, S.J., Graham, D.J. 2016. Ice-cored moraine degradation mapped and quantified using an unmanned aerial vehicle: A case study from a polythermal glacier in Svalbard. *Geomorphology* 258: 1-10.

Tonkin, T.N., Midgley, N.G., Graham, D.J., Labadz, J.C. 2014. The potential of small unmanned aircraft systems and structure-from-motion for topographic surveys: A test of emerging integrated approaches at Cwm Idwal, North Wales. *Geomorphology* 226: 35-43.

Wehr, A., Lohr, U. 1999. Airborne laser scanning – and introduction and overview. *ISPRS Journal of Photogrammetry & Remote Sensing* 54: 68-82.

Xie, Y., Sha, Z., Yu, M. 2008. Remote sensing imagery in vegetation mapping: a review. *Journal of Plant Ecology* 1(1): 9-23.

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The Zalama blanket bog

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Abstract

This article gives an overview of blanket bogs within the Atlantic zone of Northern Spain. This habitat is very dependent on specific climatic conditions, which in Europe are more frequent in northern latitudes. For these reasons, the distribution of blanket bogs in the Iberian Peninsula is limited to the North of Galicia and the Cantabrian fringe. Within this zone, it is worth highlighting the Zalama blanket bog (Sierra de Ordunte, Burgos - Bizkaia), which marks the eastern limit of the blanket bogs in the Iberian Peninsula. The Zalama blanket bog is described, including its peat deposit, historical, vegetation and conservational aspects. The challenges facing its conservation, as well as the main threats that affect all Spanish blanket bogs, are also described.

Keywords: Blanket bog; Zalama; distribution; *Eriophorum vaginatum*; Spain.

Introduction

Blanket bogs are one of the two existing types of ombrotrophic mires, i.e. fed mainly by atmospheric precipitation waters. Whilst the other type of ombrotrophic[1] mires, the raised bogs, have been originated from waterlogged areas, developed in hollows and depressions in the relief, blanket bogs are characterized by not being confined to terrain depressions. For this reason, these bogs spread out on the ground, always on gentle slopes, being able to cover large surfaces under optimum climatic conditions.

These bogs are strongly dependent on a markedly Atlantic climate, with abundant atmospheric precipitation (rain and particularly fog). The climatic conditions needed for their development (Lindsay *et al.,* 1988) are the following: an annual

[1] Basically there are two types of mires: ombrotrophic mires (bogs), exclusively or essentially fed by atmospheric precipitation; and minerotrophic mires (fens), where there is a prevalence of edaphic waters, generally subsoil water. Many of the remarkable singularities of the Zalama blanket bog are due to be the only remaining ombrotrophic mire in the Basque Mountains, where most of the other mire phenomena belong to the minerotrophic type (Heras *et al.,* 2017).

rainfall over 1,000 mm, more than 160 rainy days per year and a mean temperature of the warmest month under 15ºC. Their development also requires siliceous lithological substrates and a flat or gently undulating topography. All these special genetic needs make blanket bogs one of the rarest type of mires, since less than 3 % of the world's peatlands belong in this type (Foss *et al.,* 2001). They are also one of the most threatened mire types around the world, and are considered in the European Union as a priority natural habitat of community interest, under the code 7130*.

In Europe, due to the conditions that are needed, blanket bogs are distributed in the most strongly Atlantic regions in the Western European countries (Fig. 1). For instance, the British Isles contain 10-15 % of the world blanket bogs (Lindsay 1995, Foss *et al.,* 2001). They are especially found in the North and the Highlands of Scotland, mountainous areas in Northern England (Pennines), as well as on the moors of Cornwall, Wales and Northern and Western Ireland. In Scandinavia they extend along the western coasts of Norway, and also on the western coast of Iceland. Although doubtful and still subject to discussion, their presence has also been indicated in the Armorican Massif in France. In this case, they would be very rare, located in two small areas in the Monts d'Arrée in Brittany, probably at an early stage of development ("baby blanket bogs", according to R. Lindsay) (Gaudillat *et al.,* 2002).

The southernmost zone for blanket bogs in Europe is the Cantabrian fringe in northern Spain (Heras *et al.,* 2017). Unnoticed for a long time, they were recognized as such from the late nineties in the XXth century (Ramil *et al.,* 1996; Heras & Infante, 2003-2004). Spanish blanket bogs are unique and, as a whole, of an enormous interest.

Figure 1. Distribution of blanket bogs in Europe.

Presence of blanket bogs in Spain

Assuming the climatic conditions indicated by Lindsay *et al.* (1988) given in the preceding paragraphs, in Spain blanket bogs can only develop in a few areas of the Cantabrian fringe. Consulting the different maps in the Iberian Climatic Atlas (AEMET, 2011), the combination of areas with an annual rainfall over 1,000 mm, more than 160 rainy days per year and with a maximum mean temperature in the warmest month (August) under 15ºC, only yields a few areas suitable for the development of blanket bogs: mountainous areas in the north of Galicia, some in the west of Asturias, others in the south of Cantabria and a few isolated spots in the Basque Country and the North of Navarre (Fig. 2). A large part of Galicia, Asturias, Cantabria, the Basque Country and the Western Pyrenees, despite their high annual rainfall and numerous rainy days per year, are excluded on the account of a high summer temperature that seems to be the important limiting factor. It must also be taken into account that, apart from climate, much of this area has other important limiting factors, particularly the presence of carbonated or calcareous rocks, and a rugged mountainous terrain which does not favour the topography where blanket bogs could be developed.

The potential areas in Spain within these theoretical limits for blanket bogs fit very well with the experience in the field. Currently there are two core areas where the presence of blanket bogs has been proved: in Galicia (in the North of Lugo province), and in the mountains in the Eastern sector of the Cantabrian Range, between the South of Cantabria, the North of Burgos province and the west of the Basque Country (Fig. 2).

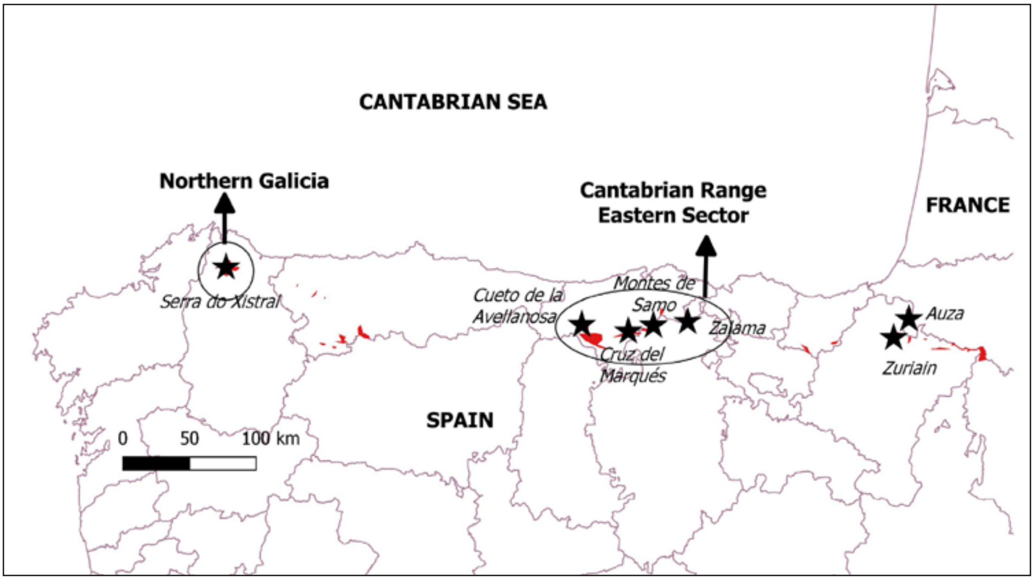

Figure 2. Location of the two known core areas of blanket bogs in Spain. Areas where climatic conditions for blankets bogs are met are shown in red. Names mentioned in the text are also indicated.

The existence of blanket bogs in Spain was revealed in quite recent times. The first record was the study of the ombrotrophic bogs in Galicia by Ramil *et al.* (1996), who mentioned the blanket bogs in the Serra do Xistral (Lugo), an area that was later studied and characterized in detail (Izco & Ramil, 2001, Martínez, 2001). A few years later, Heras & Infante (2003-2004) pointed out another blanket bog, lying 350 km to the east of the Serra do Xistral, on Zalama Mountain (Burgos and Bizkaia provinces). The gap existing between these two core areas shows how the knowledge of Spanish blanket bogs is incomplete, still in need to study their real distribution area and their characterization. It is necessary for example, to study the nature of the bogs in the *Sierras Planas* in the East of Asturias, close to Llanes (e.g. La Borbolla and Roñanzas, among others), originated at the summit of the low quartzite mountains near the coast, on ancient coastal flats. They could well be another core area for blanket bogs. Finally, some peculiar but thin peaty deposits (hardly reaching half a metre in depth) must be mentioned, located on a few summits (Auza and Zuriain) in the North of Navarre. These summits are close to the Cantabrian Sea and seem to comply with the climatic limits for the genesis of blanket bogs, but the topographical conditions and their rugged relief do not favour a good rate of peat deposition. Although a detailed study of these peaty zones is necessary, it is undeniable that they represent an interesting phenomenon closely related to blanket bogs, but at such an early stage that they could not probably be considered as such.

The Galician core area hosts the largest surface of blanket bogs in Spain. The most noteworthy area is the Serra do Xistral, which lies less than 20 km in a straight line from the sea, and contains several types of bogs and fens, among them the most representative cases of Galician blanket bogs (Fig. 3). Izco & Ramil (2001) drew a scheme of the distribution of the different mire habitats that can be found in the Serra do Xistral, including the location of these blanket bogs. In addition, the Galician blanket bogs have botanical and phytogeographical peculiarities, since they show a very grassy appearance, dominated by *Carex durieui* Steud. ex Kunze (endemic plant to the Iberian Northwest), acting as the main peat former, and *Eriophorum angustifolium* Honckeny, with the presence in much lower quantity of another endemic species, *Erica mackaiana* Bab., and *Sphagnum pylaesii* Brid. Situated in remote areas, these bogs have remained quite safe until recently when their main threat appeared: windfarms, whose massive installation have deeply artificialized this unique Galician countryside.

Figure 3. Blanket bogs in the Pena da Cadela, in the Serra do Xistral (Lugo), where windfarms have been installed.

The blanket bog core of the Cantabrian Range Eastern sector starts at the *Cueto de la Avellanosa* bog, near Tudanca in Cantabria. This bog, lying at an altitude of 1,350 m a.s.l. and at this moment practically disappeared through peat exploitation, was the first to be recognized in the literature as unique, since it was mentioned by Mariscal (1983) as "*turbera de collado*" (saddle bog), noting its development in a remarkably high topographical position. However, the main concentration of blanket bogs lies further to the East, between the Sierra del Escudo (Cantabria), the Montes de Samo (between Burgos and Cantabria provinces) and Monte Zalama, in the extreme west of the Sierra de Ordunte, between Burgos and Bizkaia provinces. In the Sierra del Escudo, the *Cruz del Marqués* bog is noteworthy, located at an altitude of 1,250 m; it is the biggest blanket bog in the area, with an original surface of around 8 hectares. Unfortunately, as in the case of *Cueto de la Avellanosa*, it is currently under exploitation and practically disappeared. On the summits of the Montes de Samo, there are a series of spots with blanket bogs, the most characteristic being *Motas de Pardo* (between Burgos and Cantabria provinces), and even reaching Las Estacas de Trueba mountain pass (Fig. 4). Finally, in the extreme west of the Sierra de Ordunte, above Los Tornos mountain pass, there are a few more patches of blanket bogs. One of them, Zalama, is the easternmost blanket bog in Spain.

The blanket bogs in the Cantabrian Range Eastern sector have a number of characteristics in common. All of them have developed on mountains with predominantly siliceous lithology, on summits with a gentle, rounded relief, at altitudes ranging from 1,150 to 1,350 m. These mountains lie in an approximately East-West alignment and are situated less than 45 km in a straight line from the sea, receiving wet winds directly from the sea. For these reasons, the conditions are very favourable for the development of blanket bogs, under an Atlantic climate of cool summers with frequent fogs.

Floristically, Cantabrian Range Eastern sector blanket bogs are very different from those in Galicia. The abundance of ericaceous shrubs (mainly *Erica tetralix* L. and *Calluna vulgaris* [L.] Hull) makes their appearance much bushier than Galician blanket bogs. It is also characteristic the more or less abundant presence of the harestail cotton-grass *Eriophorum vaginatum* L., absent from Galician blanket bogs. In fact, the blanket bogs in the Cantabrian Range Eastern sector are a key habitat for the conservation of *E. vaginatum*, a Cyperaceae that is rare in Spain because it thrives preferentially on deep peaty substrates, and protected in several regional

Figure 4. Blanket bogs at *Motas de Pardo*, in the Montes de Samo (Burgos - Cantabria).

catalogues of threatened plants. On the other hand, contrary to what it may seem, *Sphagnum* species are rare in these blanket bogs, appearing in isolated cushions, not forming a more or less extended carpet. So the plants responsible for the accumulation of peat are not *Sphagnum* species, but *E. vaginatum* (a plant with very vigorous rhizomes) and even the ericaceous shrubs. Curiously, because of their floristic composition and appearance, the Cantabrian Range Eastern sector blanket bogs look more like those found in certain areas of the British Isles (the Pennines, for example) than those in the more nearby Galicia.

Peat exploitation, as mentioned above, has caused the disappearance of certain zones of Cantabrian Range Eastern sector blanket bogs. Nevertheless, from the conservational point of view, another common feature in the blanket bogs of this sector is that they are undergoing erosion processes. At first they would seem to be natural erosion processes, but it is undoubtedly aggravated by anthropic causes that inflict a significant reduction in the original surface area of these bogs.

The Zalama blanket bog

As already indicated, the Zalama blanket bog marks the eastern limit of the distribution of Spanish blanket bogs (Figs. 1 & 2). Further to the East, even taking into account the embryonic peat deposits on *Auza* and *Zuriain* summits in Navarre, the rugged terrain and the abundance of limestone lithologies, typical in the Basque Mountains, prevent the development of blanket bogs. Even further to the East in the Pyrenees, it is the climate becoming markedly continental that prevents it.

The existence of the Zalama blanket bog was revealed by botanists working in the 1980s and 1990s. M. Onaindia discovered the important population of *E. vaginatum* that grows on this bog (Onaindia, 1986; Onaindia & Navarro, 1986/86). Later, its flora and vegetation were characterized by Heras (1990) and Prieto *et al.* (2001). However, its unique nature and significant heritage were pointed out in a study determining its environmental value (Heras, 2002), published a few years later (Heras & Infante, 2003-2004, 2005a). In the same period, a tacheometric survey was also carried out, and the first map drawn, the most detailed of the bog so far (Murillo, 2004).

The bog is situated at 1,330 m a.s.l., a little to the East of the summit of Zalama Mountain, in the site named *Terreros Negros* (Fig. 5). Its original surface area was six hectares, although at the moment only 50 % of this extension is preserved.

Given the topography around Zalama blanket bog, on a summit area where there is no hydrological feeding via springs or water courses, the water that has brought about such an important accumulation of peat could only come from the atmosphere (rain, snow and fog, the latter very frequent on these summits): it is then an ombrotrophic peatland. Since it is adapted to the relief, (slightly convex at the summit where it has developed, and even descending by the north-facing slope (Fig. 5), and therefore not confined to a depression in the terrain), we must consider it as a clear case of a blanket bog. Moreover, because of its location on a secondary summit, we can classify it within the sub-type called “spur blanket bog” (Lindsay, 1995; Charman, 2002).

The peat deposit at the Zalama blanket bog

The depth of the peat deposit in the Zalama blanket bog is remarkable (Fig. 6), although it is variable according to the part of the bog because the deposit presents an asymmetrical distribution. At the northern and eastern edges, where the deposit thins out until merging with the soil and vegetation of the steep slopes, we find the shallowest depth (approximately 50 cm deep), whilst in the central and western edges, lying on the flattest part of the summit, the peat is over two m deep (226 – 232 cm, Souto *et al.,* 2014; Pérez-Díaz *et al.,* 2016).

It is usually typical of blanket bogs to have their peat deposit lying directly on the bedrock or on thin clay deposits. In the case of the Zalama blanket bog, the peat lies on stony ground resulting from the alteration of the bedrock (Early Cretaceous sandstone with abundant quartz crystals). This stony ground is composed of a bed of loose flattened centimetric stones, some of them lift onto their sides and which usually appear concentrically, forming circles or polygons (Fig. 7). This layout is known as a type of periglacial moulding acting on a flat topography (Sanz, 1986).

Figure 5. Views of the Zalama blanket bog in 2002. Top, from Zalama summit, the development of the bog on a summit can be appreciated as well as how it descends mostly down the northern slope. Bottom, from the top of the bog, with Zalama summit in the background.

Figure 6. View of the peat deposit at the Zalama blanket bog in 2002.

Therefore, sediments of lacustrine origin, such as sand or clay resulting from sedimentation in some kind of hollow, are not seen at the base of the peat deposit of the Zalama bog. This confirms the origin of this bog via a paludification process[2].

The Zalama blanket bog has a long history. The beginning of peat accumulation goes back to around 8,000 years ago, according to the dating of the basal horizon of the peat deposit (Ua-35895: 7852 – 8053 ^{14}C cal. yr BP, Pérez-Díaz *et al.,* 2016). This is a considerable age, older than many of the British blanket bogs, which began in slightly more recent times, from 7500 BP onwards (Gallego-Sala *et al.,* 2016).

Also typical in blanket bogs, the peat deposit of the Zalama blanket bog is made up of layers of very homogeneous peat, at least to the naked eye. A more careful examination in the laboratory reveals different layers with different humification degrees, from practically no decomposition - in the more superficial layers of the peat deposit – to highly decomposed at the base, where grains of sand mixed with the peat also appear (Pérez-Díaz *et al.,* 2016). These distinct layers are witness of

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[2] Two essential types of peatbog forming are recognized:
terrestrialization: a bog is the final phase in the evolution of an open water wetland. The basin holding the water body is filled following a succession of different palustrian vegetations as the depth of the water decreases. Raised bogs have usually been originated following this kind of process.
paludification: it means that an originally dryland has become a peatland after the invasion by *Sphagnum* species under an extremely rainy climate.

Figure 7. View of the stony bedrock on which the peat of the Zalama blanket bog lies (top) and in detail one of the "stone circles" or "stone-polygons" there (bottom).

changes in the climate and hydrological conditions during the long history of the bog, as the combined study of the peat layers and the plant macro-remains has demonstrated (Souto *et al.,* 2014). This study has provided evidence of four periods of high humidity at the bog surface, the first at the beginning of its development (8000 – 7400 BP), and then later between 6400 – 5640 BP; 1050 - 480 BP and 480 - 300 BP, interspersed with dry periods and transitions from dry to humid.

