Impact of Rewetting of Irish Peatlands on Vegetation Composition and Biodiversity

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Submitted by

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Abstract

Peatlands are important wetland ecosystems with a global importance for the maintenance of biodiversity and climate regulation. However, land use changes and other human activities lead to widespread peatland loss and degradation with great negative impact on biodiversity and other ecosystem services. As a result, in Ireland only 15% of the extensive peatland area is still in a near intact state. Ecosystem restoration describes the recovery of ecosystem structure and function and aims to counteract degradation and to eventual re-establish a natural state. Due to its global importance, peatland restoration and conservation aspects became increasingly integrated into EU legislatives and international policies, including the United National Framework Convention on Climate Change (UNFCCC) or the EU Habitat Directive. Since peatland disturbances usually lead to drainage, rewetting is a fundamental and first-step approach in peatland restoration processes.

In this study, the habitat condition and vegetation composition of ten rewetted Irish bogs were investigated and compared via asking the following questions: (I) what is the current state of the different bogs in regard to vegetation composition, moisture condition and habitat heterogeneity? (II) Does rewetting impact vegetation cover and plant biodiversity? (III) How does this differ between bog types (raised versus blanket bogs) and prior types of degradation (only drained, afforested, cutaway and cutover)? It was found that rewetting in most cases indeed seems to have a positive impact on the moisture condition of the ecosystem and on the vegetation composition. The blanket bogs assessed here already showed a promising tendency towards a pristine state as did most of the raised bogs. However, special conditions could be detected for the cutaway raised bog Blackwater that still showed a high rate of disturbance followed by the afforested raised bog Sopwell. This was mainly reflected in a lower abundance of specialised bog species, especially *Sphagnum* mosses, poor habitat heterogeneity and/or a dry soil condition. The fact that in case of Blackwater a relatively good moisture level could be detected strongly indicates that successful rewetting alone does not necessarily lead automatically to a healthy bog and that the degree of degradation is probably an important factor as well. It can be concluded that rewetting is a crucial first step in a peatland restoration process, which might be in some cases even sufficient for triggering a self-regulation function of the ecosystem and finally a recovery. However, the combination of bog type and magnitude of degradation are worth to consider as these aspects could be crucial for estimating the rewetting success in a restoration process. Certain degradation states, such as cutaway raised bog, likely require more recovery time and maybe additional restoration methods. This study emphasises the general challenge we meet when dealing with the restoration of diverse ecosystems like

peatlands, the importance of long-term monitoring and the need for specific and flexible restoration approaches adapted to individual situations.

1. Introduction

Extensive exploitation and degradation in line with a constantly rising human population has led to ecosystem transformation, degradation and loss all over the world. Consequences include biodiversity loss, climate change and environmental pollution. In this context, consequences of biodiversity loss is gaining rising attention (Hector and Bagchi 2007; Convention on Biological Diversity 2010; Cardinale et al. 2012; Hooper et al. 2012). As a response of ecosystem loss and degradation, ecological restoration has become more and more important over the last years, which aims the recovery of disturbed ecosystems via establishing previous natural functions and structures (Vaughn et al. 2010; Kareksela et al. 2015). If and in how far it is possible to regain a pristine ecosystem state, strongly depends on the ecosystem complexity as well as on knowledge and good practice of ecologically effective restoration approaches (Cortina et al. 2006; Pocock et al. 2012). Among the various ecosystems which are target for exploitation and restoration, peatlands are of global relevance.

1.1. Peatlands – an introduction into the story of extraordinary ecosystems

Peatlands are important wetland ecosystems with exceptional features and they are major players in terms of biodiversity maintenance and climate regulation in a global context. There are various peatland types which differ depending on geographic region and vegetation type, but all of them are characterized by the accumulation of incomplete decomposed organic matter, called peat, derived from dead plant material under permanent condition of high water saturation (Parish et al. 2008). Hence, the formation of peat is only possible under specific climate conditions, including high precipitation rates relative to evaporation and a high production of plant material exceeding its decay. Limited decomposition is only possible under water excess and thus, healthy peatlands always show a high water table at least close to the surface (Parish et al. 2008). Peatlands represent a significant amount of global wetland resources and are present in almost every country; however, due to their dependence on appropriate climate conditions they occur mainly in tropical, boreal and subarctic zones. The wide geographic distribution leads to a large diversity of peatland types, whereby a rough but feasible separation can be made between bogs (generally rain-fed and therefore nutrient-poor) and fens (additionally fed by surface or ground water and consequently more nutrient rich) (Parish et al. 2008).

In Ireland, where this study is based on, essentially three different types of peatlands occur: raised bogs, blanket bogs and fens (Hammond 1984). As this study deals with Irish raised and blanket bogs only, henceforth the term *peatland* will refer to bogs solely.

1.2 The ecological functions and services of peatlands

All ecosystems on earth, including peatlands, are characterised by an individual set of ecological functions, which arise from their specific underlying structures and processes (Renou-Wilson 2015). Peatlands show a number of ecological functions, including carbon storage, habitat support, water filtration, soil formation, preservation or peat supply. All these functions further provide several services or benefits; for the sake of the broader environment in general, but some of them also particularly connected to human benefits only. The list of ecosystem services provided by peatlands is long. It includes for example important regulating or supporting services, like climate regulation, nutrition cycling or maintenance of biodiversity. However, also aspects like recreation and wellbeing, cultural tradition, the delivery of historical data or peat as a source of energy or horticulture are services provided by peatlands and consumed by humans (Renou-Wilson 2015). Furthermore, unoccupied peatland areas provide useful space for other human purposes, like agriculture, forestry or wind farm construction.

When looking at ecosystem functionality in general, biodiversity plays a particularly important role, as species composition and functional traits highly influence ecosystem structure and functions (Reich et al. 2004; Hooper et al. 2005). As a consequence, biodiversity can be considered as a response variable affected by environmental changes, but also as a factor directly impacting all ecosystem services as well as humans (Millennium Ecosystem Assessment 2005).

1.3 The importance of peatlands for biodiversity

Peatlands are of global importance for biodiversity conservation on all levels. They represent markedly heterogenic ecosystems and offer various different microhabitats, resulting in a unique and highly specialised flora and fauna. Due to the harsh conditions, species diversity is usually lower compared to other ecosystems in the same biographic region. However, pristine peatlands therefore hold more rare and threatened species, which are especially adapted to the water-logged, acidic and nutrient poor soil conditions. Furthermore, peatlands play an important role of providing temporary or refuge habitat to dryland species, for example via serving as breeding ground or providing food resources (Parish et al. 2008). The relevance of peatlands on biodiversity is not only reflected within the boundaries of the ecosystem since they also strongly influence hydrology and microclimate of surrounding areas. A peatland's biodiversity is directly linked to climate change via significant feedback relationships: biodiversity does not only strongly *depend on* climate, it in turn also *influences* climate in a global scale as the vegetation dictates peat-formation and therefore the carbon storing process. Due to the excellent preservation conditions in peatlands, they further provide information platforms on past biodiversity and linked climate conditions (Parish et al. 2008).

Vegetation taxonomies found in peatlands reach from algae over lichens to bryophytes and vascular plants, whereby the respective contribution varies widely depending on geographical location, bog type and health state. Peatlands usually provide particularly suitable conditions for mosses and liverworts and hence, play a significant role in maintaining bryophyte diversity. In this regard, *Sphagnum* mosses play a particular important role for bogs as they are the most important peat-forming plants and therefore good indicators of a peatland's condition (Parish et al. 2008).

1.4 Peatland disturbance and the role of human impacts

The strong interconnection of water, vegetation and peat, makes peatlands to particular delicate ecosystems. Peat formation is dependent on vegetation, which itself is determined by hydrology (and other abiotic characteristics). Water flow, in turn, is based on decay rate and peat structure. The disturbance of one of these components, even if only happening in small scale, disrupts the natural balance of the system and enables a chain reaction that sooner or later will lead to transformation of the other components as well. Consequently, human activities have strong impacts on the general health state of bogs and its biodiversity (Ivanov 1981; Davis et al. 2000; Parish et al. 2008).

The list of benefits people can obtain from peatlands is huge, and so is the utilization rate. Drainage for agriculture and forestry, grazing and peat extraction as well as land clearing, urban development or just environmental pollution are typical human activities that disturb the natural balance of peatlands. Almost all of them are directly or gradually associated with water loss. Agriculture, forestry and urban development can only happen on relative dry soils, which can be achieved via the installation of drainage ditches that lower the water table (Renou-Wilson et al. 2011b). Other activities, like peat extraction will gradually lead to drying out of the peat surface, due to vegetation loss, decreased water storage capacity and the exposure of bare peat to the surface. In other words, the bog is drying out, with far reaching consequences for climate and biodiversity.

