Hydrological and hydrochemical supporting conditions for Irish calcareous fen vegetation

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By

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Declaration

This thesis has not been submitted for a degree to this or any other university and, with acknowledged exception, is entirely my own work.

I agree that this thesis may be lent and copied in accordance with college regulations.

Elisabeth Wilhelmina Bijkerk

2023

"*Kintsugi* [is] not just a method of repair but also a philosophy. It's the belief that the breaks, cracks, and repairs become a valuable and esteemed part of the history of an object, rather than something to be hidden. That, in fact, the piece is more beautiful for having been broken.*"

Kathleen Tessaro

*kintsugi / kɪnˈtsu gi /

noun

Also called **kin·tsu·ku·roi** /kɪnˈtsu kuˌrɔɪ/. a traditional Japanese pottery repair technique in which lacquer mixed with precious metals, especially gold, is used to fill cracks and replace missing pieces.

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Abstract

Calcareous fens are listed in Annex 1 of the European Union (EU) Habitats Directive (Council Directive 92/43/EEC, habitat code 7230, 7210 and 7140) as habitats requiring special conservation measures, including the designation of suitable sites as Special Areas of Conservation (SACs). They are largely groundwater fed wetlands, being located in topographic hollows and fed by springs or seepages of water that has been in contact with mineral ground. Their principal source of nutrients is from surface or groundwater and the substrate is an alkaline to slightly acidic peat soil. The hydrogeological dynamics and hydrochemical signature supports small sedge and brown moss communities in a mosaic of different habitats.

It is acknowledged however that currently no studies of these systems exist from which the environmental supporting conditions could be determined. Therefore, the objective of this study is to investigate hydrological and hydrochemical controls that support Irish alkaline fen habitat.

An intensive multidisciplinary monitoring programme was set up in the following four fen sites containing calcareous fen covering an eco-hydrological gradient from intact to highly degraded conditions: 1) Ballymore, Co. Westmeath, with intact habitat, 2) Pollardstown, Co. Kildare, degraded by drainage and nutrient pollution, 3) Scragh Bog (fen), Co. Westmeath, with near intact habitat threatened by nutrient pollution and 4) Tory Hill, Co. Limerick, highly degraded by drainage.

Hydrological and hydrochemical was collated in order to build conceptual eco-hydrological models to represent both temporal and spatial variability in each geological setting. Data collected from surveyed good quality fen vegetation was used a representative for these models.

Higher concentrations of nutrients were found in the sediments at depth compared to the phreatic water table. This suggests that the vegetation is using up the incoming nutrients and thereby ending up in an organic form. In the natural life cycle of the vegetation, annual decay allows for these nutrients to come back out into solution and disperse into the underlying till substrate. Statistical analysis of the surface water nutrient concentrations associated with the different fen habitat types suggested typical maximum values of 37 µg-P/l for dissolved reactive phosphorus, 382 µg-P/l total phosphorus, 0.57 mg-N/l for ammonia, 0.17 mg-N/l for total oxidised nitrogen and 2.01 mg-N/l for total dissolved nitrogen. It also seemed that specific vegetation required a minimum threshold of nutrients from groundwater flow in order to survive.

The hydraulic gradients suggested that the incoming water from the aquifer enters the fen via discrete conduits straight into the phreatic zone by flowing through the underlying substrate in a more diffuse manner. Equally, in terms of water level, the field investigations suggest that the mean annual water levels are required to stay above ground level in order to support healthy fen vegetation with a threshold water level envelope of between 29 mm to 277 mm above ground level. Furthermore in order to maintain good quality fen vegetation environmental conditions should not change the vertical hydraulic gradients by more than 0.4 during a hydrological year.

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1. Introduction

1.1. Background

Fens are largely groundwater fed wetlands, being located in topographic hollows and fed by springs or seepages of water that has been in contact with mineral ground (Kuczyńska, 2008; Duval, 2010). Fens' principal source of nutrients is from surface or groundwater and the substrate is an alkaline to slightly acidic peat soil. This type of wetland often encompasses a mosaic of different habitats such as open water with reed beds and sedge vegetation as well as semi-terrestrial birch and alder woodland (Sefferova et al., 2008; Foss, 2007). Fens have permanently high water levels at or just below its surface and the organic matter accumulation supports peat growth (Kellner, 2002; McBride et al., 2010; Aggenbach et al., 2013).

The dramatic decline in area of natural fens all over the world, and in particular the species-rich calcareous fens, has resulted in the loss of highly valuable habitats and related species. Fens are therefore now considered to be among the most threatened habitat types in Europe (Joosten & Clarke 2002, Klimkowska et al. 2010, Lamers et al. 2015).

A national fen survey recognised a total number of 808 fen sites in Ireland with a total extent of 10,307 ha of Habitats Directive Annex 1 fen types. If non-designated sites are included this area increases to a total 22,180 ha. The number of protected sites designated as Special Areas of Conservation (SAC) or proposed candidate Special Areas of Conservation (cSAC) is 362 and encompasses an area of 14,086 ha. This represents 64% of the total estimated Irish fen resources (Foss, 2007). The location of fen sites is displayed in Figure 1.1.

Nationally, it is acknowledged that there have been much fewer comprehensive studies on fens compared to the other Groundwater Dependent Terrestrial Ecosystems (GWDTEs) such as bogs and turloughs from which the environmental supporting conditions can be determined. Furthermore, the national fen survey reported that the knowledge of the specific fen type of the sites is either lacking or inadequate for 268 (33%) of sites identified in the present NPWS Fen Study database. This is also true for the extent of fen habitat types present in sites where data is inadequate for around 74% of the database.

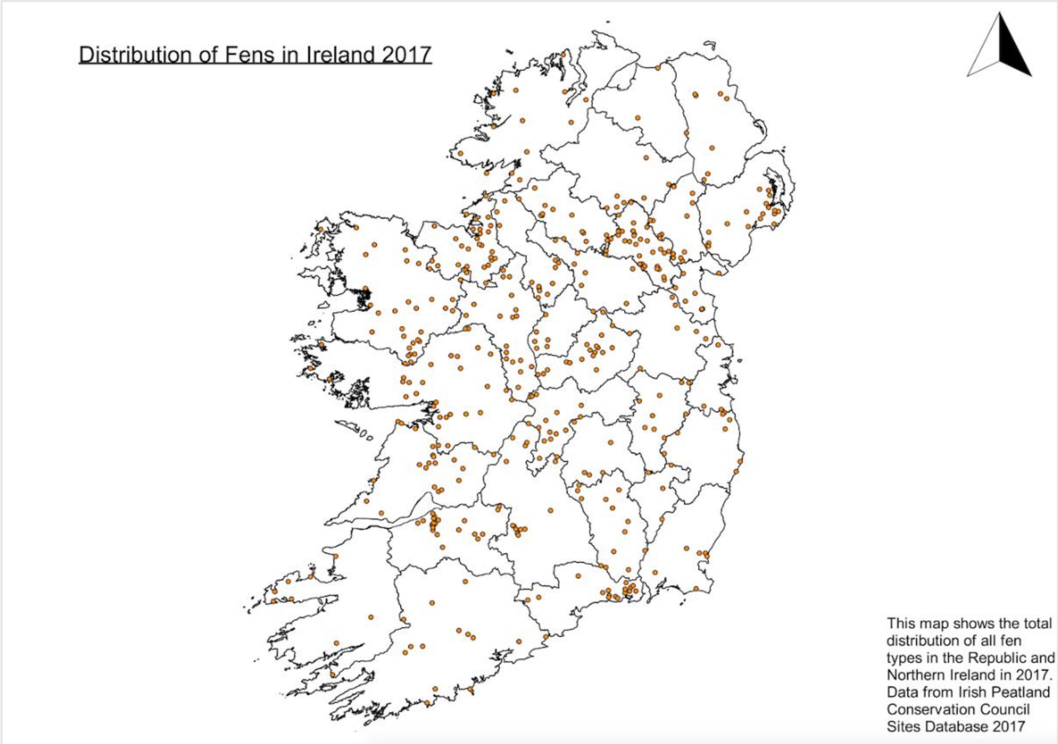


Figure 1.1 Fen distribution in Ireland in both the Republic and Northern Ireland (IPCC, 2017). The numbers reported by Foss (2017) are solely based on survey conducted in the Republic of Ireland.

Multiple classifications for calcareous fens are currently in use. These, however, seem to typify the current state of the wetland rather than to asses environmental controls on healthy fen vegetation. A hydrological approach is given in the WETMEX scheme (Wheeler et al, 2009), whereby different hydro morphologies driving the calcareous water supply to fens are outlined. Another approach often used to define a fen is based on the characteristic vegetation. However, some divergence exists between key habitats described in the EU directive and in guidance documents under the EU WFD. These different qualitative approaches therefore make a quantitative approach to defining 'healthy' calcareous fen difficult. This may have contributed to the apparent lack of quantitative research into environmental drivers that define Irish calcareous fens.

Furthermore, there remains the legal imperative of the EU WFD which has ecological criteria underlying is objectives for water management. The Directive links the management of aquifers and groundwater bodies to the state of connected, dependent wetlands – known as Groundwater Dependent Terrestrial Ecosystems (GWDTE). Calcareous fens are one such category of GWDTE. As the status of a particular groundwater body will depend upon the condition of any connected GWDTE, it is fundamental that criteria be available for determining that condition. Moreover, the Habitats Directive also demands that designated areas (NHAs, SACs) be protected from any potential impact. The intention of these Directives (and as translated into national legislation) is not only to protect and sustain natural resources but to encourage an understanding of how the relevant hydro-ecological systems function. As the literature indicates, calcareous fens present particular difficulties arising from their diversity both in terms of the range of vegetation and in underlying environmental, hydrological supporting conditions. In Ireland, there remains a gross lack of data on the hydrological conditions as well as agreement on a standardized quantitative ecological description. Nevertheless, the legislation demands the development of threshold values for both water quantity (e.g. water level) and water quality (mainly nutrients, nitrates and phosphorous) that can be used to determine risk to the sustainability of a fen.

The overall aim of the project, therefore, is to assess environmental supporting conditions for the ecosystems within fens by field studies of a number of calcareous fens. Specifically, hydrological and hydrochemical controls will be studied for this research.

1.2. Aims and objectives

The overall aim of the research is to define appropriate metrics in relation to the environmental supporting conditions for calcareous fens by carrying out field based studies of four such wetlands considered to feature a representative range of water quantity and/or quality impacts.

