Identification, geo-hydromorphological assessment and the state of degradation of the southernmost blanket bogs in Europe

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'Protect, restore, fund'

RE-DISCOVER drawn by Beth Madeley

Abstract

Blanket bogs are a globally rare type of ombrotrophic peatland internationally recognised for long-term terrestrial carbon storage, the potential to serve as carbon sinks and habitat provision. The majority of recognised areas of this habitat in Europe are found in the United Kingdom and Ireland, but the rarer examples found in Spain represent the southernmost continental edge-of-range. However, gaps in the peatland inventory suggest that a number of blanket bogs in the Cantabrian Mountains (northern Spain) are not recognised and are at increased threat of loss.

This study identifies and provides geo-hydromorphological classification for 14 unrecorded blanket bogs and one protected blanket bog located between the administrative regions of Cantabria and Castilla y León. Peat depth surveys and carbon analysis of peat cores were used to determine the amount of carbon stored within the newly identified blanket bogs and the current rate, and drivers, of peatland degradation were examined using remote sensing techniques.

A total extent of blanket bog covering 44.45 ha (>40 cm peat depth) containing more than 500,000 m^3 of peat and an estimated 44.88 ± 3.31 kt C was mapped. Approximately 30.8% of the surface of blanket bogs examined was exposed peat, and even in the protected site, exposed peat surfaces are losing a minimum of 1.7 t C m^{-2} yr^{-1}. The presence of livestock in unprotected sites is increasing the rate of erosion by over five times, and without protection exposed peat surfaces are releasing as much as 3.84 t C m^{-2} yr^{-1}.

The peatlands identified in this research extend the known limit of blanket bogs in Europe farther south than previously recorded and represent 10.5% of blanket bog currently recognised and protected in Spain. The range of anthropogenic pressures currently acting on peatlands in the Cantabrian Mountains, specifically livestock and windfarms, indicates that without protection these important landforms and stored carbon may be lost. An urgent update of European peatland inventories is thus required to preserve these valuable carbon stores and potential carbon sinks.

Publications and dissemination activities for this project

I. PUBLICATIONS

Chico, G., Clutterbuck, B., Clough, J., Lindsay, R., Midgley, N.G. & Labadz, J.C. (2020) *Geo-hydromorphological assessment of Europe's southernmost blanket bogs*. Earth Surface Processes and Landforms, 45 (12), 2747–2760.

Chico, G., Clutterbuck, B., Lindsay, R., Midgley, N.G. & Labadz, J. (2019) *Identification and classification of unmapped blanket bogs in the Cordillera Cantábrica, northern Spain.* Mires and Peat, 24 (02), 1–12.

Chico, G., Clutterbuck, B., Midgley, N.G. & Labadz, J. (2019*) Application of Terrestrial Laser Scanning to quantify surface changes in restored and degraded blanket bogs*. Mires and Peat, 24 (14), 1–24.

Chico, G. (2019) *Evaluating the importance of blanket bogs in northern Spain using traditional and remote sensing techniques*. Remote Sensing newsletter (April-2019)

Clutterbuck, B., **Chico, G.**, Labadz, J. & Midgley, N.G. (2018) *The potential of geospatial technology for monitoring peatland environments*. In: Fernández-García, J.M. & Pérez, F.J. (eds.) Inventory, Value and Restoration of Peatlands and Mires: Recent Contributions, HAZI foundation, Bizkaia, 167–181.

II. REPORTS

Clutterbuck, B., Lindsay, R., **Chico, G.** & Clough, J. (2020). *Hard Hill experimental plots on Moor House – Upper Teesdale National Natural Reserve. A review of the experimental set up.* Natural England Commissioned Report N° 321

Chico, G. & Clutterbuck, B. (2019). *Annual Report – Post LIFE Ordunte Sostenible.* Nottingham Trent University.

Chico, G. & Clutterbuck, B. (2018). Final Technical Report – LIFE+ Ordunte Sostenible. Nottingham Trent University.

III. CONFERENCES

Peat Fest (May 2020)
Talk: RE-DISCOVER – The unknown European blanket bogs
Global – Online

British Society for Geomorphology Annual Meeting (September 2019)
Poster: Geomorphological assessment of unprotected and threatened blanket bog in northern Spain
Sheffield – England

18th Biennial Australian and New Zealand Geomorphology Group meeting (February 2019)
Talk and poster: The significance of blanket bogs in the context of climate change
Inverloch – Australia

Halfway Seminar Oreka Mendian – LIFE project (November 2018)
Keynote speaker: Monitoring restoration of blanket bogs using remote sensing
Vitoria – Spain

British Society for Geomorphology Annual Meeting (September 2018)
Talk: Unmapped blanket bog in North Spain, how long before it is gone? An exploration of conventional and novel techniques
Aberystwyth – Wales

UK National Earth Observation Conference (September 2018)
Talk and poster: Unmapped blanket bog in North Spain, how long before it is gone? An exploration of conventional and novel techniques
Birmingham – England

Halfway Seminar Humberhead – LIFE+ project (May 2018)
Poster: An urgent need for identification and designation of unmapped blanket bogs
Doncaster – England

LIFE+ Ordunte Sostenible final seminar (December 2017)
Keynote speaker: Characterisation and restoration of blanket bogs. From England to Cantabrian Mountains
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BogFest (September 2017)
Poster: Application of Terrestrial Laser Scanning to quantify surface changes in blanket bogs of North Spain
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Acronyms and abbreviations

AOI: Area of interest

C: Carbon

DOC: Dissolve Organic Carbon

EC: European Commission

EU: European Union

GHG: Green House Gases

GLM: General Linear Model

Gt: Gigatonnes

LIA: Little Ice Age

m: Metres

masl: Metres about sea level

Max: Maximum

MCA: Medieval Climate Anomaly

Mha: Millions of hectares

Min: Minimum

mm: Millimetre

Mt: Megatonnes

NEE: Net Ecosystem Exchange

Kt: Kiloton

POC: Particle Organic Carbon

SD: Standard Deviation

SfM: Structure-from-Motion

spp: Species

t: Tonnes

TLS: Terrestrial Laser Scanner/Scanning

UAV: Unnamed Aerial Vehicle

UK: United Kingdom

yr: Year

Chapter 1
Introduction

1.1. European blanket bogs

Peatlands only cover around 2.84% (4.23 million km^2) of the Earth's land surface (Xu *et al.*, 2018), but represent approximately 20% of terrestrial organic carbon storage, and therefore play an important role in the global carbon cycle (Gorham, 1991; Yu *et al.*, 2010). Blanket bogs are a globally rare and unique type of ombrotrophic (rain-fed) peatland, that mainly form in areas with oceanic climates characterised by high atmospheric moisture content and precipitation (>1,000 mm yr^{-1}), low average temperatures (<15°C) and low seasonal temperature variability (Lindsay *et al.*, 1988). They can cover entire landscapes, and in addition to providing carbon storage, blanket bogs are also recognised for internationally important habitat provision (Tallis, 1998) and are protected under the European Habitats Directive (92/43/EEC; European Commission, 2019).

In Europe, the largest expanses of blanket bogs are mainly found in the United Kingdom, Ireland and Norway (Lindsay 1995), with smaller and more isolated examples in Sweden, France and Spain (Joosten et al. 2017; Figure 1.1..1). The value of isolated blanket bogs often outweighs their extent, as they contain a record of local and regional vegetation extending for potentially several millennia (Ramil-Rego and Aira-Rodríguez, 1994; Ellis and Tallis, 2000; Pontevedra-Pombal *et al.*, 2019). Moreover, the edge-of-range examples of this habitat may provide valuable insight into the effects of predicted climate change on blanket bog distribution and evolution (Gallego-Sala and Prentice, 2013).

Spanish blanket bogs are located in the north of the country along the Cantabrian Mountains, which extend from Galicia in the west of the Iberian Peninsula to the Pyrenees in the east (Figure 1.1). The role of climate and topography in blanket bog formation is clear (Lindsay, 1995), and in north Spain the climate of the Atlantic bio-geographical region is heavily influenced by the Atlantic Ocean (Heras *et al.*, 2017), with the majority of blanket bogs located above 600 masl where climatic conditions are more suitable for this habitat (Pontevedra-Pombal *et al.*, 2009). The largest areas of protected blanket bogs in Spain are in Galicia at Serra do Xistral decreasing in their size and number from west to east along the Cantabrian Mountains (European Environment Agency, 2019; Figure 1.1), although the majority of blanket bogs recorded in Asturias are suggested to be misclassified and are in fact other types of peatland (Ramil-Rego *et al.*, 2017; Figure 1.1). There is only one recognised blanket bog in the Basque Country (Zalama blanket bog) and this is currently the known southernmost edge-of-range of this habitat in continental Europe (Heras *et al.*, 2017). However, the Cantabrian Mountains are a 'hot spot' for this habitat, yet there are currently no blanket bogs recorded in the region of Cantabria, and an important gap in the inventory has been highlighted to exist between the regions of Cantabria and Castilla y León (Ramil-Rego *et al.*, 2017; Heras and Infante, 2018; Figure 1.1).

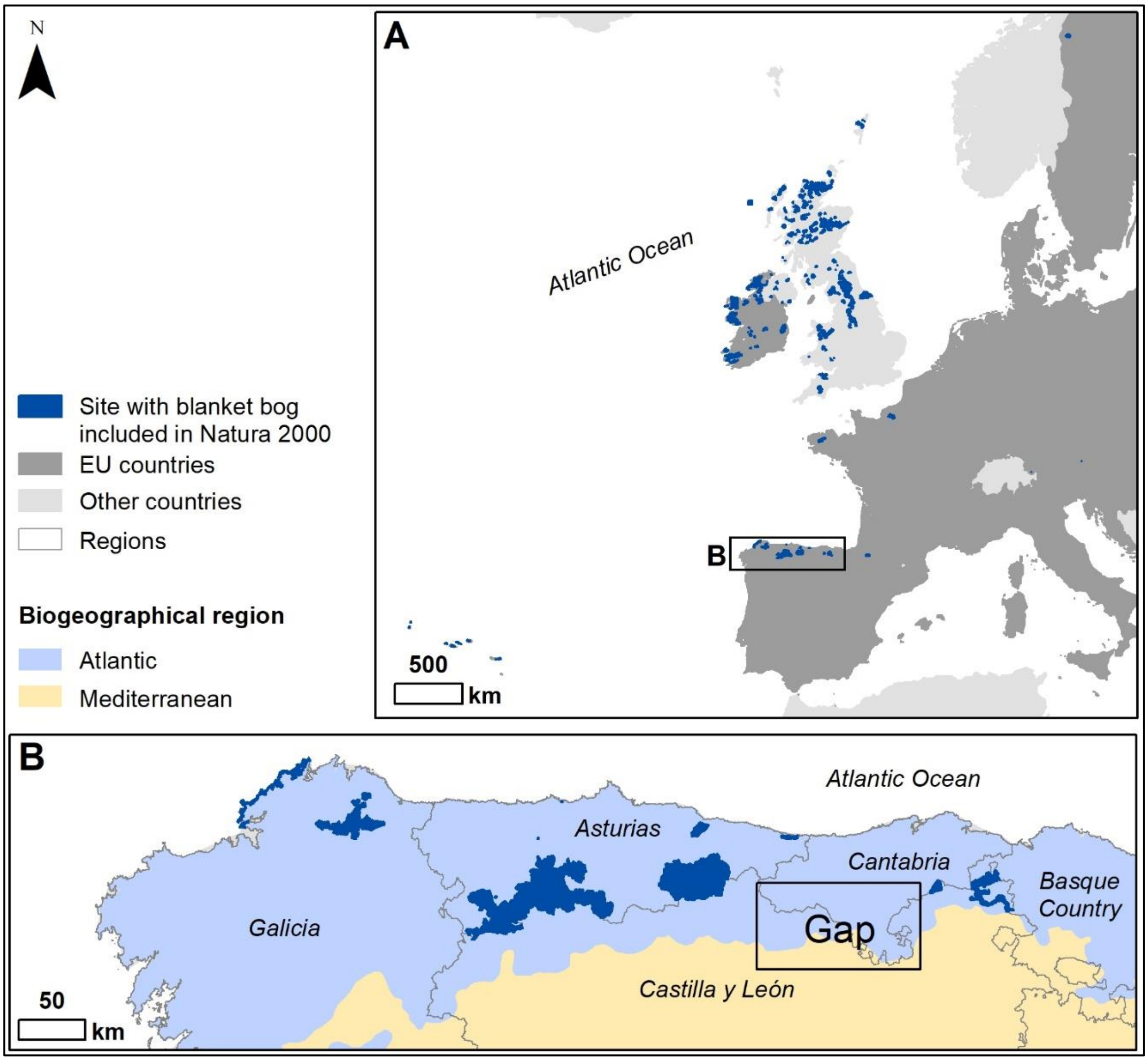


Figure 1.1. A) Natura 2000 sites with recognised blanket bog (7130) across Europe and B) in Spain highlighting the gap in the blanket bog inventory in northern Spain.

1.2. Blanket bogs and climate change

Peatlands have stored carbon from the atmosphere over millennial timescales (Pérez-Díaz *et al.*, 2016; Bunsen and Loisel, 2020), and they can continue to sequester carbon for centuries (IPCC, 2019). Protecting this store of carbon and their potential function to act as carbon sinks (Nugent *et al.*, 2018) could be an important approach to mitigating the impacts of climate change (Joosten, Tapio-Biström and Tol, 2012; Joosten *et al.*, 2016), and decrease current greenhouse gas emissions by reducing the conversion of peatlands to other land uses (e.g. agriculture; IPCC, 2019). However, a large number of peatlands, including large extents of blanket bogs, are degraded and are now acting as carbon sources (Joosten, 2009). Approximately 10% of global peatlands are either drained or undergoing mining activities, and a total of 1,298 Mt of CO_2 are emitted every year from degraded peatlands (Joosten, 2009). However, it has been demonstrated that restored peatlands can return to function as carbon sinks and retain the long-term store of carbon (Nugent *et al.*, 2018).

The total amount of carbon stored in blanket bogs within the Cantabrian Mountains (Spain) is currently unknown, although estimations at Serra do Xistral (Galicia) indicate that over 8.6 Mt of carbon are stored in this region alone, highlighting the importance of this habitat for its contribution to regional and national carbon budgets (Gómez-orellana *et al.*, 2014). While determination of the carbon stored in other recognised blanket bogs in Spain is undoubtedly required, it is not clear how currently unrecognised blanket bogs contribute to terrestrial carbon storage.

1.3. Degradation of blanket bogs

The majority of peatlands across the world are damaged or degraded as a result of a diverse range of natural and anthropogenic pressures (Evans, 1977; Price, Heathwaite and Baird, 2003; Warburton, 2003; Yeloff, Labadz and Hunt, 2006; Holden *et al.*, 2006; Evans and Warburton, 2007; Foulds and Warburton, 2007b; McHugh, 2007; Ward *et al.*, 2007; Luscombe *et al.*, 2016; Wawrzyczek *et al.*, 2018; Li *et al.*, 2018). Natural pressures that facilitate degradation, or loss of peat, primarily relate to erosion driven by aeolian, fluvial or freeze-thaw processes (Warburton, 2003; Evans and Warburton, 2007; Foulds and Warburton, 2007a; Li, Holden and Grayson, 2018). Such erosion processes mainly act on areas of exposed peat, although the relationship between natural pressures and the influence of the anthropogenic activities that may increase exposure of peat surfaces, is key to understanding degradation across blanket bogs more widely.

Anthropogenic pressures on blanket bogs include prescribed burning (Yallop *et al.*, 2006), wildfires (Yeloff, Labadz and Hunt, 2006), drainage (Holden *et al.*, 2006), overgrazing (Ward *et al.*, 2007) and windfarms (Heras and Infante, 2008; Wawrzyczek *et al.*, 2018). In northern Spain, peat extraction, windfarms and livestock have been reported to be the most significant factors causing degradation of blanket bogs (Heras and Infante, 2008; Heras *et al.*, 2017), although prescribed burning and wildfires also play an important role (Heras, 2002). Both peat extraction and the construction of windfarms can result in large-scale removal of peat, and in some cases, the complete loss of the peatland (Heras and Infante, 2008; Heras *et al.*, 2017). In contrast, burning activities to improve grazing, plus the direct impact of livestock, increase the extent of exposed peat (Heras, 2002), but the scale of damage caused by livestock is not known.

Recognised blanket bogs in Europe are protected under the Habitats Directive (European Commission, 2019; European Environment Agency, 2019) and EU funding is available to enable restoration and conservation of these ecosystems. However, unrecognised and unprotected areas of blanket bog, a number of which may exist in the gap identified (Figure 1.1), could be under increased threat. There is currently little in the way of legislation or official conservation guidance to prevent anthropogenic damage.

1.4. AIMS

To further our understanding of the number and significance of blanket bogs in the Cantabrian Mountains, this research set out four key aims:

1) To identify and classify currently unrecognised blanket bogs in the gap noted in the Cantabrian Mountains.
2) To estimate the total carbon stored across any newly identified blanket bogs and quantify the current extent of the degradation in the peatland surfaces.
3) To develop an ultra-high resolution method to measure surface change in blanket bogs using a Terrestrial Laser Scanner, and compare the rate of change in exposed peat between restored and unrestored areas.
4) To determine annual and seasonal rates of erosion, peat loss and carbon loss from blanket bogs (restored and unrestored) in Spain to enable the rate of degradation

to be placed in context with the rate of degradation of other blanket bogs across Europe.

1.5. THESIS OUTLINE

The research undertaken in this thesis is placed in the context of previous and current related work in Chapter 2. The distribution and classifications of peatlands are reviewed with particular emphasis on blanket bogs including examples in Spain. Natural and anthropogenic pressures on peatlands have also been reviewed in order to understand the current status of this habitat in Europe and Spain.

The identification and classification of unmapped Spanish blanket bogs is detailed in Chapter 3, mapping the southernmost edge-of-range of this habitat in Europe. Assessment of the current state of degradation and total carbon stored in the newly identified blanket bogs is examined in Chapter 4.

Chapters 5 and 6 focus on determining the rate of surface change, both erosion and deposition, for blanket bogs in the Cantabrian Mountains. The development of a method to measure surface change in ultra high-resolution (mm changes) is presented in Chapter 5 along with the results of a trial undertaken on three blanket bogs. This is expanded in Chapter 6 to report annual and seasonal erosion rates for one restored blanket bog and two unprotected blanket bogs. This provides estimates of the rate of natural and anthropogenically influenced erosion and evaluates the impact of restoration actions in Spanish blanket bogs. Estimates of peat loss and carbon loss presented in Chapter 6 highlight the significance of the work in Chapter 3.

The importance of the newly mapped blanket bogs identified in this research and their relevance in the context of climate change and current degradation rates in discussed in Chapter 7.

Chapter 2

Peatland & mires: Definition, distribution, classification, threats and importance

2.1. PEATLAND ENVIRONMENTS

2.1.1. Definition of peat and peatland

Peat is a type of soil consisting of at least 30% partially decomposed dry organic material (Joosten *et al.*, 2017). An area covered by peat is defined as a peatland, but only classified as a mire when peat forming species are present in the ecosystem (Kivinen, 1980; Immirzi, Maltby and Clymo, 1992; Charman, 2002; Joosten *et al.*, 2017). A peatland is an area where peat accumulated over the land surface (Gorham, 1953; Joosten *et al.*, 2017), most commonly as a consequence of waterlogging and low oxygen conditions (Sjörs, 1948). The thickness of the peat layer, or peat depth, which is required to 'define' a peatland varies between countries and regions. In Europe, peatlands are generally considered as areas where there is a minimum of 30 cm of peat (Kivinen, 1980), but this varies depending of the country. For example, the minimum peat depth required in Ireland is 40 cm (Cruickshank and Tomlinson, 1990; Evans and Warburton, 2007), compared to 20 – 30 cm in Germany (Keppeler, 1922; Schneider, 1976), and 50 cm in Scotland (Bibby, 1984). This highlights an issue in the lack of a standardised peat depth to define peatlands.

Peatlands are assumed to have two main layers, the acrotelm (top layer in contact with the surface) and the catotelm (deep layer in contact with the substrate; Ingram, 1978). The upper layer (acrotelm) is characterised by high rate of conductivity and fluctuations in the water table, which is rich in plant material and aerobic peat-forming bacteria (Ingram, 1978). The lowest water level typically defines the boundary between the acrotelm and the catotelm, the latter of which is saturated with water and contains high levels of partially decomposed organic material (Ingram, 1978). Five main factors that define peatland environments include: climate, geomorphology, geology and soils, biogeography and human activities (Charman, 2002). The interaction between these factors can instigate formation of peat and therefore the development of a peatland environment. There are two main processes for peatland initiation resulting from excess of water: terrestrialisation and paludification (Romell and Heiberg, 1931; Gorham, 1957; Charman, 2002). Terrestrialisation is a process where a shallow water body is slowly filled by organic and inorganic materials. This accumulation is continuous until a point where the water table is above the surface and peat accumulates over the previous deposit (Payette, 2001; Charman, 2002), while paludification is a process where peat accumulates over a wet substrate or mineral soil (Charman, 2002). The majority of peatlands have developed through paludification typically covering far larger areas than those peatlands initiated through terrestrialisation (Sjörs, 1983). Based on the source of water, peatlands can also be divided into two main groups, fens and bogs. If the peatland is rain-fed, it is defined as a bog or ombrotrophic while, if it is groundwater-fed, it is categorised as a fen or minerotrophic peatland environment (Du Rietz, 1954).

2.1.2. Distribution of peatland environments

2.1.2.1. Global distribution

Peatlands cover only 2.84% (4.23 million km^2) of the Earth's land surface (Xu *et al.*, 2018), but represent approximately 20% of the terrestrial organic carbon storage, thus play an important role in the global carbon cycle (Gorham, 1991; Yu *et al.*, 2010). The majority of the global peatland environments are concentrated in the Northern Hemisphere, where the largest proportion lies in Asia (38.4%) and North America (31.6%). In fact, Europe only contains 12.5% of global peatlands followed by South America (11.5%), Africa (4.4%) and Oceania (1.6%) (Xu *et al.*, 2018). Nonetheless, true estimation of the extent of peatlands is difficult and varies between authors (Figure 2.1).

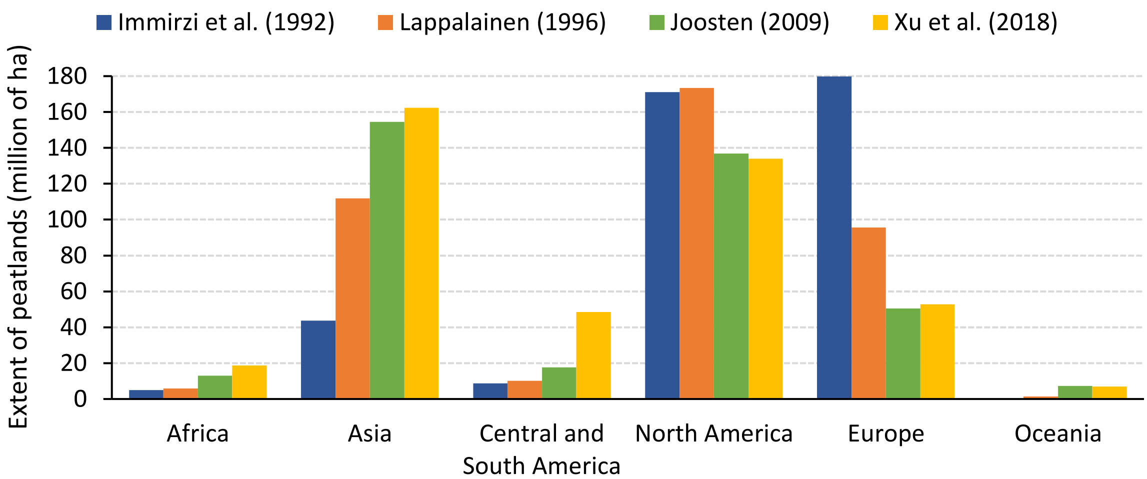


Figure 2.1. Comparison between estimations of peatland extensions by continents. Europe includes former Soviet Union in Immirzi, Maltby and Clymo (1992). Central America has been included as South America or North America in Joosten (2009) and Xu *et al.* (2018).

On a continental scale; Africa, Asia and America (Central and North) have been underestimated in previous inventories, and over time, the reported extent of peatlands has increased 3.8 times greater in Africa, 3.7 times in Asia and 5.5 times in America. However, in the case of Europe and North America, the estimated extent of peatland has reduced by 0.3 and 0.8 times respectively. Oceania shows the most extreme change in peatland extension with 37.3 times increase (Xu *et al.* 2018) in comparison with earlier estimates by Immirzi, Maltby and Clymo (1992). When considering individual countries, Russia is identified as the country with the largest extent of peatland (137.5 Mha), followed by Canada (113.4 Mha), Indonesia (26.6 Mha), the United States (22.5 Mha) and Finland (7.9 Mha) (Joosten, 2009).

The percentage of land covered by peatlands does not necessarily relate to the total extent of peatlands, and this is important for understanding conservation and protection policies. Europe has the highest percentage of land covered by peatlands and requires special attention in terms of conservation; for that reason, the European Union has protected peatlands under the Habitats Directive (92/43/ECC) (see section 2.1.3.5). Countries like Ireland, Finland and Sweden have large extents of peatlands in their territories, and in the case of Ireland, more than the 24% of the land is covered by peatland environments (Figure 2.2), which only amounts to 1.6 Mha of peatland extent.

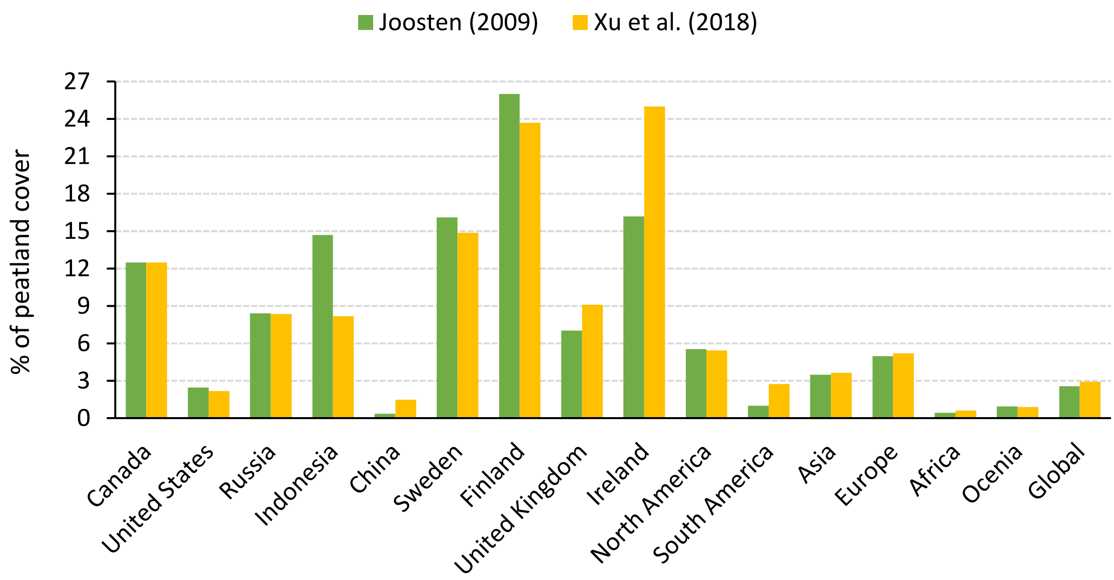


Figure 2.1 Comparison of peatland cover by percentage of land area in differing countries and continents.

2.1.2.2. European distribution

European peatlands only represent 5.2% of the Earth's land surface (Xu *et al.*, 2018); however, in some countries this environment is a very relevant part of the landscape (Figure 2.3). The distribution of peatlands in Europe is largely based on their geographical location and precipitation, which enables them to be divided into ten distinct regions. Moving from high latitudes to low latitudes, a range of habitats can be found: Artic and polygon mire region, Palsa mire regions, Northern fen region, Typical raised bog region, Atlantic bog region, Continental bog and fen region, Nemoral-submeridional fen region, Colchis mire region, Southern European marsh region and Central - southern European mountain compound region (Moen, Joosten and Tanneberger, 2017).

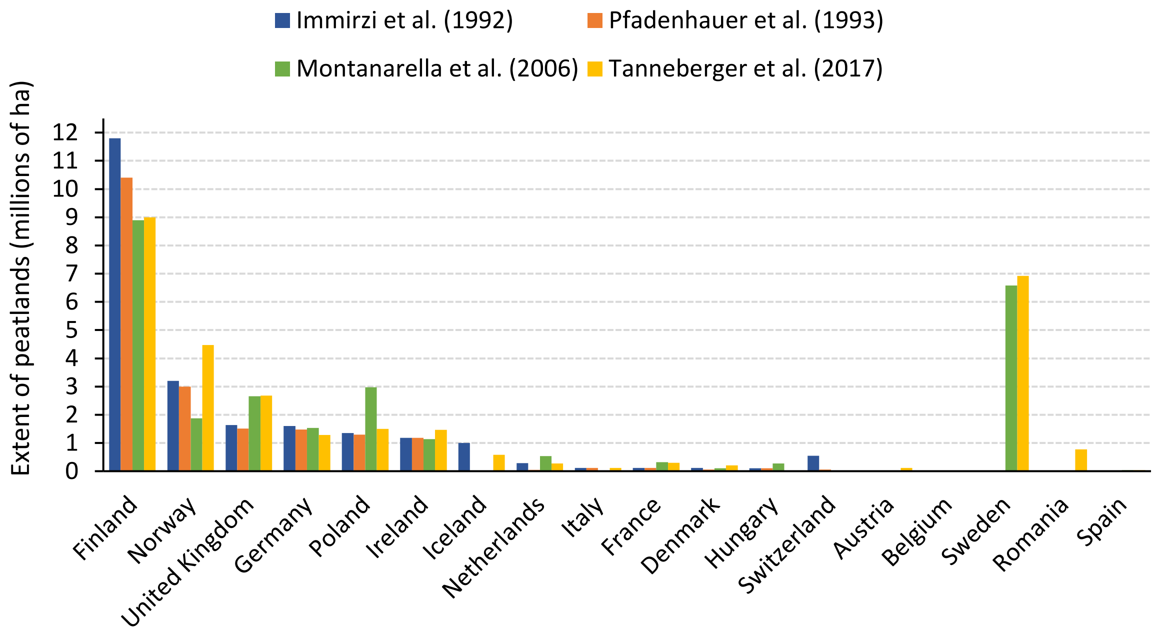


Figure 2.3. Comparison of peatlands extent of the main European countries according different authors.

The reported distribution of peatlands across Europe has changed through time for a variety of reasons; for example, peatlands distribution is now better understood than previously, particularly in light of the EU Habitats Directive that regularised and standardised peatlands types (see section 2.1.3.5), but also factors, such as, changes in country boundaries could have an impact on the total peatland extent by country (e.g. Germany).

Finland has always been noted as the European country with the greatest extent of peatland environments since the 1990s, that amount to almost a third part of the peatlands in Europe (Montanarella, Jones and Hiederer, 2006). However, there has been a clear reduction in peatland extent in this country in recent inventories (Figure 2.3), potentially as a result of peatland degradation. Peatlands are distributed throughout country with a high concentration of bogs in the south (Lindholm and Heikkilä, 2017).

Sweden has the second largest extent of peatlands of all European countries (Tanneberger *et al.*, 2017), although early inventories did not include large extents of peatlands in this country (Immirzi, Maltby and Clymo, 1992; Pfadenhauer *et al.*, 1993). There are two main concentrations of peatlands in Sweden; the first area is located in the north and central areas with a dominance of minerotrophic peatlands (fens) with exception of alpine and coastal areas where peatlands are rare. The second is located in the southwest part of the country where ombrotrophic (bogs) are dominant (Löfroth, 2017).

The final Nordic country with large extensions of peatlands is Norway. Ombrotrophic bogs are widely distributed through the country under oceanic climatic conditions, although drainage has impacted more than 100,000 ha of bogs (Moen, Lyngstad and Øien, 2017). Fens are common in this country, but they are especially noted in oceanic areas (Moen, 1990).

The United Kingdom and Ireland are further countries with a high influence of oceanic climate conditions, and therefore large extents of peatlands. Peatlands also represent a large proportion of the land in these countries covering 30% of the United Kingdom (Lindsay and Clough, 2017) and 16.8% to 20.6% of Ireland (Hammond, 1981; Connolly and Holden, 2009). In the United Kingdom, bogs are more common than fen areas, moreover, a higher concentration of peatlands are noted in Scotland covering 66% of the land, in comparison with 11% in England, 25% in Northern Ireland and 21% in Wales (Lindsay and Clough, 2017). In Ireland bogs are widespread, particularly on the west coast and in central Ireland. Ombrotrophic peatlands are the most extensive type of peatland in the country, with some fen areas located mainly in central Ireland (Hammond, 1981).

Central Europe, Germany and Poland have also large extents of peatland environments. In the case of Germany, fen peatlands are dominant and bogs are restricted to the northwest of the country, where the influence of oceanic climate conditions are stronger (Trepel *et al.*, 2017). In Poland, fens represent 92.4% of peatlands and only 4.3% are bogs with their origin associate to lake systems (Kotowski, Dembek and Pawlikowski, 2017).

In Mediterranean countries such as Italy, France and Spain, peatlands are mainly located in Atlantic, Alpine and Continental regions rather than Mediterranean regions (Bragazza *et al.*, 2017; Heras *et al.*, 2017; Julve and Muller, 2017). Peatlands represent only a small portion of these countries (0.6 – 1.1% of Italy (Montemaggiori, 1996), 0.52% of France and 0.07% of Spain (Tanneberger *et al.*, 2017)).

Finally, countries in Eastern Europe also have extensive areas of peatlands (Ukraine, Belarus, Estonia, Slovenia and Latvia), although Russia (European part) contains the largest area of peatlands, covering between 23 to 68 Mha of land (Tanneberger *et al.*, 2017).

2.1.2.3. Peatland environments in Spain

The climate conditions in Spain are diverse with oceanic climate conditions in the Cantabrian Mountains in the north and a dominance of Mediterranean climate throughout the rest of the country, although high mountain ranges also have an impact on the climate, creating a diverse range of conditions in some parts of the Mediterranean regions (Heras *et al.*, 2017). The country is divided into two biogeographical areas, being the Atlantic and Mediterranean regions (European Topic Centre on Biological Diversity, 2006), with clear differences in annual precipitation. Peatland landscapes are rare in Spain and the majority of peatlands are small and do not cover as large extents as peatlands located in other European countries, such as United Kingdom and Ireland (Figure 2.3), but are still very important in terms of carbon storage and as potential carbon sinks. The total extent of reported peatlands in Spain varies between 4,000 ha (Goodwillie, 1980) to 40,000 ha (Tanneberger *et al.*, 2017), although some gaps in the inventory have been highlighted (Heras *et al.*, 2017; Ramil-Rego *et al.*, 2017; Heras and Infante, 2018), emphasising that a national standard is required to enable better evaluation of the extent and distribution of peatlands in Spain (Heras *et al.*, 2017). The first inventory of peatlands in Spain was completed in 1903 (Calderón, 1903), and until the 1990s there was no systematic approach to inventorying Spanish peatlands. In 1996, standard classification and characterisation was adopted, classifying peatlands based on hydrology, geomorphology and vegetation (Ramil-Rego, Rodríguez-Guitián and Rodríguez-Oubiña, 1996). However, the peatland inventory for Spain remains incomplete across the country (Heras *et al.*, 2017), despite the regulations and protection that the European Habitats Directive 92/43/ECC requires of all EU state members, and continued periodical reviews.

Bogs in Spain are mainly located in the Atlantic regions due to higher precipitation and the influence of the oceanic climate, although some bogs are also occasionally found in the Mediterranean regions (Heras *et al.*, 2017). Fens are distributed across the entire country (Heras *et al.*, 2017), and in many cases bogs are interconnected to fen (minerotrophic) areas.

2.1.3. Types of peatlands

A general classification for peatlands is difficult, mainly because each country has its own classification according to the environment in which the peatland exists. Environmental factors, such as the climate, geology and vegetation, are usually utilised to classify these peatland environments (Joosten *et al.*, 2017), however there is no single classifying factor that covers all peatlands (Lindsay, 2016a). Historically, classification of peatlands was driven by their economic value and potential exploitation, however, more recently classifications are mainly based on three main variables: hydrology, vegetation and geomorphology (Joosten *et al.*, 2017).

2.1.3.1. Classification by water sources

The acidity of water in a peatland has a large impact on the vegetation cover, although this is not the only factor to consider in term of vegetation coverage (Sjörs, 1950). Based on water acidity, peatlands fit into six different classification: Moss, Extreme poor fen, Transitional poor fen, Intermediate fen, Transitional rich fen and Extreme rich fen (Sjörs, 1950). Moss peatlands are usually more acidic and poor in nutrients (pH 3.7 – 4.2) when compared with fens (pH 5.2 – 6.4), although fen areas could range from extremely poor fens (pH 3.8 – 5.0) to extremely rich fens (pH 7 – 8.4) (Sjörs, 1950). Similar classifications were utilised by Gorham (1954) on Swedish peatlands, although this classification introduced the term raised bog for those with low pH (<3.9). In 1990, pH was also adopted to define two groups of peatlands: bogs (<4.6) and fens (>5.8) (Sjörs, 1950; Gorham *et al.*, 1985); however, a transitional group of peatlands was proposed in addition to bogs and fens for those peatlands that are between pH 4.6 and 5.8 (Gorham and Janssens, 1992).

In addition to the acidity, the origin of the water is also a relevant factor for classifying peatlands. If the peatland is rainwater-fed the habitat is defined as ombrotrophic, and if the water sources are from groundwater, the habitat is defined as minerotrophic (Du Rietz, 1954). More recently, the terms *bog* for rainwater-fed and *fen* for groundwater-fed peatlands, have been widely accepted (Lindsay, 2016a).

2.1.3.2. Ecosystem classification

This classification combines different variables, such as water and vegetation, to provide a list of peatland types. Moore and Bellamy (1974) synthesized all peatland classifications based on water sources, chemistry and geomorphology. In 2002, a classification was proposed based on the water source, origin and evolution of the peatland (Joosten and Clarke, 2002), completely overlooking the fact that the peat forming mechanisms may change over time (Lindsay, 2016b). In this classification method, two main groups were defined: ombrogenous (bogs) and geogenous (fens), although fen peatlands were subdivided into three further categories based on where the terrestrialisation occurred: soligenous, for those formed in a moor pool; lithogenous, for peatlands formed in lakes; and thalanssogenous, for peatlands created from water rise in coastal transgression (Joosten and Clarke, 2002).

2.1.3.3. Geomorphological classification and relationship with water origin

Peatlands have also been defined by the topography and landforms of the substrate, as these are both relevant in defining peatland types and functions. The first attempt to classify peatlands according to the geomorphology occurred in 1903, defining three main types of peatland environments: Hochmoor, Flachmoor and Mischmoor. Hochmoor was described as a peatland with a dome, Flachmoor was a flat peatland and Mischmoor was a peatland with domes and flat areas (Weber, 1903). The topographical location and geomorphology is also relevant to classify peatlands, as they can be located in different landscapes, such as hills, slopes, flat areas, channels and basins (Semeniuk and Semeniuk, 1995), and these factors will determine the type of peatland, in addition to the slope, as a primary limitation for peatland development and expansion (Lawrence *et al.*, 2009). Early geomorphological classifications were described by Moore and Bellamy (1974). These authors classified mires and peatlands based on their location in Europe and some geomorphological features, for first time defining very distinctive peatlands such as Valley Bogs, Raised Bogs, Basin Bogs and Blanket Bogs (Moore and Bellamy, 1974).

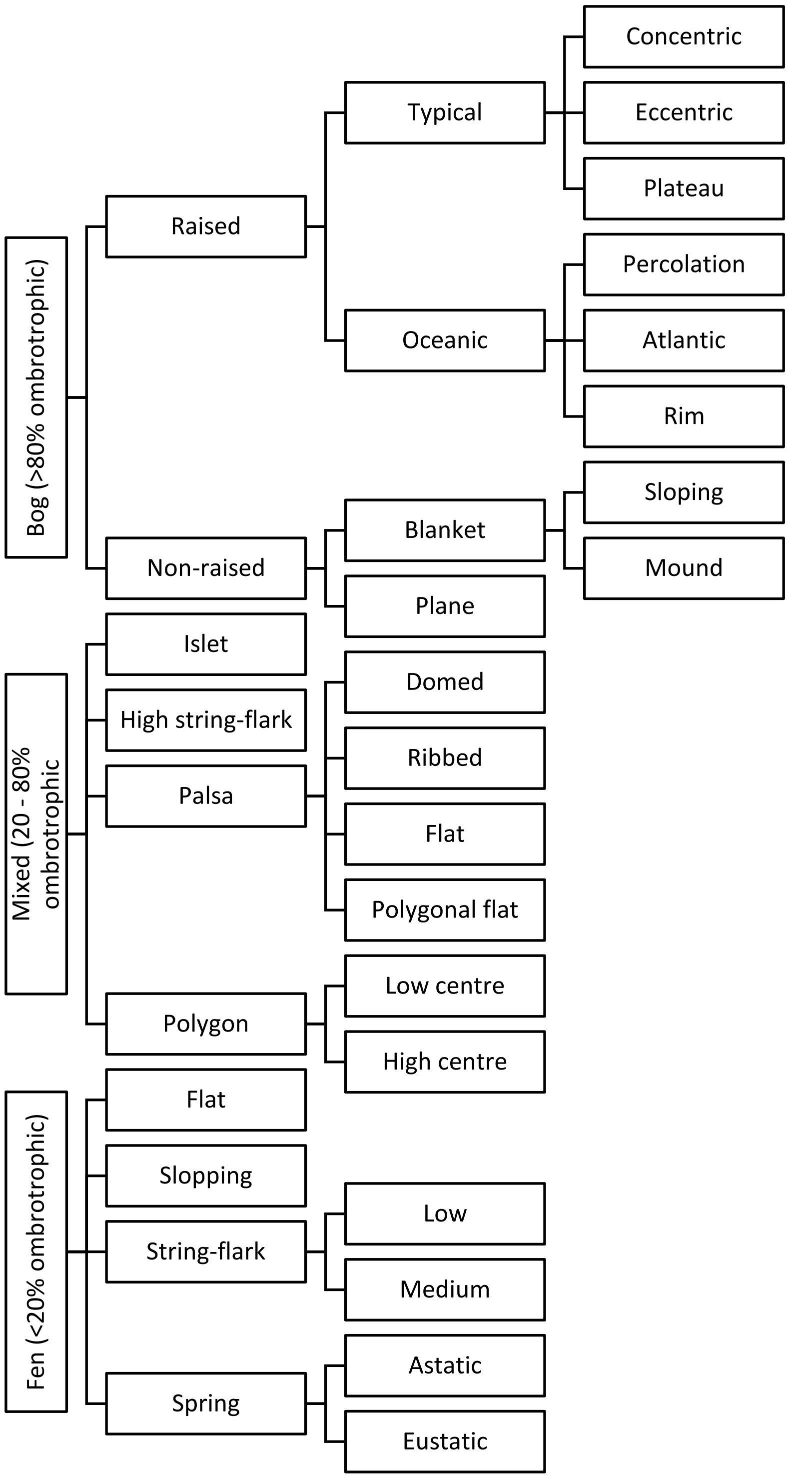


Figure 2.4. Hydromorphic classification of peatlands based on Joosten *et al.* (2017).

When geomorphology is combined with hydrology, peatland classification becomes substantially more efficient and representative (Joosten *et al.*, 2017). This classification has recently been fully described to improve peatland classification across Europe (Figure 2.4; Joosten *et al.*, 2017).

Finally, in addition to the study of the landforms and water sources, geographical scale is also relevant in the correct classification of peatlands due to the different areas and extensions that this environment could cover. Based on scale, Ivanok (1981) defined three main levels to study peatlands: microtope, mesotope and macrotope.

2.1.3.4. Hierarchical classification

Hierarchical classification is based on the scale of analysis, and has different levels of classification, with a specific set of criteria for each level within the peatland area. This classification combines vegetation, landforms and hydrology, to provide a better understanding of the complexity and relationships within the peatland environment. However, to perform this on a larger scale (in detailed surveys) requires a considerable amount of time to cover small areas, largely due to the detailed description required. This method was initially described in 1981, primarily utilising three main levels based on vegetation patterns scale (microtope), hydrological units scale (mesotope) and whole landscape scale (macrotope) (Ivanok, 1981), but this classification has developed over time seeing the introduction of further levels of classification. The current system includes six levels: Supertope, Macrotope, Mesotope, Microtope, Nanotope and Vegetation (Lindsay, 2010). All levels of classification are interconnected; for example, *Sphagnum capillofolium* could form a nanotope, such as an individual hummock. This hummock could be within an area covered in hummocks (microtope) and connected with other microtopes, such as hollows in the top of a ridge comprising a mesotope unit. This mesotope could be connected with other mesotopes, or fen areas at the edge of the peatland forming a macrotope, that could be next to other macrotopes covering the landscape (supertope).

2.1.3.4.1. Vegetation

Vegetation level is where the individual species are the most relevant feature. In some cases, methods of vegetation classification can be used at this level (Lindsay, 2016b).

2.1.3.4.2. Nanotope

This level represents all the small features within the mire/peatland, such as hummocks, tussocks, pools and ridges. Vegetation and water levels play an important role to define the types of nanotopes (Lindsay, 2016b)

2.1.3.4.3. Microtope

This level is defined by repeated surface patters, such as pool systems. This level is usually related with the hydrology of the acrotelm layer and the peatland gradient (Ivanok, 1981; Lindsay, 2016b).

2.1.3.4.4. Mesotopes

This unit is clearly defined by the hydrological boundaries and the water flows. It is the most descriptive form and classification for peatland systems (Lindsay, 2016b)

2.1.3.4.5. Macrotope

This level is when several mesotopes interlink, creating a peatland complex that is completely connected and depends on each other to be a functional peatland. A good example of this, is a raised bog connected to a fen area (Ivanok, 1981; Lindsay, 2016b).

2.1.3.4.6. Supertope

A small-scale level that covers large areas of peatland. Usually refers to the whole landscape, with several macrotopes linked via fens or streams (Joosten and Clarke, 2002; Lindsay, 2016b).

2.1.3.5. European Union Habitat Directive (92/43/EEC) classification

Since 1992, the Habitats Directive 92/43/ECC has provided a clear regulation to promote the maintenance of habitat diversity considering the economic value of the habitats, as well as the social, cultural and regional aspects (European Commission, 2019). As the European Union has grown, the Habitats Directive has been modified to include other habitats existing in new EU member states. For example, in 2004, ten countries joined the EU and these new habitats were added to the inventory to cover the new ecosystems from those countries.

The Habitats Directive protects over 1,000 species and 200 habitat types, based on vegetation distribution and hydromorphology, reporting the status of these habitats in each member state every 6 years. The last report highlighted that more than 80% of peatlands are in bad or inadequate conservation status, mainly as a result of drainage activities in this habitat (European Commission, 2015). Habitats are classified by groups, and all peatland types are in the group designated as “Raised bogs, mires and fens” (Figure 2.5; European Commission, 2013).

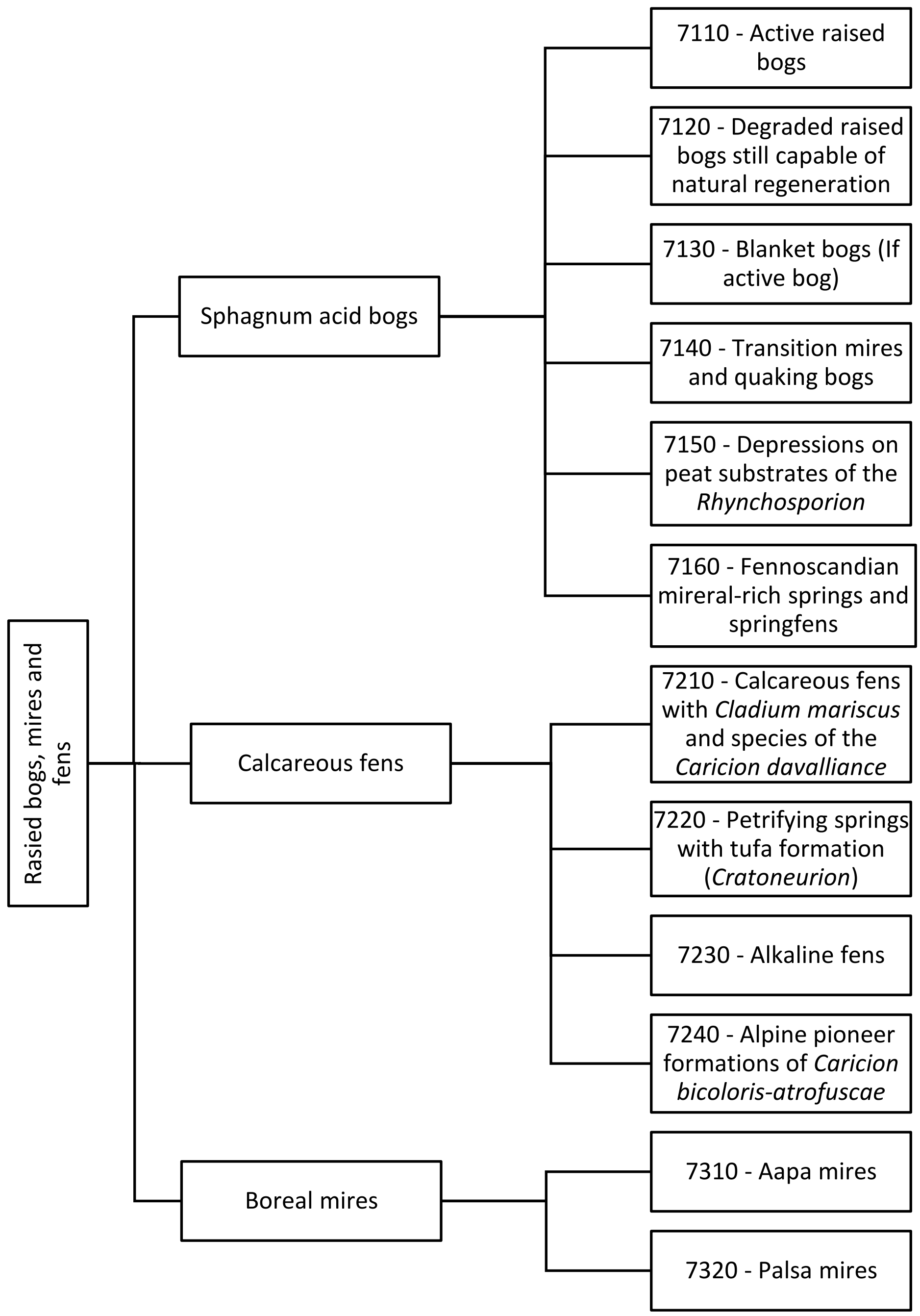


Figure 2.1. Classification of peatlands based on the Habitat Directive 92/43/EEC (European Commission, 2013).

2.1.4. The importance of peatland environments and protection

2.1.4.1. Climate change and peatlands

Peatlands play an important role in the global carbon cycle (Gorham, 1991); in fact, this ecosystem represents the largest terrestrial store of soil carbon (Limpens *et al.*, 2008). If peatlands are in pristine or restored status, they could also act as carbon sinks (Nugent *et al.*, 2018), being the most efficient terrestrial carbon store (Parish *et al.*, 2008). However, a large extent of peatlands has been reported in degraded or damaged status (Figure 2.6; Joosten, 2009) and therefore may now be acting as carbon sources (Yallop *et al.*, 2009). Therefore, their conservation and restoration could help to mitigate climate change in the current global context (Joosten *et al.*, 2016).

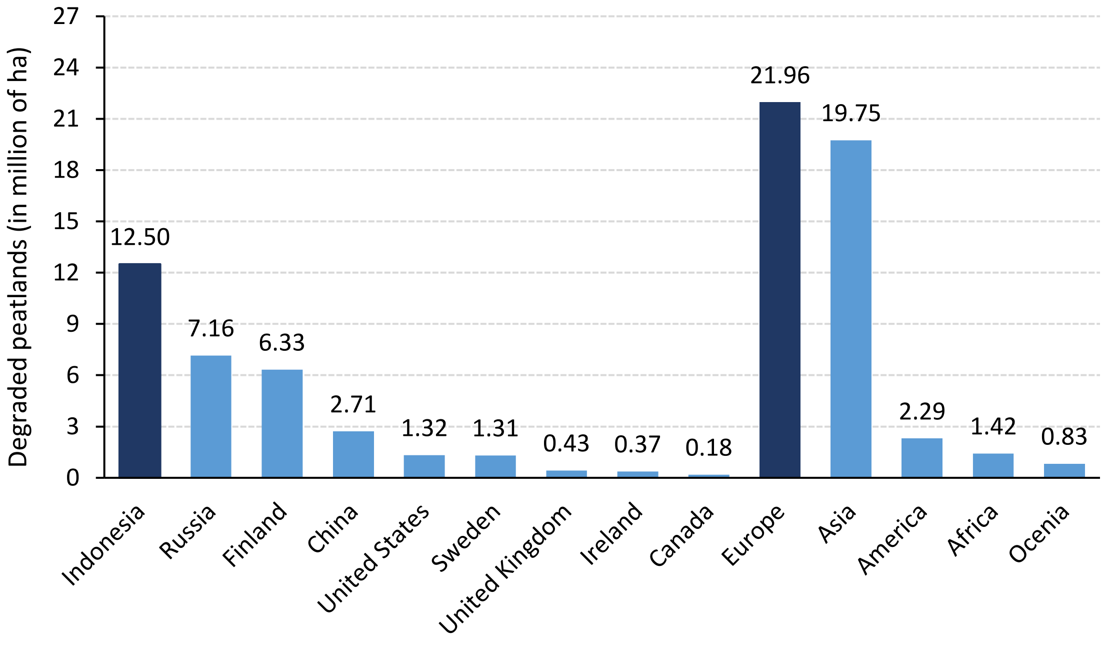


Figure 2.1.1. Extent of degraded peatlands in millons of hectares (Joosten, 2009).

The majority of degraded peatlands are concentrated in Europe and Asia (Figure 2.6). In the case of Europe, Finland (28.8%) and European Russia (28.6%) contain more than 50% of degraded peatlands, although as previously noted, there are large extents of peatlands in both areas. In Asia, 63% of degraded peatlands are reported to be in just one country, Indonesia (Figure 2.6; Joosten, 2009).

The Northern Hemisphere contains the largest amount of peatlands, and therefore the stocks of carbon are higher in this hemisphere, although carbon accumulations are predominantly controlled by the status of the peatland, where pristine peatlands could accumulate more carbon than peatlands under anthropogenic pressures such as burning (Turetsky *et al.*, 2002). Russia has the largest carbon stock in peatlands, with 214 Gt C (Botch *et al.*, 1995). North American peatlands also represent important carbon stores with 191.5 Gt C, and peatlands located in Canada are of particular importance (Bridgham *et al.*, 2006; Joosten, 2009). In Europe, between 200 – 455 Pg C has accumulated during the Holocene (Gorham, 1991; Turunen *et al.*, 2002), but these carbon estimations are dependent on the peat depth, and this has not been well studied in all the European countries (Byrne *et al.*, 2004). The total carbon stored in Europe has been estimated to be at least 43.6 Gt C (Joosten, 2009). Finally, tropical peatlands represent at least 11 to 14 % of the global peatland carbon stocks, ranging from 81.7 to 91.9 Gt C, predominantly concentrated in Indonesia (57.4 Gt C) (Page, Rieley and Banks, 2011).

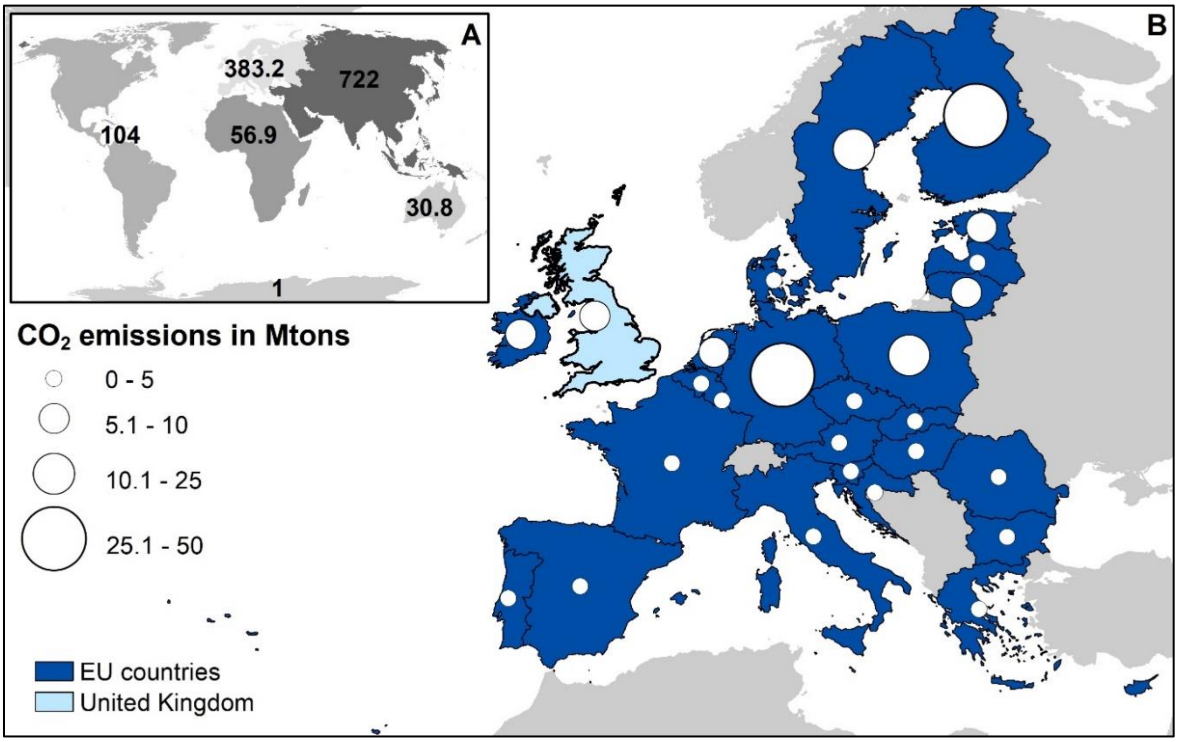


Figure 2.7. A) Global emissions of CO_2 in degraded peatlands by continents. B) CO_2 emissions from degraded peatlands in the European Union countries and United Kingdom (Joosten, 2009). All values for 2008 in Mtons = 1,000,000 tons.

Greenhouse gas (GHG) emissions from degraded peatlands are crucial in the context of global climate change. Asia represents the continent with the highest GHG emissions from degraded peatlands, followed by Europe and America (Figure 2.7A). Within the European

Union countries, Finland and Germany have the highest emissions from degraded peatlands, but emissions from Eastern European countries such as Latvia, Estonia and Lithuania are also significant (Figure 2.7B).

The net change of carbon storage (∆C) of a peatland is complex to define, and there are many variables to study before concluding if a peatland is acting as a carbon sink or source of carbon (Figure 2.8). Firstly, the atmospheric carbon and gaseous exchange needs to be evaluated, mainly studying the CO_2 and CH_4 fluxes (Figure 2.8). Furthermore, dissolved organic carbon (DOC) and particulate organic carbon (POC) are also important in understanding the carbon balance of peatland environments (Figure 2.8; Chapin *et al.*, 2006).

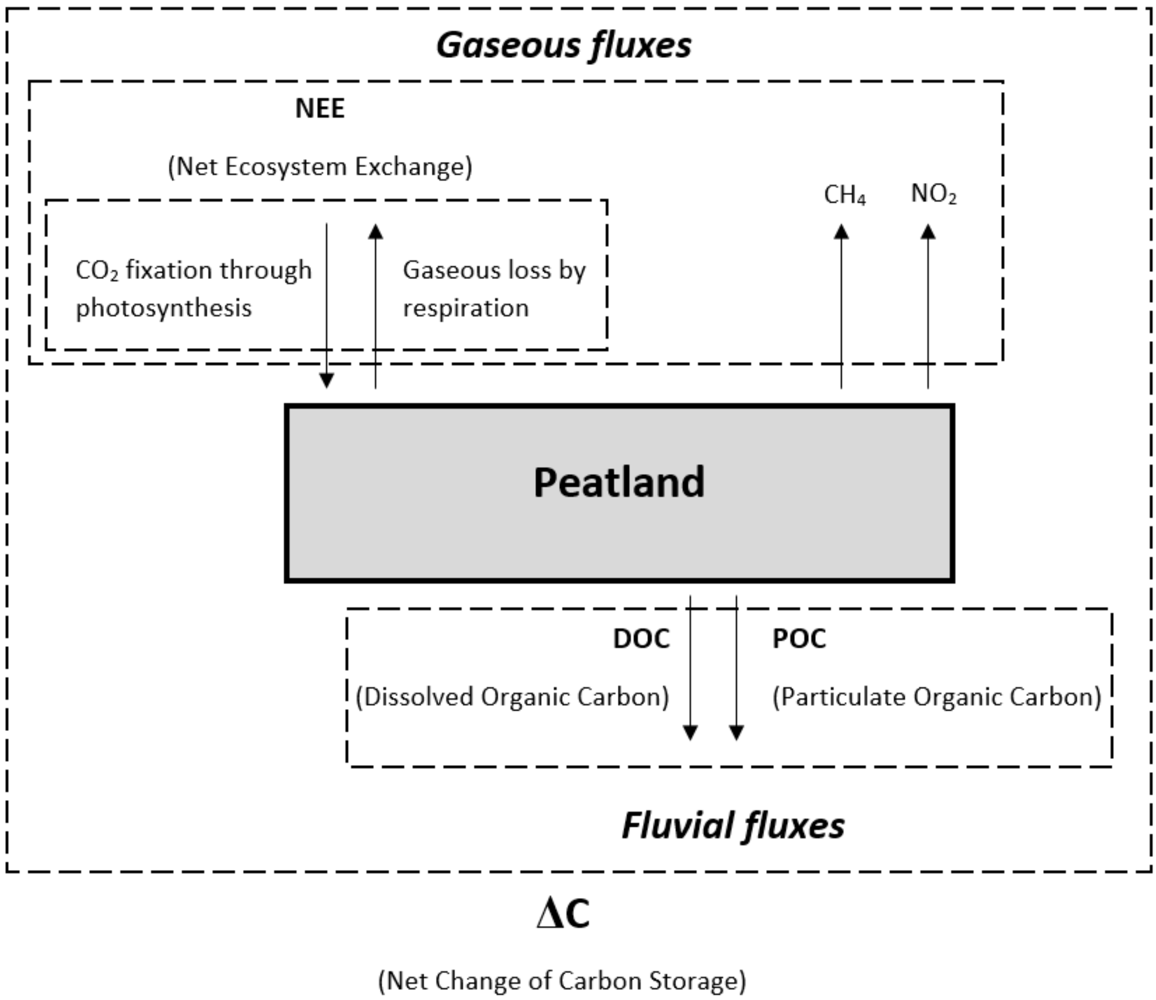


Figure 2.8. Carbon fluxes in peatlands.

The net ecosystem exchange (NEE) essentially represents the balance between the carbon fixed through photosynthesis and the loss through ecosystem respiration (Figure 2.8), and is the most important component when considering the carbon balance of peatlands (73%). The CH_4 fluxes represent 21%, but only 6% is related with DOC flux (Koehler, Sottocornola and Kiely, 2011). In terms of atmospheric gaseous exchanges, peatlands have a strong influence on the quantity and balance of three GHG (CO_2, CH_4 and N_2O), and usually have a positive effect (sink) in the sequestration of CO_2, although on a long term scale, peatlands have a negative effect (source) on CH_4 (Moore and Knowles, 1987, 1989; Parish *et al.*, 2008). If the peatlands are degraded or disturbed, the positive CO_2 balance could turn to a negative contributing to global warming (Parish *et al.*, 2008); in fact, drainage activities in peatlands could affect the total emissions of GHG, since there is a relationship between the water table and GHG (Moore and Knowles, 1989). For instance, natural peatlands or peatlands in pristine status usually act as sinks of carbon (Sottocornola and Kiely, 2005; Roulet *et al.*, 2007), whereas degraded peatlands act as a carbon source (Waddington, Warner and Kennedy, 2002). Interestingly, peatland types could have an impact on GHG capacities; for example, bogs have a higher potential in CO_2 sequestration and lower source of CH_4 when compared with fen peatlands (Parish *et al.*, 2008). Nevertheless, restoration actions could have a positive impact on the GHG sequestration, reversing the source of carbon resulting from degraded peatlands, to carbon sink status (Nugent *et al.*, 2018).

2.1.4.2. Ecosystem services

More recently, peatlands environments have been recognised for the services that they provide. In addition to climate regulation, further services such as agricultural land, peat extraction, field sports or renewable energy, can result from peatland environments, regardless of any positive or negative consequences this may have on their conservation. There is a common international classification of ecosystem services, which is divided into three main sections: Provision, Regulation and Cultural Services (BISE, 2019). In regards to peatlands, provisioning services are mainly related with the use of peat as a fuel, in food production (farming) and fresh water. In terms of regulating services, climate regulation will be the most significant, but also water regulation and purification should not be overlooked. Finally, cultural services related with recreational or educational activities are hugely important (Kimmel and Mander, 2010).

Therefore, peatlands are important for a variety of reasons, such as water quality or prevention of flooding, although in the context of climate change and global warming, caused by GHG, their climate regulation service appears to be the most relevant. Nonetheless, the current degraded status of peatlands across the world could be affecting this natural sink of carbon and they may be acting as a source of carbon, therefore increasing atmospheric GHG. Despite this, restoration and conservation actions could provide a way of mitigating climate change and preserving these carbon stores.

2.2. BLANKET BOGS

2.2.1. Definition and types

There are two main types of bogs (ombrotrophic peatlands): blanket and raised bogs (Figure 2.4; Figure 2.9). The main geomorphological difference between these two types lies in whether they are covering the substrate as a mantle (Figure 2.9; blanket bog), or if the morphology is dominated by the peat accumulation and a dome (Figure 2.9; raised bog) (Weber, 1903; Lindsay, 1995).

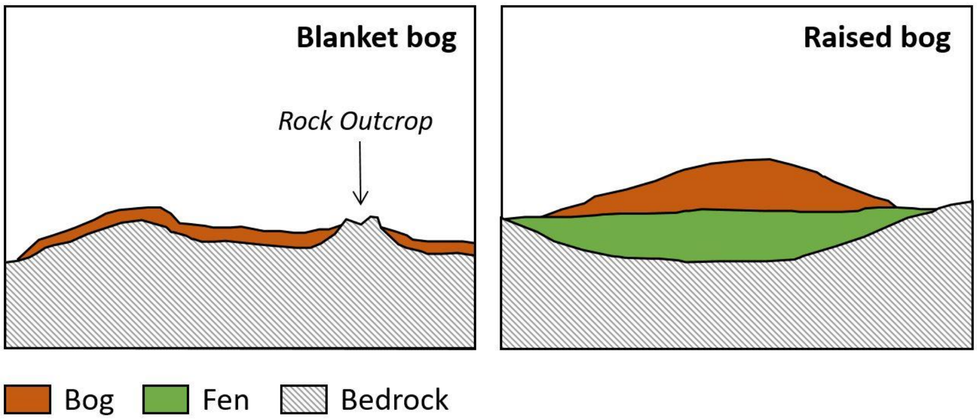


Figure 2.9. Graphical representation of a typical blanket bog and raised bog.

Blanket bogs usually cover large landscapes (Figure 2.9), but can also be relatively small depending on their location, as size may be limited by the topography (Heras, 2002). Blanket bog landscapes often have isolated areas with no-peat (e.g. outcrops, Figure 2.9; Lindsay, 1995) and the peat depth can vary from a few centimetres to 8 m (Lindsay, 1995). The most common origin of blanket bogs is through the paludification process, and landforms within the peatlands are usually related with the topography of the underlying surface (Lindsay, 1995).

The simplest classification of blanket bog is based on their hydromorphic characteristics. Based on this classification, blanket bogs are divided in three main groups: Sloping, Mound and Plane blanket bog (Figure 2.10; Joosten *et al.*, 2017). Sloping blanket bogs have a distinct slope over 3° and erosion channels in the direction of the slope. Mound blanket bogs usually cover hill summits and may have some erosion channels. Finally, Plane blanket bogs are flat or with a very weak slope (Figure 2.10).

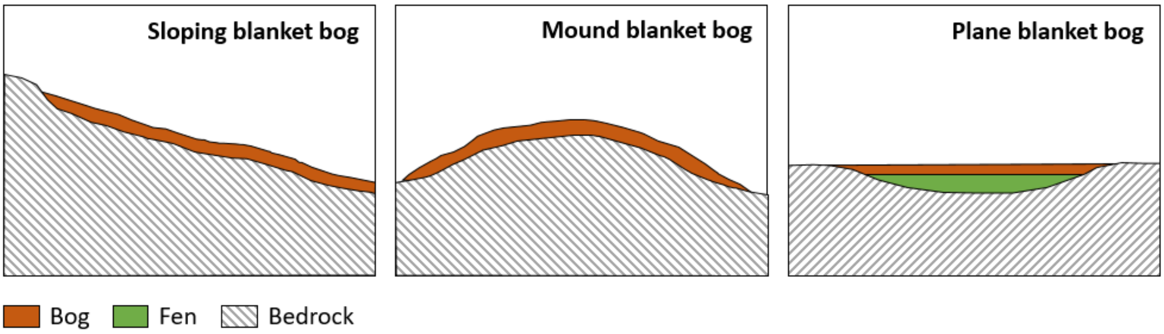


Figure 2.2. Blanket bog macrotope types. Based on Joosten *et al.* (2017).

Hierarchal classification has been developed to classify blanket bogs in more detail, particularly at macrotope and mesotope levels. At macrotope level, classification is the same as hydromorphic classification. At mesotope level, hydrology and location are key factors. Mesotopes are defined largely by water flows and the origin of the water source, although more than 80% of this water must be from direct precipitation to be considered as part of the blanket bog complex, or macrotope. There are four main mesotope types: watershed, saddle mire, spur and valleyside (Figure 2.11).

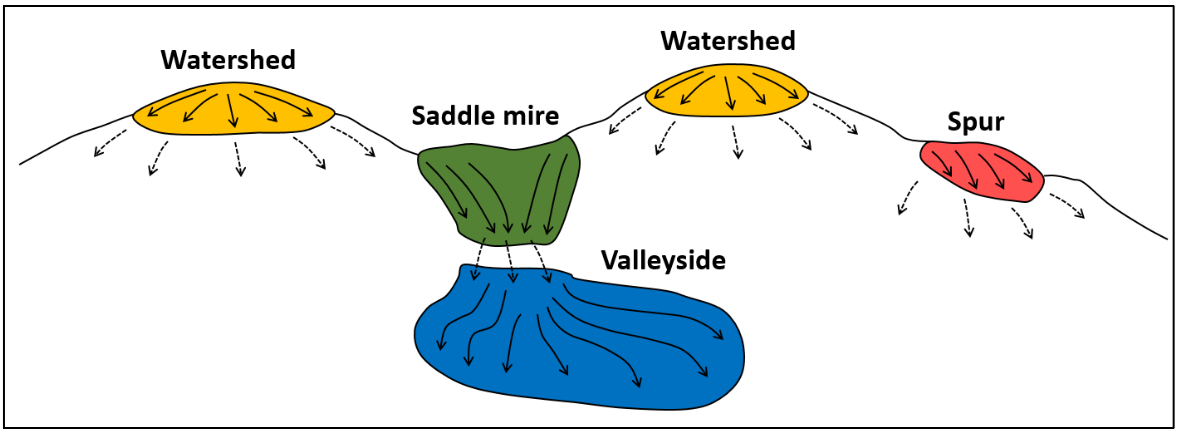


Figure 2.11. Blanket bog mesotope types. Based on Lindsay *et al.* (1988).