Drainage alters a peatlands' habitat condition by changing hydrology of the peat soil and causing eutrophication of nearby waterbodies (Turner and Haygarth 2001) via releasing nitrogen and phosphorus compounds from oxidizing peat. It results in an increased productivity and a shift from a previously bryophyte dominated wetland vegetation towards a shrubbier environment and the growth of trees, which further leads to decreased light ability and a decline of low succession wetland vegetation (Hedberg et al. 2012). Changes in hydrology towards drier conditions further lead to the loss of soil volume and subsidence, rising decay rates and net greenhouse gas (GHG) emissions as well as habitat homogenisation (Parish et al. 2008). Human induced land use change practices lead to alteration, fragmentation or even loss of peatland habitats, and often the consequences are irreversible. This has dramatic impacts on the natural biodiversity since adapted bog species cannot compete under changing conditions and might become displaced by more generalised species. Due to the tight interconnection in peatland communities, the loss of one species very likely results in subsequent losses of other species, thereby completely shifting the ecology of the ecosystem (Parish et al. 2008). Consequences on biodiversity are not limited to the bog alone, though. Also surrounding areas suffer since an alteration of peatland hydrology also lowers the ground water level in the surroundings. Habitat fragmentation, as a result of urban constructions for example, peat extraction or drainage prevents genetic exchange and recolonisation of surviving populations (Parish et al. 2008). Furthermore, human induced changes can lead to the occurrence of invasive species, especially those providing more efficient nitrogen fixation abilities than nitrogen fixing bog species (Minayeva and Cherednichenko 2005).

When looking on the type of land use change on global scale, agricultural practices bear the biggest blame for peatland loss (Parish et al. 2008). In Ireland, however, the most important issues of peatland disturbance have been domestic and industrial peat extraction, afforestation, wind farm establishment, recreation activities, the introduction of invasive species as well as climate change (Renou-Wilson et al. 2011a). Afforestation surely shows one of the largest impacts on peatland biodiversity, as it includes activities like tree planting, active drainage or the introduction of systems and infrastructure for timber transportation. Other land use practices like cattle or sheep grazing, does not necessarily lead to the destruction of a peatland's biodiversity, if managed in an appropriate way generally including low stocking rates. However, bad management and intensive grazing with large numbers of cattle, as it has been caused by the formerly EU Headage Grand Scheme, can impact the ecosystem heavily (Parish et al. 2008; Renou-Wilson et al. 2011b).

It is noteworthy, that caution should be taken if judging a bogs' health state on the basis of biodiversity or species richness. It needs to be distinguished between natural and general diversity. As mentioned previously, peatlands usually show a lower biodiversity than mineral soil habitats of the same ecoregion, but include very rare and specialised species. Peatland disturbance therefore often leads to an increase in general *common* diversity but a decrease of natural bog species (Parish et al. 2008; Renou-Wilson et al. 2011a).

1.5 Peatland restoration: The impact of rewetting on biodiversity

Peatlands are extremely vulnerable to any kind of disturbance and human impact has already let to the sad outcome that in Europe approximately 60% of previous natural peatlands have been transformed, whereby agriculture was the main reason (Joosten 1997). In order to fight the consequences on climate and biodiversity, peatland restoration became an important issue that aims to re-establish a balanced and naturally functioning mire ecosystem that possesses the ability for peat formation (Wheeler and Shaw 1995). Generally, there is a hierarchical order in applied restoration methods (Gorham and Rochefort 2003). Reintroducing and stabilising a high water table close to the peat surface level is necessary to restore the characteristic soil properties and therefore usually represents the first step of a long process (Vasander et al. 2003). It is also a premise for any subsequent recolonisation of original bog vegetation (Gorham and Rochefort 2003). Typical measures to increase the water table in drained peatlands vary depending on the situation and include ditch blocking, damming or - in case of previous afforestation – felling of trees. In Ireland this refers mainly to non-native conifer species, which had been planted because they can deal with harsh growing conditions. After felling, material can be either completely left on site or removed (Coillte 2011-2014). All these techniques aim to re-transform a drained soil into a wet soil, a process called rewetting. The restoration of drained former wetlands always includes rewetting as a compulsory step (IPCC 2013).

Several studies on boreal peatlands previously drained for forestry, including bogs and fens in Scandinavia, recorded a positive result on wetland vegetation composition development and water table rise after rewetting via ditch filling and tree cutting (Jauhiainen et al. 2002b; Haapalehto et al. 2011a; Hedberg et al. 2012). This included a fast decline of forest species (Jauhiainen et al. 2002b), increased abundance of *Sphagnum* species and a gradually increasing species richness (Hedberg et al. 2012) after restoration and a concentration of mineral elements comparable to pristine peatlands ten years after restoration, suggesting a recovery of nutrient cycling (Haapalehto et al. 2011a). In Ireland, drain blocking with peat damns had been successful in raising the water table to an appropriate level for instance in Killamuck Bog, a raised bog drained for peat extraction, and allowed a subsequent maximizing of biodiversity (Renou-Wilson et al. 2011a). This management approach was considered as being very promising.

Generally, ditch damming is a common restoration method after drainage, as it leads to the creation of pools behind the drains. However, these artificial pools are dependent on drain dimensions, quite similar in size and depth and therefore significantly different from natural pools. In this context, concern had been raised on likely ecological consequences, which are poorly understood. However, there are evidences suggesting that the creation of different sized pools (by manipulating the physical dimensions of the drains) could positively impact the community assemblage of aquatic biodiversity (Beadle et al. 2015).

Peatland restoration does not always lead to a successful final result, as the success depends on several factors. For instance, the time that has passed since drainage plays an important role (Laine et al. 1995). The less time, the more likely it is that restoration can help the ecosystem to recover. The longer the bog has been drained, the more critical it gets (Heikkilä and Lindholm 1997). Furthermore, achieving appropriate water conditions via rewetting is not trivial. Peat subsidence caused by drainage and usually most pronounced close to ditches, induces an erratic water spread across the landscape, which rather results in a mosaic of drier and wetter areas than a consistently high water table (Minkkinen and Laine 1998; Vasander et al. 2003). Furthermore, it is important that a sufficient peat amount is still available. If this is not the case the basis for an effective restoration might not be given anymore (Renou-Wilson et al. 2011b). Other studies indicated that landscape related factors might be more important for vegetation succession than abiotic soil features (Salonen and Setala 1992; Rehounková and Prach 2006; Konvalinková and Prach 2014). In this regard, the proportion of healthy and vegetated peatland area adjacent to a disturbed bog might be important for a restoration process (Campbell et al. 2003; Konvalinková and Prach 2014). Generally, the recovery of a peatland is a very protracted process and success or failure of a certain restoration technique may not be discovered in the early stage. Hence, long-term monitoring is a must (Haapalehto et al. 2011a).

There is no master plan for the restoration of degraded peatlands as each site is unique in terms of its location, site condition and type of disturbance. Consequently a flexible and adaptive management approach is necessary depending on the situation. In any case successful restoration requires the setting of realistic goals and, depending on that, an appropriate combination of processes leading to the recovery of desired peatland functions and (Renou-Wilson et al. 2011b). As restoration of degraded peatlands is also an artificial interference into natural processes and has impacts on the remaining biodiversity or other ecosystem functions, restoration projects always need to be planned and conducted carefully to avoid further losses (Parish et al. 2008). Indicators, such as protected species like the *Sphagnum* genus or marsh saxifrage (positive compositional indicator), habitat heterogeneity (positive functional indicator) or vegetation loss and tree growth (negative indicator) are appropriate to estimate and monitor a peatland's health stage and according to the BOGLAND report should be used for assessments of peatlands in Ireland (Renou-Wilson et al. 2011b).

1.6 The integration of peatland conservation and restoration into policy

In Ireland, approximately 20% of the total land area consists of peatlands (Connolly and Holden 2009), with an estimated carbon storage capacity of 1064 to 1503 gigatonnes (Tomlinson 2005; Eaton et al. 2008) and a unique fauna and flora, including many threatened and protected species (Renou-Wilson et al. 2011a). Unfortunately, most of this area is in a disturbed state, caused by human induced exploitation and transformation. As a consequence, ecosystem functions, including biodiversity, hydrology and carbon cycling, are heavily affected in more than 80% of the national peatland area (Renou-Wilson et al. 2011a). Hence, an integration of peatland conservation and restoration aspects into national and international policies is highly important.

From a national perspective, the Irish Peatland Conservation Council (IPCC), an independent not-for-profit organisation founded in 1982, takes a major responsibility of peatland conservation and protection (IPCC website). Furthermore Ireland's government published a decision catalogue on a national peatland strategy (NPS), dealing with conservation and management aspects of peatlands in the country, especially for those areas nominated for designation as Special Areas of Conservation and Natural Heritage Areas. The aims of this first NPS are meant to be consistent with international agreements, EU legislations as well as climate change regulations (NPS DRAFT 2014).

International agreements considering peatland conservation include the Convention on Biological Diversity (CBD), Ramsar and the United National Framework Convention on Climate Change (UNFCCC). The CBD, a binding agreement signed in 1992, charges its member countries to develop and implement the protection and sustainable management of biodiversity. In this regard it is highly affecting the use and management of peatlands as a key ecosystem (Rieley and Lubinaite 2014). Strong implication on peatland utilisation in relation to climate change has the UNFCCC, which overall aims to stabilise GHG concentration and which binds developed countries to individual emission reduction targets via the Kyoto protocol. The relevance of peatlands in this context is based on the emissions associated with peatland use and degradation that significantly contributes to national GHG stocks since drainage leads to an increase of CO_2 emissions (Rieley and Lubinaite 2014). In this context, the Durban agreement led to the inclusion of the wetland drainage and rewetting (WDR) category in the LULUCF (**L**and **U**se, **L**and **U**se **C**hange and **F**orestry) activities under the Kyoto Protocol – Article 3.4. That means that rewetting of peatlands can be included on a voluntary basis in national greenhouse gas accounting (Rieley and Lubinaite 2014; UNFCCC 2014). Furthermore, peatland conservation and restoration is integrated into several EU legislations, including the Habitats Directive, the Water Framework Directive and Floods Directive (NPS DRAFT 2014). In this regard, the Habitats Directive in combination with the Birds Directive somehow provides the foundation of EU conservation policy as it protects over 1000 species (Annex II-V) and more than 200 important habitat types (Annex1) including bogs, fens and mires (Council of European Communities 1992).