This will be achieved by targeting the following objectives:

- Synthesis of existing ecological / hydrological studies on fens
- A comprehensive desk study into land use, threats and impacts of four fen sites in order to find metrics for fens with a range in water quantity and/or quality impacts.
- Selection and instrumentation of four fens as exemplars for detailed field study to understand hydrology and water quality aspects
- Monitoring of the selected fens across a two year period to develop ecohydrological insights into their environmental supporting conditions
- Definition of appropriate metric(s) which can characterise these conditions required for healthy quality fen habitat (and variations)
- Provision of information on the ecohydrology of the fens to the remote sensing research in the wider EPA funded project

From these objectives the following outcomes are expected:

- Understanding of the current knowledge of Irish calcareous fens
- Monitored reference sites for further studies
- Knowledge of the hydrology of different fens types
- Knowledge of hydrochemistry of different fen types with better understanding of:

    - the nutrient cycles and fluxes in the different sediment layers of the fen as well as the role of the aquifer in nutrient fluxes
    - trace chemistry and its role in the fen
    - Redox conditions on the fen and the influence of these conditions on nutrient concentrations
- Hydrological and hydrochemical supporting conditions that indicate good quality fen vegetation in Irish calcareous fens

1.3. Thesis outline

The scope of this thesis is structured as follows:

- Review of available literature classifying calcareous fen habitats at topographical, hydrological and hydrochemical levels (Chapter 2).
- Review of available literature regarding environmental supporting conditions for calcareous fens at different scales (Chapter 2).
- Outline of research methodology regarding selection of suitable catchment and the description of the study catchment selection process as well as a description of each catchment with respect to ecology, hydrogeology, and land use (Chapter 3).
- Outline of research methodology regarding instrumentation and monitoring schedule and laboratory analysis techniques (Chapter 4).
- Hydrological and hydrochemical controls on fen habitat following from results discussed per research site. (Chapter 5 - 9)
- Quality assessment of fen vegetation of all research sites and the environmental controls that supports 'healthy' fen vegetation (Chapter 10).
- Conclusions and recommendations (Chapter 11).

2. Literature review

2.1. Introduction

Wetlands have been defined by the USEPA (2018) as follows:

> "*Wetlands are areas where water covers the soil, or is present either at or near the surface of the soil all year or for varying periods of time during the year, including during the growing season*."

The saturation of wetlands, if sustained for a long enough period, is able to promote aquatic processes indicated by various kinds of biological activity such as hydric soils and hydrophytic vegetation that are acclimated to a wet environment (Kellner, 2002). The wide variation of different wetland types encompassed within the broad definition of wetlands is caused by many regional and local differences such as climate and geology which in turn influence variations within soils, hydrology, water chemistry and vegetation within the wetlands. Furthermore, anthropogenic activity may influence the aquatic processes and alter the wetlands environmental settings. Many wetlands have a close connection between groundwater and the biosphere whereby hydrological and hydrochemical variations within the groundwater impact on this environment. Wetlands that critically depend on such groundwater variations are termed GWDTE's (groundwater dependent terrestrial ecosystems). GWDTE types are based on Annex I habitat classification under the European Union (EU) Habitats Directive (92/43/EEC) and examples of these wetlands are habitats such as species-rich Cladium fen, alkaline fens, petrifying springs, transition mires, active raised bogs, turloughs and many more.

Wetlands are among the most productive ecosystems in the world (de Groot et al., 2012; USEPA, 2018) and provide many ecosystem services such as climate regulation, hotspots of biodiversity, water purification, flood protection, recreation and ecotourism. Regulation and drainage have been common practice in wetlands all over Europe, although interventions did increase in the past 50-100 years. However, despite efforts to restore degraded wetlands, it is estimated that over two-thirds of European wetlands have been lost in the past 100 years by the EC (1995). Furthermore, an additional loss of 35% was recorded between 1970-2015 with the loss rate accelerating annually since 2000 (UNFCC, 2018).

This review focuses on fen wetlands, with the aim to provide insights as to how such wetlands function from a hydrological, biogeochemical and ecohydrological perspective, how they relate to the wider aquatic and terrestrial landscape, and how past management has influenced the wetlands and the wildlife they support. It concludes with a review of the current state of knowledge regarding fens in Ireland and provides a synthesis of contemporary ideas as to how to characterise the ecological status of such wetlands on an ongoing basis – i.e. the aim of the *Ecometrics* research project for which this research was conducted. This EPA funded project focuses on monitoring the status of Groundwater Dependent Terrestrial Ecosystems (GWDTEs) in turloughs, raised bogs and fens under the Water Framework Directive (WFD).

2.2 Legislation

Wetlands are widely recognised as endangered habitats because of their rapid decline which has resulted in a loss of the ecosystem services associated with favourable ecological, economic and social values. There is a need for preservation of these habitats for future generations and they are therefore protected through many national and international legislations and designations. Two legislative frameworks that are of particular importance in the preservation of wetlands are the Natura 2000 network and the EU Water Framework Directive (WFD) (2000/60/EC).

2.1.1. Natura 2000

The European network Natura 2000 is of prime importance for the protection and long-term survival of Europe's most valuable wetlands. This network is supported by two other directives; the EU Habitats Directive (EEC/92/43) and the EU Birds Directive (2009/147/EC). The EU Habitats Directive was implemented within Irish legislation 1997 (S.l. No. 94 of 1997) which protects endangered habitats and rare species from "adverse impacts". Sites that contain rare and endangered habitats, as well as species that have been chosen to be protected under the EU Habitats Directive, are proposed as Sites of Community Importance (SCIs). These decisions are based on habitat-types listed in Annex I and species listed in Annex II of the Directive that are represented in the wetland. Furthermore, these areas are automatically designated as Special Areas of Conservation (SACs). The EU Birds Directive (79/409/EEC) protects rare bird species and migratory species and follows Annex 1 to designate sites under Special Protection Areas (SPAs). These SCIs-SACs and SPAs designations form the main structure of the Natura 2000 network upon which EU member states have to act.

2.1.2. Water Framework Directive

The WFD is a framework for the comprehensive management of water resources in the European Union, within a common approach and with common objectives, principles and basic measures. It addresses inland surface waters, estuarine and coastal waters and groundwater. The fundamental objective of the WFD is to maintain the 'high status' of waters where it exists by holistic long-term sustainable water management, to prevent any deterioration in the existing status of waters and to achieve at least 'good status' in water bodies. Member States have to ensure that a coordinated approach is adopted for the achievement of the objectives of the WFD and for the implementation of *Programmes of Measures* for this purpose (www.wfdireland.ie). Implementation of such measures, under the WFD, will also benefit the objectives of the Birds and Habitats directives, as good ecological status in a wetland environment is fundamentally related to the status (both in terms of quantity and quality) of the source water body.

Integrated Water Resources Management (IWRM) is an approach to find a balance between the protection of natural resources for their long-term sustainability and the utilisation of the benefits and services provided by them in order to meet social and economic development imperatives. The core component of IWRM is the catchment area (River Basin), which is a physical unit within which water resources are managed. River Basin Management (RBM) focuses on all water bodies (ground and surface) within a given catchment area and attention is given to the relationship between these elements. The RBM approach has been adopted by EU legislation called the Water Framework Directive (WFD; 2000/60/EC). Wetlands that are dependent on groundwater and surface water as the primary hydrological input (fens, floodplain marshes and meadows), have been included in the framework of IWRM. It should be noted that some of the surface water dependent wetlands such as wet grasslands and wet woodlands are not considered as distinct water bodies within the WFD so can be missed out of such catchment measures which focus more on river, lake and groundwater bodies, and their dependent GWDTEs.

2.1.3. Other directives

Wetlands that are of the highest ecological importance can also be listed in The Ramsar List of Wetlands of International Importance, which designates worldwide sites of greatest significance. The Ramsar Convention requires signatory governments to designate and conserve wetlands that are considered as particularly good examples of a specific type of wetland, characteristic of its region. The convention notes the presence of rare, vulnerable, endemic or endangered plants or animals as a factor in determining international importance.

Finally, the Wildlife Act 1976 and its amendment of 2000, which provides for the conservation of plants, animals and wildlife habitats of importance is another piece of legislation that is relevant with respect to wetlands in Ireland.

2.2. Fen definition

2.2.1. Fens

Fens are peat forming wetlands that are fed by groundwater as well as surface water and have a water table near or at the surface throughout the whole year (Kellner, 2002; McBride *et al.*, 2010; Aggenbach *et al.*, 2013). Unlike bog habitats, they are not usually dominated by *Sphagnum* and are generally more alkaline and tend to be dominated by sedges and bryophytes (with the exception of base-poor fens, see Section 4.3.1). Fen occurrence is determined in large part by the discharge of groundwater near the vegetation root zone (Bedford & Godwin, 2003). Hence, topography, hydrology and geology all play important roles in determining how a fen develops and is maintained (McBride et al., 2010) – see Section 4.2.1. Fens are most commonly defined by their association with particular landscape features as well as according to the source of water which feeds the fen.

Generally, they are found to be poor in nitrogen and phosphorus, of which the latter tends to be the limiting nutrient in fen systems – see Section 5.2. However, they are also able to support a much larger amount of different flora and fauna than bogs because of their base-rich nature (Foss, 2007).

Their diverse and often rare vegetation community is one of the reasons that make this type of wetland of high conservational value. Bedford et al. (2001) report that as many as 30 vascular and bryophyte plant species can occur in these rich fens per square meter and up to 60 or more per 100 square meter (Bedford & Godwin, 2003). Fens often occur in mosaics with other wetland types such as bogs, open water and reed beds in which they occur in limited extent. However, they are also found as discrete habitats in their own right.

There are three Annex I fen habitats designated in the Habitats Directive: Transition mires (7140), *Cladium* fen (7210) and Alkaline fen (7230). Fens can have one or more of these habitats. In addition, petrifying springs with tufa formation (*Cratoneurion*) (7220) can occur in combination with the aforementioned fen habitats but are rarely found on their own.

2.2.2. Calcareous fens

Calcareous fens (or alkaline fens) are classified as a system being principally fed by base rich water with high concentrations of calcium, bicarbonate and magnesium (Kuczyńska, 2008; Duval, 2010). They are species rich and support small sedge and brown moss communities (Sefferova et al., 2008). Calcareous fens are supported by specific environmental conditions, which are detailed in Section 4.