The peat deposits in peatlands are historical archives that record changes in the climate, environment and land use by humans. The Zalama blanket bog is no exception: a study of pollen and certain geo-chemical elements has revealed the evolution of the vegetation in the surrounding area over the last 8,000 years (Pérez-Díaz *et al.,* 2016). The main landmarks in the changes in the vegetal landscape are the abundance of pine forests in the early-mid Holocene, the progressive prevalence in later times of deciduous trees, the first evidence of landscape modification due to human activities (Neolithic) around 6500 BP and the recent expansion of beech forests, around 3300 BP.

Unfortunately the peat deposit at the Zalama blanket bog is not complete. A considerable part of it has disappeared, victim of the erosion processes frequently described in similar blanket bogs, for example in Great Britain. This erosion has shaped the appearance of the Zalama blanket bog nowadays, producing some very characteristic morphological features, which can also be seen in the rest of the Cantabrian Range Eastern sector blanket bogs. The following morphological features can be distinguished (Fig. 8):

- *The preserved zone,* the area where the peat deposit remains intact, maintaining the stratigraphical profile and its water table high and close to the surface.
- *The eroded zone,* so damaged by erosion that the peat deposit got fragmented, leaving some very characteristic isolated features, the peat hags, which usually maintain the stratigraphical profile of the deposit more or less complete.

Among these, the peat deposit has disappeared to a greater or lesser extent, or even completely, exposing the bedrock. Erosion fronts appear along the edges with the preserved zone, where the peat profile can be observed. At their feet, peat slopes can be seen made of loose peat coming down from the eroded fronts.

Flora and vegetation at the Zalama blanket bog

The flora and vegetation at the Zalama blanket bog correspond in general to those outlined above for the Cantabrian Range Eastern sector blanket bogs: *Erica tetralix* and *Calluna vulgaris* heath and dominance among the monocotyledons by *E. vaginatum* and *Trichophorum cespitosum* subsp. *germanicum* (Palla) Hegi, which are the main peat-forming species.

The Zalama blanket bog flora counts up to 40 taxa (17 bryophytes -1 leafy liverwort and 16 mosses, of which 3 are Sphagna – and 23 cormophytes, Table 1).

Among these, ten taxa are typical for mire habitats (marked * in the adjoined table), whilst eleven are exclusive to this type of habitats (marked **). Moreover, a total of nine taxa, rare in the Basque Country, live (or lived) in the Zalama blanket bog (marked *r*). A bryofloristic study of the area (Heras, 1990) confirmed the presence of nine taxa of hydrophilous bryophytes, all of them typical in mires, which in later visits were not observed (marked †). Their disappearance is interpreted as a sign of the drying out of the bog surface.

As corresponds to a homogeneous biotope, the vegetation of the Zalama blanket bog is not very varied and relatively uniform. It is dominated by a dense and quite dry peaty heath, dominated by heather (*C. vulgaris*) with abundant cross-leaved heath (*E. tetralix*). However, the different morphological features impose a certain diversity level, mainly determined by soil waterlogging, which brings changes in the frequency and dominance of certain plants and allows several plant communities to be differentiated (Fig. 9).

ZALAMA BLANKET BOG

diagram of the main morphological features of the bog

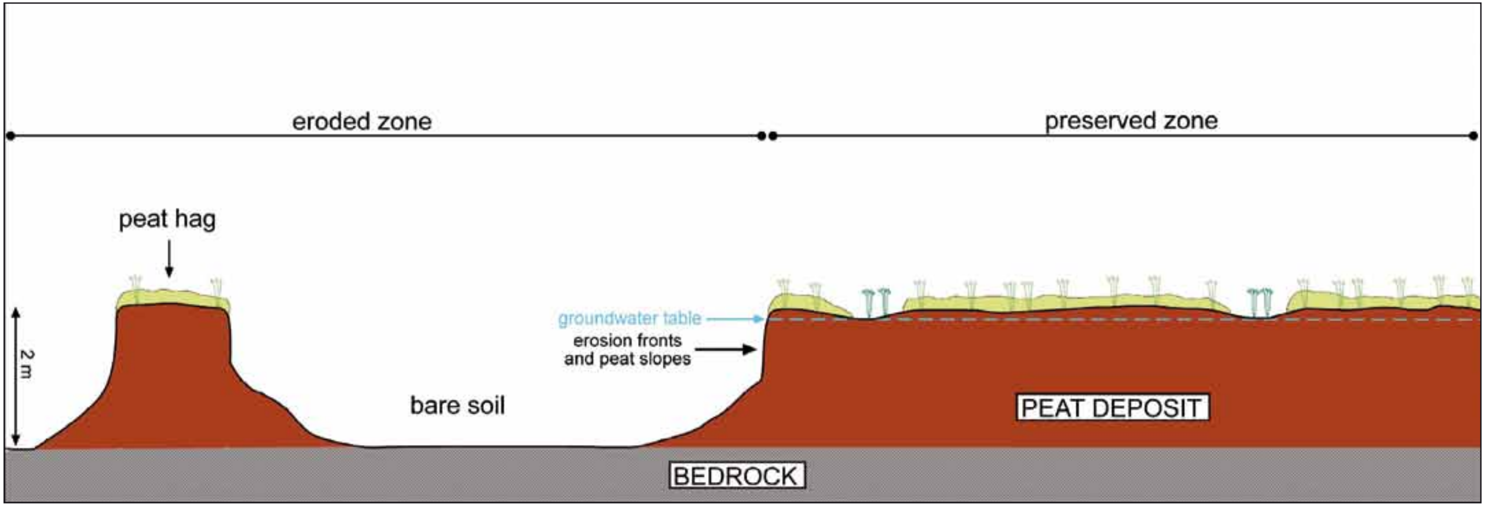

Figure 8. Diagram of the main morphological features at the Zalama blanket bog.

Agrostis capillaris L.
Agrostis curtisii Kerguélen
Aulacomnium palustre (Hedw.) Schwägr. (**) (musgo) (†)
Bryum alpinum Huds. ex With. (musgo) (†)
Calluna vulgaris (L.) Hull (*)
Campylopus flexuosus (Hedw.) Brid. (*) (musgo) (*r*) (†)
Campylopus subulatus Schimp. ex Milde (musgo) (*r*) (†)
Campylopus introflexus (Hedw.) Brid. (*) (musgo)
Carex echinata Murray (**)
Daboecia cantabrica (Hudson) C.Koch
Deschampsia flexuosa (L.) Trin.
Dicranum bonjeanii De Not. (**) (musgo) (*r*) (†)
Dicranum scoparium Hedw. (musgo)
Epilobium tetragonum L. subsp. *tetragonum*
Erica cinerea L.
Erica tetralix L. (**)
Eriophorum angustifolium Honckeny (**) (*r*)
Eriophorum vaginatum L. (**) (*r*)
Festuca laevigata Gaudin
Festuca nigrescens Lam. subsp. *microphylla* (St-Yves) Markgr.-Dannenb.
Galium saxatile L.
Gymnocolea inflata (Huds.) Dumort.(*) (hepática) (*r*) (†)
Hylocomium splendens (Hedw.) Schimp. (musgo)
Hypnum cupressiforme Hedw. var. *cupressiforme* (*) (musgo)
Juncus effusus L. (*)
Juncus squarrosus L. (*) (*r*)
Leptodictyum riparium (Hedw.) Warnst. (musgo) (†)
Molinia caerulea (L.) Moench. (*)
Pleurozium schreberi (Willd. ex Brid.) Mitt. (musgo)
Polygala serpyllifolia J.A.C. Hose
Polytrichum commune Hedw. var. *commune* (**) (musgo) (†)
Polytrichum juniperinum Hedw. (musgo)
Potentilla erecta (L.) Raeuschel (*)
Rumex acetosella L. subsp. *angiocarpus* (Murb.) Murb.
Sedum anglicum Hudson
Sphagnum auriculatum Schimp. (**) (musgo)
Sphagnum capillifolium (Ehrh.) Hedw. (**) (musgo)
Sphagnum cuspidatum Ehrh. ex Hoffm. (**) (musgo) (*r*) (†)
Trichophorum cespitosum (L.) Hartm. subsp. *germanicum* (Palla) Hegi (**) (*r*)
Vaccinium myrtillus L. (*)

Table 1. Species check-list for the Zalama blanket bog (Heras, 1990, 2002).

Current situation and state of conservation of the Zalama blanket bog

Three main threats affect Spanish blanket bogs:

1. *Erosion*. Although erosion could be initially due to the natural dynamics and evolution of blanket bogs, today the main causes are fires and burnings for cat-

tle grazing management. In dry summers, even the peat can take fire, producing scars in the upper layers of the bog, leaving more or less large parts of the peat deposit exposed to erosion by runoff and wind. Erosion is responsible for the formation of long fronts of exposed peat, more and more vulnerable to the effects of the wind, and also the formation of isolated peat hags. Moreover, cattle activity contributes to aggravate this situation. It must be remembered that 50 % of the original surface of the Zalama blanket bog has been lost through erosion.

2. *Windfarms installation*. This is the principal problem for the conservation of blanket bogs in the Sierra do Xistral in Galicia (Heras *et al.,* 2017) (Fig. 3). Although, at least for the time being, this is not a serious threat to the bogs in the Cantabrian Range Eastern sector, it must be mentioned that in recent times there was a windfarm project in the Montes de Ordunte which intended the installation, after removing the peat from Zalama, of a windmill right on the summit of *Terreros Negros* (PTS Energía Eólica 2002; Heras & Infante, 2005b, 2008). Fortunately, the windfarm project was paralyzed in 2004 and it seems that this threat to the Zalama blanket bog and its surroundings has been apparently ruled out.

3. Although peat extraction for domestic use exists in these bogs, this is at a very small scale owing to the bog's remoteness from populated areas, thus its consequences on the conservation are almost none. On the other hand, *mining exploitation* does create a serious threat to Cantabrian Range Eastern sector blanket bogs. As indicated above, peat exploitation has caused the complete disappearance of the *Cruz del Marqués* and *Cueto de la Avellanosa* bogs (Cantabria). Fortunately, this aggression has not occurred at the Zalama blanket bog.

However, there has been an aggression at the Zalama blanket bog until recent times; vehicle traffic (quads, motor bikes and mountain bikes) (Fig. 10) and the organization of sporting events such as a marathon race (*Galarleiz Mendi Maratoia*) which aggravated erosion.

Figure 10. Erosion caused by vehicle traffic at the Zalama blanket bog in 2007.

ZALAMA BLANKET BOG

diagram of the main plant communities

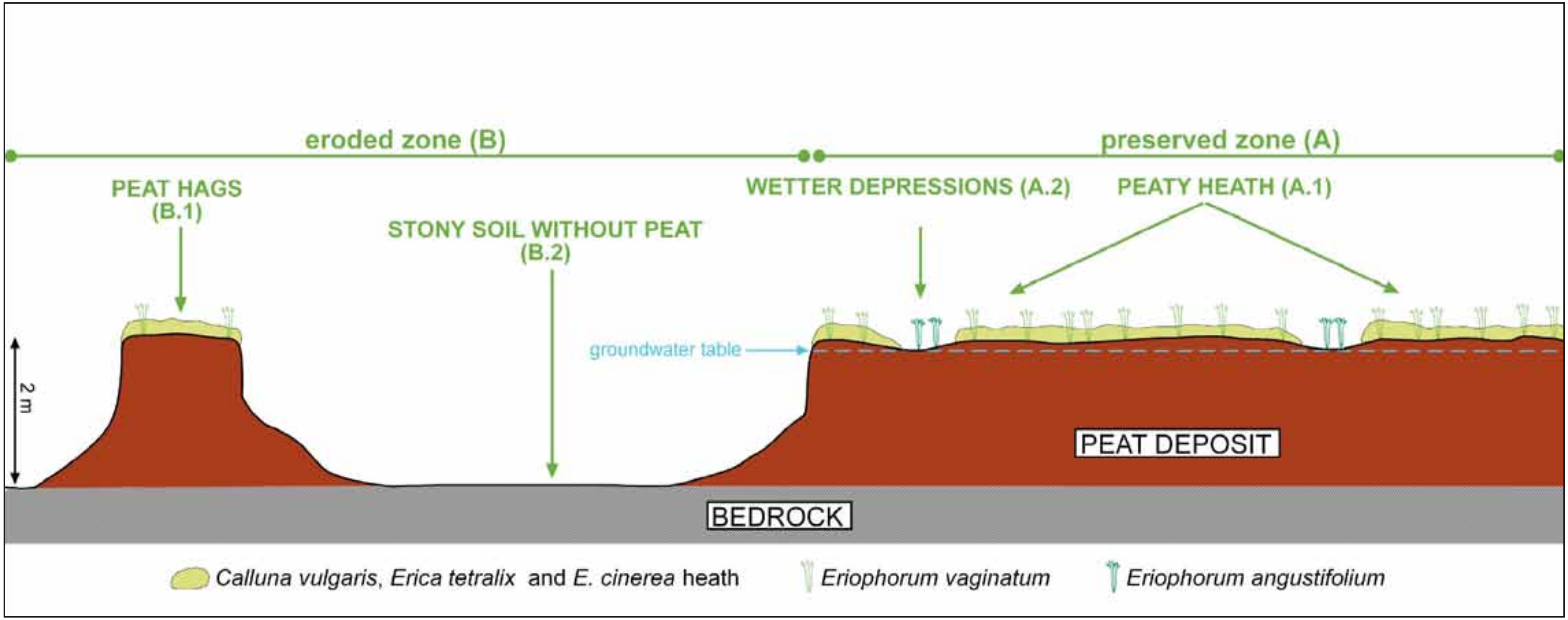

Figure 9. Diagram of the plant communities at the Zalama blanket bog.

A.1) The peaty heath dominating the preserved zone is characterized by the abundance of *Calluna vulgaris*, *Erica tetralix* and *E. cinerea*, with *Daboecia cantabrica* more or less frequent. The most frequent herbaceous plants are the Poaceae *Deschampsia flexuosa*, *Festuca nigrescens* subsp. *microphylla*, *Agrostis curtisii* and *Molinia caerulea* and the Cyperaceae *Eriophorum vaginatum* and *Trichophorum cespitosum* subsp. *germanicum*, as well as the small rosaceous *Potentilla erecta*. Sphagna are very rare, appearing only sporadically some cushions of *Sphagnum capillifolium*, protected under heath shrubs; carpets of moss *Hypnum cupressiforme* var. *cupressiforme* are more frequently seen, also under heath shrubs. A wetter variant of this heath is found in the northern part of the Zalama blanket bog, sloping down on the north-facing mountainside. The higher humidity induced by the north orientation allows the development of the blueberry (*Vaccinium myrtillus*) with a certain frequency. Besides, the steep slope causes slipping phenomena and the appearance of channels or "gullies" with a wetter bottom, hosting rushes (*Juncus effusus* and *J. squarrosus*), some cushions of *Sphagnum auriculatum* and the mosses *Pleurozium schreberi* and *Hylocomium splendens*, in the mulch under the heath shrubs.

A.2) Wetter depressions, appearing here and there within the above community. The water table closer to the surface changes the presence frequencies of most species mentioned above (more abundance of *Erica tetralix* and less of *Calluna vulgaris* and *E. cinerea*) and induces the appearance of some more hydrophilous plants, in particular *Eriophorum angustifolium*, but also *Carex echinata* and *Juncus effusus*.

B.1) In the eroded zone, on the surface of the peat hags, considerably drier than in A.1, an impoverished peaty heath develops, mainly composed of *Calluna vulgaris*, with *Eriophorum vaginatum*, *Erica tetralix* and *Potentilla erecta*.

B.2) When the peat deposit has disappeared, the stony ground and sometimes also naked dry peat are exposed. These areas show a scattered plant cover with species from the Atlantic heath and the siliceous grasses from the neighbouring areas, such as *Calluna vulgaris*, *Erica cinerea*, Ulex gallii, *Agrostis curtisii*, and also pioneer and colonizing species typical from altered areas, such as *Sedum anglicum* and *Rumex acetosella* subsp. *angiocarpus*.

It is worth noting the presence of an invasive moss, coming from the Southern hemisphere, *Campylopus introflexus*, which is an indicator of disturbance in peaty hydrophilous habitats, and acidophile heaths. At this moment, it is abundant in any area of the Zalama blanket bog where substrate has been disturbed.

When the Ordunte windfarm project was paralyzed, a series of actions followed aiming at improving the state of conservation of the Zalama blanket bog. For example, one of the first was the diversion to the aforementioned route of the sport event, the *Galarleiz Mendi Maratoia*, which was run across the bog. This action was accompanied with the publication of a leaflet explaining and justifying the motive for this diversion and providing information about the environmental value of the area (Heras & Infante 2005c).

Finally, since 2008-2009, thanks to an initiative from the Bizkaia Provincial Council (Diputación Foral de Bizkaia-DFB), a restoration plan has been set up whose first objective was to exclude cattle by enclosing the bog with a fence (Fig. 11). The second objective was to stabilize and protect the peat fronts and exposed areas by covering them with geotextile (Fig. 11). The enclosing fence tries to favour the optimum development of the bog vegetation, whilst the geotextile covering is preventing the slow but constant loss of surface by erosion. Simultaneously, a reinforcement of the *E. vaginatum* population has been made. This population is the only one in the Basque Country and one of the main environmental assets of the Zalama blanket bog (Fig. 12). This reinforcement has been made using the plants grown in a nursery from seeds taken from the population in the Zalama blanket bog. This last action is simultaneous with the monitoring of the *E. vaginatum* population, initiated in 2007. All these actions are important since the blanket bogs of the Cantabrian Range Eastern sector are decisive for the conservation of this Cyperaceae in Spain (Heras & Infante, 2015). *E. vaginatum* is listed in the regional catalogues of threatened species in Asturias, Castilla y León and the Basque Country. The actions carried out so far are benefitting this species, with the population reinforcements being considered positive, and without any competition problem with other species for the time being. The flowers and fruits of this plant are regularly produced, with oscillations arising from the variations in the meteorological conditions from one year to another. The effects of re-opening the area to grazing by cattle is still to be assessed (Prieto, 2015).

Figure 11. Restoration actions at the Zalama blanket bog. View of the protective fence in 2014 (left). View of the covering of the peat fronts with geotextile, also in 2014 (right).

References

AEMET, 2011. *Atlas Climático Ibérico: Temperatura del Aire y Precipitación (1971-2000)*. Madrid.

Charman, D. 2002. *Peatlands and Environmental Change*. John Wiley & Sons Ltd.

Foss, P.J., O'Connell, C.A., Crushell, P.H. 2001. *Bogs & Fens of Ireland Conservation Plan 2005*. Irish Peatland Conservation Council.