2.2.1.1. Watershed

This sub-type of blanket bog, or mesotope unit, occurs on hill summits or areas that slope away in all directions. This mesotope is clearly ombrotrophic because the central area of the unit is higher than the land surrounding it, and the only water source is precipitation. This precipitation could be rain, snow or occult precipitation. Erosion features are common in the mesotope unit in the direction of slope (Lindsay, 1995).

2.2.1.2. Saddle mire

This mesotope is located between two higher elevations, within a depression, and it may receive ground water with some minerotrophic influences in the blanket bog. This type of blanket bog usually develops downhill where two sides meet (Figure 2.11), depending of the slope angle (Lindsay, 1995). Therefore, this bog has two upslope and two downslopes.

2.2.1.3. Spur

This sub-type is a cross between a watershed and saddle mire mesotope. Usually it is located in a watershed area, but with a sloping side higher than the peat bog. Moreover, the central part of the Spur is usually higher than the edges, but will still receive some ground water on one of the blanket bog edges (Figure 2.11). The largest concentration of peat is usually in the uphill slope rather than in the downhill margin, and the downhill edge is generally limited by a steep slope (Lindsay, 1995).

2.2.1.4. Valleyside

This blanket bog sub-type is common on gentle slopes, where a water course or fen is located on the downhill margin. There is generally an uphill margin where ground water can pass through and the peat depth at the top is usually greater than that downhill, where some erosion or inundations may have taken place (Lindsay, 1995).

2.2.2. Distribution of blanket bogs

2.2.2.1. Global context

A global inventory of blanket bogs is difficult to compile due to differences in classification of this habitat (see section 2.1.3) and the lack of research about this habitat, particularly in the Southern Hemisphere. Despite this, based on climatic and terrain conditions, a total amount of 10 million ha of blanket bog has been estimated to exist globally (Lindsay *et al.*, 1988). Blanket bogs are predominantly located in areas with oceanic climate conditions

(Figure 2.12) characterised by high precipitation (>1,000 mm yr^{-1}) and atmospheric moisture content, low average temperatures (<15 °C) and low seasonal temperature variability (Lindsay *et al.*, 1988). In boreal areas, blanket bogs can be covered by snow for several months of the year (Solem, 1994), although the majority have little or no snow cover throughout the year (Doyle, 1997).

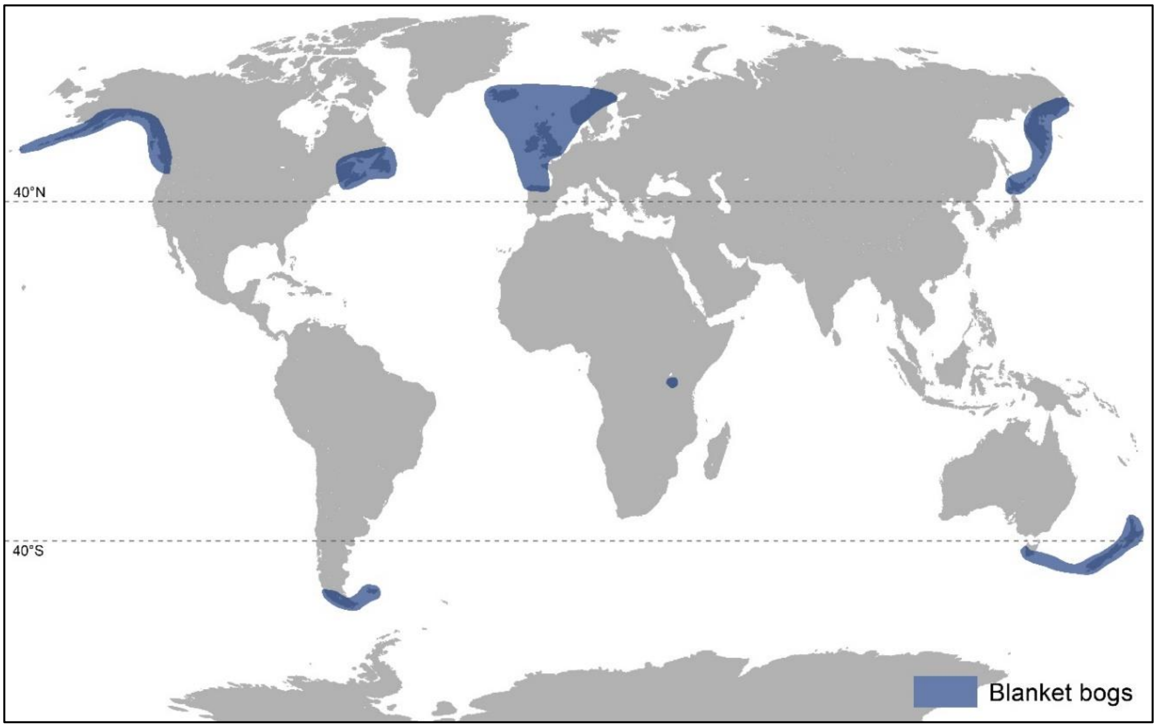


Figure 2.12. Blanket bog distribution across the world (updated from Lindsay *et al.* 1988).

The main areas with known blanket bogs are located above 40° in both hemispheres. In the Northern Hemisphere, North American blanket bogs cover large extents of Alaska (Sjors, 1985), and the west and east coasts of Canada (particularly in Quebec, Terranova and Labrador; Wells and Pollet, 1983; Price, 1992; Graniero and Price, 1999). In Asia, some examples have been described in Japan (Sakaguchi, 2001; Razzhigaeva *et al.*, 2009) and the Kamchatka Peninsula. Europe has a high concentration of this habitat across different countries, with the Atlantic climate conditions being a key factor of their distribution. In the Southern Hemisphere, the majority of blanket bogs are located in Tierra del Fuego, between Argentina (Dykes and Selkirk-Bell, 2010) and Chile (Kleinebecker, Hölzel and Vogel, 2007), although further areas have been described in South Australia, Tasmania (Jeschke and Succow, 2004) and New Zealand (McGlone, Mark and Bell, 1994). However, the research published regarding blanket bogs is limited when compared to the significant distribution of this rare environment, therefore, the global distribution of blanket bogs are well elucidated in some areas, while other areas are yet to be fully investigated (e.g. Spain).

Interestingly, research on blanket bogs (Figure 2.13) has largely been focused on examples in the Northern Hemisphere, particularly the United Kingdom and Ireland. These two countries comprise more than 84% of research conducted (Scopus, 2019; Figure 2.13). This is not surprising, as the majority of peatland research is concentrated in Europe (van Bellen and Larivière, 2020).

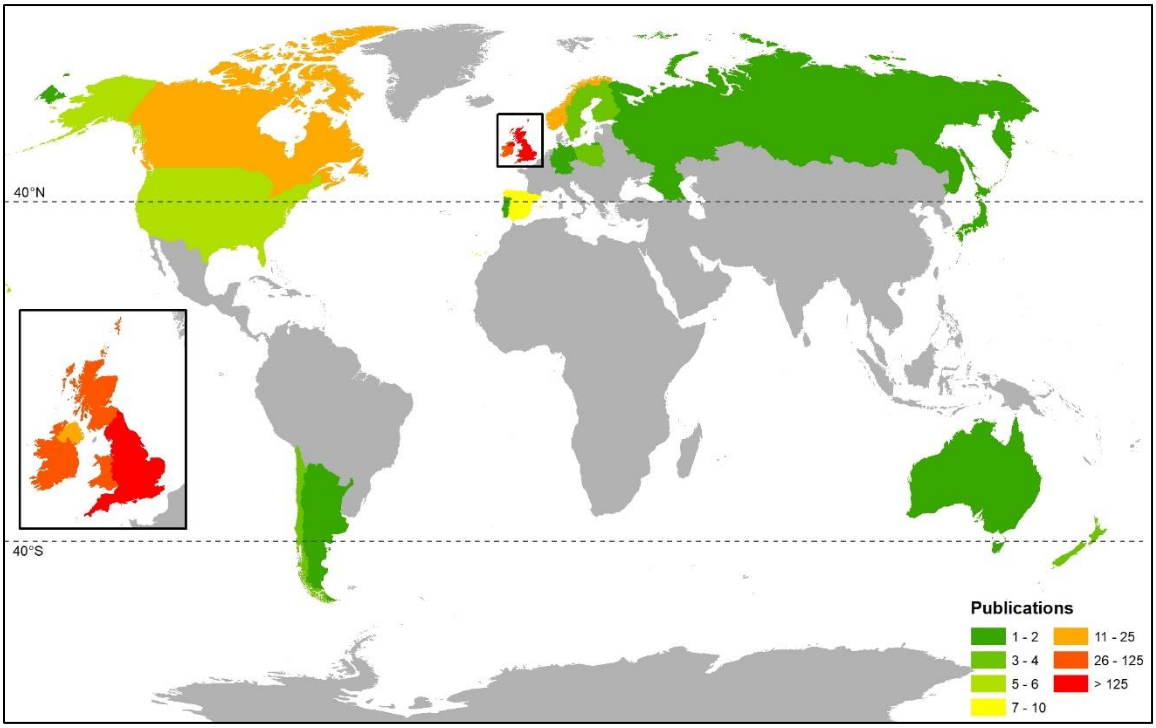


Figure 2.13. Global map outlining the quantity of research on blanket bogs by country (Scopus 2019).

Since 1956, a total of 480 studies have been published on blanket bogs, blanket mires or blanket peats across the world, according to the Scopus database in 2019. The majority of blanket bog research has focused on peatland origin and general descriptions of the geomorphology, soils and hydrology. Vegetation in peatlands has also been an important research topic, as well as anthropogenic impacts on peatland environments and their effects on degradation process and conservation. In the last 20 years, research on climate change and carbon in blanket bogs has increased their global importance since this habitat is a type of peatland and therefore an important carbon store and potential carbon sink (Figure 2.14; Figure 2.15).

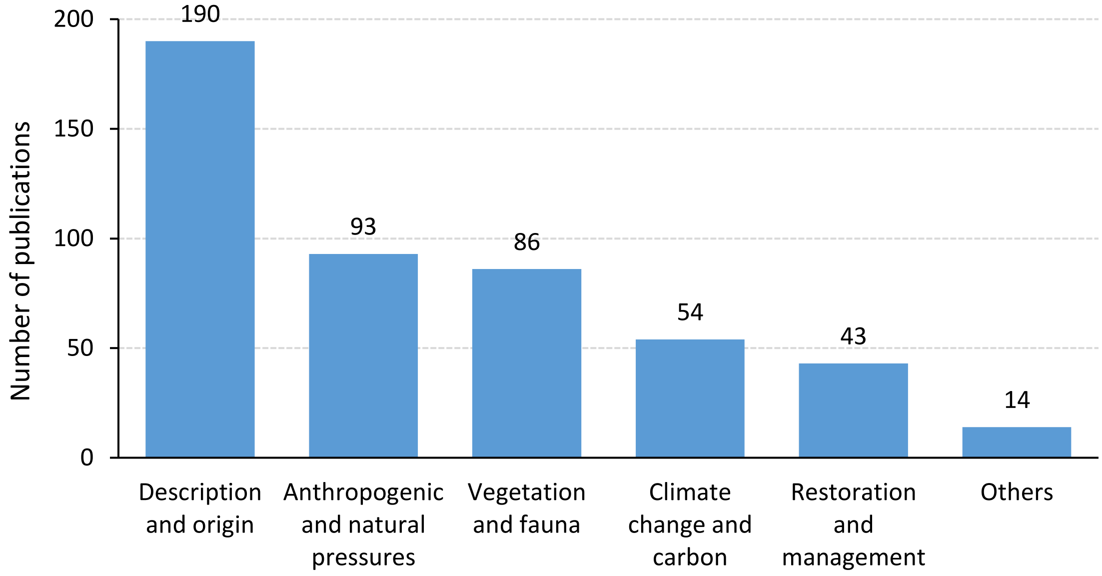


Figure 2.14. Number of research papers published about blanket bogs from 1956 to 2019 (Scopus, 2019).

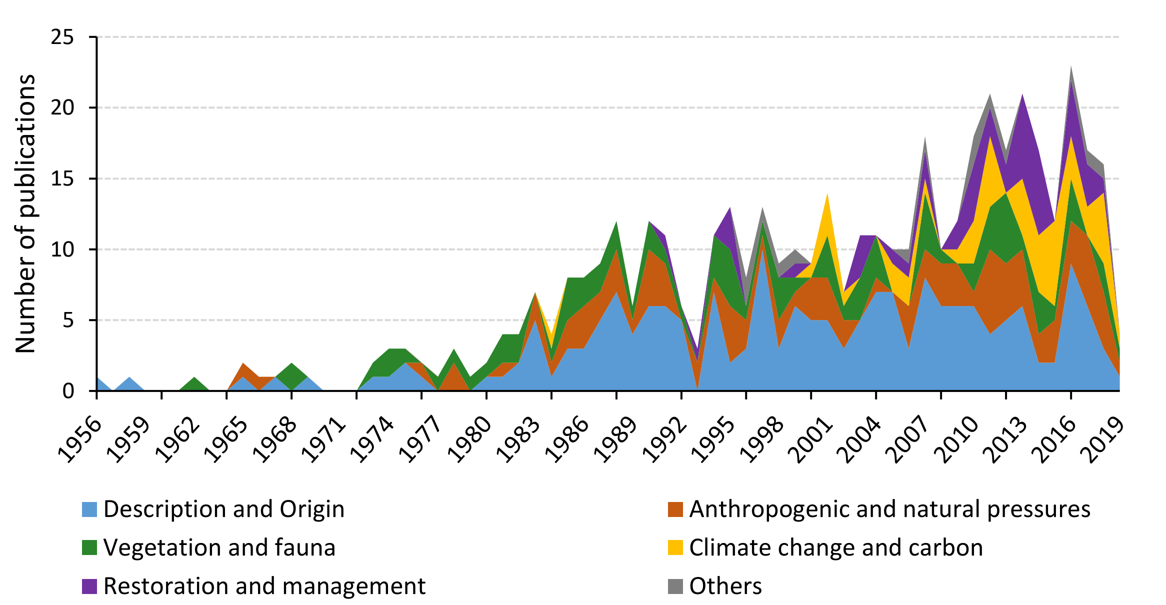


Figure 2.15. Evolution of the peatlands related research from 1956 to April 2019.

2.2.2.2. European and Spanish blanket bogs

Blanket bogs in Europe are mainly located in the United Kingdom, Ireland and Norway (Lindsay, 1995) with limited occurrence in Sweden, France, Spain, Portugal and Austria

(Moen, Joosten and Tanneberger, 2017). They are mainly limited by climatic conditions described before, and therefore tend to be concentrated in areas with a high or markedly oceanic climate (Lindsay *et al.*, 1988), although some recognised blanket bogs in Austria are located in alpine environments (European Environment Agency, 2019).

In the United Kingdom, blanket bogs are widespread across the country, mainly located in Scotland, Wales, Northern Ireland and the north of England, covering 2.28 Mha (Lindsay and Clough, 2017) and represent 90% of the peatlands in these countries (Bain *et al.*, 2011). Further blanket bogs have also been noted on hill summits in the Isle of Man (Lindsay and Clough, 2017).

In Ireland, blanket bogs are the most extensive peatland type, covering 7,739 km^2 predominantly in upland areas and on the western coast across Galway, Kerry, Mayo and Donegal (Foss and O'Connell, 2017) with a high presence of raised bogs. In this country, peatlands are separated into two main groups, lowland/Atlantic and mountain blanket bogs. Lowland blanket bogs are mainly located in the western part of the country, where precipitation is higher and oceanic climate conditions are stronger, particularly in the county of West Mayo (Hammond, 1981). Mountain blanket bogs are more widely distributed across Ireland, but are also located mainly in the western and northern areas (Hammond, 1981). It has recently been highlighted that over 2,287 km^2 of Irish blanket bogs are in an unfavourable conservation status (Foss and O'Connell, 2017).

Another country of particular interest in regards to blanket bog habitat is Norway, although the protection and designation is different to the rest of Europe as the EU Habitats Directive policy does not apply to this country. Therefore, they are classified into two main categories; mound (rare) and sloping (common) blanket bogs at macrotope level. They are mainly located along the most oceanic parts of the country (Moen, Lyngstad and Øien, 2017).

Portugal has a few examples of blanket bog habitat; however, despite the oceanic climate in the north of Portugal, all blanket bogs are located in the Azores archipelago in the Atlantic Ocean rather that in continental Europe. Blanket bogs only cover a small area in comparison with other countries and form an important part of the Azores landscape, despite the fact that more than 50% of the peatlands on this island have been destroyed (Mendes and Dias, 2017).

France, Austria and Sweden only have a few recognised blanket bogs in their inventories. The total extent of this habitat is very low, but with a great diversity. France contains the largest quantity of this habitat covering 4.3 km^2 of Britannia and the Pyrenees (Julve and Muller, 2017), followed by Austria with 1.6 km^2 of blanket bogs in pristine status (Essl and Steiner, 2017) and only 1 km^2 of Sweden is covered by blanket bogs, although these are reported to be in a favourable conservation status (Löfroth, 2017).

Spanish blanket bogs are very limited by climatic conditions and topography. They are usually located at high elevations between 600 and 1,500 masl in the Atlantic climate region with up to 5 m of peat accumulation (Pontevedra-Pombal *et al.*, 2009; Heras *et al.*, 2017). However, some protected blanket bogs with lower elevations are located close to the coast in Asturias at Sierra Plana de la Borbolla (Heras *et al.*, 2017; Ramil-Rego *et al.*, 2017; European Commission, 2019). The majority of the protected Spanish blanket bogs are located in Galicia, covering extensive areas in Serra do Xistral, Montes do Buio and Macizo da Toxiza with an approximate extent of 770 ha and representing the south-western European boundary of this habitat in the continent (Heras *et al.*, 2017). According to the current inventory included in the Habitats Directive, another administrative region with a high number of blanket bogs is Asturias (European Commission, 2019); however, recent research has suggested that several blanket bogs here have been misclassified, reducing the total number of blanket bogs to a few locations at Sierra Plana de la Borbolla (Ramil-Rego *et al.*, 2017). The final protected and recognised blanket bog is located between the administrative regions of Castilla y León and Basque Country: Zalama blanket bog (Heras, 2002). This peatland, protected under the Habitats Directive and recently restored, represents the current southernmost edge-of-range of this habitat in Europe.

2.2.3. Threats and blanket bog uses

2.2.3.1. Natural pressures in blanket bog areas

Environmental conditions and wildfires both represent pressures on blanket bogs surfaces, although these natural phenomena are usually closely related with anthropogenic pressures (see section 2.2.3.2). Moreover, peat is also removed from exposed peat surfaces through natural processes, driven by water, ice, wind and chemical oxidation (Evans and Warburton, 2007). In the following section, the main natural erosion processes caused by water and wind will be described and an overview of wildfires will be given, to define their importance in fire management activities, such as rotational or prescribed burning which is discussed in further detail later (see section 2.2.3.2.4).

2.2.3.1.1. Wind erosion

Wind erosion is a natural process which occurs when the wind picks up loose surface material and transports it (Wilson and Cooke, 1980). Peat is very prone to wind erosion due to its low density (Warburton, 2003), hence a number of studies have highlighted the importance of this process in peatland environments (Bower, 1961; Zuidhoff, 2002). Wind erosion has been recognised as an important natural pressure of peatlands as early as the XIX Century (Rastall and Smith, 1906; Samuelsson, 1910; Bower, 1960), but was not directly quantified until 2002 on degraded peatlands in Canada (Campbell, Lavoie and Rochefort, 2002), and one year later on semi-natural blanket peatlands in the United Kingdom (Warburton, 2003). Moreover, measuring wind erosion as an independent process presents difficulties, as this process is usually combined with further environmental variables, such as precipitation, defining erosion in two different processes: aeolian transport of dry particles, and crust and wind-assisted splash transport under oblique rain (Warburton, 2003; Evans and Warburton, 2007). Both of these processes are strongly related with weather patterns (Foulds and Warburton, 2007a, 2007b). Once the peat is eroded, it is transported by different aeolian processes. Under dry conditions with a desiccated, cracked and crusted surface, processes including saltation, creep, suspension, reputation and kite transport could take place. Under 'wind - rain splash' conditions, all the aeolian processes described for dry conditions could also be a way of transporting the peat, with the exception of kite transport. Furthermore, rain splash and wash could also contribute to the transport processes under wet and windy conditions (Evans and Warburton, 2007). Therefore, wind erosion is a clear aeolian process that affects exposed areas of peat within the peatland environments.

2.2.3.1.2. Fluvial erosion

The location and topography of blanket bogs (Figure 2.9) makes this type of peatland vulnerable to fluvial erosion (Kløve, 1998). Furthermore, these areas of peatlands have high precipitation regimes and are located in oceanic areas (Lindsay *et al.*, 1988), which in combination with the low infiltration capacity of peat, produces high quantities of runoff, thus impacting erosion rates and sediment transport (Evans and Warburton, 2007). Fluvial erosion in peatlands can occur through three different processes: dissection systems, sheet erosion and marginal face erosion (Bower, 1960). In blanket bogs, dissection systems are the most important fluvial erosion process and can be sub-divided into two different types (Bower, 1961). Type I erosion is the most common process in deep peat on flat areas with a well-developed gully system. Type II is generally less developed in comparison with Type I, and individual gullies are the main characteristic (Bower, 1961). Sheet erosion is also highly relevant to bare peat surfaces, often combined with wind erosion (Bower, 1961). The third important fluvial erosion process occurs on the marginal faces of peatlands where the terrain is more abrupt and causes mass movement of peat (Bower, 1960). Another important element to consider in fluvial erosion is peat pipes, usually located in the transition between the peat body and substrate (Holden and Burt, 2002).

Sediment flux of fine-grained peat particles from fluvial erosion is key to understanding the importance of this process in peatland environments. Early research demonstrated a relationship between the sediment flux and storm events, where water flows are higher and therefore increase the total sediment (peat) transported (Crisp, 1966). However, in the 1980s, some studies highlighted the importance of summer desiccation in autumn (Francis, 1990) and frost action in winter, as causes of the main sediment fluxes (Labadz, Burt and Potter, 1991). However, location and climatic conditions need to be considered to understand the main reasons for increasing sediment fluxes (Evans and Warburton, 2007), although researchers agree that the weathering processes before a storm event is key to understanding sediment fluxes rather than the direct fluvial erosion (Francis, 1990; Labadz, Burt and Potter, 1991). More recently, the role of needle ice erosion impacting on the stability of peatland surfaces has been highlighted, as well as the significant impact this has on erosion rates and sediment fluxes (Li, Holden and Grayson, 2018).

Another fluvial erosional process known as 'peat blocks' usually occurs in active gully systems where large blocks are cut from the gully banks, which subsequently collapse into the channel within the gully. It may also happen when the margin of the peatland is near to a stream (Evans and Warburton, 2007). These blocks can contribute to sediment fluxes, particularly when they are deposited within the main stream and water flow due to rapid abrasion (Evans and Warburton, 2007).

2.2.3.1.3. Wildfires

It is important to highlight the differences between wildfires and prescribed fires or burning. Wildfires are unplanned, whereas prescribed burning is usually premeditated and controlled to lower the risk of damage. Anderson *et al.* (1989) defined two categories of fires: cool fires and hot fires; cool fires refer to prescribed burning, and hot fires refer to wildfires. Cool fires will be described in detail later as they are related with anthropogenic pressures (see section 2.2.3.2.4). Hot fires are not controlled and tend to occur during the summer and dry season. They can cover large extensions with high intensity and severity (Davies and Legg, 2008). The initiation of these fires is commonly human related, mainly as a consequence of negligence (Tedim, Xanthopoulos and Leone, 2015), potentially resulting in peat burning for several weeks (Anderson, Radford and Mackenzie, 1989); however, it can also be initiated by natural causes, such as lightning, although this represents a very small proportion of the total fires (Tedim, Xanthopoulos and Leone, 2015). Hot fires have consequences on the peatland because they expose the top layers of peat to oxidation (Lindsay *et al.*, 1988), and consequently alter vegetation and peat erosion (Yeloff, 2001). In fact, the role of hot fires in the initiation of peat erosion has been well studied since 1965 and in several areas across the United Kingdom (Radley, 1965; Tallis, 1987; Anderson, Radford and Mackenzie, 1989), moreover, it has been postulated that this initiates or at least increases, peat erosion in Spanish blanket bogs (Heras, 2002).

2.2.3.2. Anthropogenic pressures and impacts on blanket bogs

There is a close relationship between humans and peatland environments; in fact, ecosystem conservation is usually related to the human services provided for this environment (see section 2.1.4.2). Drainage of peatlands appears to be the key pressure, as it is needed in order to transform peatlands and enable other uses, such as agricultural land, grazing, peat extraction or forestry. Other pressures, such as rotational burning are also relevant to the majority of blanket peatlands within the United Kingdom, in order to provide another ecosystem service: hunting (Yeloff, 2001; Yallop *et al.,* 2006). More recently, windfarm developments represent a challenge for the conservation of blanket mires globally, but particularly in the United Kingdom, Ireland (Lindsay, 2016c; Wawrzyczek *et al.*, 2018) and Spain (Heras and Infante, 2008). These aforementioned pressures lead to diverse consequences described in this section, although the ultimate issue lies in peat and carbon loss through erosion processes instigated by anthropogenic pressures.

2.2.3.2.1. Drainage

Artificial drainage has been a pressure in peatlands for centuries (Holden, Chapman and Labadz, 2004), undertaken to use the peat as an energy source, expand agricultural land, the forestry industry and horticultural purposes (Armentano and Menges, 1986). Artificial drainage has been very important in countries such as Ireland or United Kingdom, where peatlands have played an important role in farming (Williams, 1995), and are locations with more extensive drainage system in the land (Baldock, 1984). Drainage systems usually consist of ditches of 50 cm deep and 50 to 70 cm wide across the peat body (Armstrong *et al.*, 2009). The impacts of drainage could affect a number of variables, such as catchment hydrology (water tables), soil properties, water chemistry and erosion rates (Holden, Chapman and Labadz, 2004), but also the peatlands functionality and geomorphology, with processes such as subsidence of the peatland body (Lindsay, Birnie and Clough, 2014c) which in some cases, has been up to 5 – 6 m (Heathwaite, Gottlich and Cooke, 1993).

Peatlands catchments are defined by hydrological patterns; however, if drainage systems are introduced, the hydrological patterns are altered, impacting the catchment. Hydrological responses to artificial drainage have been studied for decades with a variety of aims, such as the effects on peatland water storage or annual runoff (Holden, Chapman and Labadz, 2004). Furthermore, soil properties can also be affected by peatland drainage. In this scenario, the acrotelm could change its properties, thus changing the peat-forming vegetation to drought resilient species (Lindsay, Birnie and Clough, 2014c). Changes to soil properties could also have an impact on water chemistry, short term studies have demonstrated the impacts of drainage on solute concentrations (Holden, Chapman and Labadz, 2004).

Drainage also plays a paramount role in GHG emissions. This anthropogenic pressure could produce up to 8.5 Tg yr^{-1} of CO_2 (Gorham, 1991), moreover, in cutover peatlands, the emissions of CO_2 are three times greater than that of natural peatlands (Warner, 1999; Waddington, Warner and Kennedy, 2002); however, restoration of peatlands with drainage systems could easily revert this situation and re-establish the sink function of the peatland within 6 to 10 years (Waddington, Strack and Greenwood, 2010).

Finally, soil pipes can develop as consequence of peatland drainage, increasing the particulate carbon loss (Holden, 2006), although it has been noted that soil pipes are also a natural geomorphological landform of peatland environments (Jones, 2004).

2.2.3.2.2. Peat cutting and extraction

The use of peat as fuel and in horticulture has been an anthropogenic pressure on peatlands since at least the Neolithic in the Stone Age (Joosten and Tanneberger, 2017). Peat only has half of the heating value of coal, but is easier to store and is extremely easy to obtain (Asplund, 1996; Gerding, Karel and de Vries, 2015). The first country in which large scale peat extraction for fuel occurred was The Netherlands, and in 1859, the energy obtained from peat was similar to that obtained from coal, although by 1939 peat only represented 3% of the national demand (Gerding, Karel and de Vries, 2015). On a global scale, the use of extracted peat as fuel represents 50% of total peat extraction (Joosten and Clarke, 2002), although nowadays, the main use of extracted peat is for agricultural and horticulture purposes. In some countries, such as United Kingdom, governments are encouraging the industry to replace peat for other more suitable materials (Alexander *et al.*, 2008), highlighting the importance of peat conservation for climate regulation and ecosystem services (see section 2.1.4.2).

Peat extraction in blanket bogs is less important than in another types of peatlands; however, the impact is greater as blanket bogs are rare and the rate of peat accumulation is lower in comparison with other peatland types. Despite this, in the United Kingdom, large extractions of peat occur in blanket bogs as they represent large areas of the landscape, in which the peat is primarily used for horticultural purposes (Lindsay, Birnie and Clough, 2014a). In Spain, blanket bogs are located in remote areas, where historically peat extraction did not occur until the 1940s (Heras *et al.*, 2017). In 2012, a total of 61.4 thousand tonnes of peat was extracted, mainly for horticultural and gardening purposes (Instituto Geológico y Minero de España, 1978; Heras *et al.*, 2017) and this included extraction from some important raised and blanket bogs such as Tornos and Saldropo, which have been completely destroyed as a result (Figure 2.16; Heras and Infante, 2008). Moreover, as a result of peat extraction, the southernmost edge-of-range of blanket bogs in Europe has been destroyed by this activity.

Figure 2.16. Photograph of Tornos blanket bog (Cantabria, Spain) after peat extraction.

The impacts associated with peat extraction are multiple and can be difficult to quantify. In the short term, removing vegetation and creating a drainage systems has a direct impact in the peat-forming system and on hydrology by changing groundwater patterns (Ingram, 1992). Furthermore, additional discharges of DOC and GHG may also occur in degraded peatlands (Waddington, Warner and Kennedy, 2002).

2.2.3.2.3. Forestry

Many peatlands have been commercially forested, including blanket bogs post-drainage in the United Kingdom and Ireland (Holden *et al.*, 2007), but also in other countries, such as large number of peatlands in Nordic countries, Canada and Russia (Rydin and Jeglum, 2006b). It is estimated that 15 million ha has been affected by this activity (Paavilainen and Päivänen, 1995), and that 20% of European peatlands have been drained for forestry (Drosler *et al.*, 2008). For this activity, larger ditches are required and fertilisation is also necessary in order to plant trees and grow a successful 'crop' (Holden *et al.*, 2007). The main consequences of drainage have been described previously (see section 2.2.3.2.1); for example, subsidence, although this is a lesser impact compared with peatlands drained for agricultural purposes (Minkkinen and Laine, 1998). Changes in the carbon balance of the peatland through GHG exchange could also be relevant in forestry areas (Rydin and Jeglum, 2006a), but may not be a negative impact in terms of carbon exchange (Minkkinen *et al.*, 2002), as some can remain to function as carbon sinks if there are only small changes in the water tables (Minkkinen *et al.*, 2018). A study in Irish peatlands suggested that carbon loss, as consequence of drainage and tree plantation, is compensated by the trees carbon uptake (Byrne and Farrell, 2005); however, DOC or POC were not considered in this research, suggesting that if these variables were considered, a source of carbon would be observed in the final results (Sloan *et al.*, 2018). Further research is needed to evaluate the impact of forestry in peatland environments in terms of carbon balance.

Drainage for forestry can also have consequences on the vegetation as the peat becomes drier. Peat-forming species, such as *Sphagnum* spp., could see more than a 75% reduction, and be replaced by drier mosses that require less water, such as *Polytrichum commune* (Rydin and Jeglum, 2006b).

2.2.3.2.4. Burning

Burning is a common practice in peatland environments; however, in this case, fires or 'cool fires' as Anderson *et al.* (1989) denominated, are controlled and are intended to only burn the vegetation cover; particularly *Calluna vulgaris,* for grazing and grouse activities (Yeloff, 2001; Yallop *et al.,* 2006). Burning is restricted to certain times of the year in order to protect the wildlife nesting season and avoid 'hot fires' due to non-favourable climate conditions (Clay, Worrall and Fraser, 2010). There is a long debate about the pros and cons of prescribed burning as this activity could produce several issues in peatland environments, such as changes in the water table, water quality, vegetation, as well as affecting carbon stores and sequestration, but may also bring beneficial outcomes, such as diversity, ecosystem function and reduce the risk of wildfires. In addition, burning practices could provide a range of ecosystem services.

Deeper water tables and higher variability was found in burnt peatlands (Holden *et al.*, 2015), although another study suggested that water tables were less deep in peatlands with burning practice (Worrall, Armstrong and Adamson, 2007). It should be noted that recent research has mapped significant mesoscale erosion features within the areas examined that may have greater impact than burning (Clutterbuck *et al.,* 2020)

Water quality is also important in order to quantify the impacts of rotational burning (Harper *et al.*, 2018). In the United Kingdom, water sources in upland catchments where the majority of the peatlands are located, represent 70% of the freshwater (Bonn *et al.*, 2009). DOC and water coloration are two good indicators to define water quality, but also, as previously highlighted (Figure 2.8), DOC is a source of carbon from peatlands and therefore an important component in the climate regulatory function of peatlands. DOC represents an important source of carbon loss from stream systems (Clutterbuck and Yallop, 2010) and may represent up to 30 – 50% of the net exchange carbon (Armstrong *et al.*, 2012). It is widely accepted that burnt areas have increased DOC concentrations (Clutterbuck and Yallop, 2009, 2010), although a few studies have claimed there is no correlation between rotational burning and DOC concentrations (Clay, Worrall and Aebischer, 2012). It has been noted that the scale of the study areas could explain a difference in the reported results, in which catchment scale studies showed an increase of DOC and plot scale studies showed the opposite (Harper *et al.*, 2018).

In terms of biodiversity (flora and fauna), research has highlighted that burning is beneficial and has significant beneficial impacts on biodiversity and ecosystem functions (Davies and Legg, 2008). Prescribed burning may prevent or reduce the extent, the severity and intensity of wildfires or 'hot fires' and their negative impacts in peatland environments (Davies and Legg, 2008); however, recent research did not agree with this statement, suggesting that burning does not stop wildfire expansion, and in fact areas with gully blocking and *Sphagnum* spp. were more efficient in stopping and helping to reduce the extent of the fire (Swindell, 2019). Consequently, burning practices modify the vegetation cover from bog vegetation to *Calluna* dominated vegetation (Lindsay, 2010), and significant differences in the vegetation composition have been found at regional and national scales in the United Kingdom between unburned and burned areas with less *Sphagnum* mosses and greater *Calluna vulgaris* in burned areas (Noble *et al.*, 2017). Recent research also identified no beneficial evidence of burning on the occurrence of *Sphagnum* mosses (Noble *et al.*, 2017).

Carbon dynamics can also be altered by burning practices, affecting different aspects of the carbon function and fluxes described previously in this section in relation to the importance of peatland environments (Figure 2.8). Carbon storage and sequestration are key reasons to preserve peatlands given current climate change, and the effects of burning are relevant in order to improve the conservation of the largest carbon storage globally. It has been demonstrated that rotational burning reduces the carbon storage in above ground vegetation (Ward *et al.*, 2007; Farage *et al.*, 2009), although some research has claimed that carbon loss is reduced under burning regimes (Clay, Worrall and Rose, 2010). Similarly, burning reduces the carbon accumulation in the peat surface (Ward *et al.*, 2007), and therefore the carbon storage on a long-term scale. On the face of it, and considering the ΔC in the peatland, burning seems to be beneficial in reducing the source of carbon in comparison with unburnt areas (Clay, Worrall and Rose, 2010), although again, other researchers claim the opposite saying that burning is not beneficial for the ΔC (Worrall *et al.*, 2010).

2.2.3.2.5. Agriculture, horticulture and grazing

Peatlands (both bogs and fens) have also been used for agriculture, although the ground need to be prepared in order to provide an appropriate surface for crops. Peat extraction is usually a stage of preparing land for agriculture by removing the peat vegetation and creating a drainage system (Rydin and Jeglum, 2006b). This drainage systems usually consists of shallow ditches with 10 – 15 m spacing in peatlands with a peat depth lower than 1 m (Ilnicki, 2003). After drainage, the peatland surface will sink as a consequence of a process called subsidence, that in some cases could results in a 2.5 m subsidence (Berglund, 1996). This practice of combined extraction with agricultural uses, has been very common across Europe (Göttlich *et al.* 1993).

Another common use of peatland environments is extraction of peat for horticulture. *Sphagnum* mosses, a peat-forming plant, have positive properties, such as moisture retention, good aeration and high cation exchange capacity (Rydin and Jeglum, 2006b). Seedbeds, peat pots and composting are the main horticultural purposes of peat (Bélanger *et al.*, 1988).

Finally, grazing activities are common in blanket bogs, where low levels of livestock can be supported by peatlands (0.4 sheep/ha; Lindsay, Birnie and Clough, 2014b); however, overstocking is a common practice across Europe and could damage peatland environments, not only the vegetation cover, but also the peatland surface due to trampling (Lindsay, Birnie and Clough, 2014b). This leads to the creation of small tracks or paths and bare peat surfaces where natural erosion processes (see section 2.2.3.1) could be initiated (Ellis and Tallis, 2001). Since 1980, EU sheep meat subsidies have increased this issue rather than reducing the problem of grazing on peatlands across the United Kingdom (Douglas, 1998; Fuller and Gough, 1999). In addition, another issue related with livestock is the emissions of GHG (Worrall and Clay, 2012). In fact, in the United Kingdom, sheep play an important role in GHG fluxes in peatland environments, moreover an increase in grazing activity could lead to an increase in GHG. For example, trampling will increase the loss of DOC and POC if erosion increases (Worrall and Clay, 2012).

2.2.3.2.6. Windfarms

Blanket bogs are typically located in areas with high winds and relatively good accessibility, meaning these areas are attractive for windfarm development (Lindsay, 2016c). This recent pressure in blanket mire complexes not only comprises disturbance from turbine installation, but also the tracks, sub-stations and associated infrastructure (Lindsay, 2016c; Wawrzyczek *et al.*, 2018). Windfarm infrastructures have impacts on the peatland surface, affecting mesotope units and hydrological patterns (Wawrzyczek *et al.*, 2018), as well as promoting changes in the vegetation cover and habitat loss (Fraga *et al.*, 2008). In some cases, blanket bogs have been completely removed as a consequence of windfarm development (Heras and Infante, 2008). Drainage around the tracks (Figure 2.17) is also an important factor in understanding the impact of these windfarms have on blanket bog hydrology and mesotope units. In addition, peat subsidence and erosion could be initiated by track construction, as consequence of drier conditions and exposed peat surfaces (Figure 2.17; Figure 2.18). In Scotland, 55% of windfarms are installed in deep peat (Bright *et al.*, 2008), which could have a much greater impact on carbon storage and balance, although it has been reported that windfarm developments will demonstrate a potentially positive effect in carbon balance after 25 year of wind farming (Nayak *et al.*, 2010); however, recent developments in the carbon calculator model, have concluded that blanket bogs in good conservation status may not result in a positive carbon feedback, and it might not be beneficial in terms of carbon balance in the long-term (Smith, Nayak and Smith, 2014).

In Spain, windfarms could represent an important pressure for the habitat (Heras and Infante, 2008) and vegetation diversity (Fraga *et al.*, 2008), yet they are not currently listed as an important threat (European Commission, 2012a).

Figure 2.17. Windfarm track construction at Malverde blanket bog in northern Spain (Cantabria).

Figure 2.18. Evidence of erosion adjacent to windfarm development in northern Spain (Cantabria).

2.2.3.2.7. Climate change

Since blanket bogs are limited by climatic conditions (see section 2.2.1) and are dependent on high precipitation and humidity with low temperatures to form (Lindsay *et al.*, 1988), the current situation with climate change is a challenge for this habitat, although other areas could become suitable for this habitat. Some blanket bogs are more than 10,000 years old, and therefore they have suffered from changes in climatic conditions. A clear example of this is the Medieval Climate Anomaly (MCA), where precipitation decreased and temperatures increased, producing a change in vegetation and promoting erosion in blanket bog surfaces (Tallis, 1997a; Ellis and Tallis, 2001). Changes in climate conditions likely explain the early distribution of blanket peatlands in the Holocene and later expansion (Gallego-Sala *et al.*, 2016).

Over the last millennium, peatlands in the Northern Hemisphere across North America, Russia and Europe, have accumulated carbon; however, these rates were higher during the MCA and lower in the Little Ice Age (LIA), highlighting the important role that temperatures play in peatlands, and the potential effect of global warming on peatlands (Charman *et al.*, 2013).

In the current context of climate change, several researchers have reported the importance of peatlands and their role in the carbon cycle, but also highlighted the likelihood of climate change affecting peatland environments. For example, in Canada, it is expected that an increase in the annual air temperature of 3 to 5 °C, would result in land and coastal peatlands being affected by sea level rise, releasing GHG into the atmosphere (Tarnocai, 2006). Moreover current sea level change could affect at least 7.4% (+5 m sea level rise) of the total global extent of blanket bogs (Whittle and Gallego-Sala, 2016). In the United Kingdom, large areas of bioclimatic blanket peatlands may be affected, potentially reducing their extent (Gallego-Sala *et al.*, 2010).

2.2.3.3. The erosion problem

Erosion in peatlands is a natural process caused by natural environmental drivers, such as wind and water. However, the natural or anthropogenic causes that initiate erosion in blanket bogs have been widely discussed. Initially, natural causes were more likely to explain erosion in blanket mires. Early research suggested that peat cannot accumulate indefinitely (Moss, 1913), and therefore a peat cycle was proposed based on three stages: 1) Peat accumulation; 2) Maturity; 3) Post-maturity, the last only happening when an instability in the peatland surface occurs (Johnson, 1957). Pipe erosion has been also described as a potential cause of the erosion process in peatlands and the cause of peat hags (Glaser and Janssens, 1986). In the early 1960s, it was suggested that early erosion occurred before human activities, although this problem was noted more recently and related with human activities, such as changes in vegetation (Tallis, 1964). At the end of the last century, climatic changes over the last 3,000 years were studied to understand the problem of erosion identifying two main periods of erosion; the first period was around the Early MCA (1250 – 1450 AD) and the second period was around 1750 AD (Tallis, 1997a; Table 2.1). It has also been suggested that the onset of erosion was related with climatic changes in the LIA, but the same author emphasised the importance of burning intensity as a possible initiation of erosion (Stevenson, Jones and Battarbee, 1990).

Table 2.1. Main factor of peat erosion at Holme Moss (United Kingdom) (Tallis, 1997b).

Date	*Factor of peat erosion*
1450	Desiccation MCA
1770	Fire
1800	Loss of *Sphagnum*
1940	Overgrazing
1976	Fire

The combination of human activities and climate conditions are therefore, the main reasons of peat erosion. While deforestation could change the hydrology and therefore also has an impact on the blanket peatland erosion, climatic changes could increase the impacts of this pressure, especially when combined with more contemporaneous issues, such as grazing (Ellis and Tallis, 2001).

Over the last 250 years, erosion has increased in magnitude across the world (Mackay and Tallis, 1996; Huang, 2002), at the same time grazing and agriculture has intensified (Huang, 2002), as well as an increase in pollution, particularly in the British uplands (Fergunson, Lee and Bell, 1978; Yeloff, Labadz and Hunt, 2006). Fires are also important (Yeloff, Labadz and Hunt, 2006), and since very early on, fire related erosion has been highlighted as one of the main problems for peatlands, and the current era is noted as period with greater rates of erosion (Bower, 1960).

In blanket peatlands, erosion as a process is described in three different stages. The first stage is when the blanket bog is covered in vegetation and it is removed due to natural or anthropogenic pressures, such as fire, grazing and peat extraction (see section 2.2.3.2) creating an exposed peat surface. Secondly, natural processes, such as desiccation in summer (Francis, 1990) or frost in winter (Labadz, Burt and Potter, 1991), prepares the surface for erosion. Finally, in the last stage, the surface is remove by natural erosion processes, such as wind, water, ice or oxidation (Tallis, 1998). Early research also highlighted the importance of altitude in relation with erosion, suggesting that at higher altitude the eroded area is lower and closely related with meteorological conditions, and potentially less anthropogenic pressures that are lower in high altitudes (Phillips, Tallis and Yalden, 1981).

2.2.4. Protection and legislation of blanket bogs

2.2.4.1. Current situation of degradation

Under EU legislation, the European Union states report the status of each habitat every 6 years to review the status and restoration/conservation strategies. In the United Kingdom, the country with the largest extent of blanket bogs, all peatland types are in unfavourable – bad conditions (European Commission, 2012b), the worst status of the Habitat Directive evaluation ranking, and no changes or progress have been noted between 2007 – 2012. A similar situation has been reported for Ireland, where all blanket bogs are in unfavourable – bad conditions (European Commission, 2015) and a recent report for the period 2013 – 2019 noted no change in this situation reducing in their extension, the ecosystem services and predicting the same status in the future if no action is taken (NPWS, 2019). In Spain, the last assessment of the habitat status highlighted that in 2013, only one peatland in Spain was in favourable condition. The rest of the recognised peatlands are in unfavourable – inadequate condition (European Commission, 2012a).

2.2.4.2. Current status of protection across Europe

The European Habitats Directive (92/43/EC) has identified a range of peatland types across Europe that need to be protected in order to preserve carbon storage, biodiversity and the potential of the peatland to function as carbon sinks. Blanket bogs are protected under this legislation, with a total of 200 blanket bogs noted (European Commission, 2019; European Environment Agency, 2019). The United Kingdom has more than 55% of the designated blanket bogs (Figure 2.19; Figure 2.20) which is unsurprising considering that this country is located in an area with a high influence of oceanic climate, and therefore contains the largest extent of blanket bogs in Europe and 13% of the world's blanket bogs (Bain *et al.*, 2011). Ireland is the second country with the most designated blanket bogs, followed by Spain (Figure 2.19; Figure 2.20).

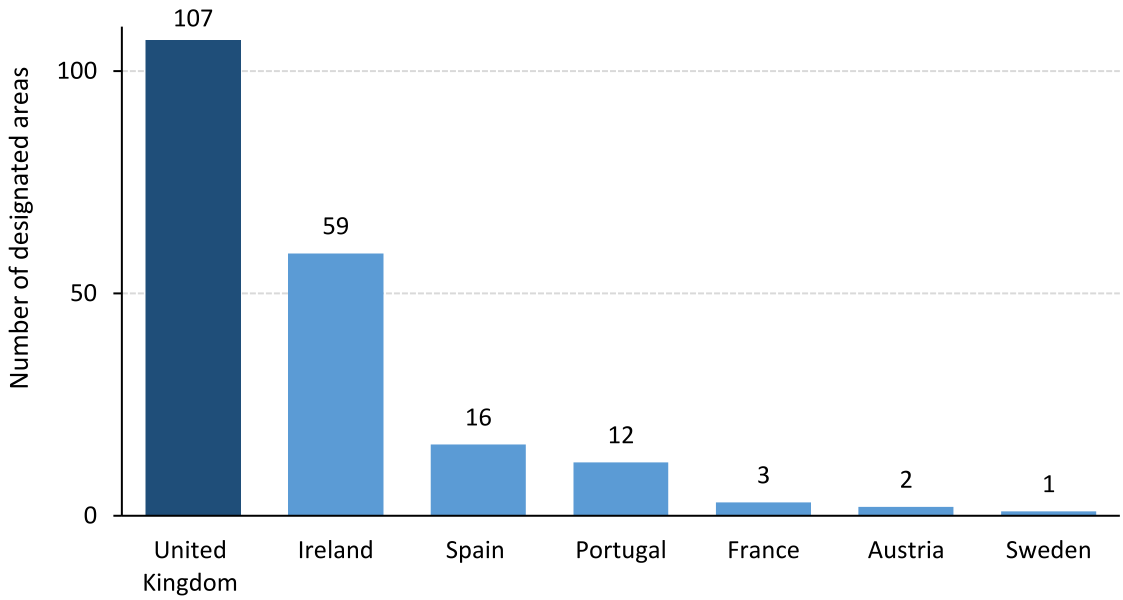


Figure 2.19. Number of designated areas with blanket bogs in the European Union by country in 2019 (European Environment Agency, 2019).

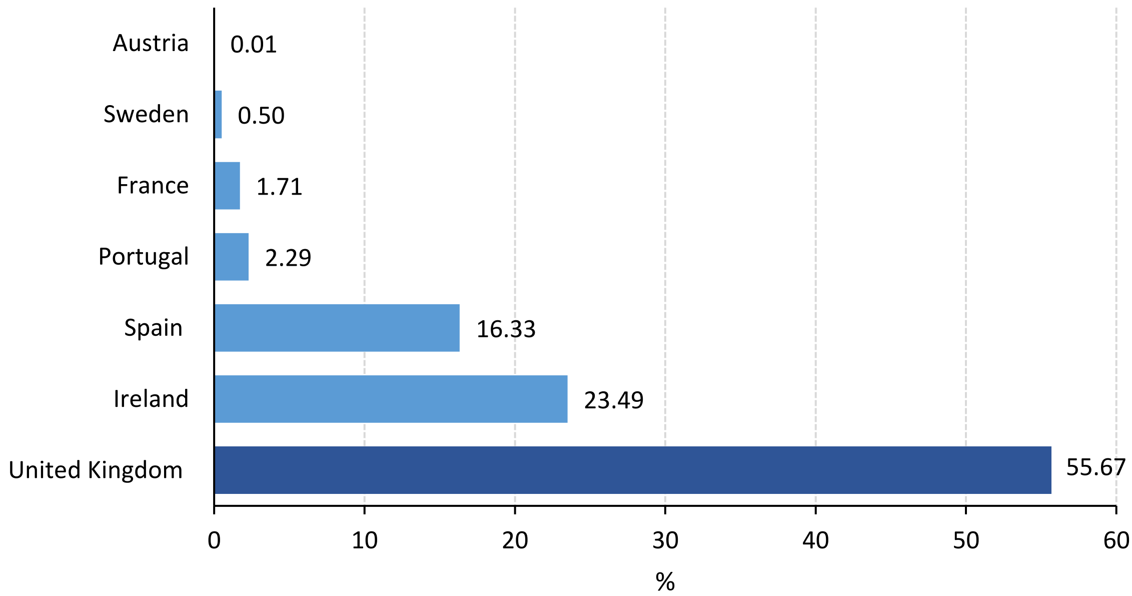


Figure 2.20. Percentage of blanket bog extent in the European Union by countries in 2019 (European Environment Agency, 2019).

Despite the importance of blanket bogs in the current context of climate change, with their diversity, rarity and unique geomorphology, there are major gaps in the Spanish blanket bog inventory (Heras *et al.*, 2017; Ramil-Rego *et al.*, 2017). Moreover, several mistakes in the Spanish inventory of blanket bogs have also overestimated the extent of this habitat in the country (see section 2.2.2.2), where the Asturias region has been defined as an important area for blanket bogs, but only one area of blanket bog has recently been confirmed (Ramil-Rego *et al.*, 2017). In 2019, the Natura 2000 network provided an estimation of the total extent of blanket bogs in Asturias of 2,499.5 ha, but in reality, only 16.98 ha are blanket bogs (Ramil-Rego *et al.*, 2017; European Environment Agency, 2019).

2.2.5. Blanket bog restoration

Peatland restoration across Europe (United Kingdom, Germany, Poland, Sweden, Ireland, and Spain) is very common, particularly in EU countries where the Habitats Directive protects peatlands and promotes their conservation. The LIFE program is a scheme by the EU to promote restoration and conservation actions in the environment and climate actions since 1992. The budget for the period of 2014 – 2020 was 3.4€ billion (European Commission, 2019). Spain has been the main receptor of funding for LIFE projects, followed by Italy and the United Kingdom (European Commission, 2019). Over 260 LIFE projects have restored peatlands through Europe, including the United Kingdom and Spain, where the LIFE program has helped to restore several blanket bogs across these countries (Joosten, Tapio-Biström and Tol, 2012).

There are several strategical elements in restoring peatland environments and improving the ecosystem services; for example, the identification of all peatlands is crucial to determining their true extent, furthermore their evaluation is key to determining the total carbon storage and potential to serve as carbon sinks (Joosten, Tapio-Biström and Tol, 2012). Recent research has showed the recovery of ecosystem services from eroded peatlands after LIFE project restorations (Alderson *et al.*, 2019).

There are several restoration techniques, such as gully blocking to increase the water tables and reduce the sediment loss, reprofiling and bare peat stabilisation to reduce the erosion process and peat loss rates, or revegetation to promote the peat-forming species recolonization in the degraded peatlands.

2.2.5.1. Bare peat stabilisation and revegetation

Bare peat, also referred to as exposed peat, is the main area exposed to natural erosion (e.g. fluvial (Evans and Warburton, 2007) and wind (Foulds and Warburton, 2007b)) and previously described pressures (see section 2.2.3). This exposed peat could develop into gully systems, thus increasing erosion rates and degradation. The aim of covering exposed peat is to create a microhabitat that allows vegetation growth and therefore reduces erosion processes (Parry, Holden and Chapman, 2014; Aguirre, Benito and Galera, 2017). In some cases, exposed peat is covered by heather brash (e.g. United Kingdom), but also geotextile techniques have been used to reduce the erosion rates and increase the vegetation cover (e.g. Spain) (Aguirre, Benito and Galera, 2017; Figure 2.21). Heather brash contains different peat-forming species apart from heather to protect the bare peat and promote the re-establishment of vegetation; however, in Spain, geotextiles seem to be enough to promote the vegetation regeneration (Figure 2.21C), although some peat species have also been planted (Aguirre, Benito and Galera, 2017).

Figure 2.21. A) Exposed peat area at Zalama blanket bog (Photo: DFB); B) Geotextile example covering exposed peat surface at Zalama blanket bog (Photo: DFB); C) Area of exposed peat after restoration actions using geotextile.

Geotextile is usually attached with metal pins to avoid loss of the geotextile, and in the case of Spain, pine wood has been used to make the area more stable in high winds (Aguirre, Benito and Galera, 2017). The geotextile is biodegradable so will disappear with time, but the metal pins should be removed after vegetation is covering the target area (Chico and Clutterbuck, 2019). This method has reduced the total extent of exposed peat, and the peat loss rates are lower than degraded and exposed areas in northern Spain (Chico and Clutterbuck, 2019).

In some cases, such as in the United Kingdom, vegetation will colonise these areas very slowly, potentially due to the peat properties not being suitable for vegetation because of high concentrations of atmospheric pollution particles and low pH (Fergunson, Lee and Bell, 1978). In this case, bare peat is limed and fertilised to increase the pH levels and re-establish the vegetation (Lunt *et al.*, 2010).

In addition to the geotextile cover, in Zalama blanket bog, livestock were also excluded from the area to prevent grazing on the peatland surface (Aguirre, Benito and Galera, 2017). A clear change in the vegetation communities has since been noted (Chico and Clutterbuck, 2019), although some grasses have been planted in the area as part of the restoration actions.

2.2.5.2. Gully blocking, dam constructions and reprofiling

Gullies are one of the main problems in blanket bogs across different countries, such as United Kingdom, Ireland and Spain. Gully erosion contributes to carbon loss (Evans and Lindsay, 2010), reduces water quality and has a negative impact on water tables (Daniels *et al.*, 2008). Gully blocking is considered to reduce the erosion rates, but also to hold the water back and increase the water table (Moors for the Future, 2020), using different materials such as plastic piling, wood, stone or peat (Parry, Holden and Chapman, 2014). Gully blocks (or dams) are built with impermeable materials (e.g. plastic) when the restoration aims are to increase water tables and trap sediment (Moors for the Future, 2020). In Spain, gully systems are not well developed due to the small extent of peatlands; however, there are several peat faces exposed to peat erosion (Figure 2.21A) and some timber dams have been constructed in some restoration projects (e.g. Zalama blanket bog) to reduce the runoff and trap sediments (Aguirre, Benito and Galera, 2017). Another action related with gully blocking and dam constructions is reprofiling, when the exposed peat is located in steep slopes or there are peat hags (e.g. Figure 2.21B). The main problem in these areas is the contribution to POC (Evans and Warburton, 2007), and this action will reduce the erosion in these areas creating a more homogenous slope by removing the over hags (Parry, Holden and Chapman, 2014).

2.2.5.3. Artificial drain blocking

Artificial drainage is one of the most common problem in peatlands (see section 2.2.3.2), and restoration actions to reverse the negative impacts of drainage is one of the main aims of restoration projects (Holden, Chapman and Labadz, 2004). Materials used to block the drainage system are usually heather bales, peat, plastic, wood or stone to block the gully, although peat is the most common material used for this restoration action (Armstrong *et al.*, 2009). Although the material used can be important for restoration success at different sites, the dam spacing is also relevant; particularly if there is a gradient or steep slope (e.g.

blanket peatlands). In this case, more dams need to be installed to compensate the flow force and topography needs to be carefully considered (Armstrong *et al.*, 2009). The most effective dams are the ones built with peat (74.1%) (Armstrong *et al.*, 2009).

2.2.5.4. Sphagnum reintroduction

Sphagnum spp. are one of the main peat-forming plants in blanket bogs; however, wildfires and burning have reduced the *Sphagnum* cover in blanket bogs (Tallis, 1964), particularly prior to the industrial revolution. More recently, with the industrial era, air pollution has dramatically affected the distribution of *Sphagnum*, because the peat is too acidic and contains sulphuric acid (Fergunson, Lee and Bell, 1978).

Early approaches for *Sphagnum* reintroduction in British blanket bogs involved including propagules in heather brash that were used to cover the bare peat to reduce the erosion and POC loss with successful results (95% reduction after 2 years; Pilkington, 2015). However, despite this method having a positive effect on reducing erosion rates and evapotranspiration, the vegetation cover developed into a sward and without *Sphagnum* mosses (Wittram *et al.*, 2015). More recently, two more direct methods have been used to cover large areas; micro-plugs and clumps, however both these methods are rather expensive and the coverage will depend on the source material (Wittram *et al.*, 2015).

2.3. MONITORING EROSION IN BLANKET BOGS

2.3.1. Traditional methods

Traditional methods to measure erosion rates can be divided into two main groups: direct and indirect. Direct methods are those that involve observing a fixed point, and surface movements and removals are measured. The indirect methods are based on trapping the sediments removed from the surface and extrapolating to estimate the total volume of peat loss (Tallis, 1973).

2.3.1.1. Erosion pins

Erosion pins are a direct method to measure erosion that have been used in multiple environments (e.g. sand dunes, gullies, river banks or peatlands) since the 1950s (Boardman and Favis-Mortlock, 2016). Erosion pins are rods that are installed and fixed into the mineral substrate to measure how much soil has been removed (or accumulated) in a period of time by measuring the total length outside the soil (Haigh, 1977; Boardman and Favis-Mortlock, 2016), and is the most common method used to measure erosion (Haigh, 1977). This method is cheap and simple, but will only cover small areas (Boardman and Favis-Mortlock, 2016). The most common pin material used is metal, but cheaper materials such as plastic and bamboo canes have been also used (Phillips, Tallis and Yalden, 1981), although for long-term studies, metal is recommended.

This method has been highlighted as ideal for exposed peat environments with high erosion rates (Boardman and Favis-Mortlock, 2016); however, livestock grazing (e.g. sheep) is common practice on blanket bogs and they can damage the erosion pins by trampling the area, resulting in loss of information for the specific study site (Birnie, 1993). In addition, four main sources of errors have been identified for this technique in peatlands: 1) movement of erosion pins (e.g. livestock, ice); 2) changes in the surface elevation (e.g. mire breathing); 3) influence of the pin on the erosion rates (e.g. dead vegetation around the erosion pin) and 4) human interference (Couper, Stott and Maddock, 2002; Evans and Warburton, 2007).

There are multiple studies that have measured peat erosion rates using erosion pins (Table 2.2) mainly located in the United Kingdom. The majority of the studies are in England (15), followed by Wales (3), Scotland (2) and Tasmania (1). The range of erosion rates is variable across England from 73.8 mm yr^{-1} in Holme Moss (Table 2.2.; Phillips, Tallis and Yalden, 1981) to 1.03 mm yr^{-1} in a recent study in Flow Moss (Table 2.2.; Baynes, 2012); however, it is important to highlight the location where measurements are taken as the angle and slope could play an important role in erosion rates (Tallis and Yalden, 1983). In this case, the highest erosion rate has been recorded in a peat margin, most exposed to weathering erosion and oxidation, whereas a lower erosion rate has been reported in a peat hag. In order to determinate erosion rates for a whole complex peatland environment, different zones within the exposed peat areas, with varying slopes, should be considered to gain an understanding of real erosion rate.

Table 2.2. Peat erosion rates measured by erosion pins across the United Kingdom and Australia

Location	*Surface change (mm* yr^{-1}*)*	*Reference*
North York (England)	40.9	Imeson, 1974
Snake Pass (England)	7.8	Phillips *et al.,* 1981
Moor House (England)	10.5	Phillips *et al.,* 1981
Holme Moss (England)	73.8	Phillips *et al.,* 1981
Snake Pass (England)	5.4	Phillips *et al.,* 1981
Holme Moss (England)	33.5	Tallis and Yalden, 1983
Cabin Clough (England)	18.5	Tallis and Yalden, 1983
Doctors Gate (England)	9.6	Tallis and Yalden, 1983
Peak District (England)	18.4 – 24.2	Anderson, 1986
Plynlimon (Wales)	30	Robinson and Newson, 1986
Mid Wales (Wales)	23.4	Francis and Taylor, 1989
Plynlimon (Wales)	16	Francis, 1990
Shetland (Scotland)	10 – 40	Birnie, 1993
Forest of Bowland (England)	20.4	Mackay and Tallis, 1994
Harrop Moss (England)	13.2	Anderson, Tallis and Yalden, 1997
Monachyle (Scotland)	59	Stott, 1997
Macquarie Island (Tasmania)	43	Selkirk and Saffigna, 1999
Moor House (England)	19.3	Evans and Warburton, 2005
Upper North Grain (England)	34	Evans, Warburton and Yang, 2006
Flow Moss (England)	1.03	Baynes, 2012

2.3.1.2. Sediment traps

An example of an indirect method to measure erosion rates, or in this case sediment loss, is sediment traps. This method can cover larger areas instead of a single point and can provide a better understanding of the sediment movement in the exposed peat. The results of this method are a total sediment budget that can be converted into surface change (retreat of peat surface), although it is difficult to compare due the nature of the method (Evans and Warburton, 2007). Similar to erosion pins, several limitations need to be considered with this indirect method. Since sediment traps measure the sediment transported, the design must be capable of trapping sediments moved by different transport methods, such as suspension or rolled in the case of wind erosion. Moreover, to collect the sediments, traps need to be inserted into the peat, and thus could accelerate the erosion process (Birnie, 1974). Evans and Warburton (2007) also highlight the importance of the location in this method, as well as the area of sediment contribution and slope. Therefore, sediment traps are better to cover larger areas when compared with erosion pins, although the experiment design (e.g. location) is important in order to obtain comparable results with other methods, furthermore sediment traps and erosion pins could be measuring different aspects of the erosion process (Evans and Warburton, 2007).

2.3.2. New geospatial techniques

Since 2010, new techniques have been developed that are capable of measuring surface changes in high resolution. Remote sensing techniques, such as aerial photography (Bower, 1961; Tallis, 1973) and airborne LiDAR (Walsh, Butler and Malanson, 1998; Evans and Lindsay, 2010), can cover surface changes over large areas in comparison with the traditional methods. However the high cost and resolution (typically 25 cm at best) are the main limitations in measuring small changes (Clutterbuck *et al.*, 2018). However, other new geospatial techniques, such as ground based photogrammetry employing Structure-from-Motion (SfM) techniques, conventional photogrammetry from Unmanned Aerial Vehicles (UAV) and Terrestrial Laser Scanning (TLS) are becoming the most popular methods to measure erosion rates in peatland environments (Grayson *et al.*, 2012; Kalacska *et al.*, 2013; Glendell *et al.*, 2017; Li *et al.*, 2018). These techniques can obtain mm resolution erosion rates and cover greater extents when compared with more traditional methods, such as erosion pins (Boardman and Favis-Mortlock, 2016).

2.3.2.1. Aerial photographs and Airborne LiDAR

Early studies have defined erosion features, such as gullies using aerial photographs (Bower, 1961; Tallis, 1973). Airborne LiDAR data has been utilised to provide high resolution maps of gullies, where erosion is significant, particularly in the United Kingdom (Walsh, Butler and Malanson, 1998; Evans and Lindsay, 2010). The use of Digital Elevation Models (DEMs) to estimate the depth and extent of gully erosion have been successfully used (Betts and DeRose, 1999; Betts, Trustrum and De Rose, 2003; Evans and Lindsay, 2010), although the resolution is important in order to fully quantify erosion. LiDAR data provides high resolution and accuracy, and has been successfully applied to map gully erosion across different environments and areas of the gully (Hancock and Evans, 2006; James, Watson and Hansen, 2007; Evans and Lindsay, 2010). However, despite the powerful information derived from Airborne LiDAR data and aerial photography for landscape assessments, erosion rates in blanket bogs are smaller than the resolution of this data (typically 25 cm), and therefore a quantification of erosion rates using this method will not provide high resolution results, although it will provide a range of useful data to define where erosion is taking place.

2.3.2.2. Unmanned Aerial Vehicles and Structure-from-motion

Unmanned Aerial Vehicles (UAV) are an extremely valuable tool to collect data in peatland environments. They have capacity to create high resolution DEMs (Chico and Clutterbuck, 2019) at centimetre resolution (and potentially mm if the UAV is flown low enough), although an aerial view will not derive 3D morphology of overhanging features such as peat hags, a typical feature in peatland environments. The cost of this technique is lower than aircraft data, although the areas covered in high resolution will take longer than an aircraft (Clutterbuck *et al.*, 2018). A benefit of UAVs is the capacity to collect photographs at high resolution, that can subsequently be used for construction of 3D models through Structure-from-Motion (SfM) at high resolution (<1 cm), although this method could also be conducted with ground cameras. SfM has been widely applied in peatland environments (Kalacska *et al.*, 2013; Knoth *et al.*, 2013; Lehmann *et al.*, 2016; Lovitt, Rahman and McDermid, 2017; Smith and Warburton, 2018), and has been successfully utilised to estimate erosion rates. Although SfM has been described as a cheaper technique (Li *et al.*, 2018), when large areas of assessment are required, UAVs appear to be the most efficient technique in terms of cost, and only SfM is more effective for plot-scale areas (Glendell *et al.*, 2017). In addition, data processing is also an important variable in terms of time - cost, which has not been considered in some research (Li *et al.*, 2018), where SfM requires more time than another techniques (Glendell *et al.*, 2017) and where ground photographs implicate disturbance of the studied area, UAV and other newer techniques, such as TLS, are less intrusive on a habitat that is very sensitive to disturbances. Therefore, the wider application of SfM for understanding erosion in peatland environments could be combine with UAV and TLS for a better assessment of surface changes.

2.3.2.3. Terrestrial Laser Scanner

TLS is capable of achieving ultra-high resolution and accuracy, thus has been used as a benchmark for other techniques, such as SfM (Castillo, *et al.*, 2012; Eltner, Mulsow and Maas, 2013; Gómez-Gutiérrez *et al.*, 2014; Ouédraogo *et al.*, 2014; Neugirg *et al.*, 2016).

This technique has rapidly advanced in the last decade, and nowadays TLS units are portable and capable of recording more than 1 million points per second at ultra-high resolution, with accuracies up to 1 mm at 10 – 15 m from the scanner (Idrees and Pradhan, 2016). Despite the potential value of TLS to measure erosion in peatlands environments, only a few studies have been conducted using this method (Grayson *et al.*, 2012; Glendell *et al.*, 2017), and in some of them, the errors in assessment were higher than the rate of erosion measured (Grayson *et al.*, 2012). Several challenges have been noted with this method, such as the limitations in assessing areas with vegetation and surface changes as consequence of mire breathing (Grayson *et al.*, 2012). In addition, given that rates of erosion can be as low as 1.03 mm yr^{-1} (Baynes, 2012) the highest resolution data that can be collected using the least intrusive method is required for assessing peatlands. It is of course key that the existence of a peatland is known first.

In the next chapters, this research will fill a gap in the Spanish inventory of blanket bogs, classifying any blanket bogs identified to mesotope level and undertake assessment of the characteristics and degradation of blanket bogs identified. A method using TLS will be developed to enable ultra high resolution assessment of erosion.

Chapter 3

Identification and geo-hydromorphological assessment of Europe's southernmost blanket bogs

The content of this chapter has been partially published

3.1. INTRODUCTION

In some countries across Europe, peatlands account for >20% of the land (e.g. Estonia, Finland and Ireland), yet in Spain these ecosystems are extremely rare and cover only 0.07% of the country (Tanneberger *et al.*, 2017) being one of the countries with least percentage of peatlands globally (Joosten, 2009). Blanket bogs are even scarcer in Spain, as their presence is specifically limited by topography and climatic conditions required for this habitat: high precipitation and humidity with low temperature variation (Lindsay *et al.*, 1988). Spanish blanket bogs are restricted, therefore, to the north of the country (Figure 3.1) in the Euro-Siberian geographical area (Heras *et al.*, 2017), where climatic conditions are suitable mainly on 'flat' areas along the summits of the Cantabrian Mountains (Martínez-Cortizas *et al.*, 2000; Heras and Infante, 2003). Despite their small extent and occurrence, these blanket bogs contain an important palaeoenviromental record and have stored carbon from the atmosphere for over 8,000 years (Pérez-Díaz *et al.*, 2016).These habitats also provide valuable contemporary biodiversity for the country, and have the potential to function as carbon sinks (Heras *et al.*, 2017).

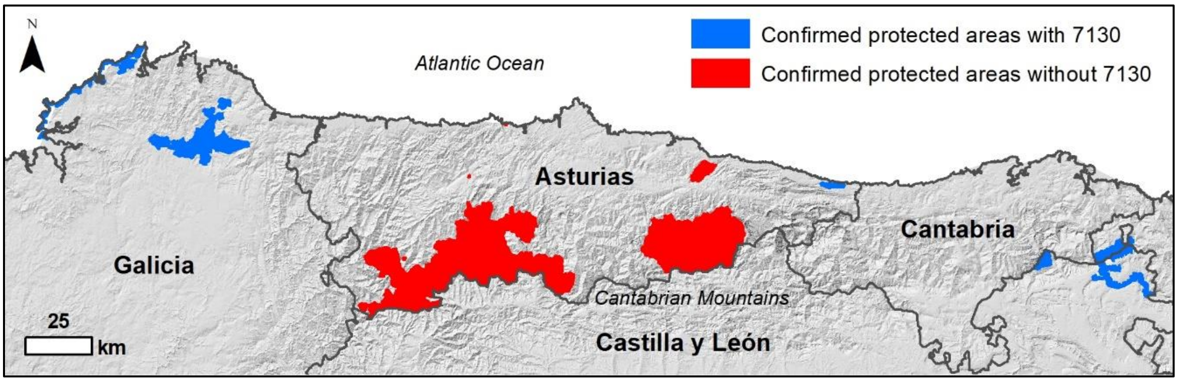


Figure 3.1. Protected areas with blanket bogs (7130) in Spain. In blue, areas with presence of blanket bogs and in red, areas with incorrectly blanket bogs classified (European Environment Agency, 2019).