1.7. Aim of this study

This study aims to evaluate the current state of ten different rewetted Irish bogs by the investigation of habitat condition and vegetation composition with the overall objective to improve the understanding of rewetted and restored peatland ecology. As part of the NEROS (Network monitoring rewetted/restored peatlands and organic soils for climate and biodiversity benefits) project, this study addresses the following questions: (I) what is the current state of the different bogs in regard to vegetation composition, moisture condition and habitat heterogeneity? (II) Does rewetting have a positive impact on vegetation cover and plant biodiversity (in comparison to a pristine landscape)? And (III) How does this differ between the bog types (raised bog versus blanket bogs) and the prior type of degradation (only drained, afforested, cutaway and cutover)?

2. Materials and Methods

2.1 Study sites

The study is based on ten different degraded and rewetted bogs, listed in **table 1**. The study sites are distributed mainly in the Midlands and North-West of Ireland (**figure 2**) and differ in type (raised or blanket bogs), disturbance prior restoration (drained only, cutaway, afforested or cutover) as well as restoration method (damming, pump stopped or felling). Depending on the ecoregion, the study regions overall show annual mean temperatures ranging from 9.3 °C to 9.8 °C and a mean monthly rainfall between 846 mm in the West-Midlands up to 1174 mm in the South-Midlands (Met Éireann Station Birr, Claremorris, Malin head or Clones, respectively – 1981-2010).

2.2 Sampling design

In order to investigate the condition of the bogs in response to rewetting, a peatland monitoring restoration programme had been constructed in line with the NEROS (Network monitoring rewetted/restored peatlands and organic soils for climate and biodiversity benefits) project (ucd.ie/neros). This project and connected monitoring system is complex and includes data on abiotic, biotic and management variables. This study focuses on the part dealing with peatland vegetation, in terms of species and composition, as well as the general habitat condition, especially heterogeneity and wetness. These aspects represent important indicators of the current ecosystem state. The vegetation survey was carried out by Florence Renou-Wilson using field methods according to the Braun-Blanquet approach (1964). Thereby five habitat quadrates (HQs, 4x4 m^2) were taken on each study site. Example photographs representing the HQs of the study sites appropriately are shown in **figure 2**. Within those, four vegetation quadrats (VQs, 1x1 m^2) were identified, which represent the vegetation plots. All vascular plant and bryophyte species were identified and recorded on VQ level following a cover value scale, based on Domin. The plant functional types (PFT) as well as bare peat and plant litter cover was estimated on HQ and VQ level following the same scale. The scale includes cover values from 0 to 4, whereby 0=absent, 1=rare (<5%), 2=occasionally (5-20%), 3=frequently (21-50%) and 4=dominant (>50%). Additionally, *calluna* or *erica* maximal height (cm) was recorded as a measure of vegetation height on VQ level. Moisture condition with a scale from 1-5 (whereby 1=relatively dry ground, 2=moist to wet ground surface, 3=water table at or just below surface, 4=pools present and

5=surface water abundant) as well as the presence or absence of habitat types (hummocks, sphagnum hummocks, pools, hollows, lawns and flats) were identified on HQ level.

2.3 Data analysis

In order to analyse the general habitat condition in terms of heterogeneity and wetness, the relative presence of habitat types based on the presence/absence data as well as the median moisture level for each bog was calculated. The relative presence of habitat types was used as a measure for habitat heterogeneity. Thereby a relative presence of 1 means that the habitat type could be observed in all habitat quadrats and a relative presence of 0 means that the habitat type could not be observed at all. A Chi² or Kruskal-Wallis test was carried out to assess differences between the study bogs in terms of heterogeneity or moisture level, respectively.

Species richness (SR) and Shannon-Wiener Index (SWI) were determined as indicators of vegetation diversity for each study bog. SR and SWI were both determined as mean values per plot and SR was further determined in total species number, total vascular plant species number and total bryophyte species number per bog. Mean maximum heights/VQ of c*alluna* and *erica* species, were calculated to further evaluate vegetation composition and vegetation height and ANOVA was used to test the difference between different bogs. Dominant PFTs and species for each study bog were determined on the basis of the respective cover value medians per VQ. Cover-weighted means of Ellenberg Indicator Values (EIV) (Ellenberg et al. 1991) were calculated from the VQ data for soil moisture (EIV-M), acidity (EIV-R) and nitrogen (EIV-N). Indicator values which were not available in Ellenberg et al. 1991, like those stated as unknown or not considered, were based on Hill et al. 1999 or Hill et al. 2007. The Kruskal-Wallis Test was applied to compare the EIVs means between the study bogs.

Non-metric multidimensional scaling (NMDS) ordination was used to investigate and visualize differences in species and plant functional type composition of the ten study bogs via using the vegan package and metaMDS function in R (version 3.2.2). Bray-Curtis index was used as the distance measure and NMDS runs were repeated with a maximum of 20 tries with random starting configurations. The appropriate number of dimension was determined via the calculation of the stress value, that basically describes the fit between the real object distances an and the final composition (Zelený 2015). Generally, the number of dimensions beyond which reduction in stress does not change much is appropriate for the analysis. For this data, the strongest decrease of stress values was discovered when shifting from one to two dimensions; hence two dimensions were used for the analysis. Stress plots were created in order help visualising and evaluating the quality of applied dimension as they show scatter around the regression between each pair of communities in the final configuration against their original dissimilarities. Small scatter around the regression line suggests that original dissimilarities are well preserved in the reduced number of dimensions (Lefcheck 2012). NMDS was applied for PFT and bryophyte species using mean cover values. Total species and vascular species cover was not considered in NMDS due to insufficient data. All statistical analysis was performed in R (version 3.2.2) or Microsoft Excel 2010. All diagrams were created in R.

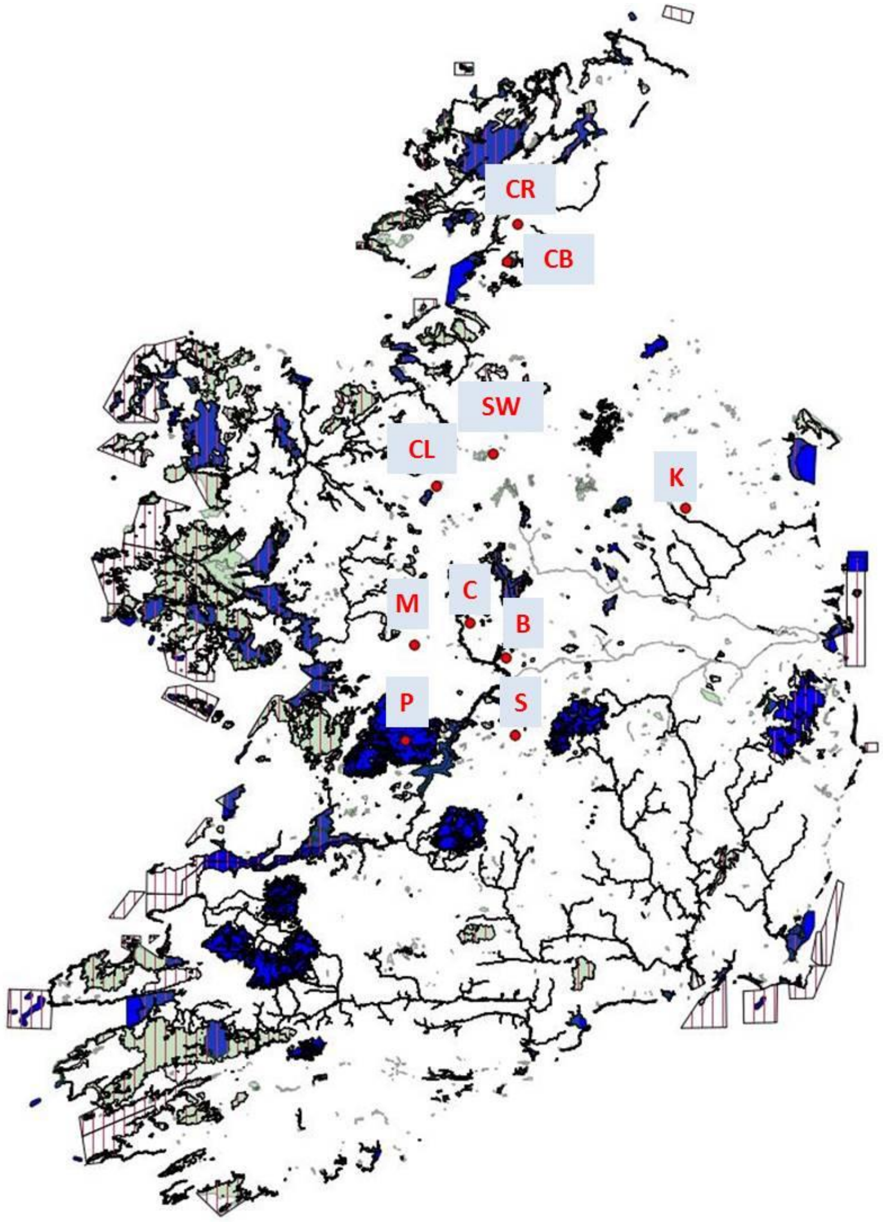


figure 1: Location of the study bogs in Ireland. M=Moyarwood, C=Cuckoo Hill, B=Blackwater, SW=Sopwell, CL=Cloonshanville, S=Sharavogue, K=Kyllyconny, CB=Carrickbarr, CR=Croaghonagh, P=Pollagoona.

table 1: Overview study sites showing management status and geographic information.