Gallego-Sala A.V., Charman D.J., Harrison S.P., Li G., Prentice I.C. 2016. Climate-driven expansion of blanket bogs in Britain during the Holocene. *Clim. Past*, 12: 129-136.

Gaudillat, V., Haury, J. (coords.) 2002: *Connaissance et gestion des habitats et des espèces d'intérêt communautaire. Tome 3: Habitats humides*. Cahiers d'Habitats Natura 2000. Ed. La Documentation Française, Paris.

Heras, P. 1990. Estudio briológico de las turberas de Los Tornos y Zalama. *Cuadernos de Sección, Ciencias Naturales* 7: 117-137. Sociedad de Estudios Vascos - Eusko Ikaskuntza.

Heras, P. 2002. *Determinación de los valores ambientales de la turbera del Zalama (Carranza; Bizkaia) y propuestas de actuación para su conservación*. Unpublished report. Dirección de Aguas. Departamento de Ordenación del Territorio y Medio Ambiente. Gobierno Vasco.

Heras, P., Infante, M. 2003-2004. La turbera de cobertor del Zalama (Burgos - Vizcaya): un enclave único en riesgo de desaparición. *Estudios del Museo de Ciencias Naturales de Álava* 18/19: 45-53.

Heras, P., Infante, M. 2005a. La turbera del monte Zalama. *Quercus* 231: 24-30.

Heras, P., Infante, M. 2005b. Zalama in Northern Spain. En: More about windmills and blanket bogs. *International Mire Conservation Group Neswsletter* 2005/1: 22.

Heras, P., Infante, M. 2005c. *Urperatutako gailurrak, Zalamako turbategia eta Saldueroko zingiradiak – Las cimas anegadas: La turbera del Zalama y los trampales de Salduero*. Diputación Foral de Bizkaia, Bilbao.

Heras, P., Infante, M. 2008: Wind farms and mires in the Basque Country and north-west Navarra, Spain. *Mires and Peat* 4: Art. 4. (Online: http://www.mires-and-peat.net/map04/map_04_04.htm)

Heras, P., Infante, M. 2015. Las turberas de cobertor del sector oriental de la cordillera cantábrica, hábitat clave para la conservación de las poblaciones españolas de *Eriophorum vaginatum. VII Congreso de Biología de Conservación de Plantas, Vitoria-Gasteiz (Araba/Álava). Libro de resúmenes*: 54.

Heras, P., Infante, M., Pontevedra-Pombal, X., Nóvoa-Muñoz, J.C. 2017. Spain. In: *Mires and peatlands of Europe. Status, distribution and conservation*. H. Joosten, F. Tanneberger & A. Moen (eds.): 639-656. Schweizerbart Science Publishers, Stuttgart.

Izco, J., Ramil, P. 2001. *Análisis y valoración de la sierra de O Xistral: un modelo de aplicación de la directiva hábitat en Galicia*. Xunta de Galicia.

Lindsay, R.A. 1995. *Bogs: The ecology, classification and conservation of ombrotrophic mires*. Scottish Natural Heritage.

Lindsay, R.A., Charman, D.J., Everingham, F., O'Reilly, R.M., Palmer, M.A., Rowell, T.A., Stroud, D.A. 1988. *The Flow Country: The peatlands of Caithness and Sutherland*. Na ture Conservancy Council.

Martínez, A. 2001. Las turberas: terminología, tipos y clasificaciones. En A. Martínez, E. García-Rodeja (coords.) *Turberas de montaña de Galicia*: 99-127. Xunta de Galicia, Consellería de Medio Ambiente.

Mariscal, B. 1983 Estudio polínico de la turbera del Cueto de la Avellanosa, Polaciones (Cantabria). VI Reunión del Grupo Español de Trabajo del Cuaternario. *Cadernos do Laboratorio Xeolóxico de Laxe* 5: 205-226.

Murillo, B. 2004. *Levantamiento taquimétrico turbera de Zalama*. Escala 1:500. Eólicas de Euskadi. Gorosti Neurketak.

Onaindia, M. 1986. *Ecología vegetal de las Encartaciones y Macizo del Gorbea –Vizcaya–*. Servicio Editorial de la Universidad del País Vasco.

Onaindia, M., Navarro, C. 1985/86. Comunidades vegetales en los ambientes de turbera de Vizcaya: vegetación de carácter relíctico en nuestro territorio. *Kobie (Serie Ciencias Naturales)* XV: 199-200.

Pérez-Díaz, S., López-Sáez, J.A., Pontevedra-Pombal, X., Souto-Souto, M., Galop, D. 2016. 8000 years of vegetation history in the northern Iberian Peninsula inferred from the palaeoenvironmental study of the Zalama ombrotrophic bog (Basque-Cantabrian Mountains, Spain). *Boreas* 45: 658-672.

PTS Energía Eólica, 2002. Comunidad Autónoma Vasca. Gobierno Vasco, Departamento de Industria, Comercio y Turismo.

Prieto, A. 2015. *Seguimiento anual de poblaciones de táxones catalogados en peligro de extinción en el Territorio Histórico de Bizkaia*. Diputación Foral de Bizkaia.

Prieto, A., Heras, P., Infante, M., Barraqueta, P.

2001. *Estudio botánico de los esfagnales del monte Zalama y Llanos de Salduero*. Saldropo, Grupo para el Estudio y Protección de los Humedales Continentales. Departamento de Agricultura y Pesca. Gobierno Vasco.

Ramil, P., Rodríguez, M.A., Muñoz, C. 1996. Distribución, génesis y caracterización botánica de las turberas ombrotróficas de Galicia. Tomo Extraordinario. 125 *Aniversario de la Real Sociedad Española de Historia Natural*: 253-256.

Sanz, C. 1986. Periglaciarismo en montaña: la Sierra de Guadarrama. *Atlas de Geomorfología*: 239-254. E. Martínez de Pisón, B. Tello (eds.). Alianza Editorial.

Souto, M., Pontevedra-Pombal, X., Castro, D., López-Sáez, J.A., Pérez-Díaz, S., García-Rodeja, E., Fraga, M.I. 2014. Reconstrucción paleoambiental de los últimos 8.000 años de la turbera de Zalama (Sierra de Ordunte, País Vasco). In Macías F., Díaz, M., Barral, M.T. (eds.). *Retos y oportunidades en la Ciencia del Suelo*, pp. 53-56.

Figure 12. *Eriophorum vaginatum*

11

Genesis and dynamics of the bogs in the Atlantic Biogeographic region of the Iberian Peninsula

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Abstract

This study analyzes the ecological processes and communities that brought about the creation of bog ecosystems in the Atlantic region of the Iberian Peninsula, in view of the important role that they have as reservoirs of biodiversity, the reason why they have been included within the several types of habitats of interest for conservation, protected by the Habitats Directive 92/43/CEE. The environmental importance of bogs also lies in their important function as natural sinks for carbon over the long-term, and in general for the importance of the ecosystem services (regulation, provisioning, cultural) that they provide. The bogs in North-west of the Iberian Peninsula are made up of the oldest representatives of the bog ecosystems in the whole of the European Atlantic region, these bog sediments have produced a series of pollen sequences longer than in more northern areas (France, UK, etc.). The importance of bog ecosystems in the Atlantic region of the Iberian Peninsula is capital, providing long continuous palaeo-ecological and palaeo-climatic records that are of high accuracy. Having this information means that changes in the climate, the biodiversity as well as in the countryside in the past can be evaluated, at the same time they provide a valuable reference for the creation of predictive models of scenarios of changes in the future.

Keywords: bogs; EUNIS; Habitats Directive; distribution; Holocene; exploitation of peat bogs.

Introduction

Bog ecosystems are a key component in the conservation of natural heritage and biodiversity at a planetary scale. Their peculiarity lies in the fact that they are made up of habitats, biocenosis and species associated with some very specific physical factors that can be combined in very different ways throughout the different biogeographic regions on the Earth. In spite of this, most of their components, especially those of the habitats, are threatened and may disappear, within their area of natural distribution, a result of global climate dynamics and their exploitation by mankind since time immemorial. The consequence being that most bog ecosystems currently show a reduced zone of distribution that is also fragmented compared to the situation found in the first half of the 20th century, or before the Anthropocene and Holocene periods.

In addition to the aspects related to biodiversity, the environmental importance of bog ecosystems lies in, among other aspects, their role as a natural sink for carbon and for the regulation of the hydrological cycle, functions directly related to the sedimentation and accumulation of organic material that occurs in these zones over prolonged periods of time. For this reason, they are an exceptional resource for carrying out palaeo-ecological studies and of palaeo-climatic dynamics in local or regional zones whose reliability, in environmental and chronological terms, is noticeably higher than those obtained from similar studies carried out in karstic zones or archaeological sites.

In addition to the eco-systemic services mentioned above, there is also the fact that bogs have provided and continue to provide an important service in the maintenance of extensive cattle-raising in zones of the European Atlantic region. In these zones the vegetation cover, together with puddles (temporary or permanent) and springs that sprout up in these bog areas have been used from their very beginning as spring pastures (high mountain meadows - *brandas, brañas, breñas,* etc.) and as zones for animals to drink. An important ethno-biological heritage was forged related to these traditional uses, particularly developed and diversified throughout the different mountainous areas in these territories, which unfortunately remains little studied or evaluated. Owing to this extensive exploitation, and in spite of their territorial regression, bogs are still an essential element in large parts of the countryside, helping them to maintain a particular natural and cultural appeal.

Peat is a sediment of organic origin, created as a result of accumulation, year after year, of the remains formed by the hydrophylic biocenosis that takes place in the upper section. Owing to the fact that this sediment is usually found saturated in water and at low oxygen pressure, the breakdown of organic remains periodically supplied by biocenosis activity is only partially completed. In this way fragments of plants (remains of vegetable tissues such as epidermic cells, stomas, fibres, and vessels, as well as resistant structures, such as seeds, pollen and spores from ferns and bryophytes), fungi (fragments of hyphae and spores) and to a lesser extent, animals (ocelli from insects, fragments of mites), which in their moment were part of the biocenosis, begin to accumulate in superimposed layers and are a reliable reflection of the dynamics of the environment and the conditions under which they carried out their life cycles. The accumulation of these remains over long periods of time brings about the formation of peat deposits whose depth and size is a consequence of the interaction of several distinct ecological factors.

The drawing up of annex I of the Habitats Directive 92/43/CEE was based on the system of classification developed via the Corine Biotopes program (EUR 13231, 1991), which currently is to be found integrated and implemented within the EUNIS Habitat Classification system (Davies *et al.,* 2004). This annex includes a series of types of habitat that make up part of the different bog ecosystems currently found within the EU. Table 1 shows the current distribution of bog ecosystems,

COD.	HABITAT (short name)	PT	AC	LU	OU	PO	AS	CA	BZ	GK	AR	NA	BU	PA	LE	ZA
7110*	Active raised bogs	•	•	•	•	•	•	•				•	•	•	•	•
7120	Degraded raised bogs still capable of natural regeneration		•	•			•	•								
7130*	Blanket bogs – still active		•	•				•	•				•			
7130	Blanket bogs			•												
7140	Transition mires and quaking bogs	•	•	•	•	•	•	•	•	•	•	•	•	•	•	•
7150	Depressions on peat substrates of *Rhynchosporion*	•	•	•	•	•	•	•			•	•	•	•	•	•
7210*	Calcareous fens with *Cladium mariscus*		•	•		•	•		•	•	•					
7220*	Petrifying spring with tufa formation (*Cratoneurion*)	•	•	•	•	•	•	•	•	•	•	•	•	•	•	•
7230	Alkaline fens	•	•	•	•	•	•	•			•	•	•	•	•	•
91D0*	Bog woodland			•	•											
4020*	Temperate Atlantic Wet heaths with *Erica ciliaris* and *Erica tetralix*	•	•	•	•	•	•	•	•	•	•	•	•	•	•	•

Portugal (PT), A Coruña (AC), Lugo (LU), Ourense (OU), Pontevedra (PO). Asturias (AS), Cantabria (CA), Bizkaia (BZ), Gipuzkoa (GK), Araba (AR), Navarra (NA), Burgos (BU), Palencia (PA), León (LE), Zamora (ZA).

Table 1. Presence of habitats of community interest (Annex I DC 92/43/CEE) and characteristics of bog ecosystems in the different administrative regions of the Atlantic Biogeographical region of the Iberian Peninsula.

based on administrative regions and according to the information compiled in the LIFE+ Tremedal, and the type of habitats of community interest representative of bog ecosystems that are thought to be present in the territory making up the Atlantic Biogeographical region of the Iberian Peninsula. Six of the types of habitats are considered by Directive 92/43/CEE as priority habitats. The definition and characteristics of these habitats have been established officially in the *Interpretation Manual of European Union Habitats* EUR28 (European Commission, 2013). This information can be complemented with the information contained in the manuals published for the distinct geographical areas: Galicia (Ramil-Rego *et al.*, 2008), Basque Country (Ihobe, 2008, 2011), Navarra (Peralta *et al.*, 2013), Castilla and León (Escudero *et al.*, 2008).

Most types of characteristic bog ecosystem habitats are included in group 7 “Raised bogs and mires and fens”. Sub-group 71 includes bogs with acidic or very acidic water and sediments and in consequence with acidophilic biocenosis, and usually in the presence of species from the genus *Sphagnum*, which at some point in the evolution of the bogs became dominant. Following the traditional scheme of bog classification (Taylor, 1983; Moore, 1984; Rodwell, 1991; Davies *et al.*, 2004;

European Commission, 2013), in subgroup 71 the classification includes, based on their ecological characteristics and especially their biocenosis, Active blanket bogs (7130*) and the Active raised bogs (7110*). Human activity (pasture, removal of undergrowth, burning, planting of species, etc) determines the fact that in some places with raised and blanket bogs they are found in a degraded state but recoverable once the factors that negatively affect them are eliminated (7120 Degraded raised bogs, 7130 Non-active blanket bogs). A third type of acidic bog corresponds to those known as Floating bogs, included in type 7140 Transition mires, which also include Transition environments between acidic bogs and other natural habitats. Finally, within the acidic bog group there are also pioneer habitats, represented by *Rhynchosporion* biocenosis (7150 Depressions on peat substrates of the *Rhynchosporion*).

The subgroup 72 Calcareous fens, include those with neutral or base water and sediments and their corresponding biocenosis adapted to the characteristics of the biotope. This defines the type 7230 Alkaline fens, represented by hydrophilous herbaceous formations (*Carex davalliana*, *C. demissa*, *C. cuprina*, *C. riparia*, *C. panicea*, *C. pulicaris*, *Juncus conglomeratus*, *Schoenus nigricans*), and can have a high level of brown moss cover (gens. *Bryum*, *Campylium*, *Fissidens*, etc.) with an absence, or very scarce presence, of species from the *Sphagnum* genus. The formations of large helophytes (*Phragmites australis*, *Scirpus maritimus*, *Scirpus lacustris*), with the presence of *Cladium mariscus*, make up a particular type of fen, and corresponds to type 7210 * Calcareous fens with *Cladium mariscus* and species of the *Caricion davallianae*, whilst habitat 7220* includes Pwith tufa formation (*Cratoneurion*).

In addition to the bog habitats included in Group 7, there are another two types of habitat present in these ecological niches. On the one hand, there are Bog woodlands, widely distributed in northern areas of the European Union, but whose presence in the Atlantic region of the Iberian Peninsula is scarce, only present in small areas. Bog woodlands are included in the subgroup of Forests of Temperate Europe set out in Directive 92/43/CEE. Secondly, there are hygrophilous and hygroturfophilous heaths belonging to habitat 4020* Temperate Atlantic wet heaths with *Erica ciliaris* and *E. tetralix*, whose bio-geographical distribution is wider than those of previously mentioned subgroups.

The oldest bogs in the Atlantic territories

With the passing of time, peat sediments undergo important changes to their physical-chemical properties, affecting the remains of vegetation and animals that they contain (taphocoenosis). On a timescale of hundreds or thousands of years, the most recent peats typically have a brown colour and a low density, and at times are over 90% water by volume. Macroscopic fragments of fibres and other remaining vegetation (macroremains) can be seen in them. On the other hand, older peats tend to have darker colours, tones of black/grey, with a lower water content, and the same with the quantity of macroremains. In these, the peat matrix has a high concentration of microremains (pollen, spores, small fragments of tissue, etc.). On a timescale of millions of years, the peat sediments that have been integrated at deep levels within the sedimentary deposits undergo important physical-chemical changes which are characterized by the reduction in the percentage of volatile compounds and humidity, whilst there is an increase in the percentage of carbon. This process allows different types of sedimentary materials to be classified, making up what is known as “the carbon series”: peat – lignite – coal – anthracite.

In the territories of the Atlantic Biogeographic region, coal and anthracite deposits can be found, formed during the Carboniferous and frequent in different towns in the provinces of Asturias, León and Palencia, as seams of brown coal (lignites and lignitic fossilized peats), whose formation can be dated to the end of the Tertiary (Neogene) and is found to be well-represented in several basins in the interior of Galicia (Médus, 1965; Médus & Nonn, 1963; Menéndez, 1975; IGME, 1974,1979; Peña, 1993).

It is more unusual to find references to the presence of sediments of peat from the older stages of the Pleistocene (1,800 – 750 thousands of years, kyr), an absence that is generalized in the rest of the territories in Europe. For the Middle Pleistocene (750-128 kyr), the number of references to peat deposits rises, worth mentioning are the sequences from Padul (Florschütz *et al.,* 1971), and the Grande Pile, Vosges, France (Woillard, 1978; De Beaulieu & Reille, 1992). In the former a mechanical probe allowed a sequence of 60 m of peat to be recovered, whose sedimentation began in the Mindel-Riss interglacial period (300,000 years ago) and ended in the middle of the Holocene (Florschütz *et al.,* 1971).

Table 2 (a & b) shows, by province, the main pollen records that were obtained from peat sediments in the North-west Iberian Peninsula. The main sequences and areas with the presence of bogs are numbered. This numeration is used in the text as a reference and in Fig. 1 to show their position.

In agreement with the climatic sequences relating to the North-atlantic zone (Martinsson *et al.,* 1987; Mix *et al.,* 1992), after the end of the Riss Stadial (MIS 6) the interglacial Riss-Würm started (MIS 5), whose first stage (MIS 5e 123-109 kyr BP), called Eemian and which witnessed a large increase in the global temperature around the planet, caused the glaciers to retreat, both at the poles as well as in continental territories. In response to this warming the sea level rose, reaching a dimension of some +6 m, or even +7 m higher than current levels. The encroachment of the sea during the Eemian brought about an important transformation to coastal zones, raising the level of erosion on cliffs and generating new deposits of pebbles and sand as well as layers created by wind in areas that had previously been occupied by continental ecosystems, which have been documented in different coastal sectors of the Atlantic region of the Iberian Peninsula.