2.3. Calcareous fens conceptual model

A conceptual model of calcareous fens has been developed that follows a top-down approach (Mitch & Gosselink, 2007). The themes in this model are thus discussed in the same order in the following sub-sections. Climate and geology determine where calcareous fens can form in the landscape and also regulate the hydrological regime, which in turn then controls the development of the physicochemical soil environment and its own microclimate. Together the hydrology and biogeochemistry dictate the range and composition of species that are capable of thriving in the habitat.

2.3.1. Development

Fens began to develop 10,000 years ago after the Ice Age left depressions in the landscape. These were subsequently occupied by shallow lakes in which anaerobic conditions occurred (see Figure 2.1). Without oxygen the complete decomposition of plant material is prevented. In time this un-decomposed plant material forms a thick layer of peat that rises towards the surface of the lake. Eventually the surface peat is invaded by sedges to form a fen.

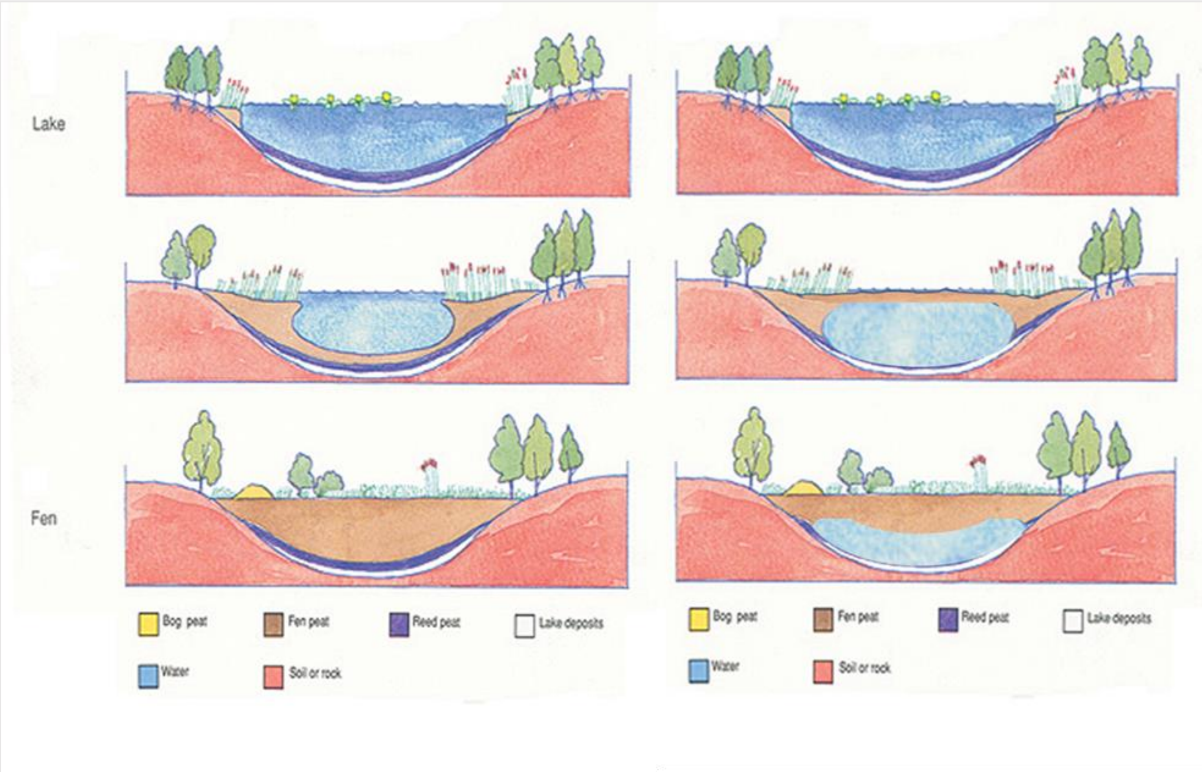


Figure 2.1. Development of an anaerobic lake into a fen (left) (IPCC, 2017) and the development of a floating raft fen (Adapted from IPCC (2017)).

There are different factors that determine how fens develop, as contained in the "Wetland Framework" (Wheeler et al., 2009; Whiteman et al. 2009) which describes the importance of water supply mechanisms in determining the distribution and composition of plant communities. Floating raft fens, for example (see Figure 2.1), may develop when root mats expand horizontally into the open water and accumulate new organic matter over time. Such fens may also be called "transition mires and quaking bogs" and the fen Scragh Bog is an excellent example of this fen type. This results in more decomposed material in the centre where the root mat is older compared to the younger edges (Stofberg et al, 2016). More information on the development of different geomorphological vegetation types can be found in Section 4.2.1.

2.3.2. Hydrology

Calcareous fens are reliant on groundwater and surface water for their predominant supply of water and nutrients, as opposed to just direct rainfall (Holden et al., 2004). The connection of this groundwater discharge zone results in a stable annual water table that is continuously at or near the land surface, but never inundated for significant lengths of time (Duval, 2010; Sampath et al., 2016). Precipitation, evapotranspiration, surface and groundwater inflows and outflows create a hydroperiod which controls the fen water balance (Mitch & Gosselink, 2007). Different partitioning of water within the different components of the water balance that influence the hydroperiod have a big influence on biogeochemical and ecological functioning of a calcareous fen.

2.3.2.1. Hydromorphologic classification

There are two major groups that subdivide fens based upon their topography and hydrology: topogenous and soligenous fens. These groups are then subdivided into a total of six fen types in Ireland that have been recognised during a nationwide fen survey by Crushell (2000). Topogenous fens are subdivided into open water transition fens, floodplain fens, and basin fen. There are also three subtypes recognised in soligenous fens: valley fens, flush fens and spring fens.

i) Topogenous fens

Topogenous fens are formed where the landscape results in a basin-type water collection system. These systems can be associated with glacial or peri-glacial processes such as kettle holes, or in solution hollows on limestone (McBride et al., 2010). This type of fen has little surface water outputs and water fluctuations occur in a vertical direction (Foss & Crushell, 2008). Surface run-off from adjacent slopes can be an important source of water in addition to groundwater, although this obviously depends on the surrounding topography.

Open water transition fen

These form on lake edges, where surface water is supplied from the open water throughout the year. The fen is situated on the landward side with emerging reed vegetation extending on the other side into deeper water regions. Fens surrounding Lough Corrib in county Galway are an example of one such fen (Crushell, 2000).

Floodplain fen

This type is situated in waterlogged floodplains alongside rivers or streams where depressions with still standing water, fed by surface water discharges, allow for the development and build-up of fen vegetation. Fens of this type can be found among the wet grasslands along the river Shannon and Suck. (Crushell, 2000).

Basin fen

Basin fens develop in waterlogged basins where little lateral water flow occurs and additional open water may be present. This type often supports floating raft vegetation where, as stated earlier, root mats expand horizontally into the open water and accumulate new organic matter over time. An example of such a fen would be at Scragh Bog, County Westmeath.

ii) Soligenous fens

Soligenous fens are formed on sloping terrain and are supplied by a continuous through-flow of water. Smaller areas of this fen type may also exist in a mosaic within bogs or mires where they are associated with runoff routes.

Valley fens

These occur on the bottom of shallow valleys. Water movement may not be obvious at first since the slope within these fens can be very gentle. This system's main source is from springs and seepages in the surrounding valley which are usually base-rich. A prime example of this fen type is Pollardstown Fen, County Kildare.

Flush fens

Flush fens develop as small areas in a complex with other fen and peatland types, more commonly in blanket bogs. The localised flow within the flush areas supplies more mineral rich water than the surrounding peatland areas, resulting in the development of rich fen vegetation.

Spring fens

Spring fens develop around permanent freshwater saturated soil or rock (spring) or discrete zone (seepage). The water often upwells from a permeable and impermeable rock or soil interface. The water from these interfaces deposits a white crust also known as tufa on the fen surface. Examples of these spring fens are found in areas within Pollardstown Fen, County Kildare.

2.3.2.2. Water sources

i) Atmospheric precipitation

All fens are fed by direct precipitation such as rain, snow, mist, frost and condensation.

ii) Surface

Surface water inputs into fens can occur as a direct response to rainfall within the catchment and have an episodic influence which is usually stronger in the winter and spring months (McBride et al., 2010). Surface water is also used for telluric water, i.e. water that has been in contact with the mineral ground (as opposed to direct precipitation) but is not groundwater.

The WETMEC study (Wheeler et al., 2009) divided surface water systems into two mechanisms that are 'upslope' and 'downslope'. Upslope water runoff enters a wetland site from the upland adjacent margins, usually as rain generated overland surface runoff, or through a stream or ditch. Downslope runoff happens in surface water bodies adjacent to the downslope side of the wetland. These may include rivers, streams, lakes as well as ditches. Examples of these different flow systems are shown in Figure 2.2.

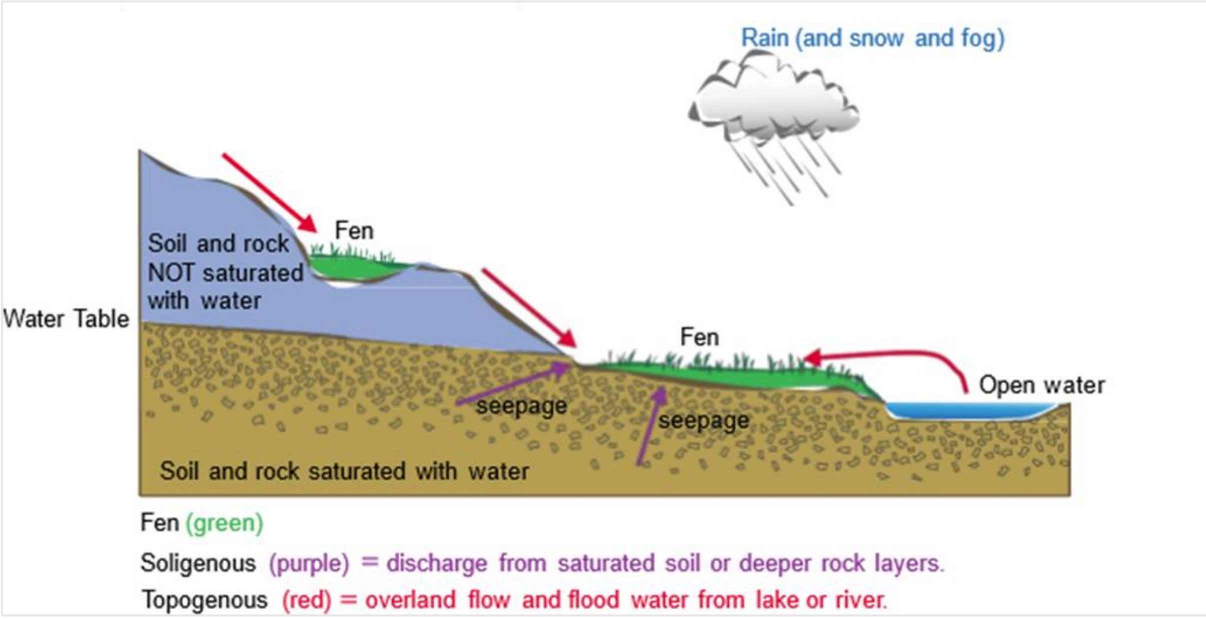


Figure 2.2. Water movement in soligenous and topogenous fens (McBride et al., 2010)

iii) Groundwater

Groundwater discharge is richer in bases (i.e. alkalinity) than rainwater. The flow into a fen from groundwater is generally considered to be a fairly stable part of the catchment-scale hydrological cycle, albeit with higher discharges from the underlying bedrock or drift aquifer during winter and spring (Wheeler et al., 2009).