The majority of recognised and protected blanket bogs in Spain are located in the regions of Galicia and Asturias (Figure 3.1; European Environment Agency, 2019) with only two examples in the eastern part of the mountain range in or on the boundaries of the Basque Country and Castilla y León: Zalama blanket bog (Heras, 2002) and Montes de Valnera Site of Community Interest (SCI; European Environment Agency, 2019). However, the majority of blanket bogs recognised in the region of Asturias under Nature 2000 are not located in areas where the topography and climatic conditions are suitable for blanket bog development (Figure 3.1) and the most recent inventory of peatlands does not recognise any blanket bog in this region (Pontevedra-Pombal *et al.*, 2017). Furthermore, some recent European projects have highlighted significant errors in the Natura 2000 inventory and indicate that <1% of the blanket bogs recognised in this region are actually blanket bog (Ramil-Rego *et al.*, 2017; Table 3.1). In addition to miss-classification, a gap in the inventory of blanket bogs has been indicated to exist between Picos de Europa in the eastern part of Asturias and the Pyrenees in the Basque Country and Navarra regions (Figure 3.2**Error! Reference source not found.Error! Reference source not found.Error! Reference source not found.**; Ramil-Rego *et al.*, 2017; Heras and Infante, 2018). These unrecorded blanket bogs could in fact represent the southernmost edge-of-range of this habitat in Europe.

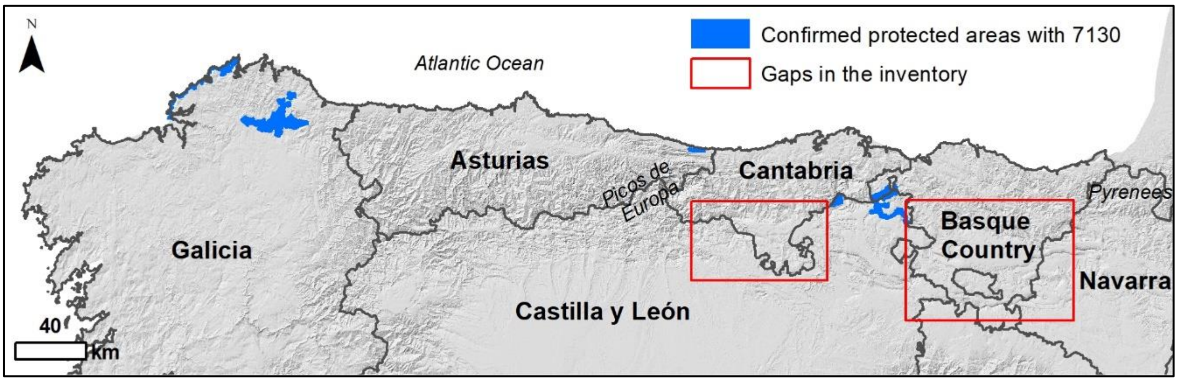


Figure 3.2. Potential gap areas with blanket bogs in northern Spain.

Table 3.1. The extent of blanket bog in Spain recorded under the Natura 2000 network (European Environment Agency, 2019) together with the rectification in Asturias region (Ramil-Rego *et al.*, 2017). * Current area of blanket bogs in Asturias.

	Region	*Area (ha)*
Protected	Asturias	2,499.5 **16.98*
	Galicia	373.4
	Castilla y León	14.61
	Basque Country	4.41

Although a large number of Spanish blanket bogs are protected, areas that are not mapped are exposed to anthropogenic and natural pressures that contribute to their degradation.

Historically domestic peat cutting may have been common in these regions for local use (Heras, 2002) although the main anthropogenic pressures today are linked to livestock (Heras and Infante, 2018), vegetation burning (Heras, 2002), commercial peat extraction (Guerrero, 1987) and more recently, windfarm infrastructures (Heras and Infante, 2008).

In light of the gaps noted in the blanket bog inventory (Ramil-Rego *et al.*, 2017; Heras and Infante, 2018), it is important to assess areas where currently unrecognised blanket bogs could exist. This will identify the wider extent and types of blanket bogs in Spain and enable understanding of the range of pressures and the current status of this rare habitat. Therefore, this chapter aims to identify and classify currently unrecognised blanket bogs in the Cantabrian Mountains. The following objectives were set:

a) Use climatic and topographical data for northern Spain to identify potential areas where blanket bogs may exist but are currently not recognised in the areas reported with gaps.
b) Undertake ground survey to confirm the presence of peat and measure the peat depth in each identified blanket bog.
c) Define the extent and volume of peat in each peatland unit based on the peat depth results.
d) Describe the landscape context of blanket bogs through aspect and slope analysis.
e) Classify the peatlands based on the hierarchical classification at mesotope level.

This chapter expands on the information presented in two publications: Chico et al. (2020) *Geo-hydromorphological assessment of Europe's southernmost blanket bogs*. Earth Surface Processes and Landforms45 (12), 2747–2760; and Chico et al. (2019) *Identification and classification of unmapped blanket bogs in the Cordillera Cantábrica, northern Spain.* Mires and Peat, 24 (02), 1–12.

3.2. METHODS

3.2.1. Identification

3.2.1.1. Climate and topographical analysis

Potential areas of unrecognised blanket bog were identified in the Atlantic Bioregion between the eastern limit of Asturias (Picos de Europa) and the north of Navarra (The

Pyrenees; Figure 3.2). Climatic model data for the period 1970-2000 were obtained from WorldClim (Hijmans *et al.*, 2005). Precipitation and temperature (maximum and mean) were analysed in ArcGIS version 10.7 to identify potential areas with >1,000 mm of precipitation per year, low maximum temperatures (<15°C) and limited seasonal variability in temperatures (Lindsay *et al.*, 1988). Additionally, precipitation was examined in July and August and temperatures were also explored by meteorological seasons. Digital Elevation Models (DEM) at 0.25 m resolution for 2017 were obtained from Instituto Geográfico Nacional (2019) and were used to exclude areas under 600 m as these areas will have low precipitation and high temperatures not suitable for blanket bogs and it has been defined as the limit of blanket bogs in Spain (Pontevedra-Pombal *et al.*, 2009). Areas of land identified with suitable climate and topography were then examined by grid squares (sectors) 30 km by 30 km to locate potential areas of blanket bog based on the presence of pool systems, erosion features and exposed peat visible in aerial photos (RGB) from 2017 (Instituto Geográfico Nacional, 2019). After exploration of each sector, topographical factors such as rock outcrops, changes in vegetation or anthropogenic features (e.g. vehicle tracks) were used to define the potential extent of the peatland. Finally, an on-site assessment was undertaken to confirm the presence of peat and redefine the potential extent of the peatland based on the field assessment.

3.2.1.2. Peat depth, volume and peatland extent

For areas of peatland that were confirmed as blanket bogs, a systematic square grid of points was created using the tool Fishnet in ArcGIS 10.7. Peat depth surveys were undertaken between May 2017 and July 2019. Each survey point was located using a Garmin GPSMAP64 handheld GNSS reporting an accuracy of ± 3 m and peat depth measurements were collected using connectable 50 cm-length sections of steel rod (6 mm in diameter). Additional survey point locations were added where peat depth on the edge of the initial extent was greater than 30 cm using the same systematic square grid. Peatlands with a potential large continuous peat surface (>200 m^2) were surveyed using a 30 m by 30 m square grid and peatlands with rock outcrops or evidence of discontinuous areas in the peatland body were surveyed using a 15 m by 15 m square grid to provide the best definition of the peatland extent. Peat cores were collected from one location at five of the study sites and the depth determined using the auger was typically within 2 cm of the depth estimated using a depth rod prior to core extraction (Table 3.2). Similar observations have been observed in another peatlands in these regions as the peat base lies directly on the bedrock with very little or no clay layer (Heras, 2002).

Table 3.2. Comparison between rod measurements and peat core length.

Peatland	*Peat depth using the rod (cm)*	*Peat core length (cm)*
Zalama	248.5	246.5
Ilsos de Zalama	230.0	229.5
Collado de Hornaza	251.5	249.0
La Marruya	145.5	145.0
Malverde	296.0	294.0

Peat depth measurements for each site were interpolated to create a map of the peatland using a spline algorithm in ArcGIS 10.7, and the main body of the peatland was delimited using a minimum peat depth of 40 cm (Cruickshank & Tomlinson, 1990). The peatland margins were identified as areas where peat depth ranged from 30 – 40 cm, and the volume of peat at each site was determined from all interpolated peat depth values.

3.2.1.3. Landscape characteristics

Landscape analysis was undertaken to understand the main characteristics of the peatland areas through slope and aspect analyses in ArcGIS 10.7 to understand the peatland geomorphology and landscape context using a DEM at 0.25 m resolution (Instituto Geográfico Nacional, 2019).

3.2.1.4. Statistical analysis

Relationships between peat depth, peat volume, peat extent, aspect, latitude and longitude were examined using Pearson coefficient and Spearman correlation tests (if the data were normally distributed or not respectively). In addition, a Generalized Linear Model (GLM) was designed to identify the most influential factors controlling development of the blanket bogs identified. All statistical analyses were undertaken in R 3.6.2.

3.2.2. Geo-hydromorphological classification

Hierarchical classification was undertaken to classify the peatlands at mesotope level. Surface water flow paths for each site were determined from the DEM at 0.25 m resolution from 2017 (Cantabria Government, 2019) using the hydrology tools in ArcGIS 10.7. Individual mesotope units were identified and mapped for each site from the hydrological flow patterns and peat depth (Ivanov, 1981; Lindsay, 2010).

3.3. RESULTS

3.3.1. Identification

3.3.1.1. Climatic conditions and topographical analysis

Analysis of precipitation excluded large areas with the potential for blanket bogs mainly in the south and west of Cantabria and the south of the Basque Country where precipitation was below 1,000 mm yr^{-1} (Figure 3.3). In terms of temperature, areas close to the coast had the highest annual maximum temperatures thereby reducing large areas of potential blanket bogs in these areas. Interestingly, some areas of Asturias where a designated blanket bog is present (Figure 3.1), were identified as having precipitation and temperatures that were suitable for this habitat and adds confidence to the method adopted in this research. However, when temperatures are examined by seasons, it is apparent that maximum temperatures across almost the entire area are greater than 15 °C (Figure 3.3). However, it is important to highlight that the climate models do not include the influence of the fog and occult precipitation, which are key for blanket bog in northern Spain (Heras and Infante, 2018).

With the inclusion of topography, a total area of land between Picos de Europa and the Pyrenees across four regions in northern Spain covering 331,517 ha was identified as being suitable for blanket bogs (Table 3.3; Figure 3.4). This is particularly interesting given that models predicting presence of blanket bogs do not include Spain as a potential area for this habitat (Gallego-Sala and Prentice, 2013).

Table 3.3. Final potential areas suitable for blanket bogs by regions.

Region	*Area (ha)*
Basque Country	60,471
Navarra	74,436
Cantabria	84,822
Castilla y León	111,788
Total	**331,517**

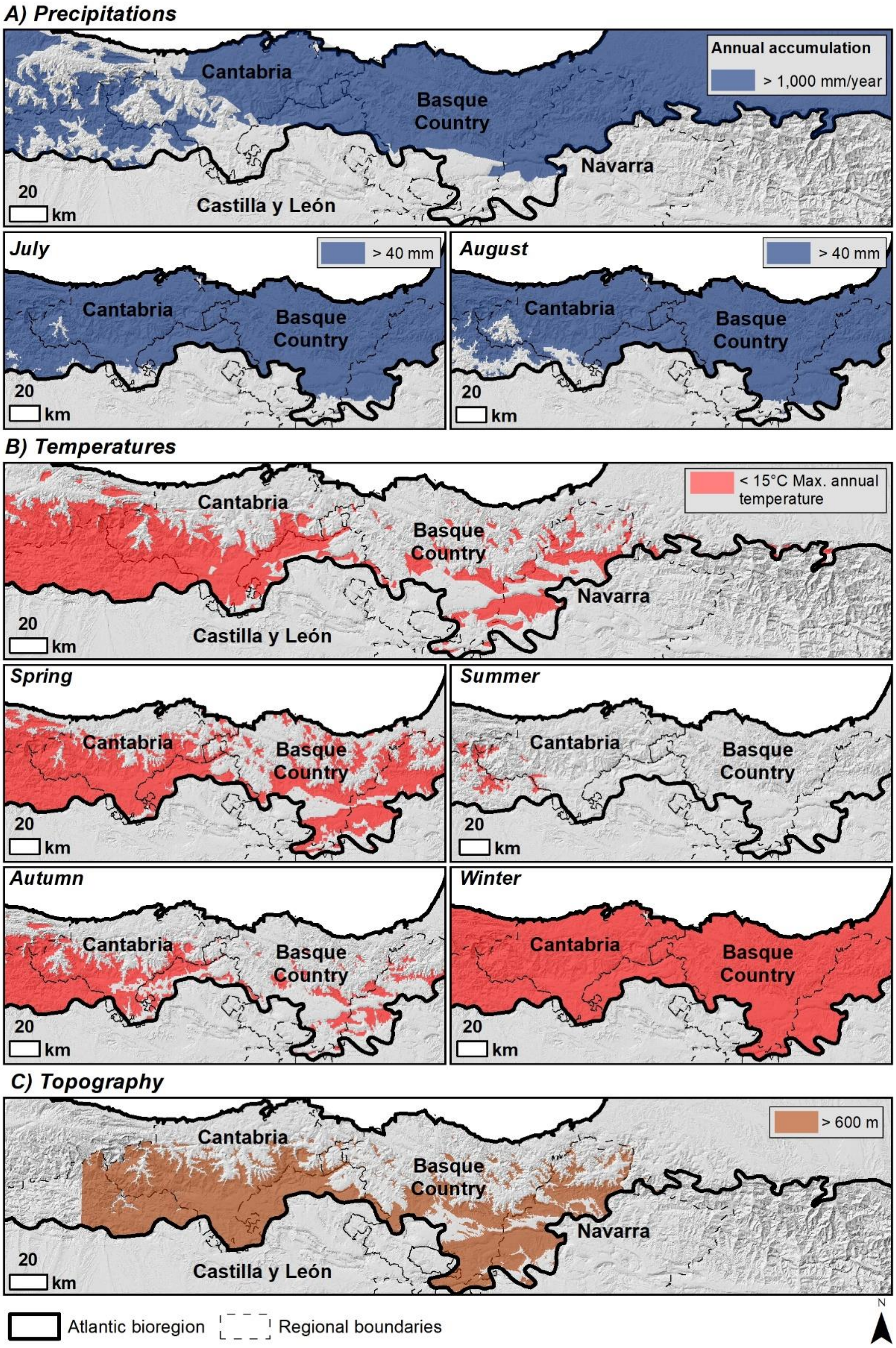


Figure 3.3. Analysis of suitable areas for blanket bogs development between Picos de Europa (Asturias) and Pyrenees (Navarra). A) Precipitations, B) Temperatures, C) Topography.

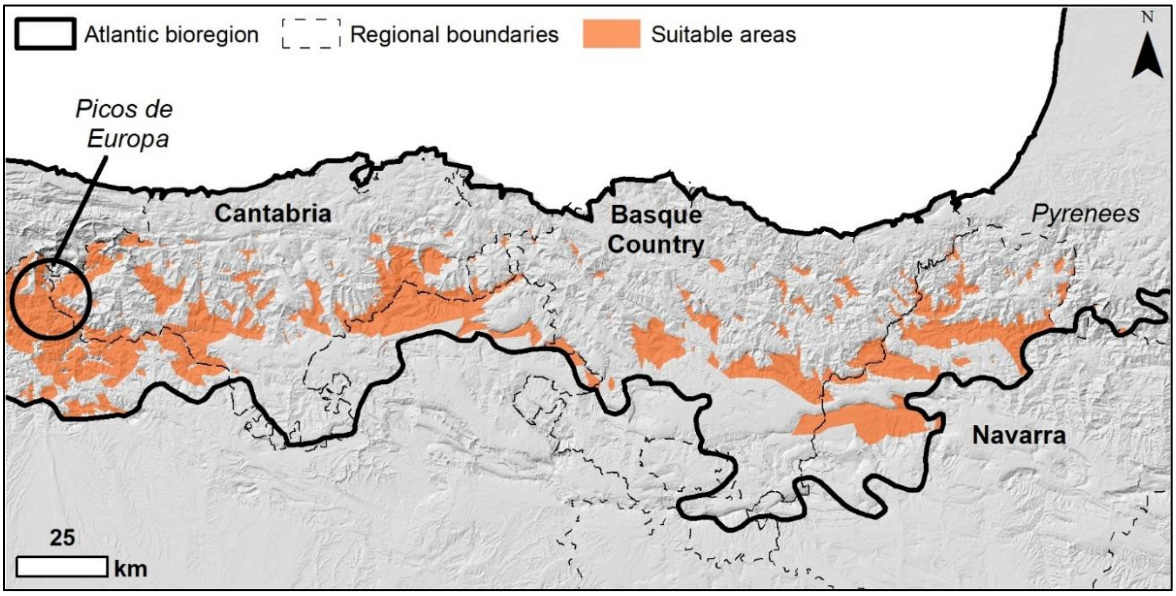


Figure 3.4. Final suitable areas for blanket bogs.

3.3.1.1.1. Identification of peatland features using aerial photos

Within the area identified by climatic and topographic analysis, a total of 42 potential blanket bogs were initially identified in three of 48 square 30 km x 30 km sectors (Table 3.4; Figure 3.5). The most common evidence indicating the presence of a blanket bog was visible erosion features or exposed peat. Only a few areas had visible pool systems such as the recognised Zalama blanket bog located between the regions of Castilla y León and the Basque Country.

Although more than 74,000 ha of the region of Navarra were identified as suitable for blanket bog (Table 3.3), no evidence of a single peatland was visible in the region from aerial photography. Only one blanket bog was identified in the Basque Country at Zalama, although the mire complex at Salduero located 2.5 km north-east may contain blanket bog elements. Salduero is, however, largely degraded and the peatland is predominantly fen (Chico and Clutterbuck, 2019) and therefore was excluded. The greatest number of potential blanket bogs identified in this research were located in the regions of Cantabria and Castilla y León, although the majority of these are located on the border of both regions along the administrative boundaries.

3.3.1.1.2. Ground validation of final areas

During ground assessment a large number of the initial 42 areas identified were reduced or removed (Table 3.4). The most common reason was where peat was found to extend between identified areas thereby indicating that these form part of the same peatland. Three areas were so severely degraded that there was almost no peat remaining, and two areas were found to be fens.

Table 3.4. Final number of blanket bog areas determined after ground assessment.

	Number of blanket bogs
Initial aerial photos exploration	42
Same blanket bog unit	-19
Degraded (peat extraction, windfarms, others)	-3
Requires ground assessment	-3
Other peatland type (e.g. fens)	-2
Final blanket bog areas	**15**

A total of 15 blanket bogs were confirmed after ground assessment although it should be noted that three additional areas still require ground assessment and need to be reported and assessed in the future. The peat surveys focused in two main areas, the Ordunte Sector and the Cantabria Sector (Figure 3.5). The Ordunte sector only contains two blanket bogs: the protected and designated Zalama blanket bog and less than 500 m to the west an unprotected and degraded area. In the Cantabria Sector, a large number of blanket bogs are concentrated along the hill summit and would appear to represent the southernmost boundary of blanket bogs in Europe.

The vegetation at all sites contained peat-forming species such as hare's-tail cotton grass (*Eriophorum vaginatum*), common cotton grass (*Eriophorum angustifolium*) and *Sphagnum* spp. including *Sphagnum palustre*, *Sphagnum fallax*, *Sphagnum denticulatum*, *Sphagnum cuspidatum* and *Sphagnum compactum.* Heather (*Calluna vulgaris*), cross-leaved heath (*Erica tetralix)* and bilberry (*Vaccinium myrtillus*) were abundant.

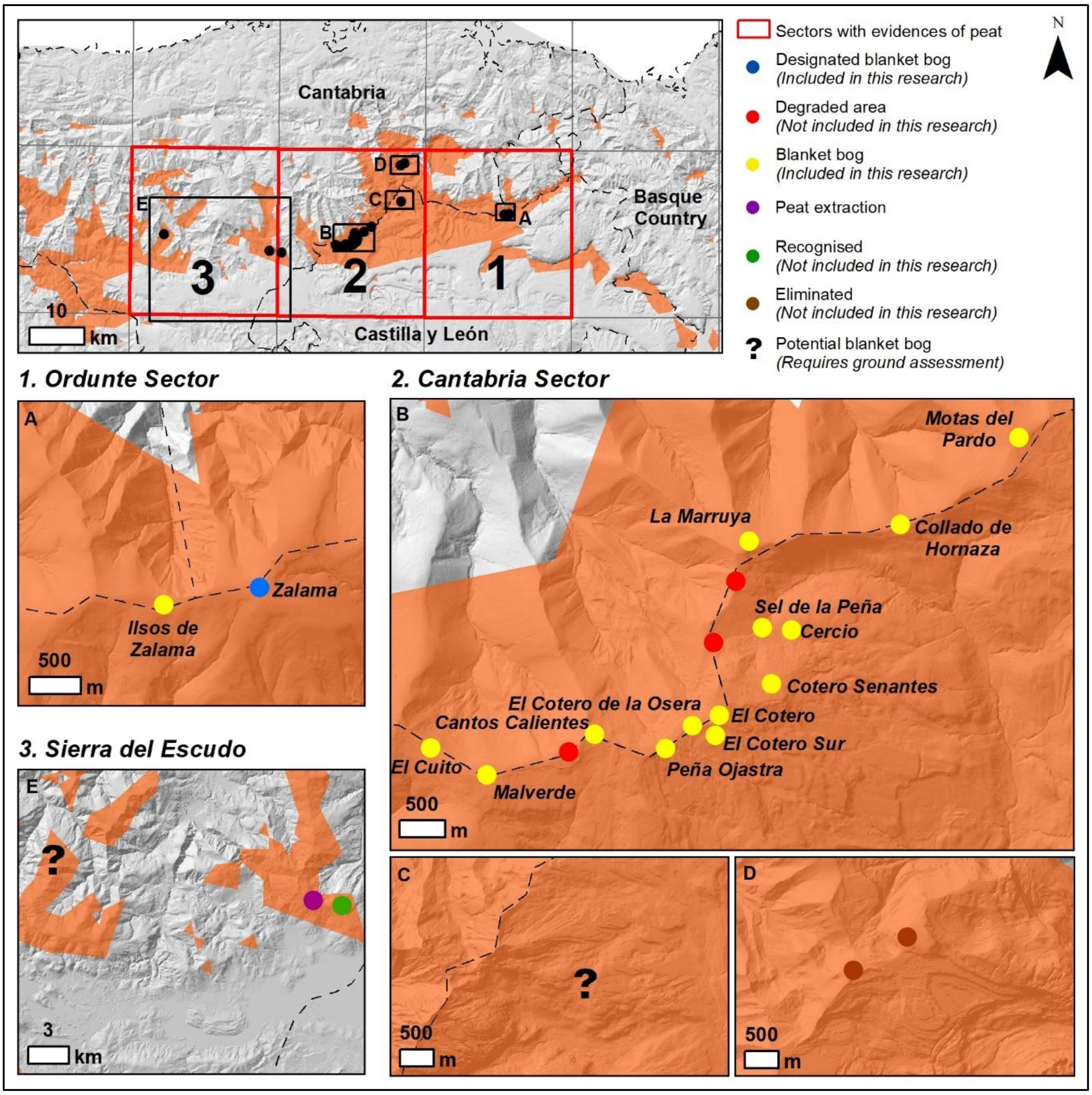


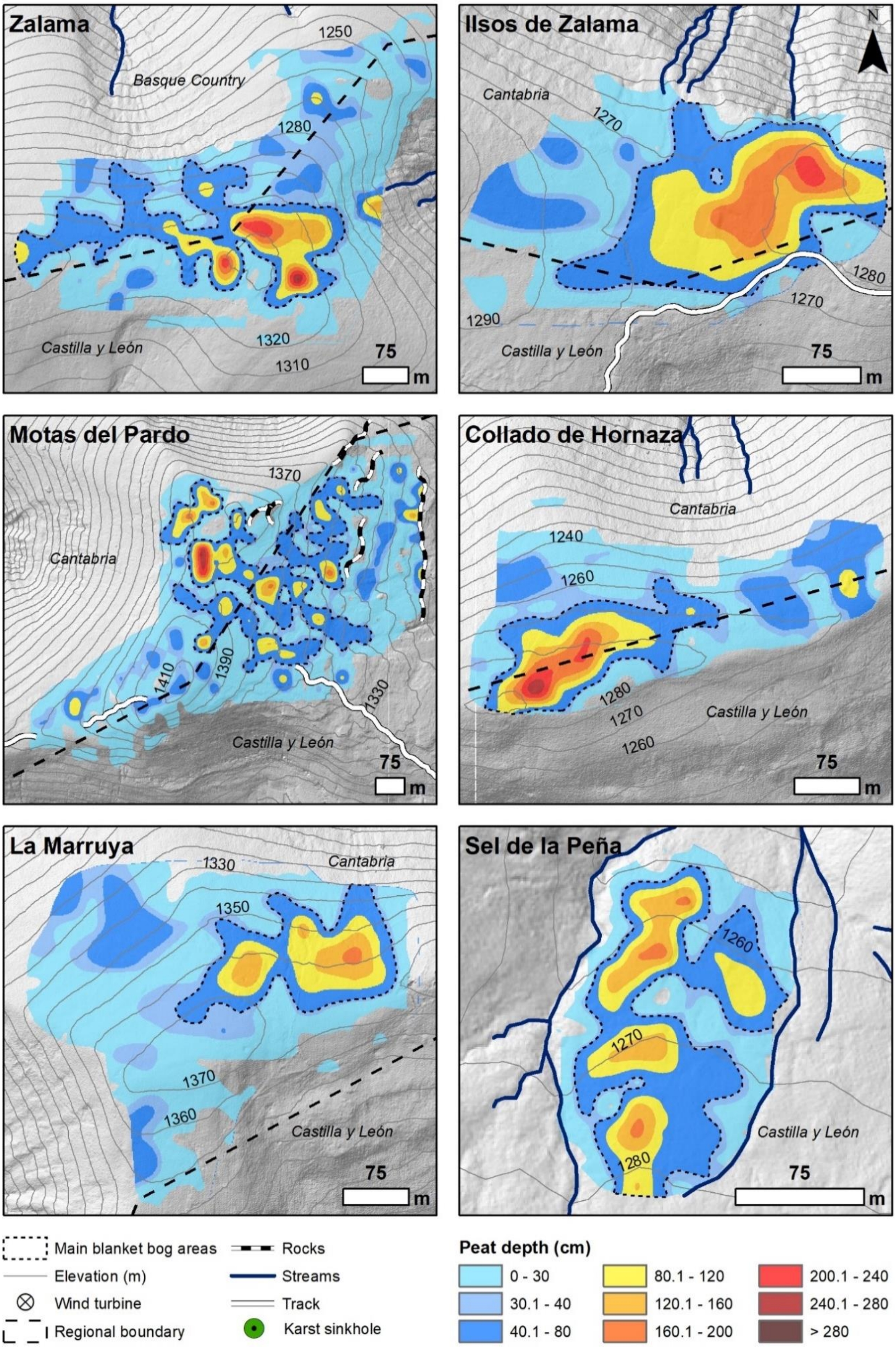
Figure 3.5. Study areas included in this research after ground assessment.

3.3.1.2. Peat depth, volume and peatland extent

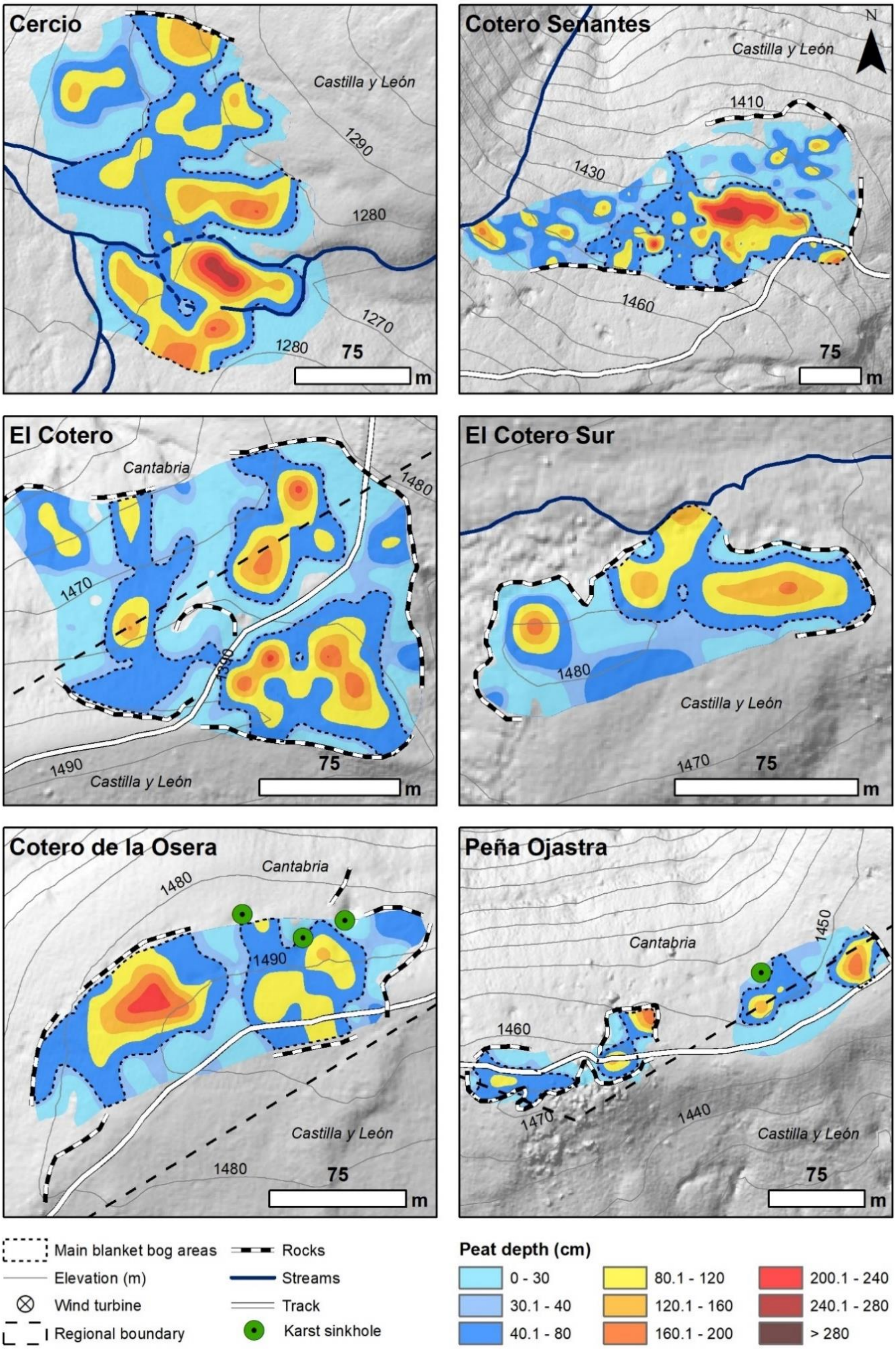
A total of 2,530 peat depth measurements were taken across the 14 currently unrecognised blanket bogs identified in this research and the one protected and restored blanket bog (Zalama). The blanket bog at Motas del Pardo covered the largest extent (10.86 ha), nearly twice the extent of Zalama blanket bog (6.49 ha) and Malverde, blanket bog (5.94 ha, Figure 3.6; Table 3.5). The maximum peat depth measured for all sites ranged from 1.61 m – 3.78 m, and interestingly the greatest peat depth was recorded at Malverde (3.78 m), over 1 m more than the greatest peat depth recorded at Motas del Pardo (2.65 m) (Table 3.5). Despite this observation, maximum peat depth, peatland extent and volume of peat were all significantly and positively correlated (see section 3.3.1.4). The mean peat depth determined for all sites was under 1 m and appears to show less variation than maximum peat depth (0.35 m – 0.80 m; Table 3.5). Using 40 cm peat depth as the limit of the peatlands, the total area of blanket bog mapped covers 44.5 ha, but increases to 64.7 ha if the margins are included (Table 3.5). The total volume of peat estimated to be contained across all sites was 554,266 m^3, an important value to enable estimation of the total amount of carbon stored in these blanket bogs in Chapter 4.

Table 3.5. Peatland characteristics by study area in the Cantabrian Mountains, Spain.

Site	*Survey area (ha)*	*Number of survey points*	*Altitude (masl)*	*Location (degrees)*	*Maximum peat depth (m)*	*Mean ± SD peat depth (m)*	*Peat extent (ha; >40 cm depth*	*Peat extent (ha; >30 cm depth*	*Peat volume (m^3)*
Motas del Pardo	49.50	516	1,390	43.1114 -3.7183	2.65	0.44 ± 0.37	10.86	20.00	153,198
Malverde	9.45	489	1,325	43.0782 -3.7887	3.78	0.84 ± 0.93	5.94	6.49	91,262
Zalama	21.37	225	1,330	43.1343 -3.4104	2.82	0.41 ± 0.43	6.49	9.87	74,341
Ilsos de Zalama	7.16	81	1,280	43.1327 -3.4214	2.16	0.80 ± 0.53	3.18	4.25	40,127
La Marruya	11.30	125	1,360	43.1015 -3.7543	1.73	0.45 ± 0.35	2.13	4.62	35,850
Collado de Hornaza	7.28	84	1,280	43.1036 -3.7324	2.75	0.58 ± 0.52	3.15	4.15	34,747
Cotero Senantes	6.14	277	1,413	43.0854 -3.7490	2.47	0.64 ± 0.52	3.47	4.26	34,414
Cercio	3.25	129	1,271	43.0923 -3.7489	2.71	0.75 ± 0.53	2.00	2.26	20,624
El Cuito	2.56	115	1,228	43.0813 -3.7989	1.61	0.71 ± 0.43	1.67	1.87	15,657
El Cotero	2.71	121	1,474	43.0840 -3.7577	2.12	0.58 ± 0.47	1.41	1.76	13,671
Sel de la Peña	2.21	102	1,246	43.0931 -3.7522	1.72	0.66 ± 0.43	1.24	1.47	11,908
Cantos Calientes	2.31	104	1,427	43.0807 -3.7781	1.78	0.63 ± 0.41	1.01	1.29	9,903
Cotero de la Osera	1.39	63	1,492	43.0832 -3.7628	2.06	0.69 ± 0.47	0.91	1.08	8,732
Peña Ojastra	1.32	62	1,452	43.0809 -3.7657	1.97	0.35 ± 0.18	0.56	0.74	5,559
El Cotero Sur	0.77	37	1,481	43.0824 -3.7580	1.87	0.68 ± 0.52	0.43	0.54	4,273
						Totals	**44.45**	**64.65**	**554,266**



Zalama
Basque Country
1250
1280
1320
1310
Castilla y León
75
m
Ilsos de Zalama
Cantabria
1270
1280
1270
1290
Castilla y León
75
m
Motas del Pardo
1370
Cantabria
1410
1390
1330
Castilla y León
75
m
Collado de Hornaza
Cantabria
1240
1260
1280
1270
1260
Castilla y León
75
m
La Marruya
1330
Cantabria
1350
1370
1360
Castilla y León
75
m
Sel de la Peña
1260
1270
1280
Castilla y León
75
m
Main blanket bog areas
Elevation (m)
Wind turbine
Regional boundary
Rocks
Streams
Track
Karst sinkhole
Peat depth (cm)
0 - 30
30.1 - 40
40.1 - 80
80.1 - 120
120.1 - 160
160.1 - 200
200.1 - 240
240.1 - 280
> 280



Cercio
Castilla y León
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Cotero Senantes
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Castilla y León
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El Cotero
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El Cotero Sur
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Castilla y León
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Cotero de la Osera
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Castilla y León
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Peña Ojastra
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Castilla y León
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Main blanket bog areas
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Wind turbine
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160.1 - 200
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> 280

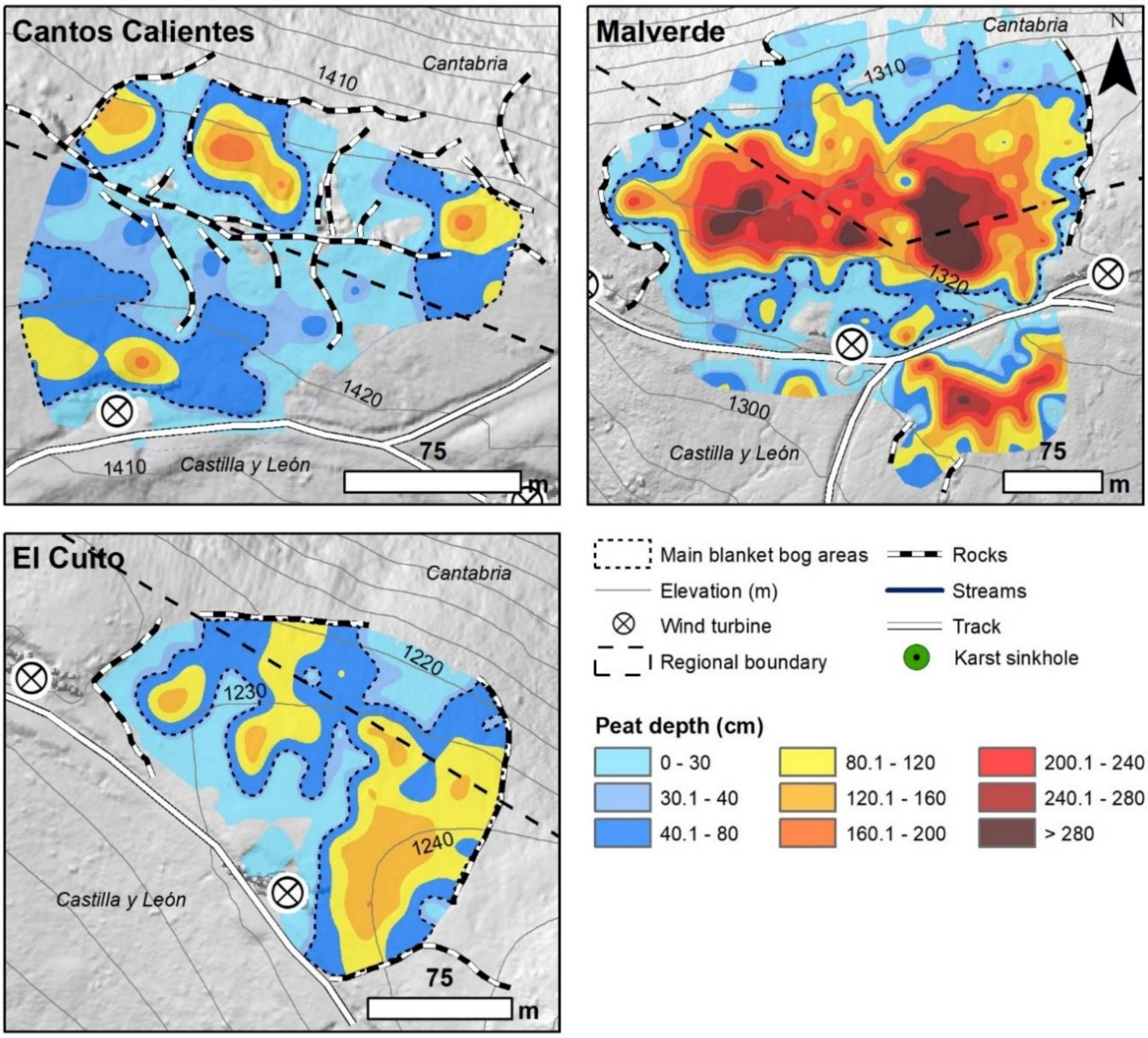


Figure 3.6. Peat depth map for each blanket bog identified in this research in the Cantabrian Mountains, northern Spain.

3.3.1.3. Landscape characteristics and anthropogenic influences

All 15 blanket bogs assessed in this research were located at an altitude of over 1,200 masl (Table 3.5). The mean slope across the blanket bogs ranged from 11.6° to 18.8° and the dominant aspect of the main peat body was north – northwest, except for two blanket bogs at Cercio and Cantos Calientes where the main peat body had a south – southwest aspect (Table 3.6). Interestingly, the maximum peat depth in each peatland was always recorded in the area orientated on north facing slopes.

Evidence of anthropogenic pressures were visible in all peatlands with exception of Zalama blanket bog. Grazing livestock comprising horses, goats and cattle were seen at all blanket bogs during field surveys, but vegetation burning was only visible in the first field survey campaign in 2017 at Motas del Pardo, Collado de Hornaza and La Marruya. However, evidence of historical burning activities was visible in all peat cores collected (see section 4.3.1). Windfarm infrastructures were present at Cantos Calientes, Malverde and El Cuito, the most westerly peatlands assessed in this research.

Table 3.6. Landscape characteristic for each blanket bog identified.

Site	*Mean ± SD slope (degrees)*	*Dominant aspect*	*Dominant aspect in relation with max. peat depth*	*Livestock*	*Windfarms*	*Tracks*	*Burning*
Zalama	14.5 ± 8.1	N	N				
Ilsos de Zalama	11.6 ± 8.4	E	NW	✓		✓	✓
Motas del Pardo	12.3 ± 6.6	E	NW	✓		✓	✓
Collado de Hornaza	17 ± 8.1	N	NW	✓			✓
La Marruya	15.9 ± 6.7	NW	N	✓			✓
Sel de la Peña	18.8 ± 10.6	N	NW	✓			
Cercio	14.9 ± 8.4	SW	NW	✓			
Cotero Senantes	13.7 ± 8.4	NE	W	✓		✓	
El Cotero	14.7 ± 7.7	N	W	✓		✓	
El Cotero Sur	12 ± 8.8	N	NE	✓			
Cotero de la Osera	12.8 ± 7.7	NW	NW	✓		✓	
Peña Ojastra	14.3 ± 9.9	NW	NW	✓		✓	
Cantos Calientes	14.9 ± 11.8	S	N	✓	✓	✓	
Malverde	14.3 ± 9.8	N	N	✓	✓	✓	
El Cuito	12.6 ± 9.7	N	NW	✓	✓	✓	

3.3.1.4. Statistical analysis

Perhaps unsurprisingly there was a strong and highly significant positive correlation between the volume of peat and the peatland extent ($r = 0.98$, $p < 0.001$) and significant positive correlations between maximum peat depth and peatland extent ($r = 0.59$, $p = 0.02$) and between maximum peat depth and peat volume ($r = 0.62$, $p = 0.02$; Table 3.7). Of particular note was the identification of significant correlations between latitude and peatland extent ($r = 0.55$, $p = 0.03$) and peat volume ($r = 0.55$, $p = 0.04$) and between longitude and peatland extent ($r = 0.53$, $p = 0.05$) and peat volume ($r = 0.53$, $p = 0.05$; Table 3.7).

Due to high collinearity between latitude and longitude (0.9), the longitude variable was excluded from the GLM. All variables combined in the model explain 96 % of peat volume (Adjusted R-squared = 0.96547), although individually the peatland extent and maximum peat depth are the most influential (Table 3.8).

Table 3.7. Statistical analysis of peatland and landscape characteristics across all study areas. r values from Spearman test except ^{A} where normal distributed data was found and Pearson test was applied. * indicates significant.

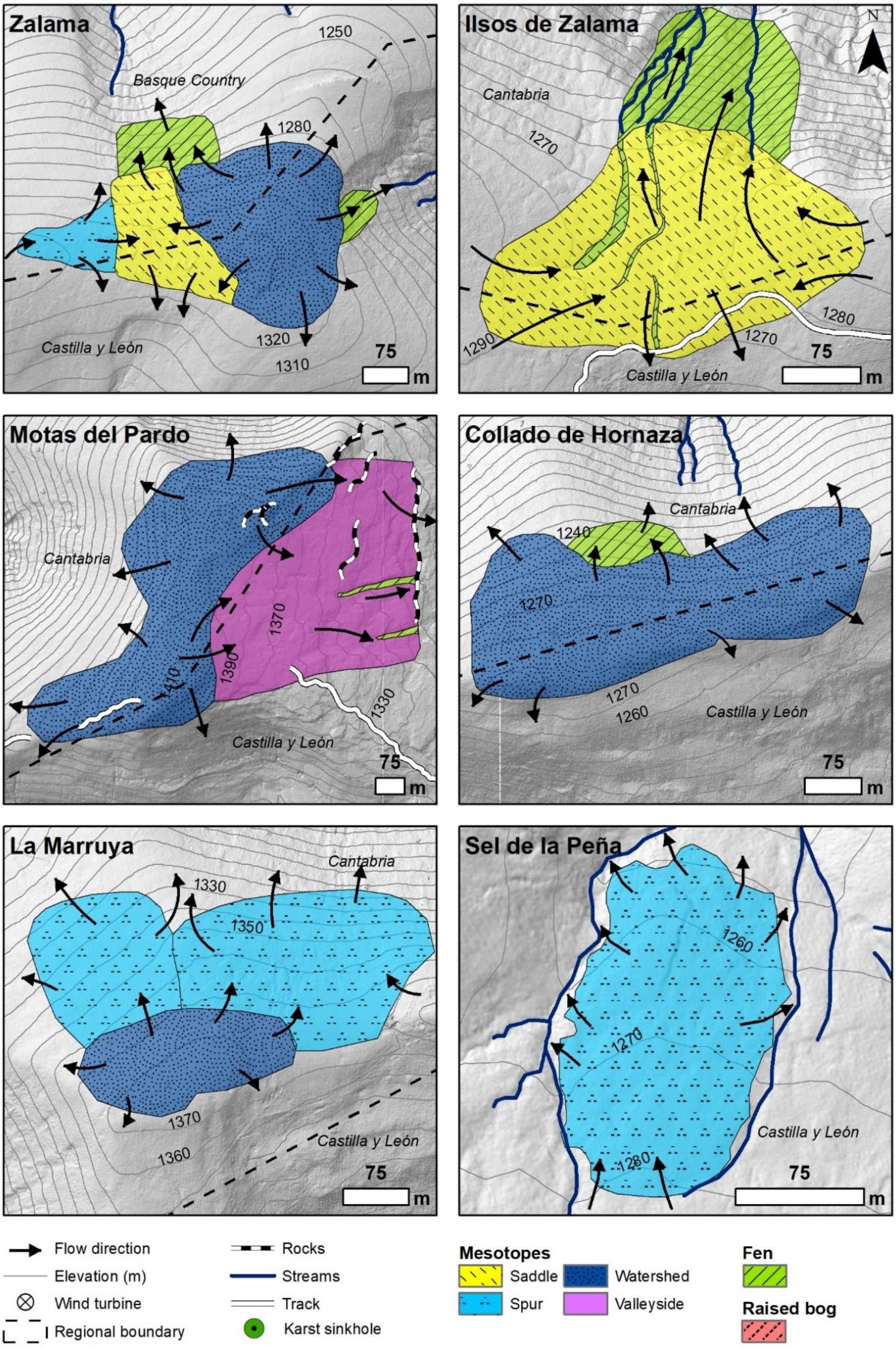
		Slope	*Maximum peat depth*	*Latitude*	*Longitude*	*Peatland extent (> 30 cm)*	*Volume*
Altitude	*r*	-0.34^{A}	-0.05	-0.31	-0.22	-0.40	-0.45
	p	0.22^{A}	0.86	0.26	0.43	0.14	0.09
Slope	*r*		-0.04	0.07	0.04	-0.001	-0.04
	p		0.89	0.80	0.88	0.99	0.89
Maximum peat depth	*r*			0.30	0.49	0.59	0.62
	p			0.28	0.07	0.02*	0.02*
Latitude	*r*				n/a	0.55	0.55
	p					0.03*	0.04*
Longitude	*r*			n/a		0.53	0.53
	p					0.05*	0.05*
Peatland extent (> 30 cm)	*r*						0.98
	p						<0.001*

Table 3.8. GLM to study the variables influence in the peat volume across all study areas. * indicates significant.

	Estimate	*SE*	*T value*	*Significance*
Intercept	1.97^{+07}	2.20^{+07}	0.90	0.397
Altitude	-3.79^{+01}	2.82^{+01}	-1.34	0.216
Slope	-6.13^{+02}	1.39^{+03}	-0.44	0.671
Maximum peat depth	1.34^{+04}	4.74^{+03}	2.83	0.022*
Latitude	-4.53^{+05}	5.06^{+05}	-0.90	0.396
Peatland extent (> 30 cm)	7.63^{+03}	8.48^{+02}	8.99	< 0.001*

3.3.2. Geo-hydromorphological classification

3.3.2.1. Mesotopes


Zalama
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Basque Country
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Castilla y León
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Ilsos de Zalama
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Cantabria
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Castilla y León
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Motas del Pardo
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Castilla y León
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Collado de Hornaza
Cantabria
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Castilla y León
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La Marruya
Cantabria
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Castilla y León
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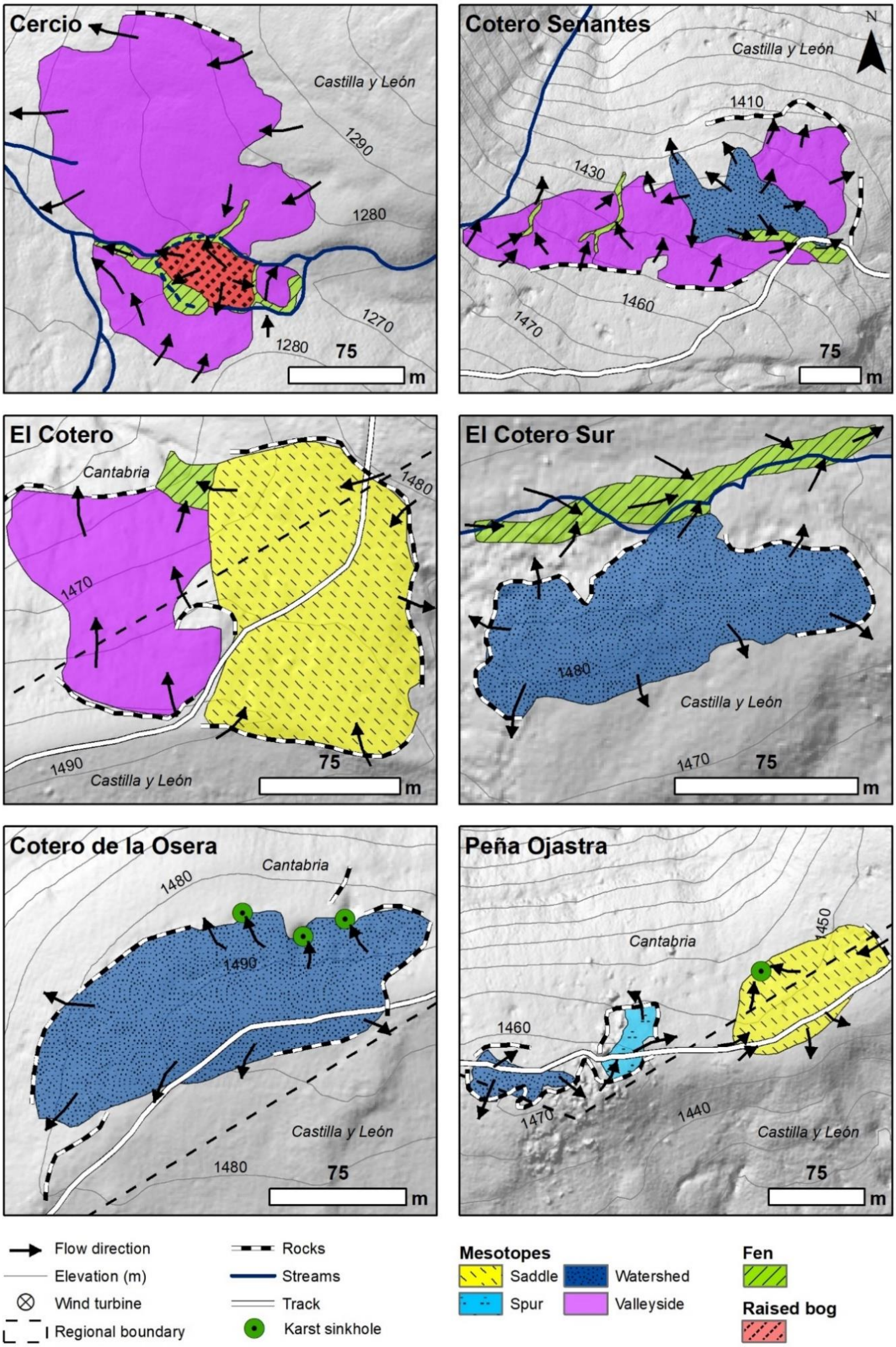
Sel de la Peña
1260
1270
1280
Castilla y León
75
m
Flow direction
Elevation (m)
Wind turbine
Regional boundary
Rocks
Streams
Track
Karst sinkhole
Mesotopes
Saddle
Watershed
Spur
Valleyside
Fen
Raised bog



Cercio
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Cotero Senantes
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El Cotero
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75 m
El Cotero Sur
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Castilla y León
1470
75 m
Cotero de la Osera
Cantabria
1480
1490
Castilla y León
1480
75 m
Peña Ojastra
Cantabria
1450
1460
1470
1440
Castilla y León
75 m
Flow direction
Elevation (m)
Wind turbine
Regional boundary
Rocks
Streams
Track
Karst sinkhole
Mesotopes
Saddle
Watershed
Spur
Valleyside
Fen
Raised bog

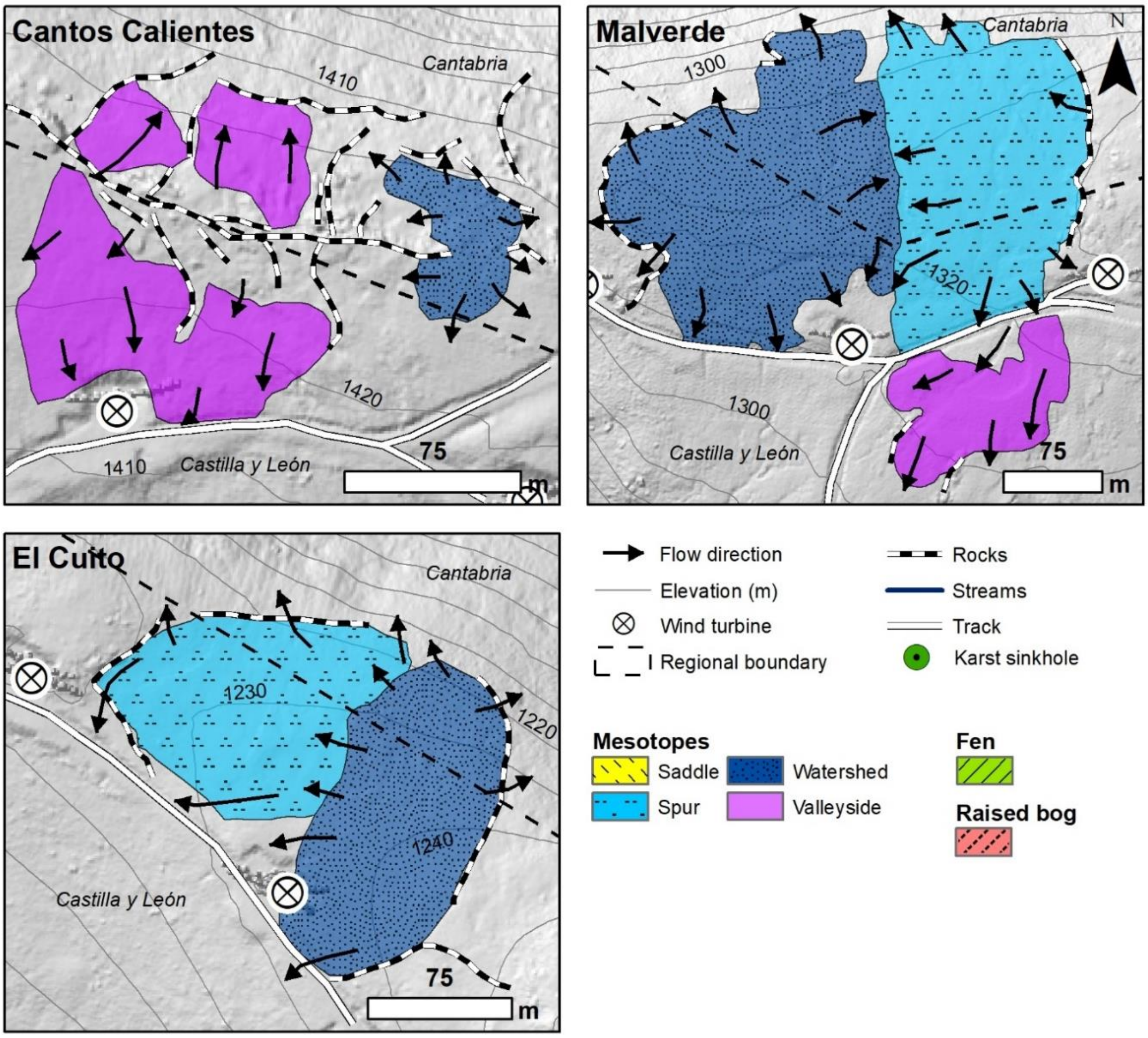


Figure 3.7. Mesotope units for each blanket bog identified in this research in the Cantabrian Mountains, northern Spain.

A total of 32 blanket bog mesotopes were identified across all the study areas. Watershed (11 units) and Valleyside (10) were the most common mesotopes followed by spur mesotopes (7). Saddle mesotopes were the least common observed (4). Areas of fen were present at 8 of the 15 blanket bogs, and at Cercio fen areas surround a potential raised bog element within the wider blanket bog (Figure 3.7)

3.4. DISCUSSION

Spanish blanket bogs are very scarce, accounting for less than 2% of the total area of peatland in Spain (Heras *et al.*, 2017; Tanneberger *et al.*, 2017). The identification of 14 currently unrecognised blanket bogs in the Cantabrian Mountains in this research highlights the importance of this area of Spain for peat formation, and these blanket bogs clearly need inclusion in the national peatland inventory (Ramil-Rego *et al.*, 2017). In addition, blanket bogs are currently not recorded in the administrative region of Cantabria (Pontevedra-Pombal *et al.*, 2009) so the fact that 12 of the blanket bogs identified and mapped in this research are partly or entirely in Cantabria is a significant gain in terms of habitat diversity for this administrative region. Furthermore, these newly described blanket bogs further our understanding of the distribution of this habitat in Europe. The blanket bog mapped at Malverde represents the southernmost edge-of-range of this habitat on the European continent, extending the limit of blanket bog 5 km farther south from Zalama blanket bog (Heras *et al.*, 2017). The blanket bog identified at Cotero de la Osera is also the highest blanket bog currently recorded in Spain (1,491 masl) and all the blanket bogs mapped in this research are at a higher altitude than blanket bogs elsewhere in Spain (Table 3.9).

Table 3.9. Comparison of the highest altitudes of blanket bogs across Spanish regions.

Blanket bog	*Region*	*Altitude (masl)*	*Reference*
Cotero de la Osera	Cantabria	1,491	This research
El Cotero Sur	Castilla y León	1,481	This research
Zalama	Basque Country	1,330	(Heras, 2002)
Serra do Xistral	Galicia	1,032	(Gómez-orellana *et al.*, 2014)
Sierra Plana de la Borbolla	Asturias	200	(European Environment Agency, 2019)

Although the largest examples and best studied blanket bogs in Spain are located in Galicia, the new areas of blanket bogs identified in this research represent a significant proportion of known blanket bogs in Spain filling an important gap in the Spanish inventory of this habitat between the regions of Asturias and Navarra. The evolution, diversity and origin of these newly described blanket bogs could help our understanding not only of the Spanish distribution of blanket bogs, but also provide key information about this habitat (7130) in a European context. These blanket bogs may even provide insight to the evolution of blanket bogs under climate change predictions. The area of blanket bog mapped in this research is equivalent to 10.5% of the area of currently protected blanket bogs in Spain (excluding the miss-classified areas in Asturias; Ramil-Rego *et al.*, 2017). Three further areas of peatland that were not visited in this research may also be blanket bog (Figure 3.5), and a full update of the inventory is needed across all of northern Spain to standardise the descriptions of blanket bog and to quantify the scale of carbon storage and the potential to function as carbon sinks.

3.4.1. Climatic and topographic influence on peat formation

In terms of peatland extent and peat accumulation, there are notable differences between the blanket bogs located in the northwest of Spain (Galicia) and the blanket bogs located along the hill summits of the Cantabrian Mountains. The largest blanket bogs are located in Galicia, accounting for more than 75% of the total designated areas (European Environment Agency, 2019) including the unrecognised blanket bogs mapped in this research. The extent and number of blanket bogs appears to reduce from the northwest to the northeast of Spain and peat accumulations are also greater in the northwest. In blanket bogs in Galicia, peat depth up to 3 to 5 m is commonly recorded (Pontevedra-Pombal and Martínez-Cortizas, 2004; Pontevedra-Pombal *et al.*, 2017), although lower values have also been noted (e.g. 1.8 m at Pena de Cadela; Pontevedra-Pombal and Martínez-Cortizas, 2004). It is interesting to note that, although not significant, there was a positive correlation between longitude and maximum peat depth in this research ($r = 0.49$; $p = 0.07$; Table 3.7) suggesting that this gradient of peat accumulation increasing westerly is detectable over a relatively short distance (30 km). While the maximum depth of peat recorded at Malverde (3.78 m) is comparable to some blanket bogs in Galicia, the mean maximum peat depth in the blanket bogs in the Cantabrian sector (2.2 m) and Ordunte sector (2.5 m) are lower in comparison with the Galician blanket bogs (Pontevedra-Pom*bal et al*., 2017). Altitude may also be a key factor influencing peat depth. Although not significant, the extent and volume of peat accumulated in the peatlands mapped in this research were both negatively correlated with altitude, suggesting that at higher altitudes, blanket bogs tend to be smaller and therefore accumulate less peat explaining why Galician blanket bogs are bigger as they are located in lower altitudes (Table 3.5). However, solid conclusions about the influence of altitude on peat accumulation should not be drawn from these data and the analysis would benefit from the inclusion of more blanket bogs; with particular interest in those located in Galicia. The significant positive correlations between both longitude and latitude and peat extent and peat volume support the suggestion that peat accumulation increases in both westerly and northerly directions. This highlights the role of climate as oceanic conditions are found nearer the coast, and on the Iberian Peninsula, the climatic zone changes from Oceanic to Mediterranean in a south easterly direction with the Cantabrian Mountains playing a key role in this climatic division.

Blanket bogs in the Cantabrian Mountains are smaller than those found in other countries such as Ireland or United Kingdom where this habitat can cover large expanses of the landscape (e.g. the Flow Country in Scotland or County Mayo in Ireland; Foss and O'Connell, 2017; Lindsay and Clough, 2017). Spanish blanket bogs are usually confined to hill summits where climatic conditions and topography are favourable for waterlogging conditions thereby allowing peat formation (see section Definition of peat and peatland). Waterlogging conditions usually result from high precipitation; however, the input from cloud or occult precipitation is also important for Spanish blanket bogs (Heras *et al.*, 2017), particularly during summer months when maximum temperatures are higher and therefore pose a potential climatic pressure to blanket bogs. Occult precipitation predominantly arrives from the Atlantic Ocean, when the ascent of a cold air mass reaches the dew point over the hill summits where blanket bogs are usually located. The potential contribution of occult precipitation to the blanket bogs in the Cantabrian Mountains is evident from topographical analysis. Out of the 15 blanket bogs mapped in this research, 11 are orientated NW-NE facing the Atlantic Ocean, two are south facing (Cercio and Cantos Calientes) and two east facing (Ilsos de Zalama and Motas del Pardo). Of particular note, however, is that the maximum peat depth at the south and east facing blanket bogs was recorded on north facing slopes (Table 3.5), suggesting a strong influence of water sources arriving from the north on all the peat accumulations. This phenomenon is also influential for other blanket bogs areas across the world; for example, in Newfoundland (Canada), where occult precipitation through fog is crucial for the formation of peatlands (Price, 1992).

Slope also seems to have a strong influence on the development of the peatlands mapped in this research. Blanket bogs usually develop on low angle slopes (Gorham, 1957), but can cover areas with slopes up to 22° (Tallis, 1973). However, peat accumulation on steep slopes tend to be unstable causing bogs bursts and often act as the limit of the blanket bog development (Pearsall, 1956; Gorham, 1957; Tallis, 1973). This is evident at several blanket bogs in this research. For example, at Motas del Pardo (Figure 3.6; Appendix A) steep slopes on the NW side of the blanket bog limit the extent of peat downslope, but on the low angle slopes (<10°) on the east side of the peatland the peat extends for around 500 m downslope until it merges with other type of peatlands, such as fens. As 'flat' areas are relatively scarce along the summits of the Cantabrian Mountains, the high slopes likely act as the geomorphological limit of the peatland, hence explaining their small extent. In contrast, hill summits in Galicia are larger and therefore provide larger 'flat' areas, so topography, location (longitude and latitude) and altitude may all be key to explaining blanket bog formation in north Spain.

Lastly, but not less important in terms of landscape characteristics, other geomorphological landforms such as rock outcrops and karst sink holes could be acting as the peatland limit and may also help to explain the south facing predominance of peat at Cercio. This peatland has a large number of rock outcrops that extend from the hill summit down the south slope. It is possible that these outcrops help to transfer water from the summit downslope thereby increasing peat formation in this area. On the other hand, the rock outcrops also seem to be the limit of many blanket bogs in this research (Figure 3.6) and the karst sink holes may be acting as drainage features in the peatland; however, sink holes are a frequent feature within blanket bog landscapes and are not reported to prevent peat formation (Smart *et al.,* 2014). Additionally, the comparatively low elevation of the main hill summit at Cercio could also be altering water sources, in particular by allowing occult precipitation to pass beyond the mountain ridge at this point and extend further south.

3.4.2. Geo-hydromorphology

The range of mesotopes mapped across all blanket bogs in this research demonstrates a large diversity of hydrological units and thus the importance of this area in contributing to the range of types of this habitat in Spain, Europe and more widely. However, further research is needed to understand the wider mire complexes and the interconnections between blanket bog mesotopes and the minerotrophic fen areas. The potential raised bog within the blanket mire complex described at Cercio is an interesting geomorphological unit, although these are not uncommon in Spain. In fact, in Galicia, some examples such as Chao de Veiga Mol have a similar topographical location, but peat accumulation is very different. Peat depth in raised bogs in Galicia can reach up to 9.2 m (Pontevedra-Pombal *et* *al.*, 2019) compared to the 2.7 m measured at Cercio raised bog unit. Perhaps the best description for this unit is an intermediate raised bog (Lindsay, 2016a) where the underlying topography (bedrock) defines the bog surface with a raised unit that is not strongly related to the surrounding fen areas. However, it is not clear if the current morphology of the unit is the 'natural' morphology. Anthropogenic pressures visible across the study sites including grazing livestock may have altered the morphology creating 'new' hydrological units and thereby adding complication to geo-hydromorphological classification. This is particularly evident at Malverde, where a vehicle access track for a windfarm, and the foundations for two turbines, have removed part of a watershed mesotope and split a spur mesotope into a valleyside and spur mesotope (Figure 3.8).

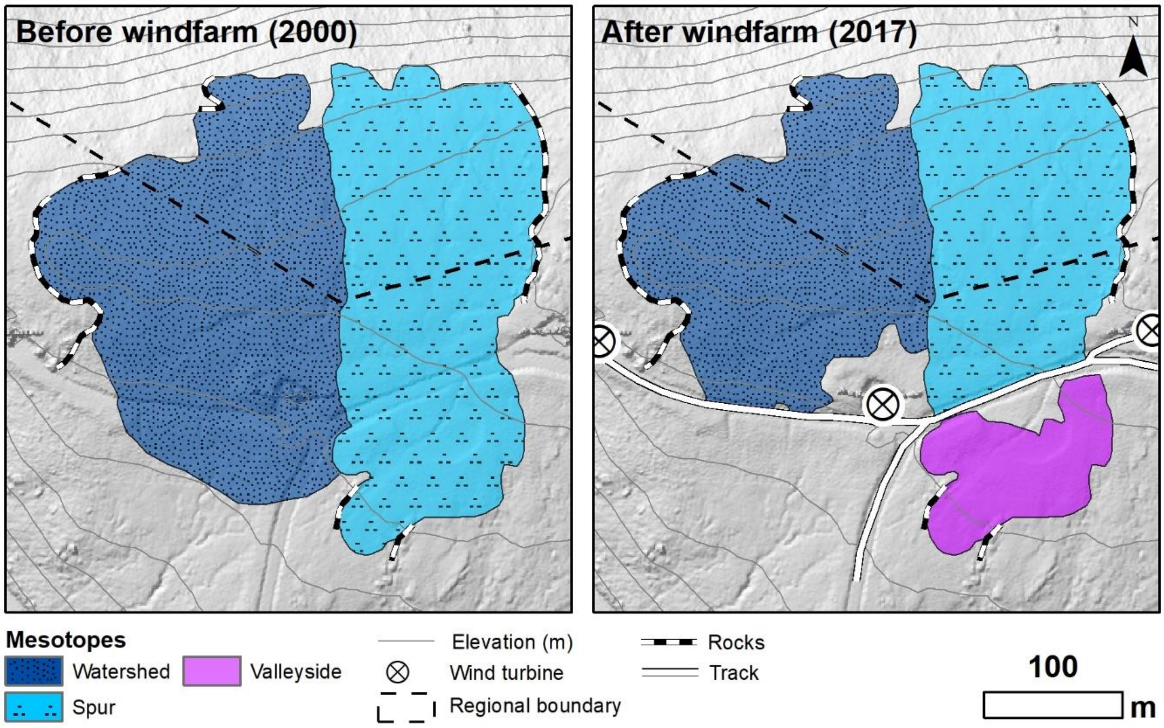


Figure 3.8. Mesotope units before (2000) and after windfarm development (2017) at Malverde blanket bog using DEM (Instituto Geográfico Nacional, 2019).

3.4.3. Influence of anthropogenic pressures

A range of anthropogenic pressures have the potential to influence the geomorphological and hydrological characteristics of Spanish blanket bogs, although in comparison with other blanket bogs in Europe fewer high impact pressures or threats are noted in habitat assessment (Table 3.10; European Environment Agency, 2019).

Table 3.10. Comparison between the principal pressures and threats (high impacts) for recognised blanket bogs under Natura 2000 network in the Atlantic region for the year period 2008-2018 (European Environment Agency, 2019)**.**

Country	*Main pressures and threats*
Spain	Peat extraction
United Kingdom	Overgrazing, burning, drainage & air pollution
Ireland	Overgrazing, burning, afforestation, peat extraction & agriculture
France	Air pollution & climate change

Historically, peat cutting was common in Spanish blanket bogs, although only for local use, potentially due to the difficulties associated with accessing mountain ranges (Heras, 2002). Interestingly, the impact of commercial peat extraction has become more important, especially since the 1940s, when peat extraction started to be more common, mainly for horticultural activities (Heras *et al.*, 2017). In the last decade, 429 kt of peat has been extracted in Spain (Heras *et al.*, 2017) and as a result, some raised bogs (e.g. Saldropo – Basque Country) and blanket bogs (Tornos – Cantabria) have been completely removed, significantly affecting the distribution of Spanish blanket bogs (Heras and Infante, 2008). In the area assessed in this study, three potential blanket bogs were so severely degraded that there was almost no peat remaining (section 3.3.1.1.2), and these all appear to have undergone peat extraction (e.g. Cueto de la Avellanosa – Cantabria; Figure 3.9A; Heras *et al.*, 2017). Peat extraction is not confined to the regions assessed in this study, and is ongoing even in protected and designated blanket bogs at Serra do Xistral in Galicia (Ramil-Rego *et al.*, 2017), and is having visibly significant impact on hydrological units and geomorphological characteristics (Figure 3.9B).