When the sea level dropped these deposits were colonized by coastal and continental vegetation. Coastal wetlands developed on some of them, with formation and sedimentation of peat-like material, several of which have been studied. This is the case of the La Franca coastal deposit (6), where the presence of macroremains of woody species has not been proved. The subsequent coastal dynamics fossilized the peat at La Franca, which was then covered by several layers of detritic materials of continental origin, some of which are mixed with marine sand. Chronologically, these layers of peat were attributed to the sub-stadial MIS5a: 82-70 kyr BP (Gómez-Orellana *et al.,* 2007).

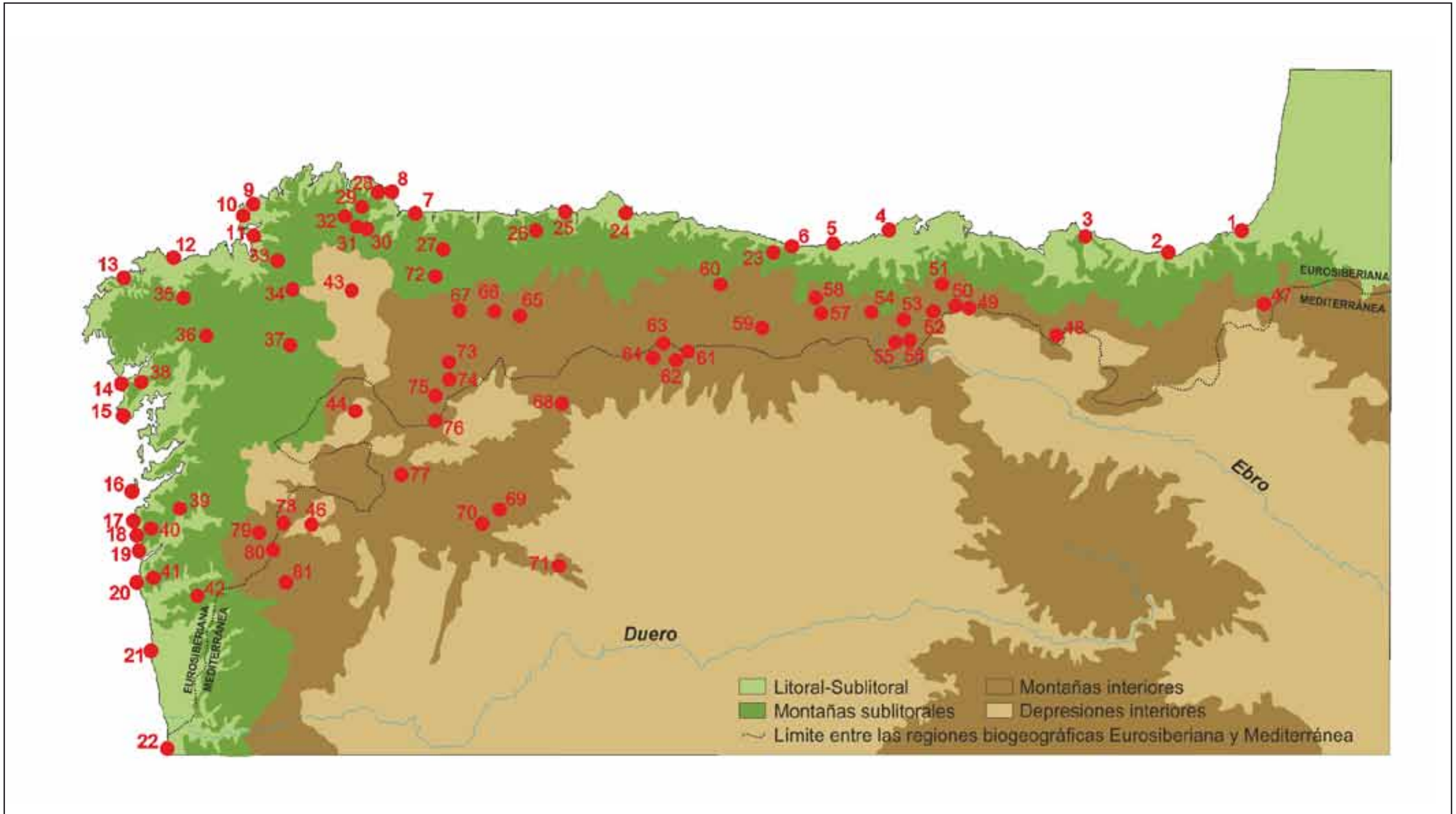

Figure 1. Geographic location of peat bogs in Northeastern Iberian Peninsula. Codes are identified in the text.

The sequence from Area Longa turns out to be more complex (8), several different sequences have been obtained, from both the lower beach and the slope from the cliff. The latter position was tested with a mechanical probe after removing the upper layers of detritus - three metres of peat sediments were extracted - these lie on a layer of clay that alternates with small layers of peat which in turn are superimposed at beach level with abundant quartz pebbles. Within the clay level several woody-remains of *Erica* (cf. *Erica australis*) were found in several parts of the deposit, some of which correspond to parts of roots and from the neck of the plant, situated in primary position; in other cases, the fragments recovered come from twigs and large branches. A change in the hydrological dynamics favoured the rapid invasion of heather by a wetland of a peaty nature, with a part of the woody fragments of *Erica* embedded in the peat layer. Pollen analysis of these materials reflect the characteristics of the environmental conditions in the MIS 5c Stadial. Above the levels attributed to MIS 5c at Area Longa (8) detritic layers of clays and sands are found, and on these lies a second packet of peat, of greater depth, whose sedimentation started at the beginning of MIS 4 (71-57 kyr BP) and which ended with MIS 3 (57-29 kyr BP), covering, in consequence, the whole of the Early Würmian Stadial and most of the Würmian interstadials (Gómez-Orellana *et al.,* 2007).

The conditions during MIS 3 were more suited to the development of vegetation than those recorded during the period of the MIS 4 Stadial. The improvement in environmental conditions favoured the appearance of wetlands both inland as well as on the coast and subcoastal zones. Several peat layers have been documented that correspond to this phase in the formation of wetlands: *Moucide* (28), *Reiro* (12), *Sorrizo* (12), *Caamaño* (14), *Oia* (18) and *San Xián* (19), that complement the records obtained from Area Longa (Gómez-Orellana, 2002; Gómez-Orellana *et al.,* 2007). The typical vegetation that can be deduced from these examinations turns out to be to a large extent similar to the current flora, deciduous species are dominant (*Quercus*, *Betula*, *Ulmus*, *Alnus*, *Fraxinus*, *Salix*, etc), although some of them (*Carpinus*, *Fagus*, *Tilia*) possess a range that is larger than the actual one. The presence in the countryside of elements such as *Castanea* and *Juglans*, was confirmed in all these cases. These were already present in pollen diagrams attributed to the Tertiary, indicating their prolonged existence in the region. As for gymnosperms, in this period pines were present (*Pinus pinaster* / *P. pinea*, *P. sylvestris*), so are yews (*Taxus*), junipers (*Juniperus*), firs (*Abies*) and spruce (*Picea*), although somewhat scarce, especially in the coastal and sub-coastal areas.

The similarity between the floral elements in Stadial MIS 3 and the actual interglacial period (Holocene) contrasts with the important differences that are observed in relation to the species of megafauna. Indeed, an important number of large-sized species, considered as archetypes of the Ice Age, present in MIS 3 were wiped out during MIS 2 (mammoth, woolly rhinoceros, giant deer, cave lions), whilst others (beaver, bison, wild bull, leopard, hippopotamus, hyena, etc), where confined to different zones, but all the same, disappeared from the Cantabrian-Atlantic territories. The final Stadial period of the Pleistocene was identified in the isotopic sequences as the MIS 2 Stadial (29-14 kyr BP) and is related to a drop in the global temperature, which unleashed an increase in the erosion processes both in mountain areas, as well as on the coasts. Both factors played a very significant role in the distribution and configuration of land ecosystems, particularly, on bogs. In consequence, the peat sediments, whose sedimentation coincides with what is called the LGM - Last Glacial Maximum - and which corresponds to interval 23-20/19 kyr BP, are scarce in the Iberian Atlantic biogeographical territories (Gómez-Orellana *et al.,* 2013), with the sequences obtained from the deposits from the *Río Boó* (31), *Caamaño* (14) and *Oia* (18) standing out. In the same way, the prevailing environmental conditions in this period had a negative impact on the preservation of deposits formed in previous times. So, many of them were completely eroded, and only a few retained a part of their sequences buried under large amounts of detritic material. The formation of peat in these deposits stopped many thousands of years ago, and in consequence are considered as fossilized bogs.

The formation of bogs after the LGM

In the Northern hemisphere, the end of the LGM and the start of deglaciation has been calculated at between 20/19 kyr BP (Clark *et al.,* 2009). From that moment on, the increase in the temperature brought about a modification to the ecosystems in the Cantabrian-Atlantic territories. In mountainous areas, the retreat of the ice exposed many depressions where shallow type lagoons, either temporary or permanent, as well as deep lakes, formed. The palaeo-ecological analysis of the sediments from these lake-type systems shows that, in many of them, they have retained the ecological conditions of free water media up to the present day, in which herbaceous taxa (*Isoetes* cf. *velata*, *Ranunculus*) were dominant.

Cantabrian - Atlantic coast			
●	Cantabrian Coast: País Vasco	●	Atlantic coast - Costa da Morte
01	Irún (Edeso *et al.,* 1989)	12	Reiro I (Sáa, 1985)
02	Herriko-Barra (Altuna *et al.,* 1989)	12	Reiro II (Gómez-Orellana, 2002)
03	Urdaibai (Iriarte *et al.*, 2006)	12	Sorrizo (Gómez-Orellana, 2002)
●	Cantabrian coast: Cantabria	13	Traba (Bao *et al.,* 2007)
04	Liencres (Clark & Menéndez, 1975)	●	Atlantic coast: Rías Baixas
05	Jerra (Mary *et al.,* 1975)	14	Caamaño (Gómez-Orellana *et al.*, 2013)
05	Río Berdena (Mary *et al.,* 1975)	15	Corrubedo (Gómez-Orellana, unpublished)
●	Cantabrian coast: Asturias	16	Lagoa dos Nenos (Costas *et al.*, 2009)
06	La Franca I, II (Mary *et al.,* 1975)	●	Atlantic coast: Baixo Miño – N. Portugal
06	La Franca III (Gómez-Orellana, 2002	17	Mougás (Gómez-Orellana *et al.*, 1998)
●	Cantabrian coast: Galicia	18	Oia (Gómez-Orellana *et al.*, 2013)
07	Reinante (Gómez-Orellana *et al.*, 2013)	19	San Xián (Gómez-Orellana, 2002)
08	Area Longa I (Mary *et al.,* 1977)	19	Fedorento (Gómez-Orellana, 2002)
08	Area Longa II (Gómez-Orellana *et al.,* 2007)	20	Moledo (Gómez-Orellana, 2002)
●	Atlantic coast - Costa Artabra	21	Aguçadoura (Gómez-Orellana *et al.*, 2001)
09	Ponzos (Gómez-Orellana & Ramil, unpublished)	22	Cortegaça (Gómez-Orellana *et al.*, 2001)
10	Doniños (Santos *et al.*, 2001)		
11	Seselle (Santos *et al.*, 1993)		

Table 2a. Main pollen sequences obtained from peat sediments in the Iberian Atlantic region.

In other cases, after a phase short or long of existing as a lake-type medium, the development of biocenosis that grow in peaty environments, dominated by Poaceae, Cyperaceae, Juncaceae and *Sphagnum*, and the sedimentation of peat began. These biocenosis may correspond to herbaceous formations which, in general, grow from the sides of the depression towards the centre, meaning that part of the vegetation was floating on the body of water. On other occasions, these formations would float in the centre of the lagoon without touching the sides. The remains of the vegetation from these floating plants accumulated at the bottom of the lagoon and its development could have been that, after the passing of sufficient time, to completely fill it, in such a way that the depth of the permanent body of water dropped gradually until it became temporary, existing only in periods of heavy rain or after ice-melt.

Mountains and sub-coastal valleys			
•	Asturias	30	Schwejk (Ramil-Rego, 1992)
23	Llano de Roñances (Menéndez, 1950)	30	Sever (Ramil-Rego, 1992)
23	Buelna (Menéndez, 1950)	31	Río Boó (Gómez-Orellana, 2002)
23	Pendueles (Menéndez, 1950)	32	Gañidoira (Ramil-Rego, 1992)
23	Vidiago (Menéndez, 1950)	•	Galicia: Western mountains and valleys
23	Llano de la Mesa (Menéndez, 1950)	33	Mandeo (Méndez, 1969)
24	Monte Areo (López *et al.*, 2010)	34	Boedo (Bellot & Vieitez 1945)
25	Dueñas (López *et al.*, 2006)	35	Alcaián (Törnqvist & Joosten, 1989)
26	La Espina (López *et al.*, 2009)	36	Brins (Van Mourik, 1986)
27	La Bobia (Gómez-Orellana, unpublished)	37	Insua (Aira *et al.*, 1992)
•	Galicia: Northern mountains and valleys	37	Ameneiros (Aira *et al.*, 1992)
28	Moucide (Gómez-Orellana, 2002)	38	Sabuceda (Aira, 1986)
29	Montes do Buio (Menéndez *et al.*, 1961),	•	Galaico - Miñotos
29	Montes do Buio, Vilacampa (Van Mourik,1986)	39	Budiño I (Nonn, 1996)
30	Chan do Lamoso (Ramil-Rego, 1992)	39	Budiño III (Gómez-Orellana *et al.*, 1997)
30	Río das Furnas I (Ramil-Rego, 1992)	40	A Portela (Gómez-Orellana, unpublished)
30	Río das Furnas II (Ramil-Rego *et al.*, 1996)	40	Cruz de Pao (Gómez-Orellana, unpublished)
30	Chan da Cruz (Ramil-Rego, 1992)	41	Serra de Arga (Gómez-Orellana *et al.*, 2010)
30	Pena Vella (Ramil-Rego, 1992)	42	Châo da Cheira (Ramil-Rego *et al.*, 1996)
30	Pena Veira (Ramil-Rego, 1992)		
Inland sedimentary depressions			
•	Galicia: Terra Chá	•	Galicia: A Limia
43	Lagoa de Cospeito (Ramil-Rego, 1992)	46	Antela I (Van Mourik, 1986)
43	Alligal (Ramil-Rego, unpublished)	46	Antela II (Gómez-Orellana *et al.*, 1997)
43	Legua Dereita (Ramil-Rego, unpublished)		
•	Galicia: Terra de Lemos		
45	Touriz ((Van Mourik, 1986)		

In these cases, the biocenosis vegetation that covers the surface of wetlands when they lack free water are communities of therophytes; amphibious or hydrophiles. Examples of these dynamics are found in the *Lagoa de Lucenza* (76), *Laguna de la Lleguna* (70), *Pena Velosa* (67), *Pozo de Carballal* (75), *Turbera de San Isidro* (64) and *Laguna de la Roya* (70), among others.

Not all the depressions that witnessed the formation of lagoons have a glacial origin. In granite massifs hollows of varying sizes were formed, from one square meter to kilometres in extent, originating from processes such as weathering and tectonic activity, and which were also seen to be affected by cold processes. The shape and dimensions of these hollows are very variable, along with the depth of the detritic material, resulting from the erosion of the granite itself, which fills in the depression. These detritic layers aid the leaching of rainwater and slow down the hydrological dynamics. In some cases, there are well-defined drainage networks, so that any water that accumulates quickly passes through the rest of the basin, but in others, the lack of definition of these drainage networks helps water to accumulate on the surface, bringing about the creation of different types of wetlands in its heart.

At the end of the LGM, the abundance of water in the depressions on the granite massifs favoured the formation of Raised bogs. This process has been listed, although with different chronologies for the starting date, in places such as *Montes do Buio* (29), *Sierra del Xistral* (30), *Serra de Queixa* (77), *Peneda –Gerês – Xures* (80) and *Macizo de Sanabria* (70). In the *Serra de Gerês*, a large granite hollow modified by ice developed into a lagoon medium, the *Lagoa do Marinho* (80), after the most recent deglaciation. In these wetlands, the biocenosis of floating bogs developed, just like in the lagoons formed in glacial depressions. Here all the remains of vegetation fell to the bottom of the depression until it reached a depth of 290 cm. The chronology obtained for the accumulation de organic sediments process would have started at 12,845 cal. yr BP.

In the *Sierra del Xistral* (N of Lugo), in the *Pena Vella* zone (30), there are many depressions in the granite that host small lagoons and shallow ponds, with hydrophilous and hygroturfophilous vegetation on their edges, existing in some cases since over 10,000 years ago. This biocenosis is responsible for the accumulation of organic silts (gyttja) which mix with fine detritic material from the edges of the depressions. Pollen analysis of these layers confirm the existence, among the local vegetation of these wetlands, of free water species, together with others typical in peaty and hydrophile media (*Sphagnum*, Cyperaceae, Juncaceae, Poaceae, *Drosera*, *Pinguicola*). Coinciding with the end of the Younger Dryas, the free water vegetation of these media was replaced by another but with a peaty character, bringing a change in the sedimentology of the wetlands, and the start of peat sedimentation. The development of the *Pena Vella* bog continued in later periods, reaching its maximum size and depth in the mid-Holocene, coinciding with the Atlantic period.