		Management status						Geographic information		
Site	**Code**	**Type**	**Prior restoration**	**Peat extraction**	**Restoration type**	**Macro-topography**	**Active drains**	**Eco-region**	**T (°C)** annual mean	**Rainfall (mm)** mean monthly total
Moyarwood	**M**	RB	drained	no	Drain Damming	Flat	outskirts	West-midlands	9,8	845,7
Cuckoo Hill	**C**	RB	drained	no	Drain Damming	Flat, Sloping	outskirts	West-midlands	9,8	845,7
Blackwater	**B**	RB	cutaway	yes (milled)	Pump stopped	Flat	yes	West-midlands	9,8	845,7
Sopwell	**SW**	RB	afforested	no	Clearfelled	Flat	outskirts	South-midlands	9,3	1173,6
Cloonshanville	**CL**	RB	afforested	no but adjacent	Plastic dam	Flat	no	West-midlands	9,3	1173,6
Sharavogue	**S**	RB	cutover	no but adjacent	peat dam	Domed	outskirts	South-midlands	9,8	845,7
Kyllyconny	**K**	RB	cutover	cutover	plastic dam	Glentle slope	outskirts	East	9,4	960,4
Carrickbarr	**CB**	BB	afforested	no	Clearfelled	Gentle slope	no	north-west	9,8	1076
Croaghonagh	**CR**	BB	afforested	no but adjacent	Plastic dam	Sloping	outskirts	North-west	9,8	1076
Pollagoona	**P**	BB	afforested	unknown	Clearfelled	unknown	unknown	SW midlands	9,8	845,7

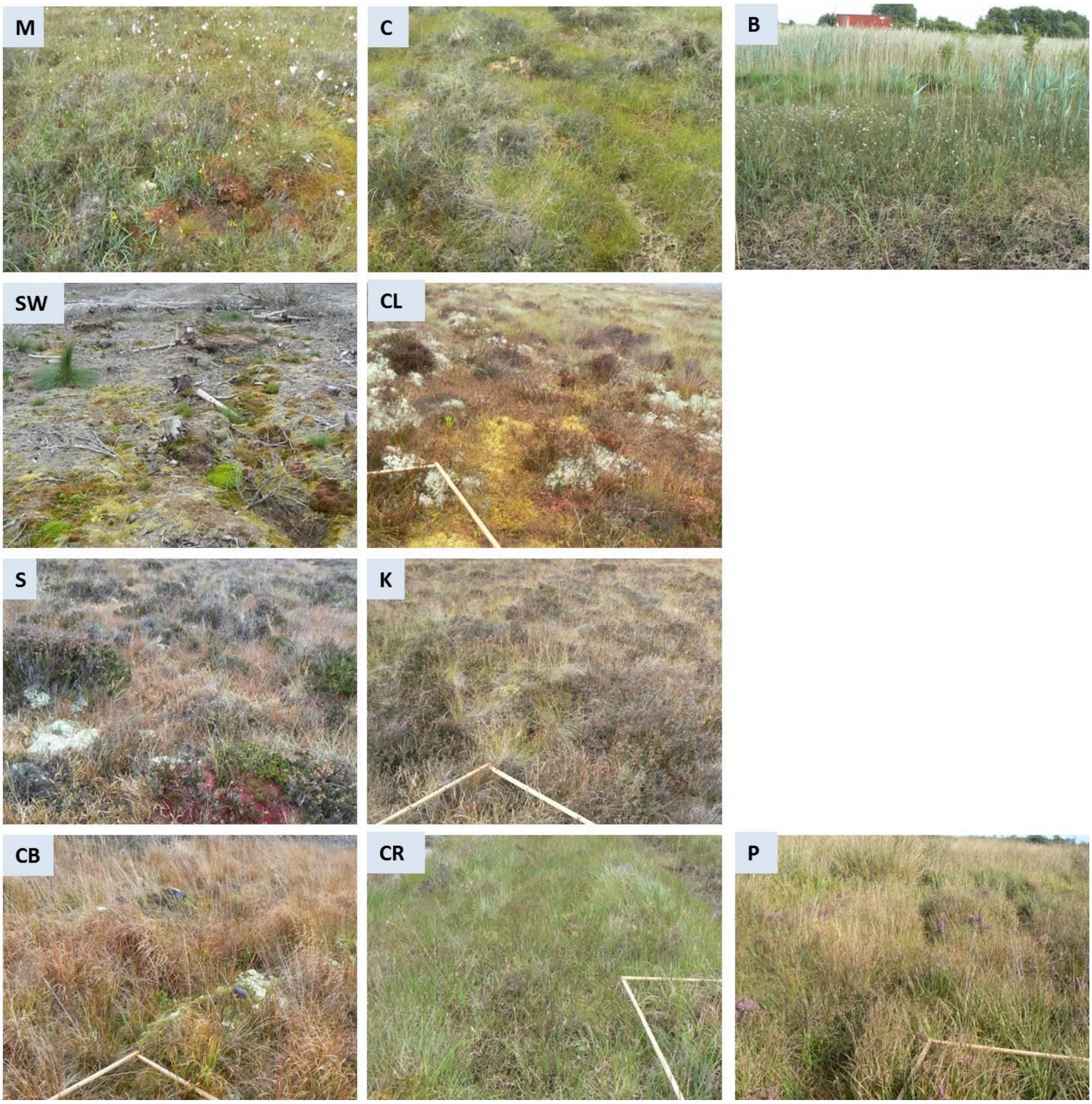


figure 2: **Exemplary habitat quadrat representation for each site.** M=Moyarwood, C=Cuckoo Hill, B=Blackwater, SW=Sopwell, CL=Cloonshanville, S=Sharavogue, K=Kyllyconny, CB=Carrickbarr, CR=Croaghonagh, P=Pollagoona.

3. Results

3.1 Habitat analysis

The general habitat condition was investigated based on the relative presence of different habitat types, which was used as a measure for habitat heterogeneity, and moisture level (**table 2**). In regard to habitat heterogeneity, Blackwater and Pollagoona significantly vary from the other sites since they completely lack different habitat types. No significant difference in general habitat heterogeneity could be detected between the other bogs (p=0.06). However, it is noticeable that Cuckoo Hill shows the highest degree of habitat heterogeneity (with an average relative presence of 0.73) and is the only site where pools could be detected. While the moisture level was significantly different between the sites if considering all bogs (p= 0.008) or only raised bogs (p=0.01), this is not the case between the three blanket bogs which all showed a median moisture level of 2 (moist to wet ground surface). In general all bogs showed a median moisture level of at least 2 or even 3 (water table at or just below surface), except from Sopwell that shows a relatively dry ground. Furthermore, the moisture level is usually quite consistent throughout the study sites, except for Cuckoo Hill where relatively high habitat variation could be detected, ranging from 1 (relatively dry ground) to 4 (pools present).

table 2: Habitat analysis. Top table shows the relative presence of habitat types as well as median moisture level for each bog (standard deviation in parentheses). The average of relative habitat type presence is used as a measure of relative heterogeneity. Lower tables show significant tests for habitat heterogeneity and moisture level. ns = not significant, *p < 0.05, **p < 0.01, ***p < 0.001, ****p < 0.0001.