At a more local scale for fens, when surface soil and drift deposits happen to be permeable, groundwater will flow into the overlying fen peat via diffuse upward discharge. In cases where the underlying bedrock is not permeable, groundwater will most likely discharge at the margins of soil or drift deposits through discrete springs or seepages. In reality, a fen receives groundwater through a combination of these two mechanisms as seen in Figure 2.2 (McBride et al., 2010). However, the groundwater flow in shallow drift is normally derived within a close distance of the wetland with consequential low capacity of groundwater storage. This means that groundwater discharge can be significantly reduced or even interrupted during the summer and autumn periods (Wheeler et al., 2009) making the wetland vulnerable to droughts. The Wetland Framework (Wheeler et al., 2009) describes the different water supply mechanisms into fens.

In order for a fen to receive (and discharge) a constant flow of groundwater, an underlying or adjacent aquifer usually needs to be present. This phenomenon is usually described as 'groundwater outflow from a mineral aquifer' (Wheeler et al., 2009). The high storage capacity ensures a more constant flow of groundwater discharge, which will also be maintained over the summer and autumn periods.

If groundwater directly discharges onto a wetland surface, forming pools and streams, it is still referred to as groundwater. However, in the case where streams or other surface water bodies feed the wetland that originate well outside the fen, then the water source is described as surface water (Wheeler, et al., 2009).

2.3.3. Hydrochemistry

2.3.3.1. Chemical and floristic classification

i) Base poor

Fens that are base poor are mainly associated with lowland heaths, where the wetland receives water from base-poor rock such as sandstones and granites (McBride et al., 2010). The water found in base poor fens is characterised with a pH below 5.5 and its floristic composition is characterised by short vegetation dominated by *Sphagnum* mosses. Small sedges found in the fen belong to the *Caricetalia nigrae* order with species such as bottle sedge (*Carex rostrate*) abs Black sedge (*Carex nigra*). Rushes such as soft rush (*Juncus effusus*) and Jointed Rush (*Juncus articulatus*) may also be an indicator species for this fen type (Crushell, 2000).

ii) Base rich

Fens categorised as base rich are fed by mineral rich calcareous water and are usually confined to lowlands or upland areas with underlying limestone, therefore making the peat substrate alkaline. Water in these fens is found with a pH higher than 5.5 and the mineral rich water tends to support a higher amount of biodiversity than base poor fens (McBride et al., 2010).

Base rich fens are dominated by vegetation of the order *Caricetalia davallianae* and appear brown in colour due to the upstanding part of this vegetation being dead throughout most of the year. Vegetation of base rich fens mostly consists of Black bog-rush (*Schoenus nigricans*) and small to medium size sedges from the *Caricetalia nigrae* order. However, this fen type supports a lot more vegetation types such as rushes, brown mosses and insectivorous plant and orchids. Reeds, bulrush and aquatic plants flourish in the wetter part of the fen and the wooded part (also known as fen carr) supports willow and birch (Crushell, 2000).

2.3.3.2. Nutrients

i) Key nutrients

The most significant nutrients in fens, also referred to as plant macronutrients, consist of nitrogen (N), phosphorus (P) and potassium (K) (McBride et al., 2010). These nutrients typically limit plant growth in a fen which makes them significant factors in nutrient enrichment except for potassium which is rarely a limiting factor in enrichment. Elements such as oxygen (O) and carbon (C) are needed for plant growth, but again are not often seen as the limiting factor (see Section 5.2).
Based on C: N ratios, a subdivision can be made between oligotrophic, mesotrophic and eutrophic fens. Base rich fens mainly seem to be characterised by mesotrophic and eutrophic conditions, however some highly calcareous fens were found with extremely low nutrient concentrations due to the immobilisation of phosphorus by co-precipitation with calcite (Wheeler and Proctor, 2000). Ions such as calcium (Ca^{2+}), magnesium (Mg^{2+}) and sodium (Na^{+}) have an ameliorating effect on the acidity of a wetland habitat. Elements referred to as micronutrients (aluminium, magnesium, copper, iron, selenium) also have important roles in fen hydrochemistry. Base poor conditions are characterised by low calcium concentrations and the presence of chloride (Cl^{-}) and sulphate (SO^{4-}) as main inorganic anions whereas base rich conditions are dominated by high calcium and bicarbonate (HCO_{3-}) (Wheeler and Proctor, 2000).
The ecological characteristics of fens as described by Wheeler et al., (2009) in WETMEC 8 (Groundwater-Fed Bottoms with Aquitard) and WETMEC 9 (Groundwater-Fed Bottoms) confirms that fens exist in a range from base-poor to base-rich and from oligotrophic to eutrophic, but that this mainly depends on groundwater source and substratum characteristics. However, most examples found in that study appeared to be base-rich/sub-neutral and mesotrophic.

ii) Nutrient sources

Nutrients can be derived from both natural and anthropogenic sources and transmitted via terrestrial and atmospheric pathways. Water is the main transport mechanism, providing a pathway for nutrient enrichment in fens. The source of nutrients are often linked to catchment land use (different inputs from agriculture, on-site wastewater treatment etc.), but may also be affected by catchment geology (Mitch & Gosselink, 2007; Wheeler et al., 2009). Surface water which might enter fens via streams or other surface flow often carries nitrates and phosphorus. Soil erosion, for example, often leads to surface water transportation of phosphorus-rich sediments into a fen (McBride et al., 2010).
Atmospheric deposition is also considered to be a significant pathway for nitrogen to GWDTEs, carrying key pollutants such as nitrous oxides and ammonia. Ammonia is associated with highly intensive agricultural systems whereas nitrous oxides typically originate from fossil fuel burning.

This makes fens near farms and roads more vulnerable to nutrient pollution. Furthermore, point sources from sewage works, farm discharge or diffuse sources received by aerial or water borne nutrient enrichment can also be a major cause for eutrophication in fens. Fens also have internal nutrient cycling that involves plant available inorganic nutrient. These cycles are strongly affected by the water table and hydrological changes such as drainage may result in increased nutrient concentrations (see later discussion in Section 5.2).

2.3.4. Vegetation

2.3.4.1. Habitat classification

Fens have also been classified looking primarily at the floristic composition of vegetation types that correspond to certain topographic and hydrological types listed earlier. Certain key habitat features are considered in this classification. Using this information, the National Fen survey of Ireland (NFS) have recognised a total of six fen categories which are different to the six hydromorphological types defined in Section 4.2.1 (Foss & Crushell, 2008). Four of these types are being recognised in this study; 1) alkaline fen, 2) calcareous fens with Cladium, 3) transition mire and 4) petrifying spring with tufa.

This research study will mainly focus on two fen habitat types: alkaline fen and calcareous fens with *Cladium mariscus*. However, transition mire and petrifying spring with tufa are included here as well since they are featured existing in areas together with alkaline and calcareous fens. Other classification schemes exist that relate to these habitat types such as the EU Habitats Directive (Council Directive 92/43/EEC), CORINE Habitat and Fossit Habitat schemes (Fossit, 2000; EPA, 2003). These schemes and their relationship to the NSF classification are presented in Table 2.1.

Table 2.1. Comparison of different existing fen habitat classifications (Foss, 2007; Foss & Crushell, 2008).

NFS Fen Classification Scheme	**EU Habitats Directive Habitat**	**CORINE Habitat**	**Fossitt Habitat Scheme**
Alkaline fen	7230 Alkaline fens	542 Rich Fens Caricion davallianae, 5421 Black bog rush fens, 5422 Fens not Schoenus dominated	PF1 Rich fens and flushes
Calcareous fens with *Cladium mariscus*	7210 *Calcareous fens with Cladium mariscus and species of the Caricion davallianae	533 Fen Sedge Beds, 5331 Fen Cladium Beds	PF1 Rich fen and flush
Transition Mire	7140 Transition mires and quaking bogs	545 Transition mires	PF3 Transition mire and quaking bog

Petrifying Spring with Tufa	7220 * Petrifying springs with tufa formation (Cratoneurion)	5412 Hard Water Springs Cratoneurion	FP1 Calcareous Springs

i) Alkaline fen

The phytosociological classification for this fen type is *Caricetalia davallianae* (Foss & Crushell, 2008). Wetlands mostly or largely occupied by peat- or tufa-producing small sedge and brown moss communities developed on soils permanently waterlogged, with a soligenous or topogenous base-rich, often calcareous water supply, and with the water table at, or slightly above or below, the substratum (EEA, 2019; JNCC, 2019).

ii) Calcareous fens with Cladium mariscus

The phytosociological classification for this fen type is *Cladietum marisci* and *Caricetalia davallianae* (Foss & Crushell, 2008; EEA, 2019). Sites may exist under three different conditions: 1) sites that have a mixture of closed, species-poor *Cladium* beds, which at their margins have transitions to species-rich small-sedge mire vegetation, 2) sites where *Cladium* beds retain their species-richness owing to management and 3) situations where *Cladium* fen is inherently species-rich, possibly owing to the fact that conditions do not allow the *Cladium* to grow vigorously and dominate the vegetation (JNCC, 2019).

iii) Transition mire

The phytosociological classification for this fen type is *Scheuchzerietalia palustris, Caricetalia nigrae* and *Caricetalia davallianae* (Foss & Crushell, 2008). Transition mires have floristic composition and general ecological characteristics that are transitional between acid bog and 7230 Alkaline fens, in which the surface conditions range from markedly acidic to slightly base-rich (JNCC, 2019).

iv) Petrifying Spring with Tufa

The phytosociological classification for this fen type is Cratoneurion (Foss & Crushell, 2008). Hard water springs with active formation of travertine or tufa. These formations are found in such diverse environments as forests or open countryside (EEA, 2019). On contact with the air, carbon dioxide is lost from the water and a hard deposit of calcium carbonate (tufa) is formed. These conditions occur most often in areas underlain by limestone or other calcareous rocks (JNCC, 2019).