A third case of bog formation after the LGM is linked geographically to the summit zones of mountain *massifs* not made of limestone. Here, the retreat of the ice left large rocky surfaces exposed along with piles of stony materials, whose topography was not conducive to the retention of large volumes of water, hence, not at all favourable to the formation of lakes and lagoons. In some massifs that receive heavy rainfall, because of their location regarding the entry of cyclones, the land surfaces

Inland Cantabrian-Atlantic mountains and transition areas with the Iberian plateau			
•	Basque mountains	70	Lago de Sanabria (Watts, 1986)
47	Belate (Peñalba, 1989)	70	Laguna de La Roya I(Allen *et al.,* 1996)
48	Saldropo (Peñalba, 1989)	70	Laguna de La Roya II(Muñoz *et al.,* 2013)
49	Zalama (Pérez-Díaz *et al.*, 2016)	71	Arroyo de las Ciervas (Muñoz, 2001)
•	Eastern Cantabrian Mountain Range	71	El Portillo (Muñoz, 2001)
50	Los Tornos I (Peñalba, 1989)	•	Western Sierras Galicia
50	Los Tornos II (Muñoz, 2001)	72	San Ciprian (Muñoz, 2001)
51	Sotombo (Pérez-Díaz *et al.*, 2016)	73	Subarbol (Muñoz, 2001)
52	Estacas de Trueba (Mariscal, 1995)	73	Porto Ancares (Muñoz, 2001)
53	Escudo (Muñoz, 2001)	73	Cespedosa Ancares (Muñoz, 2001)
54	T. de Alsa (Mariscal, 1993)	74	Brañas de Lamela (Muñoz, 2001)
54	Pico Ano (Salas, 1993)	75	Pozo del Carballal (Muñoz, 2001)
55	Santa Gadea (Muñoz *et al.*, 2003)	76	Lagoa de Lucenza (Muñoz, 2001)
55	San Mames de Abar (Iriarte *et al.,* 2001)	•	Sierra de Queixa
55	La Piedra (Muñoz, 2001)	77	Queixa-I (Menéndez, 1971)
56	Valle de la Nava (Menendez, 1968)	77	Queixa-II (Menéndez, 1971)
57	Cueto de Avellanosa (Mariscal, 1983)	77	Manzaneda (Aira, 1986)
58	Pico Sertal (Mariscal, 1986)	77	Xuncos (Aira *et al.,* 1986)
•	Cantabrian Central Mountain Range	77	As Lamas (Maldonado, 1994)
59	Río Frío (Menéndez *et al.*, 1963)	77	Prada (Maldonado, 1994)
60	Lago Enol (Moreno *et al.*, 2011)	•	Sierras Galaico - Minhotas
60	Lago Ercina (Moreno *et al.,* 2011)	78	Monte de Vieiro (Alvarez, 1993)
60	Vega de Comeya (Ruíz *et al.,* 2001)	79	Souto Fiscal (Ruiz *et al.,* 1995a,b)
61	La Tarna (Ruíz, 2000)	79	Lorderlo (Ruiz *et al.,* 1995a,b)
62	Pinar de Lillo (García *et al.,* 1997)	79	Lamas de Movio (Ruiz *et al.,* 1995a,b)
62	Lillo II (Muñoz, 2001)	79	Branda de Gorbelas (Ruiz *et al.,* 1995a,b)
63	Corteguero (Ruíz, 2000)	79	Lamas de Vez (Coudé, 1981)
64	San Isidro (Fombella, 2001)	80	Lagoa de Marinho I (Coudé, 1981)
65	Lago de Ajo (Allen *et al.*, 1996)	80	Lagoa de Marinho II (Ramil-Rego *et al.,* 1996)
65	Laguna de la Mata (Belet *et al.,* 1994)	80	Lama de Porto Châo (Fonticova, unpublished)
•	Western Cantabrian Mountain Range	80	Lama do Pastor (Fonticova, unpublished)
66	Puerto de Leitariegos (García *et al.*, 2004)	80	Turbera de Leonte (Bellot, 1950)
67	Pena Velosa (Muñoz *et al.*, 2012)	80	Pé de Cabril (Bellot, 1950
68	Brañuelas (Muñoz, 2001)	80	Borrageira (Bellot, 1950)
•	Sanabria – Cabrera - Culebra	80	Carris (Bellot, 1950)
69	La Baña (Janssen, 1996)	81	Lama de Mira (Ruiz *et al.,* 1997)
70	LLeguna (Muñoz *et al.*, 2004)	81	Encostra do Trobaô (Ruiz *et al.,* 1997)
70	Sanguijuelas (Muñoz *et al.*, 2004)		

Table 2b. Main pollen sequences obtained from peat sediments in the Iberian Atlantic region.

have had throughout the Holocene, a high level of humidity during the whole year, although the water only manages to accumulate in small seasonal pools, square metres in size, made up of fragments of stone that cover the summits. In these environments, the small ponds served as the pioneering media for the colonization of lagoon type plants (*Glyceria*, *Ranunculus*), whilst the permanently damp surfaces were colonized progressively by species typical in waterlogged zones with peat, or plants that grow in peaty environments, spreading depending on the water availability on the summit flats and descending from these via the windward mountainside towards lower altitudes, creating a special type of bog called blanket bog.

In the *Sierra del Xistral* (30), the oldest sediments in the Summit bogs correspond to the wetlands of *Tremoal do Chao do Lamoso*, situated at an altitude 1,039 m. In this deposit upon a layer of quartzite materials of pre-glacial origin, at the end of the Tardi-glacial Interstadial, a system of wetlands containing both plants that grow in peaty environments and small systems of lagoons appeared. At the beginning of the Holocene, the wetlands changed into a summit bog, which progressively spread through the flat areas and mountainsides (blanket bog). The maximum depth of the peat sediments reaches 420 cm. The dating of the base corresponds to 9,833 cal. yr BP, whilst the upper part reflects the latest changes in the nearby countryside, marked by the expansion of forest plantations of different exotic species.

Similarly, the formation of blanket bogs throughout the Holocene took place in other mountainous areas in the Northern mountain chain of Galicia and down to levels quite a bit lower than in the sector where the *Chao do Lamoso* is situated. So, in the bogs at *Tremoal das Furnas II* (30), the start of the peat sedimentation has been dated to over 8,255 cal. yr BP, obtaining a depth of 155 cm; in the *Montes do Buio* this is around 8,613 cal. yr BP (29), whilst at the *Vilacampa* deposit (29) it is 8,504 cal. yr BP. Referring to the *Tremoal da Charca do Chan da Cruz* bogs (30), the start of peat sedimentation was more recent, around 6,706 cal. yr BP.

The existence of Blanket bogs at the start of the Holocene was also recorded in other mountainous areas of the Atlantic region of the Iberian Peninsula. Specifically, at the *Monte Zalama* bog (49), whose start would have begun later than the date for *Chan do Lamoso*, put at 7,973 cal. yr BP. There is also evidence that this type of bog activity also occurred at the *Puerto del Escudo* (53) dating of the peat layer at 8,573 cal. yr BP, and at the *Puerto de los Tornos* bog (50), whose start of organic sedimentation is calculated to be 8,596 cal. yr BP.

The period of maximum Atlantic bog cover

Climatic reconstructions for the Holocene (Davis *et al.,* 2003; Mauri *et al.,* 2015; Peyron *et al.,* 1998; Prentice *et al.*, 1998; Rensen *et al.,* 2012) state that there has been a significant increase in rainfall on the Cantabrian-Atlantic face of the Iberian Peninsula coinciding with the thermal maximum of the Holocene. These climatic conditions favoured the development of peat deposits that began in previous periods, and favoured the formation of new bogs in different territories.

Coinciding with this stage, the Blanket bogs reached their maximum spread. In the *Serra del Xistral*, this expansion meant that the area covered by the Blanket bog ecosystems went from the summit areas (1,050-900 m) until they reached other existing bogs in the large systems of hollows and on the valley floors (750-600 m). Similar dynamics, but with an unequal spread around the zone, has been documented in other mountainous areas of the Atlantic region. This is the case of the sub-coastal *sierras* in the west of Asturias, the bog *de la Bobia* (27) shows the start of a peat sequence around 6,085 cal. yr BP, which ends after 1,480 cal. yr BP. On the other hand, in the Eastern zone of Asturias residual witnesses can also be found of what was once a wide-ranging system of Holocene blanket bogs in the *Cordal del Cuera* (23). During this same period, the maximum size of the summit bog complexes was reached, comparable to a large extent to the Blanket bog, in the South of Cantabria, as confirmed by the ages of formation obtained at places like *Puerto de los Tornos* (50), *Solombo* (51), *Estaca de Trueba* (52), *Puerto del Escudo* (53), *Bog de Alsa* (54), *Pico Ano* (54) and *Cueto de Avellanosa* (57).

In addition to the blanket bogs, throughout the first half of the Holocene the formation of innumerable complexes of raised bogs and mires and fens took place at other mountainous zones, which also reached their maximum size related to the Thermal Maximum: *Brañuelas* (68), *A Fonsagrada* (72), *Ancares* (73), *Pedrafita* (74), *Queixa* (77), *Saldropo* (48), *Belate* (47).

Other environments, unrelated to areas of siliceous mountains but with territories that receive heavy rains and hidden-precipitation, were also the setting for the appearance of bog ecosystems throughout the first third of the Holocene. Likewise, it has been shown that in this period peat sediments were formed in areas that received less rain, but which received significant supplies of water from river courses or from surface run-off. The example most representative of these media is the *Lagoa de Antela*, (46), a large wetland lake area which in the first half of the 20th century occupied a surface of approximately 7,000 ha, of which 3,700 ha were covered by water, with a maximum depth of 3 m. Surrounding the sheet of water were different types of amphibia and hydrophile biocenosis, the most common of these was a large area made up of reeds (*Scirpus lacustris*, *Phragmites communis*) and bulrushes (*Sparganium, Iris*). The communities dominated by these species formed brown peat (Guitián & Carballas, 1982) which is considered a characteristic of a fen type bog. The age obtained for the start of the peat layers is put at 7,493 cal. yr BP.

Likewise, similar media were formed around the large systems of coastal lagoons and marshes where large areas of reeds such as (*P. communis*, *Cladium mariscus*, *Scirpus maritimus*), rushes (*Juncus maritimus*), bulrush (*Sparganium*, *Iris*), grew, which gave way to hydrophilous zones with scrub (*Erica ciliaris*, *E. tetralix*, *E. Erigena*). These in turn gave way to complex swamp woods and alluvial forests (*Alnus*, *Salix*, *Fraxinus*). Examples of these wetlands with sedimentation of peat have been found in *Irún* (1), *Herriko-Barra* (2), *Urdaibai* (3), *Liencres* (4), *La Jerra* (5), *Río Berdena* (5), *Reinante* (7), *Ponzos* (9), *Doniños* (10), *Seselle* (11), *Reiro* (12), *Traba* (13), *Corrubedo* (15), *Lagoa dos Nenos* (16), *Aguçadoira* (21) and *Cortegaça* (22).

Coinciding with the period of largest area of bog ecosystems, some recorded significant changes in their biocenosis, marked by the invasion of arboreal vegetation. This process was documented in some of them as they contain arboreal remains in the organic sediments, generally *Betula*, *Salix*, and *Alnus*. In the *A Gañidoria* bog (32), the invasion of birch was dated at 7,744 cal. yr BP (Ramil-Rego, 1992), after which new layers of peat were deposited. Among the remains are fragments of branches, of differing lengths and girths, some over 70 cm long and 10 cm wide. Other remains correspond to roots, some of which kept their primary position. In the *Ponzos* (9) and *Reinante* bogs (7) the trees that colonized the old bog were oaks (*Quercus*), willows (*Salix*) and alders (*Alnus*), where different woody remains have been recovered, from large fragments to trunks and branches, some of which are over 3 m long.

The reduction in the size of bogs

In the last third of the Holocene the first villages appear and the natural countryside was rapidly transformed into a rural one. The need to have land for cultivation and cattle pasture brought significant alterations in the vegetation cover, particularly with respect to trees. The repeated use of fire, grazing, the cutting down of plants along with their uprooting, had an accumulative effect on the appearance of the countryside and on the biodiversity of the different territories.

During this period, many pollen sequences demonstrate that the effect of human activities tends to mask the climatic changes that took place and their possible effects on the type of vegetal cover. The most marked, in the territories in the Iberian-Atlantic region, was related to the occurrence of successive thermal oscillations, whose maximum temperatures were separated by cold periods of short duration when compared to those recorded during the Pleistocene, as well as a drop in rainfall, if compared to those recorded at the Holocene Thermal Maximum. There is a lack of data to evaluate the incidence of these processes on the first rural societies, but, on the contrary, there is a large amount of documentation from the Middle Ages and in the *Ancién Regime* concerning their catastrophic effects on harvests and links to events such as famine, epidemics, and death rates of the human population.

An analysis of the sediments, the pollen and the chronological data allow us to establish that, after 3,0-2,5 kyr BP, the peat biocenosis of many bog ecosystems in the Atlantic Biogeographic region were replaced by mesophilic communities of vegetation (eurihydric), typically by scrub or sub-humid/ dry pastures. Moreover, in areas where there was a reduction and/or a greater seasonality to the rainfall, but which retained large supplies of hidden precipitation, above all in the spring or summer months, the bog biocenosis was substituted by Wet heaths (*Erica ciliarias*, *E. mackayana*, *E. tetralix*) or mosaics of heather patches with hydrophile pastures (*Nardus*, *Juncus*, *Molinia*). In these cases, the wet heaths were able to occupy most of the wetlands, with the bog biocenosis relegated to residual zones. The location of these media around the territory, although being determined, to a large extent, by natural factors (climate, altitude, type of geological material, etc), may have been modified by human activity, especially via burning, grazing, construction of small drainage networks, etc.

Human settlements were, above all, located in areas of low altitude during the first stages of ruralization of the territory. After the end of the Metal Ages, these areas suffered an important transformation through the widespread and repetitive use of fire, the elimination of native vegetation through grazing and the manual uprooting of plants. In the first agricultural societies significant areas of wetlands were preserved, not surprising given the difficulties and efforts that were necessary for their transformation, although they were frequently subjected to temporary use, for grazing and water sources, as well as for timber and brushwood. This situation was modified after the 18th century, when new agrarian movements emphasised the need to improve and transform primitive terrain, especially wetlands (Cónsul & Tineo, 1786). These proposals acquired a greater influence at the beginning of the 20th century, when the principal norms that affect land (legislation of mountain land, legislation of waters, agrarian legislation), promoted a reshaping of land that had been idle for many years. A policy that was maintained and extended during the Franco dictatorship and which, currently in the middle of the "sustainable development" era, underlies, camouflaged, many of the activities that are carried out in the territory.

The signing of the Maastricht Treaty (1993), put the environment into the official policy of the European Union, underpinned by the principles of caution, prevention, correction of the pollution at source and of «who pollutes pays». In agreement with this new orientation, the EU Directive 92/43/CEE, turned its nature conservation policy towards the creation of a network of protected zones, Nature 2000, based on the protection of habitats and of the species of community interest using ecological, measurable and assessable criteria. The approval of Habitats Directive 92/43/CEE and the naming of the first proposals for inclusion in the Natura 2000 in the Atlantic Biogeographic region of the Iberian Peninsula (DOCE, 2004), were not however, able to mitigate the loss of biodiversity and so at the moment the obligations put forward in the Directive itself are not being followed, neither are the international commitments inherent in the Convention on Biological Diversity. In the last 25 years, the territories of the Iberian-Atlantic region have suffered a dramatic loss of biodiversity, which has had a very significant effect with respect to the different types of wetlands linked to the groups of Wet Heaths and bogs.

Among the causes of this deterioration that must be mentioned are the execution of forest repopulation projects on Wet heaths and bog habitats, especially those that use species alien to these media, most of them considered as non-native or invading in different territories (*Pinus pinaster*, *P. sylvestris*, *P. radiata*, *Eucalyptus globulus*, *E. nitens*). A second factor affecting the loss of biodiversity in bog ecosystems, is connected to the process of land consolidation and improvements in the exploitation of agricultural land or of cattle, by the opening of tracks, digging of drainage systems, white-washing, application of manure or fertilizers, sowing of seeds of species used for grazing, planting of forest species (pines, eucalyptus, birches, etc), etc.

There is also exploitation through mining, one of whose branches has orientated itself in the last few years to extracting substrates for gardening, this has left a big footprint in the bogs in the Atlantic region. The *Saldropo* bog, originated 5,600 years ago, was considered as the reference peat ecosystem (raised bog) in The Basque Country. In spite of this, in the two decades of the 1970s and 1980s it was exploited until it completely disappeared. Even so, exploitation through the mining of bogs has continued after the introduction of Directive 92/43/CEE, including some of the mines within Nature 2000 network. The statistical data published by the IGME (Marchán & González, 2014) indicate that between 2008 and 2012 the exploitation of peat reached 102,130 tonnes and it was concentrated in the Atlantic area, in three provinces: Lugo, Cantabria and Burgos.

At this moment in time, infrastructure (tracks, roads) and drainage pipes, continue to generate a significant level of destruction and alteration of bogs and Wetland heath bogs, although less than that caused by the establishment of wind farms. One of the most dramatic and destructive examples of the loss of biodiversity for the installation of "green energies" in mountain bog ecosystems occurred in the north of Galicia. An area that boasts several Special Area of Conservation (SAC) in Nature 2000 network, where over 600 wind-turbines were installed between 1995-2016, at the same time hundreds of kilometres of tracks were opened, along with the appearance of numerous channels both underground and on superficial. The

PROVINCE	2008	2009	2010	2011	2012	TOTAL
Lugo	6,379	6,187	5,112	3,225	2,464	**23,367**
Cantabria	3,500	4,000	2,500	3,500	3,000	**16,500**
Burgos	10,580	10,580	7,493	6,444	7,718	**42,815**
Valladolid	105	714	157	74	50	**1,100**
Tarragona	3,092	1,507	-	546	142	**5,287**
Castellón	19,379	33,300	46,600	46,600	35,340	**181,219**
Valencia	-	-	1,600	1,182	3,000	**5,782**
Granada	38,190	860	1,500	25,000	9,665	**75,215**
TOTAL	**81,225**	**58,678**	**64,962**	**86,571**	**61,379**	**351,285**

Table 3. Peat exploitation in Spain. Values in tonnes per year. Source: Estadística Minera de España (2014).

environmental impact evaluations and environmental reports under-estimated the importance of the bogs, as well as the repercussions coming from the building and running of the wind farms. A significant part of these mountains occupied by blanket bogs, raised bogs and wet heath bogs was destroyed. The construction of tracks fragmented the original surfaces of the wetlands, altering their surface hydrology, exposing the detritic material on the surface that was normally covered by deposits of peat. The tracks and their lateral drainage systems collect and channel a large quantity of water in rainy periods and these have had a large impact on the mountainsides, covered by heather and bogs, causing serious erosion and similar phenomena in places where the hygroturfophilous vegetation has impeded it for thousands of years. In consequence, the sub-surface levels of the bogs show the sedimentation of layers of detritic material (dense silts, sands, gravel and even stones) originating from the clearing of mountainsides and the construction of tracks which afterwards have been swept away by rainwater. Materials that in the future will serve, as a "guiding fossil", to date the exact moment when the wind farms were built and their impact on the bogs.

References

Clark, P.U., Dyke, A.S., Shakun, J.D., Carlson, A.E., Clark, J., Wohlfarth, B., Mitrovica, J.X., Hostetler, S.W., McCabe, A.M. 2009. The Last Glacial Maximum. *Science* 325, Issue 5941, pp. 710-714.

Cónsul J. y Tineo, F. 1786. *Memoria sobre el conocimiento de las tierras; Verdadero i Económico Método de cultivarlas adaptado al clima, i circunstancias de Galicia, i Asturias, Santiago*: Imprenta Aguayo.

Davies, C.E., Moss, D., O Hill, M. 2004. *Eunis Hábitat Classification. Report to the European Topic Centre on Nature Protection and Biodiversity*, European Environment Agency.

Davis, B.A.S., Brewer, S., Stevenson, A.C., Guiot, J. 2003. The temperature of Europe during the Holocene reconstructed from pollen data. *Quaternary Science Reviews*. 22: 1701-1716.

De Beaulieu, J.L., Reille, M. 1992. The last climatic cycle at la Grande Pile (Vosges,France): A new pollen profile. *Quaternary Science Reviews* 11: 431–438.