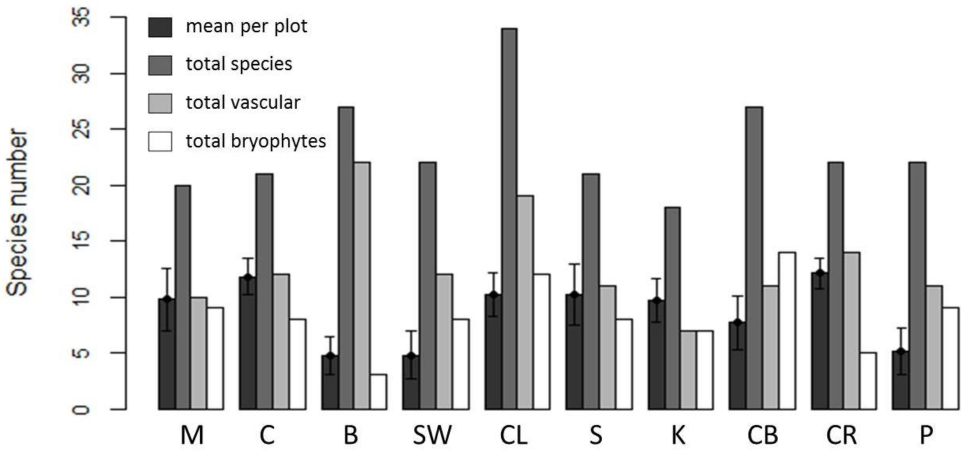
	Raised Bogs							**Blanket Bogs**		
	M	**C**	**B**	**SW**	**C**	**S**	**K**	**CB**	**CR**	**P**
Habitat types (rel. presence)										
Hummocks	0.6	1	0	0.8	0.4	0.8	0.8	0	0.4	0
Sphagnum hummocks	0.6	0.8	0	0.8	0.4	0.8	0.8	0	0.4	0
Pools	0	0.6	0	0	0	0	0	0	0	0
Hollows	0.4	0.6	0	0	0.2	0.4	0.2	0.8	0.2	0
Lawns	0.4	0.6	0	0	0	0.2	0	0.6	0	0
Flats	1	0.8	0	0	0.4	1	1	0.6	1	0
Average (rel. heterogeneity)	**0.5**	**0.73**	**0**	**0.27**	**0.23**	**0.53**	**0.47**	**0.33**	**0.33**	**0**
Moisture level (median)	**3** (0.9)	**3** (1.1)	**3** (0.4)	**1** (0.0)	**2** (0.5)	**3** (0.4)	**2** (0.0)	**2** (0.5)	**2** (0.5)	**2** (0.0)

	p (all bogs)	**p (without B+P)**	*test*
Habitat heterogeneity	****	ns	*Chi²*

	all bogs		**only RB**		**only BB**		*test*
	F	**p**	**F**	**p**	**F**	**p**	
Moisturelevel	2.5	**	8.5	**	0.4	ns	*Krustal Wallis*

3.2 Species richness and biodiversity

Species richness, in terms of mean species number per plot, total species number, total vascular and bryophyte species number (Fehler! Verweisquelle konnte nicht gefunden werden.) as well as the mean Shannon Wiener Index (SWI) per plot as a measure of species diversity (**figure 4**), was calculated for all study sites. Total species number ranged from 18 for Kyllyconny up to 34 for Cloonshanville. Also the proportion between vascular species and bryophytes differs between the sites and is not necessarily reflected in the total species number. The highest proportion of bryophyte species (52%) could be detected on Carrickbarr which has a total species amount of 27, similar to the cutaway raised bog Blackwater. However, the latter shows the lowest moss proportion of all bogs (11%). Except from Blackwater and Croaghonagh, the bryophyte proportion of all study sites represents over 30%. A comparison between mean species number per plot with total species number gives an impression about the general spread of the species and to what degree several species occur together. For instance, Blackwater shows a quite low species number per plot (4.8±1.7) with a relatively high total species number of 27, indicating that only few species occur together (18%). Equal situations can be observed for Sopwell (22%) and Pollagoona (23%). On the other hand, in case of Moyarwood, Cuckoo Hill, Croaghonagh, Sharavoque and Kyllyconny in average almost 50% or more of the total species occur simultaneously in one plot. The average SWI value per plot was 1.79±0.42, with Croaghonagh and Cuckoo Hill showing the highest species diversity (SWI=2.34 or 2.27, respectively) and Sopwell and Blackwater showing the lowest (SWI=1.29 and 1.34).



	M	C	B	SW	CL	S	K	CB	CR	P
share bryo.	0.45	0.38	0.11	0.36	0.35	0.38	0.39	0.52	0.23	0.41
mean/total	0.49	0.56	0.18	0.22	0.3	0.48	0.54	0.29	0.55	0.23

figure 3. Species richness in mean species number per plot, total species number, total vascular species number and total bryophyte species number. Table shows proportion of bryophytes species of total species number and ratio of mean species number per plot and total species number. M=Moyarwood, C=Cuckoo Hill, B=Blackwater, SW=Sopwell, CL=Cloonshanville, S=Sharavogue, K=Kyllyconny, CB=Carrickbarr, CR=Croaghonagh, P=Pollagoona.

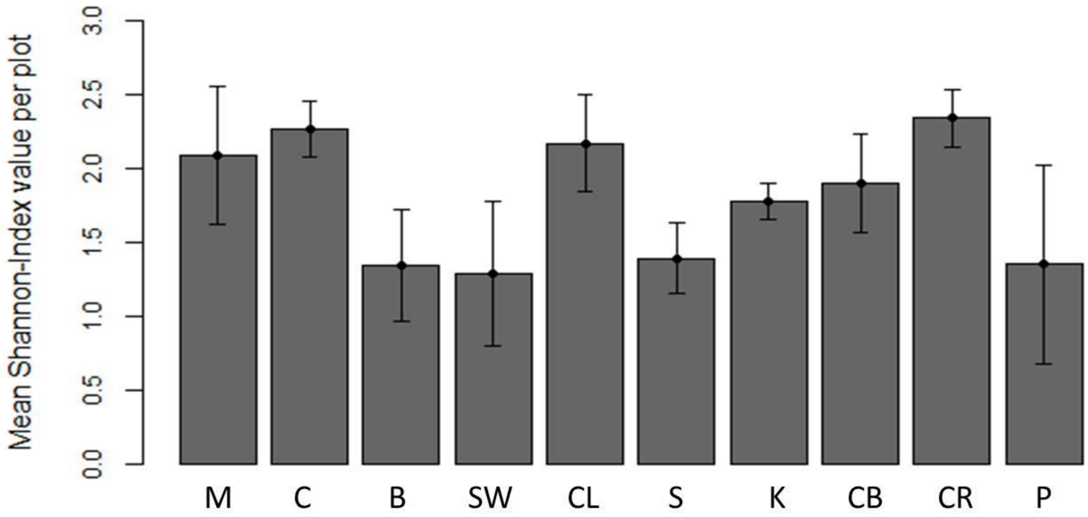


figure 4: species diversity as mean Shannon Wiener Index per plot. Standard deviation as error bars. M=Moyarwood, C=Cuckoo Hill, B=Blackwater, SW=Sopwell, CL=Cloonshanville, S=Sharavogue, K=Kyllyconny, CB=Carrickbarr, CR=Croaghonagh, P=Pollagoona.

3.3 Vegetation composition

In regard to plant functional type cover, the presence of woody species generally was very rare on all sites and could only be detected at very low cover on Blackwater, Sopwell and Cloonshanville. The same accounts for ferns (with a low abundance on Blackwater, Sopwell, Pollagoona and Carrickbarr) and algae cover (low abundance on Sopwell, Croaghonagh, Cloonshanville and Sharavoque). The most dominate PFTs were sedges (S) and *Sphagnum* mosses (Sp.M) followed by ericoid plants (E), and non-*Sphagnum* mosses (nSp.M) (**table 3**). Grasses (G) are only dominant at the blanket bogs, plant litter (L) was detected at high cover on Blackwater as well as Sopwell and Blackwater furthermore showed a dominance of bare peat (P) cover. The most abundant species that could be detected on the study sites were *Carex vulgaris* (*C.vul.*) and S*phagnum capillaris (S.Cap)* followed by *Eriophorum vaginatum (E.vag.)* (on raised bogs) and *Molinia caerulea (M.cae.)* (on blanket bogs). Noticeably apart from all other bogs in terms of vegetation composition is Blackwater with a high amount of bare peat and plant litter cover and the dominant species *Carex nigra (C.nigra)* and *Polytrichum commune (P.com.)*, as well as the plant litter-rich Sopwell bog, dominated by *Rhytidiadelphus spp. (Rhy.sp.)* and *Erica tetralix (E.tet.)* (**table 3**). The average height of *Calluna* and *Erica*, used as a measure to further evaluate vegetation composition, bog state and vegetation height differs significantly between the bogs (**table 3**). Blackwater does not show any evidence of ericoid species at all. Sopwell, Carrickbarr and Pollagoona show an average height below 10 cm and the others between 10 and 20 cm. The height within the bogs is quite variable and is reflected by high standard variations.

3.3.1 Abiotic site characteristics based on Ellenberg Indicator Values

The abiotic site characteristics for moisture, acidity and nitrogen were estimated based on the vegetation species composition by using the appropriate Ellenberg Indicator Values (EIVs). All three parameters differ significantly between the ten bogs. However, the average EIVs for moisture (7.5±0.5), acidity (2.4±0.8) and nitrogen (1.9±0.7), generally indicate a soil that is rather damp to wet, quite acid and infertile (**table 3**). The detected difference seems to be mainly pronounced between the different degradation types (drained, cutaway, cutover) since no or only a relatively low significant variation can be detected between drained raised bogs and cutover raised bogs (**table 3**, bottom). Furthermore, Blackwater shows quite different acidity and nitrogen conditions compared to the other study bogs; with a tendency towards a lower acidity and higher fertility (both moderate). A lower acidity and slightly higher fertility could also be observed for Sopwell.

table 3: Vegetation composition. Top: dominant plant functional type (PFT) and species based on cover value medians per VQ (median in parentheses), *calluna* and *erica* mean maximum height per VQ (standard deviation in parentheses) and mean weighted indicator values per VQ (standard deviation in parentheses), after Ellenberg et al. 1991 (EIVs). E=Ericoid, G=Grass, L=Plant litter, nSp.M=non *Sphagnum* mosses, S=Sedges, Sp.M= *Sphagnum* mosses, P=bare peat; *C.vul.=Carex vulgaris, E.tet.=Erica tetralix, E.ang.=Eriophorum angustifolium, E.vag.=Eriophorum vaginatum, J.eff.=Juncus effuses, M.cae.=Molinia caerulea, P.aus.=Phragmites australis, P.com.=Polytrichum commune, R.lan.=Racomitrium lanuginosum, Rhy.sp.=Rhytidiadelphus spp, S.cap.=Sphagnum capillaris, S.mag.=Sphagnum magellanicum.* **Bottom:** significant tests for vegetation height and EIVs, while considering all bogs, only drained raised bogs (drained RB), cutover raised bogs (cutover RB), afforested raised bogs (afforest. RB) or only blanket bogs (BB). ns = not significant, *p < 0.05, **p < 0.01, ***p < 0.001, ****p < 0.0001.