2.4. Environmental controls on calcareous fen vegetation

Section 4 followed a top-down approach of calcareous fens in order to outline a general perceived conceptual model. In this section a more bottom-up approach will be explored in order to help identify the components of water supply and chemical conditions that sustain habitat features of conservation importance in wetlands.

2.4.1. Hydrology

As stated previously, calcareous fens have water levels near or just below their surface throughout the hydrological year and can be supplied by groundwater under base rich conditions, or in some cases by calcium rich surface waters (for example, in East Anglia, UK). Large variations in biodiversity exist as well as differences in hydrological influences in calcareous fens. Furthermore the balance between system inputs and outputs, such as precipitation and potential evapotranspiration, are controlled significantly by type of land cover and land management and are also the drivers for potential drainage or flooding (Holden et al., 2004; Verhoeven et al., 2011). Even though the literature states that land management has great control on the hydrology in fens, little research appears to have been carried out on how these controls, causing hydrological variation, affect the quality of fen vegetation. Nevertheless, an attempt is made here to describe and quantify hydrology and its effect on calcareous fen vegetation. For the rest of the review, the fens discussed are characterised by high-productive and species rich vegetation and a base-rich groundwater supply unless stated otherwise.

2.4.1.1. Reaction to catchment and climate characteristics

The literature states that the type of ecosystems found in fens depends strongly on the dominant hydrological regime (McBride et al., 2010). However, this regime is strongly influenced by both the connection to and isolation from other waters. Fen biological diversity is not only controlled by groundwater flows and its chemistry (Simkin, 2012; Stofberg et al., 2016) but it also relies some sort of isolation from other surface waters (Amon et al., 2002) which implies fens are usually saturated by groundwater but rarely flooded by adjacent surface waters (Bedford & Godwin, 2003). Furthermore, the hydrological regime is controlled by different hydraulic gradients which vary between recharge and discharge zones in both groundwater and surface waters. These gradients may occur naturally, or may have been created by some anthropogenic interference at some point. Furthermore, variations within climatic characteristics influence the hydrological regime which in turn controls the water table within these wetlands. In order to sustain a good quality fen ecosystem, an approach to understanding how catchment and climatic characteristics drive the internal water table therefore needs to be made. It should also be appreciated that larger systems are often interconnected with the implication that investigations (and conservation efforts) must be expanded from focusing on individual fens and their immediate surroundings, to studying the much larger and inter-connected hydrologic network that sustains multiple fens. (Sampath et al., 2016).

i. Internal water table

Mettrop et al. (2015) studied the interactive effects of short-term water table fluctuations on extracted peat cores from two species-rich fens in the Netherlands. The main difference between these fens was in their dominant form of water supply. The '*Stobbenribben'* fen is characterised by a supply of base-rich surface water and dominated by the highly endangered brownmoss *Scorpidium scorpioides* and multiple species of the family Cyperaceae. Lowering of the water table in a mesocosm experiment by 15 cm for a period of seven weeks resulted in soil subsidence and an increase of N-availability which then may have led to the decline of *Scorpidium scorpioides*.
A lower water table proved to negatively affect the dominant *Calliergon giganteum* in the groundwater supplied 'Binnenpolder Tienhoven' fen in the same manner. However, this species did show a clear recovery when inundated for seven weeks after the experimental drought whereas the vitality of *S. scorpioides* remained low. A similar approach was investigated on *H. vernicosus* and *Scorpidium scorpioides* cores taken from *Stobbenribben* for a longer period of time (30 weeks) (Cusell et al., 2013). Again, low water levels negatively affected the brownmoss species, whereas inundation had yielded high levels, which was also reflected by higher growth rates than in the control cores.
Other authors also agree that maintaining high water tables is favourable for fen target communities as well as indicator species. Maassen et al. (2015) argue that the key to restore decomposed fen is to provide the site with a continuous high water table. However, restoring a site by raising the water table may result in a shift of biotic processes. Wyatt et al. (2012) found that primary productivity in algae was considerably higher with low water tables and that algal productivity peaked following seasonal maxima in nutrient concentrations. Since algae are able to rapidly assimilate available nutrients, rewetting of fens after droughts may have great effect on the algal productivity possibly affecting the existing fen species in the site such as brown moss, *Sphagnum*, and emergent vascular flora, including *Equisetum, Carex*, and *Potentilla*.
However, it has been argued that more natural fluctuating water levels are required to improve conditions in specific conservation areas which have been isolated as a management option in order to prevent the input of polluted surface water, such as a species rich hay meadow located on the flood plain of a river (Loeb et al., 2008). This may cause a seasonal decrease in dry periods and increase of inflow of base-rich water during wetter periods which can also have an adverse effect on internal biotic processes in the fen peat (Cusell et al., 2013). Acidification by oxidation and higher mineralisation rates (Lucassen et al., 2002) may occur during periods of drainage, while periods of inundation may lead to leaching of nutrients by surface water supply, resulting in mobilisation of phosphates and eutrophication (Surridge et al., 2005) as well as sulphide and

ammonium toxicity following reduction processes (Smolders & Roelofs, 2009). Acidification by oxidation might occur in numerous ways depending on the elements present in the peat. For example oxidation in a sulphur rich fen will result in a shift of sulphide to sulphate through the exchange of electrons.

More complex processes in fens that control fen target vegetation make it a challenge to predict biotic reactions to fluctuations of the water table. For example, it has been found that the redox reactions involving nitrate, sulphate and phosphate in groundwater fed wetlands may support a chemical reaction where sulphate ions may also cause phosphorus release (Lucassen et al., 2004). The apparent optimum water table range can however vary within different fen ecological communities. Hydrological guidelines for NVC class have been described by Wheeler et al. (2004) for fen communities such as *Schoenus nigricans – Juncus subnodulosus* mire (NVC class M13). Examples of these habitats with a high species richness occur only in locations that exhibit a water table generally at the fen surface in winter and summer. Here, groundwater discharge causes flushing within the site. However, a fluctuating subsurface water table may be the 'natural' condition of some (less rich) stands occupying intermittent seepages. In contrast to type M13, fen communities such as *Molinia caerulea - Cirsium dissectum* (NVC class M24) may occupy a broad band of subsurface summer water tables. Sites where relatively high summer water tables can be observed tend to show affinity towards M13. A relatively deep subsurface water table may be also be great for supporting these community M24. Furthermore, M24 is not normally associated with inundation, except to a very minor degree in the winter at particularly wet sites.

Attempts have been made to directly correlate hydrological distributions to vegetation communities. Large et al. (2007) attempted to correlate the potential response in vegetation communitiees to different management options, creating a conceptual model of these correlations (see Figure 2.3). The model shows the transition of fen communities along moisture gradients with favourable hydrological conditions of fen target communities such as *Schoenus nigricans – Juncus subnodulosus* mire (NVC class M13) and *Phragmites australis – Eupatorium cannabinum* tall-herb (NVC class S25). Conceptual models like this may be suited to offer some quantification of suitable hydrological conditions that have to be maintained over longer timescales in order to aid site restoration (Wheeler and Shaw, 2010). However, the metrics these models provide are greatly dependant on surrounding land use as well as climate, making this approach useful for fens in the UK. Fens not sharing the climate and land use presented by this study would have to be assessed separately in order to provide metrics better suited to its current conditions. It is also important to note that in recent years the UK Natural England are moving away from defining water level regimes towards more focus on the restoration natural hydrological functioning.

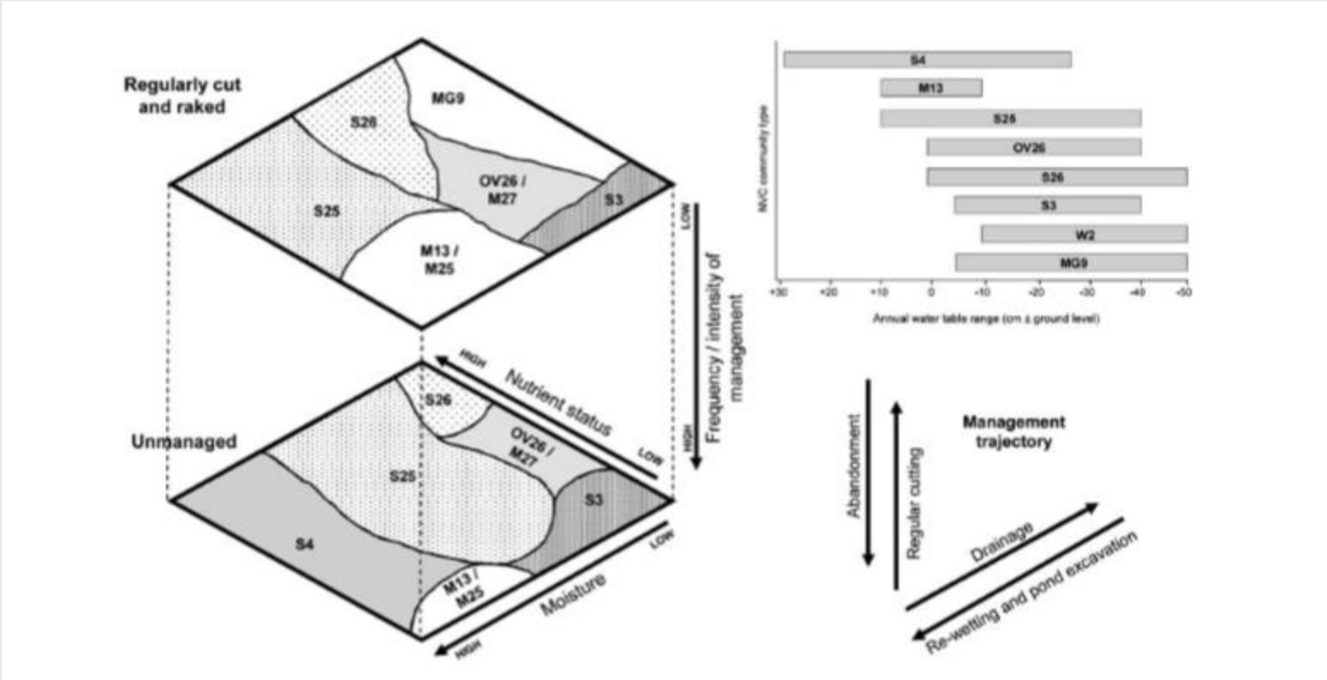


Figure 2.3. Conceptual model of potential vegetation community ranges in response to management. The target of rich fen communities such as Schoenus nigricans – Juncus subnodulosus *mire (NVC class M13) seems to have a small annual water table change compared to the other vegetation classes (Large et al., 2007).*

ii. Recharge

According to Sampath et al. (2015) a multi-scale groundwater- surface water modelling approach is needed in order to understand the hydrologic processes that support the resilience of fens. Using this approach on a geographically-isolated fen in southern Michigan, local recharge from an adjacent small pond and associated shallow outwash aquifer was found to be the most important source of water measured in the flux quantity and the calculated travel time in flow paths to the fen.