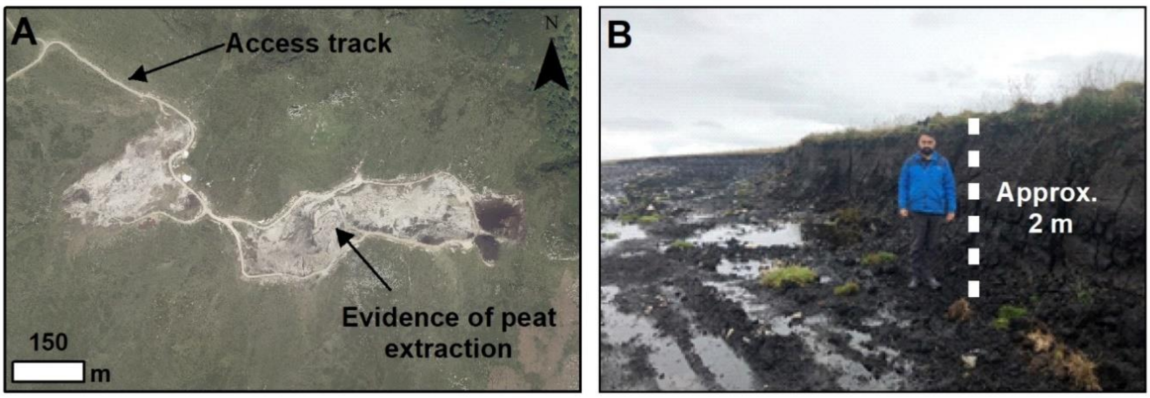


Figure 3.9. A) Peat extraction at Cueto de la Avellanosa (Cantabria); B) Peat extraction at Serra do Xistral (Galicia).

Although not mentioned in habitat assessment, the installation of windfarms along the hill summits of the Cantabrian Mountains is also becoming increasingly common. The foundations for wind turbines and associated infrastructures such as vehicle access tracks not only alter the hydrological function of the peatlands as is evident at Malverde (Figure 3.8) and in the United Kingdom (Wawrzyczek *et al.*, 2018), but also change the species biodiversity (Fraga *et al.*, 2008) and in some extreme cases, can destroy the habitat (Heras and Infante, 2008). In addition, peat extracted to construct stable turbine foundations and construct tracks, represent large-scale loss of peat and almost certainly changes the ability of the adjacent peatland to function as a carbon sink (at least in the short-term).

Other anthropogenic pressures including grazing and associated burning practices have been reported to initiate major erosion events in blanket bog in the United Kingdom (Tallis, 1997b; Table 2.1) and at the restored blanket bog included in this research, Zalama, 50% of the original peatland surface has been removed by natural erosion processes that are suggested to have been enhanced by overgrazing, prescribed burning and wildfire (Heras, 2002). Aeolian, fluvial or ice erosion inevitably also influence the geomorphological and hydrological characteristics of Spanish blanket bogs, but the rate of erosion of exposed peat surfaces in Spain has not been quantified to date.

3.5. CONCLUSION

This chapter has identified and provided geo-hydromorphological assessment of 14 formerly unrecorded blanket bogs and one recognised area (Zalama blanket bog) in the Cantabrian Mountains (north Spain). These blanket bogs represent the southernmost known limit of this habitat in Europe and may represent 10.5% of the blanket bogs currently recognised in Spain and the recognition of these blanket bogs would fill an important gap in the Spanish peatland inventory between the regions of Asturias and Navarra. This research has also identified the highest blanket bog known in Spain, identified the first blanket bogs in the region of Cantabria, and suggests that topography, location (latitude and longitude) and altitude combined with occult precipitation are key factors influencing the development and accumulation of peat in the Cantabrian Mountains. The 32 individual mesotopes mapped demonstrate a large diversity of hydrological units and thus highlight the importance of this area in contributing to the range of types of this habitat in Spain, Europe and more widely. However, despite the potential importance of these landforms for terrestrial carbon storage and associated palaeoenvironmental archive, high levels of anthropogenic pressures have had, and continue to have, substantial negative impacts on these newly identified areas affecting the described landforms and the hydrological units. The total volume of peat estimated to be contained across all sites was 554,266 m^3, but it is not clear how important Spanish blanket bogs are for carbon storage or how quickly this store is being lost. The amount of carbon stored in these bogs will be examined in Chapter 4, and the rate of degradation will be examined in Chapters 5 and 6.

Chapter 4

The extent of degradation and carbon stored in Spanish blanket bogs

The content of this chapter has been partially published

4.1. INTRODUCTION

Peatlands cover a small proportion of the Earth's land surface (Xu *et al.*, 2018), but they represent the largest terrestrial carbon store (Limpens *et al.*, 2008). However, an estimated 78 Mha of known peatlands are reported as damaged or degraded (Figure 2.7), and release 5–6% of global greenhouse gases (Joosten, 2009). Peatlands are mainly located in the subarctic and boreal zones in the Northern Hemisphere, and peatlands in tropical and temperate climatic zones contain most of the remaining carbon stored in these habitats (Gorham, 1991). Peatlands act as carbon sinks when they accumulate organic material at a faster rate than microbial decomposition processes can break it down (Gorham, 1957). Accumulation rates of peat vary significantly, largely due to climate conditions (Lindsay, 2010), but the presence of peat-forming vegetation species and waterlogging conditions are usually a good indicator of the formation of peat (Joosten *et al.*, 2017). It is not clear if the blanket bogs mapped in chapter 3 are actively forming peat or serving as carbon sinks, as all the blanket bogs identified and described in this research have evidence of degradation, at least in the form of erosion features and areas of exposed peat. However, peat-forming species and waterlogging conditions were found in all sites suggesting that environmental conditions are suitable for peatland formation.

To define the status or level of degradation in a peatland, the main variables of a peatland ecosystem (water, peat and vegetation) need to be considered (Schumann and Joosten, 2008). Early degradation classifications highlighted the importance of these variables and defined stages of degradation, although each variable could affect the peatland in a different way (Schumann and Joosten, 2008). For example, when the peat landform or deposits are affected (i.e. peat is lost), the status of degradation will be maximal and therefore, restoration efforts will increase (Schumann and Joosten, 2008). More recent degradation classifications categorise peatlands in four classes: active, degraded peat, bare peat and wasted peat (Bruneau and Johnson, 2014). Active peatlands are those with peat-forming vegetation that covers the peatland surface and the hydrology is unmodified. Bare peat simply refers to areas where vegetation has been removed, but the land use has not changed. Degraded peat is the transitional stage between an active and bare peat peatland, and finally, wasted peat is when the peatland has been heavily modified and peat-forming species are no longer present (Bruneau and Johnson, 2014). All the blanket bogs mapped in this research could be categorised as degraded peat as they have areas of bare peat, but also several areas where peat-forming species are present, despite some clear changes in land use and anthropogenic pressures (e.g. windfarm at Malverde; Figure 2.17).

Erosion of peat through natural, anthropogenic or sometimes both pressures, is one of the main issues in degraded blanket mires (Li *et al.*, 2018) and therefore, quantifying the extent of exposed peat in a peatland is critical to understand the scale of the problem. Exposed peat is common across European blanket mires, although there is considerable spatial variation. In the United Kingdom, the proportion of the area of blanket mire that is exposed peat varies between countries, with estimates suggesting a higher exposure in Wales than in Scotland or England (Table 4.1). However, these overall statistics mask significant variation within each country. In Scotland, for example, some areas of peatland in Caithness affected by gully erosion are estimated to have only 0.2% exposed peat, while in contrast, 76% of some peatlands in Inverness are exposed peat (Coupar, Immirzi and Reid, 1997).

Table 4.1. Percentage of exposed peat area within blanket mire across United Kingdom and Ireland.

Country	*% exposed peat areas based on total blanket mire cover*	*Reference*
Scotland	19%	(Coupar, Immirzi and Reid, 1997)
Wales	30%	(Marcus, 1997)
England (only Moor House)	20%	(Garnett and Adamson, 1997)
Ireland	27 – 33%	(Cooper and Loftus, 1998)
Northern Ireland	29%	(Cruickshank and Tomlinson, 1990)

Exposed peat has been reported to be a wide issue in blanket bogs in Spain (Heras and Infante, 2003), and is evident from both aerial and ground surveys in this research. However, there are currently no statistics on the level of degradation for Spanish blanket bogs, including the extent of exposed peat. Areas of exposed peat are vulnerable and a potential hot spot for carbon loss (Ward *et al.*, 2008) and direct anthropogenic pressures can also reduce the carbon trapped in the peatland surface (Garnett, Ineson and Stevenson, 2000; Ward *et al.*, 2007). These changes may affect the ability of peatlands environments to function as long-term carbon sinks (Gorham, 1991). To fully understand the significance of the scale of degradation of a peatland it is therefore also important to quantify the amount of carbon stored.

The aim of this chapter is to estimate the total carbon stored across the blanket bogs identified in chapter 3 and quantify the extent of the degradation on blanket bog peatland surfaces. These will enable better context of the loss of carbon from these peatlands to be determined in chapter 6. The following objectives were set:

a) Collect a peat core from as many peatlands as possible across the study area.
b) Determine the carbon content of each core using the Loss On Ignition (LOI) method.
c) Explore any variation in the carbon content of peat across the peatlands.
d) Estimate the total carbon stored in each blanket bog identified in this research.
e) Map the areas of exposed peat surface in each blanket bog area using the most recent aerial photography.
f) Assess the impact of restoration activities at Zalama blanket bog on the area of exposed peat using historical aerial photography.

This chapter has been partially published in one peer review publication; Chico et al., (2020) *Geo-hydromorphological assessment of Europe's southernmost blanket bogs*. Earth Surface Processes and Landforms, 45 (12), 2747–2760.

4.2. METHODS

4.2.1. Carbon stored

4.2.1.1. Peat cores collection

Five peat cores were collected from Zalama, Ilsos de Zalama, Collado de Hornaza, La Marruya and Malverde blanket bogs (Figure 4.1) in June 2018 and August 2019 using a 5 cm diameter semi-cylindrical Russian auger. The number of cores collected was restricted by permission from the local government bodies and the coring location was guided by the peat depth data obtained in previous chapter 3 (i.e. taken if possible where the peat depth was greatest). Zalama represents the most easterly blanket bog mapped in this research, and Malverde was selected over El Cuito (300 m west) as the most westerly blanket bog as

Malverde had the greatest peat depth. Malverde is also the most southerly blanket bog mapped in this research and represents the southernmost edge-of-range of this habitat in Europe (see section 3.3.1.2). Ilsos de Zalama was selected for comparison with the protected and restored Zalama 500 m east (Figure 4.1) and Collado de Hornaza and La Marruya were the other two sites where permission to collect a core was granted by the Cantabrian government. Peat cores were examined in the laboratory prior to analysis and a number of thin black layers of charcoal were evident throughout all cores indicating previous fire events. Charcoal analysis was not undertaken as the full core was required for determination of carbon content.

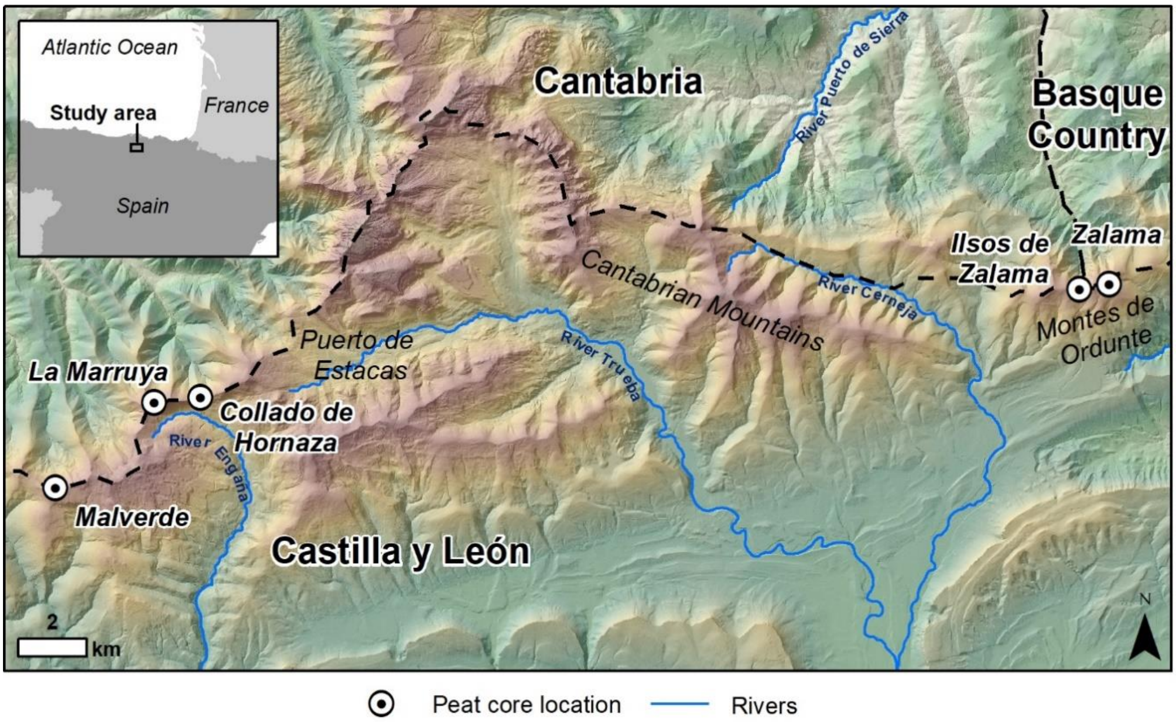


Figure 4.1. Location of peat cores obtained at locations in the Cantabrian Mountains, northern Spain.

The peat characteristics have been described previously at Zalama blanket bog with a range of different statuses depending on the peat depth (Pérez-Díaz *et al.,* 2016). Top layers of peat (up to 6 cm) were completely undecomposed with almost clear water when peat was squeezed (H1 – von Post scale) with the plant remains easily identifiable. Peat between 6 to 12 cm, 18 to 32 cm, and 120 to 160 cm also contained easily identifiable plant remains, but water was more yellowish. Some grade of decomposition has been reported in the rest of the peat core which was slightly decomposed between 12 to 18 cm and 86 to 94 cm, moderately decomposed between 32 to 48 cm and 94 to 112 cm, and highty decomposed between 48 to 86 cm and 112 to 120 cm. The bottom part of the core (160 to 232 cm) was practically fully decomposed (Pérez-Díaz *et al.,* 2016).

4.2.1.2. Estimation of carbon content (% Carbon, Bulk Density and Organic Carbon Content)

The total organic carbon in the peat cores was determined using the Loss On Ignition (LOI) method (Agus, Hairiah and Mulyani, 2011). For peat soils, this method has been used widely for estimating organic matter and % of carbon in peatlands environments globally (Chambers, Beilman and Yu, 2010; Loisel *et al.,* 2014); however, this method could be inaccurate in soils with high content on mineral or clay (Bhatti and Bauer, 2002) or if an incorrect correction factor is used. Since the mineral and clay layer in the peatlands presented appear minimal as at Zalama blanket bog (Heras, 2002), LOI has been selected due to the cost-effective approach and simplicity for estimation of organic carbon (Bhatti and Bauer, 2002). The correction factor used for this research has been widely used for boreal and temperature peat deposits, although this correction factor should vary if other peatland types are analysed (e.g. tropical peatlands; Paramananthan *et al.,* 2018). Peat cores were analysed in 5 cm sections and 3 subsamples in each 5 cm section were analysed to allow assessment of variation within each 5 cm section. Samples were analysed using porcelain crucibles that were weighed at all stages of the process using a Sartorius Entris224-1s balance. Peat samples were first dried in an oven at 105°C for 24 hours or until a consistent dry weight was achieved (M_s). The dry samples were subsequently burned at 550°C for 6 hours to remove organic matter leaving the residual or ash (M_{ash}). The volume of each section (5 cm) was determined from the dimensions of the auger and used to calculate dry bulk density (BD, Equation 1).

$$BD = \left\{\frac{M_s}{V}\right\} \tag{1}$$

where M_s is the dry mass of the peat section (g) and V is the volume of the sample (cm^3).

The organic carbon content (C_{org}) of the organic matter and weight of organic carbon per unit volume of peat (C_v) were then estimated using the generalized relationship between organic matter and carbon content (Agus, Hairiah and Mulyani, 2011; Equation 2 & 3).

$$C_{org} = \left\{\frac{M_s - M_{ash}}{M_s}\right\}/1.724 \quad (2)$$

where C_{org} = Organic carbon content of the organic matter (%), M_s = dry mass of the peat sample (g), M_{ash} = mass of sample remaining (ash) after LOI (g).

$$C_v = BD * C_{org} \quad (3)$$

where Cv = weight of organic carbon per unit volume of peat (g/cm^3).

4.2.1.3. Statistical analysis

The % carbon, BD and organic carbon content were compared for all cores to assess any differences between the peat at each blanket bog. This was undertaken using data from the whole core and from the top 1 m of the core to remove any potential impact of the bottom part of the core on the peat composition as this is where it is in contact with the bedrock. All data were tested for normality prior to analysis using the Shapiro-Wilk's test. For normally distributed data, a One-Way ANOVA test was undertaken followed by a Post HOC analysis using the Tukey HSD test to find individual differences between peatlands. For one dataset that was not normally distributed, a Krustal-Wallis rank sum test was undertaken followed by a Pairwise Wilcox test to define individual differences between peatlands. All statistical analysis was undertaken in R 3.6.2.

4.2.1.4. Total carbon stored

The volume of peat estimated for each peatland in chapter 3 (see section 3.3.1.2) was used to calculate the total carbon stored in each peatland. For the five peatlands where a peat core was taken, the weight of organic carbon per unit volume of peat was determined using that core. For the remainder of the peatlands, the weight of organic carbon was estimated using the peat core from the closest peatland with a peat core available (Table 4.2), rather than the mean of all cores as spatial variation in carbon content was notable across the cores. For all calculations, the weight of organic carbon per unit volume for the whole core was used.

Table 4.2. Location of peat core used at each site to estimate the stored total carbon content.

Site	*Peat core*
Zalama	Zalama
Ilsos de Zalama	Ilsos de Zalama
Motas del Pardo	Collado de Hornaza
Collado de Hornaza	
La Marruya	La Marruya
Sel de la Peña	
Cercio	
Cotero Senantes	
El Cotero	
El Cotero Sur	
Cotero de la Osera	Malverde
Peña Ojastra	
Cantos Calientes	
Malverde	
El Cuito	

4.2.2. Extent of degradation

4.2.2.1. Current extent of exposed peat

Orthorectified aerial photography from 2017 at 0.25 m resolution was acquired from Instituto Geográfico Nacional (2019) and was used to digitise the visible areas of exposed peat across all sites in ArcGIS 10.7. In order to compare the differences between peatlands, the area of exposed peat was standardised by dividing the total area of exposed peat by the total area of the peatland (>30 cm) defined in chapter 3.

4.2.2.2. Statistical analysis

To assess whether the level of peatland degradation is influenced by peatland characteristics, any potential relationships between the extent of exposed peat and altitude, slope, maximum peat depth, latitude, longitude, peatland extent and volume of peat were examined. As altitude and slope data were normally distributed relationships were examined using the Pearson correlation test in R 3.6.2. For the remainder of the variables (maximum peat depth, latitude, longitude, peatland extent and volume of peat), a Spearman rank test was undertaken as the data were not normal distributed. A Spearman rank test was also undertaken to explore relationship between all variables and the standardised area of exposed peat.

4.2.2.3. Impact of restoration actions on areas of exposed peat

In order to explore the impacts of restoration actions on the area of exposed peat, orthorectified historical aerial photography from 1977, 2002 and 2009 was acquired (Instituto Geográfico Nacional, 2019) and used to digitise changes in the area of exposed peat at the restored Zalama blanket bog, and for comparison at the unprotected and unrestored blanket bog Ilsos de Zalama located only 500 m from Zalama (Figure 4.1).

4.3. RESULTS

4.3.1. Carbon stored

4.3.1.1. % of Carbon

The carbon content of peat in all cores showed largely similar patterns, with a gradual and consistent increase with depth from the top and a rapid decrease at the base of the core where the peat is in contact with the bedrock (Figure 4.2). The most notable variation from this pattern was in the peat core from Ilsos de Zalama, where in the last 60 cm several alternating sections of increasing and decreasing carbon content were observed (Figure 4.2). This variation has clear impact on the distribution of the data as a higher number of outliers with comparatively low carbon content are present in this core compared to the other four (Figure 4.3). The mean carbon content in the cores from Zalama, Collado de Hornaza, La Marruya and Malverde only varied by 2% (between 53.2% and 55.2%), but the mean value for Ilsos de Zalama was notably lower (46.9%; Figure 4.3). The One-Way ANOVA identified a significant difference in the carbon content between the peat cores ($F = 4.918$, $p < 0.001$) and the Post *HOC* Tukey HSD (Honest Significant Differences) test identified that the peat from Ilsos de Zalama was significantly different to that from Zalama, Collado de Hornaza and La Marruya, but interestingly not significantly different to the peat from Malverde (Table 4.3).

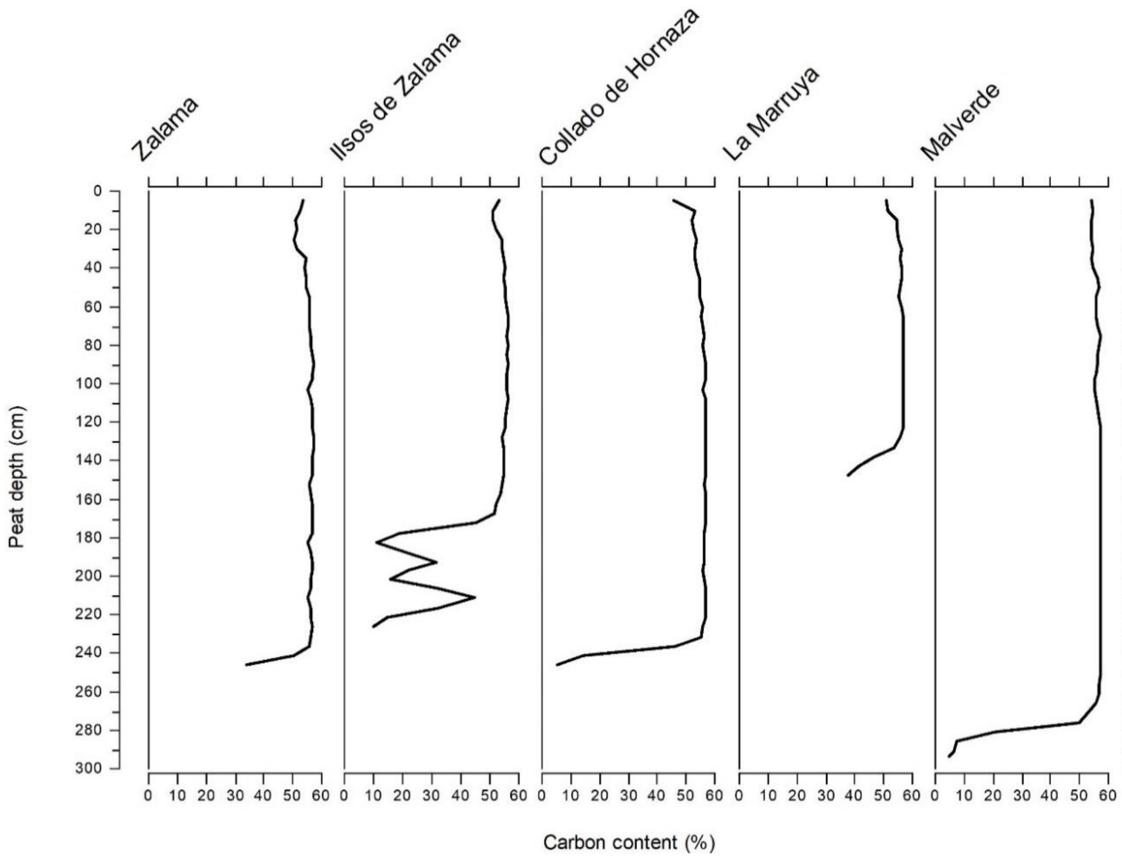


Figure 4.2. Carbon content (%) for each peat core at Zalama, Ilsos de Zalama, Collado de Hornaza, La Marruya and Malverde, northern Spain.

When examining just the top 1 m of the cores, a Kruskal-Wallis rank sum test identified a significant difference in the carbon content of the peat between the cores (chi-squared = 9.6833, p = 0.04611), but a Pairwise Wilcox test did not identify that any one core was significantly different from the others (Table 4.4).

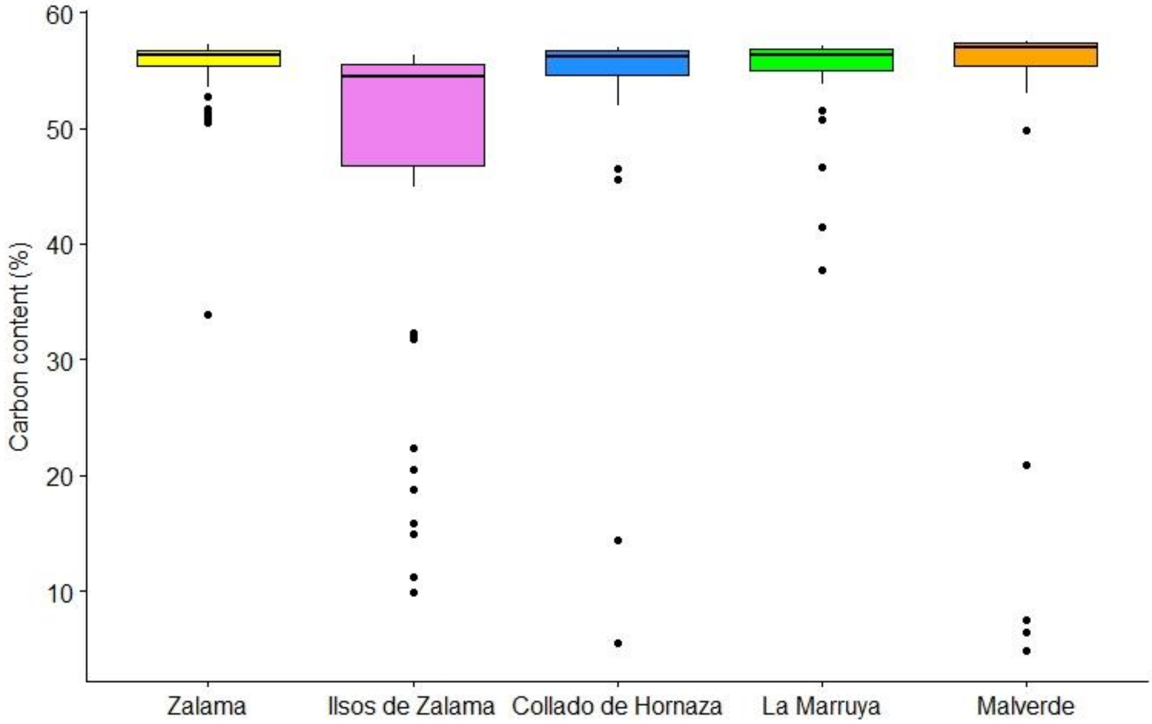


Figure 4.3. Boxplot of % of carbon content at Zalama, Ilsos de Zalama, Collado de Hornaza, La Marruya and Malverde, northern Spain.

Table 4.3. p values from statistical test Tukey HSD for the % of carbon where * indicates significant difference.

	Ilsos de Zalama	*Collado de Hornaza*	*La Marruya*	*Malverde*
Zalama	< 0.001*	0.932	0.998	0.851
Ilsos de Zalama		0.012*	0.014*	0.135
Collado de Hornaza			0.995	0.999
La Marruya				0.982

Table 4.4. p values from statistical test Pairwise Wilcox for the % of carbon of the first top 1 m of the core.

	Ilsos de Zalama	*Collado de Hornaza*	*La Marruya*	*Malverde*
Zalama	0.947	0.947	0.241	0.527
Ilsos de Zalama		0.947	0.056	0.368
Collado de Hornaza			0.056	0.255
La Marruya				0.241

4.3.1.2. Bulk Density

The dry bulk density of peat in all cores showed largely similar patterns as was found with the carbon content, although the bulk density shows a gradual and consistent decrease with depth from the top of the core (Figure 4.4). Interestingly only the peat from Ilsos de Zalama, Collado de Hornaza and Malverde showed an increase in bulk density at the base of the core where the peat is in contact with the bedrock (Figure 4.4), and is evident as outliers in the distribution of the data (Figure 4.5). The mean bulk density of the peat in four of the cores was similar ranging from 0.14 to 0.18 g/cm^3 (Table 4.5), but a higher mean bulk density of 0.22 g/cm^3 was determined for the peat from Malverde (Figure 4.5; Table 4.5). The mean bulk density of peat in the top 1 m of the core from Zalama (0.19 g/cm^3) and Malverde (0.19 g/cm^3), are notably higher than in the peat from the other sites (Figure 4.5; Table 4.5).

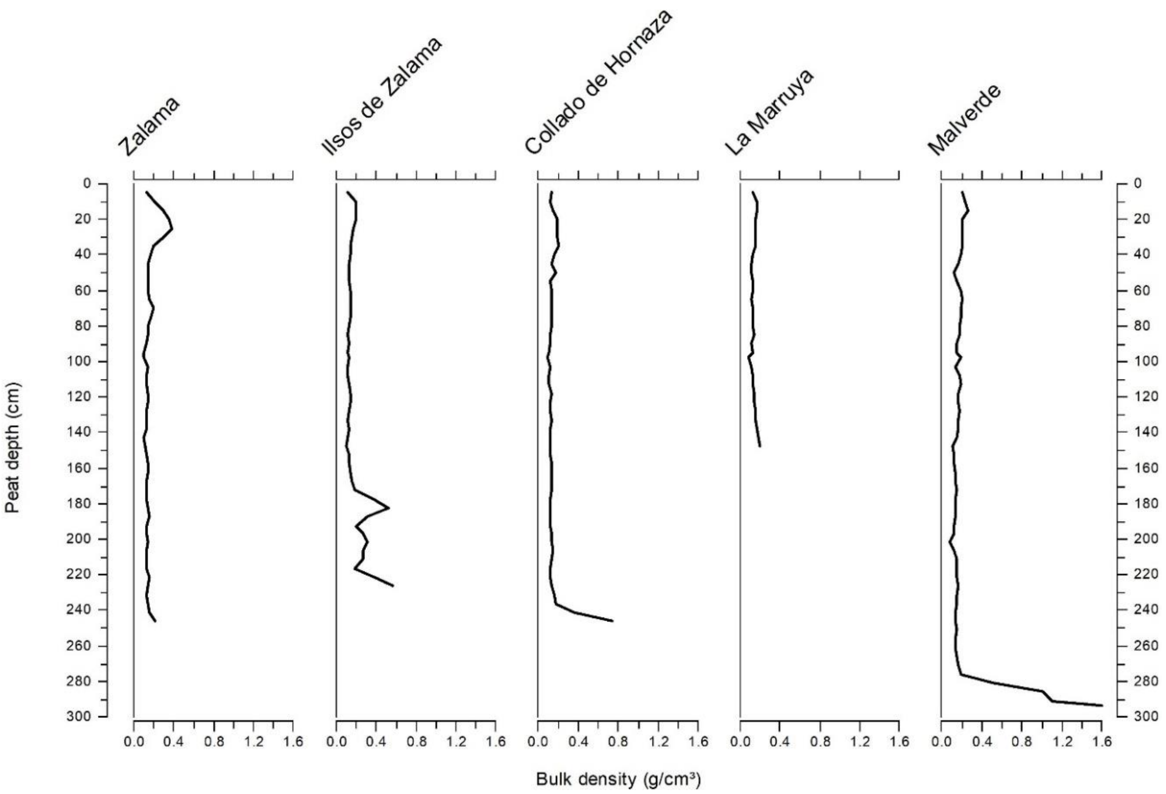


Figure 4.4. Bulk Density (g/cm^3) every 5 cm for the peat cores collected at Zalama, Ilsos de Zalama, Collado de Hornaza, La Marruya and Malverde, northern Spain.

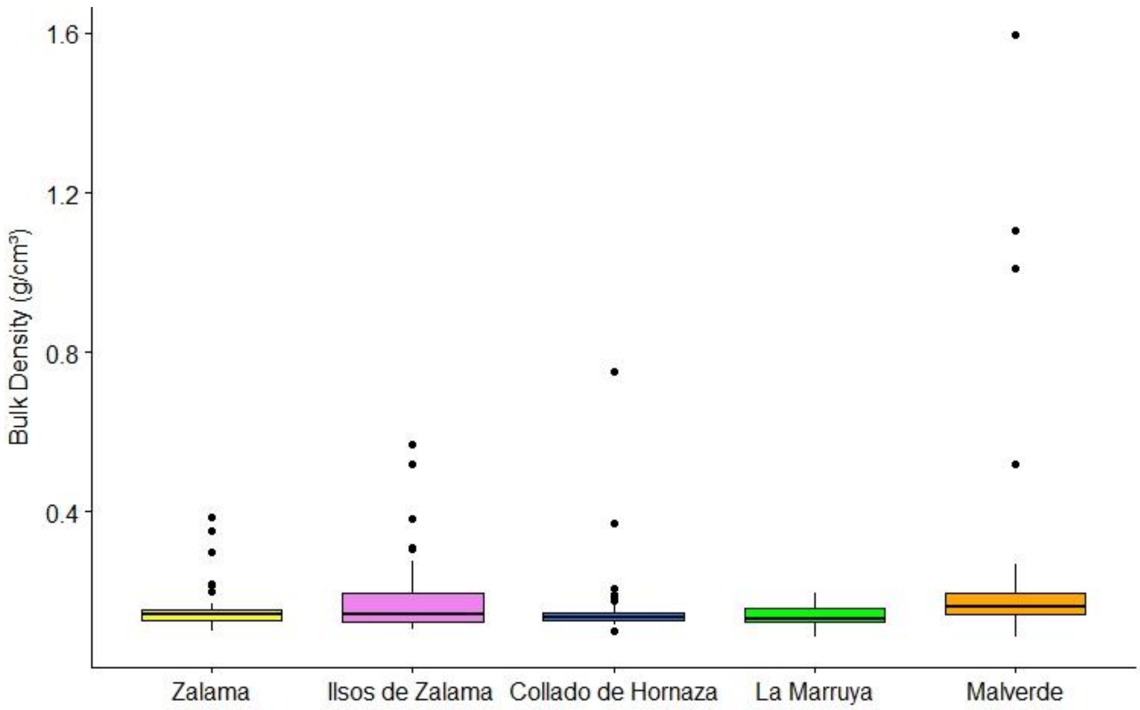


Figure 4.5. Boxplot of the bulk density (g/cm^3) for each peat core at Zalama, Ilsos de Zalama, Collado de Hornaza, La Marruya and Malverde, northern Spain.

A Kruskal-Wallis rank sum test identified a significant difference between the bulk density of peat in the cores (chi-squared = 24.108, $p < 0.001$) and a Pairwise Wilcox test identified that the bulk density of peat from Malverde was significantly different to the bulk density of peat from all other peatlands except Ilsos de Zalama (Table 4.6).

Table 4.5. Mean bulk density for each peat core and the top 1 m for each site.

	Mean BD whole core (g/cm^3)	*Mean BD top 1 m (g/cm^3)*
Zalama	0.15	0.19
Ilsos de Zalama	0.18	0.14
Collado de Hornaza	0.15	0.15
La Marruya	0.14	0.13
Malverde	0.22	0.19

Table 4.6. p values from statistical test Pairwise Wilcox where * indicates significant difference for the whole core.

	Ilsos de Zalama	*Collado de Hornaza*	*La Marruya*	*Malverde*
Zalama	0.612	0.612	0.475	0.001*
Ilsos de Zalama		0.496	0.353	0.098
Collado de Hornaza			0.475	< 0.001*
La Marruya				< 0.001*

Table 4.7. p values from statistical test Pairwise Wilcox where * indicates significant difference for the first top 1 m of peat.

	Ilsos de Zalama	*Collado de Hornaza*	*La Marruya*	*Malverde*
Zalama	0.163	0.317	0.033*	0.152
Ilsos de Zalama		0.414	0.338	< 0.001*
Collado de Hornaza			0.045*	< 0.001*
La Marruya				< 0.001*

When examining the top 1 m of the cores, a Kruskal-Wallis rank sum test also identified a significant difference between the bulk density of the peat (chi-squared = 29.897, $p < 0.001$) and a Pairwise Wilcox test identified a significant difference between the bulk density of peat from Malverde and the bulk density of peat from all other sites except Zalama. In addition, the bulk density of peat from La Marruya was significantly different to the bulk density of peat from Zalama and Collado de Hornaza blanket bogs (Table 4.7).

4.3.1.3. Total carbon stored

The carbon content of the peat in the cores ranged from 73.21 kg/m^3 at Ilsos de Zalama to 92.08 kg/m^3 at Malverde (Table 4.8) and the combined carbon stored in all 15 blanket bogs mapped in this research was estimated to be 44.88 kt C (Table 4.8). As the amount of carbon stored in each peatland was determined using the volume of peat calculated in chapter 3, it is not surprising to note that, more than the 50% of the total carbon stored across all peatlands included in this research is contained in the three largest blanket bogs mapped (Motas del Pardo, Malverde and Zalama; Table 4.9).

Table 4.8. Total carbon for each peat core.

Site	*Total carbon ± SD (kg/m^3)*
Zalama	85.76 ± 27.87
Ilsos de Zalama	73.21 ± 13.76
Collado de Hornaza	76.70 ± 12.04
La Marruya	75.13 ± 9.09
Malverde	92.08 ± 16.89

Table 4.9. Carbon stored in each peatland based on the volume of peat for each peatland and the closest peat core carbon content.

Site	*Carbon stored ± SD (kt)*
Zalama	6.38 ± 2.07
Ilsos de Zalama	2.94 ± 0.55
Motas del Pardo	11.75 ± 1.84
Collado de Hornaza	2.67 ± 0.42
La Marruya	2.69 ± 0.33
Sel de la Peña	0.89 ± 0.11
Cercio	1.55 ± 0.19
Cotero Senantes	2.59 ± 0.31
El Cotero	1.03 ± 0.12
El Cotero Sur	0.32 ± 0.04
Cotero de la Osera	0.80 ± 0.15
Peña Ojastra	0.51 ± 0.09
Cantos Calientes	0.91 ± 0.17
Malverde	8.41 ± 1.54
El Cuito	1.44 ± 0.26
Total	44.88 ± 3.31

4.3.2. Extent of degradation

4.3.2.1. Current extent of exposed peat

All blanket bogs assessed in this research had areas of exposed peat visible in aerial photography from 2017, but the standardised area of exposed peat at each site varied considerably from 44.6 m^2/ha at El Cotero Sur to 505.9 m^2/ha at Cotero Senantes (Table 4.10). A total area of 13.7 ha of exposed peat was mapped (Table 4.10), which equates to 30.8% of total surface area (Table 4.10) of the new mapped blanket bogs in chapter 3 when the peatland extent is defined using peat depth >40 cm, and 21.2% of the total surface area of the blanket bogs if the peatland extent includes peat depth >30 cm.

Table 4.10. Extent of exposed peat area across all study areas.

Site	*Total area of exposed peat (m^2)*	*Standardised area of exposed peat (m^2/ha)*	*% of exposed peat in relation with total peatland area (>40 cm peat depth)*
Zalama	1,632.9	165.4	25.2
Ilsos de Zalama	744.1	175.1	23.4
Motas del Pardo	2,593.2	129.7	23.9
Collado de Hornaza	709.2	170.9	22.5
La Marruya	1020.6	220.9	47.9
Sel de la Peña	193.2	112.3	15.6
Cercio	529.7	195.5	26.5
Cotero Senantes	1,249.5	505.9	36.1
El Cotero	669.6	315.8	47.5
El Cotero Sur	83.4	44.6	19.4
Cotero de la Osera	301.3	146.3	33.1
Peña Ojastra	383.2	197.5	68.4
Cantos Calientes	883.2	496.2	87.4
Malverde	1,782.9	471.7	30.1
El Cuito	939.3	502.3	56.2
Total	13,715.3		30.8

A highly significant and very strong positive correlation was found between the total area of exposed peat and both, the peatland extent ($r = 0.88$, $p < 0.001$) and volume of peat ($r = 0.84$, $p < 0.001$; Table 4.11). This correlation did not hold with the standardised area of exposed peat. It is interesting to note that, although not significant, there was a weak correlation between the standardised area of exposed peat and both latitude ($r = 0.45$, $p = 0.09$) and longitude ($r = 0.44$, $p = 0.10$; Table 4.11).

Table 4.11. Correlation statistical results between the exposed peat area and the variables defined in Chapter 3. [A] indicates Pearson test. For the rest, Spearman test has been undertaken. * indicates significant relationship between the variables.

		Total area of exposed peat	*Standardised area of exposed peat*
Altitude	*r*	-0.22	-0.07[A]
	p	0.43	0.79[A]
Slope	*r*	-0.12	-0.06[A]
	p	0.67	0.83[A]
Maximum peat depth	*r*	0.43	0.05
	p	0.11	0.85
Latitude	*r*	0.20	-0.45
	p	0.47	0.09
Longitude	*r*	0.18	-0.44
	p	0.52	0.10
Peatland extent (> 30 cm)	*r*	0.88	0.15
	p	< 0.001*	0.60
Volume	*r*	0.84	0.10
	p	< 0.001*	0.72

4.3.2.2. Historical evolution of exposed peat areas in restored and unrestored blanket bogs

Since 2009, Zalama blanket bog has been under restoration actions (Aguirre, Benito and Galera, 2017) and there is a marked visible change in the area of exposed peat as a result of restoration activities (Figure 4.6). Between 1977 and 2002 there was a 25% increase in the area of exposed peat at both Zalama and Ilsos de Zalama (Table 4.12), but between 2002 and 2009 the area of exposed had changed little at Zalama (-0.1%) while an 11% increase was observed at Ilsos de Zalama (Table 4.12). The most notable change is between 2009 and 2017 where an 80% reduction in exposed peat was observed at Zalama, although interestingly a decrease in exposed peat was also noted at Ilsos de Zalama, albeit far smaller (12%; Table 4.12). It is also worth noting that in 2017 new areas of erosion have appeared at Zalama around the edge of new fencing installed under the LIFE+ Ordunte Sostenible project (Figure 4.6), presumably as the fence has instigated a new route for livestock and vehicles. Exposed peat in an old vehicle track (running in a SE direction from the NW corner of Zalama across the peatland) reduced notably once the fence was installed in 2008.

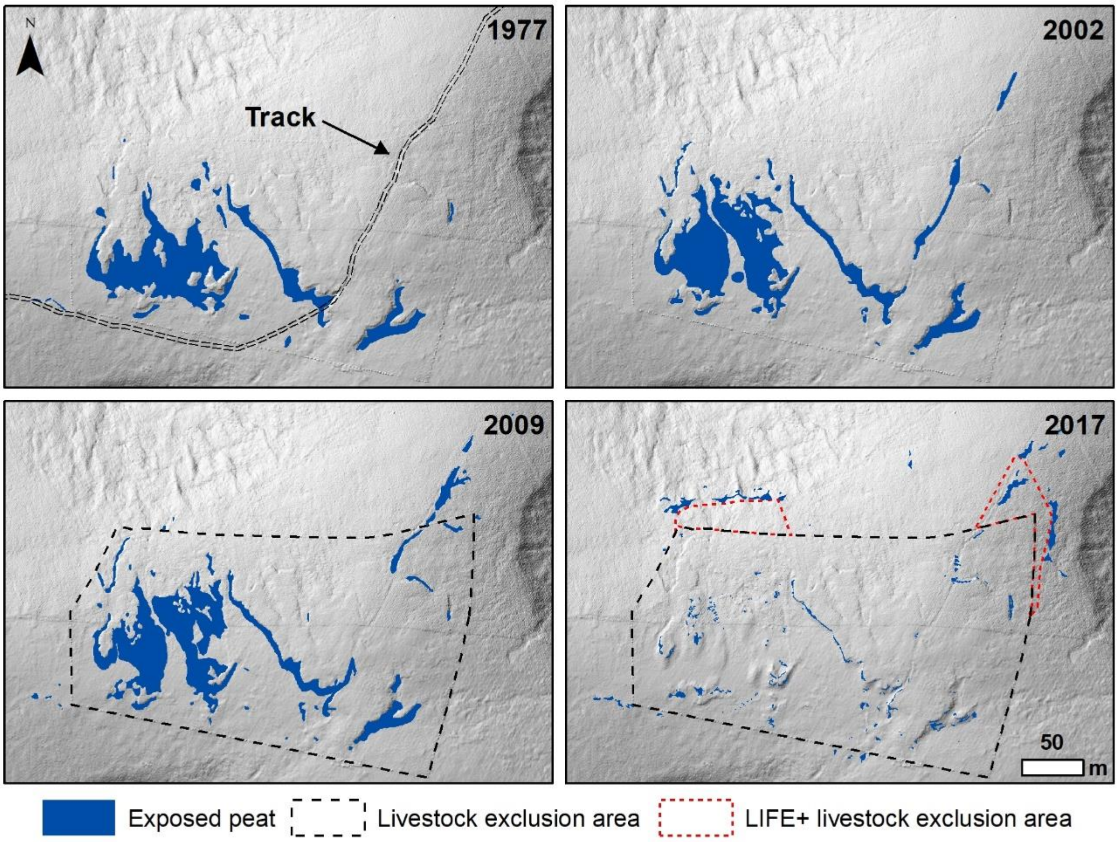


Figure 4.6. Historical evolution of exposed peat areas at Zalama blanket bog before and after restoration.

Table 4.12. Comparison of the historical evolution of exposed peat areas at Zalama blanket bog (protected and restored) and Ilsos de Zalama (unrestored).

Site	*Exposed peat area (m^2)*			
	1977	*2002*	*2009*	*2017*
Zalama	6,632.6	8,295.2 ↑ +25.1%	8,267.2 ↓ -0.1%	1,632.9 ↓ -80.2%
Ilsos de Zalama	608.0	759.9 ↑ +25.0%	845.5 ↑ +11.3%	744.1 ↓ -12.0%

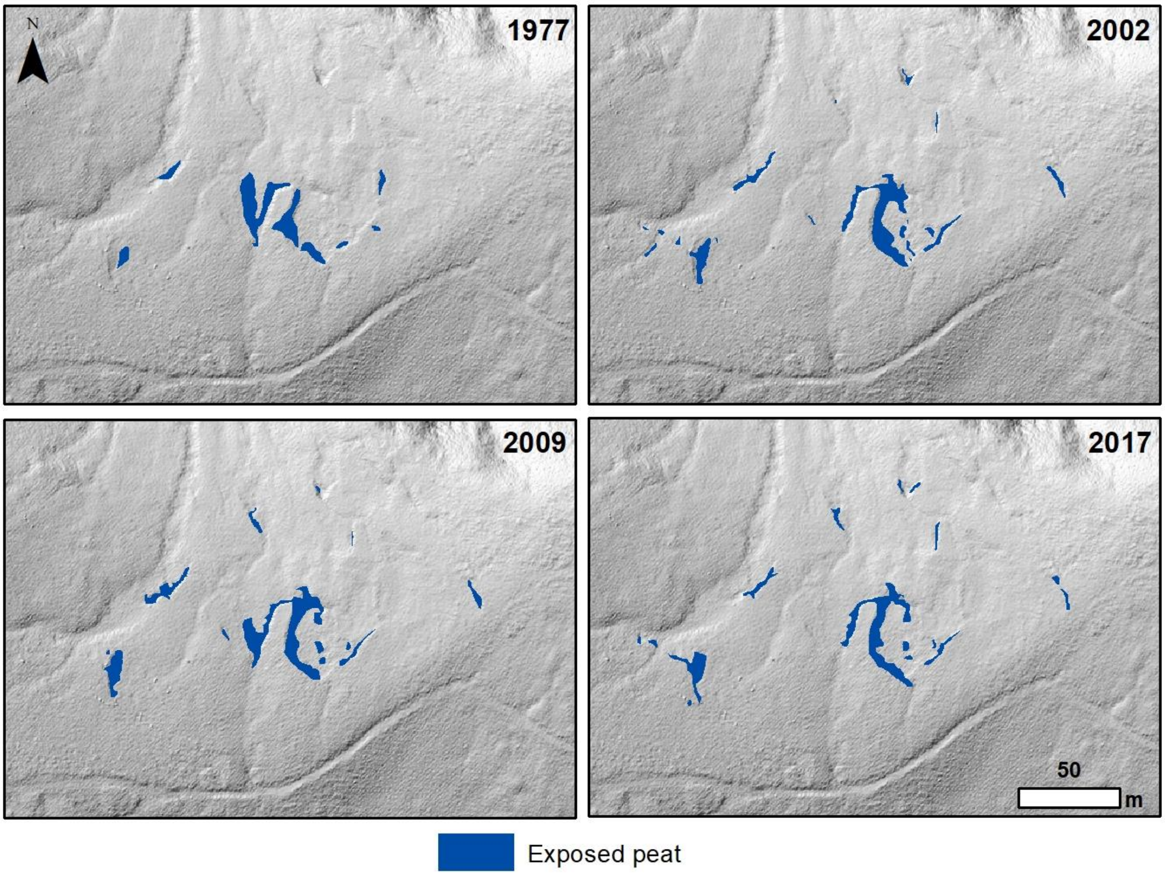


Figure 4.7. Historical evolution of exposed peat areas at Ilsos de Zalama, unrestored and unprotected blanket bog.

4.4. DISCUSSION

4.4.1. Carbon stored

The first aim of this chapter was to estimate the amount of carbon stored in each of the blanket bogs mapped in this research. Adopting the LOI method to analyse the peat cores enabled determination of the dry bulk density (BD) of the peat, the organic carbon content of the organic matter (%) and the weight of organic carbon per unit volume of peat (carbon stored). The BD of peat can directly influence the carbon stored, because BD typically increases as peat is compressed (Lindsay, 2010). Analysing the peat cores every 5 cm, rather than deriving a mean value for a whole core, should, therefore, give an indication of how these two characteristics vary, and how BD affects carbon content in Spanish blanket bog. Models of carbon storage have indicated that acrotelm peat with BD of 0.03 – 0.09 g/cm^3 contains between 14 to 54 kg of C per m^3, and that catotelm peat with BD of 0.10 – 0.20 g/cm^3 contains between 49 to 97 kg of C per m^3 of peat (Clymo, 1992). The values of bulk density and carbon content from the peat cores analysed in this research are comparable to the figures provided by Clymo (1992) for the catotelm, and it is likely that there is no true acrotelm in the blanket bogs assessed in Spain. The model for catotelm peat suggests that for every 0.01 g/cm^3 increased in bulk density, a 1 m^3 block of peat adds 4.85 kg C more to the carbon store (Lindsay, 2010). Linear regression of BD and weight of organic carbon per unit volume of peat showed that these characteristics of the peat in the blanket bogs presented in this research are comparable to those in bog peat in the long-standing model (Table 4.13).

Table 4.13. Relationship between BD and weight of organic carbon per unit volume of peat for the top 1 m of each peat core.

Blanket bog	R^2	*Equation*	*Increase in carbon (kg) stored per m^3 for each 0.01g/cm^3 increase in BD*
Zalama	0.9978	y = 480.74x + 10.739	4.81
Ilsos de Zalama	0.9799	y = 458.79x + 12.44	4.59
Collado de Hornaza	0.9448	y = 491.5x + 7.4699	4.92
La Marruya	0.9571	y = 505.18x + 6.7769	5.05
Malverde	0.9889	y = 510.45x + 8.3073	5.10

The BD of peat in bogs in the United Kingdom typically ranges from 0.07 to 0.15 g/cm^3, although higher values of BD (up to 0.20 g/cm^3) have been recorded near the peatland surface, particularly in the top 45 cm (Cannell, Dewar and Pyatt, 1993). Similar results were found for the peat from at least two of the blanket bogs in this research (Zalama and Malverde), where the mean BD of the peat in the top 1 m of the core (0.19 g/cm^3 for both sites) was notably higher than the BD of peat in the top 1 m for the other three peatlands (ranging from 0.13 to 0.15 g/cm^3). It was also notable at Zalama that the BD of the peat in the top 1 m of the core was higher than the BD of the peat in the remainder of the core, but all cores show a general increase in the BD of the peat with depth up to the basal layers of the core. Interestingly, similar observations of increased BD in surface layers of a peat core had also been recorded in other blanket bogs in Spain (Galicia), where the peat surface had been significantly impacted by livestock (Lindsay, 2010). With the exception of two values of BD for peat in the top layers of the core from Zalama and the BD of the peat/material in the bottom layers of cores (where the peat merges with the bedrock), the values of BD determined in all peat cores in this research were under 0.30 g/cm^3. This value of BD is a limit below which blanket bogs in Spain are reported to be optimal (Pontevedra-Pombal *et al.*, 2009).

The significant difference between the BD of peat determined for Malverde and the BD for Ilsos de Zalama, Collado de Hornaza and La Marruya for the top 1 m of the cores (Table 4.7) may simply result from different environmental conditions (Heinemeyer, Berry and Sloan, 2019). The peat core for Malverde was collected one year after the other cores and expansion and shrinkage of the peatland (mire breathing; Heinemeyer, Berry and Sloan, 2019) would impact on the BD. However, it does not explain why no significant difference was identified between the BD of peat from Malverde and the BD of peat from Zalama in the top 1 m of the cores. Another possible explanation for the difference in the BD of peat from Malverde compared to the BD of peat in the three other cores cited previously (Table 4.7) could be the presence of a windfarm. Malverde is the only peatland with this pressure, and the peat core was collected only 30 m away from the main vehicle track affecting the hydrological units as was seen in chapter 3 (Figure 3.8). It is highly likely that machinery to create the track and excavate peat to create the turbine foundations travelled over large areas of the peatland and there is also a drainage system associated with the track (Figure 2.17). The observation of the higher BD of peat in the top of the core from Zalama may also support this suggestion as Zalama is protected and has undergone restoration, which involved the use of machinery to move equipment across the peatland. In addition, a now revegetated vehicle track was running across the peatland surface indicating previous vehicular activity at Zalama that could explain the higher BD (up to 0.39 g/cm^3) on the top layers in this site (Figure 4.6). The use of machinery would result in compression of peat near the surface, and a resultant increase in the BD of the peat (Lindsay, 2010). However, the impact of machinery on the BD of peat in the surface of peat has been reported to be less significant than the impact of mire breathing (Heinemeyer, Berry and Sloan, 2019), although it is not clear if the study only assessed one single travel of machinery rather repeated travel, and the study did not assess the impact of vehicles turning. If mire breathing is the main cause for the significant differences found in BD of peat in the top layers of the peat cores from Malverde and the other peatlands (Table 4.7), it would not explain why the BD of peat from the top of the core from Zalama was not significantly different. Zalama is located only 500 m from Ilsos de Zalama and the cores were collected on the same day and there is no evidence of machinery use at Ilsos de Zalama.

Prescribed burning and wildfire can also modify the BD of peat, as the BD of surface layers of burned peat is higher than the BD of unburned areas (Thompson and Waddington, 2013; Holden *et al.*, 2014). Vegetation burning was visible in the first field survey campaign in 2017 at Motas del Pardo, Collado de Hornaza and La Marruya blanket bogs, but black layers indicating burning activities in the past were visible in all peat cores. The impact of burning may therefore be widespread across the study areas. The practice of burning is commonly used to improve grazing for livestock, an activity that can have further impact on the BD of peat from trampling and compaction (Worrall and Clay, 2012). Livestock including cattle, horses and goats were observed in all study areas assessed in this research with exception of Zalama (livestock exclusion), especially between May and October. All livestock have an impact on the BD of soils (Greenwood *et al.*, 1998), but in peatlands only the impacts of sheep on BD have been reported (Worrall and Clay, 2012). As the mean BD for the top 1 m of all the peat cores determined here (0.13 to 0.19 g/cm^3) are all at the higher end of typical values in peat 0.07 to 0.15 g/cm^3 (Cannell, Dewar and Pyatt, 1993), this could suggest that the presence of livestock on Spanish blanket bogs has a measurable impact on the BD of the peat. Since La Marruya is located within a farm that may have seen continued livestock for centuries, and this might explain why the BD of peat in the top 1 m of the core was significantly different to the BD of peat at Zalama, Malverde and Collado de Hornaza. However, the mean BD of peat at La Marruya was actually the lowest out of all five cores, and while livestock will affect the BD, it appears that the use of vehicles may have a greater impact.

The organic carbon content of the organic matter (%) is also often used to provide an indication of the status of peatlands. For blanket bog in Spain the optimal carbon content is reported as 45% (Pontevedra-Pombal *et al.*, 2009), and it is promising that the mean carbon content in all the blanket bogs in this research is greater than this figure (46.9% to 55.2%) and comparable to the carbon content reported for Galician blanket bogs between 46% (Ramil-Rego and Aira Rodríguez, 1994) and 51% (Gómez-Orellana *et al.*, 2014). The lowest carbon content in this study (46.9%) was measured at Ilsos de Zalama, and this was the only blanket bog that had a saddle mire mesotope. It is possible that mesotope type has an impact on carbon content, but further examples of saddle mire are required to assess this. In comparison with other blanket bogs, the % of carbon content of the other four blanket bogs (Zalama, Collado de Hornaza, La Marruya and Malverde) are comparable to those found in Scotland (53.5%, Chapman *et al.*, 2009), Eastern Canada and Western European Islands with Atlantic climate conditions (Loisel *et al.*, 2014).

Based on the volume of peat determined in chapter 3 and the carbon content determined for the peat cores, it was possible to estimate that the blanket bogs in this research contain 44.88 kt of C. All types of peatland combined in Spain were reported to contain 5,398 Mt of carbon (Joosten, 2009), and the blanket bogs in Galicia contain 4.47 Mt of carbon (Gómez-Orellana *et al.*, 2014). The figure of 0.04 Mt of carbon contained within the blanket bogs in this research may therefore appear less significant. However, it is important to note that the blanket bogs assessed here represent the southernmost edge-of-range of this habitat in Europe, and therefore any carbon stored has a disproportionately high value for preservation. The threat to such small areas of blanket bog is very apparent as this research found that three potential areas of blanket bog in the study area were so severely degraded that there was almost no peat left (see section 3.3.1.1). It is also worth considering the contribution of carbon storage in peatlands with that stored in rainforests. The amount of carbon contained in 1 ha of peatland with 30 cm of peat is equal to the carbon stored in 1 ha of rainforest (Lindsay *et al.*, 2019). Based on the mean peat depth mapped across the study sites, the 64.65 ha of blanket bog assessed in this research contain the same amount of carbon as an area of rainforest of 120.34 ha. This further highlights the importance of this habitat to store carbon, and when restored, blanket bogs also have the capability of functioning as carbon sinks (Nugent *et al.*, 2018).

4.4.2. Extent of Degradation

Having determined the amount of peat stored in each blanket bog, this chapter was also set out to quantify the extent of exposed peat to provide a proxy for the level of degradation and to subsequently determine the impact of restoration on the area of exposed peat. The physical characteristics of the peatlands were used to assess the drivers of degradation.

The strong positive correlation between the area of exposed peat and both the extent of the peatland and volume of peat suggest that degradation is consistent across all sites and not obviously influenced by location or mesotope type. This is particularly evident as the standardised area of exposed peat did not correlate with any physical peatland characteristic. In countries with the largest extent of blanket bogs (United Kingdom and Ireland; Moen, Joosten and Tanneberger, 2017), the proportion of peatland with exposed peat area in relation with the peatland extent (19% to 33%; Table 4.1) is comparable to the proportion of peatland exposed in this study (31% when the peatland extent is defined using peat depth > 40 cm). This is particularly interesting considering the blanket bogs in Spain are more than 1,000 km farther south and located at a higher altitude (particularly compared to those in England and Ireland), where natural pressures such as aeolian, fluvial or ice erosion might be expected to be greater (Phillips, Tallis and Yalden, 1981).

Protection and restoration measures undertaken at Zalama indicate that this is extremely effective at reducing the area of exposed peat (Figure 4.6; Figure 4.7) and likely reducing carbon loss. Historical aerial photography enabled baseline data to be collected and demonstrated that between 1977 and 2002, the increase in exposed peat at Zalama was equal to the increase in exposed peat at Ilsos de Zalama (25%), but since intervention, specifically from 2009 to 2017, there has been an 80% reduction in exposed peat at Zalama (Table 4.12); however, and perhaps counter-intuitively, a reduction in the area of exposed peat was also observed at Ilsos de Zalama (12%) over this same time period without protection or restoration. The reduction of exposed peat mapped at Ilsos de Zalama could then be a natural re-vegetating response of the peatland resulting from reduced anthropogenic pressures although alternatively, and of greater concern, the reduction of exposed peat mapped at Ilsos de Zalama may reflect complete loss of the peat deposit leaving the mineral substrate as the new surface and habitat. There was no evidence of an increase in vegetation cover in the aerial photography at Ilsos de Zalama, but there was a clear increase in the total area of mineral substrate visible in the main area of exposed peat at Ilsos de Zalama. An increase in the area of mineral substrate was also observed at Zalama blanket bog prior to restoration where up to 50% of the original peat deposit was removed by erosion processes leaving mineral substrate exposed (Heras, 2002). This could indicate that anthropogenic pressures such as burning or livestock are promoting erosion up to a point where the peat deposit disappears completely, as was described at Zalama blanket bog (Heras and Infante, 2003). In fact, the government livestock inventory for the last century in Cantabria shows a clear increase in the number of cattle since 1900, although a slight decrease has been observed in the last 20 years (Figure 4.8). The number of sheep have decreased since the early 1900s, and goat numbers have remained relatively stable. In the municipality of Soba, the nearest town to Ilsos de Zalama, the number of cattle, sheep and goats have all remained relatively stable since 2001 (Figure 4.9), but there are over five times more cattle than either sheep or goats.

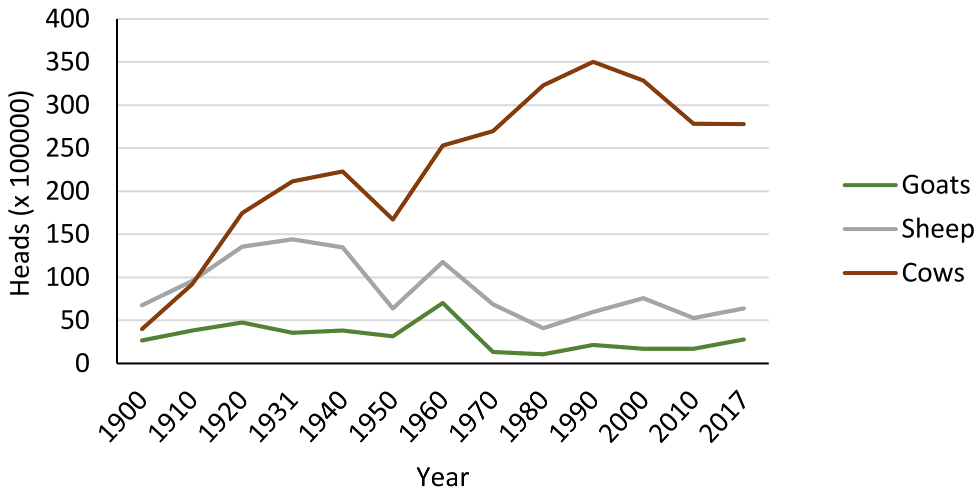


Figure 4.8. Livestock numbers from 1900 to 2017 in Cantabria administrative region (Spain) (Instituto Cántabro de Estadística, 2020).

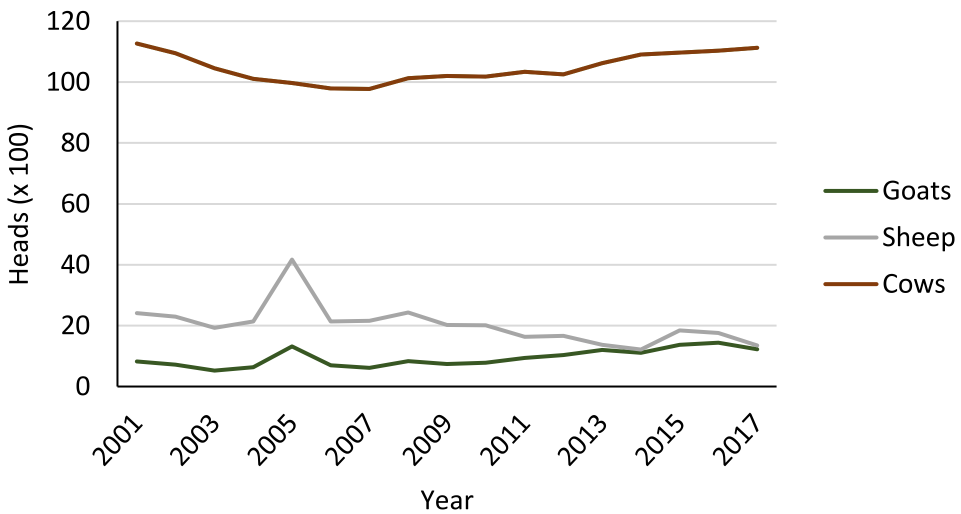


Figure 4.9. Livestock numbers from 2001 to 2017 in Soba municipality, Cantabria (Spain) (Instituto Cántabro de Estadística, 2020).

Given that there are more than 10,000 cattle in the municipality of Soba (Figure 4.9), there is a clear need to quantify the impact that this has on blanket bogs, particularly the southernmost edge-of-range examples in the Cantabrian Mountains.

4.5. CONCLUSION

The carbon content of peat in the Cantabrian Mountains has been determined for four of the newly identified blanket bogs and also for the restored Zalama blanket bog. The organic carbon content of the organic matter (%) in all cores is above the reported optimal value for blanket bog in Spain, and the values are comparable with other blanket bogs in Galicia, Scotland and Eastern Canada. The dry bulk density appears to show a gradual but consistent decrease with depth and trampling from livestock may be increasing BD in the top layers. At Zalama and Malverde notably higher BD of the peat was recorded near the surface and may be a result of machinery used for restoration and windfarm infrastructure. The 15 blanket bogs in this research are estimated to contain 44.88 kt C, and while this may appear low in relation to the carbon stored in other Spanish peatlands, as the blanket bogs assessed here represent the southernmost edge-of-range of this habitat in Europe, any carbon stored here has a disproportionately high value for preservation.

The area of exposed peat at each site varied considerably from 44.6 m^2/ha to 505.9 m^2/ha, and does not appear to relate to any physical characteristic of the peatlands. The total area of exposed peat mapped (13.7 ha) equates to 30.8% of total surface area of the blanket bogs. Protection and restoration activities have had a marked positive impact on the area of exposed peat at Zalama, while only 500 m away at Ilsos de Zalama, an apparent reduction in exposed peat was found to arise from total loss of the peat deposit. The presence of livestock is implicated in this loss, and given that there are more than 10,000 cattle that could potentially graze Ilsos de Zalama, there is a clear need to quantify the impact of livestock on peat loss. As there are currently no estimates of the rate of erosion of peat in Spain, a method of assessment that enables ultra-high resolution of change is required. This is presented in Chapter 5, and expanded in Chapter 6 to determine the impact of livestock on erosion and peat loss.

Chapter 5

The current degradation status of blanket bogs in northern Spain – Application of terrestrial laser scanning to quantify surface changes in restored and degraded blanket bogs

The content of this chapter has been published

5.1. INTRODUCTION

Erosion and peat loss from blanket peat has been studied extensively in the United Kingdom (Table 2.2; Table 5.1), and while it is commonly highlighted as a problem in other countries, it is not always quantified: e.g. in Ireland (McGreal and Larmour, 1979), Canada and Sweden (Foster *et al.*, 1988), and in Spain (Castillo *et al.*, 2001; Heras and Infante, 2003, 2018). The rate of erosion is affected by natural processes, while peat loss can arise from a combination of natural and anthropogenic influences, though aeolian, fluvial and freeze-thaw processes have been identified as the key drivers of surface change (Bower, 1961; Labadz, 1988; Campbell, Lavoie and Rochefort, 2002; Li, Holden and Grayson, 2018). Anthropogenic pressures on peatlands including drainage (Holden *et al.*, 2006; Luscombe *et al.*, 2016), peat extraction (Price, Heathwaite and Baird, 2003; Lindsay, 2010), overgrazing (Ward *et al.*, 2007), prescribed burning (Yallop *et al.*, 2006; Clutterbuck and Yallop, 2009) and wildfires (Yeloff, Labadz and Hunt, 2006; Heras and Infante, 2018) have all been highlighted as influencing peat degradation. Bog-bursts may also be initiated by windfarms and associated infrastructure (Lindsay and Bragg, 2005), and the installation of wind turbines on blanket bog is a contentious issue (Wawrzyczek *et al.*, 2018), particularly in north Spain (Heras and Infante, 2008), where a number of areas of peat, including the new areas of blanket bog identified in chapter 3, are currently not protected and under this pressure (Gobierno de Cantabria, 2017).

Table 5.1. Annual rates of peat erosion for England, Wales and Scotland derived from Evans, Warburton and Yang (2006) and Li *et al.* (2018).

Country	*Number of studies*	*Min (mm yr^{-1})*	*Max (mm yr^{-1})*	*Mean (mm yr^{-1})*
England	18	1.03	73.8	22.4
Wales	3	16	30	23.1
Scotland	2	10	59	36.3

The mean rate of erosion for bare peat surfaces across the United Kingdom is estimated at 23.1 mm yr^{-1} (Table 26; Evans and Warburton, 2007), although in some places rates of change are less than 10 mm yr^{-1} (Table 2.2). Such fine-scale erosion in peatlands has traditionally been determined using erosion pins (e.g. (Labadz, Burt and Potter, 1991; Evans, Warburton and Yang, 2006) and sediment traps (Evans and Warburton, 2007), although the spatial extent of assessment using these approaches is significantly limited (Boardman and Favis-Mortlock, 2016). In addition, erosion pins can be moved and their presence influences the erosion process (Couper, Stott and Maddock, 2002). Assessments of erosion features over larger areas of peatland have employed remote sensing techniques such as conventional aerial photography (Bower, 1961; Tallis, 1973) and airborne LiDAR (Walsh, Butler and Malanson, 1998; Evans and Lindsay, 2010), but the spatial resolution of both these technologies (typically 25 cm at best for commercial off-the-shelf (COTS) products covering areas of peatlands) constrains the scale of erosion detectable (Clutterbuck *et al.*, 2018). For the study areas in this research even the most recent aerial photography available from Instituto Geográfico Nacional (2019) is 25 cm resolution (see section Current extent of exposed peat). While higher resolution data can be obtained from bespoke, commissioned surveys, these technologies are still suited to longer-term assessment of change in peatlands owing to the accuracy (5 – 10 cm horizontal. 5 – 15 cm vertical; e.g. Bluesky International Ltd., 2019).

The development of new techniques such as Structure-from-Motion (SfM) photogrammetry marked a major enhancement in geoscience (Westoby *et al.*, 2012), and SfM approaches using UAVs and ground-based cameras are seeing wide application in peatland environments (Kalacska *et al.*, 2013; Knoth *et al.*, 2013; Lehmann *et al.*, 2016; Glendell *et al.*, 2017; Lovitt, Rahman and McDermid, 2017; Smith and Warburton, 2018). Ultra-high resolution imagery achievable with the techniques (<1 cm) are beginning to see direct application for quantifying rates of peat erosion (Glendell *et al.*, 2017) although this technique is commonly validated with benchmark data derived from Terrestrial Laser Scanning (TLS) techniques (Glendell *et al.*, 2017; Godfrey *et al.*, 2020).