	Raised Bogs							Blanket Bogs		
	M	**C**	**B**	**SW**	**CL**	**S**	**K**	**CB**	**CR**	**P**
Dominant PFT										
1	S (4)	S, Sp.M (4)	S (4)	L (4)	E, Sp.M (3)	Sp.M (4)	Sp.M (4)	Sp.M, G (3)	G, S, nSp.M (3)	G (4)
2	Sp.M (3.5)	E (2.5)	P, L (2)	nSp. M (3)	nSp.M, S (2)	S (3)	S, E (3)	n.Sp.M (2)	E (2)	nSp.M (2.5)
3	E (3)			Sp. M (1)		E (2.5)	nSp.M (2.5)			Sp.M, S (1)
Dominant species										
1	*C. vul. (3)*	*Sp. cap. (3)*	*C. nigra (3.5)*	*Rhy. sp. (3)*	*C. vul. (2.5)*	*C. vul. (2)*	*C. vul. (3)*	*M. cae. (2)*	*M. cae. (3)*	*M. cae. (4)*
2	*Sp. cap. (3)*	*Sp. mag.(2.5)*	*P. au.s (1)*	*E. tet. (0.5)*	*E. vag. (2)*	*Sp. cap. (2)*	*Sp. cap. (3)*	*Sp. cap. (2)*	*R. lan. (2.5)*	*P. com. (2)*
3	*E. vag. (2)*	*C. vul (2)*	*E. ang. (0)*	*Sp. cap. (0)*	*Rhy. Sp. (1)*	*E. vag. (2)*	*E. vag. (2)*	*P. com. (2)*	*C. vul. (2)*	*J. eff. (0)*
Calluna/Erica (cm)	**16.6** (5.0)	**18.05** (7.5)	**0** (0.0)	**6.2** (7.4)	**17.2** (6.2)	**14** (3.9)	**20.55** (7.2)	**7.95** (8.7)	**16.25** (5.0)	**9.65** (11.9)
EIV moisture	**8.0** (0.4)	**8** (0.3)	**8.4** (0.6)	**6.7** (0.5)	**7.5** (0.7)	**7.9** (0.6)	**7.4** (0.3)	**7.2** (0.7)	**7** (0.4)	**7.1** (0.5)
EIV aciditiy	**2** (0.3)	**2** (0.2)	**4.5** (1.0)	**3** (0.8)	**2.1** (0.3)	**1.9** (0.1)	**1.8** (0.1)	**2.2** (0.3)	**2.1** (0.2)	**2.5** (0.4)
EIV nitrogen	**1.5** (0.2)	**1.6** (0.2)	**3.5** (1.3)	**2.4** (0.6)	**1.7** (0.3)	**1.4** (0.1)	**1.4** (0.1)	**1.9** (0.2)	**1.5** (0.1)	**2.2** (0.3)

	all bogs		RB (drained)		RB (cutover)		RB (afforest.)		only BB		
	F	p	F	p	F	p	F	p	F	p	*test*
Calluna/Erica	1.9	****									*ANOVA*
EIV moisture	11.1	****	0.1	ns	5.3	*	11.1	***	0.4	ns	*Krustal Wallis*
EIV aciditiy	12.6	****	1.2	ns	1.8	ns	11.9	***	5.4	**	*Krustal Wallis*
EIV nitrogen	14.9	****	4.4	*	0.02	ns	13.4	***	20	***	*Krustal Wallis*

3.3.2 Vegetation composition differences measured by ordination

The ecosystem structure, measured as plant functional type composition and bryophyte species composition was also investigated via NMDS, which clearly visualised the differences between the various study bogs (**figure 5**). The most substantial difference can be recognised between the cutaway raised bog Blackwater and all the others, for both PFT and bryophyte species composition. This difference is mainly described on the second axis for PFT cover and on the first axis for bryophyte cover and is connected to the high abundance of bare peat cover and the total lack of *Sphagnum* mosses of Blackwater (**figure 5, table 3**). In terms of PFT composition, Sopwell also represents an outsider position described by axis 2 which is based on a dominance of plant litter (**figure 5A, table 3**). Such a difference cannot be observed for bryophyte species composition (**figure 5B**). Besides Blackwater and Sopwell (yellow shaded), another separation can be detected between raised (red shaded) and blanket bogs (blue shaded) for PFT composition shown on the first axis, although a bit less pronounced. Thereby, blanket bogs show a tendency towards non-*Sphagnum* mosses and grasses. The raised bogs on the other hand, are characterised by a high abundance of *Sphagnum* mosses in combination with ericoid vegetation and sedges. A difference between raised and blanket bog becomes less clear if looking only on bryophyte composition, since in all of study bogs (with the exception of Blackwater) mosses are among the dominant vegetation cover types. In both ordinations the use of two dimensions was appropriate as it can be visualized in the low scatter pattern of the stress plots (**figure 5, bottom**).

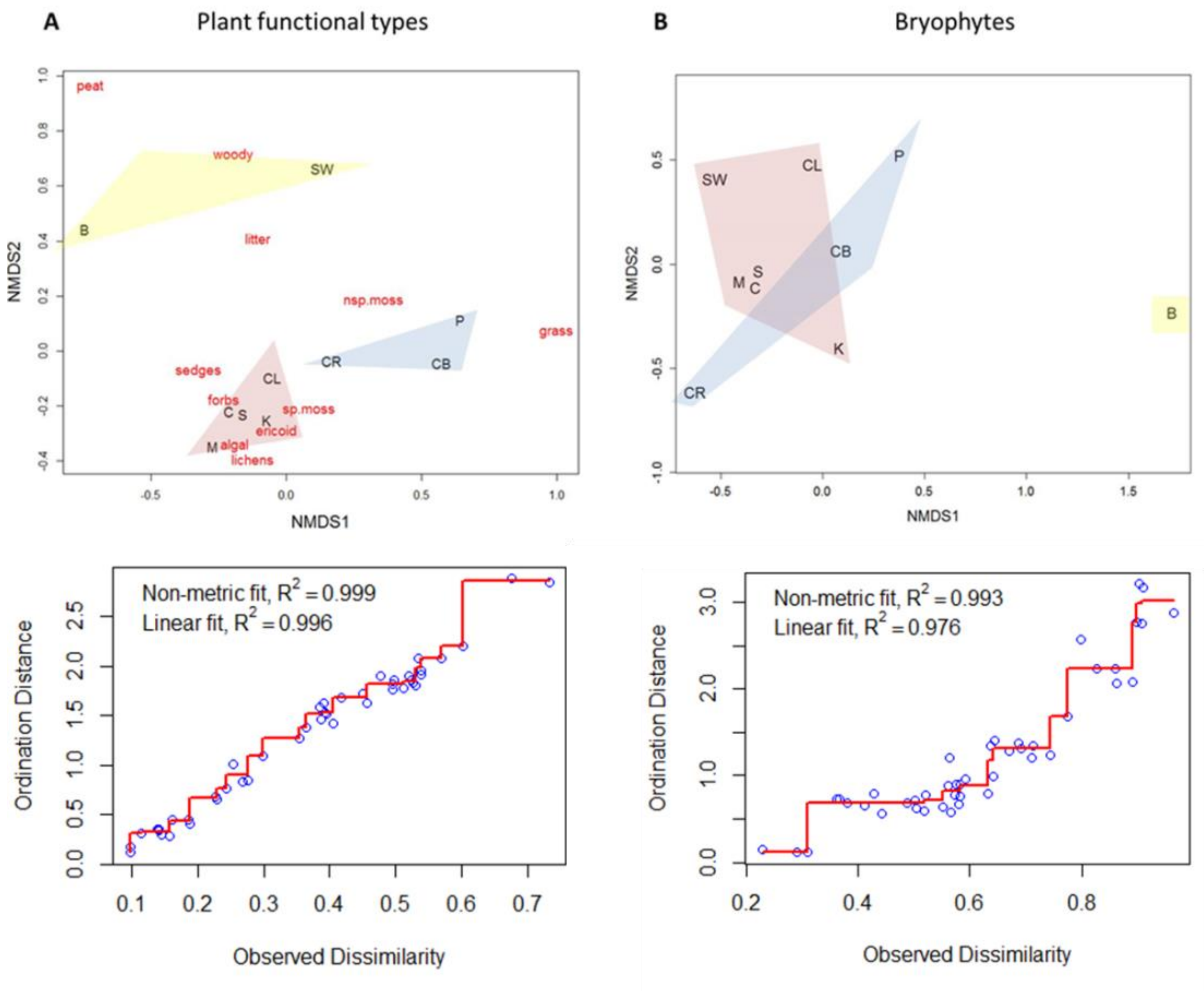


figure 5: Vegetation composition difference between study sites. NMDS was applied for plant functional types (PFT) and bryophyte species using mean cover values, both NMDS ordination with Bray Curtis distance and 2-dimensional solutions. Stress plots were created for both to visualize the quality of applied dimension (bottom graphs). **A.** Plant functional types, where most pronounced difference between the sites is shown on axis 2 and where Blackwater and Sopwell separates from the rest, stress= 0.03. **B.** Bryophytes, where most pronounced difference between the sites is shown on axis 1, where Blackwater separates from the rest, stress= 0.08). Raised bogs shaded in red, blanket bogs shaded in blue, bogs showing a pronounced difference to the others shaded in yellow.