Knowing where local recharge reaches the fen is also important in order to understand the influence that the water chemistry might have on fen ecology. Van Wirdum (1991) found that during dry weather the change in hydraulic head gradient causes lateral flow to travel from open water into floating root mat fens in the Netherlands. Furthermore, his QUAGSOLVE model simulated lateral flow and transport of tracers in the ‘preferential flow channel’ under the root mat and also found a vertical exchange between this channel and the root mat under steady state conditions. Therefore, this shows that it is not only interactions between atmospheric and surface water boundaries that can affect the flow and transport solutes, but subsurface seepage as well.

Low water tables following droughts within fens generally create negative effects on fen vegetation, although Schilling & Jacobson (2016) explain that in certain situations fens can also exhibit resilience to severe long-term droughts. For example, a tallgrass fen community in Iowa, dominated by *Carex* spp., *Juncus* spp. and *Schoenoplectus tabernaemontani*, proved to be resilient against a water table fluctuation of 55 cm along its boundary with a highly agricultural terrace during an extreme drought that lasted approximately one and half years. Although, the hydraulic head in the fen itself varied by only 18 cm due to buffering by the steady regional groundwater flow, thus providing resilience against larger fluctuations in the smaller, more local groundwater tables.

iii. Drainage

Even though fens are generally situated in geological settings that prevent them from drying out, thus helping their resilience, many are still sensitive to relative small changes in the hydrological cycle (van Diggelen et al., 2006). Anthropogenic interference such as large-scale drainage of the surroundings for agricultural purposes or for irrigation, including groundwater abstraction for management and construction within local and regional catchments, can lead to decreased groundwater flows to calcareous fens. This drainage in or near the fen may cause the water table to drop, creating a number of detrimental changes. These changes have been described in the Wetland Framework for England and Wales by Wheeler et al. (2009) in numerous cases. Most examples of groundwater-fed bottoms with aquitard (WETMEC 8) have been influenced by drainage to some degree. It has been suggested that marginal peat cuttings probably increase the penetration of base-rich water into the margins of the valley bottoms, due to the creation of sub-surface water flow paths. WETMEC 8 sites can be vulnerable to further drainage, especially in sites which support vegetation often associated with higher water tables. If, however, the sites support established vegetation which is compatible with fairly low water tables (such as *Molinia caerulea* – *Cirsium dissectum* fen, NVC Class 24) it may be that partial drainage is part of the conservation interest.

Many groundwater-fed bottoms (WETMEC 9) seem to have become drier than once was the case due to groundwater abstraction as well as an increase of drainage of the valley bottoms. In some cases, such as the Broadland sites or Bugg's Hole (Thelnetham) and Thelnetham Middle Fen, drainage occurs in the peat, separating the sites from the river. In other areas some the sites that are now referred to as groundwater-fed bottoms would have been designated to seepage percolation basins (WETMEC 13) before drainage occurred and in some sites small, wetter areas within WETMEC 9 still have clear connections to WETMEC 13.

If the water table within WETMEC 9 is strongly influenced by deepening of adjoining watercourses and by groundwater abstraction, then further drainage may occur. The absence of basal aquitard layers may make these sites more susceptible to drainage than examples of WETMEC 8. There has been a reported dramatic loss of *Carex rostrata – Calliergon cuspidatum / giganteum* (NVC class M9) and *Schoenus nigricans– Juncus subnodulosus* (NVC class M13) by Ballamy and Rose (1961)

Degradation of organic soil in the surface peat can result in decomposition, shrinkage and subsidence of the soil surface up to several decimetres (Amon et al., 2002; Holden et al., 2004; Cabezas et al., 2013; Grygoruk et al., 2015). As the water table is lowered shrinkage occurs because the upper peat collapses, causing the bulk density to increase (Holden et al., 2004). For example, increases up to 63% in the upper 40 cm in a forest peatland in Alberta were found within a few years of drainage (Silins & Rothwell, 1998). Equally, Gebhardt et al. (2010) found 65% shrinkage in peats underlain by clay in northern Germany. Shrinkage is also increased when aerobic bacterial activity more readily decomposes near surface peat which is no longer anaerobic (Holden et al., 2004).

Subsidence is also associated with the result of decomposition as well as consolidation of dry peat in surface layers. Furthermore, collapsing of macropores which are important runoff pathways in peat (Silins & Rothwell, 1998) enhances capillary action, resulting in an increase of water loss through the subsurface layers (Holden et al., 2004). Therefore, the whole peat mass dries out more and shrinks since undecomposed peatland can yield as much as 80% of their saturated water content to drainage (Boelter, 1968).

Other effects, such as an increase in the relative importance of rainwater (due, for example to increased groundwater abstractions) can slowly cause acidification of the top layer, and are often not immediately noticed in the water table inside the fen as it depends on the amount of acid produced and the buffering capacity inside the soil (Wassen, 1995; van Diggelen et al., 2006). Examples of this have been shown to occur in the Norfolk Broads. A lower water table can also lead to increased mineralisation rates. This can increase biomass production, but can also cause a shift with respect of the limiting nutrient (van Diggelen et al., 2006; Wassen et al., 2009). This abiotic shift can create an altered competition intensity, causing vegetation and productivity rates to change to accordingly.

Changes in the hydrological regime can prove to have a long-lasting effect. Palaeo-hydrological reconstruction in intensively managed low-productive fen reserves (van Loon et al., 2009) proved that a shift in the predominant groundwater discharge mechanism from regional overland flow to local drain discharge caused environmental degradation of fens and loss of fen vegetation species in intensively managed regions.

There are a lot of examples of fens that were 'managed' by drainage in order to create arable land for agriculture. A peatland in Rostock, Germany was drained by ditches and tile drains and is now managed as an intensive meadow utilising mineral fertiliser (Tiemeyer & Kahle, 2014). The arable land around the fen was not drained but was used for intensive conventional crop production and fertilised using cattle slurry. This hydrological setting had a major impact on the control of both dissolved organic carbon (DOC) and nitrogen (N) concentrations in the fen, particularly during higher discharges. Drainage and peat extraction in another fen in Germany (Cabezas et al., 2013) in the 19th century was followed by a complex dewatering and more intense agriculture around the mid-1970s that has caused the almost complete disappearance of species of natural occurring *Caricetalia davallianae*. Furthermore, these activities caused severe peat degradation and subsidence of the soil surface by several cm. Another fen in Middle Biebrza Basin, Poland was also transformed as a result of the construction of Woznawiejski Canal which has drained the basin systematically since 1950 (Grygoruk et al., 2015). The peat forming wetland, which previously had an average water table of 0.4 m below the ground level (and for which plans are now being made to re-establish such levels) was transformed into a peat losing fen meadow and caused CO_2 to release into the atmosphere. Additionally, the meadow gradually lost nutrients from the mineralised peat into the ground and surface water. However, due to long term hay removal, low productive habitat conditions were created (Wassen, 1995) and the area still remains inhabited by *Molinia* fen meadows (Natura 2000 habitat 6410) and contain several rare species. These species, however, are not found in typical alkaline or calcareous fens. This site has subsequently been included in a restoration project (Zockler et al, 2000) with the objectives to restore of the peat formation and water purification functions. So far the increase of water table by Biebrza was successful; a considerable increase was achieved in only a few months and further steps to ensure long-term success system of the river flood are reportedly planned.

Finally, it should be noted that artificial drainage rarely occurs in isolation; burning, grazing, afforestation, fertilization can all accompany drainage. Thus, the effectiveness of any restoration strategy does not rest solely on the restoration technique adopted but on how well integrated the catchment management schemes are and how well we understand the interacting mechanisms. Nonlinear restoration strategies are often needed and much more work is required to examine the hydrological and hydrochemical processes surrounding artificial drainage and peatland restoration (Holden et al., 2004).

iv. Flooding

In contrast to drainage, flooding may happen in calcareous fens as well, although it is reported less often (van Diggelen et al., 2006). Altering water regimes, such as building dams and other activities can flood the area, but the effect on ecological conditions in fens depends on the regularity and the duration of the inundation. If this happens regularly, the physical as well as chemical properties that support the existing vegetation will change. Flooding will cause an increase of the surface water component and change in water chemistry, causing nutrient dynamics to change (van Diggelen et al., 2006). If greater nutrient loads and/or sediment loads enter a fen it could promote an increase of biomass productivity, which could be an additional reason why the wetland loses species diversity (Grace, 1999). Furthermore, Bedford & Godwin (2003) explain that repeated flooding can also eliminate those plant species that are not able to adapt to these conditions, which will further reduce species diversity since many fen specific species are not found in wetlands that have long periods with standing water.