Terrestrial laser scanning has advanced rapidly in the last decade, with TLS units now more portable and capable of recording 1 million points per second (pts s^{-1}) providing ultra-high-resolution 3D data (< 2 mm points spacing) with accuracies of 1 mm at 10 – 15 m from the scanner (Idrees and Pradhan, 2016). The high-resolution area captured using TLS is significantly less than areas covered in airborne surveys, but point cloud data derived from TLS retain the complex morphology of surfaces such as overhanging topography that is very common in peatland environments and allow 3D comparison of change (Ordóñez *et al.*, 2018). TLS technology is seeing wide application for assessing change in a range of environments including alpine (Schürch *et al.*, 2011) and proglacial rivers channels (Milan, George and Hetherington, 2007), meandering gravel beds (O'Neal and Pizzuto, 2011), bedrock rivers (Lague, Brodu and Leroux, 2013), rill (Lu *et al.*, 2017) and bluff erosion (Day *et al.*, 2013), badland landforms (Neugirg *et al.*, 2016), sub-tropical vertosol gullies (Goodwin *et al.*, 2016) and coastal geomorphology (Godfrey *et al.*, 2020). However, to date, TLS has seen limited application for assessing geomorphological changes such as erosion in peatland environments (Grayson *et al.*, 2012; Glendell *et al.*, 2017). Several challenges have been noted in the application of TLS in peatlands (Grayson *et al.*, 2012), as dense vegetation may inhibit assessment of the surface, and morphological change arising from 'mire breathing', where the peat surface can change vertically and horizontally in response to gaseous exchange or water content in the peat body (Schlotzhauer and Price, 1999; Glaser *et al.*, 2004), could be greater than the scale of erosion occurring.

Although erosion has been highlighted as a significant issue for peatlands in northern Spain (Heras and Infante, 2003), there are currently no published data on erosion or peat loss. As the rate of change is unknown, any assessment of erosion must be able to detect as fine-scale change as possible. The aim of this chapter is to develop a method to measure surface change in blanket bogs using TLS, and to compare the rate of change of exposed peat between restored and unrestored areas. The following objectives were set:

a) Explore the operation, application and characteristics of the TLS equipment and data.
b) Develop a method to measure ultra-high-resolution changes in exposed peat surfaces with a high degree of accuracy
c) Assess differences between the rate of surface change between a designated and restored blanket bog and two other comparable blanket bogs in the Cantabrian Mountains.

This chapter has been published in one peer review publication; Chico et al., (2019*) Application of Terrestrial Laser Scanning to quantify surface changes in restored and degraded blanket bogs*. Mires and Peat, (24) 14, 1-24.

5.2. METHODS

5.2.1. Initial method development

5.2.1.1. Terrestrial laser scanner characteristics

A FARO Focus3D X330 was selected for assessment in this research as the unit is suited to remote surveys owing to the relatively small size and low weight (5.2 kg). The X330 scanner has a maximum range of 330 m with a ranging error of ± 2 mm (FARO technologies Inc., 2015). The scanner is phase based measuring phase shift in pulses sent at a wavelength of 1550 nm to determine distance and can send up to 970,000 points per second at maximum scan resolution (FARO technologies Inc., 2015). The step size at this resolution is 0.009°, which is reported to achieve a point spacing of 1.5 mm at a distance of 10 m from the scanner (FARO technologies Inc., 2015). The scanner also allows to the user to select a 'quality', which relates to the confidence in a distance measurement determined by the number of repeat measurements used to derive an average distance for each individual scan 'point'. Higher quality settings should reduce potential noise in the data caused by moving objects such as vegetation, but it will not increase the number of points collected, although both higher quality and higher scan resolutions will increase scan time (Table 5.2). There is inevitably a trade-off between the time available for scanning and the resolution/quality of the scan data required.

In order to explore the limitations of the X330 scanner to measure surface change in peatland environments and to determine the optimal scanner parameters prior to remote surveying, three series of experiments were undertaken between December 2016 and March 2017. The experiments were designed to explore: a) the impact of changes in light conditions on the number of scan points achieved in an area of interest (AOI); b) to quantify the number of scan points achieved in an AOI at different resolutions for a range of distances; and c) to understand how the quality setting impacts on scan accuracy. Scan data were processed using FARO SCENE version 7.1.1.81 and the resulting point cloud data were exported in .pts format and imported into CloudCompare for evaluation.

Table 5.2. TLS scanning times in relation with the best 3 resolution and all quality settings (FARO technologies Inc., 2015).

Resolution	*Quality*	*Scanning time*
1/1	1x	14 min
1/1	2x	29 min
1/1	3x	57 min
1/1	4x	1h 55 min
1/2	1x	4 min
1/2	2x	7 min
1/2	3x	14 min
1/2	4x	29 min
1/4	1x	1 min
1/4	2x	2 min
1/4	3x	4 min
1/4	4x	7 min

5.2.1.2. Determination of optimal scanner parameters

5.2.1.2.1. Light conditions

During surveys in peatland environments it is possible that cloud cover will frequently change the level of illumination. This could occur during a scan or mean that two independent scans will be taken under different light conditions. To understand the impact of changes in illumination on the scan data, repeat scans were undertaken under three conditions: No light, Artificial light and Natural (Sun) light. A room with windows and blackout blinds on two sides of the room was selected to enable manipulation of all conditions. Five sheets of white A3 paper with a red boundary to define the edge of the AOI were attached to a wall, and five targets (spheres) were positioned across the room to align scan data (Figure 5.1). Three scans were undertaken under each light condition to compare the variability for each light condition. The location (coordinates) of the spheres from the first scan were used to position the spheres in all subsequent scans to align the point clouds, since spheres and scanner were not moved between scans. Five rectangular AOI's were created in CloudCompare using the A3 sheets and used to report the number of scans points in each rectangle in each scan (Table 5.3). A Kruskal – Wallis rank test was conducted using R 3.6.2 to identify any potential difference between the total number of scan points in all rectangles for each replication under each light condition.

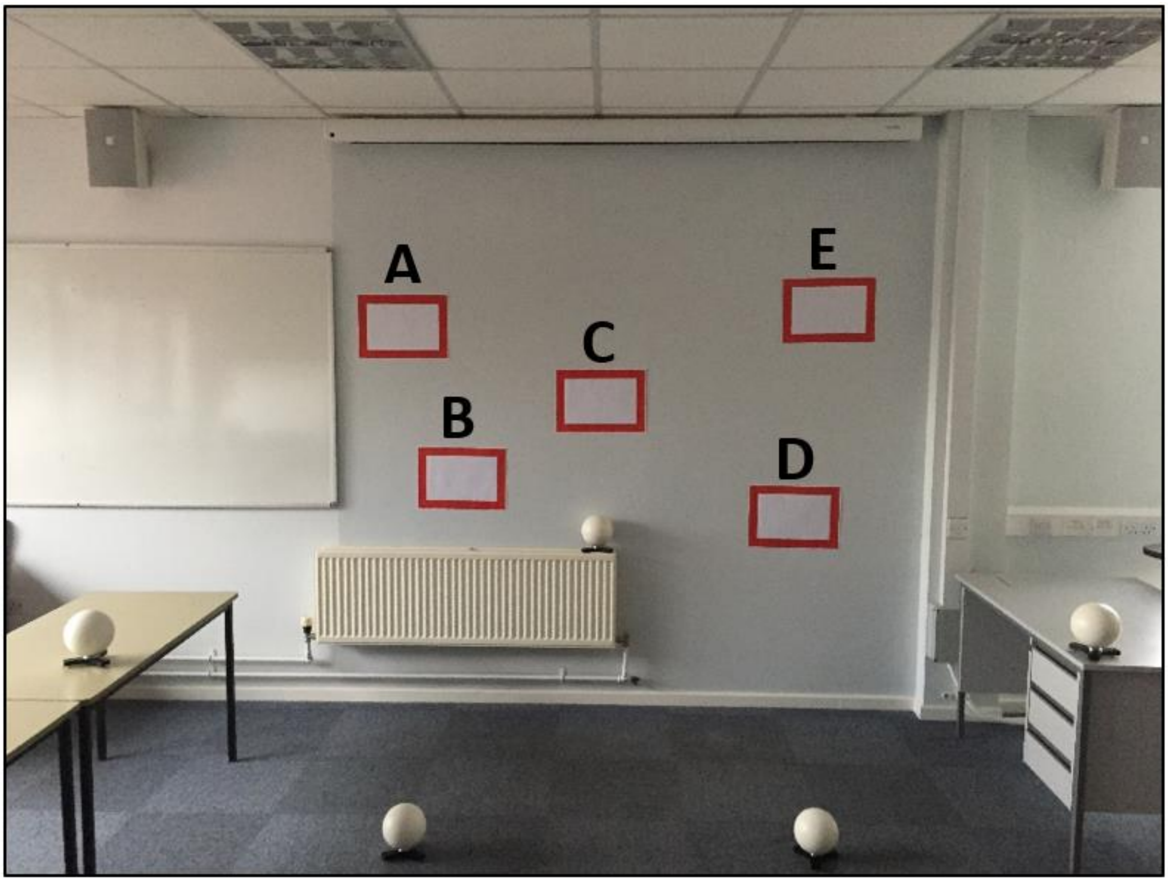


Figure 5.1. Light conditions experiment design.

With the exception of AOI C under natural light, the number of points in replicate scans varied by <0.7% (Table 5.3). The variation in the number of points between scans for AOI C under natural light was slightly higher at 1.05% although, the Kruskal-Wallis rank test did not identify any significant difference between the number of scan points in the replication of each light condition (Artificial light, chi-squared = 0.18, p = 0.91; Dark, chi-squared = 0.26, p = 0.88; Sun light, chi-squared = 0.02, p = 0.99) and also found no significant difference in the total number of points in all AOIs between each light condition (chi-squared = 0.19, p = 0.91). This assessment demonstrates that changes in illumination during field survey will not have a significant impact on the number of scan points collected using the FARO X330 TLS.

Table 5.3. Number of TLS data points under different light conditions.

Artificial light					
	A	*B*	*C*	*D*	*E*
Scan 1	*12424*	*12530*	*13713*	*12623*	*12452*
Scan 2	12382	12593	13547	12709	12373
Scan 3	12383	12604	13568	12800	12371
Mean	12396.33	12575.67	13609.33	12710.67	12398.67
SD	23.97	39.93	90.39	88.51	46.20
Variation (%)	0.19	0.32	0.66	0.70	0.37
Dark					
	A	*B*	*C*	*D*	*E*
Scan 1	12537	12534	13667	12666	12468
Scan 2	12442	12555	13585	12707	12477
Scan 3	12398	12542	13593	12707	12477
Mean	12459	12543.67	13615	12663.33	12446
SD	71.04	10.60	45.21	45.06	46.12
Variation (%)	0.57	0.08	0.33	0.36	0.37
Natural light					
	A	*B*	*C*	*D*	*E*
Scan 1	12516	12537	13474	12727	12359
Scan 2	12408	12493	13742	12702	12402
Scan 3	12408	12611	13695	12686	12391
Mean	12444	12547	13637	12705	12384
SD	62.35	59.63	143.10	20.66	22.34
Variation (%)	0.50	0.48	1.05	0.16	0.18

5.2.1.2.2. Resolution and distance

To assess the impact of resolution and distance on the number of points achieved in an AOI, 12 identical targets (spheres measuring 145 mm diameter) were positioned at a range of distances from the scanner between 3 m (the minimum distance required to obtain data based on the height of the scanner) and 20 m (Table 5.4; Figure 5.2) in a semi-circular distribution to retain a clear line of sight between the scanner and all targets (Figure 5.2). Nine scans were undertaken from the same position using each scan resolution available in the FARO X330 (1, 2, 4, 5, 8, 10, 16, 20 and 32) and the same quality setting. The experiment was repeated on two different days under similar environmental conditions. A clip box measuring 5 x 5 x 5 cm was used in CloudCompare to extract the scan points from the centre of each sphere.

Table 5.4. Distances from scanner to the spheres in experiment 1 and 2.

Experiment 1		*Experiment 2*	
Sphere	*Distance (m)*	*Sphere*	*Distance (m)*
1	3.30	1	3.45
2	4.12	2	4.76
3	5.03	3	5.96
4	6.03	4	7.40
5	7.22	5	8.97
6	8.41	6	9.98
7	10.01	7	11.33
8	12.13	8	12.63
9	14.26	9	14.33
10	16.25	10	15.66
11	18.23	11	17.39
12	19.95	12	19.17

Figure 5.2. Distance and resolution experiment design.

The number of points achieved in the centre of each sphere reduced markedly with scan resolution, as around 75% fewer points were achieved between the highest resolution and the second highest setting (Figure 5.3; Figure 5.4). The number of points in the centre of each sphere also decreased exponentially with distance from the scanner (Figure 5.3; Figure 5.4). As the rate of erosion in Spain was not known, it was envisaged that mm resolution data might be required to detect change accurately. At 15 m from the scanner the highest scan resolution is still able to achieve a point spacing of around 1 mm (100 points per cm^3) and therefore, only the highest resolution was deemed acceptable to measure erosion in this research.

A Mann-Whitney U test was undertaken to explore potential differences between the number of points with distance in both experiments. No significant different was found at resolution 1, 2 or 4 ($p = 0.79$, resolution 1; $p = 0.76$, resolution 2; $p = 0.89$, resolution 4) demonstrating the consistency of the TLS to collect comparable point density at different resolutions and distances on different occasions.

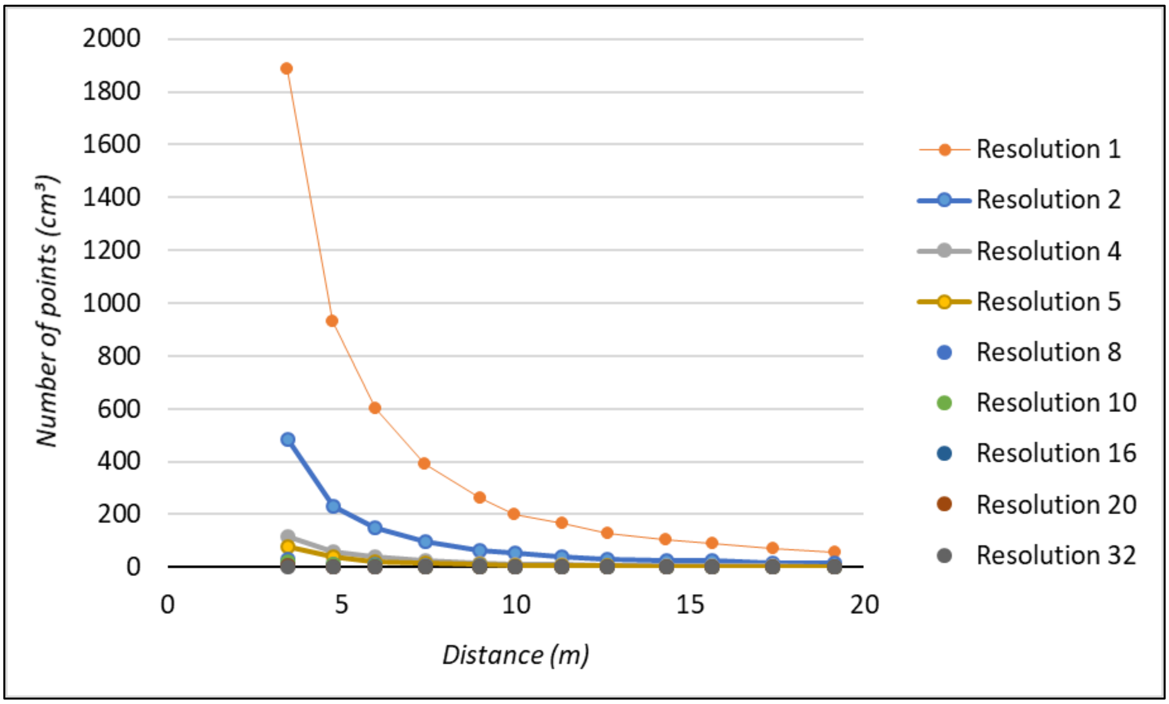


Figure 5.3. Number of points per cm^3 obtained with each resolution setting against distance from scanner location (experiment one).

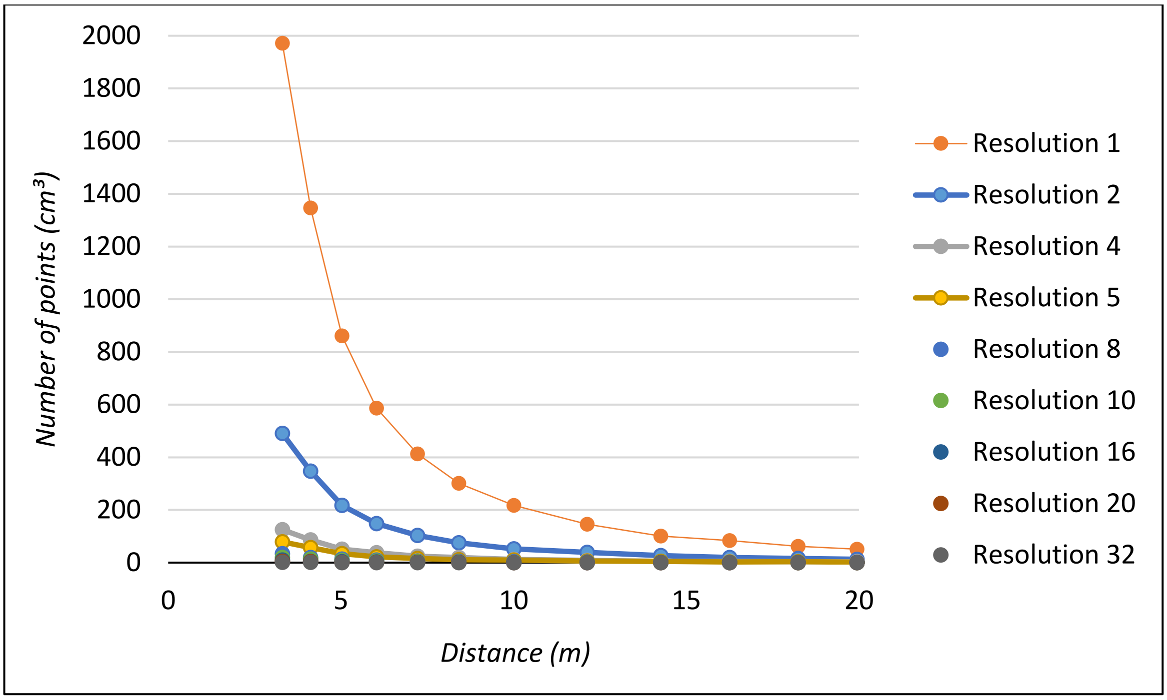


Figure 5.4. Number of points per cm^3 obtained with each resolution setting against distance from scanner location (experiment two).

5.2.1.2.3. Quality

To explore the impact of the quality setting on the accuracy of the distance of a scanned object, three identical targets (same spheres than previous experiments) were positioned at different distances from the scanner (Table 5.5). The X330 has four settings for quality: x1, x2, x3 and x4. Quality x4 undertakes the most repeat measurements and the spheres were scanned first using this setting to provide a reference dataset. Scans were then repeated with the other three settings. All scans were undertaken using resolution 1 for consistency and because it was selected as the target resolution for this research in previous experiment. A clip box of 5 x 5 x 5 cm was created for each sphere and a point cloud extracted for each sphere at each quality setting. Using the C2C (Cloud to Cloud) tool in CloudCompare, the distance between the points in the reference scan (x4) and the points in the scans at x1, x2 and x3 were determined for all spheres.

Table 5.5. Distances of the spheres on the quality experiment

Sphere	*Distance (m)*
1	3.06
2	4.60
3	5.65

Table 5.6. Average and maximum distances between quality x4 (baseline) and the other qualities (x1, x2 and x3). On green the best performances.

Sphere	*Quality settings*	*Average distance ± SD (mm)*	*Maximum distance (mm)*
1	X3	0.86 ± 0.61	3.20
	X2	0.67 ± 0.37	2.39
	X1	0.65 ± 0.46	2.80
2	X3	0.54 ± 0.25	1.86
	X2	0.45 ± 0.21	1.32
	X1	1.09 ± 0.72	3.99
3	X3	1.36 ± 0.75	4.12
	X2	0.97 ± 0.60	4.09
	X1	1.19 ± 0.70	3.84

Scan data using quality x1 and x2 showed the best performance (i.e. lower distance between points in the reference scan and points in the assessed scan, Table 5.6). Interestingly, the results for quality x3 show this setting produced higher average distances and a higher maximum distance (Table 5.6). In addition, quality x3 will requires at least 57 minutes (Table 5.6) to complete a scan at the highest scan resolution, and since fog is common in the Cantabrian Mountains (Heras, 2002), a short period of data collection may only be possible on some days. Quality x2 was selected as a compromise between time for data collection and the quality of the data to reduce potential noise.

5.2.2. Application of TLS for measuring peat erosion

5.2.2.1. Study areas

The application of TLS for measuring peat erosion was trialled in three areas of blanket bog identified in chapter 3 (Zalama, Ilsos de Zalama and Collado de Hornaza; Figure 5.5) that are located on the regional mountain borders of Cantabria, Basque Country and Castilla y León. These three sites were selected to allow comparison between a restored and unrestored blanket bog at Zalama and Ilsos de Zalama located only 500 m apart, and comparison between these two blanket bogs in the Ordunte Sector and the unrestored and unprotected Collado de Hornaza blanket bog in the Cantabrian sector (Figure 3.5). The area is dominated by oceanic climatic conditions (Heras, 2002) and during the research period for this trial (May 2017 – June 2017), climatic conditions were comparable across all sites with a mean air temperature ranging from 13.1°C to 13.8°C, mean wind speed between 11.6 km/h – 12.5 km/h, humidity from 79.5% – 80.4% and rainfall from 124.6 mm to 135.1 mm (Meteoblue, 2017). There is no arid season as in summer months, occult precipitation continues from cloud that encompasses the mountain tops (see chapter 3; Figure 3.3).

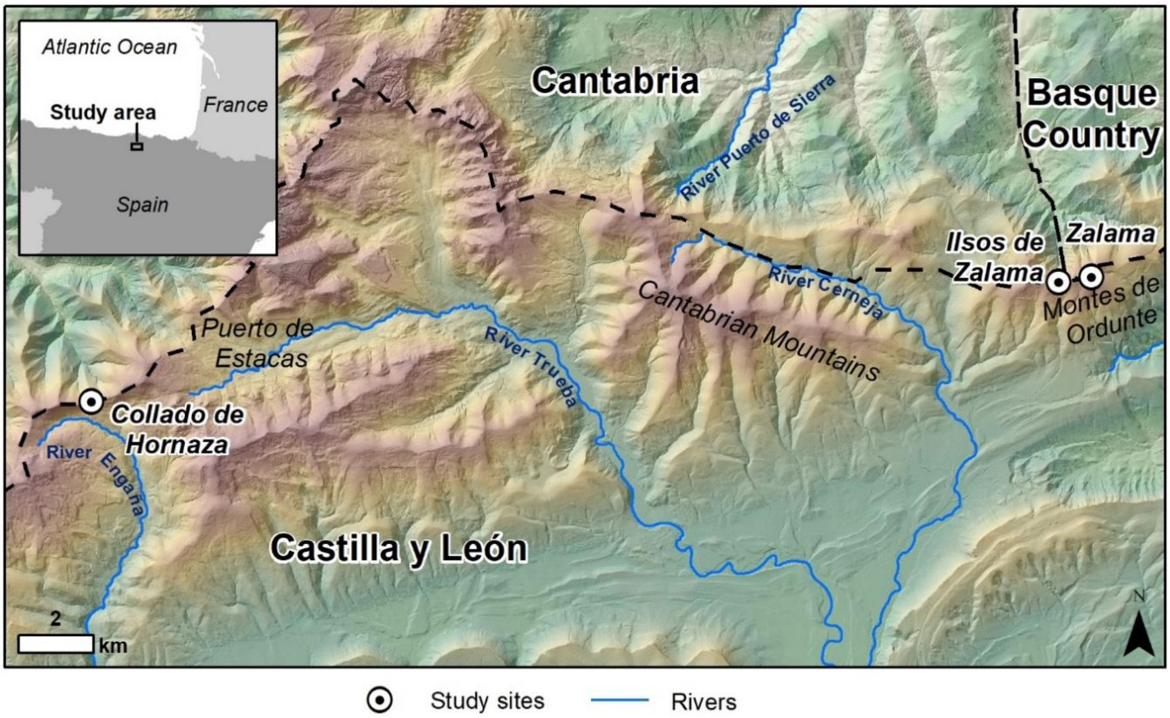


Figure 5.5. Study area locations of TLS erosion experiments.

5.2.2.1.1. Zalama

Zalama is a blanket bog (Heras, 2002) with three distinctive mesotopes units (spur, watershed and saddle mire; Figure 3.7) located in Montes de Ordunte between the administrative regions of Basque Country and Castilla y León at an altitude of 1330 masl. Peat covers an area of approximately 6.3 ha with a peat depth up to 2.82 m (Table 3.5) and basal layers have been dated at 8,000 years old (Pérez-Díaz *et al.*, 2016). Zalama is currently the only site included that has been designated under Natura 2000 as blanket bog (7130) and, although some areas remain degraded, undergoing restoration actions are reporting a favourable trend in terms of vegetation and water retention (Chico and Clutterbuck, 2019). In 2008, a fence was installed around the perimeter of the main peat body covering 3.4 ha (Figure 4.6) to exclude large grazing livestock (specifically cattle and horses) and additional fencing was installed in 2017 to protect a further area of 2.6 ha of the bog margin. Subsequently, areas of exposed peat ranging from horizontal to slopes of up to 30° have been covered with geotextile (coconut fibre sheets held in place using square wooden frames; Figure 5.6A) and planted with peatland species such as *Eriophorum vaginatum* under the project LIFE+ Sustainable Ordunte funded by the European Union and Bizkaia Provincial Council. A number of vertical, concave peat faces are, however, still exposed to erosion processes (Figure 5.6A).

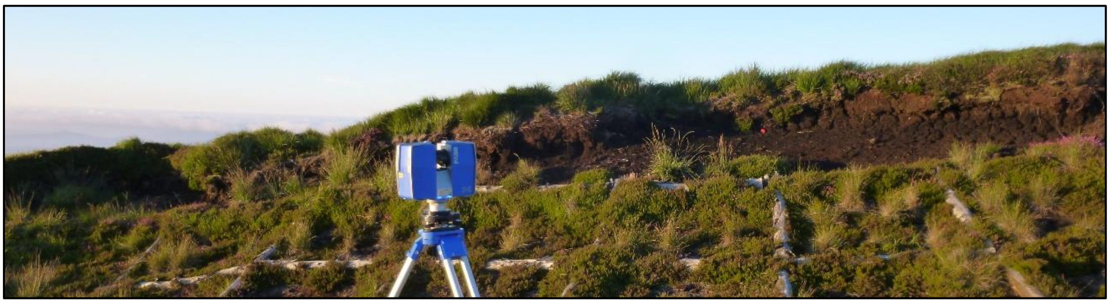
5.2.2.1.2. Ilsos de Zalama

This blanket bog is located approximately 500 m to the west of Zalama blanket bog at an altitude of 1280 masl. Peat covers an area of 3.1 ha with a maximum peat depth of 2.16 m (Table 3.5). The peatland is a good example of saddle mire (Figure 3.7) with a central raised portion indicating the ombrotrophic status. The blanket bog is degraded with several near horizontal areas of exposed peat (Figure 5.6B) and a raised 'tongue' of intact peat with vertical, concave exposed peat faces on all sides. In contrast to Zalama blanket bog, there are no structures to protect the exposed peat as highlighted in Chapter 4 and livestock graze the area between April and October.

5.2.2.1.3. Collado de Hornaza

Collado de Hornaza is located 3 km south-west of the mountain pass Estacas de Trueba between Cantabria and Castilla y León administrative regions (Figure 5.5). This site is approximately 25 km west from Zalama blanket bog at an altitude of 1280 masl. Peat accumulation at Collado de Hornaza extends northwards (Figure 3.6) and covers an area of 3 ha with a maximum peat depth of 2.75 m (Table 3.5). Exposed peat in this area consists primarily of two 'islands' with vertical, concave exposed peat faces on all sides (Figure 5.6C). Similar to Ilsos de Zalama, there is no protection of the peat from livestock, and in addition to grazing, burning of vegetation to improve browse is undertaken locally from November to April.

A *Zalama*



B *Ilsos de Zalama*

C *Collado de Hornaza*

Figure 5.6. Study areas. A) Zalama blanket bog with an example of restoration geotextile material, B) Ilsos de Zalama blanket bog, C) Collado de Hornaza blanket bog.

5.2.2.2. Experimental design

Each study site was scanned on one day between 22nd and 23rd of May 2017 and again between 16th and 18th of July 2017 using a FARO Focus3D X330 terrestrial laser scanner. All scans were completed using the optimal settings determined previously (resolution 1 and quality x2). To exclude potential large errors introduced by registering/aligning multiple point clouds, also known as methodological errors (Smith, 2015), the approach for this study adopted a single scan on each site. A fixed ground reference marker was installed in the bedrock on the first scan to allow precise positioning of the tripod and scanner in repeat surveys. On each survey date, scans were repeated from the same location to allow assessment of instrumental errors. This error is related with the device and they are usually systematic (Smith, 2015). All survey areas were within 16 m of the scanner and 60% of the areas were within 10 m thereby obtaining very high density of points in the data collected over the AOI (Figure 5.7).

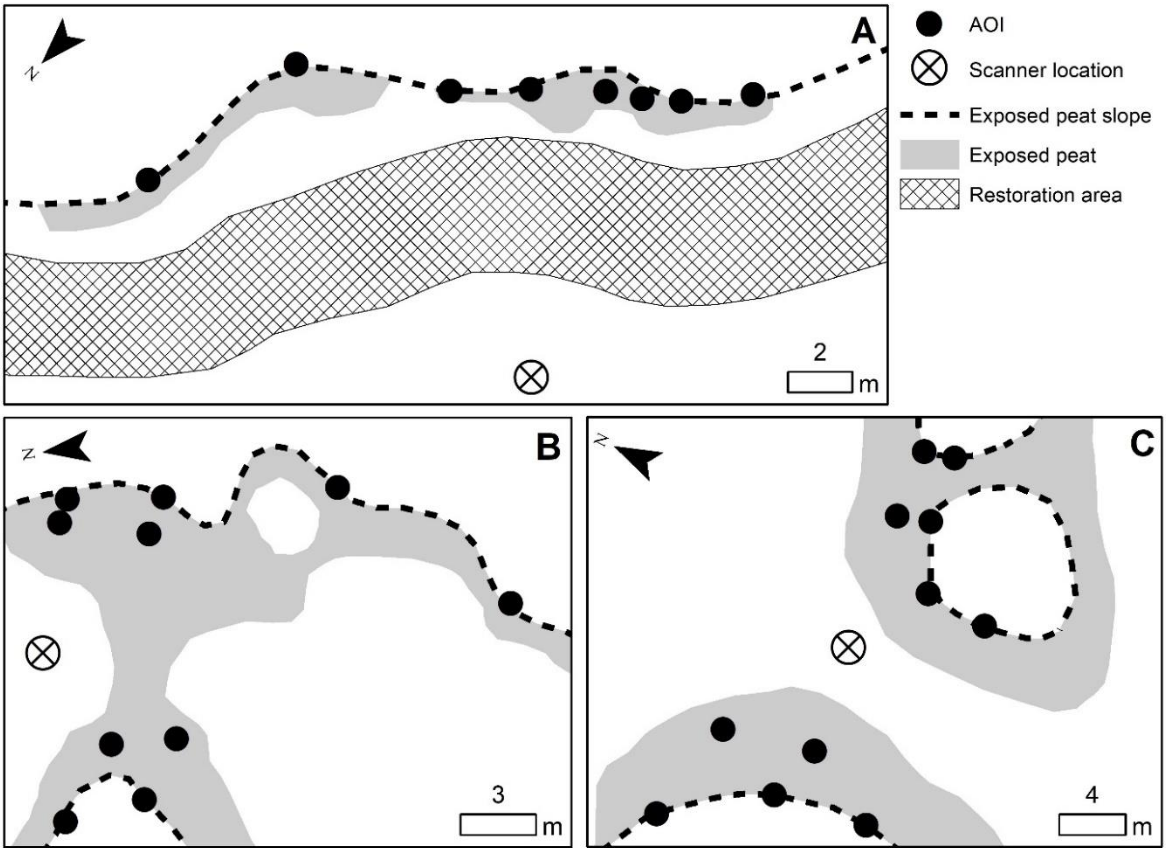


Figure 5.7. Schematic of study areas and survey strategy. A) Zalama, B) Ilsos de Zalama, C) Collado de Hornaza. Black circles indicate the centre of the selected AOI.

5.2.2.3. Survey area selection and fixed reference markers

To mitigate damage to the sensitive vegetation and exposed peat across the areas, a single scanning location for the largest area of exposed peat present at each site, oriented N – NW, was selected. At Zalama the survey area comprises a near vertical peat face with low angle sloping surfaces extending out from the base, although the area of exposed peat available was considerably reduced as a result of the restoration actions. At Ilsos de Zalama and Collado de Hornaza, it was possible to capture several near vertical peat faces and low angle sloping surfaces. Four fixed markers were inserted into the peat faces at each site to improve the accuracy of multi-temporal scan data alignment. Makers were 60 cm in length with a red circular end 5 cm in diameter that contrasts against the colour of exposed peat surfaces (Figure 5.8).

Figure 5.8. Example of fixed marker (5 cm diameter red disk) positioned on a near-vertical exposed peat surface that was used to improve the accuracy of multi-temporal scan data alignment.

5.2.2.4. Scan data processing and registration

Data from the scanner were imported and processed initially using FARO SCENE version 7.1.1.81. Point clouds were colorized using the photographs captured by the scanner to improve identification of fixed reference markers. The data were then filtered in FARO SCENE using a stray point algorithm provided in the software to remove erroneous data points resulting from dust particles or water vapour in the air. Subsequently an edge artefact filter was used to remove noise around edges of features such as slumped blocks of peat.

Scan point clouds were registered using the fixed reference markers as reference points and were left in a local coordinate system with the scanner location as the origin. To align scan point clouds, the x, y and z local coordinate for each fixed reference marker from the first scan was applied to the same markers in subsequent scans and registered. The point cloud for each scan was exported separately for comparison and clipped to the area of exposed peat to remove returns from objects outside the AOI. It was apparent in the July survey that at Ilsos de Zalama two fixed reference markers had been covered by slumping peat and at Collado de Hornaza two fixed reference makers had been physically removed by an unknown person or animal. For registration of these scan data, two areas of bedrock were used as extra reference points in addition to the two remaining fixed makers.

5.2.2.5. Areas of interest selection

Between May and July 2017 changes in vegetation obscured parts of each study site (e.g. *Eriophorum vaginatum* at Zalama blanket bog) that were surveyed or included in the AOI in May. Therefore, the total point cloud for each site was reduced to comprise multiple individual AOI (Figure 5.8) that were visible in both datasets and can be used for future comparison (e.g. annual and seasonal rates of erosion in chapter 6). In addition to vegetation, areas with obstructions such as rocks or geotextiles (in the case of Zalama) were also excluded as change in their morphology was not of interest for this research. The number of AOIs identified ranged from 8 at Zalama covering the area of exposed peat above the restored area (Figure 5.7A), 10 AOI at Ilsos de Zalama (Figure 5.7B) and 12 at Collado de Hornaza (Figure 5.7C). For all sites these AOI cover a range of near vertical peat faces and low-angle sloping areas.

5.2.2.6. Error assessment

To further assess instrumental errors (TLS), the point cloud for the repeat scan at each AOI was compared with the first scan using the point to point comparison tool (C2C) in CloudCompare software. This tool measures distances between the two clouds using the Hausdorff distance (Girardeau-Montaut, Roux and Thibault, 2005) and any difference determined in point location here quantifies variation in scan geometry. Potential increase in error with distance from scanner in the point clouds was tested using Pearson’s correlation in R 3.6.2. Methodological error from multi-temporal alignment of the scans taken in May and July 2017 at each site were reported in FARO SCENE software and extracted for each study area.

5.2.2.7. Determining surface difference and volume change

To quantify change between May and July 2017, the point cloud for the scan taken in May at each survey site was converted to a 3D mesh using FARO SCENE software with maximum data resolution in the output (Figure 5.9; Table 5.7). The mesh was exported and compared with the point cloud from July data for each AOI using the Mesh to Cloud (M2C) algorithm in CloudCompare. This method determines the signed distance between each point in the cloud from July and the mesh data created from May. It creates a new point cloud (e.g. Figure 5.10) where each point has the signed distance assigned (Monserrat and Crosetto, 2008).

To compare overall change between sites, points from all AOI at each site were combined and the mean (overall) surface difference between May and July calculated for all sites. As the mean difference in each site will obscure the magnitude of both positive and negative changes, the data were split by signed values and the mean negative and positive surface changes determined separately for each site. Subsequently, the mean difference between May and July surfaces was calculated for individual AOI in all sites. Mean negative and mean positive surface change for each AOI were determined separately and volume change by unit area determined for each AOI.

Figure 5.9. Example of a point cloud generated in FARO SCENE and visualised in CloudCompare.

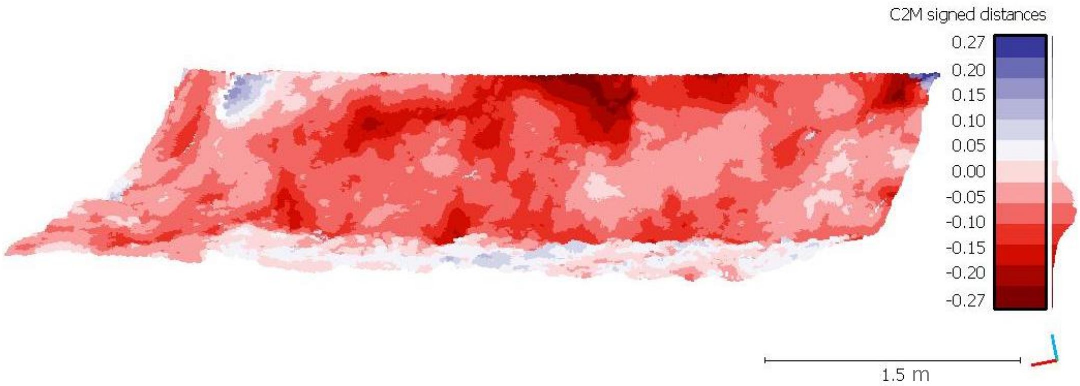


Figure 5.10. Example of point cloud after C2M calculations between both periods (May – July 2017). On red, erosion and on blue, deposition.

As the data for all sites were not normally distributed, the values of difference (change) quantified for each site were compared to the values of difference for the other two sites using the Mann-Whitney test. This was undertaken first using all difference values and then on all negative and all positive difference values separately. All analyses were undertaken using R 3.6.2.

5.3. RESULTS

The total surface area of all combined AOIs varied between sites, with the smallest area assessed in Zalama (11.7 m^2; Table 5.7). This difference largely arises from the nature of the sites and exposed peat surface available for survey, particularly the reduction in exposed peat at Zalama following restoration actions (Figure 5.6A). Mean point cloud densities across the sites ranged from 163,698 to 555,260 pts m^{-2} (Table 5.7), and relate to distance from the scanner (Table 5.8). If the points were evenly distributed in the data this would equate to a mean point spacing of 1.3 – 2.3 mm. Of particular note is that for each site the mean point density between surveys varied by < 0.1 % (Table 5.7), indicating a consistent survey strategy.

Table 5.7. Survey areas, point densities and mesh resolutions in each study area for May and July 2017.

Site		*Zalama*	*Ilsos de Zalama*	*Collado de Hornaza*
Total survey surface area (m^2)		11.70	26.88	91.37
Number of points covering survey area				
	May	1,915,269	14,925,396	25,710,537
	July	1,903,390	14,962,164	24,556,916
Mesh resolution (Number of faces, May)		5,501,743	9,320,604	14,231,857
Variation in point density (May, pts m^{-2})				
	Mean	163,698	555,260	281,389
	Maximum	209,494	1,167,405	556,348
	Minimum	95,430	59,020	54,807
Variation in point density (July, pts m^{-2})				
	Mean	162,683	556,628	268,763
	Maximum	212,698	1,165,508	534,173
	Minimum	86,770	57,485	53,389

5.3.1. Error assessment

5.3.1.1. Instrumental error

Analysis of repeat scans showed that with the exception of two AOI at Ilsos de Zalama in May and two AOI at Collado de Hornaza in July (Table 5.8), the mean distance difference was < 2 mm and falls within the reported ranging error of the scanner (± 2 mm; FARO technologies Inc., 2015). The four instances of higher mean error were less than 4 mm (Table 5.8) indicating that at the distance surveyed, error introduced by variations in scan geometry appears minimal. Error of scan geometry did not correlate with the distance from the scanner ($r = 0.22$, $p > 0.05$).

Table 5.8. Surface area and instrumental error for AOIs in each study site for May and July 2017.

			May		***July***	
AOI	*Surface area (m^2)*	*Distance from scanner (m)*	*Mean error (mm)*	*SD (mm)*	*Mean error (mm)*	*SD (mm)*
Zalama						
LZ1	1.51	11.2	0.4	1.3	0.5	2.1
LZ2	1.26	9.8	0.01	2.1	0.2	2
LZ3	1.38	9.3	0.3	0.9	0.2	0.9
LZ4	1	9.2	0.6	0.7	0.6	0.7
LZ5	4.10	8.9	0.6	1.8	0.4	2.3
LZ6	0.81	9.2	0.7	1.9	0.2	6
LZ7	0.32	12.1	1.4	1.6	0.5	2.9
LZ8	1.32	13.4	1.4	3.4	0.05	13.5
Mean		**10.4**	**0.7**	**0.5**	**0.3**	**0.2**
Ilsos de Zalama						
ILZ1	3	6.8	1.7	2.6	1.2	1.9
ILZ2	6.33	4.6	0.4	2	1.4	2.4
ILZ3	0.55	7.1	2.7	2.6	0.3	1.6
ILZ4	3.18	6.3	1.1	2.1	0.4	3.2
ILZ5	1.57	15.9	0.7	3.3	1.2	3.5
ILZ6	0.66	13.5	3.9	3.6	0.1	2.6
ILZ7	2.41	7.8	1.2	2.2	0.3	2.1
ILZ8	6.63	6.3	0.7	2	0.1	2.5
ILZ9	1.41	6.2	0.1	2.1	0.5	1.7
ILZ10	1.14	5.2	0.01	1.8	0.7	1.2
Mean		**8**	**1.6**	**1.2**	**0.6**	**0.5**
Collado de Hornaza						
CH1	5.23	15.9	1.5	1.3	2.7	3.3
CH2	1.93	12.6	0.9	1.4	1.3	3
CH3	6.44	8.4	0.8	1.3	1.7	2.4
CH4	4.71	13	0.4	1	0.8	2.2
CH5	4.40	9	0.9	1.1	1.2	3.9
CH6	19.86	5.8	0.04	1.5	3.2	3.7
CH7	3.14	8.3	0.3	1.2	0.7	4.7
CH8	11.66	10.8	1.4	1.8	0.5	3.7
CH9	11.35	10	0.4	2.3	0.7	3.6
CH10	14.31	6.7	0.9	1.5	0.2	2.7
CH11	1.97	9.1	0.3	0.8	0.3	2.6
CH12	5.97	15.4	0.1	1.7	0.8	3.6
Mean		**10.4**	**0.7**	**0.5**	**1.5**	**1**

5.3.1.2. Methodological error

For Zalama, where all four fixed reference markers were present in both surveys, the maximum error of registration of May and July 2017 point clouds ranged from 0.6 – 0.8 mm (Table 5.9). Despite the loss of two fixed reference markers at both Collado de Hornaza and Ilsos de Zalama noted in July, scan registration error was still < 4 mm for Collado de Hornaza and < 7 mm at Ilsos de Zalama.

Table 5.9. Methodological error per site derived from the multi-temporal point cloud alignment.

Site	*Mean (mm)*	*SD (mm)*	*Min (mm)*	*Max (mm)*
Zalama	0.7	0.1	0.6	0.8
Ilsos de Zalama	5.4	0.8	4.8	6.5
Collado de Hornaza	2.7	1.4	0.7	3.7

5.3.2. Determining surface differences and morphological changes

5.3.2.1. Peatland surface change by study sites

A negative surface difference was identified for the majority of points at each site between May and July 2017 (63 – 72 %; Table 5.10), indicating that erosion at Zalama blanket bog and peat loss at Collado de Hornaza and Ilsos de Zalama blanket bog are the dominant surface processes occurring in the areas assessed. The overall mean surface difference for each site ranged from -2.8 ± 6.9 mm (mean ± SD) for Zalama, to -6.8 ± 28.7 mm at Ilsos de Zalama and -19.9 ± 40.7 mm at Collado de Hornaza (Table 5.10; Figure 5.11). The range of difference from the 1 % to 99 % percentiles determined at Zalama (36 mm) was four times lower than the range of values determined at Ilsos de Zalama (149 mm) and six times lower than the range of values determined at Collado de Hornaza (215 mm; Figure 5.11). A Mann-Whitney test identified that overall change at Zalama was significantly different to the change at both Ilsos de Zalama ($W = 1.64e+13$, $p < 0.001$) and Collado de Hornaza ($W = 3.01e+13$, $p < 0.001$), and that overall change was also significantly different between Ilsos de Zalama and Collado de Hornaza ($W = 2.13e+14$, $p < 0.001$).

Table 5.10. Mean surface difference for all study sites.

Site		*Mean (mm)*	*SD (mm)*	*Proportion of point data (%)*
Zalama				
	Overall	-2.8	6.9	100
	Erosion	-5.9	4.6	72
	Deposition	4.9	5.5	28
Ilsos de Zalama				
	Overall	-6.8	28.7	100
	Erosion	-22.9	20.5	63
	Deposition	20.9	17.4	37
Collado de Hornaza				
	Overall	-19.9	40.7	100
	Erosion	-35.8	37	70
	Deposition	17.3	18.8	30

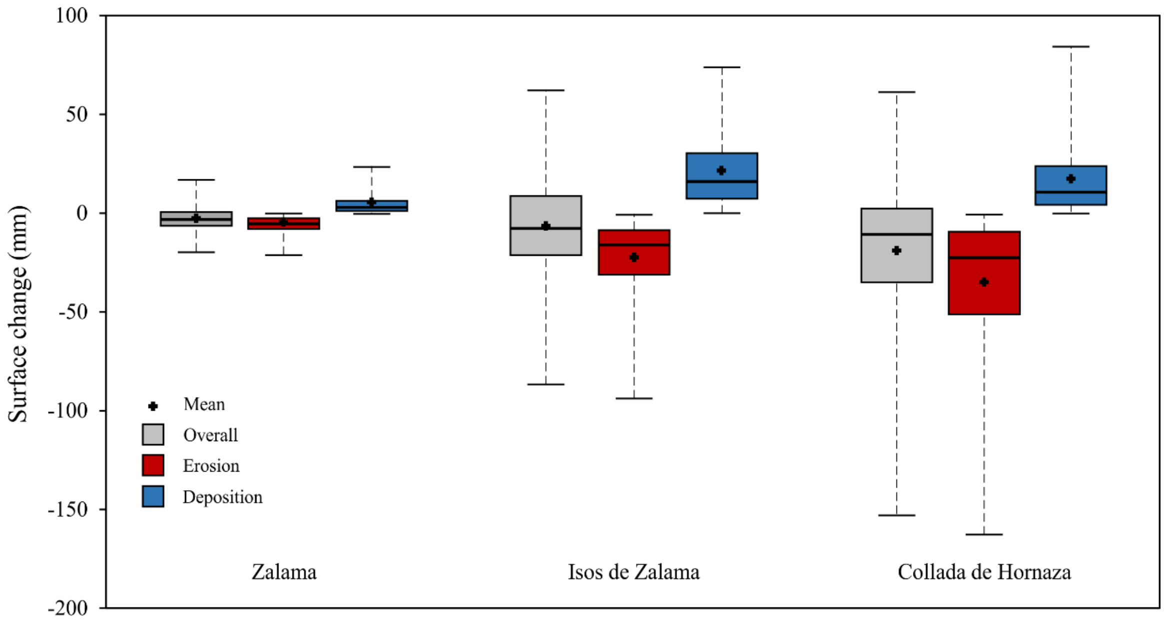


Figure 5.11. Distribution of surface change values for Zalama, Ilsos de Zalama and Collado de Hornaza showing 1 %, 25 %, 50 %, 75 % and 99 % percentiles.

Treating positively and negatively signed difference data separately highlights a greater magnitude of change at each site. The mean surface difference determined from negative values at Zalama indicated larger change compared to the overall mean difference (-5.9 ± 4.6 mm; Table 5.10), but difference values of up to -22.9 ± 20.5 mm and -35.8 ± 37 mm were identified at Ilsos de Zalama and Collado de Hornaza (Table 5.10). In addition, the mean surface difference determined from positive values (29 – 37 % of points) identified deposition ranging from 4.9 ± 5.5 mm at Zalama to 17.3 ± 18.8 mm and 20.9 ± 17.4 mm at Collado de Hornaza and Ilsos de Zalama, respectively (Table 5.10; Figure 5.11). While the maximum scan methodological error at Ilsos de Zalama (6.5 mm) was of the same magnitude as the mean overall change detected at this site (6.8 mm), the methodological error was 3 – 4 times lower than the mean negative and mean positive change determined. A Mann-Whitney test identified that negative change at Zalama was significantly different to the change at Ilsos de Zalama ($W = 1.07e+13$, $p < 0.001$) and Collado de Hornaza ($W = 1.97+e13$, $p < 0.001$) and that the negative change was also significantly different between Ilsos de Zalama and Collado de Hornaza ($W = 9.42e+13$, $p < 0.001$). For positive values, a Mann-Whitney test identified a significant difference between positive change at Zalama and both Ilsos de Zalama ($W = 4.94e+11$, $p < 0.001$) and Collado de Hornaza ($W = 9.65e+11$, $p < 0.001$). Positive change was also significantly different between Ilsos de Zalama and Collado de Hornaza ($W = 2.37e+13$, $p < 0.001$).

5.3.2.2. Peatland surface change by AOI

At Zalama, the mean overall surface change measure for all AOI between May and July 2017 was negative (Figure 5.12). Mean overall surface change measured in the AOI at Collado de Hornaza and Ilsos de Zalama was also predominantly negative, expect for two AOI at Collado de Hornaza and three at Ilsos de Zalama that produced a positive overall mean change, indicating that deposition may be a dominant surface process in some AOI (Figure 5.12). There was a clear difference in the variability and magnitude of the overall change between the sites assessed, as for Zalama mean overall change for all AOI ranged from -8 mm to -1 mm while at Ilsos de Zalama and Collado de Hornaza mean AOI change figures of -40 mm to +6mm and -67 to +9mm respectively were noted (Figure 5.12). This contrast between sites is even greater when positively and negatively signed points were analysed separately; for Zalama, the range of mean erosion rates in the AOI compared with the overall change increases by 1.4 mm to -9.4 mm, whereas for Ilsos de Zalama and Collado de Hornaza particularly, mean erosion rates plus peat loss of up to -42.6 mm and -82.1 mm were noted (Figure 5.13). Interestingly the proportion of points contributing negative values for the larger estimates in these two AOI was 95% at Ilsos de Zalama and 87% at Collado de Hornaza (Table 5.11).

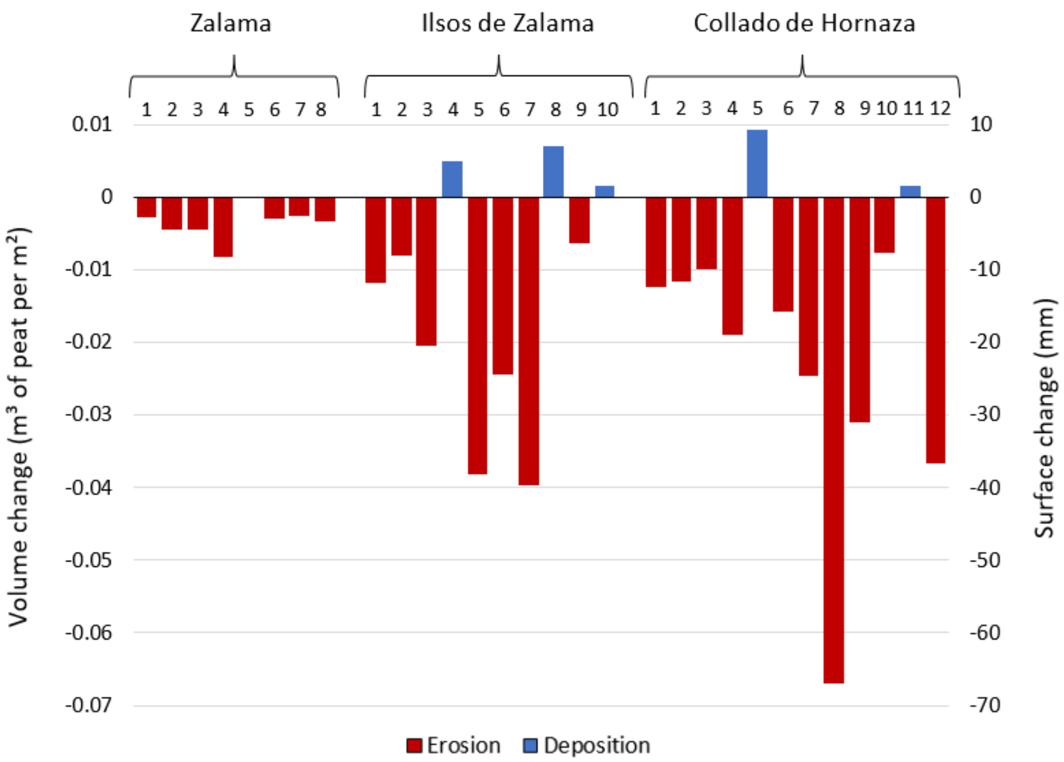


Figure 5.12. Mean peat volume change and mean surface change in each AOI.

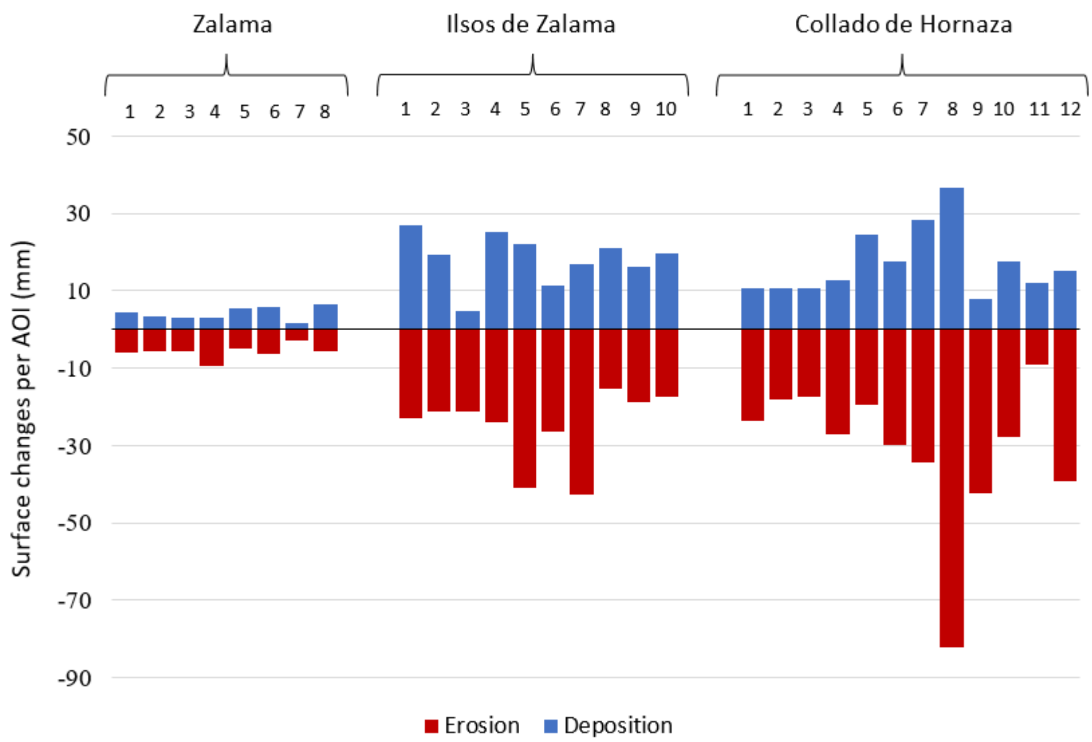


Figure 5.13. Mean positive (deposition) and mean negative (erosion) surface change measured by AOI.

It is also interesting to note that when analysing positively signed points separately, all AOI indicated some level of deposition occurring (Figure 5.13). At Zalama blanket bog all deposition estimations were lower than 7 mm, but at Ilsos de Zalama, 10 of the 11 AOI indicated rates of deposition over 10 mm with two AOI indicating up to 28 mm. At Collado de Hornaza, 11 of the 12 AOI indicated rates of deposition over 10 mm and the AOI where -82 mm of erosion/peat loss was recorded and up to 34 mm of deposition had occurred. It is interesting, therefore, that the AOI with the highest negative change also had the highest value of deposition at Collado de Hornaza. It should be noted that the large deposition identified at this AOI (number 8) was derived from only 13 % of the points recorded; at Ilsos de Zalama, both larger estimates of deposition were derived from 22 – 59 % of the points for the AOI (Table 5.11).

Table 5.11. Proportion of points in surface differences for each AOI and study site.

AOI	*% Accumulation*	*% Erosion/Peat loss*
Zalama		
LZ1	30.3	69.7
LZ2	99.3	0.7
LZ3	13	87
LZ4	9.3	90.7
LZ5	47.8	52.2
LZ6	27.7	72.3
LZ7	8.3	91.7
LZ8	17.5	82.5
Ilsos de Zalama		
ILZ1	22.3	77.7
ILZ2	32.1	67.9
ILZ3	2.8	97.2
ILZ4	58.6	41.4
ILZ5	4.3	95.7
ILZ6	5.2	94.8
ILZ7	5	95
ILZ8	61.7	38.3
ILZ9	35.6	64.4
ILZ10	51.2	48.8
Collado de Hornaza		
CH1	33.1	66.9
CH2	22.7	77.3
CH3	26.2	73.8
CH4	20.4	79.6
CH5	65.1	34.9
CH6	29.5	70.5
CH7	15.3	84.7
CH8	12.7	87.3
CH9	22.6	77.4
CH10	44.6	55.4
CH11	50.2	49.8
CH12	4.1	95.9

At least two of the AOI assessed at each site solely comprised sections of near vertical peat and it is clear that the two highest measurement of erosion in AOI at Ilsos de Zalama (-40.9 to -42.7 mm) and the three highest measurements of erosion in AOI at Collado de Hornaza (-39.2, -42.3 and -82.1 mm) were identified for near vertical section of exposed peat (Appendix C). For the AOI with the highest erosion measured at -82.1 mm, the overall volume change was -0.068 m^3 m^{-2} (Figure 5.12).

5.4. DISCUSSION

5.4.1. Applicability of TLS for assessing peatland erosion and peat loss

Rates of erosion and peat loss in peatlands vary spatially and temporally. Recent advances in the spatial resolution of data derived from geospatial techniques have allowed the increased application of remotely sensed data for monitoring peatlands (Clutterbuck *et al.*, 2018) and offer finer scale measurement of change in peatlands compared with traditional techniques such as erosion pins or sediment traps. Data from UAV-mounted, ground-based cameras (Glendell *et al.*, 2017) and TLS (Grayson *et al.*, 2012; Glendell *et al.*, 2017) are seeing direct application for quantifying rates of erosion in peatlands with mm resolution.

Estimation of change using any remote sensing approach though, requires comparable resolution of repeat survey data and quantification of errors associated with the technique to ensure that errors derived from the instruments or the methods applied are not greater than the rate of change. Terrestrial laser scanning has been used widely in erosion studies (e.g. Milan, George and Hetherington, 2007; O'Neal and Pizzuto, 2011; Schürch *et al.*, 2011; Goodwin *et al.*, 2016; Dąbek *et al.*, 2018) capturing ultra-high resolution data comprising over 80,000 pts m^{-2}. Where multiple scans overlap, resolutions of over 390,000 pts m^{-2} have been reported close to the scanner (Brasington, Vericat and Rychkov, 2012), yet in studies assessing peatlands, resolution of TLS data range from 24 pts m^{-2} (Höfle, Griesbaum and Forbriger, 2013) to 4,800 pts m^{-2} (Glendell *et al.*, 2017). The application of TLS was explored in this study to quantify the rate of surface change in three blanket bogs in northern Spain and mean resolution of data obtained across the sites ranged from 163,698 – 555,260 pts m^{-2} (Table 5.7). If the points were evenly distributed across the surface this would equate to a mean point spacing of 1.3 – 2.3 mm highlighting the need for extremely high levels of accuracy to enable comparison of the survey data.

Errors in TLS data arise for a number of sources including the scanner specification and the operational settings (ranging error, beam divergence and data ‘quality’), environmental factors (presence of vegetation, dust particles or water vapour) and methodological errors such as point cloud registration/alignment, specifically relating to registration target type, number, distribution and position (often recorded using differential GNNSS; Grayson *et al.*, 2012; Smith, 2015; Hall, 2016).

For all three sites assessed, the scanner was located within 15.9 m (Table 5.8) of the survey areas and this may explain the far higher resolution of data obtained compared with other studies assessing peatland surface changes (Grayson *et al.*, 2012; Höfle, Griesbaum and Forbriger, 2013; Glendell *et al.*, 2017). However, it is of particular interest that the resolution of scan data collected in this study varied by < 0.1% between surveys. Smith (2015) highlighted the impact of using different scanner resolution or data quality settings on methodological error and this was mitigated in this research by adopting the same scanner settings in each scanning campaign. Variation of scan geometry (instrumental error) could also impact upon the accuracy of the data, although comparison of repeat scan data indicated that the mean error of point location between scans was < 2 mm for 26 of the 30 AOI assessed in this research (Table 5.8). Larger instrumental error recorded for four of the AOI was still < 4 mm (2.7 – 3.9 mm; Table 5.8) and did not increase with the distance from scanner. These observations indicate that at the range employed here, the impact of variation in scan geometry and beam divergence on data collected using the FARO X330 appears minimal and within the range provided in the manufacturer report (FARO technologies Inc., 2015).

In order to avoid any environmental related error, all scans were undertaken when conditions were clear (e.g. no fog or visible water vapour in the air), although stray point filtering incorporated in FARO SCENE software was used to remove erroneous data points that may have included dust particles or water vapour. The impact of vegetation obstructing survey areas has been commonly noted (e.g. Grayson *et al.*, 2012; Hall, 2016; Dąbek *et al.*, 2018) and changes in vegetation between surveys in this study did present a problem by obscuring sections of the survey area. Filters could be used to remove vegetation (Dąbek *et al.*, 2018; Ordóñez *et al.*, 2018; Clutterbuck *et al.*, 2020a), but to prevent false identification of change, only those areas where exposed peat was openly visible were selected by excluding areas of vegetation, rocks or geotextile. This reduced the total area evaluated but enabled a far greater extent of peat to be assessed compared to more traditional techniques such as erosion pins.

The largest source of error in TLS data occurs in the registration of multiple scan point clouds, also called methodological error (Smith, 2015). To assess change in areas larger than that assessed here, multiple contemporaneous TLS scan data are frequently combined (Milan, George and Hetherington, 2007; Schürch *et al.*, 2011; Grayson *et al.*, 2012; Höfle, Griesbaum and Forbriger, 2013; Hall, 2016; Glendell *et al.*, 2017), but as the rate of erosion for blanket bog in Spain was not known, and could potentially be mm, any additional error introduced by aligning multiple scans was considered too high. Therefore, in this study a single scan strategy was used for each temporal survey to remove any potential error associated with the process. It was noted that this strategy reduced the potential scan area available and also reduced the ability to monitor complex 3D morphology that would require scans from more than one angle. Methodological error was therefore limited to the alignment of individual multi-temporal point clouds for each site. As the instrumental error (scan geometry) here appears minimal, the error of alignment is likely to reflect the relative geometry of marker locations. It is also relevant to remark that the nature of peat and other factors including 'mire-breathing' can influence the ability to align multi-temporal data (Grayson *et al.*, 2012), and largest fluctuations in vertical, but also horizontal peat surfaces are reported to occur between seasons of the year as a result of water content (Schlotzhauer and Price, 1999; Glaser *et al.*, 2004). As the errors of alignment determined for all sites here are lower than scan registration errors reported in non-peat soils (11 – 13 mm; Goodwin *et al.*, 2016), this suggests that the phenomenon of 'mire-breathing' was minimal in the sites assessed between May and July 2017, but this may have a greater influence on longer term data (Chapter 6).

Terrestrial laser scanning is being used to assess geomorphological change in a range of complex environments (Milan, George and Hetherington, 2007; Schürch *et al.*, 2011; Day *et al.*, 2013) and is increasingly being used to provide benchmark data to assess the accuracy of other surveying technologies (Castillo *et al.*, 2012; Glendell *et al.*, 2017; Godfrey *et al.*, 2020). The resolution of data obtained in this study (mostly <2 mm) and the mean combined instrumental and methodological errors (1.7. mm at Zalama, 7.6 at Ilsos de

Zalama and 4.8 mm at Collado de Hornaza; Table 5.8; Table 5.9) indicate that TLS, particularly with the use of fixed reference markers, is an appropriate approach to quantify mm resolution surface change in peatlands. The cost and weight of TLS units have been highlighted as a disadvantage for this approach compared to UAV or ground-based photogrammetry (Glendell *et al.*, 2017), but with a weight of 5 kg, the FARO X330 is extremely portable and appropriate for remote field surveys. Although the cost of TLS units are in excess of GBP30,000, the FARO X330 can be hired in the United Kingdom for less than GBP250 per day. In addition, the combined reported error in this study was lower than errors reported for both UAV and ground-based SfM techniques for mapping erosion in peatland environments (Glendell *et al.*, 2017). The approach of mesh to cloud (M2C) to quantify surface change was preferred to the option of creating and comparing DEM data (e.g. Grayson *et al.*, 2012) as in a DEM each pixel can only have one value of z. In a 3D point cloud, it is possible for multiple points to have the same x and y coordinate but different z value, and therefore the M2C approach retains the ability to assess change of complex 3D morphology at fine-scale resolution (Monserrat and Crosetto, 2008; Lague, Brodu and Leroux, 2013) and thereby better understand the surface change in peatlands .

5.4.2. Assessing peatland surface changes in restored and degraded peatlands

The three areas of blanket bog assessed in this study are located within 25 km of each other, oriented in a N – NW aspect and located at an altitude ranging from 1270 – 1330 masl. During the period May 2017 to July 2017 all sites also experienced comparable air temperature, rainfall, humidity and wind speeds. As the rate of surface change determined at Zalama was identified to be significantly different to the rate of surface change determined at both Ilsos de Zalama and Collado de Hornaza, this indicates that other factors may be influencing the rate of surface change in exposed peat assessed here. It is worth noting that the rate of surface change determined at Ilsos de Zalama was also identified to be significantly different to the rate of surface change determined at Collado de Hornaza.

The identification of negative mean overall surface change for all sites indicates that erosion or peat loss was the dominant geomorphological process occurring in all sites. It is not surprising that the two highest measurements of erosion/peat loss at Ilsos de Zalama and the three highest measurements of erosion/peat loss at Collado de Hornaza were found in AOI that solely comprise near vertical peat faces (Appendix C). It is, however, of note that some degree of deposition was identified in all AOI at all sites, including those AOI that solely comprise near vertical peat faces. While it is possible that the far higher rates of erosion and peat loss determined at Ilsos de Zalama and Collado de Hornaza compared to Zalama might relate to slumping of peat as a result of fluvial and aeolian processes (Evans and Warburton, 2007), trampling by livestock has been suggested to increase natural erosion processes in this region (Heras and Infante, 2003). The comparable climatic conditions across the sites during the study and particularly at Zalama and Ilsos de Zalama (located only 500 m apart), support the suggestion of external anthropogenic influences in addition to natural erosion processes (section 2.2.3.3). Both cattle and horses were observed at Ilsos de Zalama and Collado de Hornaza on both surveys and striations/incisions resulting from livestock rubbing and scratching heads (horns) were visible in vertical peat faces (Figure 5.14B). In addition, where livestock trample over the peat, there was evidence of disturbance from hooves (Figure 5.14A). This disturbance from livestock might explain the apparent deposition determined on near vertical exposed peat faces, and it will be interested to monitor the morphology of the peat over a longer period (Chapter 6) as livestock are mainly in the area from May to September.

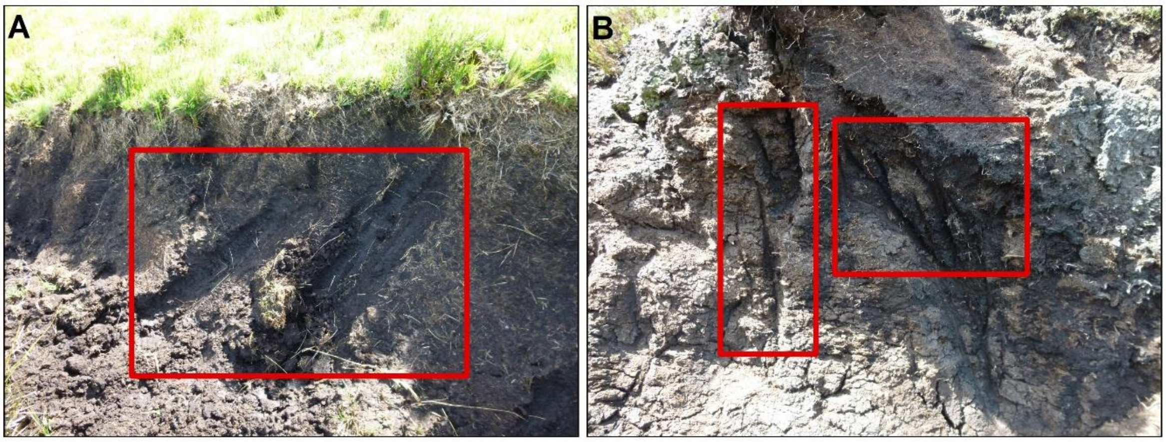


Figure 5.14. A) Peat disturbance from hooves and B) striations in exposed peat left by livestock horns.

The primary difference between Zalama, where very low rates of surface change (erosion) were identified and both Ilsos de Zalama and Collado de Hornaza, where significantly higher rates of change (peat loss plus erosion) were identified, is the restoration intervention and presence of a fence to exclude cattle and horses (Figure 5.15). The results indicate that, while surface change is occurring at Zalama, the presence of livestock is significantly increasing surface change at Ilsos de Zalama and Collado de Hornaza. However, the fence at Zalama did not exclude smaller livestock, and in both May and July 2017, herds of goats were observed to enter the fenced area at Zalama. It is possible, therefore, that some of the surface change determined at Zalama is not caused solely by natural erosion processes. It is also important to highlight that due to trampling of livestock in unprotected areas, some negatives changes could be a consequence of peat compaction, although the deposition showed clear movement of peat in the areas.

Figure 5.15. Visual difference in density and diversity of vegetation between restored and unrestored areas at Zalama highlighted by UAV-derived aerial imagery.

As highlighted in chapter 4, the numbers of livestock in the region of Cantabria has changed significantly over the last century (Figure 4.8) and in particular, there has been a nine-fold increase in the number of cattle. This trial application of TLS has only assessed two months (May – July 2017), but already reports the first rates of erosion (Zalama) and of peat loss

(Ilsos de Zalama and Collado de Hornaza) for blanket bogs in northern Spain. The erosion determined at Zalama over two months is around a quarter of the annual mean erosion figures for exposed peat in England and Wales (Table 5.1). However, and of great concern, the rates of peat loss and erosion determined in unprotected blanket bogs in this study are already equal to the annual rate of peat erosion in England and Wales (Ilsos de Zalama) and equal to the higher mean rates determined in Scotland (Collado de Hornaza; Table 5.1; Table 5.10). It will be important to continue to monitor this surface change over several years, but the method applied here provides rapid indication of the rate of surface change in blanket bogs.