4. Discussion

Overall, without considering any differences between rewetting method or bog type yet, the rewetting measures seem to have a positive impact on the water table of almost all study sites since the moisture condition was throughout at least moderate (moist to wet ground surface). This counts for all bogs with the exception of Sopwell, a raised bog with rather dry conditions. A rapid rise of the water table is a common short term-effect of peatland restoration (Jauhiainen et al. 2002a; Worrall et al. 2007) and a premise for a subsequent change in vegetation composition and biodiversity towards a pristine-like ecosystem. However, despite more or less similar moisture levels, further habitat characteristics and vegetation composition varies greatly between the different sites in this study, which is not solely based on the typical differences of raised and blanked bog ecosystems. This indicates that other individual site factors play a role as well. The presence and variety of different (micro-) habitats, including pools, hummocks or hollows is a good positive indicator of the bog condition (Renou-Wilson et al. 2011b) and can serve as a first hint for potential differences in the restoration success after rewetting. In this regard, the once raised bog Blackwater as well as the blanket bog Pollagoona both do not show any habitat heterogeneity, suggesting a poorer general health condition compared to other bogs of the same type.

4.1 The special cases of Blackwater and Sopwell

Probably the most outstanding observation could be made in case of Blackwater, that shows a clear outsider position throughout all further analysis, whereby all of them point out to a relatively poor health state of this bog. Blackwater clearly differs from the other raised (as well as blanket bogs) not only by lacking any habitat heterogeneity but also in terms of biodiversity, vegetation composition and soil characteristics. It does not show any *Sphagnum* moss species at all and a very poor moss cover in general. In fact there are only eight vegetation squares out of twenty where mosses are slightly present and the bog contains only three bryophytes species (*Campylopus introflexus, Hypnum spp and Rhytidiadelphus spp*). With a SWI of 1.34±0.38 it generally has a relatively low biodiversity. Instead, it shows a high degree of bare peat cover and a tendency towards a less acid and more fertile soil reflected by respective Ellenberg Indicator Values. All those features are far off from an intact raised bog ecosystem and might be connected to the management status. Blackwater is a prior cutaway bog where in addition peat extraction is still happening in the surroundings and some drains are still active. 'Cutaway' means, that prior to restoration, the complete vegetation was removed from the bog, and thereby differs evidently from cutover (where the peat is removed from the edges of the bog), afforested or only drained practices. While all these disturbances affect the hydrology of the ecosystems, cutaway bogs prior restoration further lack vegetation that can respond to increasing moisture conditions and hence, the recovery of vegetation after rewetting on cutaway peatlands is generally much slower (Jauhiainen 1998). Considering these aspects, it seems very likely that the overall poor condition of Blackwater is correlated with its previous type of degradation. However, since Blackwater is the only cutaway bog analysed here and therefore no other comparison can be made on basis of this data, formulating a clear statement remains difficult. In order to draw a clear conclusion about the role of the degradation type for a restoration success after rewetting, it would be necessary to include several sites for each degradation and restoration type under the same monitoring conditions.

In addition to Blackwater, Sopwell also differs significantly in its ecosystem structure from the other bogs, mainly in regard to vegetation composition through its high abundance of plant litter, a lower species diversity (SWI = 1.29±0.49) and with an average of 4.8±2.1 also a lower species number per plot. However, in this special case it seems likely that the current condition results from the relatively poor moisture level and that the type of degradation or other factors are rather secondary.

4.2 Raised versus blanket bogs

Raised and blanket bogs examined here differ clearly from each other in terms of vegetation composition, with blanket bogs showing a dominance of grasses, particularly purple moor-grass (*M. caerulea)*, followed by some mosses (*Sphagnum capillaris and Polytrichum commune*) and raised bogs (excluding Blackwater and Sopwell) mainly dominated by sedges, ericoid species and *Sphagnum* mosses (particularly *C. vulgaris, Sp. capillifolium* and *E. vaginatum*).

The detected blanket bog vegetation shows a strong tendency towards a typical intact lowland blanket bog composition, usually dominated by grass species such as *M.caerulea*, *E. vaginatum* or *E.tetralix* (Conaghan 2009) and a moss cover, lower compared to raised bogs though, but generally exceeding 30% ground cover. Common species include *Sp. capillifolium, Sp. papillosum, R. lanuginosum* and *Hy. cupressiforme* (Conaghan et al. 2000). However, the strong dominance of *M. careula* detected here and particularly distinct for Pollagoona, could point out to a fluctuating or partly lower water table, since this species is quite tolerate towards changing water conditions (Conaghan et al. 2000). Another characteristic species of atlantic blanket bogs is the purple liverwort *Pleurozia purpurea* (Conaghan et al. 2000), which could be found almost constantly at low cover in Croaghonagh and scattered in Carrickbarr (data not shown). Summarised, especially Croaghonagh represents already a fairly typical blanket bog vegetation composition, suggesting a good and active state of this study site. The restoration success is followed by Carrickbarr, which also shows a decent species composition related to a pristine blanket bog. Pollagoona, however, differs from the other in terms of species composition and a lower biodiversity (SWI= 1.35±0.67) and species number per plot (5.1±2.4), which might be connected to the different eco-region in the southwest Midlands. In fact, the structure of blanket bogs shows much variation throughout Ireland, whereby much of variation in vegetation is due to several region-dependent factors, including rainfall, temperature or altitude (Conaghan et al. 2000).

In contrast to blanket bogs, raised bogs are usually dominated by *Sphagnum* mosses (Walsh and Barry 1958), which are major important for peat formation. Due to its high water holding capacity, *Sphagnum* is responsible for keeping the peat surface waterlogged and for an acidic environment leading to further *Sphagnum* establishment (Renou-Wilson et al. 2011b). Indeed, *Sphagnum* mosses could be identified as major species for the raised bogs in this study, and for most of them the cover median exceeds 50%, which is a quite promising result. However, the moss species number proportion generally is quite modest compared to vascular plants, mainly in the range of 30-40% and also the *Sphagnum* moss diversity is quite low. The most dominant species found was by far *Sphagnum capillifolium* and although being an important peat forming species as well, important other key species like *Sp. magellanicum*, *Sp. papillosum* or *Sp. rubellum* (Smolders et al. 2003), were basically rare or completely absent, with the exception of Cuckoo Hill. Other common plant species found on raised bogs, like heather (*C. vulgaris*) or bog cotton (*E. angustifolium* and *E. vaginatum*) (Renou-Wilson et al. 2011b), could be identified on the study bogs as well. Particularly remarkable was the presence of *E. vaginatum,* a species already reported in other studies of having a strong response to bog restoration. In Haapalehto et al. 2011b the abundance of this bog cotton was constantly increasing through a 10-year monitoring period. Suggested by Jauhiainen et al. 2002a, this condition could be explained as a transitional stage towards a *Sphagnum*-dominated vegetation. However, as the early development and composition of vegetation after rewetting does probably differ between peatland types, region and other parameters, it is difficult to draw a final conclusion. Nevertheless, the results suggest that rewetting generally had a positive impact on the recovery of the studied raised bogs. However, the establishment of a continuous and more diverse *Sphagnum* layer seems to take more time and maybe requires some additional measures. For instance this could be accelerated by harvesting branches of specific *Sphagnum* species from *Sphagnum*-rich areas of the same site or even another bog and transplanting them to the poor zones (Conaghan 2009).

No clear evidence of a difference in biodiversity, species richness or derived soil characteristics based on Ellenberg could be detected between raised and blanket bogs as these parameters are varying to a similar extent between all bogs.

4.3 The role of different degradation types

The outstanding position of the cutaway bog Blackwater has been already discussed. Apart from this it remains difficult to draw any conclusion of the role of the different degradation types on the restoration success. In addition to Blackwater, the prior afforested bog Sopwell also differs clearly from the other raised bogs in terms of vegetation composition including a less pronounced *calluna/erica* height, a situation that might be based on the poor moisture state showing that rewetting was less successful in this case. However, Cloonshanville, the other afforested raised bog in this study does not show any similar features comparable to Sopwell, except from maybe that both bogs show relatively few species occurring together on one plot. Apart from this, the moisture level is satisfactory and also the plant functional type and species composition is rather comparable to the remaining raised bogs. Overall, no relevant difference could be detected between only drained, afforested and cutover bogs within raised bogs, neither on habitat level nor on biodiversity or vegetation composition level. The only noticeable observation made, is the stable abiotic soil features (based on Ellenberg) within drained and cutover bogs. No conclusion can be made here for blanket bogs as only afforested blanket bogs were included in this study.