Koerselman (1989) studied Dutch fens that were highly influenced through water management using a regional matrix of ditches within which fens are imbedded. The 'polder boards' area is either supplied with water or drained in order to maintain water level suitable for the agricultural pastures used for milk production. During dry periods, water tables were maintained by pumping water into the area which caused the natural water flow directions to reverse. This change in flow pattern caused nutrient rich surface water from the ditches to move into the fen and reduced the influence of groundwater. This type of management had a devastating effect on the delicate balance between groundwater, surface water and precipitation inputs and can eventually cause a decline the species-rich mesotrophic plant communities found there such as *Carex diandra, Carex curta, Potentilla palustris, Menyanthes trifolia, Caltha palustris, Equiseturn fluviatile.* A similar example can be found in Chippenham Fen, Cambridgeshire, UK where a borehole was used to maintain ditch water levels in the fen, which changed the pattern of water flow and chemistry (M. Whiteman 2020, *pers. comm.*).

Finally, regular inundation may support less diverse communities such as the *Carex rostrata-Potentilla* palustris tall-herb fen (NVC class S27). Schutten (2019) has found that this community in Scottish fens is associated with long lasting flooding in the spring or summer months. S27 tall herb fen is generally found in transition mires or quaking bogs, but with higher pH than found in a poor fen (Wheeler et al., 2009).

v. Climate response

Anthropogenic activities are not the only influence on the hydrological dynamics of calcareous fens: all wetlands primarily depend on the quantity and quality of their water supply (Erwin, 2009), and fens may be extremely sensitive to the predicted changes in precipitation (Essl et al., 2012; Fernández-Pascual et al., 2015). Natural climate shifts can also cause a decrease or increase of water supply into these wetlands. For example, Large et al. (2007) found a clear control of moisture conditions in English rich-fen communities harbouring the rare *Schoenus nigricans-Juncus subnodulosus* mire community (NVC class M13). This was quantified by using DCA analysis and Ellenberg F-values on long-term hydrological databases and vegetation data collected periodically over a 12 year period. The change from grassland communities into mire communities revealed a recovery of fen communities after drought conditions. The found a strong correlation between moisture and vegetation community structure that can be used for the restoration of communities of conservation interest. It is also likely that predicted increases in temperature will increase the CO_2fluxes from the wetland.

2.4.1.2. Hydrological thresholds for calcareous fens

i. Water balance

Mitch & Gosselink (2007) distinguish three major factors controlling the water budget of a wetland: 1) the water mass balance, 2) the surface contour of the landscape and subsurface soil 3) and the geological and groundwater conditions. The first factor only deals with inflows and outflows while the latter factors depict the capacity of a wetland to store water.

A full water balance for wetlands showing storage for any time interval can be expressed in the following equation:

$$\frac{\Delta V}{\Delta t} = P_n + S_{in} + G_{in} - ET - S_{out} - G_{out}$$

Where:

$\frac{\Delta V}{\Delta t}$ – change in volume of water storage in wetland per unit time [T]

P – precipitation [L^3/T]

S_{in} – surface inflows [L^3/T]

G_{in} – groundwater inflows [L^3/T]

ET – evapotranspiration [L^3/T]

S_{out} – surface outflows [L^3/T]

G_{out} – groundwater outflows [L^3/T]

Quantification of the different parts of the water balance (inflows, outflows etc.) can reveal if any issues are arising that may negatively affect ecological conditions in calcareous fens. If such monitoring data is available from which to derive an overall water balance, then the change in water storage (and thus more specifically the change in water level in the fen) can be predicted over time. From this, assessment can be made as to the sensitivity of the water table and its threshold envelope and the water sources required to support the fen vegetation with respect to the various other parts of the hydrological cycle.

Another water balance was made in the UK that outlines the different mechanisms that support wetlands controlled by both groundwater and surface water (Lloyd & Tellam, 1995). As seen in Figure 2.4 the water balance equation is largely comparable to the equation outlined above. However, and important difference is seen in the storage where Lloyd & Tellam (1995) distinguishes between groundwater storage and surface water storage as different mechanisms have an influence on this as seen in Figure 2.4. It is therefore important to measure all these different inputs and outputs properly in order to calculate a water balance that is as close to the actual conditions as possible.

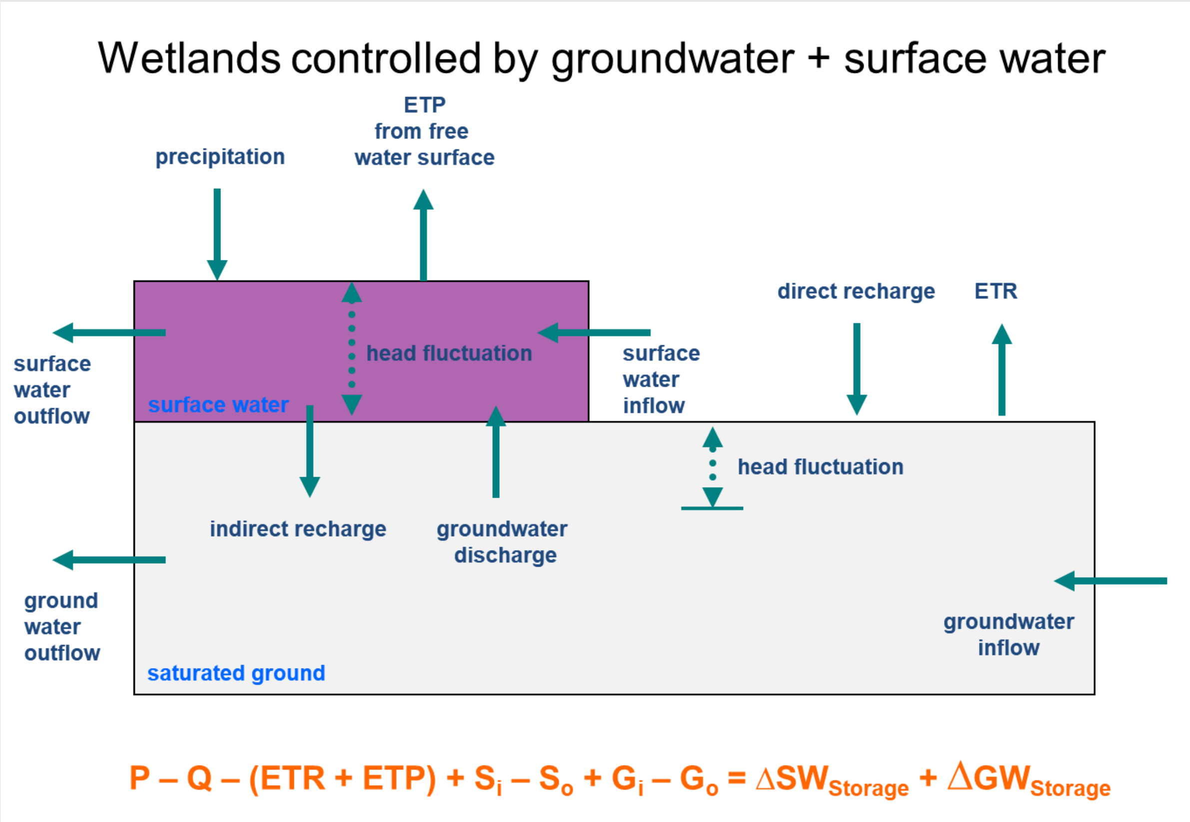


Figure 2.4. Schematic diagram of Groundwater Dependent Terrestrial Ecosystems (GWDTEs) by Lloyd and Tellam in Birmingham (1995).

ii. Internal water table

The internal water levels of calcareous fens are generally found to be higher and more stable compared to surface water fed fens and show higher water levels, associated with larger flows, in the winter and spring months. For example, two groundwater fed fens that were monitored in Scotland were inundated throughout most of the year, 75% of the time in winter and 50% of the time in summer, corresponding to water tables of approximately -0.10 m and -0.25 m respectively (Schutten, 2019).

As discussed previously, Wassen (1995) studied different variables of the different types of wetlands found in the Bierbza river valley in Poland, amongst which was surface water level. The results (see Table 2.2) reveal how mean surface water levels values between summer and winter level differ the least in the rich fens, which relate to UK based NVC class fen vegetation communities associated with EU habitat alkaline fen (7230). However, it should be noted that the data does seem to suggest that fluctuations do occur in these fen types across different times of the year. Data from alkaline fens presented by Wheeler et al. (2009) in Table 2.2 shows summer water table with more extreme fluctuations than measured by Wassen, (1995).

Table 2.2. Mean values of groundwater levels in fens (from Wassen, 1995 and Wheeler et al., 2009)

Wetland ecosystem (from Wassen, 1995)	**Mean water table (cm bgl)** *winter*	*summer*
Floodplain (n = 1)	-54.8 ±6.8	4.0 ±6.5
Rich fen (n = 1)	-3.5 ±6.2	2.2 ±3.8
Transitional fen (n = 1)	-0.6 ±1.3	6.1 ±4.2
Alkaline fen type (from Wheeler et al., 2009)	**Mean summer water table (cm bgl)**	
M9 *Carex rostrata – Calliergon cuspidatum / giganteum* mire Variant 1 (n = 52) Variant 2 (n = 37) Variant 3 (n = 6)	3 (min -25, max 36) 4.5 (min –14, max 24) –7.3 (min –26.2, max 3.2)	
M10 *Carex dioica – Pinguicula vulgaris* mire (n = 121)	–1.5 (min –16.2, max 3.4)	
M13 *(Schoenus nigricans– Juncus subnodulosus*) mire (n =117)	- 4.6 (min –38.6, max 5.0)	

iii. Internal flow rate

Hydraulic conductivity

The depth and variants in peat stratum have a significant effect on mean vertical permeability which is disproportional to its relative permeability. The depth of low permeability stratum has a bigger effect on mean permeability than the depth of high permeability strata. A proper understanding of local geology and the physical properties of fen peat can be understood by determining the hydraulic conductivity since it is a parameter that is very sensitive to the geological structure of the profile. This also includes the impact of preferential flow paths through such a medium. For example, several years of field measurements in Pollardstown fen in Ireland showed that the key process in the conceptual hydrological model for the wetland was the moisture balance in the upper layer of peat, which is controlled by the amount of an upwelling flow and the evapotranspiration from the peat surface. The complex geology of the fen margin also seemed to be a leading factor controlling seepage (Kuczyńska, 2008).

The hydraulic conductivity in floating root mat fens such as De Stobbenribben, the Netherlands (van Wirdum, 1991) was estimated to be somewhere between 500 and 1000 m/d on the basis of the hydraulic gradient and the water balance (as well as being checked through local field tests with piezometers). More extreme values have been found in De Wieden of up to 1500 m/d (van der Perk & Smit, 1975), whilst, in contrast, Koerselman (1989) found lower average hydraulic conductivities of 64.5 m/d for similar floating Dutch fens.