Escudero, A., Olano, J.M., García, R.; Bariego, P., Molina, C., Arranz, J.A., Molina, J.I., Ezquerra, F.J. 2008. *Guía básica para la interpretación de los hábitats de interés comunitario en Castilla y León*. Junta de Castilla y León. Consejería de Medio Ambiente. Valladolid.

EUR 13231. 1991. CORINE biotopes. *The design, compilation and use of an inventory of sites of major importance for nature conservation in the European Community*. Office for Official Publications of the European Communities. Luxembourg.

European Commission. 2013. *Interpretation Manual of European Union Habitats - EUR 28*. DG Environment - Nature and Biodiversity.

Florschütz, F., Menéndez, J., Wijmstra, T.A. 1971. Palynology of a thick Quaternary succession southern Spain. *Palaeogeography, Palaeoclimatology, Palaeoecology* 10: 233-264.

Guitian, F., Carballas, T. 1982. *Suelos naturales de la provincia de Orense*. Consejo Superior de Investigaciones Científicas (CSIC). Santiago de Compostela.

Gómez-Orellana, L.; Ramil-Rego, P., Muñoz, C. 2007. The Würm in NW Iberia, A pollen record from Area Longa (Galicia). *Quaternary Research* 67 (3): 438-452.

IGME 1974. *Investigación de lignitos en Meirama. (La Corufia)* Colección-Informe. Madrid: IGME.

IGME 1979. Aspectos geológicos de la cuenca Terciaria de Puentes de García Rodríguez. (La Coruña). *Boletín Geológico y Minero* 95: 451-461.

Ihobe 2008. *Bases generales para la elaboración del inventario y la caracterización de los hábitats hidroturbosos (turberas, medios paraturbosos y afines) en la CAPV*. Gobierno Vasco Bilbao.

Ihobe 2011. *Primera evaluación del estado de conservación de los hábitats hidroturbosos de interés comunitario en el País Vasco*. Gobierno Vasco. Bilbao.

Marchán, C., Regueiro, M. 2014. *Panorama minero 2014*. Instituto Geológico y Minero de España (IGME). Ministerio de Economía y Competitividad. Madrid.

Mauri, A., Davis, B.A.S., Collins, P.M., Kaplan, J.O. 2015. The climate of Europe during the Holocene: a gridded pollen-based reconstruction and its multi-proxy evaluation. *Quaternary Science Reviews* 112: 109-127.

Médus, J., Nonn, H. 1963. Prerniers résultats d'analyses polliniques à Puentes de García Rodríguez (province de Galice. Espagne) et conclusions géomorphologiques qui en découlent. *Comptes Rendus de l'Accadémie des Sciences* 256: 1570-1572.

Médus, J. 1965. L'evolution biostratigraphique dune lagune néogéne de Galicie (Espagne). *Pollen et Spores* 8: 381-393.

Menéndez, J. 1975. Análisis paleobotánico de algunas muestras de lignitos proccedentes de Puentes dc García Rodríguez (La Coruña). *Boletín de la Real Sociedad Española de Historia Natural (Sección Geológica)* 73: 121-124.

Moore, P.D. 1984. *European mires*. Academic Press. London.

Peña, A. 1993. Estudio paleontológico en las cuencas terciarias de Galicia. *Revista de la Sociedad Geológica de España* 6: 19-28.

Peralta, J. Biurrun, I., García-Mijangos, I., Remón, J.L. Olano, J.M., Lorda, M. Loidi, J., Campos, J.A. 2013. *Manual de Hábitats de Navarra*. GANASA y Gobierno de Navarra, Departamento de Desarrollo Rural, Medio Ambiente y Administración Local.

Peyron, O., Guiot, J., Cheddadi, R., Tarasov, P., Reille, M., de Beaulieu, J.L., Bottema, S., Andrieu, V. 1998. Climatic reconstruction in Europe for 18,000 yr B.P. from pollen data. *Quaternary Research* 49: 183-196.

Prentice, I.C., Harrison, S.P., Jolly, D., Guiot, J. 1998. The climate and biomes of Europe at 6000 yr BP: comparison of model simulations and pollen-based reconstructions. *Quaternary Science Reviews* 17: 659-668.

Ramil-Rego, P., Rodríguez, M.A., Hinojo, B.A., Rodríguez, P.M., Ferreiro da Costa, J., Rubinos, M., Gómez-Orellana, L., de Nóvoa, B., Díaz, R.A., Martínez, S., Cillero, C. 2008. *Os Hábitats de Interese Comunitario en Galicia. 2 Vol.* IBADER. Universidade de Santiago de Compostela. Lugo.

Renssen, H., Seppä, A., Crosta, X., Goosse, H., Roche, D.M. 2012. Global characterization of the Holocene Thermal Maximum. *Quaternary Science Reviews* 48: 7-19.

Rodwell J.S. 1991. *British Plant Communities. Vol. 2. Mires and heaths*. Cambrigde Univ. Press.

Taylor, J.A. 1983. The peatlands of Great Britain and Ireland. In: Gore A.P.J. (ed.), *Ecosystems of the World 4B. Mires: Swamp, Bog, Fen and Moor. Regional Studies,* pp. 1-46. Elsevier Science, Amsterdam.

Woillard, G.M. 1978. Grande Pile peat bog: A continuous pollen record for the last 140,000 years. *Quaternary Research* 9 (1): 1-21.

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Origin and evolution of the blanket bog of Zalama (Montes de Ordunte, Northern Iberian Peninsula)

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Abstract

The Zalama blanket bog, because of its location and typology, is an exceptional eco-system in bio-geographical and paleo-environmental terms. This study analyzes the Holocene evolution of the Zalama bog beginning with the behaviour of some of its geo-chemical properties and a pollen analysis, as well as a paleo-environmental reading of the principal changes in these properties. The Zalama bog began its formation 8,000 years ago on periglacial materials that generated a generalized paludification of the surface allowing a rapid and sudden transition from minerotrophic conditions to others more ombrotrophic that were consolidated in 7,750 cal. years BP. During this time, the growth rates of the bog oscillated between values of 0.07 and 0.02 cm year^{-1} coupled with the main climatic changes, which demonstrates the suitability of this bog for paleo-environmental reconstruction studies.

Keywords: genesis of bogs; Holocene; Geo-chemistry; climatic coupling.

Introduction

Bogs are ecosystems of important environmental value (Pontevedra-Pombal *et al.*, 2001), in spite of not being abundant, only occupying 3% of the Earth´s surface (Gorham, 1991). Their distribution is assymetrical and their presence decreases as one moves from northern to southern latitudes. On the Iberian Peninsula they are particularly scarce (Heras *et al.*, 2016), blanket bogs are especially infrequent, and until very recently were considered as absent (Evans, 2006). However, several nuclei of blanket bogs have been described on the Iberian Peninsula, the Southern limit of distribution of this type of bog. One of these is the Zalama blanket bog, which is located in a sector of high bio-geographical interest spanning both the Atlantic and Mediterranean regions.

Although this class of bog is especially valuable for retrospective studies into the evolution of the climate and its paleo-ecological reconstruction (De Jong *et al.*, 2010), owing to the fact that their genesis makes them very sensitive to environmental changes, their relative scarcity has meant that they have received little attention. Several paleo-environmental studies were made in the region, concentrating especially on the minerotrophic bogs (Menéndez-Amor, 1968; Mariscal, 1993; Muñoz-Sobrino, 2001) and very occasionally the ombrotrophic bogs (García-Antón *et al.*, 1989; Peñalba, 1994; Pérez-Obiol *et al.*, 2016).

In any case, the study of these bogs focused on their use as a paleo-botanic archive (Pérez-Díaz *et al.*, 2016) and, to a lesser extent, on the analysis of their current vegetation (Heras, 2002). In no previous study has the evolution of the bog been studied.

Therefore, the bio-geochemical nature, the bio-geographical distribution and the low number of paleo-environmental studies in the area, increase enormously the relevance of the Zalama blanket bog. In this study, we present a high-resolution model of the genesis and evolution of the Zalama bog over the last 8,000 years, through the analysis of its geo-chemical and pollinic properties.

Area of study

The Zalama bog (43º08'04'' N - 3º24'35'' W) at an altitude of 1,330 m (Fig. 1), is located on the summits of the Montes de Ordunte, in the South-east of the Basque-Cantabrian mountains (North of the Iberian Peninsula). It covers the summits and mountainsides of the main mountain chain that is the water shed between the Cantabrian and the Mediterranean basins. The bog develops directly on the periglacial material of fragments of strongly superimposed quartzitic sandstone (Fig. 1) that have favoured the paludification of the edaphic system. The superficial part of the bog has been significantly altered by processes of compaction and erosion (Fig. 1) caused, very probably, by the systematic burning to create grazing for cattle (Heras & Infante, 2005).

The area falls in the Atlantic bio-geographical region, has an annual average temperature of 7.5 ºC and a total annual precipitation of over 1,600 mm (Heras, 2002). The occurrence of fog associated with the barrier effect, when masses of oceanic air from the Atlantic pass over the Cantabrian mountains, assures a high level of environmental humidity during the whole year and especially in the hottest months.

The current vegetation (Heras, 2002), is made up of a central part dominated by common heather, *Calluna vulgaris* (L.) Hull and *Erica tetralix* L., with *Eriophorum vaginatum* L., *Molinia caerulea* (L.) Moench., *Daboecia cantábrica* (Huds.) K. Koch., *Erica cinerea* L., *Potentilla erecta* (L.) Rauschel, *Galium saxatile* L., *Vaccinium myrtillus* L., *Juncus effusus* L. and the mosses *Sphagnum carillifolium* (Ehrh.) Hedw., *Hypnum cupressiforme* Hedw., *Dicranum scoparium* Hedw. and *Campylopu*s *flexuosus* (Hedw.) Brid.; and a wetter outer zone, of minerotrophic character dominated by *E. tetralix*, *C. vulgaris*, *D. cantabrica*, with *Juncus squarrosus* L., *E. vaginatum*, *E. angustifolium* Honck, *Scirpus cespitosus*, *M, caerulea*, *J. effusus* and the moss-

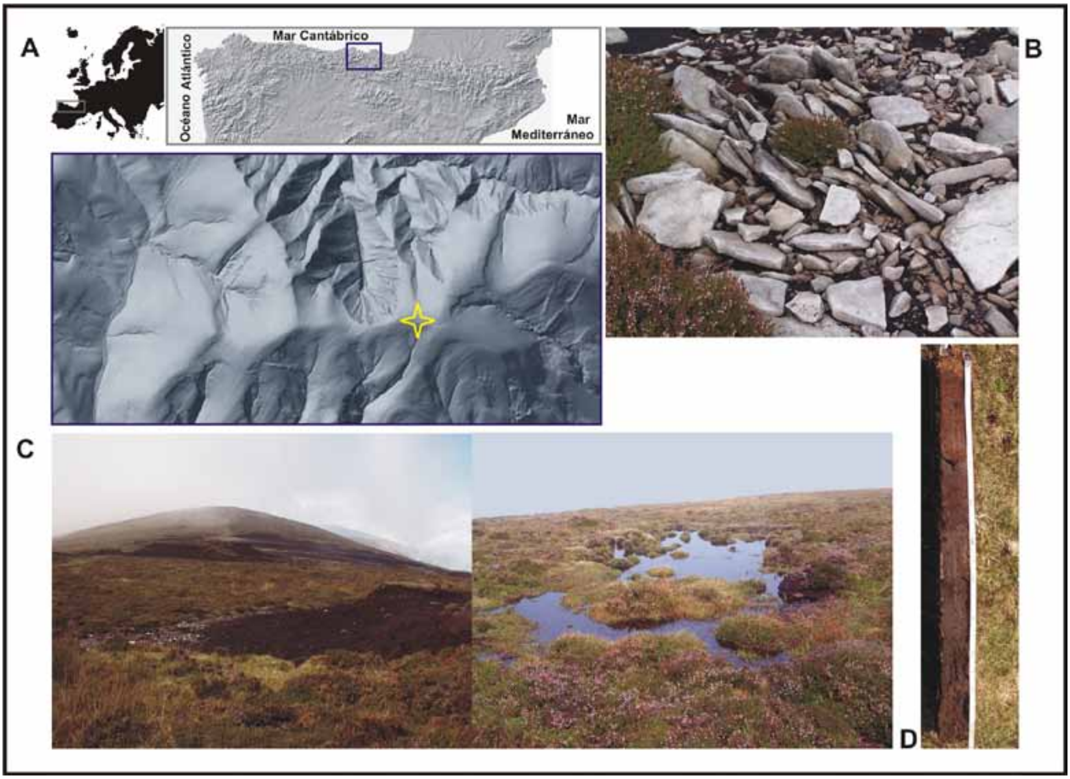

Figure 1. Zalama bog A) General geographical context and location (yellow star); B) View of the periglacial quartzite deposits; C) Form taken by the bog and superficial features; D) Stratigraphical profile of the first metre of peat.

es, *Sphagnum cuspidatum* Hoffm., *Sphagnum tenellum* (Brid.) Brid., *Pleurozium schreberi* (Willd ex Brid.) Mitt., *Aulacomnium palustre* (Hedw.) Schwaegr., *Dicranum bonjeanii* De Not. and *Polytrichum commune* Hedw.

Material and methods

Collection of samples

In the least altered sector of the bog, isolated from the main flow of induced erosion, two complete profiles down to the bedrock were taken, a total of 232 cm, from which 130 samples were obtained. The first 100 cm (Fig. 1) were extracted with a Wardenaar corer (Wardenaar, 1987) and the rest with a Russian peat borer. In the laboratory, the superficial 30 cm were sectioned into samples 1 cm thick, and the rest of the profile was cut every 2 cm. Two samples that represent the transition interphase, between the vegetation and the peat, were also taken, with each sample being divided into 4 sub-samples. One was frozen to below -10ºC, the second was maintained cool and in the dark at 4ºC, a third was freeze-dried and the fourth dried at 105ºC. The freeze-dried and the dry samples were finely ground, homogenized and stored in a dry dark place.

Chronology

To establish the development patterns of the bog, 17 samples of the peat were radiocarbon (^{14}C AMS) dated in the Ångström Laboratory, Div. of Ion Physics, ^{14}C-lab (Uppsala - Finland) and at the Centro Nacional de Aceleradores de Sevilla (Spain). The data was calibrated with the CALIB 7.1 program using the IntCal13 curve (Reimer *et al.*, 2013). The model age/depth was established by applying the software CLAM 2.2 (Blaauw, 2010). The details of the model of the age of the Zalama bog can be consulted in Pérez-Díaz *et al.* (2016). All the ages are expressed in years, callibrated before the present (cal. yr BP).

Physico-Chemical analysis

To determine the characteristics related to the genesis of the Zalama bog and its trophic evolution, several of its properties were analyzed. The bulk density (BD) was determined by following the method described by Lynn *et al.* (1974). The inorganic ash content of the peat was obtained by incineration at 600 ºC for 8 hours and was expressed as a percentage of dry weight at 105 ºC. The evolution of the organic material was established for all the peat samples from the determination of the Von Post scale (Von Post, 1937) and from the pyrophosphate index, IP (Lynn *et al.*, 1974).

The molar ratio Ca:Mg, an index of atmospheric and edaphic flows, was determined from the total content of both elements. Details concerning the methodology employed can be consulted in Pérez-Díaz *et al.* (2016).

Total carbon (TC) was measured with a Leco CHN 1000 auto-analyzer, with a dried ground sample, in complete combustion at 1,000 ºC and detection of the infrared band of the CO_2 liberated. The absolute error was under 0.3 %. The equipment was callibrated and verified using different reference materials (EDTA-502-092, SOIL-502-308, COAL-501-002). The low inorganic ash content, the pH of the peat - acid to very acid - and the lithology of the area ruled out the presence of carbonates, and so the value of the TC is comparable to the organic carbon content.

Growth rate of the peat

The vertical growth rate (VGR) of the bog in cm per year (cm $year^{-1}$) was calculated from the age of a sample and from the accumulated depth that said sample represented. The apparent total vertical growth rate or average vertical growth was calculated from the age of the sample from the base and the total depth of the bog, eliminating the data corresponding to the active oxic layer or acrotelm in the calculation (Turunen *et al.*, 2004).

Pollinic analysis

The chemical treatment of the 131 samples used for pollinic analysis, were made using the method proposed by Girard & Renault-Miskovsky (1969), with the samples concentrated in Thoulet liquor (Goeury & de Beaulieu 1979). The identification was carried out in agreement with the criteria of Fægri & Iversen (1989), Moore *et al.* (1991) and Reille (1999). The non-pollinic microfossils were identified in agreement with van Geel (1978, 2001, 2006); van Geel *et al.* (1981, 1989, 2003) and Cugny (2011). A minimum number of 500 pollens in each sample were identified, excluding Cyperaceae, aquatics and spores.

Results and discussion

Chronology, pedogenic and trophic development

From the age–depth model established for the Zalama bog (Pérez-Díaz *et al.*, 2016) it was possible to identify the most significant changes in the characteristics of the bog within a chronological framework that covers the last 8,000 years.

In the profile of the Zalama bog it is possible to differentiate significant stratigraphical changes. The uppermost 32 cm are of brown/red peat, 'fíbric' type, with a low level of decomposition and Von Post values from H1 to H4. This layer of peat is strongly impacted and shows an intense process of compaction. The following 54 cm are of a dark-brown greyish peat (H5-7) with "sapric" features. Fragments of carbon can be distinguished at the bottom of this layer. Between a depth of 86 and 112 cm the peat is less decomposed (H4) giving rise to Brown 'fíbric-hemic' peats. A new phase with a higher level of degradation of the peat can be identified between 112 and 120 cm, with dark-brown peat (H7) 'sapric' type, and which contains abundant fragments of wood at its base. The next 40 cm are of a very fibrous reddish peat (H2) rich in *Sphagnum* remains. Between this stratographic level and the remaining 72 cm there is a very intense change to very decomposed black peats (H8-9) with 'sapric' features and an increasing content of very fine quartzitic sand.