5.5. CONCLUSION

This study demonstrates the application of TLS to quantify the rate of surface change in three recently mapped blanket bogs in the Cantabrian Mountains (northern Spain) from May – July 2017. With the use of fixed reference markers, portable TLS units such as the FARO X330 are able to collect mm resolution data and enable determination of surface change with mm level accuracy. The mean rate of erosion determined over two months for the area of exposed peat assessed in this study for the protected blanket bog (Zalama) was quantified at -5.9 mm. However, the mean peat loss/erosion in the areas assessed in the unprotected blanket bogs is 4 – 6 times greater (-22.9 mm at Ilsos de Zalama; -35.8 mm at Collado de Hornaza) and is already comparable to annual rates of erosion determined for exposed peat in the United Kingdom. These are the first quantified measurements of peat erosion/loss for Spain and the application of TLS has highlighted a significant impact of livestock on rates of peat loss. It is important to explore the spatial variation in erosion and peat loss across blanket bogs in northern Spain, and any seasonal variation resulting from the absence of livestock. This will be explored in Chapter 6.

Chapter 6

The current degradation status of blanket bogs in northern Spain – Seasonal and annual erosion and peat loss rates in restored and degraded blanket bogs

The content of this chapter has been partially published

6.1. INTRODUCTION

Anthropogenic pressures on peatlands are diverse, but in Spain peat extraction, windfarms and livestock have been reported to be the most significant factors that require intervention to facilitate conservation of blanket bogs (Heras and Infante, 2008; Heras *et al.*, 2017). Both peat extraction and the construction of windfarms can result in large-scale removal of peat and in some cases, the complete loss of the peatland (Heras and Infante, 2008; Heras *et al.*, 2017), but the rate of erosion of blanket bog in northern Spain, and the impact of livestock on erosion and peat loss, were previously not known. Assessment of surface change over just two months indicated that livestock result in a 4 – 6 times greater rate of peat loss for Spanish blanket bogs in comparison with blanket bogs exposed only to natural erosion processes (see section 5.4.2). There is a clear need to understand seasonal variation in erosion (periods with and without livestock), and to understand what this loss of peat means in terms of loss of carbon.

The impact of livestock has been studied on a range of soils (Evans, 1998; McHugh, 2000; McHung, 2007; Bilotta, Brazier and Haygarth, 2007; Newton *et al.*, 2009) and was specifically noted as having a negative impact on peatlands (Evans, 1977; Grant, Bolton and Torvell, 1985; McHugh, 2000; Yeloff, Labadz and Hunt, 2006). Livestock (grazing) has been described as a main cause of blanket bog degradation in protected areas of blanket bog in England (Yeloff, Labadz and Hunt, 2006; O'Brien, Labadz and Butcher, 2007) and Ireland (NPWS, 2019), but has also been listed as the main pressure and threat to the habitat 7130 – blanket bogs across Europe (European Environment Agency, 2012). In Spain, overgrazing has only been highlighted as a medium pressure and threat to blanket bogs (European Commission, 2012a), but overgrazing and associated burning practice has been cited as the reason for the reduction of 50% of the original extent of Zalama blanket bog (Heras and Infante, 2003). The pressure of livestock can cause a range of problems in peatlands environments including changes to the physical properties of the soil (Greenwood *et al.*, 1998), changes in vegetation cover (particularly reducing peat-forming species such as *Eriophorum vaginatum; e.g.* Ward *et al.*, 2007; Milligan, Rose and Marrs, 2015), impacts on the carbon balance of the peatland (Worrall and Clay, 2012) and increased areas of exposed peat (Grant, Bolton and Torvell, 1985; McHugh, 2007).

Trampling by livestock can increase the BD of the top layers of peat (Langlands and Bennett, 1973; Lindsay, 2010), and by increasing soil compaction (Langlands and Bennett, 1973; Worrall and Clay, 2012) reduce the rate of infiltration (Gifford and Hawkins, 1978). The BD of peat in cores taken from five blanket bogs examined in this research appears to show a gradual, but consistent decrease with depth (see section 4.3.1.2) suggesting that trampling from livestock may be increasing BD in the top layers of peat across the Cantabrian Mountains. Interestingly the BD of peat in the top layers at Zalama were notably higher than the BD of peat at any of the other four sites examined, despite the exclusion of livestock from this site since 2009, but it is worth noting that recovery of peat properties is a long-term process (Lindsay, 2010).

Grazing also reduces the cover of key peat-forming species such as *Sphagnum* (Ward *et al.*, 2007; Noble *et al.*, 2017), and burning activities undertaken to improve grazing can shift the vegetation from a dominance of peat-forming species (e.g. sedges or *Sphagnum*) to a dominance of other species such as *Calluna vulgaris* (Hobbs, 1984; Noble *et al.*, 2017). Such effects are clear at Zalama where there is a striking difference in the density and diversity of species within the fenced area compared to the grazed areas outside (Figure 5.15). Such change in vegetation composition can have a negative impact on the carbon balance of the peatland (Noble *et al.*, 2019), specifically reducing carbon stocks in the vegetation above ground (Ward *et al.*, 2007). Interestingly grazing also can increase the amount of CO_2 being absorbed by a peatland, but the balance of GHG is offset by an increased release of CH_4 at the same time (Ward *et al.*, 2007).

While changes in soil properties and vegetation may affect the ability of a peatland to sequester carbon and impact on gaseous exchanges, other impacts of grazing are more important in terms of direct peat loss, particularly by increasing the extent of exposed peat (Evans, 1977; McHugh, 2007). Hooves of livestock can displace chunks of peat (Figure 5.14), but exposing the peat surface leaves the peat susceptible to natural erosion processes. The nature and scale of natural erosion varies seasonally as snow cover will protect the peatland surface from other agents, such as wind or rainfall. However, temperature fluctuations during winter months can initiate freeze-thaw processes (Labadz, 1988; Li, Holden and Grayson, 2018). In summer, desiccation of the peat can lead to cracks in the surface enlarging also the extent of the exposed peat area (Evans, 1977) making the peat more susceptible to aeolian and fluvial erosion (Radley, 1962). Erosion of peat by wind can result in significant loss of peat in blanket mires, particularly if the peat surfaces are aligned with the direction of the wind where peat fluxes could be up to 13 times greater under moderate intensity frontal rainfall conditions (Foulds and Warburton, 2007a) . When wind and rainfall are combined, it can produce a wind-splash effect commonly reported across European Atlantic blanket bogs (Warburton, 2003; Foulds and Warburton, 2007a). Areas of blanket bog located in exposed, high altitude locations such as those in the Cantabrian Mountains might reasonably be expected to experience a range of alternating natural erosion processes.

Quantifying the rate of erosion and peat loss is key to understanding the future of peatlands and to enable practitioners to target restoration efforts and manage anthropogenic pressures. To date there are no published data reporting the rate of erosion in Spanish blanket bogs, and in addition, while livestock have been implicated as being a key driver of the loss of blanket bogs in the Cantabrian Mountains (Heras, 2002; Heras *et al.*, 2017), the contribution of livestock to erosion of blanket bogs in Spain is not known. Therefore, this chapter aims to provide annual and seasonal rates of erosion and peat loss of blanket bog in Spain, enabling the rate of degradation to be placed in context with the rate of degradation of other blanket bogs across Europe, and determine the volume of peat and associated carbon lost through erosion. To achieve these aims, the following objectives were set:

a) Extend the assessment of surface change in three blanket bogs in the Cantabrian Mountains over a period of at least one year.
b) Estimate annual rates of surface change, specifically erosion and peat loss for the blanket bogs examined.
c) Assess seasonal variation in surface changes between spring – summer and autumn – winter periods.
d) Compare surface changes between restored (Basque Country) and unrestored (Cantabria and Castilla y León) blanket bogs, and determine the rate of erosion driven by natural processes and the influence of livestock.
e) Quantify the loss of peat from each site.

f) Quantify annual carbon loss from each site and the contribution of livestock.

This chapter has been partially published in one peer review publication; Chico et al., (2019*) Application of Terrestrial Laser Scanning to quantify surface changes in restored and degraded blanket bogs*. Mires and Peat, (24) 14, 1-24.

6.2. METHODS

6.2.1. Study areas

Assessment of seasonal and annual surface change was undertaken over the period May 2017 to June 2018 at the three blanket bogs examined in chapter 5. These sites allowed further comparison between a restored and unrestored blanket bog at Zalama and Ilsos de Zalama located only 500 m apart, and comparison between these two blanket bogs in the Ordunte Sector and the unrestored and unprotected Collado de Hornaza blanket bog in the Cantabrian Sector defined in chapter 3 (Figure 5.5). All sites have clear examples of Type 1 erosion features mainly due to wind erosion (Bower, 1960), although in some cases these features form the peatland margin due to the small dimensions of the peatland overlapping with Type 2 erosion – Marginal face development (Bower, 1960). Climate data were obtained from WorldClim models (Hijmans *et al.*, 2005) and indicate that mean air temperature (ranging from 7.0°C to 7.4°C) and mean wind speed (between 13.5 km/h – 14.1 km/h) for the period of assessment were comparable across all sites (Table 6.1). Interestingly the strongest wind direction for Zalama and Ilsos de Zalama was suggested to have come from a SW direction, while at Collado de Hornaza the strongest winds came from the NW (Table 6.1). Despite the fact that all sites are located at a comparable altitude (1,280 m to 1,330 m), the total precipitation (5933.1 mm) and snowfall accumulation (1665.4 mm) modelled for Collado de Hornaza were both more than double the amount modelled for Zalama and Ilsos de Zalama (Table 6.1). The seasonal report for climatic variables in Spain highlights that in the north of Cantabria the winter between 2017 and 2018 was extremely humid (AEMET, 2018), but the models are susceptible to errors particularly on high altitude ridges (Meteoblue, 2017). From May 2017 to October 2017 livestock were able to access Ilsos de Zalama and Collado de Hornaza; however, from October 2017 to late April 2018, livestock were not at either site due to adverse weather conditions during this period (Hernández, personal communication, 2017).

Table 6.1. Climatic conditions across each study area within the Cantabrian Mountains.

Variable	*Zalama*	*Ilsos de Zalama*	*Collado de Hornaza*
Temperature (°C)			
Mean	7	7.3	7.4
Minimum	-12.7	-12.4	-15.7
Maximum	29.6	29.8	29.5
Precipitation (mm)			
Accumulation	2265.8	2265.8	5933.1*
Snowfall (cm)			
Accumulation	721.9	721.9	1665.4*
Wind speed (km/h)			
Mean	13.5	13.5	14.1
Maximum	63.5	63.5	63.9
Dominant wind direction	N / WSW	N / WSW	NW - NNW
Strongest wind direction	SW	SW	NW
Altitude (masl)	1,330	1,280	1,280

6.2.2. Materials and experiment design

The protocol developed in chapter 5 to assess rates of erosion using TLS was continued to analyse the annual and seasonal surface changes. In addition to the scans from May 2017 and July 2017, three further scans were collected at Zalama and Ilsos de Zalama in October 2017, April 2018 and June 2018 and at Collado de Hornaza in October 2017 and June 2018 (Table 6.2). Access to Collado de Hornaza was not possible in April 2018 due to adverse weather (snowfall).

Table 6.2. Data collection dates for all sites. Data not available for Collado de Hornaza due to adverse weather conditions.

	Zalama	*Ilsos de Zalama*	*Collado de Hornaza*
Scan	*Day of data collection*		
May 2017	22/05/2017	23/05/2017	24/05/2017
July 2017	16/07/2017	16/07/2017	26/07/2017
October 2017	23/10/2017	23/10/2017	24/10/2017
April 2018	06/04/2018	06/04/2018	No data available
June 2018	15/06/2018	15/06/2018	07/06/2018

6.2.3. Determining annual and seasonal surface changes

Scan data were processed using FARO SCENE version 7.1.1.81 to undertake point cloud colorisation, filtering and scan alignment (see section 5.2.2.4). For each scan the fixed markers were aligned to the location of the fixed markers in the previous scan. The scan data were clipped to the same 30 AOIs examined in chapter 5 (8 at Zalama, 10 at Ilsos de Zalama and 12 at Collado de Hornaza; Figure 5.7) to extend the previous analysis. Surface change between scans was determined using the M2C algorithm and comparing the later point cloud with a mesh for the previous scan. To derive an estimate of annual change (380-390 days; Table 6.3), the scans from June 2018 were compared directly with the scans from May 2017 at each site. For seasonal changes, differences were determined between scans for four periods: May 2017 – July 2017, July 2017 – October 2017, October 2017 – April 2018 and April 2018 – June 2018 (Table 6.3). Difference data were standardised to a monthly difference based on the total number of days for comparison between periods. Owing to the lack of scan data for April 2018 at Collado de Hornaza, only three periods were assessed with the latter extending from October 2017 to June 2018.

Table 6.3. Total number of days for each period and total for all year. * For Collado de Hornaza, the last period assessed was from October 2017 to July 2018.

	Zalama	*Ilsos de Zalama*	*Collado de Hornaza*
Period		*Number of days*	
May 2017 – July 2017	56	55	64
July 2017 – October 2017	99	99	90
November 2017 – April 2018	165	165	No data available
April 2018 – June 2018	70	70	226*
Total days	390	389	380

6.2.4. Statistical analysis

The distribution of the surface change point cloud data between May 2017 and June 2018 were assessed using the Anderson-Darling test. As the data were not normally distributed ($p < 0.001$) any potential difference in the annual rate of surface change between all sites was assessed first using a Kruskal-Wallis rank sum test, and then using a Pairwise Wilcox test to identify individual differences. All statistical analyses were undertaken in R version 3.6.2.

6.2.5. Error assessment

Instrumental error was taken to be the manufacturer specification (±2 mm; FARO technologies Inc., 2015) following assessment of error during method development (see section 5.3.1.1). The methodological error generated from the alignment of scans at each site was determined for each period and the maximum error of scan alignment by site was taken as the methodological error for annual surface change. Instrumental and methodological errors were combined to produce the cumulative error for each site.

6.2.6. Total peat and carbon loss

The annual surface change for each peatland was standardised by unit area (per m^2) and multiplied by the total area of exposed peat in each study area (see section 4.3.2.1) to determine the total annual peat loss for each peatland assuming that the surface change was consistent across the site. The carbon content determined for the peat at each site (see section 4.3.1.3) was then used to estimate the total annual carbon loss for each peatland.

6.3. RESULTS

All scans collected between October 2017 and June 2018 showed a far larger variation in point density across the combined AOIs at each site than the variation in point density found between May 2017 and July 2017. This is particularly evident in the scans from April 2018 at Zalama and Ilsos de Zalama, where the point density was 7.6% and 18.9% lower than the point density in either May 2017 or July 2017 respectively (Table 6.4). During the field campaign in April 2018 areas of surface water lay in some of the AOIs and as the laser beam in the FARO X330 operates at a wavelength of 1550 nm, this is absorbed by water leaving 'holes' in the data. This would reduce the number of points returned, thereby reducing the number of faces generated in a mesh and also reducing the overall point density. Although no scan was taken at Collado de Hornaza in April 2018, the point density in the scan from June 2018 is 12 % lower than the point density achieved in May 2017 or July 2017. The fact that a greater reduction in point density was observed at both the unprotected sites could mean that morphological change in these two sites was greater than at Zalama, and that some areas of the AOI have been obscured in later scans, by peat that had been deposited in the AOI.

Table 6.4. Survey areas, total number of points, point densities and mesh resolutions in each study area for all periods.

Site	*Zalama*	*Ilsos de Zalama*	*Collado de Hornaza*
Total survey surface area (m^2)	11.7	26.88	91.37
Number of points covering survey area			
May 2017	1,915,269	14,925,396	25,710,537
July 2017	1,903,390	14,962,164	24,556,916
October 2017	1,791,001	14,362,573	23,819,234
April 2018	1,780,456	12,582,787	
June 2018	1,843,216	13,838,699	22,961,094
Mesh resolution (Number of faces)			
May 2017	5,501,743	9,320,604	14,231,857
July 2017	5,569,347	9,516,194	14,010,549
October 2017	5,635,281	9,061,787	14,174,700
April 2018	3,887,797	8,003,120	
Mean point density (pts m^{-2})			
May 2017	163,698	555,260	281,389
July 2017	162,683	556,628	268,763
October 2017	153,077	534,322	260,690
April 2018	152,176	468,110	
June 2018	157,540	514,833	251,298

6.3.1. Error assessment

The maximum error of scan alignment at Ilsos de Zalama and Collado de Hornaza was determined in the first period (May 2017 - July 2017; ± 5.4 mm and ± 2.65 mm respectively), and progressively improved in each subsequent period (Table 6.5). Interestingly at Zalama, where the error of scan alignment was consistently lower than at the other two sites, the greatest error was determined in the period October 2017 - April 2018 (± 1.08 mm; Table 6.5).

The combined error (instrumental and methodological) margin determined for estimates of annual change range from ± 3.08 mm at Zalama to ± 7.40 mm at Ilsos de Zalama (Table 6.6).

Table 6.5. Methodological error per site for each period derived from the multi-temporal point cloud alignment. On green the methodological error considered for the annual rates.

Site		*Zalama*	*Ilsos de Zalama*	*Collado de Hornaza*
May 2017 – July 2017				
	Mean (mm)	0.69	5.4	2.65
	SD (mm)	0.08	0.76	1.41
	Min (mm)	0.63	4.78	0.66
	Max (mm)	0.8	6.47	3.66
July 2017 – October 2017				
	Mean (mm)	0.73	3.35	1.51
	SD (mm)	0.3	0.22	0.9
	Min (mm)	0.33	3.07	0.7
	Max (mm)	1.05	3.6	3.49
October 2017 – April 2018				
	Mean (mm)	1.08	2.79	
	SD (mm)	0.46	1.48	
	Min (mm)	0.44	3.7	
	Max (mm)	1.45	3.95	
April 2018 – July 2018				
	Mean (mm)	0.13	4	0.53
	SD (mm)	0.05	1.31	0.32
	Min (mm)	0.06	2.84	0.07
	Max (mm)	0.19	5.83	0.76

Table 6.6. Combined instrumental and methodological error for estimates of annual change determined at each site.

Site	*Zalama*	*Ilsos de Zalama*	*Collado de Hornaza*
Total error (mm)	± 3.08	± 7.40	± 4.65

6.3.2. Determining surface differences and morphological changes

6.3.2.1. Annual surface changes

6.3.2.1.1. Peatland surface change by study area

The overall surface change determined for all AOI combined at all sites between May 2017 and July 2018 was negative (Figure 6.1; Table 6.7), with a markedly higher proportion (60 to 76%) of the change measurements (i.e. points) showing negative change (Table 6.7). The total negative change (i.e. just erosion and peat loss) was determined to be highest at Collado de Hornaza (-50.13 mm), 2.2 times higher than the negative change determined at Zalama (-22.84 mm; Table 6.7). Interestingly the highest total positive change (deposition) was determined at Ilsos de Zalama, 2.7 times higher than the positive change determined at Zalama (Table 6.7). This positive change at Ilsos de Zalama impacted on the overall change, which if assessed alone suggests that, the scale of surface change occurred at Ilsos de Zalama (-7.73 mm) was lower than at the protected and restored Zalama (-14.76 mm; Table 6.7).

Table 6.7. Annual surface change by study area between May 2017 and July 2018.

Site		*Mean (mm)*	*SD (mm)*	*Proportion of point data (%)*
Zalama				
	Overall	-14.76	25.24	100
	Erosion	-22.84	22.88	75.89
	Deposition	10.70	11.86	24.11
Ilsos de Zalama				
	Overall	-7.73	38.31	100
	Erosion	-31.28	26.34	60.51
	Deposition	28.37	22.35	39.49
Collado de Hornaza				
	Overall	-30.69	54.62	100
	Erosion	-50.13	49.74	72.96
	Deposition	21.75	24.27	27.04

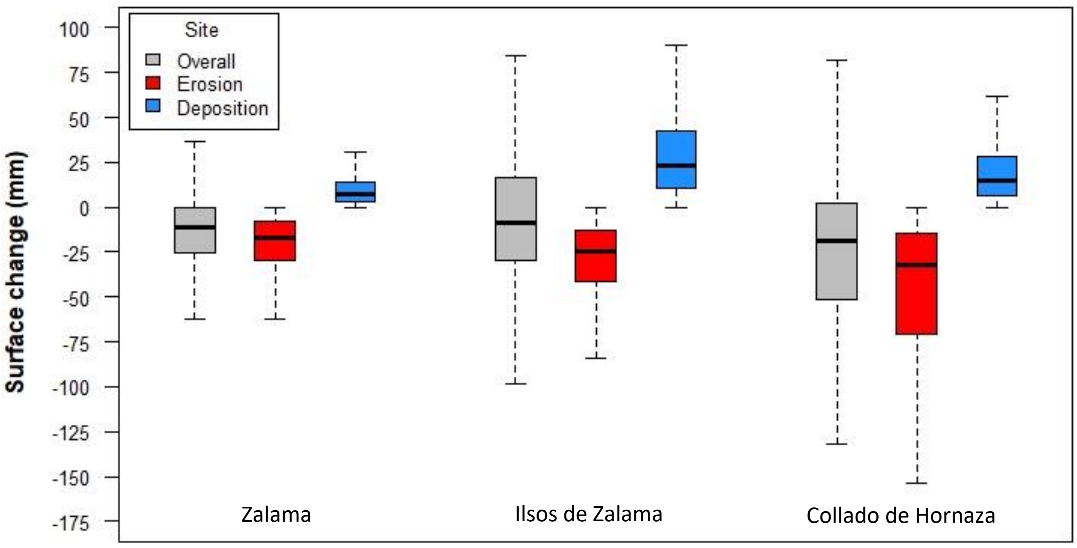


Figure 6.1. Annual surface changes at Zalama, Ilsos de Zalama and Collado de Hornaza.

Examples of high positive and negative change were visible in several AOIs at Ilsos de Zalama (Figure 6.2) and Collado de Hornaza (Figure 6.3), where the majority of negative values appear on sloping peat faces with positive values at the base. Marks left by hooves are common in the sloping faces (e.g. Figure 6.2).

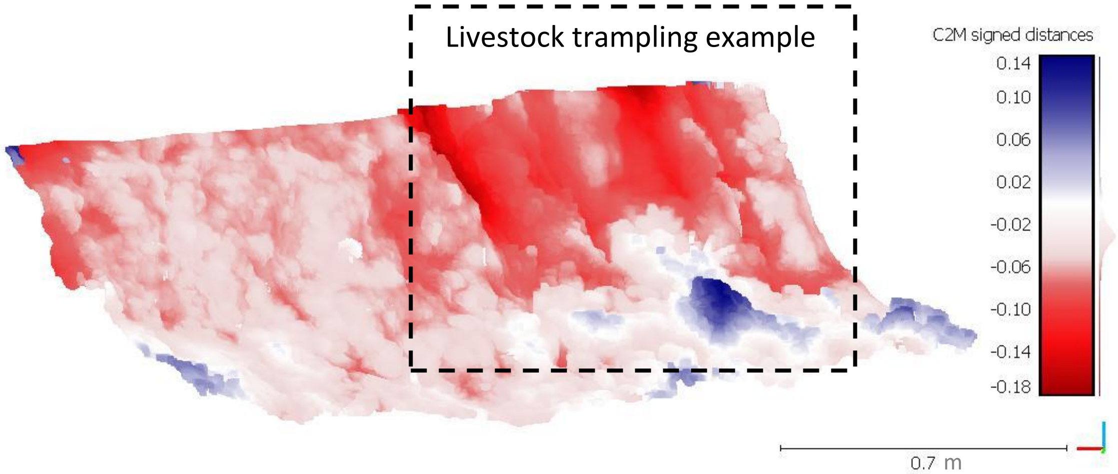


Figure 6.2. AOI 2 at Ilsos de Zalama with clear evidences of livestock trampling on the peat face (red) and accumulation on the bottom of the site (blue).

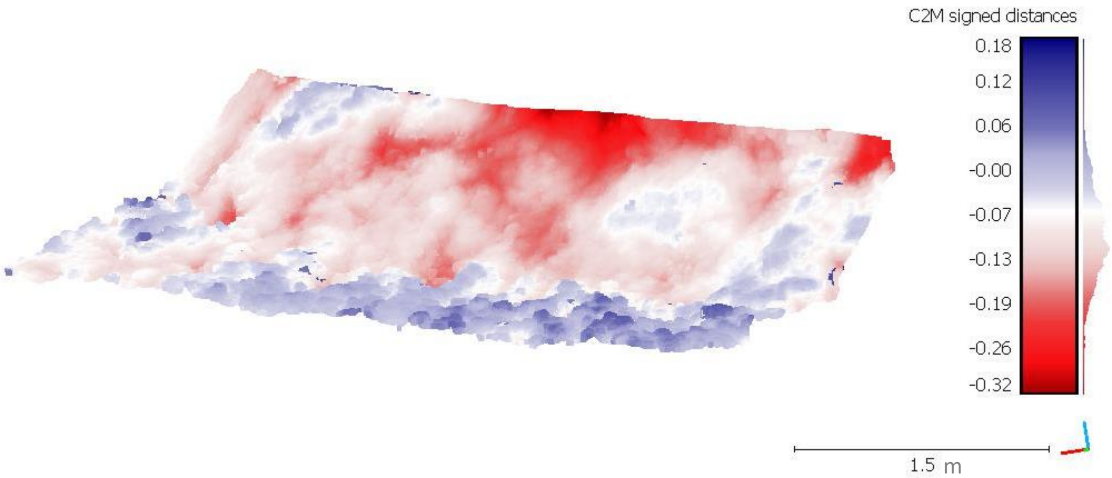


Figure 6.3. Examples of peat erosion/loss (red) and peat deposition (blue) at Collado de Hornaza.

A Krustal-Wallis rank sum test identified a significant difference between the overall surface changes (chi square = 1357654, df = 2, $p < 0.001$), the erosion surface changes (chi square = 910587, df = 2, $p < 0.001$) and deposition surface changes (chi square = 656252, df = 2, $p < 0.001$) across the sites. A Pairwise Wilcox test identified that overall change, erosion and deposition were significantly different at all three sites (Table 6.8).

Table 6.8. p value between sites for overall, erosion and deposition changes.

	Overall	
	Zalama	*Ilsos de Zalama*
Ilsos de Zalama	< 0.001	-
Collado de Hornaza	< 0.001	< 0.001
	Erosion	
	Zalama	*Ilsos de Zalama*
Ilsos de Zalama	< 0.001	-
Collado de Hornaza	< 0.001	< 0.001
	Deposition	
	Zalama	*Ilsos de Zalama*
Ilsos de Zalama	< 0.001	-
Collado de Hornaza	< 0.001	< 0.001

6.3.2.1.2. Peatland surface change by AOI

A trend in the overall annual surface change across AOI was comparable to the trend in overall surface change determined across AOI between May 2017 and July 2017, with a notably lower range of values at Zalama than at Ilsos de Zalama and Collado de Hornaza (Figure 6.4). The mean surface change for AOI at Zalama ranged from around -1.04 to -26.18, while at Ilsos two AOI showed overall change of more than -40 mm (ILZ5 and ILZ7)) and at Collado de Hornaza four AOI showed overall change of more than -40 mm (CH7, CH8, CH9 and CH12; Figure 6.4).

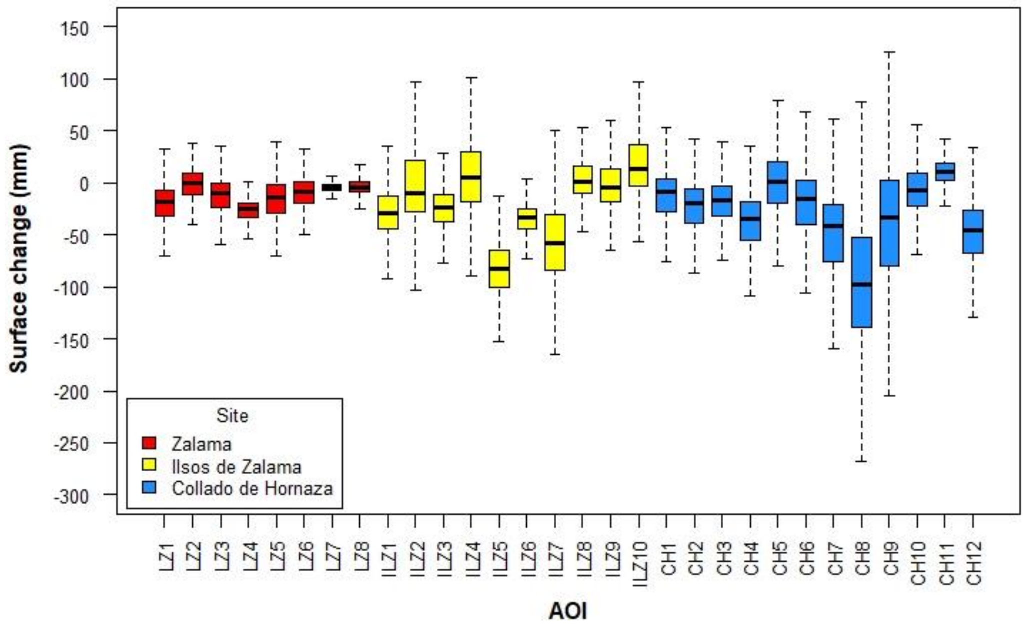


Figure 6.4. Overall surface change by AOI at Zalama, Ilsos de Zalama and Collado de Hornaza.

6.3.2.2. Seasonal surface changes

At Zalama and Ilsos de Zalama, the overall rate of change in autumn - winter (October 2017 – April 2018) was lower than for any of the other three periods assessed (0.56 mm and 0.03 mm respectively; Table 6.9; Figure 6.5; Figure 6.6), and results from lower rates of both erosion and deposition in this period. Comparing the results from Collado de Hornaza was not straightforward due to the lack of a scan in April 2018. However, the rate of overall change, erosion and deposition at Collado de Hornaza from October 2017 to July 2018 were notably lower than the rate of change in the other two periods assessed (Table 6.9; Figure 6.7). The rate of erosion at Zalama varied by less than 3 mm from -1.78 mm to -4.54 mm, but the variation at the unprotected sites was > 10 mm ranging from -1.55 mm to -12.50 mm at Ilsos de Zalama and from -2.43 mm to -16.8 mm at Collado de Hornaza (Table 6.9). It was also notable that the highest rates of erosion at Ilsos de Zalama and Collado de Hornaza were determined between May and October when livestock were present (for the periods May 2017 to July 2017 and July 2017 to October 2017; Table 6.9).

Table 6.9. Monthly surface change by periods for each study site.

Site	*Zalama*		*Ilsos de Zalama*		*Collado de Hornaza*	
	Mean	*% of data*	*Mean*	*% of data*	*Mean*	*% of data*
May 2017 – July 2017						
Overall (mm)	*-1.51*	*100*	*-3.71*	*100*	*-9.39*	*100*
Erosion (mm)	*-3.14*	*71.31*	*-12.50*	*63.22*	*-16.80*	*70.08*
Deposition (mm)	*2.55*	*28.69*	*11.40*	*36.78*	*7.98*	*29.92*
July 2017 – October 2017						
Overall (mm)	*-2.62*	*100*	*-1.72*	*100*	*-2.18*	*100*
Erosion (mm)	*-4.54*	*74.70*	*-4.34*	*67.21*	*-6.75*	*63.50*
Deposition (mm)	*3.04*	*25.30*	*3.66*	*32.79*	*5.77*	*36.50*
October 2017 – April 2018						
Overall (mm)	*0.56*	*100*	*0.03*	*100*		
Erosion (mm)	*-1.78*	*33.59*	*-1.55*	*50.04*		
Deposition (mm)	*1.74*	*66.41*	*1.62*	*49.96*		
April 2018 – June 2018					Oct. 2017 - June 2018	
Overall (mm)	*-2.12*	*100*	*3.35*	*100*	*-0.22*	
Erosion (mm)	*-3.80*	*72.44*	*-4.22*	*50.71*	*-2.43*	*54.79*
Deposition (mm)	*2.29*	*27.56*	*11.14*	*49.29*	*2.46*	*45.21*

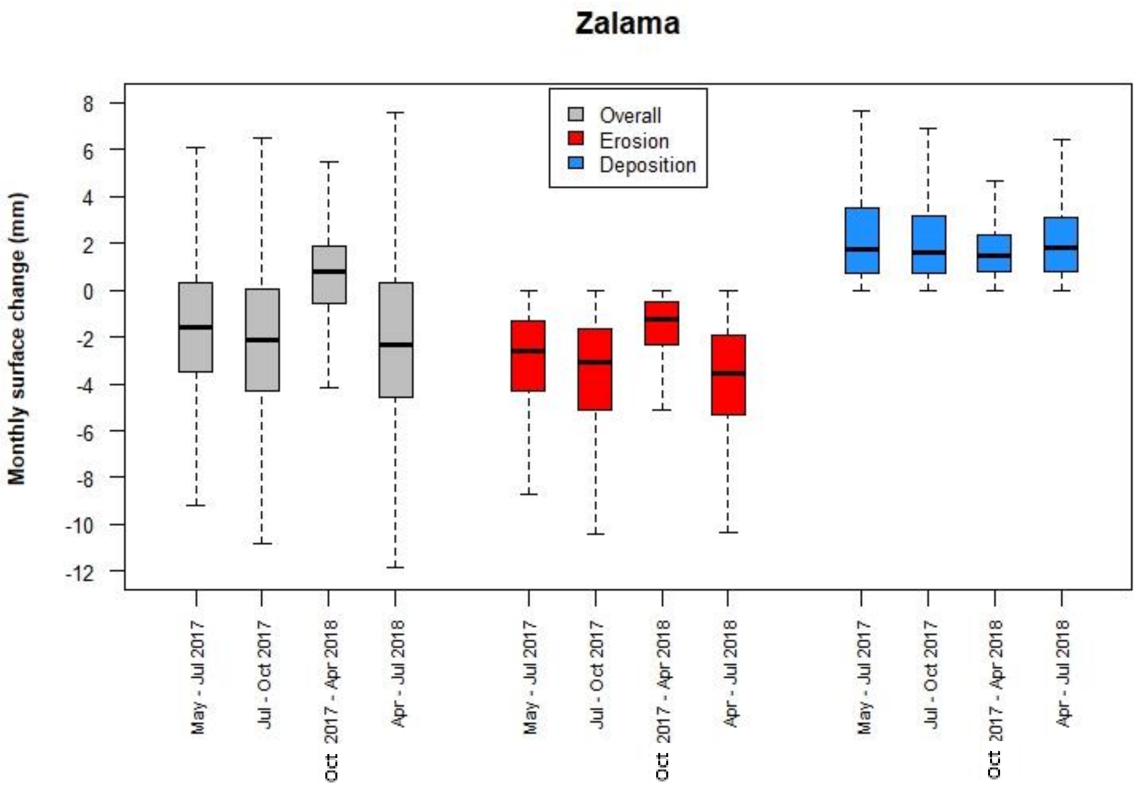


Figure 6.5. Monthly surface changes at Zalama blanket bog between May – July 2017, July – October 2017, October 2017 – April 2018 and April 2018 – June 2018.

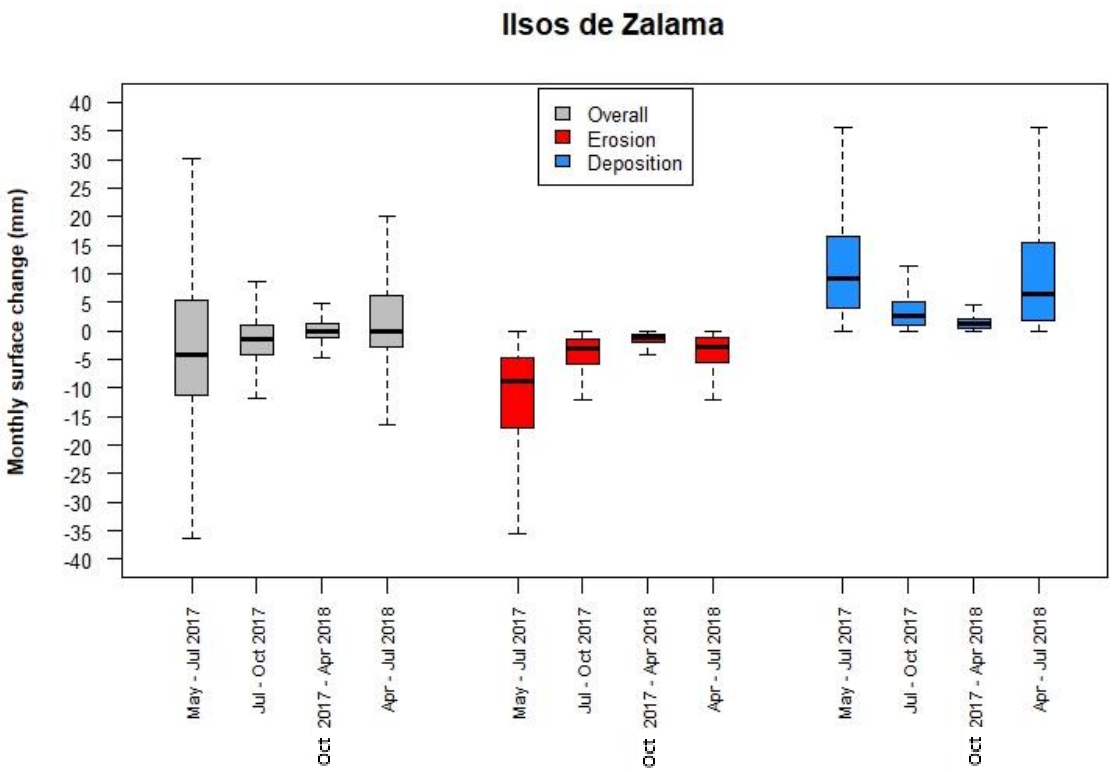


Figure 6.6. Monthly surface changes at Ilsos de Zalama blanket bog between May – July 2017, July – October 2017, October 2017 – April 2018 and April 2018 – June 2018.

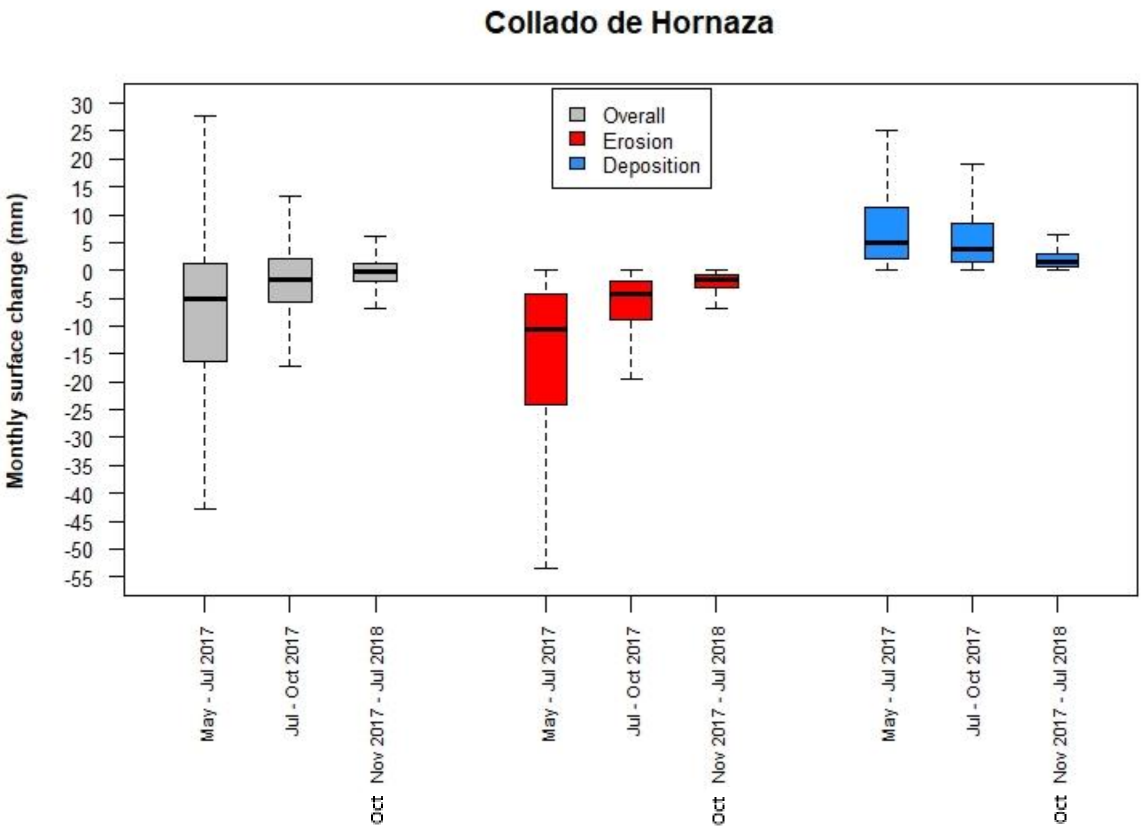


Figure 6.7. Monthly surface changes at Collado de Hornaza blanket bog between May – July 2017, July – October 2017 and October 2017 – June 2018.

The scan data from April 2018 at Ilsos de Zalama and Zalama provide greater ability to assess seasonal change including periods with and without livestock. At both sites the rate of erosion and deposition was higher for the period April 2018 to June 2018 compared to the rates in October 2017 to April 2018 (Table 6.9; Figure 6.5; Figure 6.6). However, while at Zalama the rate of erosion and deposition in the period April 2018 to June 2018 was 2.1 and 1.3 times higher than the period October 2017 to April 2018 respectively, at Ilsos de Zalama erosion was 2.7 times greater and deposition was 6.9 times higher (Table 6.9). It was also apparent that notably higher rates of deposition were determined at Ilsos de Zalama in the periods May 2017 to July 2017 and April 2018 to June 2018 in comparison with Zalama (Table 6.9).

6.3.3. Total peat and carbon loss

The annual rate of change of peat per unit area is directly proportional to the rate of erosion. From the rate of peat loss at each site it was possible to estimate that the lowest rate of carbon loss per unit area was occurring at Zalama 1.70 kg C m^2 yr^{-1}; Table 6.10). More than twice (2.3 times) the amount of carbon per unit area was lost from Collado de Hornaza (3.84 kg C m^2 yr^{-1}) and 1.3 times more carbon was lost from Ilsos de Zalama (2.27 kg C m^2 yr^{-1}; Table 6.10).

As there was a greater area of exposed peat at Zalama, there was currently more long-term carbon being lost from Zalama (-3.20 t C yr^{-1}) than from Ilsos de Zalama and Collado de Hornaza (-1.70 and -2.73 t C yr^{-1} respectively; Table 6.10). However, based on the area of exposed peat at Zalama mapped prior to restoration (8,267.2 m^2; Table 4.12), the annual loss of carbon has reduced by 5.1 times from 16.20 t C yr^{-1} to 3.20 t C yr^{-1}.

Table 6.10. Total annual peat and carbon loss by site.

Site	*Zalama*	*Ilsos de Zalama*	*Collado de Hornaza*
Area of combined AOI (m^2)	11.70	26.88	91.37
Annual rate of erosion (m yr^{-1})	-0.023	-0.031	-0.050
Error in rate of erosion (m yr^{-1})	±0.003	±0.007	±0.005
Annual volume of peat change (m^3 yr^{-1})	-0.267	-0.841	-4.580
Annual rate of change of peat (m^3 of peat per m^2 yr^{-1})	-0.023	-0.031	-0.050
Peat loss			
Total area of exposed peat (m^2)	1632.9	744.1	709.2
Annual peat loss (m^3 yr^{-1}) ± error (m^3)	-37.30 ± 5.03	-23.28 ± 5.51	-35.55 ± 3.30
Carbon loss			
Carbon content of peat (kg/m^3)	85.76	73.21	76.7
Annual rate of carbon loss (kg of carbon per m^2 yr^{-1})	-1.70	-2.27	-3.84
Annual carbon loss from peatland (t yr^{-1}) ± error (t)	-3.20 ± 0.43	-1.70 ± 0.40	-2.73 ± 0.25

6.4. DISCUSSION

The protocol for assessing erosion in peatlands using TLS developed in chapter 5 produced mm resolution surface morphology data, and the single scan strategy enabled assessment of change over three months with a margin of error 3 – 4 times lower than the mean erosion and mean deposition change determined (section 5.3.2.1). It is particularly noteworthy that this level of error was not only maintained, but for Ilsos de Zalama and Collado de Hornaza reduced, while aligning four additional scans collected over the period of a year (May 2017 to June 2018; combined error ranging from ± 3.08 mm at Zalama to ± 7.40 mm at Ilsos de Zalama; Table 6.6). This level of accuracy indicates a high level of confidence in the estimations of erosion across the sites examined, and this research has now provided the first annual estimates of erosion for blanket bogs in Spain.

The annual rate of peat erosion determined at Zalama (-22.54 mm yr^{-1}; Table 6.7) was comparable to annual rates of peat erosion in England (-22.4 mm yr^{-1}; Table 5.1) and Wales (-23.1 mm yr^{-1}; Table 5.1), while the annual rate of erosion and peat loss at Ilsos de Zalama (-31.28 mm yr^{-1}; Table 6.7) was more comparable to annual rates of erosion in Scotland (-36.3 mm yr^{-1}; Table 5.1). However, the annual rate of erosion and peat loss at Collado de Hornaza were much larger (-50.13 mm yr^{-1}; Table 6.7) and closer to the annual rate of erosion and peat loss estimated in a peat margin at Holme Moss in England (-73.8 mm yr^{-1}; Phillips, Tallis and Yalden, 1981).

6.4.1. Landscape and environmental influences

At landscape scale, there are some notable differences between the study areas selected that may explain the higher rates of erosion and peat loss determined at Collado de Hornaza in comparison with Zalama and Ilsos de Zalama. Firstly, the areas of exposed peat assessed at Zalama and Ilsos de Zalama are located in the central area of both peatlands, whereas at Collado de Hornaza, the area of exposed peat assessed is on the current margin of the peatland. As these areas are characterised by different types of erosion feature (Bower, 1960), this may explain why the rate of erosion and peat loss determined at Collado de Hornaza was significantly higher (Table 6.8) than the rate of erosion and peat loss determined at both Ilsos de Zalama and Zalama. However, this does not explain why the rate of erosion and peat loss determined at Ilsos de Zalama was significantly higher than the rate of erosion determined at Zalama.

The slope of the land can have a strong influence on erosion and revegetation processes, as mass movements of peat are more frequent on higher angle sloping peatlands (Bower, 1960), and revegetation of exposed peat surfaces has been reported to be more successful on slopes up to 11° (McHugh, 2000). Despite the fact that at all sites the AOI examined covered a range of near vertical peat faces and low-angle sloping areas (Appendix C), the mean slope of the peatland at Collado de Hornaza (17° ± 8.1; Table 3.6) was notably higher than the mean slope at Ilsos de Zalama (11.6° ± 8.4; Table 3.6) and at Zalama (14.5° ± 8.1; Table 3.6). This difference in slope may explain the lower overall annual erosion rates and peat loss at Ilsos de Zalama and Zalama, as a consequence of greater stability in the exposed peat surfaces.

Aspect may also be important as the maximum peat depth at all 15 blanket bogs assessed in this research was measured on N-NW facing slopes (Table 3.6) demonstrating a clear influence of the direction of precipitation on peat formation (see section 3.4.1). However, the dominant peatland body aspect of Ilsos de Zalama faces east, while both Collado de Hornaza and Zalama predominantly face north (Table 3.6). It may therefore be necessary to assess the position of the blanket bogs within the landscape to include both topographical and hydrological influences. In terms of hydrological influences, Ilsos de Zalama is a saddle mire, and erosion was assessed near the centre of the mesotope. At Zalama erosion was assessed between a watershed and a saddle mire mesotope, and at Collado de Hornaza erosion was assessed on the margin of a watershed mesotope. As there are two inputs of water flow in addition to direct precipitation on the saddle mire mesotope, this might explain why the rate of deposition determined across the AOI assessed at Ilsos de Zalama between May 2017 – July 2017 and April 2018 – July 2018 were higher than the rate of deposition across the AOI assessed at Collado de Hornaza and Zalama for the same time periods. It does not explain though why this difference was not observed in the period July 2017 – October 2017, and there appears to be a strong seasonal influence on the rate of surface change.

A limitation to the analysis in this research was the availability of climatic data. The data sourced for the Cantabrian Mountains are modelled from meteorological stations, but there are no official meteorological stations within 44 km of the study sites and the closest station is located at 42 masl. Modelled temperature data appeared to be similar to the values reported at Zalama in other research (Heras, 2002) and modelled rainfall for the period assessed (May 2017 – June 2018) at Zalama and Ilsos de Zalama are consistent with data provided by the government (AEMET, 2018). However, the total precipitation (5933.1 mm; Table 6.1) and snowfall accumulation (1665.4 mm; Table 6.1) modelled at Collado de Hornaza for the meteorological winter 2017-2018 appeared to be an overestimation despite the wetter winter reported (AEMET, 2018). It is likely that Zalama and Ilsos de Zalama experience very similar weather conditions due to their proximity (<500 m), so in the absence of 'real' climate data, a greater confidence can be placed on evaluating the impact of climatic variation on changes at these two sites. If modelled wind data are correct though, there is notable difference in wind direction across the sites, where the strongest wind for Zalama and Ilsos de Zalama was suggested to have come from a SW direction, while at Collado de Hornaza the strongest winds came from the NW (Table 6.1). As the area of exposed peat selected at each site was oriented in an approximately N-NW aspect (section 5.2.2.2), this may provide additional reason for why the highest rates of erosion were determined at Collado de Hornaza (Foulds and Warburton, 2007a).

A consistent change in the scale of surface change was evident across all sites, where the lowest rates of both erosion and deposition were determined over the winter months (from October 2017 to April 2018 for Zalama and Ilsos de Zalama, and from October 2017 to June 2018 at Collado de Hornaza (owing to the lack of a scan in April 2018); Table 6.9). The reduction in surface change over this period is most likely to be a consequence of the peat surfaces being covered by snow. As temperatures increase thaw processes can increase the rate of erosion (Li, Holden and Grayson, 2018) and desiccation of peat over summer months can lead to cracks in the peat thereby making the peatlands more susceptible to wind erosion (Evans, 1977) and accelerating the erosion process during wet season (Tallis, 1973; Francis, 1990). A net accumulation of peat is predicted in exposed peat surfaces over winter months and net erosion in summer (Evans, 1974; Evans, 1977). This increase in erosion is perhaps evident in the data between May 2017 and April 2017, where the highest rate of erosion was determined at both Ilsos de Zalama and Collado de Hornaza (Table 6.9). However, if seasonal change and associated 'natural' erosion processes were the primary driver of surface change in the blanket bogs examined, it might be reasonably expected that the rate of erosion at Zalama would be of the same magnitude in the same time period.

While there is some evidence that topographical, hydrological and other climatic factors could explain some of the seasonal variation in surface change observed across all sites, there was a clear impact of an external pressure: livestock. Trampling by livestock disturbs the peat surface and will potentially increase natural erosion processes. This pressure offers the only explanation as to why at Zalama, an area with livestock exclusion, the monthly rate of erosion for the periods assessed varies by less than 3 mm from -1.78 mm to -4.54 mm, while far larger monthly changes (up to -12.5 mm and -16.8 mm) were observed at the two unprotected and grazed sites.

6.4.2. The role of livestock

The presence of sheep has been shown to lead to the expansion of exposed peat surfaces thereby increasing the area that is affected by erosion processes (Evans, 1977; McHugh, 2007). Trampling from hooves increases soil compaction (Phillips, Tallis and Yalden, 1981; Worrall, Armstrong and Adamson, 2007) resulting in reduced soil infiltration rates (Gifford and Hawkins, 1978) and increased rates of runoff from exposed peat surfaces (Evans, 1977). Estimates of erosion resulting from sheep have been reported (e.g. Evans, 1977; McHugh, 2007), but there are currently no estimates of the impact of cattle on erosion from peatlands. Evidence of hooves, striations from horns (Figure 5.14) and livestock paths (with areas of exposed peat) were visible at all blanket bogs identified in this research with exception of Zalama. During field survey campaigns, cows, horses and goats were observed grazing the AOIs at Collado de Hornaza and Ilsos de Zalama with the exception of October 2017 and April 2018. The absence of livestock in the Cantabrian Mountains over winter months therefore allowed assessment of the impact of livestock on erosion from blanket bogs in Spain.

Owing to the lack of scan data from Collado de Hornaza in April 2018, and potential errors in the climate data, the impact of livestock on erosion was assessed using data for Zalama and Ilsos de Zalama. The lowest rates of erosion were determined at both these sites in the period October 2017 to April 2018, and the rate of monthly change was remarkably similar (-1.55 mm at Ilsos de Zalama and -1.78 mm at Zalama; Table 6.9). These figures most likely reflect the rate of erosion resulting almost entirely from natural processes. Examining the monthly rate of erosion at Zalama between May 2017 and July 2017 (-3.14 mm; Table 6.9) and between April 2018 and June 2018 (-3.80 mm; Table 6.9) indicates that 'natural' erosion may increase by 1.4 to 2.0 mm over summer months. The monthly rate of erosion at Ilsos de Zalama between May 2017 and July 2017 (-12.50 mm; Table 6.9) and between April 2018 and June 2018 (-4.22 mm; Table 6.9) demonstrates that with livestock erosion increases by 2.7 to 11.0 mm. The lower increase of 2.7 mm quantified between April 2018 and June 2018 is likely an underestimate, as in both summer periods the monthly rate of deposition was remarkably similar (11.14 to 11.40 mm; Table 6.9). It is therefore possible to determine that livestock in Spain can increase the rate of erosion by at least five times. In addition, large morphological change in peatland surfaces (>±40 mm) have been associated with disturbances from sheep hooves (Evans, 1977), and the fact that the median overall surface change at 8 AOI across Ilsos de Zalama and Collado de Hornaza during the summer was greater than 40 mm, indicates the scale of change caused by cattle (Figure 6.4).

6.4.3. Implications of erosion rates and peat loss on carbon stored

On the assumption that the rate of erosion of peat determined at Zalama represents the rate of erosion driven by natural erosion processes, this research indicates that a minimum of 1.7 kg C m^{-2} yr^{-1} (Table 6.10) were being lost from exposed peat surfaces in blanket bogs in the regions of the Cantabrian Mountains examined. The assessment of surface change at Collado de Hornaza indicated that the rate of carbon loss in some blanket bogs was 2.3 times as high (3.84 kg C m^{-2} yr^{-1}) and while the presence of livestock was implicated to be driving a key part of this increase, teasing out the contribution of landscape and environmental factors from the influence of livestock was problematic. The most confident assessment of the impact of livestock on carbon loss from the blanket bogs based on the comparison of surface change at Zalama and Ilsos de Zalama indicated that the presence of livestock results in an additional 34% of loss (2.27 kg C m^{-2} yr^{-1}). It is worth noting that while this was as an annual estimate, due to weather conditions in the Cantabrian Mountains, livestock typically only have access to the blanket bogs for a period of six months (between May and October). It could be argued therefore, that if livestock were present all year round, an increase in carbon loss from these blanket bogs of over 60% might be observed.

Based on the area of exposed peat mapped from 2017 aerial photography (Figure 4.10; Appendix B), this research estimates that 7.63 ± 1.08 t C were being lost every year from just three blanket bogs in the Cantabrian Mountains (Zalama, Ilsos de Zalama and Collado de Hornaza) assuming that the rate of erosion and peat loss is consistent across exposed peat in each peatland. Over 40% of this carbon loss was occurring at Zalama (-3.2 t C yr^{-1}; Table 6.10) despite restoration interventions. However, given the clear change in vegetation within the fenced area at Zalama (Figure 5.15), and the increased presence of some peat-forming species (e.g. *Eriophorum vaginatum* (Chico and Clutterbuck, 2019), it is possible that some parts of Zalama are in equilibrium or may even be sequestering carbon. Determination of the carbon budget of Zalama is a key area for further research.

6.5. CONCLUSION

Extending the assessment of surface change over a longer period of time than that assessed in chapter 5, has revealed clear seasonal variation in the rate of erosion and deposition in blanket bogs in the Cantabrian Mountains. At all three blanket bogs assessed the rate of erosion and deposition was lowest over autumn-winter months (October to April) and is suggested to be largely influenced by snow cover during these periods. The rate of peat erosion determined at the protected Zalama likely reflects the rate of erosion driven almost entirely by natural processes and varies from 1.78 mm per month over winter to 3.80 mm per month over summer. The annual rate of peat erosion determined at Zalama (-22.54 mm yr^{-1}) is comparable to lower estimates of erosion across the United Kingdom.

At the two unprotected blanket bogs, a far greater range in the rate of erosion (> 10 mm) was determined ranging from -1.55 mm to -12.50 mm per month at Ilsos de Zalama and from -2.43 mm to -16.8 mm per month at Collado de Hornaza. Annual rates of peat erosion were significantly higher ($p < 0.001$) at both the unprotected sites (-31.28 mm yr^{-1} at Ilsos de Zalama and -50.13 mm yr^{-1} at Collado de Hornaza). Although not evident in the annual rates of erosion determined, livestock increase the rate of erosion by at least five times during the months they were present.

The data suggest that a minimum of 1.7 kg C m^{-2} yr^{-1} are being lost from exposed peat surfaces in blanket bogs in the regions of the Cantabrian Mountains examined, and where livestock are present this increases by at least 34% (2.27 kg C m^{-2} yr^{-1}). Carbon loss at Collado de Hornaza may be as high as 3.84 kg C m^{-2} yr^{-1}, but it was not possible to separate the direct influence of livestock on erosion from the influence of other factors at this site.

The loss of carbon from all these sites is concerning, particularly given that Zalama has been restored. It will be important to determine the full carbon budget of Zalama to quantify the impact of restoration and revegetation across the blanket bog.

Chapter 7

The importance of the southernmost blanket bogs in Europe

The protection and restoration of peatlands have international significance as these habitats contain the largest store of terrestrial carbon (Limpens *et al.*, 2008) and have the potential to return to function as carbon sinks when restored (Nugent *et al.*, 2018). Peatland restoration is therefore seen as a key approach to mitigating the impacts of climate change (Joosten, Tapio-Biström and Tol, 2012), and some countries have identified peatland restoration as a key measure to deliver Net Zero GHG emissions targets (e.g. UK Committee on Climate Change, 2020). However, some blanket bogs go unnoticed, meaning this globally important habitat and carbon store may be at threat of loss as there is no protective legislation.

The aim of the research presented in this thesis has been to identify currently unrecognised blanket bogs in part of the Cantabrian Mountains in north Spain where a gap in the peatland inventory was noted to exist. In addition, the research aimed to determine the significance of any blanket bog identified in terms of carbon storage, quantify the rate of peat erosion and assess the drivers of peat loss to study the grade of degradation. The evidence presented in this thesis highlights an urgent need to update the peatland inventory for Spain to enable the protection and restoration of the southernmost blanket bogs in Europe since, all blanket bogs identified in this research, face significant anthropogenic pressures and are eroding at a significantly faster rate than protected areas. A further three blanket bogs identified in the study area, and the associated priority habitat, carbon store and paleaoenvironmental archive, have already been lost (e.g. Cueto de la Avellanosa; Figure 3.9A).

7.1. EUROPEAN BLANKET BOGS

The research presented in this thesis has identified 14 currently unrecorded blanket bogs located in the regions of Cantabria and Castilla y León in the Cantabrian Mountains, providing geo-hydromorphological classification (mesotopes) for these blanket bogs and for the protected Zalama blanket bog on the administrative boundaries of Castilla y León and the Basque Country. The number of blanket bogs and range of mesotope types identified highlights the importance of this region for peat formation, and completes a gap highlighted to exist in the Spanish peatland inventory (Heras *et al.*, 2017; Ramil-Rego *et al.*, 2017). This gap is particularly significant since 12 of these blanket bogs lie partly or entirely within the region of Cantabria, and to date no blanket bog has been recognised in this region. In addition, the 14 newly identified blanket bogs redefine the southernmost edge-of-range of blanket bogs in Europe and would add 10.5% (by area) more to the current inventory of protected blanket bogs in Spain if they were recognised under Natura 2000.

The total area of blanket bogs mapped (>40 cm peat depth) covers 44.5 ha and contains more than 0.5 million m^3 of peat (Table 3.5). Maximum peat depth ranged from 1.61 – 3.78 m (Table 3.5), which is notably lower than the peat depth in other blanket bogs in Spain (e.g. up to 5 m in Galicia, Heras *et al.*, 2017). This research indicates that topography, location (latitude and longitude) and altitude combined with occult precipitation are key factors influencing the development and accumulation of peat in the Cantabrian Mountains. All the blanket bogs examined in this research are located at an altitude of 1,200 to 1,500 masl (Table 3.5), which was higher than any other blanket bog recorded in Spain. As temperature is a critical factor in blanket bog formation (Lindsay, 1995), this would provide a logical explanation for altitudinal constraint. Interestingly, although not significant, the extent and volume of peat accumulated in the peatlands mapped in this research were both negatively correlated with altitude (Table 3.7), suggesting that at higher altitudes, Spanish blanket bogs tend to be smaller. This would also explain why blanket bogs in the Cantabrian Mountains were smaller than examples found at lower altitudes in Galicia, or indeed in Ireland or the United Kingdom (European Environment Agency, 2019).

The Cantabrian Mountains chain typically comprises a series of peaks and ridges interspersed with flatter areas, and although blanket bogs can develop on steep slopes (up to 22°), they become unstable as slope is a key limitation to peat development (Tallis, 1973). The mean slope across all the blanket bogs examined ranged from 11.6° to 18.8°, but within individual blanket bogs, greater accumulations of peat were measured in 'flatter' areas with significant reductions on steeper slopes that define the natural edge of the blanket bogs studied (e.g. Zalama or Motas del Pardo; Figure 3.6). In addition to slope, other geomorphological formations such as rock outcrops and sink holes also appeared to act as the limit of blanket bog development (e.g. Cotero de la Osera).

It was also evident that the aspect of the terrain influences the development of blanket bogs in the Cantabrian Mountains. The maximum peat depth in all 15 blanket bogs was measured on NE – NW facing slopes, most likely as a result of the supply of water from the

Atlantic Ocean being deposited on the north faces of the primary mountain ridge. Interestingly, while the majority of the main peat body for 11 of the blanket bogs lies on north-facing slopes, greater proportions of the peat body can lie on south facing (Cercio and Cantos Calientes; Table 3.6) and east facing (Ilsos de Zalama and Motas del Pardo; Table 3.6) slopes. There was an indication from Cantos Calientes that rock outcrops may be influential in transferring water down south facing slopes, but occult precipitation is common year-round (Heras, 2002) and this source of water is key for blanket bog formation as has been found for blanket bogs in Canada (Price, 1992).

Analysis of peat cores in chapter 4 may provide some indication of a difference in peatland genesis. A notable difference was observed in the BD and carbon content at the base of the core taken from Ilsos de Zalama compared to the BD and carbon content at the base of the other four cores. The blanket bog at Ilsos de Zalama is the only saddle mire where peat was sampled, and the alternating sections of increasing and decreasing BD and carbon content in the last 60 cm (Figure 4.2; Figure 4.4) may indicate episodes of mineral input from the two mountain summits on either side until the central portion of the blanket bog had accumulated sufficient peat to become raised above the directly surrounding area.

This research therefore furthers our understanding of the origins and distribution of blanket bogs in Spain, and the 14 newly identified and classified blanket bogs (Figure 3.5; Figure 3.6; Figure 3.7) clearly warrant inclusion in the Spanish and European peatland inventories. However, designation and protection of these blanket bogs is most urgent as large-scale loss of peat is occurring within the area of the Cantabrian Mountains examined. Commercial peat extraction has already removed three areas of blanket bog (e.g. Figure 3.9), and the increased accessibility provided by windfarm tracks could put the blanket bogs now identified at Malverde, Cantos Calientes and El Cuito at heightened risk. In addition, the windfarm tracks highlight a greater contemporary concern, as windfarm development plans proposed by Cantabria and Castillo y León governments have identified locations for construction that coincide with some of the newly identified blanket bogs (Figure 7.1). Despite growing evidence demonstrating the negative impacts of windfarms on peatland environments (Fraga *et al.*, 2008; Heras and Infante, 2008; Lindsay, 2016c; Ramil-Rego *et al.*, 2017; Wawrzyczek *et al.*, 2018), without designation and protection there is little in the way of statutory or official procedural process to oppose windfarm installations. In 2017 one such new windfarm installation was proposed for construction across Ilsos de Zalama by the Cantabrian Government (Gobierno de Cantabria, 2017). The data mapping the extent of the blanket bog at Ilsos de Zalama collected in this research was instrumental in assembling a case to oppose the installation. Strong opposition from a number of organisations, in particular the Provincial Council of Bizkaia who raised additional concern over the proximity to the protected and restored Zalama blanket bog (500 m), has led to the proposal being withdrawn.

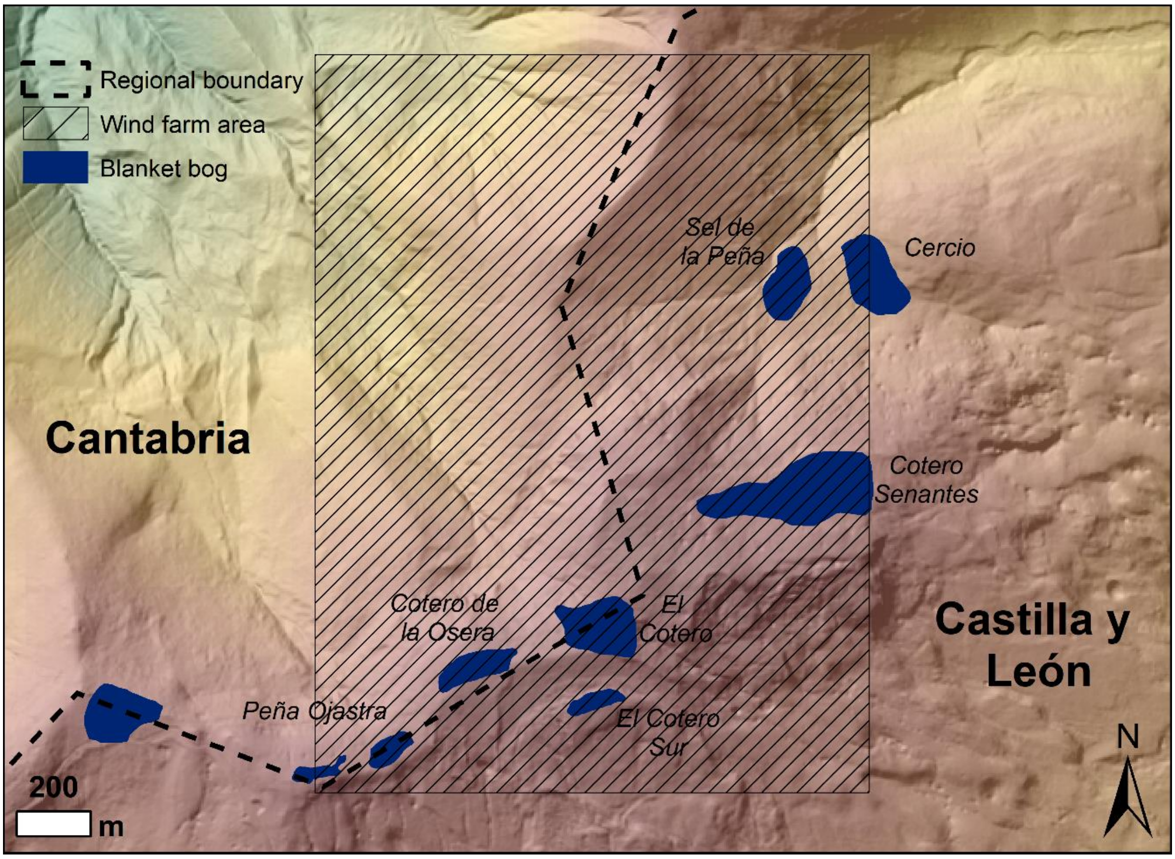


Figure 7.1. Proposed area for windfarm development (BOE, 2015) on the regional boundary between Castilla y León and Cantabria regions in comparison with the newly mapped blanket bogs in this research.

However, recent windfarm installations have already significantly damaged the blanket bog at Malverde (Figure 2.17; Figure 3.8) and the vehicle access tracks at El Cuito and Cantos Calientes are likely influencing the hydrology of the blanket bogs at these two sites (Figure 3.6; Appendix A). A further 8 of the blanket bogs identified in the Cantabria Sector in this research from Peña Ojastra to Cercio are still under threat from a further proposal put forward by the Castilla y León government (Figure 7.1). As the foundations for individual turbines cover a circular area 90 m in diameter (BOE, 2015), some of the smaller blanket bogs, such as Peña Ojastra, could be irreversibly damaged, or at worst, lost. It is worth noting that this issue is not constrained to the regions examined in this thesis, as extensive windfarm networks have also been constructed on some of the best examples of blanket bogs in Galicia (Ramil-Rego *et al.*, 2017), and across Great Britain extensive vehicle track networks (including for windfarm access) exist on blanket bog (Clutterbuck *et al.,* 2020b). The threat and pressures of windfarm infrastructure on blanket bog in Spain needs to be modified to high – important and included in all habitat assessment by the European Environment Agency.

7.2. BLANKET BOGS AND CLIMATE CHANGE

Analysis of peat cores in chapter 4 demonstrated that the mean carbon content of the organic matter in the peat for the five sites sampled (46.9 – 55.2%) was comparable to the carbon content of organic matter in peat determined at other blanket bogs in Galicia (51% (Gómez-orellana *et al.*, 2014); 46% (Ramil-Rego and Aira-Rodríguez, 1994)), and more widely in Scotland (53%; Chapman *et al.*, 2009), Eastern Canada and Western European islands (Loisel *et al.*, 2014). No significant difference was found between the carbon content in the top 1 m of each core, which may suggest that the peat at each site in this research formed from comparable vegetation and is perhaps of comparable age (4,000 years BP determined for the top 1 m at Zalama blanket bog; Pérez-Díaz *et al.*, 2016).

Based on the volume of peat determined in chapter 3 and the carbon content determined for the peat cores in chapter 4, it was possible to estimate that the blanket bogs in this research contain 44.88 kt of C. While this figure is small in comparison to the amount of carbon stored in all peatlands in Spain (5,398 Mt; Joosten, 2009), it is important to note the geographical significance of these blanket bogs. It is also worth highlighting that the 64.65 ha of blanket bog assessed in this research contains the same amount of carbon as an area of rainforest covering 120.34 ha. In addition, this research has not accounted for the carbon stored in areas of shallow peat (<30 cm) or in fen environments. In many cases, individual blanket bog 'units' (delimited by peat depth >30 cm for the margins) were connected to other blanket bogs by a continuous layer of shallow peat (e.g. between Zalama and Ilsos de Zalama; Figure 7.2). This could suggest that the blanket mire landscape in the Cantabrian Mountains was larger in the past. Blanket bogs in Spain developed during the Holocene

(Castillo *et al.*, 2001; Gómez-Orellana *et al.*, 2014; Pérez-Díaz *et al.*, 2016) and the presence of peat-forming species observed at all the blanket bogs in this research, and specifically the increase in peat-forming vegetation recorded at Zalama after restoration (Chico and Clutterbuck, 2019), indicates that climatic conditions may well still be favourable for peat formation. If restoration activities extended beyond the 'margins' of the blanket bogs identified, this could mean a far greater area for potential carbon sequestration would be realised.