4.4 The overall impact of rewetting: in a nutshell

In summary, for most study sites the above results indicate that rewetting measures successfully initiated the restoration process and the ability of the bog to re-gain its self-regulation function. This was mainly independent on the type of degradation. This reflects the major importance of water-logging conditions as a basis for a healthy bog and suggests that the type of degradation might not play a principal role for the final restoration outcome. It further indicates that rewetting is not only necessary to trigger recovery but in many cases might be also sufficient. Probably it is rather the magnitude of degradation (e.g. total area of disturbance, remaining amount of vegetation, remaining peat depth or acrotelm condition) and the time that has passed since the disturbance occurred than the degradation type that dictates rewetting success.

However, despite the overall rewetting success this study also shows that there are still profound differences in the composition of plant species between the rewetted sites compared to pristine landscapes of the same type, especially with regard to the raised bogs. It is difficult to identify any clear reasons for this, since all bogs studied here differ significantly in terms of management status and/or region. In case of cutover bogs, as we had on the example of Blackwater, it is likely to be more challenging to initiate the process that leads to the recovery of ecosystem functionality and structure. The transformation towards a typical bog vegetation might therefore require additional measures besides rewetting. It is important to keep in mind, that here we studied the rewetting successes only on a quite short time scale. The year of rewetting is not known for most sites, but it has been just four years ago in case of Cuckoo Hill and for Moyarwood even only three years, and it is likely that the others do not differ so much in this respect. If considering that the Irish blanked bogs started developing around 4000 years ago and raised bogs even have been evolved since the last ice age (Renou-Wilson et al. 2011b), these few years of rewetting represent only a glance in a whole life-story. In fact, during the first years of restoration, ecosystems are markedly dynamic (Sarr 2002), including quick variations in their communities (Kellogg and Bridgham 2002). Furthermore, the impacts of drainage on hydrology can last for decades after drainage (Holden et al. 2006) and the recovery of a bog is not only dependent on a high water table alone but also on the quality of water (Grootjans et al. 2002a; Grootjans et al. 2002b). Hence, early assessments after rewetting are important to make first estimates of an initial recovery success, but are not enough for making final conclusions of the impact of rewetting on the peatland health state. In order to properly assess any restoration process, long-term monitoring is therefore inevitable (Verhagen et al. 2001). Additionally, it would be necessary to include pristine bogs of the same type and region with the same data collection approach used, in order to make straight-forward comparisons between natural and restored bogs. This is important to clearly evaluate an eventual restoration success. In order to compare the impact of rewetting on bog biodiversity and vegetation composition between different degradation types (only drained, afforested, cutaway, cut over) it would be further necessary to include more peatlands of the various degradations types.

4.5. Policy relevance

Due to the important and meanwhile rare status of active bogs in Ireland, it is in the interest of the government to enable and promote the restoration of degraded bogs. Although the awareness of the ecological importance of Irish peatlands – for Ireland and beyond borders – has been fortunately rising, widespread damage and exploitation is still happening.

Indeed, there are several EU and international policies that do consider peatlands, including those dealing with climate change, biodiversity or water, and which commit Ireland to undertake appropriate measures to ensure peatland protection and restoration.

The United Nations Framework Convention on Climate Change (UNFCCC) has strong implications on peatland use and management in Ireland. It obligates Ireland to report on GHG emissions of peatlands that are degraded or influenced by human impact from all land use sectors since disturbances and drainage leads to a rise in CO_2 emissions (Renou-Wilson et al. 2011b; Rieley and Lubinaite 2014) . In this context, the Durban agreement led to the inclusion of the wetland drainage and rewetting (WDR) category in the LULUCF (**L**and **U**se, **L**and **U**se **C**hange and **F**orestry) activities under the Kyoto Protocol, with the consequence that rewetting of peatlands can be voluntary included in national greenhouse gas accounting (Rieley and Lubinaite 2014; UNFCCC 2014). The Convention on Biological Diversity (CBD) as well as the Convention on Wetland called Ramsar play an important role for the conservation of biodiversity in an international context and also specifically consider peatlands. The binding agreement CBD, charges its member countries since 1992 to develop and implement the protection and sustainable management of biodiversity, whereby peatlands are considered as key ecosystem (Rieley and Lubinaite 2014). In contrast to CBD which includes many different ecosystem types, the Ramsar Convention is an international treaty that particularly aims for a conservation of wetlands and their resources based on national actions and international cooperation (Ramsar convention secretariat, 2014). Furthermore, peatland conservation, under the aspect of the protection of biodiversity, is integrated into EU legislations, including the EU Habitats Directive and EU Birds Directive (NPS DRAFT 2014), which together are responsible for the protection of over 1000 species and more than 200 important habitat types including bogs, fens and mires (Council of European Communities 1992). In regards to water, the Water Framework Directive (WFD) directs EU water quality management and generally aims for a good status of all waters, including surface, underground and coastal waters (Kallis and Butler 2001). Since peatlands feed into adjacent river catchments and provide crucial functions like water supply, pollution control or groundwater recharge, the WFD has also great implications on Irish peatlands including actions to overcome peatland nutrient pollution (Malone and O'Connell 2009).

Despite the widespread integration of peatland conservation into policies, based on the BOGLAND report the management status of peatlands has mainly not been sustainable and even has been negatively affecting important ecosystem services, including biodiversity (Renou-Wilson et al. 2011a). However, there are several reasons to further push and implement the protection of bogs in Ireland, since they are

(1) **unique ecosystems:** The peatland areas of Ireland are of international importance. With approximately 8% Ireland possesses a significant part of the global blanket bog resource (Foss and O' Connell 1996) and also Irish raised bogs are of national and international conservation importance (Renou-Wilson et al. 2011b).

(1) **crucial for climate regulation:** active bogs have the potential to counteract global warming through the high capacity to store CO_2 (Conaghan et al. 2000).

(2) **important for biodiversity maintenance:** Irish bogs provide habitat to many rare species, whereby the Marsh Saxifrage (*Saxifrage hirculus*), Shining Sicklemoss (*Drepanocladus vernicosus*) or the Bog Orchid (*Hammarbya paludosa*) are only some few examples (Conaghan et al. 2000).

(3) **relevant for the national economy:** peatlands have played and still do play an important role in the economy of Ireland. Especially blanket bogs provide space for cattle and sheep. However, land-use practices need to be managed in a sustainable way including recovering periods and low stocking rates (Conaghan et al. 2000).

The restoration and conservation of Irish bogs cannot be seen solely as a national interest since the functions peatlands provide go far beyond borders. Furthermore, as Ireland contains a significant proportion of the global (not only blanket) bog resources (Conaghan et al. 2000), it has a major responsibility of taking action in peatland restoration and conservation. This study as part of the NEROS project helps to gain more knowledge of the impact of rewetting on peatland structure and hence should support good practice in peatland restoration and management. It emphasized the importance of rewetting in the early restoration process of drained bogs of all types but also shows how variable individual bogs do respond to rewetting. It highlights the importance of flexible and long term restoration approaches of peatlands which is necessary to integrate into policies.

5. Conclusion

Bogs and fens used to be extensive habitats in Ireland but now roughly only 15% are remaining in a near intact state (Renou-Wilson et al. 2011b). According to the Millennium Assessment (MA 2005) "the degradation and loss of wetlands (including peatlands) is more rapid than that of other ecosystems". It describes fatal conditions that emphasize the importance of (1) finally stopping further peatland degradation in Ireland and (2) restoring degraded areas as soon as possible since it might be too late someday if transformation proceeds.

Rewetting usually is the first step of a long-lasting restoration process. Based on this data, rewetting approaches can be concluded as being a crucial first step in peatland restoration for a various types of peatlands in order to enhance the water table. In 90% of the bogs investigated here, moisture conditions of the soil had been shown to be in a satisfactory to good state and the same counts for soil acidity and fertility. However, vegetation cover and species diversity, particularly in terms of *Sphagnum* mosses, suggests that the restoration process is still in an early stage and that some peatland types might require not only more time for recovery but also additional restoration measures in order to achieve the best result.

As almost every site in this study is specific, if considering all parameters including previous use, rewetting method or location, this work helps to collect individual information but it is not sufficient to draw general conclusions. Consequently, we can formulate first ideas about the difference of impact between raised and (lowland) blanket bogs. However, it is not possible to make profound suggestions in how far rewetting impact differs depending on the type of previous degradation or rewetting method. The early assessment of restoration efficiency based on changes in vegetation shortly after rewetting is crucial, but it is also vague due to at least two reasons: Firstly because of the slow recovery of peatlands and secondly because of the high diversity of these ecosystems, even within one geographic region. In order to estimate clear trends in the condition of bog habitats and the efficacy of restoration measures, constant long-term monitoring studies are required. Principally we need to be careful with generalising monitoring results of one restored peatland type to another. This study emphasizes quite well, how diverse peatlands even in one ecoregion are and how different they therefore might react on degradation and rewetting. This difference is very likely based on the combination of the various bog characteristics, not only type and ecoregion, but also degradation amplitude and length, topography or adjacent landscapes. Hence, restoration projects must be flexible and as diverse as the peatlands are.

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