The peat density (Fig. 2) varied between 0.12 and 0.34 g cm^{-3} with higher average values in the acrotelm than in the catotelm (Table 1). This distribution can also be observed in the ash content with average values of 7.8 % in the acrotelm

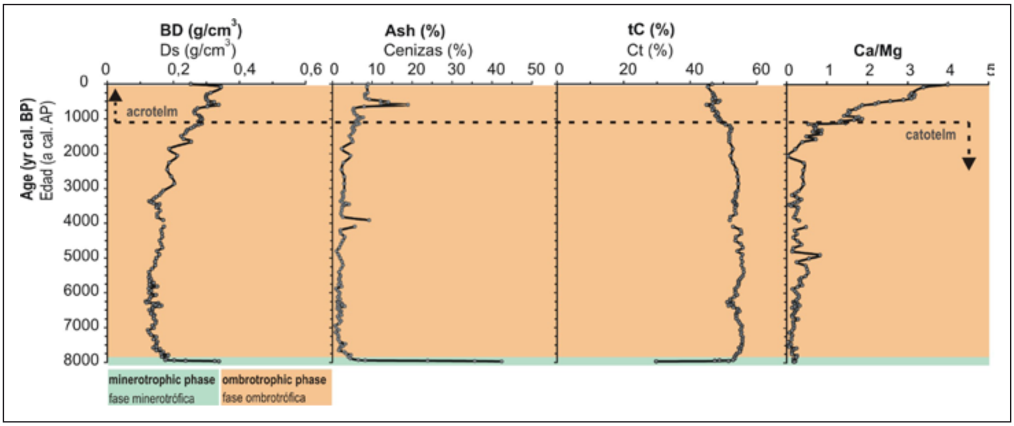

Figure 2. Evolution over time of the properties of the Zalama bog. The peat density (PD), total carbon content (Ct), total molar ratio Ca:Mg.

compared to 3.6 % in the catotelm. If the minerotrophic phase of the bog is eliminated in the calculation of the latter, which accounts for up to 42 % of the ash, the difference between the two functional layers of the bog would be even sharper. The total carbon content (Ct) varies between 29.8 and 56.1 % with an average of 48.3 and 53.6 % in the acrotelm and in the catotelm respectively. These organic carbon values are comparable to those calculated for other ombrotrophic bogs in the North-east of the Iberian Peninsula (Pontevedra-Pombal *et al.*, 2006) and the British Isles (Loisel *et al.*, 2014). The average value of the molar ratio Ca/Mg is clearly under 1.0 in the catotelm, exceeding the value in the acrotelm.

The values (Table 1) and distribution in the profile (Fig. 2) of bulk density (BD), ash content, the molar ratios Ca/Mg and carbon (C) content are within the range of values representative of bogs of an ombrotrophic nature (Shotyk *et al.*, 1996). The dynamics shown by these properties reflect the existence at 7,900 cal. yr BP, of a rapid and drastic transition between the initial formation-phase of the Zalama bog under minerotrophic conditions to an ombrotrophic one, the process being established after 7,750 cal. yr BP.

	Ds (g/cm^3)		Ash (%)		Ct (%)		Ca/Mg		VGR(cm/a)
	acrotelm	catotelm	acrotelm	catotelm	acrotelm	catotelm	acrotelm	catotelm	catotelm
Average	0.287	0.160	7.82	3.57	48.36	53.57	1.93	0.26	0.047
S.D.	0.024	0.038	3.02	5.78	1.98	2.86	0.93	0.15	0.041
Min.	0.243	0.117	5.06	0.80	44.94	29.80	0.54	0.03	0.010
Max.	0.344	0.338	18.98	42.46	52.48	56.10	3.99	0.85	0.195

Table 1. Average values, standard deviation (S.D.), maximum and minimum peat density (PD), ash content and total carbon (Ct), for the ratio Ca:Mg y and the vertical growth rate (VGR) of the Zalama bog.

The interdependency of the ratios that are established between the different properties determined discriminate clearly (Fig. 3) between the minerotrophic and ombrotrophic phases of the bog. Equally, it is possible to define the zone of contact between the acrotelm and the catotelm. The greater density of the samples from the base is conditioned by a greater content of inorganic ash and a reduction in the carbon concentration resulting from a mixing with the underlying mineral bedrock. The variations in these properties in the superficial samples of the acrotelm reflect the nutrient recycling mechanisms, especially Ca, and the incomplete humification of the vegetal necromass.

When the carbon percentage is corrected to the ash and organic material content (C/(cenizas*MO), a net separation between those samples that make up the minerogenic part with a greater influence of the mineral substrate and the samples coming from the surface of the bog and the ombrotrophic samples from the catotelm can be defined.

The behaviour of these properties in the profile also reveals the absence of any hiatus, that is, there has been a single continuous cycle of bog formation over the last 8,000 years. This period of activation in the formation of ombrotrophic raised

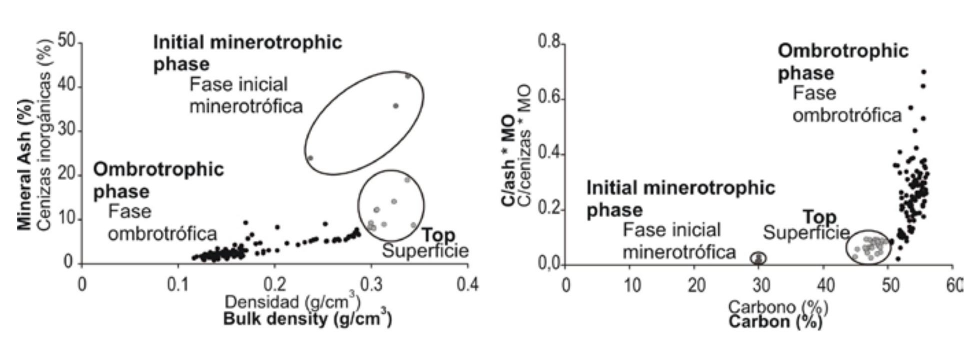

Figure 3. Relationship between the organic carbon, peat density and the mineral material content for the distinct phases in the evolution of the Zalama bog.

and blanket bogs has been described as one of the most intense in the Northern-hemisphere at more northerly latitudes (Malmer 1975; Sjors 1982; Moore *et al.*, 1984; Averdieck *et al.*, 1993; Laine *et al.*, 1996; Mäkilä 1997; Hughes & Barber 2003; MacDonald *et al.*, 2006; Gorham *et al.*, 2007; Gallego-Sala *et al.*, 2015).

The carbon (C) distribution in depth (Fig. 2) shows the typical tendency in ombrotrophic bogs. Although it is possible to define an increase in the carbon content with the depth, associated with the processes of decomposition and loss of mass, this increase is not constant. The rise in C in the bog is faster in the uppermost 30 cm, coinciding with the limit between the acrotelm and the catotelm, and reproduces the same pattern as other Iberian blanket bogs (Pontevedra-Pombal 2002, Pontevedra-Pombal *et al.*, 2004; Kaal *et al.*, 2007). In this type of bogs, the concentrations and the distribution of C in the profile has been linked to the selective accumulation of recalcitrant organic compounds during the decomposition of the organic material (Lu *et al.*, 2000; Pontevedra-Pombal *et al.*, 2001; Disnar *et al.*, 2008; Schellekens *et al.*, 2015).

Evolution in the Holocene

The development of the Zalama bog has been continuous over the last 8,000 years, with an average vertical growth rate of 0.047 cm year^{-1} (21 y cm^{-1}), consistent with the interval of 0.047-0.036 cm year^{-1} determined for other Iberian ombrotrophic bogs (Pontevedra-Pombal *et al.*, 2016), and with the value (0.05 cm year^{-1}) established on a global scale (Gorham, 1991).

However, the calculation of the VGRs for chronological Holocene intervals (Fig. 4) show that the intensity of development was variable. This fact is reasonable as it is a bog whose typology is especially sensitive to changes in the climate.

During the first two thousand years of the existence of the Zalama bog there was a higher rate of growth, with a VGR of around 0.07 cm year^{-1}. This period, after resisting the climate crisis of 8,200 cal. yr BP (Alley & Ágústsdóttir, 2005), is considered as the moment when there was a recovery in the climate which favoured the development of bogs in the northern-hemisphere (Gallego-Sala *et al.*, 2015). Over the following 2,000 years the development of the bog gradually slowed down, reaching a VGR of 0.02 cm year^{-1}, less than half the average in the Holocene, at the end of 4,000 cal. yr BP. Between 6,000 and 4,000 cal. yr BP two intense global scale climatic crises have been identified, known as the neo-glaciation (Magny *et al.*, 2006) and the crisis in 4,200 cal. yr BP (Mayewski *et al.*, 2004). The impact of the neo-glaciation on the Iberian Peninsula has been detected at other bogs (Martínez-Cortizas *et al.*, 1999; Ortiz *et al.*, 2010) and in more Northern sectors of Europe (Loisel *et al.*, 2014).

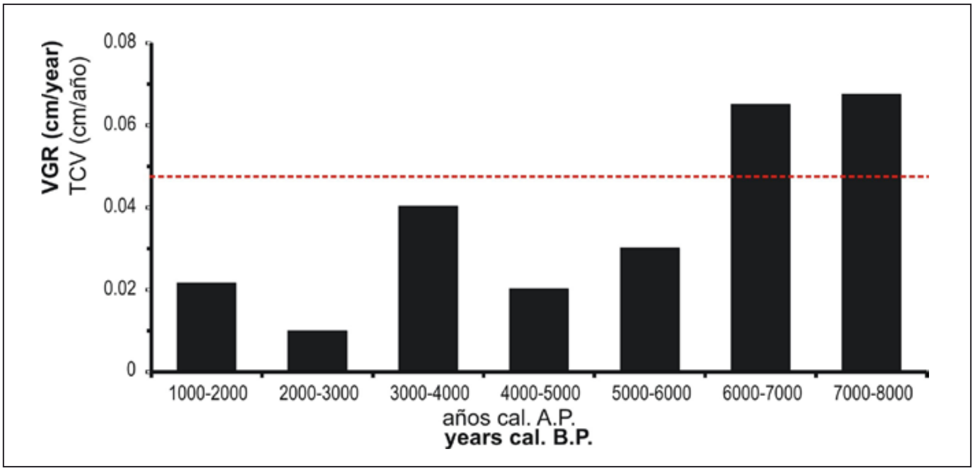

Figure 4. Vertical growth rates (VGRs) of the Zalama bog in the different dated periods of the Holocene (excluding the acrotelm). The dotted red line indicates the average value of the VGR for the whole of the Holocene.

A recovery in the growth rates at Zalama occurred between 4,000 and 3,000 cal. yr BP, with a VGR of 0.04 cm year^{-1}. This phase is synchronized with a thermal/humidity recovery which has been registered at different geographic scales in several types of environmental archives (Martínez-Cortizas *et al.*, 1999; Magny, 2004; Moreno *et al.*, 2011; Ruppel *et al.*, 2013).

In the following millenium (3,000 – 2,000 cal. yr BP) a lower intensity in the growth of the bog was detected, with a VGR of 0.01 cm year^{-1}. This feature has been found in a large number of northern bogs (Loisel *et al.*, 2014) and has been called an event of climatic crisis at a great scale (Yu, 2006).

Between 2,000 and 1,000 cal. yr BP there is a significant recovery that reached rates higher than 0,02 cm year^{-1}. This period coincides with the Roman Warm period and the initial phase of the Medieval Warm period, which has been described from the indicators stored in ombrotrophic bogs (Martínez-Cortizas *et al.*, 1999; Mauquoy *et al.*, 2002; Castro *et al.*, 2015) as a warm-wet period.

Evolution of the vegetation

The oldest part of the pollen recorded at the Zalama bog (Fig. 5) shows maximum values of pine pollen to be 44 %, specifically from *Pinus sylvestris*, the highest in the whole of the sequence. However, owing to its high level of pollen production and dispersal, only values of around 60% allow local presence of pine forests to be confirmed (López-Sáez *et al.*, 2013). This is the reason why, in this case, it can be suggested that there was a certain regional presence of these pines, as well as the existence of isolated pines in the bog surroundings, therefore admitting its indigenous origin in the area of the Eastern Cantabrian mountains.

These pine forests recede notably from 7,585 cal. yr BP, in parallel with the development of deciduous forests. Above all, hazel trees and deciduous oaks - *Quercus* – come to dominate the zone. It is also worth mentioning the presence of walnut trees, although at low densities (<2%). This is a species that is considered as introduced at the end of the Holocene, however, research carried out in the last couple of decades suggest its indigenous character on the Iberian peninsula, the same applying to the Montes de Ordunte. Likewise, riparian species, such as alders, ash and willow, have also been identified, possibly located on a regional scale.

From 6,500 cal. yr BP an interesting phenomenon was detected, the modification of the zone by the first neolithic communities in the Cantabrian mountains, with the diagram showing a certain decline in the forest cover. The forest masses that were most affected by these emerging processes of deforestation are deciduous trees, above all hazels and oaks, owing to their local presence. This decline coincided with the development of the vegetation of anthropogenic nitrophile origin. The affected communities are those closely linked to human exploitation, such as *Aster*, Cardueae, Cichorioideae and *Dipsacus fullonum*, whose percentage increase shows a clear anthropogenic phase.

The processes that modified the countryside seem to be caused by the human activities that have been carried out since Neolithic times. The presence of anthropozoogenous type vegetation (Chenopodiaceae, *Plantago lanceolata*, *Plantago major/media*, *Urtica dioica*), together with the presence of ascospores of coprophilous fungi (*Sordaria* sp.) suggests that the predominant activity in the zone was that of cattle raising (Pérez-Díaz *et al.*, 2016). This evidence is also related to the means that were used to open-up the countryside.

At the same time that there was a decline in the tree density, an increase in the pastures containing species of Gramineae, in the anthropogenic nitrophilous communities, and evidence of grazing (anthropozoogenic communities and *Sordaria* sp.) were detected, as well as the presence of a pyrophilous taxus such as *Asphodelus albus*, which suggests an incidence of phenomena such as local fires. The same is suggested by the presence of ascospores from an equally pyrophilous non-fungal micro-fossile, *Chaetomium* sp. One theory is that this incidence of pyrophilous species, and therefore of fires, could have a natural origin. However, its coincidence with other evidence of anthropization mentioned above show without a doubt the anthropogenic origin of these fires, which were used as a tool to open-up the countryside.

Another interesting phenomenon documented at the Zalama bog is the expansion of the beech forests. Their first appearance in the pollen register is around 3,900 cal. yr BP, however from 3,300 cal. yr BP on, their continuous presence and later expansion is detected, making them into one of the most notable formations at local level.

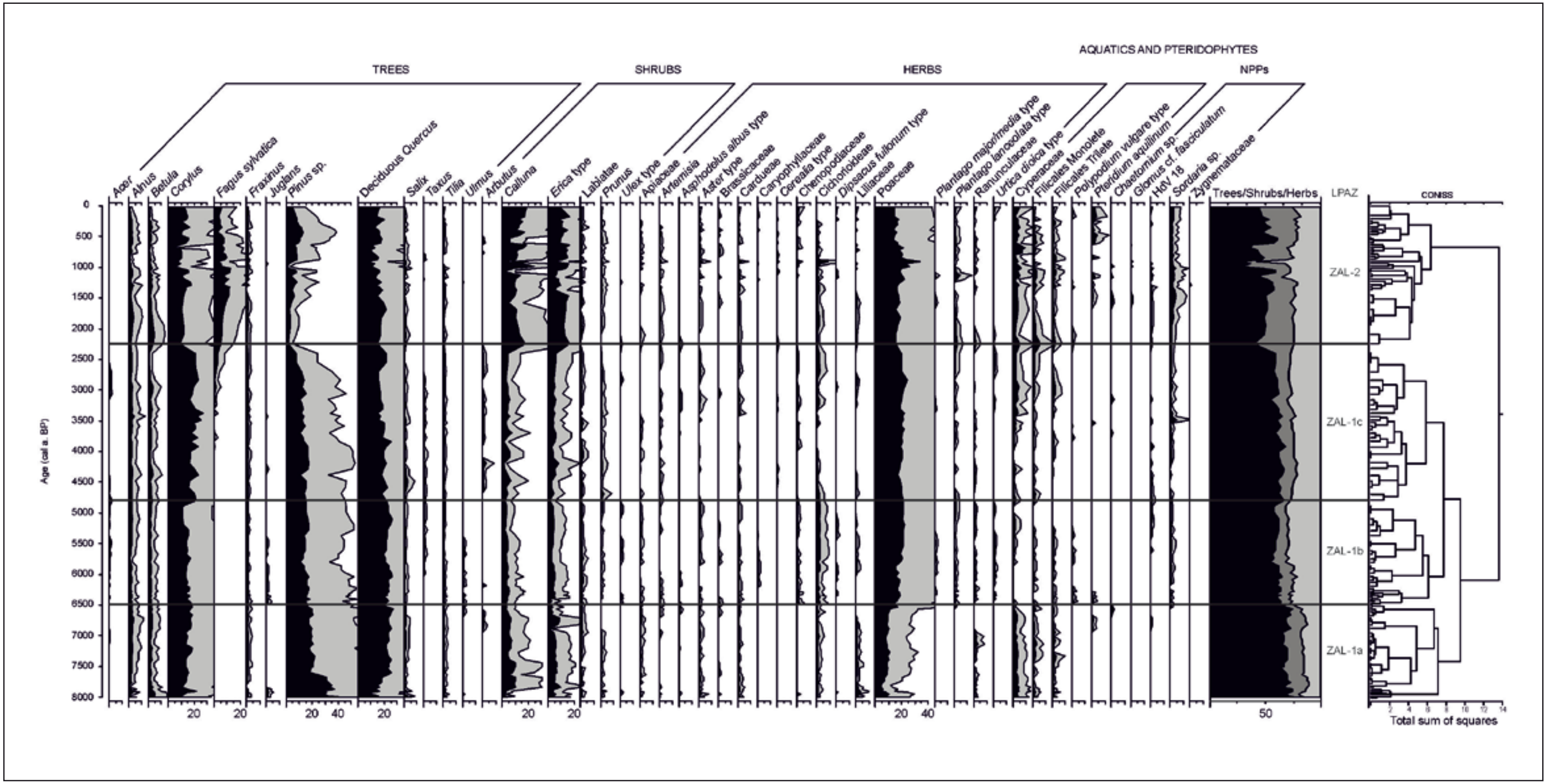

Figure 5. Palynological records from the Zalama blanket bog.

Conclusions

The geo-chemical properties determined at the Zalama bog are characterístic of ombrotrophic blanket bogs, and their behaviour over the period of the profile indicates that the pattern is not affected to a great depth by the mechanisms of degradation that have damaged the surface of the bog. Therefore, it continues to be a useful and essential record for the understanding of the local and regional patterns associated with climate dynamics and anthropization.

The general response of the bog to the climatic evolution in the Holocene, interpreted from its growth rates, is very consistent with the patterns of evolution established at regional scale for Iberian blanket bogs, and at a global scale in bogs all over the Northern hemisphere.

The evolution of the vegetation over the last 8,000 years shows a notable initial presence of pine forests but after that it shows a declining trend. Next to these, the deciduous forests played a leading role in the Montes de Ordunte. This zone was not unaffected by historical processes, such as the adoption of a productive economy during the Neolithic, which left a clear mark on this mountainous countryside.

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References

Alley, R.B., Ágústsdóttir, A.M. 2005. The 8k event: cause and consequences of a major Holocene abrupt climate change. *Quaternary Science Reviews* 24: 1123–1149.