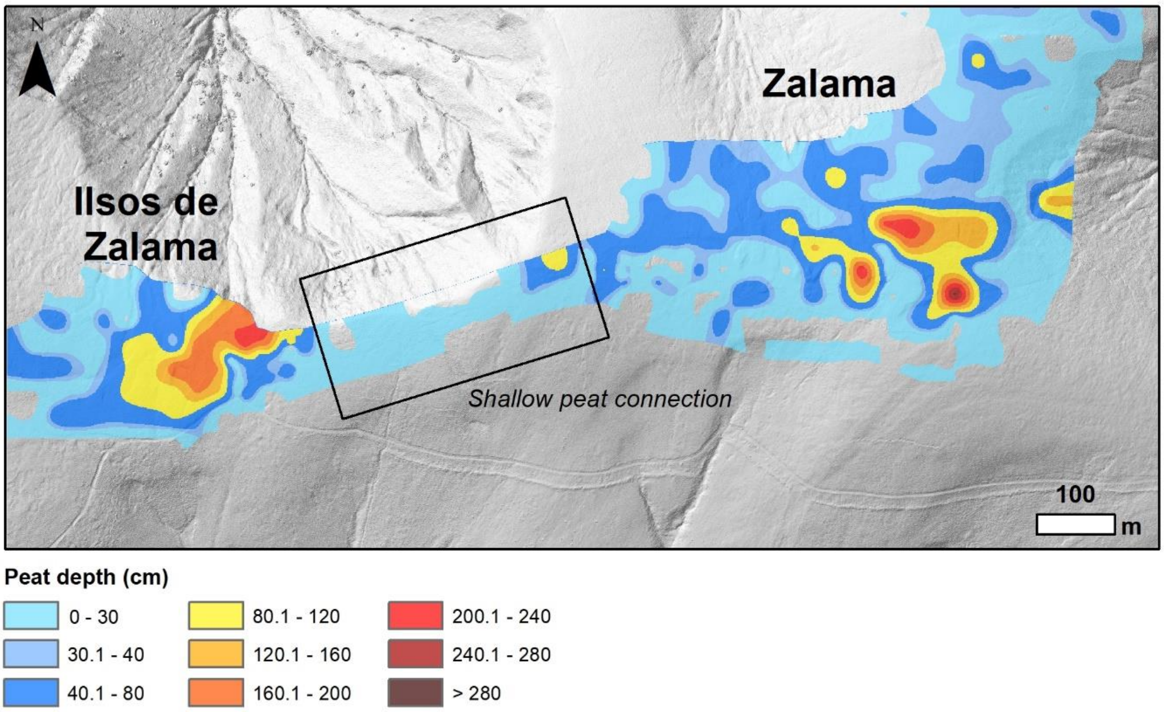


Figure 7.2. Peat depth between Zalama and Ilsos de Zalama, northern Spain.

7.3. DEGRADATION OF BLANKET BOGS

The data presented in chapter 4 indicated that 30.8% of the surface of the 14 blanket bogs identified in this research (peat depth >40 cm) were exposed peat and therefore susceptible to aeolian, fluvial and freeze-thaw erosion processes (Table 4.10; Appendix B). This level of peat exposure was not exceptional as similar values have been reported in Ireland (27 – 33%; Cooper and Loftus, 1998), Wales (30%; Marcus, 1997) and Northern Ireland (29%; Cruickshank and Tomlinson, 1990). However, while restoration at Zalama resulted in a significant decrease (80%; Table 4.12) in the area of exposed peat (2009 to present), an apparent reduction in the area of exposed peat at Ilsos de Zalama (unprotected) was found to arise from total loss of the peat deposit over this same time period (Table 4.12). The presence of livestock was implicated in this loss of peat and therefore as a factor increasing the degradation of blanket bogs in the Cantabrian Mountains.

In the absence of any data reporting the scale of peat erosion in Spain, a method that enabled ultra-high resolution of change using TLS was developed in chapter 5. The protocol adopted a single scan strategy and demonstrated that with the use of fixed reference markers, portable TLS units are able to collect mm resolution data and enable determination of surface changes in peatlands with mm level accuracy. A trial application undertaken on three blanket bogs over a period of two months indicated that compared to the rate of erosion determined at the protected Zalama blanket bog (-5.9 mm; Table 5.10; Figure 5.11), the mean peat loss/erosion in the areas assessed in the unprotected and grazed blanket bogs was 4 – 6 times greater (-22.9 mm at Ilsos de Zalama; -35.8 mm at Collado de Hornaza; Table 5.10; Figure 5.11).

The data presented in chapter 6 provide the first annual rates of peat erosion for blanket bogs in Spain and reveal clear seasonal variation in the rate of erosion and deposition in blanket bogs in the Cantabrian Mountains. At all three blanket bogs assessed, the rate of erosion and deposition was lowest over autumn-winter months (October to April) and is suggested to be largely influenced by snow cover during these periods (Table 6.9). The annual rate of peat erosion determined at Zalama (-22.54 mm yr^{-1}; Table 6.7; Figure 6.1) was comparable to annual rates of peat erosion in England and Wales (Evans, Warburton and Yang, 2006; Li *et al.*, 2018), while the annual rate of erosion and peat loss at Ilsos de Zalama (-31.28 mm yr^{-1}; Table 6.7; Figure 6.1) was more comparable to annual rates of erosion in Scotland (Stott, 1997). However, the annual rate of erosion and peat loss at Collado de Hornaza were much larger (-50.13 mm yr^{-1}; Table 6.7; Figure 6.1) and closer to the annual rate of erosion and peat loss estimated in a peat margin at Holme Moss in England (Phillips, Tallis and Yalden, 1981).

There was evidence that topographical, hydrological and other climatic factors could explain some of the variation in surface change observed across the sites, but the impact of livestock was clear and was quantified at Ilsos de Zalama and Collado de Hornaza. The rate of erosion assumed to be driven almost entirely by natural processes at Zalama varied from 1.78 mm per month over winter to 3.80 mm per month over summer (Table 6.9). The rate of erosion at Ilsos de Zalama over winter was comparable to the rate at Zalama (1.55 mm per month; Table 6.9), but over summer, when livestock were present, the rate of erosion was over 5 times higher (up to -12.50 mm per month; Table 6.9).

The data presented in this thesis suggest that a minimum of 1.7 kg C m^{-2} yr^{-1} were being lost from exposed peat surfaces in blanket bogs in the regions of the Cantabrian Mountains examined, and where livestock were present this increased by at least 34% (2.27 kg C m^{-2} yr^{-1}). Carbon loss at Collado de Hornaza may be as high as 3.84 kg C m^{-2} yr^{-1}, but it was not possible to separate the direct influence of livestock on erosion from the influence of other factors at this site.

This research estimates that 7.63 ± 1.08 t C were being lost every year from just three blanket bogs in the Cantabrian Mountains (Zalama, Ilsos de Zalama and Collado de Hornaza; Table 6.10). If the further 10,629.10 m^2 (Table 4.10) of exposed peat at the other 11 unprotected blanket bogs examined were losing carbon at the same rate as Ilsos de Zalama, a further 24.13 ± 5.47 t C may be being lost. It is therefore likely that more than 30 t C are being lost every year from the 15 blanket bogs examined in this research, excluding any additional loss associated with peat extraction and windfarm installation.

7.4. LIMITATIONS AND RECOMMENDATIONS FOR FUTURE RESEARCH

The research presented in this thesis has focussed on one area of the Cantabrian Mountains between Picos de Europa in the eastern part of Asturias and the Pyrenees in the Basque Country and Navarra regions, identifying 14 areas of currently unrecorded blanket bog. Three further potential areas were preliminarily identified, but owing to access constraints these sites were not visited during field surveys. In addition, the identification of potential areas of blanket bogs for survey was constrained to areas where climatic conditions (Lindsay, 1995) and altitude (>600 masl; Pontevedra-Pombal *et al.*, 2009) are reported to be favourable for blanket bog development. It is possible that further blanket bogs exist outside of these constraints (e.g. the blanket bog in Asturias at 200 masl; Table 3.9; European Environment Agency, 2019). A key area for future research will therefore be to expand the search area, ideally to the whole of the Atlantic biogeographical region of northern Spain, and also to offer improved classification of the peatlands in Asturias that appear to be incorrectly reported as blanket bogs.

Assessment of the current state and rate of degradation was constrained to three blanket bogs. The evidence for the impact of livestock on peat erosion and loss is compelling, but it was noted that only one protected blanket bog was assessed. Based on the evidence presented in this research, the Cantabrian government are initiating protection of at least the two unprotected blanket bogs identified at Ilsos de Zalama and Collado de Hornaza. It is suggested that the rate of erosion at all three blanket bogs examined in this research was continued, as the installation of protection will provide further data on natural erosion rates, the impact of livestock and the impact of restoration on blanket bog in the Cantabrian Mountains.

The method developed to determine surface change in this research aimed to keep the error as small as possible as the scale of change occurring in the Cantabrian Mountains was not known and the erosion rates in the other blanket bogs in Europe have been reported to be small (Table 2.2). The data presented in chapters 5 and 6 demonstrate that a slightly higher level of error will not significantly affect the confidence in the scale of change determined. Rather than using a single scan strategy, it is suggested that multiple scans should be collected and aligned if reported erosion rates are high. This would increase the area of exposed peat that can be assessed and would also overcome some issues of occlusion in the data where peat that was deposited in the AOI created small areas of shadow in the data.

Although the proximity of Ilsos de Zalama and Zalama (500 m) enabled climatic influence to essentially be removed from interpretation of the data, it was clear that the installation of meteorological stations at several places in the study area would provide valuable input to future determination of change and the role of climatic variation. Occult precipitation is indicated to be extremely important for blanket bogs in the Cantabrian Mountains and quantification of the amount and importance would further our understanding of the role of occult precipitation in blanket bog development.

The identification of areas of shallow peat that connect individual blanket bog units (Figure 7.2) suggests that blanket mire complexes may have been more extensive in the Cantabrian

Mountains in the past. It is suggested that carbon dating of such shallow peat connections and adjacent blanket bogs be undertaken to provide insight into past extents. This information may be crucial in promoting wider protection measures that extend beyond the margins of current blanket bogs.

Given the clear change in vegetation within the fenced area at Zalama (Figure 5.15), and the increased presence of some peat-forming species (e.g. *Eriophorum vaginatum* (Chico and Clutterbuck, 2019), it is possible that some parts of Zalama are in equilibrium or may even be sequestering carbon. Determination of the carbon budget of Zalama and other blanket bogs is a further key area for future research to fully understand the role of Spanish blanket bogs in the current climate change situation.

7.5. CONCLUDING REMARKS

The work presented in this research has provided valuable contribution to our understanding of the distribution, diversity and significance of the southernmost edge-of-range of this habitat in continental Europe. Peat extraction, windfarms and livestock present the most serious threats to blanket bogs in the Cantabrian Mountains, but recognition and protection of blanket bogs significantly reduces the rate of degradation.

It is hoped that the research presented in this thesis can be used to instigate recognition of the blanket bogs identified in this research as habitat 7130 – blanket bogs by the European Environment Agency and Spanish Government when updating the Spanish peatland inventory. This will require action from the local governments, and designation as part of the Natura 2000 network enabling the EU to provide financial support for restoration. Urgent action is required.

References

AEMET (2018) *Resumen estacional climatológico*, *Invierno (2017/2018)*. Madrid: Agencia Estatal de Meteorología.

Aguirre, M., Benito, I. and Galera, A. (2017) 'Overview of the Zalama peat bog (Bizkaia) over the last thirty years', in Fernández-García, J. M. and Pérez, F. (eds) *Inventory, value and restoration of peatlands and mires: recent contributions*. Bizkaia: HAZI foundation, pp. 23–36.

Agus, F., Hairiah, K. and Mulyani, A. (2011) *Measuring carbon stocks in peat soils: practical guidelines*. Bogor: World Agroforestry Centre (ICRAF) Southeast Asia Regional Program, Indonesian Centre for Agricultural Land Resources Research and Development.

Alderson, D. M. *et al.* (2019) 'Trajectories of ecosystem change in restored blanket peatlands', *Science of the Total Environment*, 665, pp. 785–796.

Alexander, P. D. *et al.* (2008) 'Peat in horticulture and conservation: The UK response to a changing world', *Mires and Peat*, 3(8), pp. 1–10.

Anderson, P. (1986) *Accidental moorland fires in the Peak District*. Bakewell: Peak Park Joint Planning Board.

Anderson, P., Radford, E. and Mackenzie, S. (1989) *Marsden Moor Estate: Moorland erosion, its extent, cause and restoration*. England: Penny Anerson Associates.

Anderson, P., Tallis, J. H. and Yalden, D. W. (1997) *Restoring moorland - Peak District moorland management project - Phase III report*. Bakewell: Peak District Moorland Management Project.

Armentano, T. V. and Menges, E. S. (1986) 'Patterns of Change in the Carbon Balance of Organic Soil-Wetlands of the Temperate Zone', *Journal of Ecology*, 74(3), pp. 755–774.

Armstrong, A. *et al.* (2009) 'Drain-blocking techniques on blanket peat: A framework for best practice', *Journal of Environmental Management*, 90, pp. 3512–3519.

Armstrong, A. *et al.* (2012) 'Multi-scale relationship between peatland vegetation type and dissolved organic carbon concentration', *Ecological Engineering*, 47, pp. 182–188.

Asplund, D. (1996) 'Energy use of peat', in Vasander, H. (ed.) *Peatlands in Finland*. Helsinki: Finnish Peatland Society, pp. 107–113.

Bain, C. G. *et al.* (2011) *IUCN UK Commission of inquiry on peatlands*. Edinburgh: IUCN UK Peatland Programme.

Baldock, D. (1984) *Wetland drainage in Europe*. Nottingham: International Institute for Environment and Development.

Baynes, E. (2012) *Peat bog restoration: Implications of erosion and sediment transfer at*

Flow Moss, North Pennines. Durham University.

Bélanger, A. *et al.* (1988) *Peat. A resource of the future*. SainteFoy: Centre Québécois de Valorisation de la Biomass.

van Bellen, S. and Larivière, V. (2020) 'The ecosystem of peatland research: a bilbiometric analysis', *Mires and Peat*, 26(15), pp. 1–30.

Berglund, K. (1996) *Cultivated organic soils in Sweden: properties and amelioration*. Swedish University of Agricultural Sciences.

Betts, H. D. and De Rose, R. C. (1999) 'Digital elevation models as a tool for monitoring and measuring gully erosion', *ITC Journal*, 1(2), pp. 91–101.

Betts, H. D., Trustrum, N. A. and De Rose, R. C. (2003) 'Geomorphic changes in a complex gully system measured from sequential digital elevation models, and implications for management', *Earth Surface Processes and Landforms*, 28, pp. 1043–1058.

Bhatti, J. S. and Bauer, I. E. (2002) 'Comparing Loss-on-ignition with dry combustion as a method for determing carbon content in upland and lowland forest ecosystems'. *Communications in soil science and plant analysis,* 33(15–18), pp. 3419–3430.

Bibby, J. S. (1984) *Organization and methods of the 1:250 000 soil survey of Scotland.* Aberdeen: The Macauley Institute for Soil Research.

Bilotta, G. S., Brazier, R. E. and Haygarth, P. M. (2007) 'The impacts of grazing animals on the quality of soils, vegetation and surface waters in intensively managed grasslands', *Advances in Agronomy*, 94, pp. 237–280.

Birnie, R. V. (1974) *Glen Quoich: an investigation into the forms and processes associated with meander-scar terraces*. University of Aberdeen.

Birnie, R. V. (1993) 'Erosion rates on bare peat surfaces in Shetland', *Scottish Geographical Magazine*, 109(1), pp. 12–17.

BISE (2019) *Common International Classification of Ecosystem Services (CICES)*. Available at: https://biodiversity.europa.eu/maes/common-international-classification-of-ecosystem-services-cices-classification-version-4.3 (Accessed: 2nd July 2020).

Bluesky International Ltd. (2019) *Aerial data*. Available at: https://www.bluesky-world.com/ (Accessed: 24th June 2020).

Boardman, J. and Favis-Mortlock, D. (2016) 'The use of erosion pins in geomorphology', *Geomorphological Techniques*, 3(5), pp. 1–10.

BOE (2015) 'Boletin Oficial del Estado', *V. Anuncios*. Madrid: Spanish government, pp. 37260–37261.

Bonn, A. *et al.* (2009) *Drivers of Environmental Change in Uplands*. London: Routledge.

Botch, M. S. *et al.* (1995) 'Carbon pools and accumulation in peatlands of the former Soviet Union', *Global Biogeochemical Cycles*, 9, pp. 37–46.

Bower, M. M. (1960) 'The erosion of blanket peat in the southern pennines', *East*

Mindlands Geographer, 2(13), pp. 22–33.

Bower, M. M. (1961) ‘The Distribution of Erosion in Blanket Peat Bogs in the Pennines’, *Transactions of the Institute of British Geographers*, 29, pp. 17–30.

Bragazza, L. *et al.* (2017) ‘Italy’, in Joosten, H., Tanneberger, F., and Moen, A. (eds) *Mires and peatlands of Europe. Status, distribution and conservation*. Stuttgart: E. Schweizerbart’sche Verlagsbuchhandlung, pp. 462–477.

Brasington, J., Vericat, D. and Rychkov, I. (2012) ‘Modeling river bed morphology, roughness, and surface sedimentology using high resolution terrestrial laser scanning’, *Water Resources Research*, 48(11), pp. 1–18.

Bridgham, S. D. *et al.* (2006) ‘The carbon balance of North American wetlands’, *Wetlands*, 26(4), pp. 889–916.

Bright, J. A. *et al.* (2008) ‘Spatial overlap of wind farms on peatland with sensitive areas for birds, *Mires and Peat*, 4(7), pp. 1–12.

Bruneau, P. and Johnson, S. M. (2014) *Scotland’s peatland - definitions and information resources. Scottish Natural Heritage Commissioned Report No 701.* Edinburgh: Scottish Natural Heritage.

Bunsen, M. S. and Loisel, J. (2020) ‘Carbon storage dynamics in peatlands: Comparing recent- and long-term accumulation histories in southern Patagonia’, *Global Change Biology*, 0, pp. 1–18.

Byrne, K. A. *et al.* (2004) *EU peatlands: Current carbon stocks and trace gas fluxes*. Lund: CarboEurope Cluster.

Byrne, K. A. and Farrell, E. P. (2005) ‘The effect of afforestation on soil carbon dioxide emissions in blanket peatland in Ireland’, *Forestry*, 78(3), pp. 217–227.

Calderón, S. (1903) ‘Nota preliminar sobre la turba y los humedales de España’, *Boletín de la Real Sociedad Española de Historia Natural*, 3, pp. 417–428.

Campbell, D. R., Lavoie, C. and Rochefort, L. (2002) ‘Wind erosion and surface stability in abandoned milled peatlands’, *Canadian Journal of Soil Science*, 82(1), pp. 85–95.

Cannell, M. G. R., Dewar, R. C. and Pyatt, D. G. (1993) ‘Conifer plantations on drained peatlands in Britain: a net gain or loss of carbon?’, *Forestry: An International Journal of Forest Research*, 66(4), pp. 353–369.

Cantabria Government (2019) *Mapas Cantabria*. Available at: http://mapas.cantabria.es/ (Accessed: 29th November 2019).

Castillo, C., James, M.R., *et al.* (2012) ‘Comparing the Accuracy of Several Field Methods for Measuring Gully Erosion’, *Soil Science Society of America Journal*, 76, pp. 1319–1332.

Castillo, F. *et al.* (2001) *Turberas de montaña de Galicia*. Vigo: Xunta de Galicia.

Chambers, F. M., Beilman, D. W. and Yu, Z (2010) ‘Methods for determining peat humification and for quantifying peat bulk density, organic matter and carbon content for palaeostudies of climate and peatland carbon dynamics’, *Mires and Peat,* 7(7), pp. 1–10.

Chapin, F. S. *et al.* (2006) 'Reconciling carbon-cycle concepts, terminology, and methods', *Ecosystems*, 9(7), pp. 1041–1050.

Chapman, S. J. *et al.* (2009) 'Carbon stocks in Scottish peatlands', *Soil Use and Management*, 25(2), pp. 105–112.

Charman, D. (2002) *Peatlands and Environmental Change*. Chichester: John Wiley & Sons, Ltd.

Charman, D. J. *et al.* (2013) 'Climate-related changes in peatland carbon accumulation during the last millennium', *Biogeosciences*, 10, pp. 929–944.

Chico, G. and Clutterbuck, B. (2019) *Informe anual. Seguimiento de la restauración*. Southwell: Nottingham Trent Univertsity.

Clay, G. D., Worrall, F. and Aebischer, N. J. (2012) 'Does prescribed burning on peat soils influence DOC concentrations in soil and runoff waters? Results from a 10 year chronosequence', *Journal of Hydrology*, 448–449, pp. 139–148.

Clay, G. D., Worrall, F. and Fraser, E. D. G. (2010) 'Compositional changes in soil water and runoff water following managed burning on a UK upland blanket bog', *Journal of Hydrology*, 380, pp. 135–145.

Clay, G. D., Worrall, F. and Rose, R. (2010) 'Carbon budgets of an upland blanket bog managed by prescribed fire', *Journal of Geophysical Research*, 115, pp. 1–14.

Clutterbuck, B. *et al.* (2018) 'The potential of geospatial technology for monitoring peatland environments', in Fernández-García, J. M. and Pérez, F. (eds) *Inventory, value and restoration of peatlands and mires: recent contributions*. Bizkaia: HAZI foundation, pp. 167–182.

Clutterbuck, B. *et al.* (2020a) *Hard Hill experimental plots on Moor House – Upper Teesdale National Nature Reserve - A review of the experimental set up*. Natural England Commissioned Report No321.

Clutterbuck. B. *et al.* (2020b) 'Vehicular tracks and the influence of land use and habitat protection in the British uplands', *Science of The Total Environment,* 737, pp. 1402–1443.

Clutterbuck, B. and Yallop, A. R. (2009) 'Land management as a factor controlling dissolved organic carbon release from upland peat soils 1: Spatial variation in DOC productivity', *Science of the Total Environment*, 407(12), pp. 3803–3813.

Clutterbuck, B. and 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, pp. 6179–6191.

Clymo, R. S. (1992) 'Models of peat growth', *Suo*, 43(4–5), pp. 127–136.

Committee on Climate Change (2020) *Reducing UK emissions - Progress Report to Parliament.* Available at: https://www.theccc.org.uk/publication/reducing-uk-emissions-2020-progress-report-to-parliament/ (Accessed: 27th August 2020).

Connolly, J. and Holden, N. M. (2009) 'Mapping peat soils in Ireland: Updating the derived Irish peat map', *Irish Geography*, 42(3), pp. 343–352.

Cooper, A. and Loftus, M. (1998) 'The application of multivariate land classification to vegetation survey in the Wicklow Mountains, Ireland', *Plant Ecology*, 135(2), pp. 229–241.

Coupar, A., Immirzi, P. and Reid, E. (1997) 'The nature and extent of degradation in Scottish blanket mires', in Tallis, J. H., Meade, R., and Hulme, P. D. (eds) *Blanket mires degradation: Causes, consequences and challenges*. Aberdeen: British Ecological Society, pp. 90–100.

Couper, P., Stott, T. and Maddock, I. (2002) 'Insights into river bank erosion processes derived from analysis of negative erosion-pin recordings: Observations from three recent UK studies', *Earth Surface Processes and Landforms*, 27(1), pp. 59–79.

Crisp, D. T. (1966) 'Input and Output of Minerals for an Area of Pennine Moorland: The Importance of Precipitation, Drainage, Peat Erosion and Animals', *Journal of Applied Ecology*, 3(2), pp. 327–348.

Cruickshank, M. M. and Tomlinson, R. W. (1990) *Northern Ireland Peatland Survey*. Belfast: Belfast Department of Enviroment.

Dąbek, P. B. *et al.* (2018) 'The use of terrestrial laser scanning in monitoring and analyses of erosion phenomena in natural and anthropogenically transformed areas', *Cogent Geoscience*. Cogent, 4(1), pp. 1–18.

Daniels, S. M. *et al.* (2008) 'Water table variability and runoff generation in an eroded peatland, South Pennines, UK', *Journal of Hydrology*, 361, pp. 214–226.

Davies, G. M. and Legg, C. J. (2008) 'The effect of traditional management burning on lichen diversity', *Applied Vegetation Science*, 11(4), pp. 529–538.

Day, S. S. *et al.* (2013) 'Measuring bluff erosion part 1: Terrestrial laser scanning methods for change detection', *Earth Surface Processes and Landforms*, 38(10), pp. 1055–1067.

Douglas, C. (1998) 'Blanket bog conservation', in O'Leary, G. and Gormley, F. (eds) *Towars a conservation strategy for the bogs of Ireland*. Dublin: Irish Peatland Conservation Council, pp. 205–211.

Doyle, G. J. (1997) 'Blanket bogs: an intrepretation based on Irish blanket bogs', in Parkyn, L., Stoneman, R. E., and Ingram, H. A. P. (eds) *Conserving peatlands*. Wallingford: CAB International, pp. 25–34.

Drosler, M. *et al.* (2008) 'Observations and status of peatland greenhouse gas emissions in Europe', in Dolman, A. J. and Valentini, R. (eds) *The Continental-scale Greenhouse Gas Balance of Europe.* New York: Springer, pp. 243–261.

Dykes, A. P. and Selkirk-Bell, J. M. (2010) 'Landslides in blanket peat on subantarctic islands: Causes, characteristics and global significance', *Geomorphology*, 124(3–4), pp. 215–228.

Ellis, C. J. and Tallis, J. H. (2000) 'Climatic control of blanket mire development at Kentra Moss, north-west Scotland', *Journal of Ecology*, 88(5), pp. 869–889.

Ellis, C. J. and Tallis, J. H. (2001) 'Climatic control of peat erosion in a North Wales blanket mire', *New Phytologist*, 152, pp. 313–324.

Eltner, A., Mulsow, C. and Maas, H. G. (2013) ‘Quantitative Measurement of Soil Erosion From Tls and Uav Data’, *ISPRS - International Archives of the Photogrammetry, Remote Sensing and Spatial Information Sciences*, XL-1/W2, pp. 119–124.

Essl, F. and Steiner, G. M. (2017) ‘Austria’, in Joosten, H., Tanneberger, F., and Moen, A. (eds) *Mires and peatlands of Europe. Status, distribution and conservation*. Stuttgart: E. Schweizerbart’sche Verlagsbuchhandlung, pp. 257–265.

European Commission (2015) *Informe sobre el estado y las tendencias de los tipos de hábitats y las especies regulados por las Directivas de Aves y de Hábitats durante el período 2007-2012 exigido en virtud del artículo 17 de la Directiva de Hábitats y del artículo 12 de la Directiva de Aves*. Brussels: European Commission.

European Commission (2019) *The Habitats Directive*. Available at: https://ec.europa.eu/environment/nature/legislation/habitatsdirective/index_en.htm (Accessed: 1st July 2020).

European Commission (2012a) *National Summary for Article 17 - Spain*. Brussels: European Commission.

European Commission (2012b) *National Summary for Article 17 - United Kingdom*. Brussels: European Commission.

European Commission (2013) *Interpretation Manual of European Union Habitats*. Brussels: European Commission.

European Commission (2019) *LIFE Programme*. Available at: https://ec.europa.eu/easme/en/life (Accessed: 12th August 2020).

European Environment Agency (2012) *Report under the Article 17 of the Habitats Directive Period 2007-2012: 7130 Blanket bog (*active only)*. Brussels: European Environment Agency.

European Environment Agency (2019) *Blanket bogs (*if active bog)*, *Habitat types*. Available at: https://eunis.eea.europa.eu/habitats/10144 (Accessed: 29th November 2019).

European Topic Centre on Biological Diversity (2006) *The indicative Map of European Biogeographical Regions: Methodology and development*. Paris: Muséum National d'Histoire Naturelle.

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

Evans, M. and Warburton, J. (2005) ‘Sediment budget for an eroding peat-moorland catchment in northern England’, *Earth Surface Processes and Landforms*, 30(5), pp. 557–577.

Evans, M. and Warburton, J. (2007) *Geomorphology of Upland Peat: Erosion, Form and Landscape Change*. Australia: Blackwell Publishing.

Evans, M., Warburton, J. and Yang, J. (2006) ‘Eroding blanket peat catchments: Global and local implications of upland organic sediment budgets’, *Geomorphology*, 79, pp. 45–57.

Evans, R. (1974) *A study of selected erosion processes acting on slopes in a small drainage basin*. Sheffield University.

Evans, R. (1977) 'Overgrazing and soil erosion on hill pastures with particular reference to the Peak District', *Journal of the British Grassland Society*, 32, pp. 65–76.

Evans, R. (1998) 'The erosional impacts of grazing animals', *Progress in Physical Geography*, 22(2), pp. 251–268.

Farage, P. *et al.* (2009) 'Burning management and carbon sequestration of upland heather moorland in the UK', *Australian Journal of Soil Research*, 47, pp. 351–361.

FARO technologies Inc. (2015) *FARO Laser Scanner Focus3D x330*, *FARO Laser Scanner Focus3D x330*. Available at: https://faro.app.box.com/s/g3ssqjb1xadhs6wv3qilybxnj3mr0hli (Accessed: 10th June 2020).

Fergunson, P., Lee, J. A. and Bell, J. N. B. (1978) 'Effect of sulphur pollutants on the growth of sphagnum species', *Environmental Pollution*, 16(2), pp. 151–162.

Foss, P. and O'Connell, C. (2017) 'Ireland', in Joosten, H., Tanneberger, F., and Moen, A. (eds) *Mires and peatlands of Europe. Status, distribution and conservation*. Stuttgart: E. Schweizerbart'sche Verlagsbuchhandlung, pp. 449–461.

Foster, D. R. *et al.* (1988) 'Bog development and landform dynamics in Central Sweden and South-Eastern Labrador, Canada', *Journal of Ecology*, 76(4), pp. 1164–1185.

Foulds, S. A. and Warburton, J. (2007a) 'Significance of wind-driven rain (wind-splash) in the erosion of blanket peat', *Geomorphology*, 83, pp. 183–192.

Foulds, S. A. and Warburton, J. (2007b) 'Wind erosion of blanket peat during a short period of surface desiccation (North Pennines, Northern England)', *Earth Surface Processes Landforms*, 32, pp. 481–488.

Fraga, M. I. *et al.* (2008) 'Assessing the impact of wind farms on the plant diversity of blanket bogs in the Xistral Mountains (NW Spain).', *Mires and Peat*, 4(6), pp. 1–10.

Francis, I. S. (1990) 'Blanket peat erosion in a mid-wales catchment during two drought years', *Earth Surface Processes and Landforms*, 15(5), pp. 445–456.

Francis, I. S. and Taylor, J. A. (1989) 'The effect of forestry drainage operations on upland sediment yields', *Earth Surface Processes and Landforms*, 14(1), pp. 73–83.

Fuller, R. J. and Gough, S. J. (1999) 'Changes in sheep numbers in Britain', *Biological Conservation*, 91, pp. 73–89.

Gallego-Sala, A. V. *et al.* (2010) 'Bioclimatic envelope model of climate change impacts on blanket peatland distribution in Great Britain', *Climate Research*, 45, pp. 151–162.

Gallego-Sala, A. V. *et al.* (2016) 'Climate-driven expansion of blanket bogs in Britain during the Holocene', *Climate of the Past*, 12(1), pp. 129–136.

Gallego-Sala, A. V. and Prentice, I. C. (2013) 'Blanket peat biome endangered by climate change', *Nature Climate Change*, 3, pp. 152–155.

Garnett, M. and Adamson, J. (1997) ‘Blanket mire monitoring and research at Moor House national nature reserve’, in Tallis, J., Meade, R., and Hulme, P. (eds) *Blanket mires degradation: Causes, consequences and challenges*. London: British Ecological Society, pp. 116–117.

Garnett, M. H., Ineson, P. and Stevenson, A. C. (2000) ‘Effects of burning and grazing on carbon sequestration in a Pennine blanket bog, UK’, *Holocene*, 10(6), pp. 729–736.

Gerding, M. A. W., Karel, E. H. K. and de Vries, G. E. (2015) ‘The history of the peat manufacturing industry in The Netherlands: Peat moss litter and active carbon’, *Mires and Peat*, 16(10), pp. 1–9.

Gifford, J. M. and Hawkins, R. H. (1978) ‘Hydrologic impact of grazing on infiltration: a critical review’, *Water Resources Research*, 14, pp. 305–313.

Girardeau-Montaut, D., Roux, M. and Thibault, G. (2005) ‘Change Detection on Points Cloud Data Acquired With a Ground’, *ISPRS Workshop - Laser scanning 2005 – Enschede, the Netherlands, September 12-14*, pp. 30–35.

Glaser, P. H. *et al.* (2004) ‘Surface deformations as indicators of deep ebullition fluxes in a large northern peatland’, *Global Biogeochemical Cycles*, 18(GB1003), pp. 1–15.

Glaser, P. H. and Janssens, J. A. (1986) ‘Raised bogs in eastern North America - transitions in landforms and gross stratigraphy’, *Canadian Journal of Botany*, 64, pp. 1951–2007.

Glendell, M. *et al.* (2017) ‘Testing the utility of structure-from-motion photogrammetry reconstructions using small unmanned aerial vehicles and ground photography to estimate the extent of upland soil erosion’, *Earth Surface Processes and Landforms*, 42(12), pp. 1860–1871.

Gobierno de Cantabria (2017) *Boletín Oficial de Cantabria*, *Boletín Oficial de Cantabria (BOC)*. Santader: Gobierno de Cantabria.

Godfrey, S. *et al.* (2020) ‘Monitoring coastal morphology: The potential of low-cost fixed array action cameras for 3D reconstruction’, *Earth Surface Processes and Landforms*.

Gómez-Gutiérrez, A. *et al.* (2014) ‘Comparing two photo-reconstruction methods to produce high density point clouds and DEMs in the Corral del Veleta Rock Glacier (Sierra Nevada, Spain)’, *Remote Sensing*, 6, pp. 5407–5427.

Gómez-orellana, L. *et al.* (2014) ‘El sistema de turberas de la sierra de O Xistral como reservorio de carbono, valoración, estado de conservación y amenazas’, *Boletín de la Real Sociedad Española de Historia Natural*, 108, pp. 5–17.

Goodwillie, R. (1980) *Les tourbières en Europe*. Strasbourg: Comité Européen pour la Sauvegarde de la Nature et des Ressources Naturelles, Conseil de l’Europe.

Goodwin, N. R. *et al.* (2016) ‘Assessing the repeatability of terrestrial laser scanning for monitoring gully topography: A case study from Aratula, Queensland, Australia’, *Geomorphology*, 262, pp. 24–36.

Gorham, E. (1953) ‘A Note on the Acidity and Base Status of Raised and Blanket Bogs’, *Journal of Ecology*, 41(1), pp. 153–156.

Gorham, E. (1954) ‘On some factors affecting the chemical composition of Swedish fresh waters’, *Geochimica et Cosmochimica Acta*, 7(3–4), pp. 129–150.

Gorham, E. (1957) ‘The Development of Peat Lands’, *The Quarterly Review of Biology*, 32(2), pp. 145–166.

Gorham, E. *et al.* (1985) ‘Chemistry of bog waters’, in Stumm, W. (ed.) *Chemical Processes In Lakes*. New York: John Wiley & Sons Ltd, pp. 339–362.

Gorham, E. (1991) ‘Northern Peatlands : Role in the Carbon Cycle and Probable Responses to Climatic Warming’, *Ecological Applications*, 1(2), pp. 182–195.

Gorham, E. and Janssens, J. (1992) ‘Concepts of fen and bog re-examined’, *Acta societatis botanicorum poloniae*, 61(1), pp. 7–20.

Graniero, P. A. and Price, J. S. (1999) ‘The importance of topographic factors on the distribution of bog and heath in a Newfoundland blanket bog complex’, *Catena*, 36, pp. 233–254.

Grant, S. A., Bolton, G. R. and Torvell, L. (1985) ‘The responses of blanket bog vegetation to controlled grazing by hill sheep’, *Journal of Applied Ecology*, 22(3), pp. 739–751.

Grayson, R. *et al.* (2012) ‘Improving particulate carbon loss estimates in eroding peatlands through the use of terrestrial laser scanning’, *Geomorphology*, 179, pp. 240–248.

Greenwood, K. L. *et al.* (1998) ‘Soil physical properties after grazing exclusion’, *Grass and Forage Science*, 14, pp. 19–24.

Guerrero, F. (1987) *Study of physical and chemical properties of some Spanish peats and possible agricultural use properties*. Universidad Autónoma de Madrid.

Haigh, M. J. (1977) ‘The use of erosion pins in the study of slope evolution’, *British Geomorphological Research Group Technical Bulletin*, 18, pp. 31–49.

Hall, L. R. (2016) *Monitoring Bluff Erosion Rates Using Terrestrial Laser Scanning on Minnesota’s North Shore Streams*. University of Minnesota.

Hammond, R. F. (1981) *The Peatlands of Ireland*. 2nd edn. Dublin: An Foras Taluntais.

Hancock, G. R. and Evans, K. G. (2006) ‘Channel head location and characteristics using digital elevation models’, *Earth Surface Processes and Landforms*, 31, pp. 809–824.

Harper, A. R. *et al.* (2018) ‘Prescribed fire and its impacts on ecosystem services in the UK’, *Science of the Total Environment*, 624, pp. 691–703.

Heathwaite, A. L., Gottlich, K. H. and Cooke, J. (1993) *Mires: Process, Exploitation and Conservation*. Michigan: Wiley.

Heinemeyer, A., Berry, R. and Sloan, T. J. (2019) ‘Assessing soil compaction and micro-topography impacts of alternative heather cutting as compared to burning as part of grouse moor management on blanket bog’, *PeerJ*, 7, pp. 1–22.

Heras, P. (2002) *Determinación de los valores ambientales de la turbera del Zalama y propuestas de actuación para su conservación*. Vitoria: Diputación Foral de Bizkaia.

Heras, P. *et al.* (2017) ‘Spain’, in Joosten, H., Tanneberger, F., and Moen, A. (eds) *Mires and peatlands of Europe. Status, distribution and conservation*. Stuttgart: E. Schweizerbart’sche Verlagsbuchhandlung, pp. 639–656.

Heras, P. and Infante, M. (2003) ‘La turbera cobertor del Zalama (Burgos - Vizcaya): Un enclave unico en riesgo de desparicion’, *Estudios Museo de Ciencias Naturales de Alava*, 18–19, pp. 49–57.

Heras, P. and Infante, M. (2008) ‘Wind farms and mires in the Basque Country and north-west Navarra, Spain’, *Mires and Peat*, 4(04), pp. 1–14.

Heras, P. and Infante, M. (2018) ‘The Zalama blanket bog’, in Fernández-García, J. M. and Pérez, F. J. (eds) *Inventory, value and restoration of peatlands and mires: recent contributions*. Bizkaia: HAZI foundation.

Hijmans, R. J. *et al.* (2005) ‘Very high resolution interpolated climate surfaces for global land areas’, *International Journal of Climatology*, 25(15), pp. 1965–1978.

Hobbs, R. J. (1984) ‘Length of Burning Rotation and Community Composition in High-Level Calluna- Eriophorum Bog in N England’, *Vegetatio*, 57, pp. 129–136.

Höfle, B., Griesbaum, L. and Forbriger, M. (2013) ‘GIS-Based detection of gullies in terrestrial lidar data of the cerro llamoca peatland (peru)’, *Remote Sensing*, 5(11), pp. 5851–5870.

Holden, J. *et al.* (2006) ‘Impact of land drainage on peatland hydrology’, *Journal of Environmental Quality*, 35(5), pp. 1764–1778.

Holden, J. (2006) ‘Sediment and particulate carbon removal by pipe erosion increase over time in blanket peatlands as a consequence of land drainage’, *Journal of Geophysical Research: Earth Surface*, 111(2), pp. 1–6.

Holden, J. *et al.* (2007) ‘Environmental change in moorland landscapes’, *Earth-Science Reviews*, 82(1–2), pp. 75–100.

Holden, J. *et al.* (2014) ‘Fire decreases near-surface hydraulic conductivity and macropore flow in blanket peat’, *Hydrological Processes*, 28(5), pp. 2868–2876.

Holden, J. *et al.* (2015) ‘Impact of prescribed and repeated vegetation burning on blanket peat hydrology’, *Water Resources Research*, 51, pp. 1–13.

Holden, J. and Burt, T. P. (2002) ‘Piping and pipeflow in a deep peat catchment’, *Catena*, 48(3), pp. 163–199.

Holden, J., Chapman, P. J. and Labadz, J. C. (2004) ‘Artificial drainage of peatlands: hydrological and hydrochemical process and wetland restoration’, *Progress in Physical Geography*, 28(1), pp. 95–123.

Huang, C. C. (2002) ‘Holocene landscape development and human impact in the Connemara Uplands, Western Ireland’, *Journal of Biogeography*, 29, pp. 153–165.

Idrees, M. O. and 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), pp. 71–88.

Ilnicki, P. (2003) ‘Agricultural production systems for organic soil conservation’, in Parent, L. E. and Ilnicki, P. (eds) *Organic soils and peat materials*. Boca Ratón: CRC Press, pp. 99–187.

Imeson, A. C. (1974) ‘The origin of sediment in a moorland catchment with particular reference to the role of vegetation’, *Institute of British Geographers*, 6, pp. 59–72.

Immirzi, C. P., Maltby, E. and Clymo, R. S. (1992) *The global status of peatlands and their role in carbon cycling*. London: Friends of the Earth.

Ingram, H. A. P. (1978) ‘Soil layers in mires - Function and terminology’, *European journal of soil science*, 29(2), pp. 224–227.

Ingram, H. A. P. (1992) ‘Introduction to the ecohydrology of mires in the context of cultural perturbation’, in Bragg, O. M. et al. (eds) *Peatland Ecosystems and Man: an Impact Assessment*. Finland: International Peat Society, pp. 67–93.

Instituto Cántabro de Estadística (2020) *Sector Primario*. Available at: https://www.icane.es/economy/primary-sector (Accessed: 18th May 2020).

Instituto Geográfico Nacional (2019) *Centro de descargas SIG*. Available at: http://centrodedescargas.cnig.es/CentroDescargas/index.jsp (Accessed: 29th November 2019).

Instituto Geológico y Minero de España (1978) *Inventario y reconocimiento de indicios de turba en España*. Madrid: Gobierno de España.

IPCC (2019) ‘Climate Change and Land: An IPCC Special Report on climate change, desertification, land degradation, sustainable land management, food security, and greenhouse gas fluxes in terrestrial ecosystems - Summary for Policymakers’, in Shukla, P. R. et al. (eds) *Climate Change and Land: an IPCC special report on climate change, desertification, land degradation, sustainable land management, food security, and greenhouse gas fluxes in terrestrial ecosystems*. Intergovernmental Panel on Climate Change, p. 41.

Ivanok, K. (1981) *Water movement in mirelands*. Edited by A. Thompson and H. A. P. Ingram. London: Academic Press.

James, L. A., Watson, D. G. and Hansen, W. F. (2007) ‘Using LiDAR data to map gullies and headwater streams under forest canopy: South Carolina, USA’, *Catena*, 71, pp. 132–144.

Jeschke, L. and Succow, M. (2004) ‘Hydrogenetic mire types in South Australia and Tasmania’, *Telma*, 34, pp. 65–92.

Johnson, R. H. (1957) ‘Observations on the stream patterns of some peat moorlands in the southern Pennines’, *Memoirs of the Manchester Literary and Philosophical Society*, 99, pp. 1–18.

Jones, J. A. A. (2004) ‘Implications of natural soil piping for basin management in upland Britain’, *Land Degradation and Development*, 15(3), pp. 325–349.

Joosten, H. (2009) *The Global Peatland CO2 Picture*, *Wetlands International*. Ede: Wetlands International.

Joosten, H. *et al.* (2016) 'The role of peatlands in climate regulation', in Bonn, A. et al. (eds) *Peatland Restoration and Ecosystem Services: Science, Policy and Practice*. Cambridge: Cambridge University Press, pp. 63–76.

Joosten, H. *et al.* (2017) 'Mire diversity in Europe: mire and peatland types', in Joosten, H., Tanneberger, F., and Moen, A. (eds) *Mires and peatlands of Europe. Status, distribution and conservation*. Stuttgart: E. Schweizerbart'sche Verlagsbuchhandlung.

Joosten, H. and Clarke, D. (2002) *Wise use of mires and peatlands -Background and principles including a framework for decision-making*, *International Mire Conservation Group and International Peat Society*. Saarijärvi: International Mire Conservation Group and International Peat Society.

Joosten, H. and Tanneberger, F. (2017) 'Peatland use in Europe', in Joosten, H., Tanneberger, F., and Moen, A. (eds) *Mires and peatlands of Europe. Status, distribution and conservation*. Stuttgart: Schweizerbart Science Publishers, pp. 151–172.

Joosten, H., Tapio-Biström, M. L. and Tol, S. (2012) *Peatlands - guidance for climate change mitigation through conservation, rehabilitation and sustainable use*. 2nd edn. Rome: The Food and Agriculture Organization of the United Nations and Wetlands International Mitigation.

Julve, P. and Muller, F. (2017) 'France', in Joosten, H., Tanneberger, F., and Moen, A. (eds) *Mires and peatlands of Europe. Status, distribution and conservation*. Stuttgart: E. Schweizerbart'sche Verlagsbuchhandlung, pp. 395–402.

Kalacska, M. *et al.* (2013) 'Videographic analysis of Eriophorum vaginatum spatial coverage in an Ombotrophic bog', *Remote Sensing*, 5(12), pp. 6501–6512.

Keppeler, G. (1922) *Moornutzung und Torfverwertung*. Halle: Wilhelm Knapp.

Kimmel, K. and Mander, Ü. (2010) 'Ecosystem services of peatlands: Implications for restoration', *Progress in Physical Geography*, 34(4), pp. 491–514.

Kivinen, E. (1980) 'Proposal for a general classification of virgin peat', in Grubich, D. M., Farnham, R. S., and Itkoten, B. (eds) *Proceedings of the 6th International Peat Congress*. Buluth: International Peat Society, p. 47.

Kleinebecker, T., Hölzel, N. and Vogel, A. (2007) 'Gradients of continentality and moisture in South Patagonian ombrotrophic peatland vegetation', *Folia Geobotanica*, 42(4), pp. 363–382.

Kløve, B. (1998) 'Erosion and sediment delivery from peat mines', *Soil and Tillage Research*, 45(1–2), pp. 199–216.

Knoth, C. *et al.* (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(3), pp. 509–517.

Koehler, A. K., Sottocornola, M. and Kiely, G. (2011) 'How strong is the current carbon sequestration of an Atlantic blanket bog?', *Global Change Biology*, 17(1), pp. 309–319.

Kotowski, W., Dembek, W. and Pawlikowski, P. (2017) 'Poland', in Joosten, H., Tanneberger, F., and Moen, A. (eds) *Mires and peatlands of Europe. Status, distribution and conservation*. Stuttgart: E. Schweizerbart'sche Verlagsbuchhandlung, pp. 549–571.

Labadz, J. C. (1988) *Runoff and sediment production in blanket peat moorland studies in the South Pennines*. University of Huddersfield.

Labadz, J. C., Burt, T. P. and Potter, A. W. R. (1991) 'Sediment yield and delivery in the blanket peat moorlands of the southern pennines', *Earth Surface Processes and Landforms*, 16, pp. 255–271.

Lague, D., Brodu, N. and Leroux, J. (2013) 'Accurate 3D comparison of complex topography with terrestrial laser scanner: Application to the Rangitikei canyon (N-Z)', *ISPRS Journal of Photogrammetry and Remote Sensing*, 82, pp. 10–26.

Langlands, J. P. and Bennett, I. L. (1973) 'Stocking intensity and pastoral production. I. Changes in the soil and vegetation of a sown pasture grazed by sheep at different stocking rates', *Journal of Agricultural Science*, 81, pp. 193–204.

Lawrence, R. *et al.* (2009) *The Geography and Hydrology of High Country Peatlands in Victoria - Part 1. Geography and Classification*. Melbourne: Department of Sustainability and Environment.

Lehmann, J. R. K. *et al.* (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(3), pp. 1–19.

Li, C. *et al.* (2018) 'Erosion in peatlands: Recent research progress and future directions', *Earth-Science Reviews*, 185, pp. 870–886.

Li, C., Holden, J. and Grayson, R. (2018) 'Effects of Needle Ice on Peat Erosion Processes During Overland Flow Events', *Journal of Geophysical Research: Earth Surface*, 123(9), pp. 2107–2122.

Limpens, J. *et al.* (2008) 'Peatlands and the carbon cycle: From local processes to global implications - A synthesis', *Biogeosciences*, 5(5), pp. 1475–1491.

Lindholm, T. and Heikkilä, R. (2017) 'Finland', in Joosten, H., Tanneberger, F., and Moen, A. (eds) *Mires and peatlands of Europe. Status, distribution and conservation*. Stuttgart: E. Schweizerbart'sche Verlagsbuchhandlung, pp. 376–394.

Lindsay, R. *et al.* (1988) *The Flow Country: The Peatlands of Caithness and Sutherland*. Peterborough: Nature Conservancy Council.

Lindsay, R. (1995) *Bogs: The Ecology, Classification and Conservation of Ombrotrophic Mires*. Scotland: Scottish Natural Heritage.

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*. Scotland: Royal Society for the Protection of Birds.

Lindsay, R. (2016a) 'Peatland (Mire Types): Based on Origin and Bahavior of Water, Peat Genesis, Landscape Position, and Climate', in Finlayson, C. M. et al. (eds) *The Wetland Book*. Springer Netherlands.

Lindsay, R. (2016b) ‘Peatland classification’, in Finlayson, C. M. et al. (eds) *The Wetland Book*. Dordrecht: Springer Netherlands.

Lindsay, R. (2016c) ‘Peatlands and Windfarms: Conflicting Carbon Targets and Environmental Impacts’, in Finlayson, C. M. et al. (eds) *The Wetland Book*. Dordrecht: Springer Netherlands.

Lindsay, R. *et al.* (2019) ‘Peatlands: the challenge of mapping the world’s invisible stores of carbon and water’, *Unasylva*, 251, pp. 46–57.

Lindsay, R., Birnie, R. and Clough, J. (2014a) ‘Commercial peat extraction’, *IUCN UK Committee Peatland Programme*, 6, pp. 1–6.

Lindsay, R., Birnie, R. and Clough, J. (2014b) ‘Grazing and trampling’, *IUCN UK Committee Peatland Programme*, 7, pp. 1– 4.

Lindsay, R., Birnie, R. and Clough, J. (2014c) ‘Impacts of Artificial Drainage on Peatlands’, *IUCN UK Committee Peatland Programme*, 3, pp. 1–8.

Lindsay, R. and Bragg, O. (2005) *Wind farms and blanket peat*. London: University of East London.

Lindsay, R. and Clough, J. (2017) ‘United Kingdom’, in Joosten, H., Tanneberger, F., and Moen, A. (eds) *Mires and peatlands of Europe. Status, distribution and conservation*. Stuttgart: E. Schweizerbart’sche Verlagsbuchhandlung, pp. 705–720.

Löfroth, M. (2017) ‘Sweden’, in Joosten, H., Tanneberger, F., and Moen, A. (eds) *Mires and peatlands of Europe. Status, distribution and conservation*. Stuttgart: E. Schweizerbart’sche Verlagsbuchhandlung, pp. 664–675.

Loisel, J. *et al.* (2014) ‘A database and synthesis of northern peatland soil properties and Holocene carbon and nitrogen accumulation’, *Holocene*, 24(9), pp. 1028–1042.

Lovitt, J., Rahman, M. M. and McDermid, G. J. (2017) ‘Assessing the value of UAV photogrammetry for characterizing terrain in complex peatlands’, *Remote Sensing*, 9(7), pp. 1–13.

Lu, X. *et al.* (2017) ‘The effect of grid size on the quantification of erosion, deposition, and rill network’, *International Soil and Water Conservation Research*, 5(3), pp. 241–251.

Lunt, P. *et al.* (2010) *Peatland Restoration*. United Kingdom: IUCN UK Committee Peatland Programme.

Luscombe, D. J. *et al.* (2016) ‘How does drainage alter the hydrology of shallow degraded peatlands across multiple spatial scales?’, *Journal of Hydrology*, 541, pp. 1329–1339.

Mackay, A. W. and Tallis, J. H. (1994) ‘The recent vegetational history of the Forest of Bowland, Lancashire, UK’, *New Phytologist*, 128(3), pp. 571–584.

Mackay, A. W. and Tallis, J. H. (1996) ‘Summit-type blanket mire erosion in the Forest of Bowland, Lancashire, UK: Predisposing factors and implications for conservation’, *Biological Conservation*, 76, pp. 31–44.

Marcus, Y. (1997) ‘Blanket mire degradation in Wales’, in Tallis, J., Meade, R., and Hulme,

P. (eds) *Blanket mires degradation: Causes, consequences and challenges*. Aberdeen: British Ecological Society, pp. 101–115.

McGlone, M. S., Mark, A. F. and Bell, D. (1994) 'Late Pleistocene and Holocene vegetation history, Central Otago, South Island, New Zealand', *Journal of the Royal Society of New Zealand*, 25(1), pp. 1–22.

McGreal, W. S. and Larmour, R. A. (1979) 'Blanket peat erosion: Theoretical considerations and observations from selected conservation sites in slieveanorra forest national nature reserve, County Antrim', *Irish Geography*, 12(1), pp. 57–67.

McHugh, M. (2000) *Extent, causes and rates of upland soil erosion in England and Wales*. Cranfield University.

McHugh, M. (2007) 'Short-term changes in upland soil erosion in England and Wales: 1999 to 2002', *Geomorphology*, 86(1–2), pp. 204–213.

Mendes, C. F. M. and Dias, E. M. F. (2017) 'Azores', in Joosten, H., Tanneberger, F., and Moen, A. (eds) *Mires and peatlands of Europe. Status, distribution and conservation*. Stuttgart: E. Schweizerbart'sche Verlagsbuchhandlung, pp. 282–287.

Meteoblue (2017) *Historical climatic data*. Available at: https://www.meteoblue.com/en/historyplus (Accessed: 20th June 2018).

Milan, D. J., George, L. H. and Hetherington, D. (2007) 'Application of a 3D laser scanner in the assessment of erosion and deposition volumes and channel change in a proglacial river', *Earth Surface Processes and Landforms*, 32, pp. 1657–1674.

Milligan, G., Rose, R. J. and Marrs, R. H. (2015) 'Winners and losers in a long-term study of vegetation change at Moor upland vegetation', *Ecological Indicators*, 68, pp. 89–101.

Minkkinen, K. *et al.* (2002) 'Carbon balance and radiative forcing of Finnish peatlands 1900-2100 - The impact of forestry drainage', *Global Change Biology*, 8, pp. 785–799.

Minkkinen, K. *et al.* (2018) 'Persistent carbon sink at a boreal drained bog forest', *Biogeosciences*, 15(11), pp. 3603–3624.

Minkkinen, K. and Laine, J. (1998) 'Long-term effect of forest drainage on the peat carbon stores of pine mires in Finland', *Canadian Journal of Forest Research*, 28(9), pp. 1267–1275.

Moen, A. (1990) *The plant cover of the boreal uplands of central Norway. I. Vegetation ecology of Sølendet nature reserve; haymaking fens and birch woodlands*. Trondheim: Universitetet i Trondheim Vitenskapsmuseet.

Moen, A., Joosten, H. and Tanneberger, F. (2017) 'Mire diversity in Europe: mire regionality', in Joosten, H., Tanneberger, F., and Moen, A. (eds) *Mires and peatlands of Europe. Status, distribution and conservation*. Stuttgart: Schweizerbart Science Publishers, p. 134.

Moen, A., Lyngstad, A. and Øien, D. (2017) 'Norway', in Joosten, H., Tanneberger, F., and Moen, A. (eds) *Mires and peatlands of Europe. Status, distribution and conservation*. Stuttgart: E. Schweizerbart'sche Verlagsbuchhandlung, pp. 536–548.

Monserrat, O. and Crosetto, M. (2008) 'Deformation measurement using terrestrial laser scanning data and least squares 3D surface matching', *ISPRS Journal of Photogrammetry* *and Remote Sensing*, 63(1), pp. 142–154.

Montanarella, L., Jones, R. J. A. and Hiederer, R. (2006) 'The distribution of peatland in Europe', *Mires and Peat*, 1(1), pp. 1–11.

Montemaggiori, A. (1996) *Le Zone Umide in Italia*. Roma: WWF Italia.

Moore, P. D. and Bellamy, D. J. (1974) *Peatlands*. 1st edn. London: Paul Elek Ltd.

Moore, T. R. and Knowles, R. (1987) 'Methane and Carbon Dioxide evolution from subarctic fens', *Canadian Journal of Soil Science*, 67, pp. 77–81.

Moore, T. R. and Knowles, R. (1989) 'The influence of water table levels on methane and carbon dioxide emissions from peatland soils', *Canadian Journal of Soil Science*, 69, pp. 33–38.

Moors for the Future (2020) *Conservation work*. Available at: https://www.moorsforthefuture.org.uk/our-work/restoring-blanket-bog (Accessed: 12th August 2020).

Moss, C. E. (1913) *Vegetation of the Peak District*. New York: Cambridge University Press.

Nayak, D. R. *et al.* (2010) 'Calculating carbon budgets of wind farms on Scottish peatlands', *Mires and Peat*, 4(9), pp. 1–23.

Neugirg, F. *et al.* (2016) 'Erosion processes in calanchi in the Upper Orcia Valley, Southern Tuscany, Italy based on multitemporal high-resolution terrestrial LiDAR and UAV surveys', *Geomorphology*, 269, pp. 8–22.

Newton, A. C. *et al.* (2009) 'Impacts of grazing on lowland heathland in north-west Europe', *Biological Conservation*, 142(5), pp. 935–947.

Noble, A. *et al.* (2017) 'Prescribed burning, atmospheric pollution and grazing effects on peatland vegetation composition', *Journal of Applied Ecology*, 55(2), pp. 559–569.

Noble, A. *et al.* (2019) 'Peatland vegetation change and establishment of re-introduced Sphagnum moss after prescribed burning', *Biodiversity and Conservation*, 28(4), pp. 939–952.

NPWS (2019) *The Status of EU Protected Habitats and Species in Ireland*. Dublin: NPWS.

Nugent, K. A. *et al.* (2018) 'Multi-year net ecosystem carbon balance of a restored peatland reveals a return to carbon sink', *Global Change Biology*, 24(12), pp. 5751–5768.

O'Brien, H., Labadz, J. C. and Butcher, D. P. (2007) *Review of blanket bog management and restoration*. Southwell: Nottingham Trent University.

O'Neal, M. A. and Pizzuto, J. E. (2011) 'The rates and spatial patterns of annual riverbank erosion revealed through terrestrial laser-scanner surveys of the South River, Virginia', *Earth Surface Processes and Landforms*, 36(5), pp. 695–701.

Ordóñez, C. *et al.* (2018) 'Study of the evolution of the topography of the Comeya bog (Picos ed Europa National Park) over time with a terrestrial laser scanner', in Fernández-García, J. M. and Pérez, F. (eds) *Inventory, value and restoration of peatlands and mires: recent contributions*. Bizkaia: HAZI foundation, pp. 85–104.

Ouédraogo, M. M. *et al.* (2014) ‘The evaluation of unmanned aerial system-based photogrammetry and terrestrial laser scanning to generate DEMs of agricultural watersheds’, *Geomorphology*, 214, pp. 339–355.

Paramananthan, S., Lee, P., Wong, M., Van Ranst, E., Wüst, R. A. J. and Vijiandran, J. R. (2018) ‘A comparative study of the use of organic carbon and loss on ignition in defining tropical organic soil materils’, *Communications in Soil Science and Plant Analysis,* 49(5), pp. 626–634.

Paavilainen, E. and Päivänen, J. (1995) *Peatland forestry – ecology and principles (Ecological Studies 111)*. Berlin: Springer.

Page, S. E., Rieley, J. O. and Banks, C. J. (2011) ‘Global and regional importance of the tropical peatland carbon pool’, *Global Change Biology*, 17, pp. 798–818.

Parish, F. *et al.* (2008) *Assessment on Peatlands, Biodiversity and Climate change: Main Report*, *Assessment on Peatlands, Biodiversity and Climate Change: Main Report. Global Environment Centre, Kuala Lumpur and Wetlands International*. Wageningen: Global Environment Centre, Kuala Lumpur and Wetlands International.

Parry, L., Holden, J. and Chapman, P. (2014) ‘Restoration of blanket peatlands’, *Journal of Environmental Management*, 133, pp. 193–205.

Payette, S. (2001) ‘Les principaux types de tourbières’, in Payette, S. and Rochefort, L. (eds) *Écologie des tourbières du Québec-Labrador*. Québec: Les Presses de l’Université Laval, pp. 39–89.

Pérez-Díaz, S. *et al.* (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(4), pp. 658–672.

Pfadenhauer, J. *et al.* (1993) ‘Mire distribution’, in Heathwaite, A. L. and Göttlich, K. (eds) *Mires: Process, Exploitation and Conservation*. Chichester: Wiley, pp. 77–121.

Phillips, J., Tallis, J. and Yalden, D. (1981) *Peak District moorland erosion study: Phase 1 report*. Bakewell: Peak Park Joint Planning Board.

Pilkington, M. G. (2015) ‘Annex 1: Background, location, design and restoration’, in *Restoration of Blanket bogs; flood risk reduction and other ecosystem benefits. Final report of the Making Space for Water project*. Edale: Moors for the Future Partnership.

Pontevedra-Pombal, X. *et al.* (2009) *7130 Turberas de cobertor (*) (para las turberas activas)*, *W.AA., Bases ecológicas preliminares para la conservación de los tipos de hábitat de interés comunitaroi en España*. Madrid: Gobierno de España.

Pontevedra-Pombal, X. *et al.* (2017) ‘Iberian acid peatlands: types, origin and general trends of development’, *Mires*, 19(October), pp. 1–19.

Pontevedra-Pombal, X. *et al.* (2019) ‘10,000 years of climate control over carbon accumulation in an Iberian bog (southwestern Europe)’, *Geoscience Frontiers*, 10(4), pp. 1521–1533.

Pontevedra Pombal, X. and Martínez Cortizas, A. (2004) ‘Turberas de Galicia: procesos formativos, distribución y valor medioambiental. El caso particular de las “Serras Septentrionais”’, *Chioglossa*, 2, pp. 103–121.

Price, J. S. (1992) ‘Blanket bog in Newfoundland. Part 2. Hydrological processes’, *Journal of Hydrology*, 135, pp. 103–119.

Price, J. S., Heathwaite, A. L. and Baird, A. J. (2003) ‘Hydrological processes in abandoned and restored peatlands: an overview of management approaches’, *Wetlands Ecology and Management*, 11, pp. 65–83.

Radley, J. (1962) ‘Peat erosion on the high moors of Derbyshire and West Yorkshire’, *East Midland Geographer*, 17.

Radley, J. (1965) ‘Significance of Major Moorland Fires’, *La Nature*, 205(4978), pp. 1254–1259.

Ramil-Rego, P. *et al.* (2017) ‘Información Territorial: Región Atlántica del Norte de la Península Ibérica’, in Ramil-Rego, P. and Rodríguez Guitián, M. A. (eds) *Hábitats de turbera en la Red Natura 2000*. Lugo: Horreum-Ibader, pp. 245–376.

Ramil-Rego, P. and Aira-Rodríguez, M. J. (1994) ‘Análisis polínico y sedimentológico de dos turberas en las Sierras Septentrionales de Galicia (N.O. de España)’, *Revue de Paléobiologie*, 13(1), pp. 9–28.

Ramil-Rego, P., Rodríguez-Guitián, M. A. and Rodríguez-Oubiña, J. (1996) ‘Valoración de los Humedales Continentales del NW Ibérico: Caracterización Hidrológica, Geomorfológica y Vegetacional de las Tuberas de las Sierras Septentrionales de Galicia’, in Pérez-Alberti, A. and Martínez-Cortizas, A. (eds) *Avances en la Reconstrucción Paleoambiental de las Áreas de Montaña Lucenses*. Lugo: Diputación Provincial de Lugo, pp. 165–187.

Rastall, R. H. and Smith, B. (1906) ‘III.—Tarns on the Haystacks Mountain, Buttermere, Cumberland’, *Geological Magazine*, 3(9), pp. 406–412.

Razzhigaeva, N. G. *et al.* (2009) ‘Wetlands in the Southern Kuril Islands: Origin, age, evolution’, in *International Wetlands: Ecology, Conservation and Restoration*, pp. 191–231.

Du Rietz, G. E. (1954) ‘Die Mineralbodenwasserzeigergrenze als Grundlage einer natürichen Zweigliederung der nord- und mitteleuropäischen Moore’, *Vegetatio*, 5–6, pp. 571–585.

Robinson, M. and Newson, M. D. (1986) *Comparison of Forest and Moorland Hydrology in an Upland Area with Peat Soils.*

Romell, L. G. and Heiberg, S. O. (1931) ‘Types of Humus Layer in the Forests of Northeastern United States’, *Ecology*, 12(3), pp. 567–608.

Roulet, N. T. *et al.* (2007) ‘Contemporary carbon balance and late Holocene carbon accumulation in a northern peatland’, *Global Change Biology*, 13, pp. 397–411.

Rydin, H. and Jeglum, J. K. (2006a) ‘Productivity and carbon balance’, in Rydin, H. and Jeglum, J. K. (eds) *The Biology of Peatlands*. Berlin: Springer, pp. 240–261.

Rydin, H. and Jeglum, J. K. (2006b) ‘Uses, functions, and management of peatlands’, in Rydin, H. and Jeglum, J. K. (eds) *The Biology of Peatlands*. New York: Oxford University Press, pp. 262–288.

Sakaguchi, Y. (2001) 'The Ozegahara peat-bog - Striking features and genesis of a special geotope in Japan', *Telma*, 31, pp. 151–181.

Samuelsson, G. (1910) 'Scottish peat mosses. A contribution to the knowledge of the late-quaternary vegetation and climate of north-western Europe', *Bulletin of the Geological Institute of Upsala*, 10, pp. 197–260.

Schlotzhauer, S. M. and Price, J. S. (1999) 'Soil water flow dynamics in a managed cutover peat field , Quebec: Field and laboratory investigations, *Water Resources Research*, 35(12), pp. 3675–3683.

Schneider, S. (1976) 'Verteilung der Moore auf der Erde', in Göttlich, K. (ed.) *Moor- und Torfkunde*. Stuttgart: Schweizerbart, pp. 27–40.

Schumann, M. and Joosten, H. (2008) *Global peatland restoration manual*. Available at: http://www.imcg.net/media/download_gallery/books/gprm_01.pdf (Accessed: 6th May 2020).

Schürch, P. *et al.* (2011) 'Detection of surface change in complex topography using terrestrial laser scanning: application to the Illgraben debris-flow channel', *Earth Surface Processes and Landforms*, 36(14), pp. 1847–1859.

Selkirk, J. M. and Saffigna, L. J. (1999) 'Wind and water erosion of a peat and sand area on subantarctic Macquarie Island', *Arctic, Antarctic, and Alpine Research*, 31(4), pp. 412–420.

Semeniuk, C. A. and Semeniuk, V. (1995) 'A Geomorphic Approach to Global Classification for Inland Wetlands', *Vegetatio*, 118(1/2), pp. 103–124.

Sjörs, H. (1948) 'Myrvegetation i Bergslagen', *Acta Phytogeographica Suecica*, 21, pp. 1–299.

Sjörs, H. (1950) 'On the Relation between Vegetation and Electrolytes in North Swedish Mire Waters', *Oikos*, 2(2), pp. 241–258.

Sjörs, H. (1983) 'Mires of Sweden', in Gore, A. J. P. (ed.) *Ecosystems of the World 4B. Mires: Swamp, Bog, Fen and Moor*. Amsterdam: Elsevier, pp. 69–94.

Sjors, J. (1985) 'A comparison between mires of southern Alaska and Fennoscandia', *Aquilo Series Botanica*, 21, pp. 89–94.

Sloan, T. J. *et al.* (2018) 'Peatland afforestation in the UK and consequences for carbon storage', *Mires and Peat*, 23(1), pp. 1–17.

Smith, J., Nayak, D. R. and Smith, P. (2014) 'Wind farms on undegraded peatlands are unlikely to reduce future carbon emissions', *Energy Policy,* 66, pp. 585–591.

Smith, M. W. (2015) 'Direct acquisition of elevation data : Terrestrial Laser Scanning', *Geomorphological Techniques*, 1, pp. 1–14.

Smith, M. W. and Warburton, J. (2018) 'Microtopography of bare peat: a conceptual model and objective classification from high-resolution topographic survey data', *Earth Surface Processes and Landforms*, 43, pp. 1557–1574.

Solem, T. (1994) 'Age, origin and development of blanket mires in Central Norway', *Norwegian Journal of Geography*, 48, pp. 93–98.

Sottocornola, M. and Kiely, G. (2005) 'An Atlantic blanket bog is a modest CO2 sink', *Geophysical Research Letters*, 32(23), pp. 1–4.

Stevenson, A. C., Jones, V. J. and Battarbee, R. W. (1990) 'The cause of peat erosion: a palaeolimnological approach', *New Phytologist*, 114(4), pp. 727–735.

Stott, T. (1997) 'A comparison of stream bank erosion processes on forested and moorland streams in the Balquhidder Catchments, central Scotland', *Earth Surface Processes and Landforms*, 22(4), pp. 383–399.

Swindell, S. F. (2019) *Assessment of the effect of pre-fire vegetation and management on wildfire damage and recovery using remote sensing and ground data*. Nottingham Trent University.

Tallis, J. H. (1964) 'Studies on Southern Pennine Peats: II. The Pattern of Erosion', *Journal of Ecology*, 52(2), pp. 333–344.

Tallis, J. H. (1973) 'Studies on Southern Pennine Peats: V. Direct Observations on Peat Erosion and Peat Hydrology at Featherbed Moss, Derbyshire', *The Journal of Ecology*, 61(1), pp. 1–22.

Tallis, J. H. (1987) 'Fire and Flood at Holme Moss: Erosion Processes in an Upland Blanket Mire', *Journal of Ecology*, 75(4), pp. 1099–1129.

Tallis, J. H. (1997a) 'The Pollen Record of Empetrum Nigrum in Southern Pennine Peats : Implications for Erosion and Climate Change', *Journal of Ecology*, 85(4), pp. 455–465.

Tallis, J. H. (1997b) 'The Southern Pennine experience: An overview of blanket mire degradation', in Tallis, J. H., Meade, R., and Hulme, P. D. (eds) *Blanket mires degradation: Causes, consequences and challenges*. Aberdeen: Macaulay Land Use Research Institute, pp. 7–16.

Tallis, J. H. (1998) 'Growth and degradation of British and Irish blanket mires', *Environmental Reviews*, 6(2), pp. 81–122.

Tallis, J. H. and Yalden, D. W. (1983) *Peak District moorland restoration project - Phase 2 Report: Re-vegetation trials*. Bakewell: Peak Park Joint Planning Board.

Tanneberger, F. *et al.* (2017) 'The peatland map of Europe', *Mires and Peat*, 19(22), pp. 1–17.

Tarnocai, C. (2006) 'The effect of climate change on carbon in Canadian peatlands', *Global and Planetary Change*, 53(4), pp. 222–232.

Tedim, F., Xanthopoulos, G. and Leone, V. (2015) 'Forest Fires in Europe: Facts and Challenges', in Paton, D. et al. (eds) *Wildfire Hazards, Risks and Disasters*. Amsterdam: Elsevier, pp. 77–99.

Thompson, D. K. and Waddington, J. M. (2013) 'Peat properties and water retention in boreal forested peatlands subject to wildfire', *Water Resources Research*, 49(6), pp. 3651–3658.

Trepel, M. *et al.* (2017) 'Germany', in Joosten, H., Tanneberger, F., and Moen, A. (eds) *Mires and peatlands of Europe. Status, distribution and conservation*. Stuttgart: E. Schweizerbart'sche Verlagsbuchhandlung, pp. 413–424.

Turetsky, M. *et al.* (2002) 'Current disturbance and the diminishing peatland carbon sink', *Geophysical Research Letters*, 29(11), pp. 1–4.

Turunen, J. *et al.* (2002) 'Estimating carbon accumulating rates of undrained mires in Finland - application to boreal and subarctic regions', *The Holocene*, 12(1), pp. 69–80.

Waddington, J. M., Strack, M. and Greenwood, M. J. (2010) 'Toward restoring the net carbon sink function of degraded peatlands: Short-term response in CO2 exchange to ecosystem-scale restoration', *Journal of Geophysical Research*, 115(G1008), pp. 1–13.

Waddington, J. M., Warner, K. D. and Kennedy, G. W. (2002) 'Cutover peatlands: A persistent source of atmospheric CO2', *Global Biogeochemical Cycles*, 16(1), pp. 1–7.

Walsh, S. J., Butler, D. R. and Malanson, G. P. (1998) 'An overview of scale, pattern, process relationships in geomorphology: a remote sensing and GIS perspective', *Geomorphology*, 21, pp. 183–205.

Warburton, J. (2003) 'Wind-splash erosion of bare peat on UK upland moorlands', *Catena*, 52, pp. 191–207.

Ward, S. *et al.* (2008) 'Gully hotspot contribution to landscape methane (CH4) and carbon dioxide (CO2) fluxes in a northern peatland', *Science of The Total Environment*, 404(2–3), pp. 354–360.

Ward, S. E. *et al.* (2007) 'Long-Term Consequences of Grazing and Burning on Northern Peatland Carbon Dynamics', *Ecosystems*, 10(7), pp. 1069–1083.

Warner, K. (1999) *C02 exchange in cutover and restored peatlands*. McMaster University.

Wawrzyczek, J. *et al.* (2018) 'The ecosystem approach in ecological impact assessment: Lessons learned from windfarm developments on peatlands in Scotland', *Environmental Impact Assessment Review*, 72, pp. 157–165.

Weber, C. A. (1903) 'Über Torf, Humus und Moor. Versuch einer Begriffsbestimmung mit Rücksicht auf die Kartiertrog und die Statistik der Moore.', *Abhandlungen des Naturwissenschaftlichen Vereins zu Bremen*, 17, pp. 465–484.

Wells, E. D. and Pollet, F. C. (1983) 'Peatlands (of Newfoundland).', in *Biogeography and ecology of the island of Newfoundland*, pp. 207–265.

Westoby, M. J. *et al.* (2012) '"Structure-from-Motion" photogrammetry: A low-cost, effective tool for geoscience applications', *Geomorphology*, 179, pp. 300–314.

Whittle, A. and Gallego-Sala, A. V. (2016) 'Vulnerability of the peatland carbon sink to sea-level rise', *Scientific Reports*, 6, pp. 1–11.

Williams, R. (1995) *Wetlands - Tealham Moor*. London: Immel.

Wilson, S. J. and Cooke, R. U. (1980) 'Wind erosion', in Kirkby, M. J. and Morgan, R. P. (eds) *Soil erosion*. Chichester: Wiley, pp. 51–217.

Wittram, B. W. *et al.* (2015) *A Practitioners Guide to Sphagnum Reintroduction*. Edale: Moors for the Future Partnership

Worrall, F. *et al.* (2010) *Impacts of Burning Management on Peatlands*. England: IUCN UK Peatland Programme Commission.

Worrall, F., Armstrong, A. and Adamson, J. K. (2007) 'The effects of burning and sheep-grazing on water table depth and soil water quality in a upland peat', *Journal of Hydrology*, 339, pp. 1–14.

Worrall, F. and Clay, G. D. (2012) 'The impact of sheep grazing on the carbon balance of a peatland', *Science of the Total Environment*, 438, pp. 426–434.

Xu, J. *et al.* (2018) 'PEATMAP: Refining estimates of global peatland distribution based on a meta-analysis', *Catena*, 160, pp. 134–140.

Yallop, A. R. *et al.* (2006) 'The extent and intensity of management burning in the English uplands', *Journal of Applied Ecology*, 43(6), pp. 1138–1148.

Yallop, A. R., Clutterbuck, B., Thacker, J. (2009) 'Increases in humic dissolved organic carbon export from upland peat catchments: the role of temperature, declining sulphur deposition and changes in land management', *Climate Research,* 45, pp. 43–56.

Yeloff, D. E. (2001) *Causes of blanket peat erosion in the pennines: A palaeoenvironmental approach*. University of Huddersfield.

Yeloff, D. E., Labadz, J. C. and Hunt, C. O. (2006) 'Causes of degradation and erosion of a blanket mire in the southern Pennines, UK', *Mires and Peat*, 1(4), pp. 1–18.

Yu, Z. *et al.* (2010) 'Global peatland dynamics since the Last Glacial Maximum', *Geophysical Research Letters*, 37(13), pp. 1–5.

Zuidhoff, F. S. (2002) 'Recent decay of a single palsa in relation to weather conditions between 1996 and 2000 in Laivadalen, northern Sweden', *Geografiska Annaler, Series A: Physical Geography*, 84(2), pp. 103–111.

Appendix A

Typical features of the identified blanket bogs

A.1. ZALAMA

Overview

Peat-forming vegetation

Degraded areas

Pools

A.2. ILSOS DE ZALAMA

Overview

Peat-forming vegetation

Degraded areas

Pools

A.3. MOTAS DEL PARDO

Overview	*Peat-forming vegetation*
	
Degraded areas	*Pools*
	

A.4. COLLADO DE HORNAZA

Overview	*Peat-forming vegetation*
	
Degraded areas	*Pools*
	

A.5. LA MARRUYA

Overview	*Peat-forming vegetation*
	
Degraded areas	*Pools*
	

A.6. SEL DE LA PEÑA

Overview	*Peat-forming vegetation*
	
Degraded areas	*Pools*
	

A.7. CERCIO

Overview	Peat-forming vegetation
	
Degraded areas	**Pools**
	

A.8. COTERO SENANTES

Overview	Peat-forming vegetation
	
Degraded areas	**Pools**
	

A.9. EL COTERO

Overview	*Peat-forming vegetation*


Degraded areas	*Pools*


A.10. EL COTERO SUR

Overview	*Peat-forming vegetation*


Degraded areas	*Pools*


A.11. COTERO DE LA OSERA

Overview	*Peat-forming vegetation*
	
Degraded areas	*Pools*
	

A.12. PEÑA OJASTRA

Overview	*Peat-forming vegetation*
	
Degraded areas	*Pools*
	

A.13. CANTOS CALIENTES

Overview	*Peat-forming vegetation*
	
Degraded areas	*Pools*
	

A.14. MALVERDE

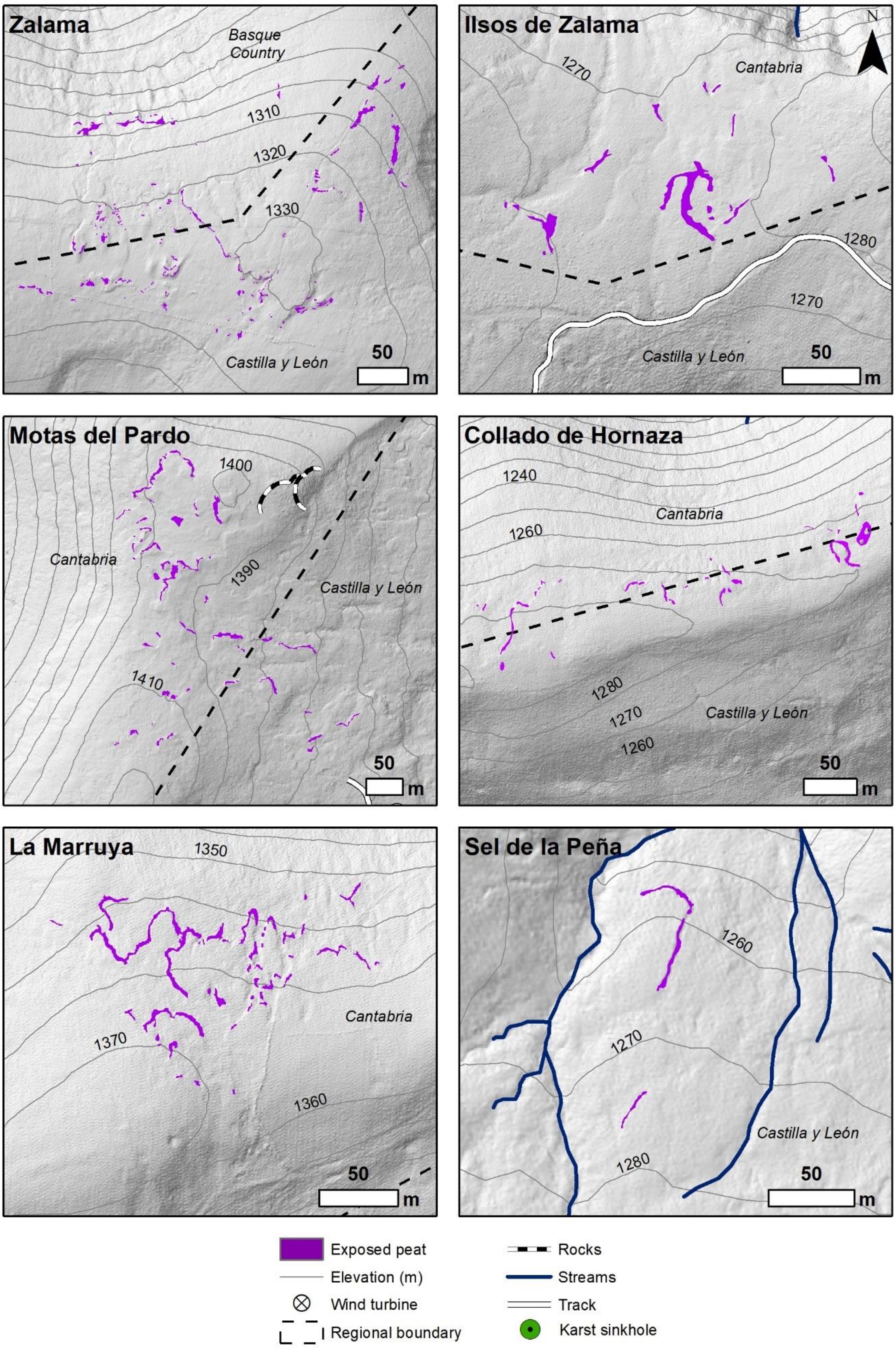
Overview	*Peat-forming vegetation*
	
Degraded areas	*Pools*
	

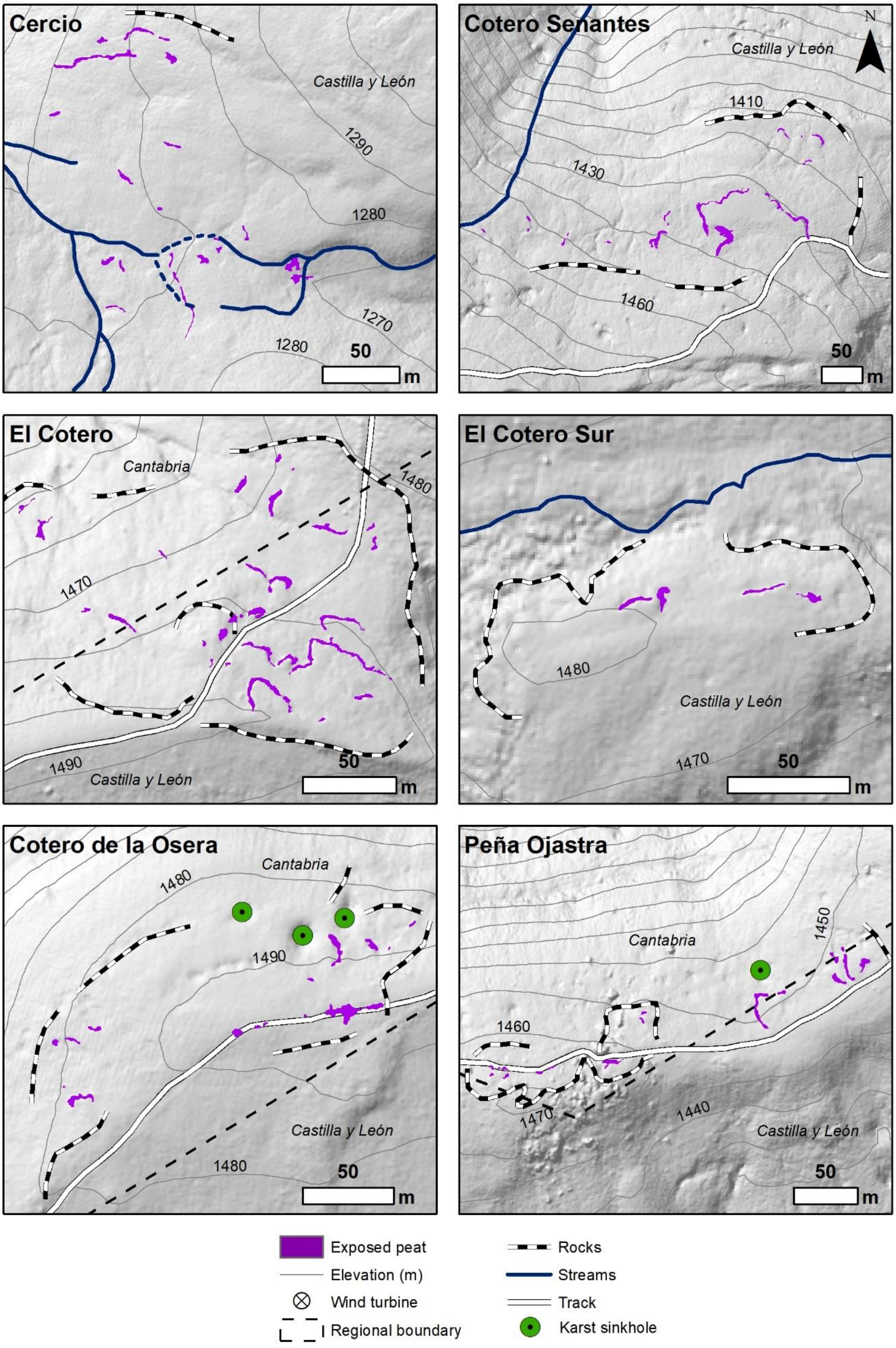
A.15. EL CUITO

Overview	*Peat-forming vegetation*
	
Degraded areas	*Pools*
	

Appendix B

Current areas of exposed peat mapped at all blanket bogs


Zalama
Basque Country
1310
1320
1330
Castilla y León
50
m
Ilsos de Zalama
N
Cantabria
1270
1280
1270
Castilla y León
50
m
Motas del Pardo
1400
Cantabria
1390
Castilla y León
1410
50
m
Collado de Hornaza
1240
Cantabria
1260
1280
1270
Castilla y León
1260
50
m
La Marruya
1350
Cantabria
1370
1360
50
m
Sel de la Peña
1260
1270
Castilla y León
1280
50
m
Exposed peat
Elevation (m)
Wind turbine
Regional boundary
Rocks
Streams
Track
Karst sinkhole



Cercio
Castilla y León
1290
1280
1270
1280
50
m
Cotero Senantes
N
Castilla y León
1410
1430
1460
50
m
El Cotero
Cantabria
1480
1470
1490
Castilla y León
50
m
El Cotero Sur
1480
Castilla y León
1470
50
m
Cotero de la Osera
Cantabria
1480
1490
Castilla y León
1480
50
m
Peña Ojastra
1450
Cantabria
1460
1470
1440
Castilla y León
50
m
Exposed peat
Elevation (m)
Wind turbine
Regional boundary
Rocks
Streams
Track
Karst sinkhole

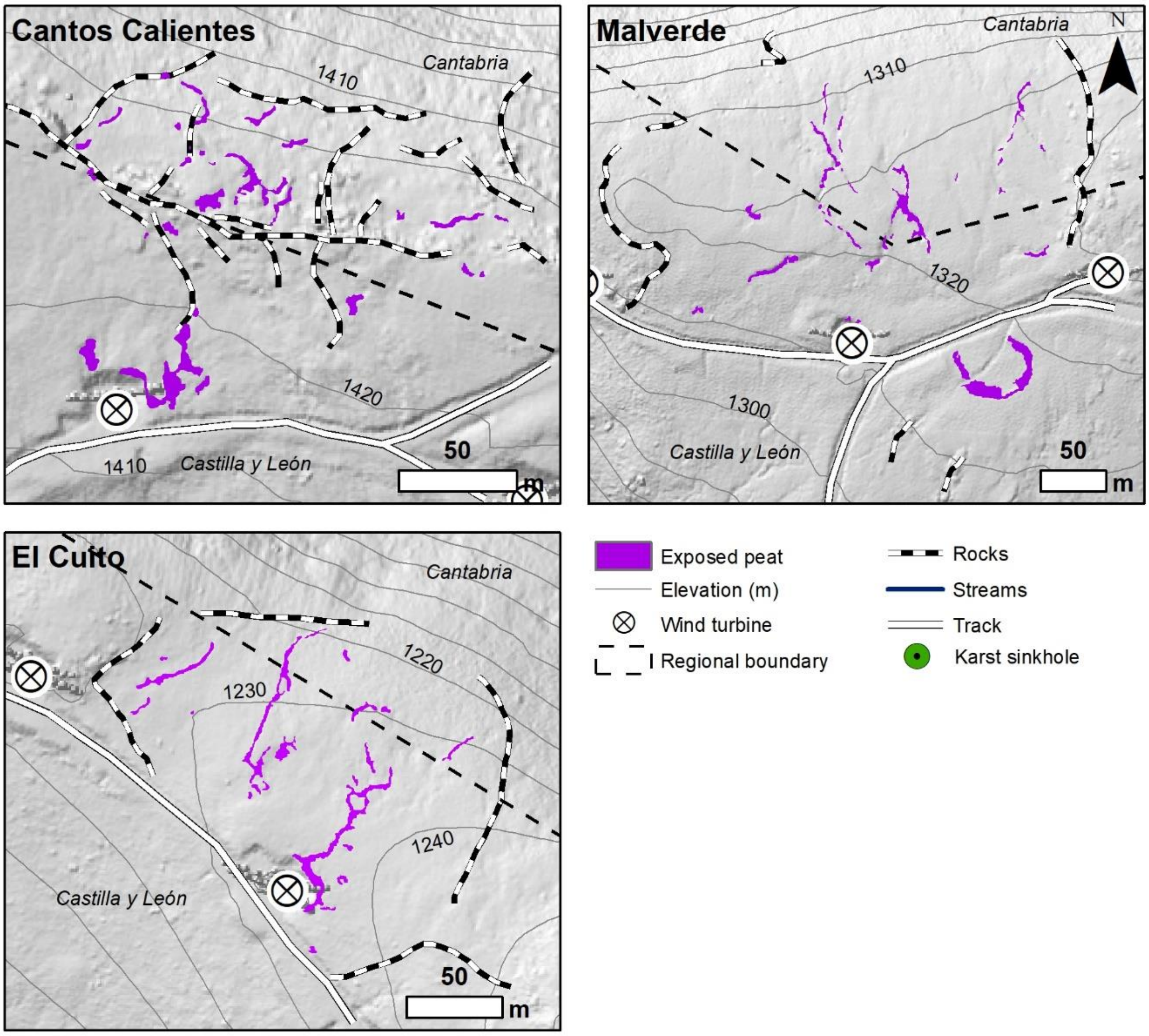


Figure B1. Exposed peat areas across all sites mapped from year 2017 aerial photography.

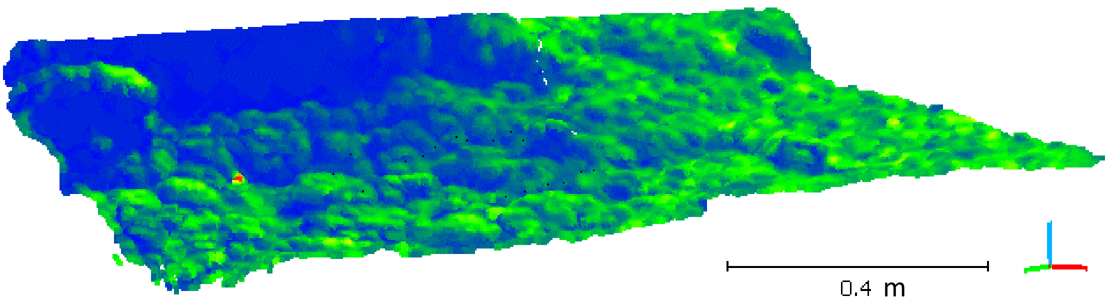
Appendix C

TLS point clouds by AOI

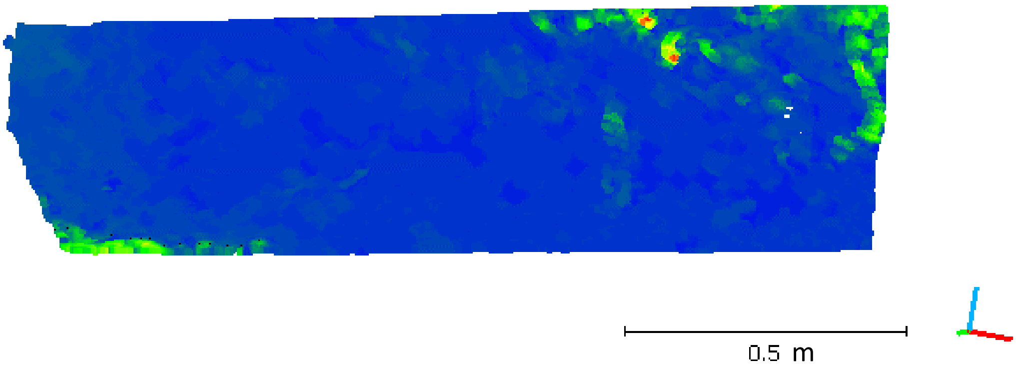
C.1. ZALAMA

All scales in metres

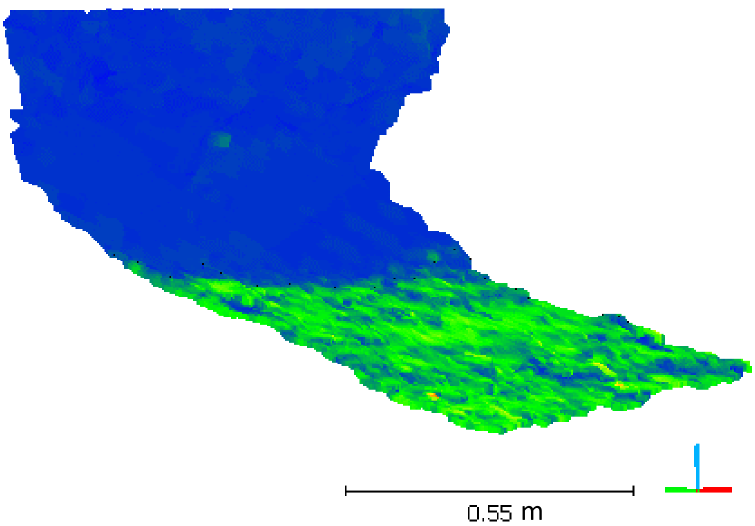
LZ1



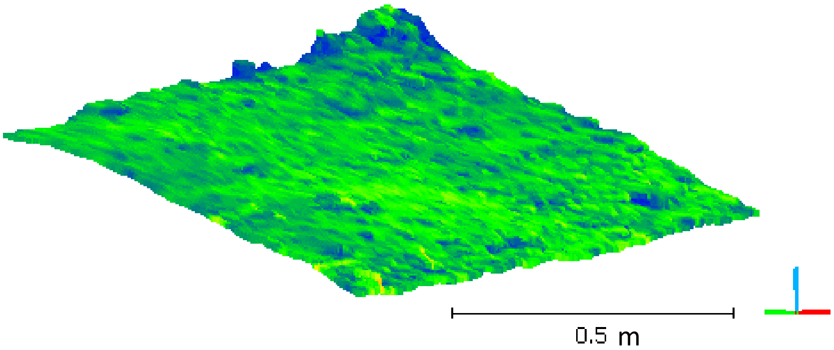
LZ2



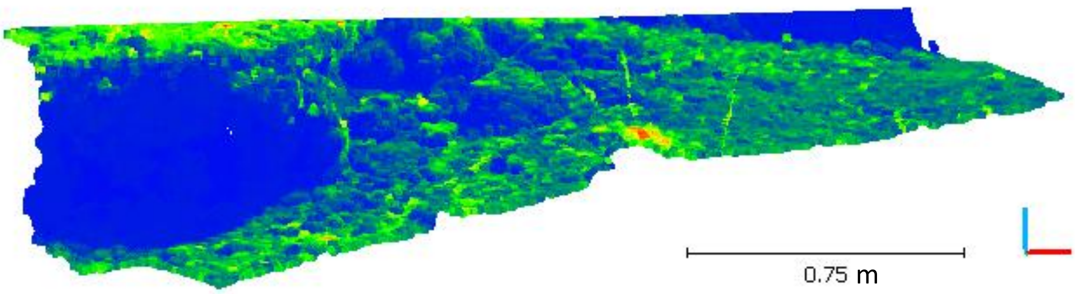
LZ3



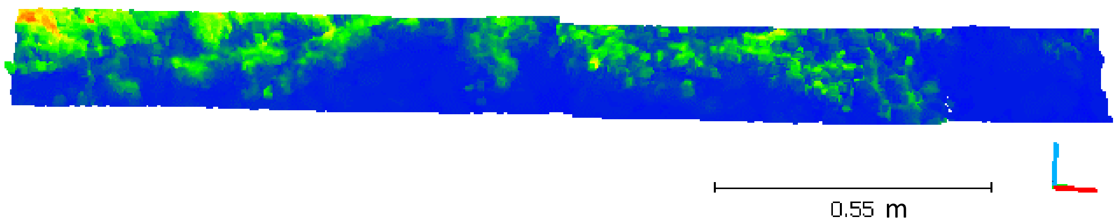
LZ4



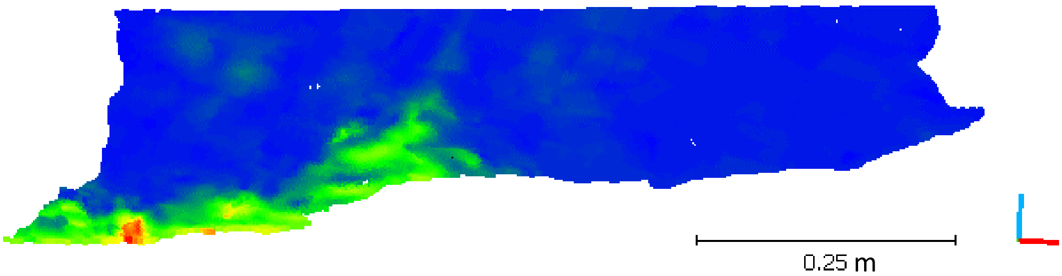
LZ5



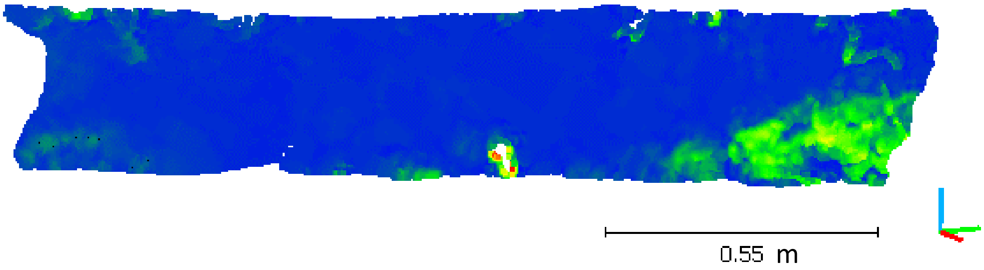
LZ6



LZ7



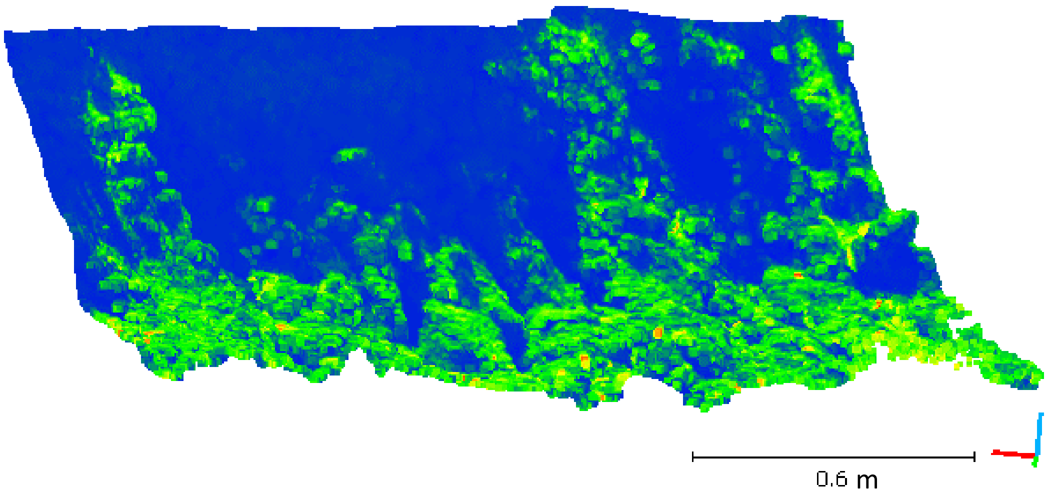
LZ8



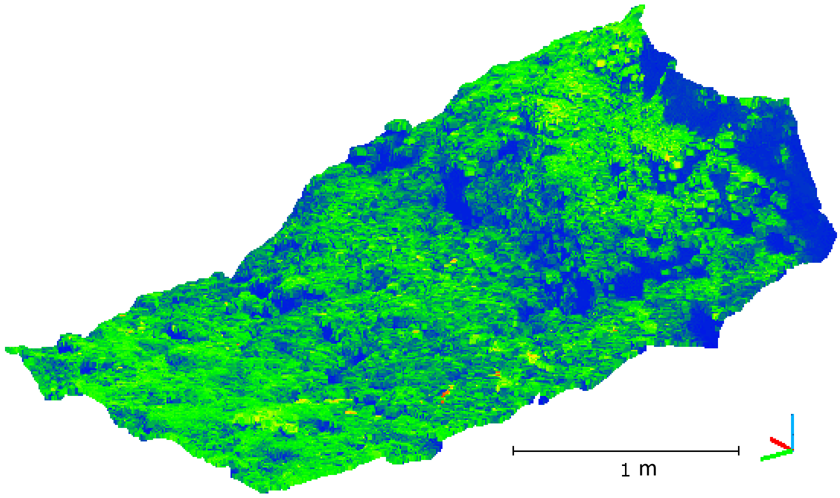
C.2. ILSOS DE ZALAMA

All scales in metres

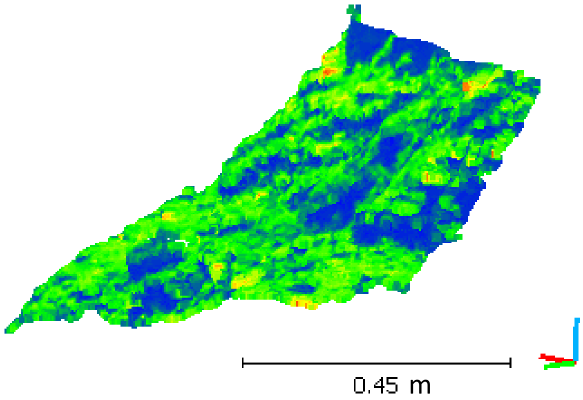
ILZ1



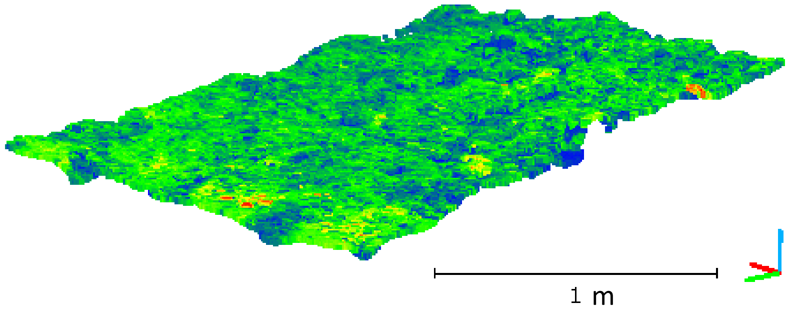
ILZ2



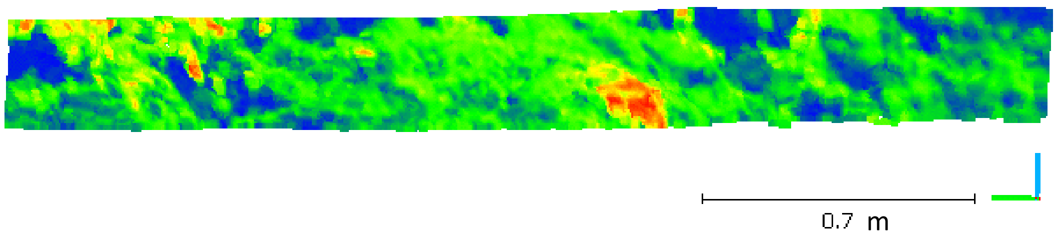
ILZ3



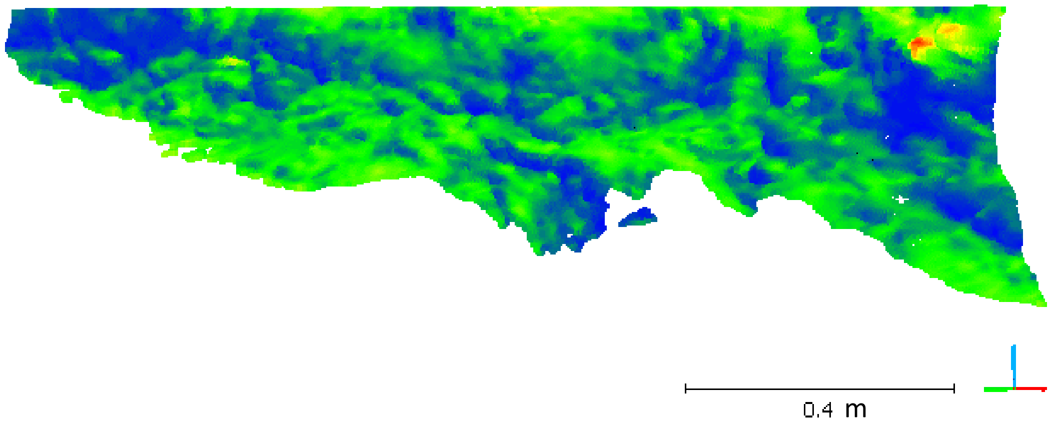
ILZ4



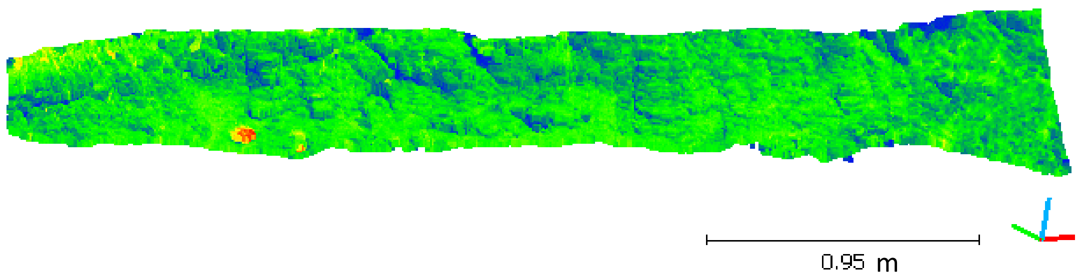
ILZ5



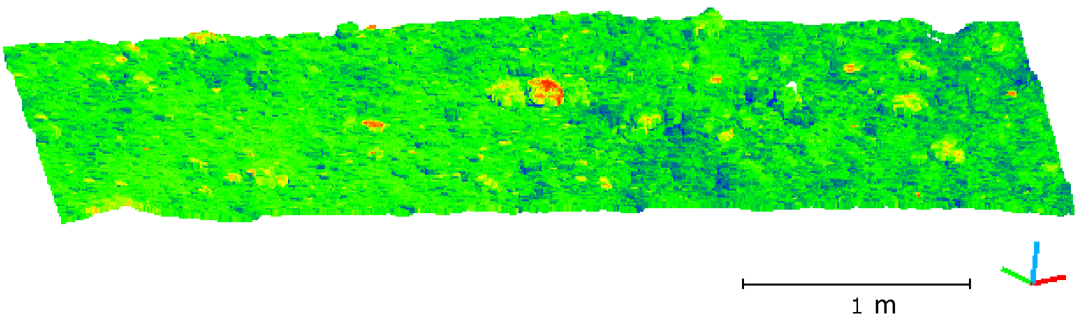
ILZ6



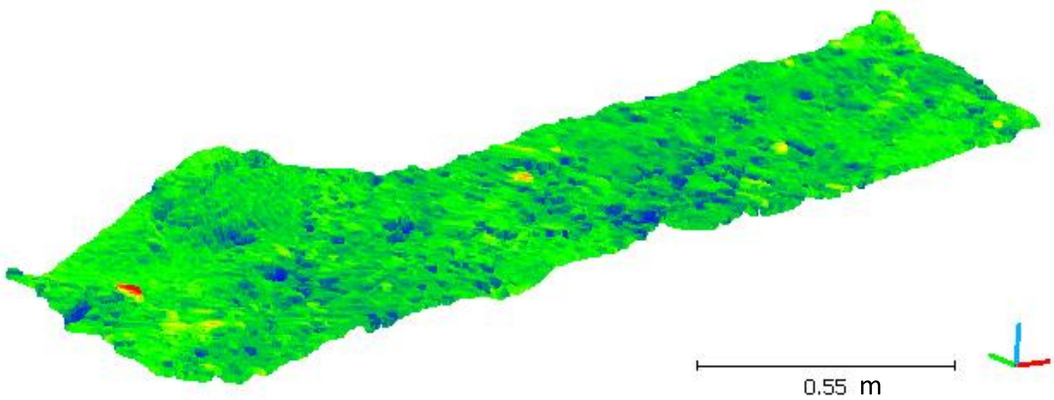
ILZ7



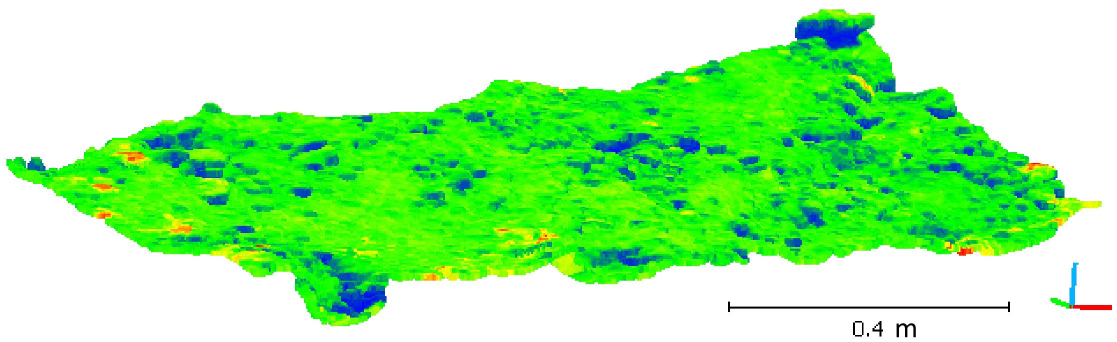
ILZ8



ILZ9



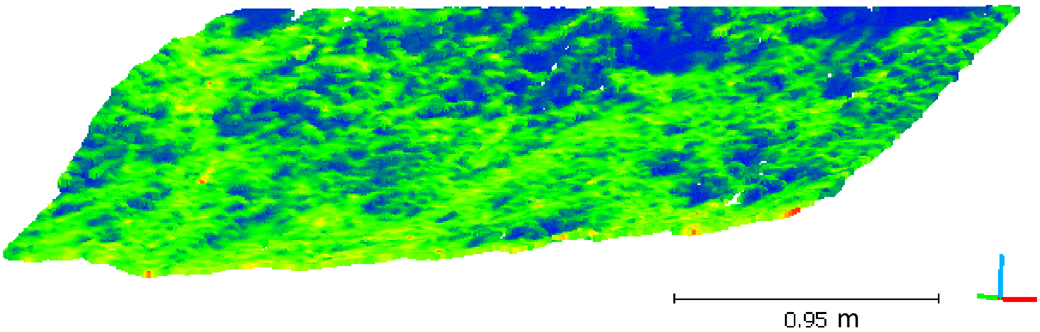
ILZ10



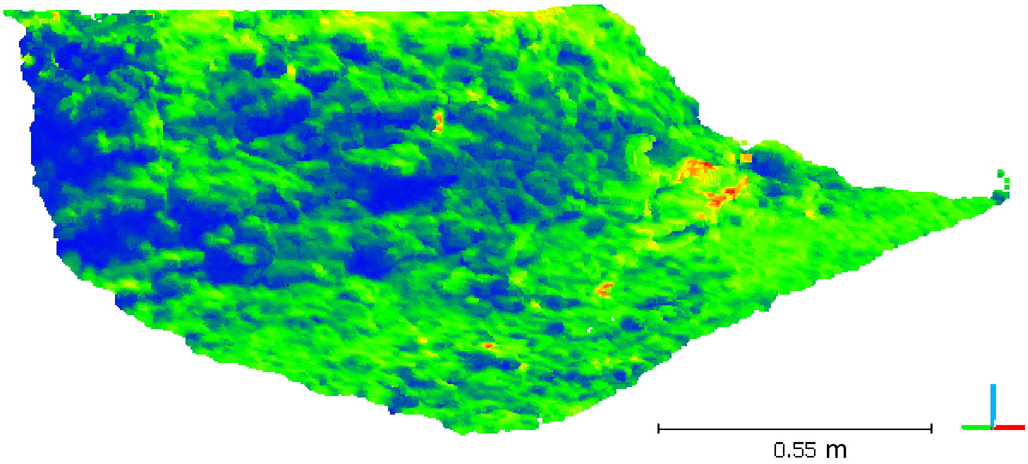
C.3. COLLADO DE HORNAZA

All scales in metres

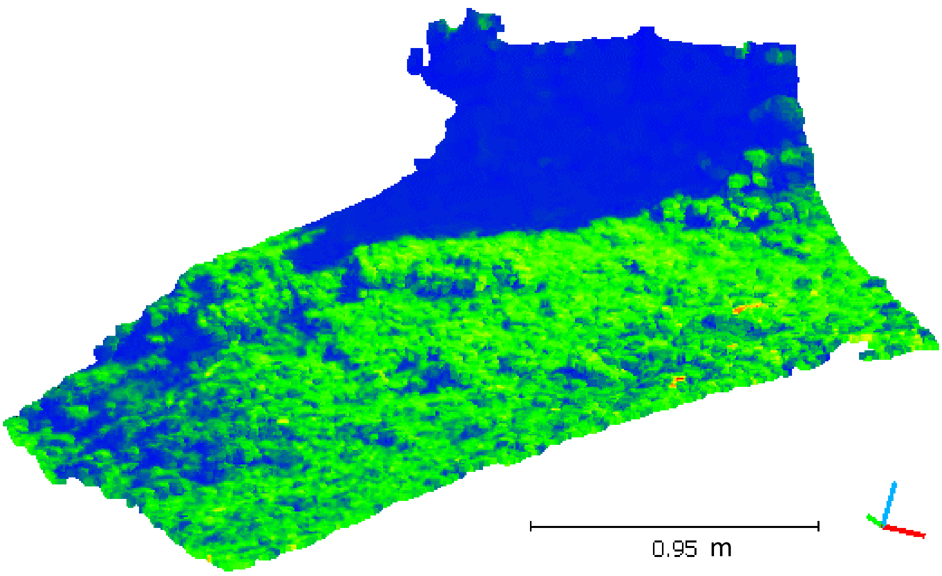
CH1



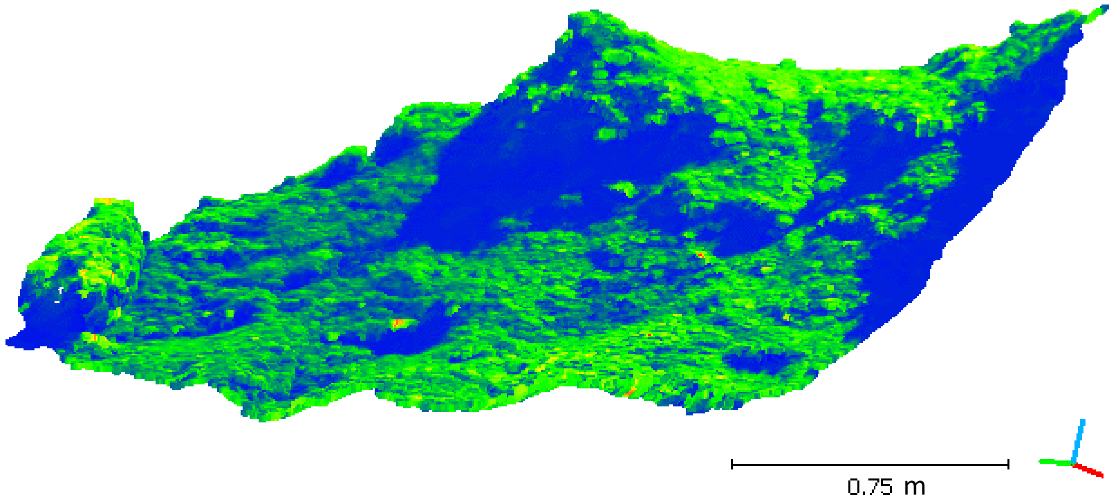
CH2



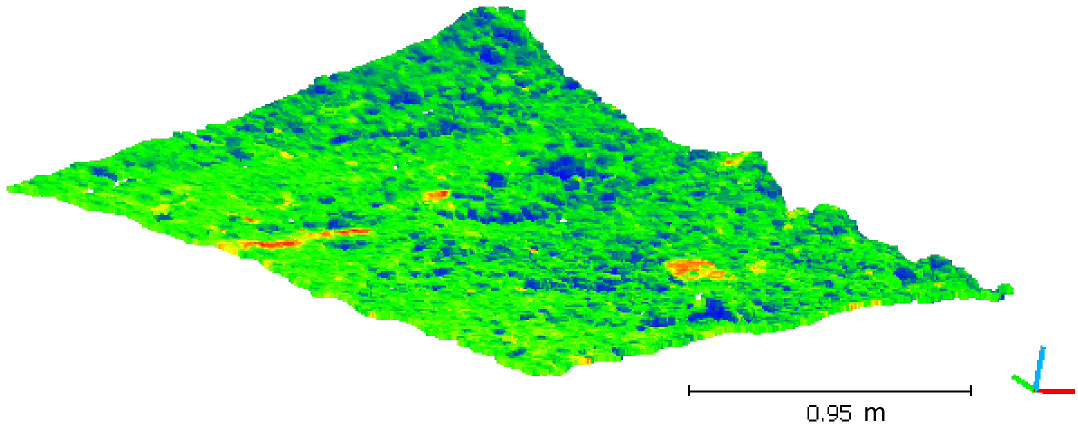
CH3



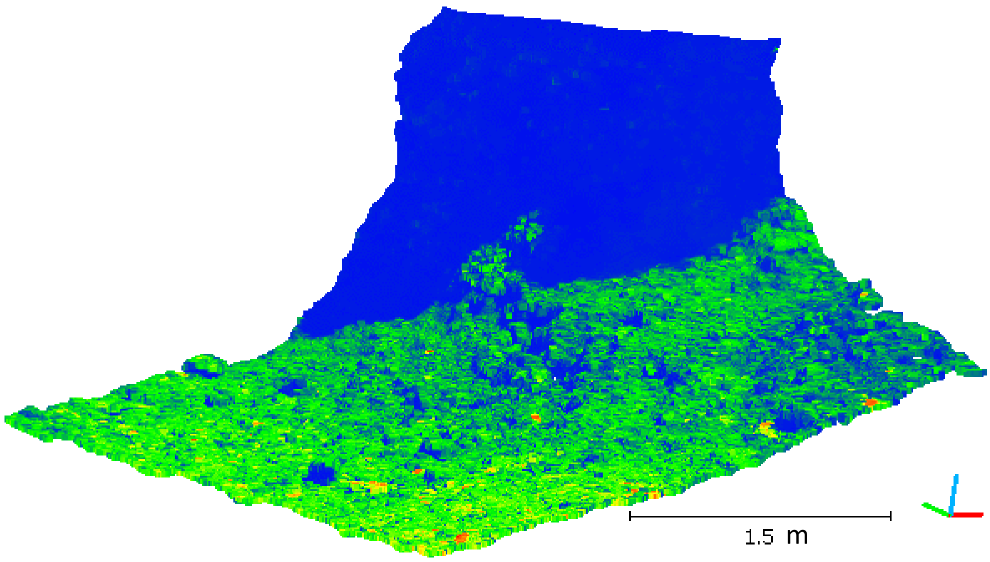
CH4



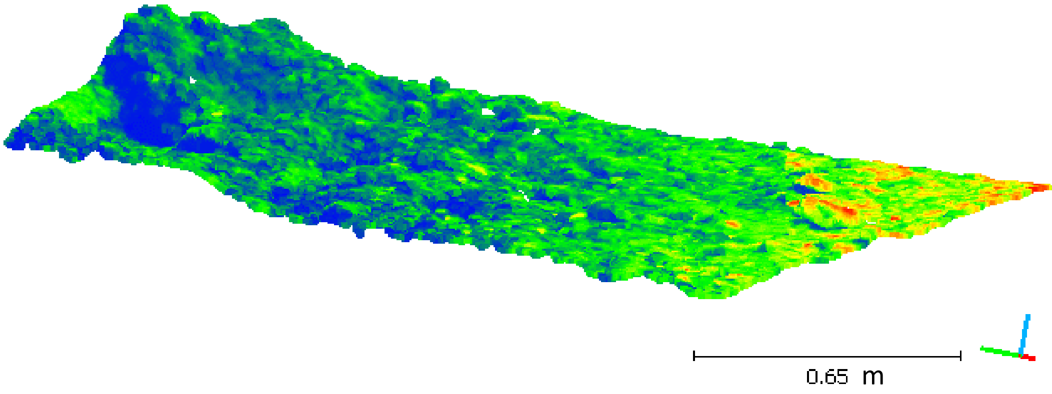
CH5



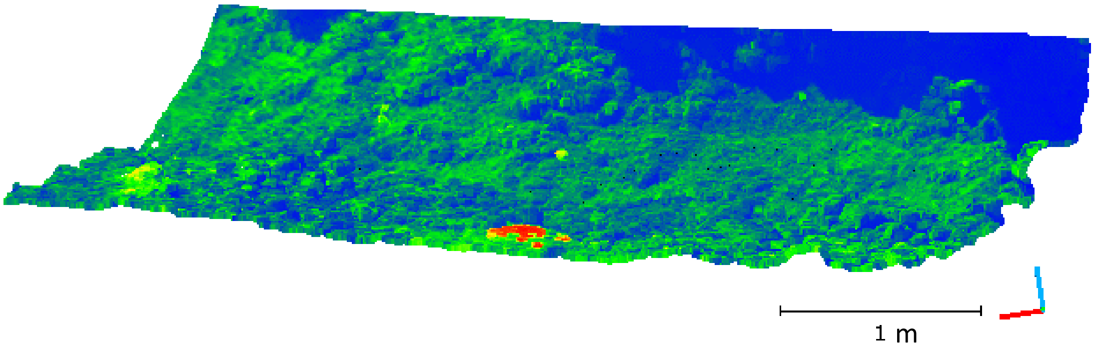
CH6



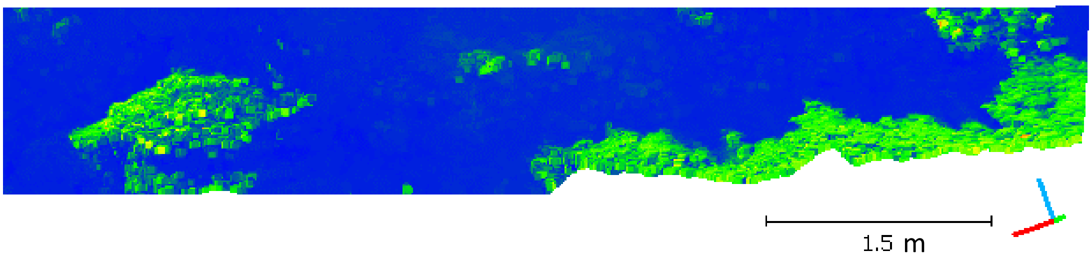
CH7



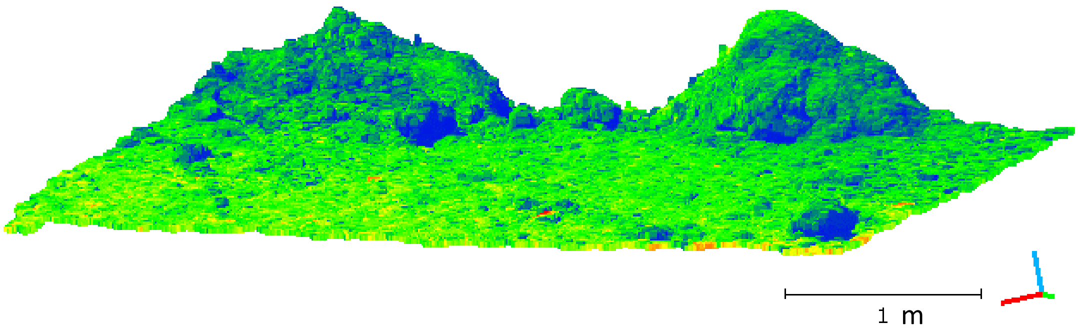
CH8



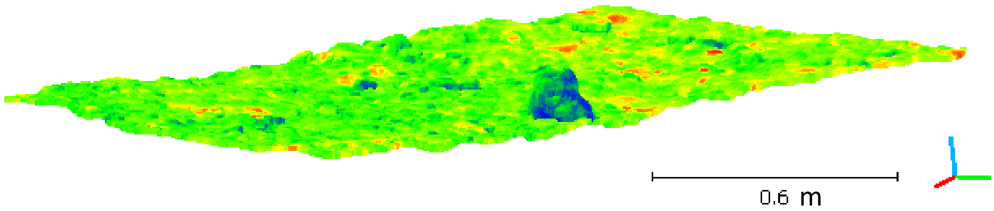
CH9



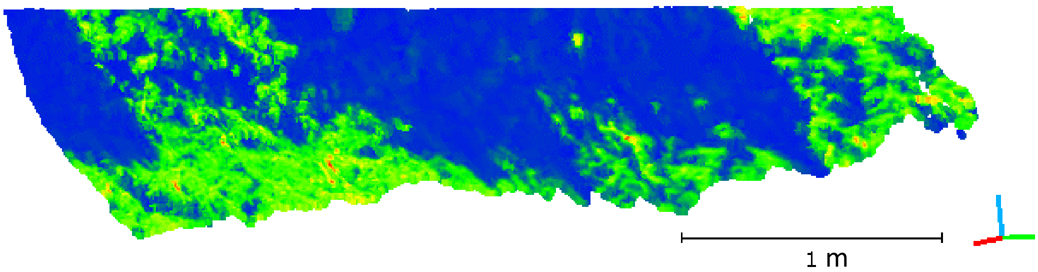
CH10



CH11



CH12



Appendix D

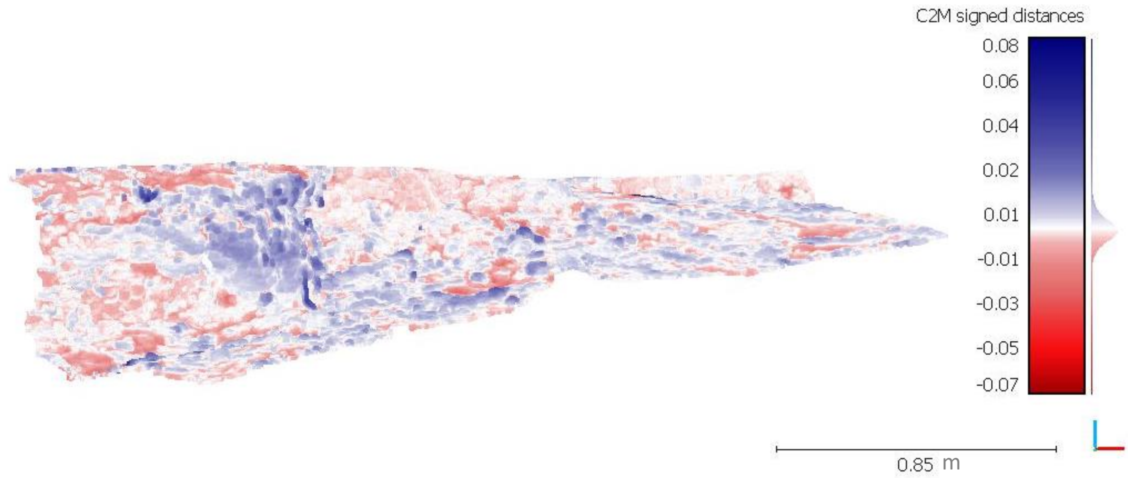
Example sequences showing multi-temporal surface changes

D.1. ZALAMA

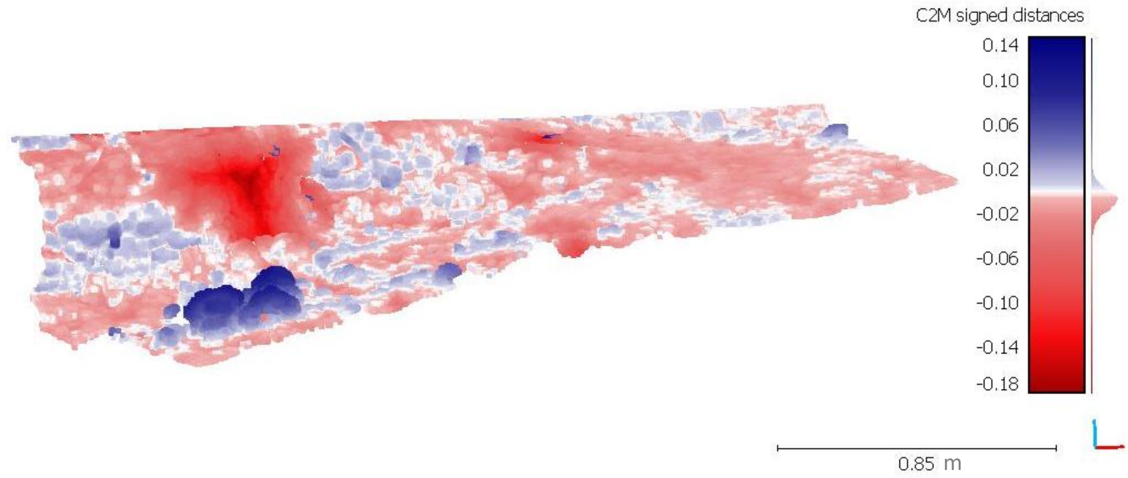
All values are presented on metres

Multi-temporal surface change at Zalama blanket bog for LZ5

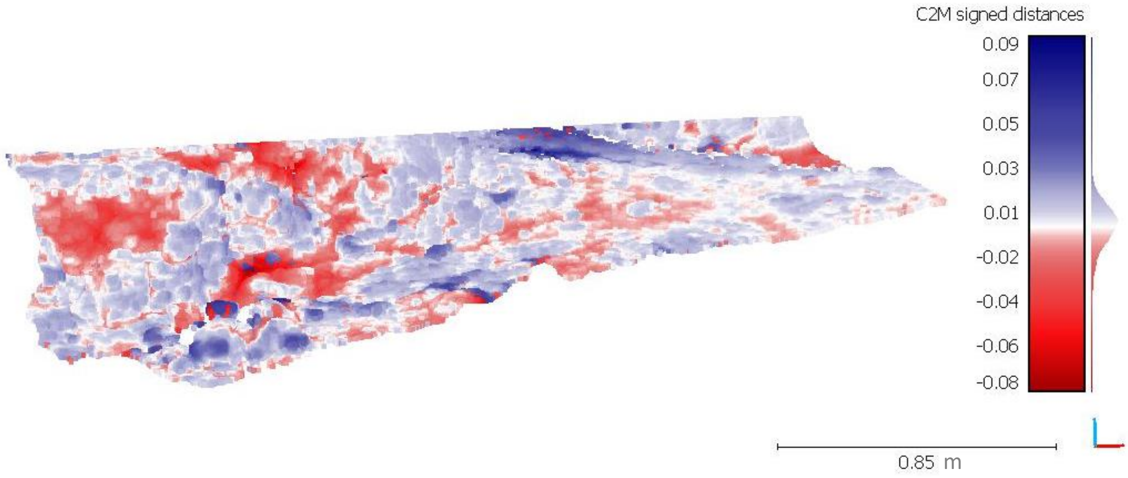
May 2017 to July 2017



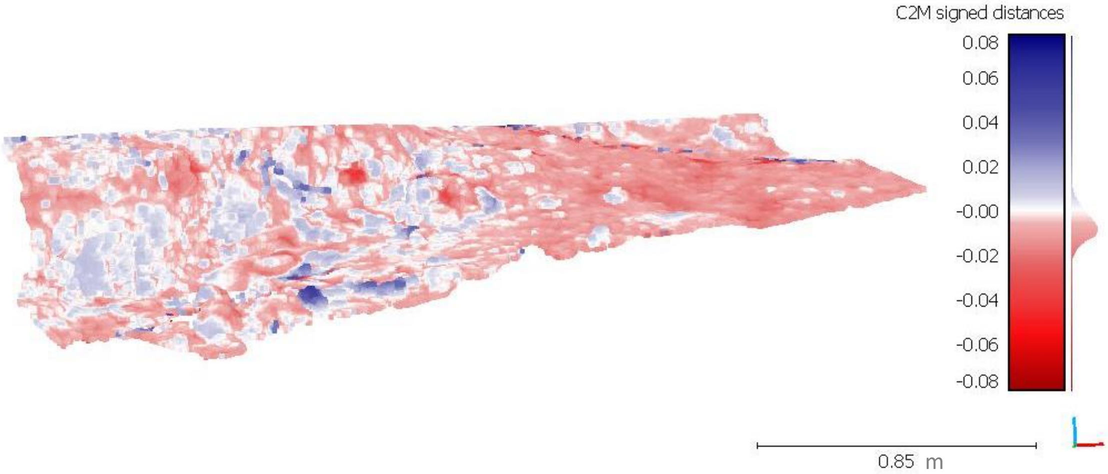
July 2017 to October 2017



October 2017 to April 2018



April 2018 to June 2018

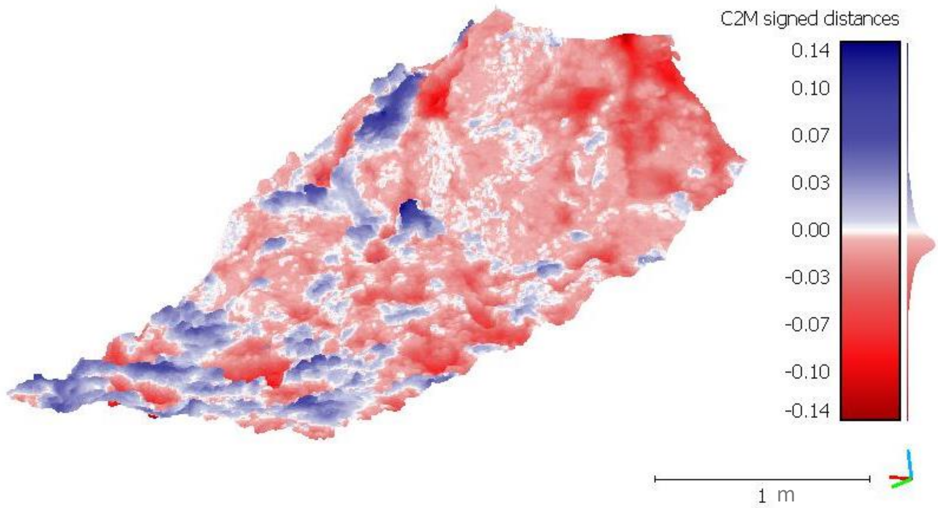


D.2. ILSOS DE ZALAMA

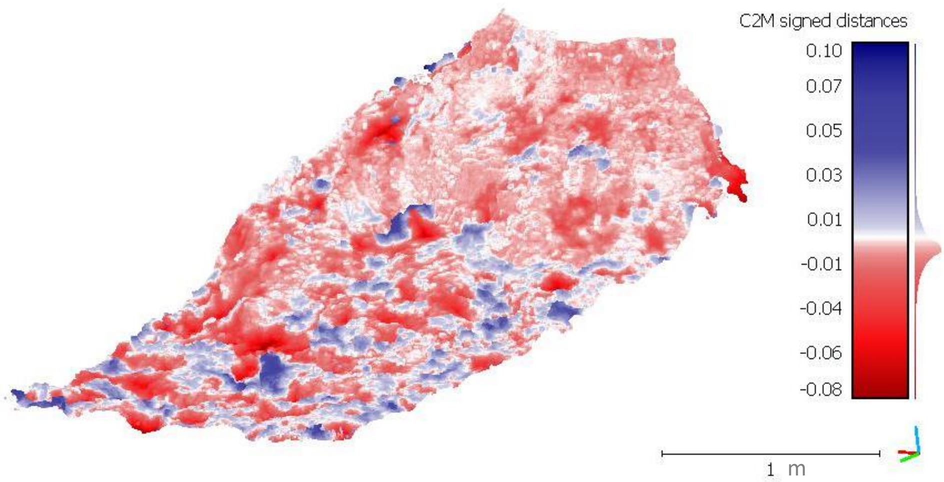
All values are presented on metres

Multi-temporal surface change at Ilsos de Zalama blanket bog for ILZ2

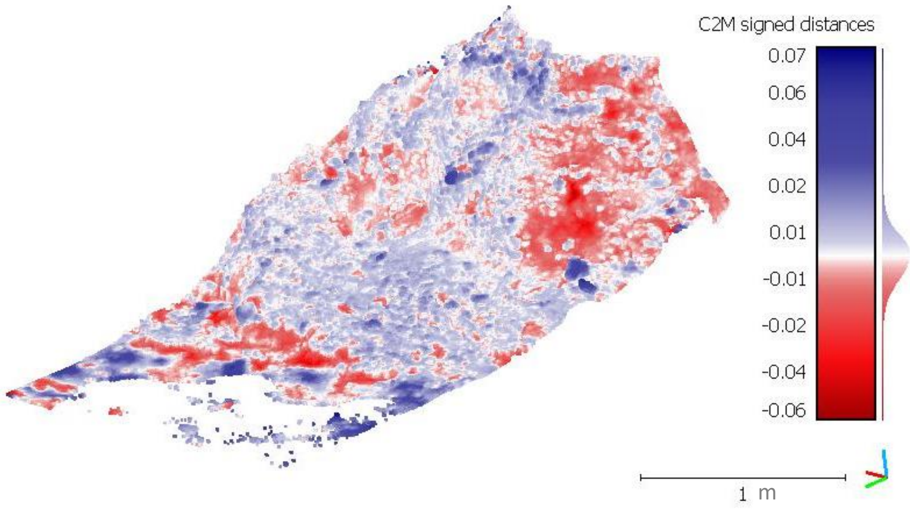
May 2017 to July 2017



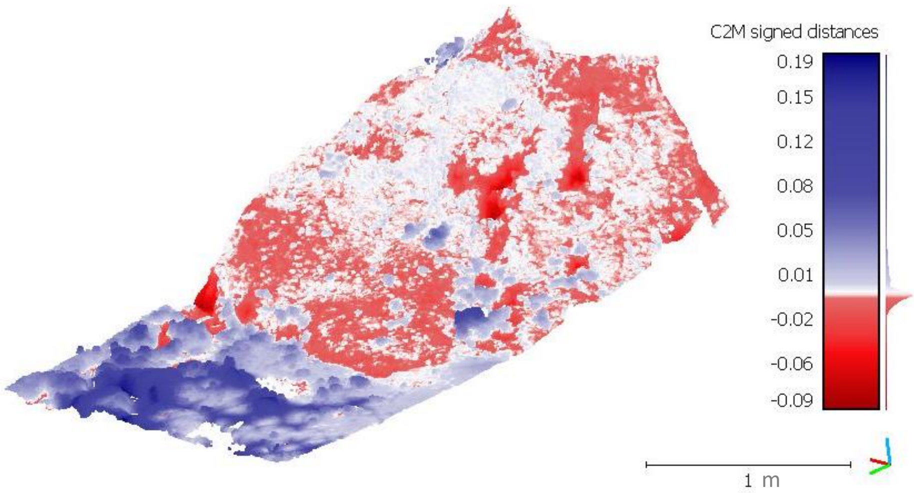
July 2017 to October 2017



October 2017 to April 2018



April 2018 to June 2018

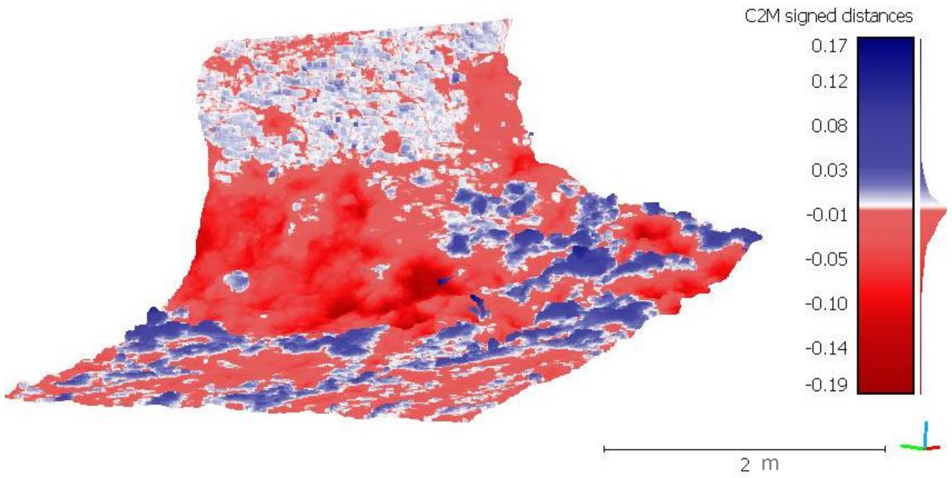


D.3. COLLADO DE HORNAZA

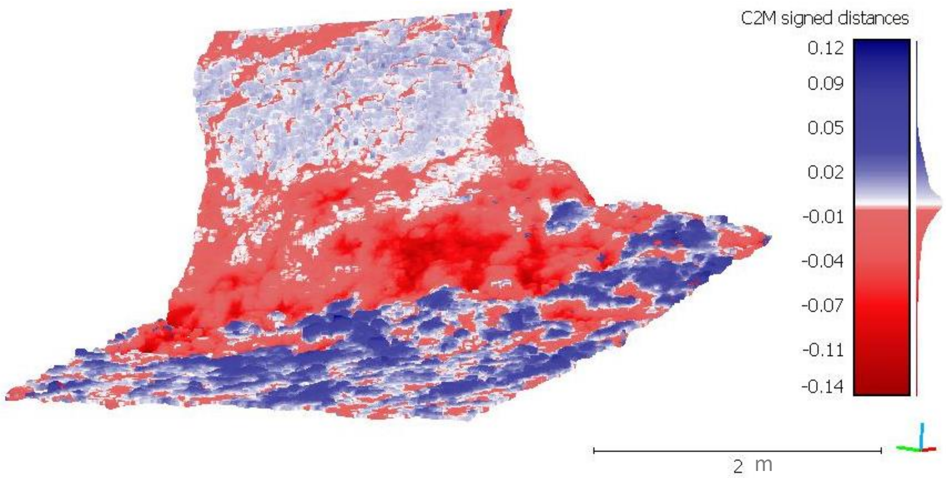
All values are presented on metres

Multi-temporal surface change at Collado de Hornaza blanket bog for CH6

May 2017 to July 2017



July 2017 to October 2017



October 2017 to June 2018