Averdieck, F.R., Hayen, H., Heathwaite, A.L., Willkomm, H. 1993. The chronology of mire development. In: Heathwaite, A. L. & Göttlich, K. (eds.) *Mires: Process, Explotation and Conservation*, John Wiley & Sons, Chichester, England, 123–170.

Blaauw, M. 2010. Methods and code for "classical" age-modelling of radiocarbon sequences. *Quaternary Geochronology* 5: 512–518.

Castro, D., Souto, M., García-Rodeja, E., Pontevedra-Pombal, X., Fraga, M.I. 2015. Climate change records between the mid- and late Holocene in a peat bog from Serra do Xistral (SW Europe) using plant macrofossils and peat humification analyses. *Palaeogeography, Palaeoclimatology, Palaeoecology* 420: 82–95.

Cugny, C. 2011. *Apports des microfossiles non-polliniques à l´histoire du pastoralisme sur le versand nord pyréneén. Entre référentiels actuels et reconstitution du passé*. Vol. 1. Texte et figures/Vol. 2. Illustrations et descriptions de microfossiles, Ph.D. thesis. University Toulouse II Le Mirail.

de Jong, R., Blaauw, M., Chambers, F.M., Christensen, T.R., de Vleeschower, F., Finsinger, W., Fronzek, S., Johansson, M., Kokfelt, U., Lamentowicz, M., Le Roux, G., Mauquoy, D., Mitchell, E.A.D., Nichols, J.E., Samaritani, E., van Geel, B. 2010. Climate and peatlands. In Dodson, J. (ed): *Changing Climates, Earth Systems and Society*, Springer, Dordrecht. pp: 85–121.

Disnar, J.R., Jacob, J., Morched-Issa, M., Lottier, N., Arnaud, F. 2008. Assessment of peat quality by molecular and bulk geochemical analysis: application to the Holocene record of the Chautagne marsh (Haute Savoie, France). *Chemical Geology* 254: 101–112.

Evans, D. 2006. The habitats of the European Union Habitats Directive. *Biology and Environment: Proceedings of the Royal Irish Academy* 106B: 167–173.

Faegri, K., Iversen, J. 1989. *Text-book of pollen analysis.* John Wiley & Sons, Chichester. 328 pp.

Gallego-Sala, A.V., Charman, D.J., Harrison, S.P., Li, G., Prentice, I.C. 2015. Climate-driven expansion of blanket bogs in Britain during the Holocene. *Climate of the Past Discussions* 11: 4811–4832.

García-Antón, M., Ruiz, B. & Ugarte, F.Mª. 1989. Análisis geomorfológico y palinológico de la turbera de Saldropo (Barazar. Zeanuri/Bizkaia). *Lurralde* 12: 25–44.

Girard, M., Renault-Miskovsky, J. 1969. Nouvelles techniques de préparation en palynologie appliquées à trois sédiments du Quaternaire final de l´Abri Cornille (Istres, Bouches du Rhône). *Bulletin de l´Association Française pour l´Etude du Quaternaire* 1969: 275–284.

Goeury, C., de Beaulieu, J. L. 1979. À propos de la concentration du pollen à l´aide de la liqueur de Tholet dans les sédiments minéraux. *Pollen et Spores* 21: 239–251.

Gorham, E. 1991. Northern peatlands: role in the carbon cycle and probable responses to climatic warming. *Ecological Applications* 1: 182–195.

Gorham, E., Lehman, C., Dyke, A., Janssens, J., Dyke, L. 2007. Temporal and spatial aspects of peatland initiation following deglaciation in North America. *Quaternary Science Reviews* 26: 300–311.

Heras P. 2002. *Determinación de los valores ambientales de la turbera del Zalama (Carranza; Bizkaia) y propuestas de actuación para su conservación*. Dirección de Aguas del Departamento de Ordenación del Territorio y Medio Ambiente del Gobierno Vasco.

Heras, P., Infante, M. 2005. More about windmills and blanket bogs. Zalama in Northern Spain. *IMGC Newsletter* 2005, 22.

Heras, P., Infante, M., Pontevedra-Pombal, X., Nóvoa-Muñoz, J.C. 2016. Mires and peatlands of Spain. En: Joosten, H., Tanneberger, F., Moen, A. (eds.), *Mires and peatlands of Europe: Status, distribution, and nature conservation*, Schweizerbart Science Publishers, Stuttgart.

Hughes, P.D. M., Barber, K.E. 2003. Mire development across the fen-bog transition on the Teifi floodplain at Tregaron Bog, Ceredigion, Wales, and a comparison with 13 other raised bogs. *Journal of Ecology* 91: 253–264.

Kaal, J., Baldock, J.A., Buurman, P., Nierop, K.G.J., Pontevedra-Pombal, X., Martínez-Cortizas, A. 2007. Evaluating pyrolysis–GC/MS and ^{13}C CPMAS NMR in conjunction with a molecular mixing model of the Penido Vello peat deposit, NW Spain. *Organic Geochemistry* 38: 1097–1111.

Laine, J., Silvola, J., Tolonen, K., Alm, J., Nykänen, H., Vasander, H., Sallantaus, T., Savolainen, I., Sinisalo, J., Martikainen, P.J. 1996. Effect of water-level drawdown on global climatic warming: northern peatlands. *Ambio* 25: 179–184.

Loisel, J., Yu, Z., Beilman, D.W., Camill, P., Alm, J., Amesbury, M.J., Anderson, D., Andersson, S., Bochicchio, C., Barber, K.E., Belyea, L.R., Bunbury, J., Chambers, F.M., Charman, D.J., De Vleeschouwer, F., Fiałkiewicz-Kozieł, B., Finkelstein, S.A., Gałka, M., Garneau, M., Hammarlund, D., Hinchcliffe, W., Holmquist, J., Hughes, P.D.M., Jones, M.C., Klein, E.S., Kokfelt, U., Korhola, A., Kuhry, P., Lamarre, A., Lamentowicz, M., Large, D., Lavoie, M., MacDonald, G., Magnan, G., Makila, M., Mallon, G., Mathijssen, P., Mauquoy, D., McCarroll, J., Moore, T.R., Nichols, J., O'Reilly, B., Oksanen, P., Packalen, M., Peteet, D., Richard, P.J.H., Robinson, S., Ronkainen, T., Rundgren, M., Sannel, A.B.K., Tarnocai, C., Thom, T., Tuittila, E.S., Turetsky, M., Valiranta, M., van der Linden, M., van Geel, B., van Bellen, S., Vitt, D., Zhao, Y., Zhou, W. 2014. A database and synthesis of northern peatland soil properties and Holocene carbon and nitrogen accumulation. *The Holocene* 24: 1028–1042.

López-Sáez, J. A., Sánchez-Mata, D., Alba-Sánchez, F., Abel-Schaad, D., Gavilán, R. G., Pérez-Díaz, S. 2013. Discrimination of Scots pine forests in the Iberian Central System (Pinus sylvestris var. iberica) by means of pollen analysis. Phytosociologial considerations. *Lazaroa* 34: 191–208.

Lu, X. Q., Hanna, J. V., Johnson, W. D. 2000. Source indicators of humic substances: an elemental composition, solid state ^{13}C CPMAS-NMR and Py-GC/MS study. *Applied Geochemistry* 15: 1019–1033.

Lynn, W., McKinzie, W.F., Grossman, R.B. 1974. Field laboratory tests for characterization of Histosols. In: Aandahl, A.R., Buol, S.W., Hill, D.E., Bailey, H. (eds.), *Histosols: their characteristics, classification, and use*. Madison: Soil Science Soc America Inc. 11–20.

MacDonald, G.M., Beilman, D.W., Kremenetski, K.V., Sheng, Y., Smith, L.C., Velichko, A.A. 2006. Rapid Early Development of Circumarctic Peatlands and Atmospheric CH4 and CO2 variations. *Science* 314: 285–288.

Magny, M., Leuzinger, U.R.S., Bortenschlager, S., Haas, J.N. 2006. Tripartite climate reversal in central Europe 5600–5300 years ago. *Quater-*

nary Research 65: 3–19.

Mäkilä, M. 1997. Holocene lateral expansion, peat growth and carbon accumulation on Haukkasuo, a raised bog in southeastern Finland. *Boreas* 26: 1–14.

Malmer, N. 1975. Development of Bog mires. In: Hassler, A. D. (ed.) *Coupling of land and water systems*. Ecological Studies 10, Springer – Verlag, New York 85-92.

Mariscal, B. 1993. Variación de la vegetación holocena (4300-280 BP) de Cantabria a través del análisis polínico de la turbera de Alsa. *Estudios Geológicos* 49: 63–68.

Martínez-Cortizas, A., Pontevedra-Pombal, X., Nóvoa-Muñoz, J.C., García-Rodeja, E., Shotyk, W. 1999. Mercury in a Spanish peat bog: archive of climate change and atmospheric metal deposition. *Science* 284: 939–942.

Mauquoy, D., Engelkes, T., Groot, M.H.M., Markesteijn, F., Oudejans, M.G., van der Plicht, J., van Geel, B. 2002. High-resolution records of late-Holocene climate change and carbon accumulation in two north-west European ombrotrophic peat bogs. *Palaeogeography, Palaeoclimatology, Palaeoecology* 186: 275–310.

Mayewski, P.A., Rohling, E.E., Stager, J.C., Karlén, W., Maasch, K.A., Meeker, L.D., Meyerson, E.A., Gasse, F., van Kreveld, S., Holmgren, K., Lee-Thorp,J., Rosqvist, G., Rack, F., Staubwasser, M., Schneider, R.R., Steig, E.J. 2004. Holocene climate variability. *Quaternary Research* 62: 243– 255.

Menéndez, J. 1968. Estudio esporo-polínico de una turbera en el Valle de la Nava (provincia de Burgos). *Boletín de la Real Sociedad Española de Historia Natural (Geología)* 66: 35–39.

Moore, P. D., Webb, J. A., Collinson, M. E. 1991. *Pollen Analysis*, Blackwell Scientific Publications, London. 216 pp.

Moore, P.D., Merryfield, D.L., Price, M.D.R. 1984. The vegetation and development of blanket mires. En: Moore, P. (ed.), *European mires*. Academic Press, London, 203–235.

Moreno, A., López-Merino, L., Leira, M., Marco-Barba, J., González-Sampériz, P., Valero-Garcés, B.L., López-Sáez, J.A., Santos, L., Mata, P., Ito, E. 2011. Revealing the last 13,500 years of environmental history from the multiproxy record of a mountain lake (Lago Enol, northern Iberian Peninsula). *Journal of Paleolimnology* 46: 327–349.

Muñoz-Sobrino, C. 2001. *Cambio climático y dinámica del paisaje en las montañas del noroeste de la Península Ibérica*. Ph D Thesis, Universidade de Santiago de Compostela.

Ortiz, J.E., Gallego, J.L.R., Torres, T., Díaz-Bautista, A., Sierra, C. 2010. Palaeo-environmental reconstruction of Northern Spain during the last 8000 cal yr BP based on the biomarker content of the Roñanzas peat bog (Asturias). *Organic Geochemistry* 41: 454–466.

Peech, M., Alexander, L.T., Dean, L., Reed, J.F. 1947. *Methods of soil analysis for soil fertility investigations*. U.S. Deparment of Agriculture Circular 757.

Peñalba, M.C. 1994 The History of the Holocene Vegetation in Northern Spain from Pollen Analysis. *Journal of Ecology* 82: 815–832.

Pérez-Díaz, S., López-Sáez, J.A., Pontevedra-Pombal, X., Souto-Souto, M., Galop, D. 2016. 8000 years of vegetation history in the northern Iberian Peninsula inferred from the palaeoenvironmental study of the Zalama ombrotrophic bog (Basque-Cantabrian Mountains, Spain). *Boreas* 45: 658–672.

Pérez-Obiol, R., García-Codron, J.C., Pélachs, A., Pérez-Haase, A., Soriano, J.M. 2016. Landscape dynamics and fire activity since 6740 cal yr BP in the Cantabrian region (La Molina peat bog, Puente Viesgo, Spain). *Quaternary Science Reviews* 135: 65 – 78.

Pontevedra Pombal, X., Nóvoa Muñoz, J.C., Martínez Cortizas, A., García-Rodeja, E. 2001. ¿Por qué preservar las turberas?. En: Martínez, A. & Rodeja-Gayoso, E. (eds.), *Turberas de Montaña de Galicia*. Colección Técnica – Medio Ambiente, Xunta de Galicia, Santiago de Compostela. pp: 177-190.

Pontevedra–Pombal, X. 2002. *Turberas de Montaña de Galicia. Génesis, propiedades y su aplicación como registros ambientales geoquímicos*. 1st ed. Tesis Doctoral. Univ. Santiago de Compostela Editorial, Compostela (Spain).

Pontevedra-Pombal, X., Castro, D., Carballeira, R., Souto, M., López-Sáez, J.A., Pérez-Díaz, S., Fraga, M.I., Valcárcel, M., García-Rodeja, E. 2016. Genesis and evolution of the Iberian peatlands. *Mires and Peats*.

Pontevedra-Pombal, X., Martínez-Cortizas, A., Buurman, P. 2004. Las turberas de montaña de Galicia como sumideros de carbono. *Edafología* 11: 295–307.

Pontevedra-Pombal, X., Martínez-Cortizas, A., García-Rodeja, E., Buurman, P., Van Lagen, B. 2001. Composición y transformación de la materia orgánica de un histosol de la Serra do Xistral (norte de Galicia) mediante CPMAS ^{13}C-NMR. *Edafología* 8: 67–79.

Pontevedra-Pombal, X., Nóvoa-Muñoz, J.C., García-Rodeja, E., Martínez-Cortizas, A. 2006. Mountain mires from Galicia (NW Spain). En: Martini, I.P., Martínez, A. & Chesworth, W. (eds.), *Peatlands: basin evolution and repositories of records of environmental and climate changes*. Elsevier (Development in Earth Surface Processes), London, 83–108.

Reille, M. 1999. *Pollen et spores d´Europe et* *d´Afrique du Nord*, Laboratoire de Botanique Historique et Palynologie, Marseille. 535 pp.

Reimer, P. J., Bard, E., Bayliss, A., Beck, J. W., Blackwell, P.G., Bronk, Ramsey. C., Buck, C.E., Cheng, H., Edwards, R.L., Friedrich, M., Grootes, P.M., Guilderson, T.P., Haflidason, H., Hajdas, I., Hatté, C., Heaton, T.J., Hoffmann, D.L., Hogg, A.G., Hughen, K.A., Kaiser, K.F., Kromer, B., Manning, S.W., Niu, M., Reimer, R.W., Richards, D.A., Scott, E.M., Southon, J.R., Staff, R.A., Turney, C.S.M., van der Plicht, J. 2013. Intcal13 and marine13 radiocarbon age calibration curves 0–50,000 years cal BP. *Radiocarbon*, 55: 1869–1887.

Ruppel, M., Väliranta, M., Virtanen, T., Korhola, A. 2013. Postglacial spatiotemporal peatland initiation and lateral expansion dynamics in North America and northern Europe. *The Holocene* 23: 1596 – 1606

Schellekens, J., Buurman, P., Kuyper, T.W., Abbott, G.D., Pontevedra-Pombal, X., Martínez-Cortizas, A. 2015. Influence of source vegetation and redox conditions on lignin-based decomposition proxies in graminoid-dominated ombrotrophic peat (Penido Vello, NWSpain). *Geoderma* 237: 270–282.

Shotyk, W. 1996. Peat bog archives of atmospheric metal deposition: geochemical evaluation of peat profiles, natural variations in metal concentrations, and metal enrichment factors. *Environmental Reviews* 4: 149–183.

Sjörs, H. 1982. The zonation of northern peatlands and their importance for the carbon balance of the atmosphere. In: Gopal, B., Turner, R. E., Wetzel, R. G. & Whigham, D. F. (eds.), *Wetlands: ecology and management*. National Institute of Ecology and International Scientific Publ., Jaipur, India. 11–14.

Turunen, J., Roulet, N.T., Moore, T.R., Richard P. J. H. 2004. Nitrogen deposition and increased carbon accumulation in ombrotrophic peatlands in eastern Canada. *Global Biogeochemical Cycles* 18: GB3002.

Van Geel, B. 1978. A palaeoecological study of Holocene peat bog sections in Germany and The Netherlands. *Review of Palaeobotany and Palynology* 25: 1–120.

Van Geel, B. 2001. Non-pollen palynomorphs. In Smol, J. P., Birks, H. J. B., Last, W. M. (eds.): *Tracking environmental change using lake sediments; volume 3: Terrestrial, algal and silicaceous indicators*, Kluwer Academic Publishers, Dordrecht. pp: 99–119.

Van Geel, B. 2006. Fossil ascomycetes in Quaternary deposits. *Nova Hedwigia* 82: 313–329.

Van Geel, B., Bohncke, S. J. P., Dee, H. 1981. A palaeoecological study of an Upper Late Glacial and Holocene sequence from 'De Borchert', The Netherlands. *Review of Palaeobotany and Palynology* 31: 367–448.

Van Geel, B., Buurman, J., Brinkkemper, O., Schelvis, J., Aptroot, A., van Reenen, G., Hakbijl T. 2003. Environmental reconstruction of a Roman Period settlement site in Uitgeest (The Netherlands), with special reference to coprophilous fungi. *Journal of Archaeological Science* 30: 873–883.

Van Geel, B., Coope, G. R., van der Hammen, T. 1989. Palaeoecology and stratigraphy of the Lateglacial type section at Usselo (The Netherlands). *Review of Palaeobotany and Palynology* 60: 25–129.

Von Post, L. 1937. The geographical survey of Irish bogs. *Irish Naturalist's Journal* 6: 210–227.

Wardenaar, E.P.C. 1987. A new hand tool for cutting peat profiles. *Canadian Journal of Botany* 65: 1172–1173.

Yu, Z. 2006. Fen peatlands in boreal western canada: a complex ecosystem response to climate variation and disturbance. *Ecosystems* 9: 1278–1288.

This book gathers articles from specialists who describe their latest researches and experiences about the study and conservation of peatlands. It aims, therefore, to disseminate technical knowledge and promote actions favouring this type of unique, scarce and particularly threatened habitats in Southern Europe. The LIFE+ Sustainable Ordunte (2012-2017), a project co-financed by the LIFE program of the European Union and implemented by the Provincial Council of Bizkaia and the Hazi Foundation, has focused on the restoration of the Zalama blanket bog.

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Drone photograph of the Zalama blanket bog (Bizkaia), where the cells protected by coconut fiber mesh can be seen. This helps to stabilize slopes exposed to erosion, limit the loss of peat and restore the vegetation (University of Nottingham Trent).

Cover: *Eriophorum vaginatum*
(Sergio González Ahedo)

Inventory, value and restoration of peatlands and mires:

recent contributions

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