Stofberg et al. (2015) quantified hydraulic conductivities in a fen surrounded by intensively drained agricultural fields the Netherlands, finding variations over several orders of magnitude (10^{-3}–10^{-1} m/d). The subsoil in the fen however, consisting of thick peat and underlain by clay layers, had water infiltrating at a rate of 0.6 mm/d, as estimated by a water balance. The authors also proved that saturated hydraulic conductivity of floating fen material was negatively correlated with the degree of decomposition, using a mixed regression model.

Hence, different values of hydraulic conductivity may be found depending on the degree of decomposition of the peat. Romanov (1968) found that slightly decomposed fen had a hydraulic conductivity of 4.32 m/d. This number decreased for fen peat with moderately to highly decomposed fen peat with rates obtained such as 0.68 m/d and 0.09 m/d respectively.

Other hydraulic conductivities found in other literature are listed below:

- lesser humified peat (6.78 m/d at 0.7 m blg and 7.80 m/d at 0.9 m blg) (Baird, 1997)
- peat (1 -4 m/d) (Gilvear et al., 1993)
- peat stratum, Pollardstown fen, Ireland (0.10 m/d) (Kuczyńska, 2008)
- sandy valley fen, Nebraska (0.36 m/d mean) (Harvey et al., 2007)

Hence, it is clear that there are large variations in hydraulic conductivities between different fen types.

iv. Inflows

Groundwater contributions to the water balance of typical fens usually exceed that from direct precipitation onto the fen. Groundwater inputs as high as 90% have been determined found by Gilvear et al. (1993), whilst Koerselman (1989) attributed 55% of the total inflow to groundwater recharge and only 43% to precipitation. As an example of flow rate, Harvey et al., (2007) calculated values of vertical groundwater inflow, across one of the sandy valley fens in Nebraska to give a total estimated groundwater contribution to the fen of 5.1×10^{-1} m^3/s over the ~100 ha area of wetland. They noted that while it was evident that fen plants are growing in areas of groundwater discharge, no specific correlations could be identified between species type and either discharge magnitude or chemical concentration. The absence of correlations was attributed to the coarseness of both data sets (Harvey et al., 2007).

Figure 2.5 shows the contribution of each water source, (surface water, rainfall as well as groundwater) in the wetland framework for England and Wales (Wheeler et al., 2009). It should be noted that M92, M93, M10 and M13 (NCV classes) are all connected to the EU Alkaline fen (7230) habitat classification.

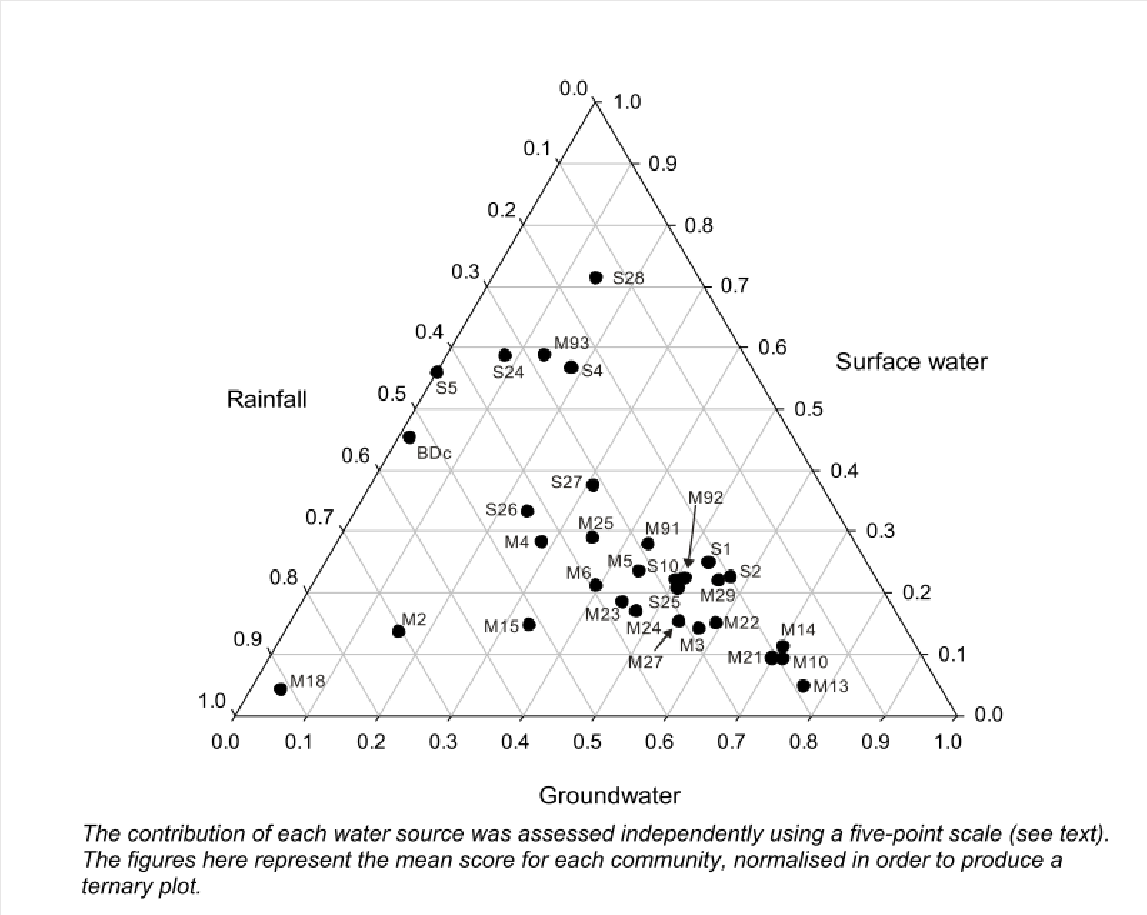


Figure 2.5. The contribution of rainfall, surface water and groundwater to different wetlands within the wetland framework for England and Wales (Wheeler et al., 2009).

v. Outflows

The losses of water via evapotranspiration from fens will clearly depend on the local climate, the rate being affected by meteorological parameter such as solar radiation, wind speed, air temperature, atmospheric temperature and relative humidity.

Kellner, (2002) provides a summary of evapotranspiration levels determined for several different wetlands in northern temperate climates, as shown on Table 2.3.

Table 2.3. *Examples of evapotranspiration for different wetland types (from Kellner, 2002).*

Reference	Wetland site, type, vegetation	Period	E mm/day	E / PE
/Price, 1994/	Lakeshore Typha marsh, Lake Ontario	June–August	4.8	0.97
/Lafleur, 1990/	Lakeshore Carex marsh, Southern Hudson Bay	May–August	2.75	0.8
/Souch et al, 1996/	Lakeshore mixed open marsh, Lake Michigan	June	3.3	1.0
/Kim and Verma, 1996/	Poor open fen, Minnesota	May–October	(0.2–4.7) 3.0	1.0
/Lafleur and Roulet, 1992/	Poor open fen, Southern Hudson Bay Lowland	July–August	2.5	≈0.7
/Kellner, 2001/	Open bog ,Central Sweden	June–August	2.1–2.4	0.7
/Moore et al, 1994/	Mixed open mire, Northern Quebec	June–August	2.8	0.8

Of more relevance to the Irish climate and to the fens under investigation, Wheeler et al. (2009) provides UK data for potential evaporation related to NVC class fen vegetation communities associated with EU habitat alkaline fen (7230) in Table 2.4. The other main outflow will be via surface water discharge routes in the form of a stream(s).

Table 2.4. Potential evaporation (mm a-1) in alkaline fens (7230) (from Wheeler et al., 2009).

Alkaline fen type	**Mean summer PE (mm a^{-1})**
M9 *Carex rostrata – Calliergon cuspidatum / giganteum* mire Variant 1 Variant 2 Variant 3	 547 (min 454, max 646) 565 (min 467, max 626) 625 (min 625, max 625)
M10 *Carex dioica – Pinguicula vulgaris* mire	539 (min 462, max 614)
M13 (*Schoenus nigricans– Juncus subnodulosus*) mire	613 (min 564, max 646)

2.4.2. Hydrochemistry

2.4.2.1. Reaction to catchment and climate characteristics

i. Base richness

In the mesocosm experiment mentioned earlier Cusell et al. (2013) showed that inundation with base rich water was beneficial for fen mosses such as *H. vernicosus* and *S. scorpioides*. The flooded peat cores were found to have an increased soil buffering capacity and internal chemical reduction processes caused a considerable increase in pH and alkalinity as well as a decrease in sulphate (Loeb et al., 2008). Cusell et al. (2013) argue that the delay in the expected increase of pH and alkalinity was probably caused by the consumption of bicarbonate through oxidation processes. Furthermore, following the continuous infiltration of bicarbonate-rich water there was an increase in chloride and calcium concentrations. The infiltration had a stronger effect during the summer, due to the high evapotranspiration in that season, as well as other factors such as higher microbiological activity as a result of the warmer temperatures.

In contrast, decreasing inputs of base rich water to the surface of fens causes rich fen mosses to be replaced by Sphagnum species, and these mosses actively acidify these wetlands (Kooijma et al., 2016). This was found in the Dutch fen 'Stobbenribben' where the wetlands were hydrologically isolated with peat ridges. Furthermore base-rich ditch water became increasingly mixed with rainwater the further the distance into the isolated parts of the fen. This resulted in a vegetation gradient from alkaline loving brown moss (*S. scorpioides*) communities near the ditch towards *Sphagnum*-dominated communities in the more isolated parts (van Wirdum, 1991).
Grootjans et al. (2016) found evidence of a previously existing ground water fed fen in a blanket bog in Connemara, Ireland. Remnants of *Schoenus nigricans*, an indicator species for calcareous habitats, indicated that the blanket bog used to be a groundwater-fed fen for a long time. Indeed, evidence for the inflow of base-rich water was found at multiple sites and depths indicated by high electrical conductivity and pH values as well as high concentrations of calcium and bicarbonate.
The reason that ions such as calcium may have such a large effect on the vegetation in fens is that it affects the bioavailability of other nutrients (McBride et al., 2010). At high concentrations, calcium reacts with soluble orthophosphates and forms insoluble calcium phosphates, thus effectively removing bio-available phosphorus.

ii. Oxidation state (redox)

As stated previously, reducing conditions induced by inundation through base-rich water may not only lead to desired alkalinisation in calcareous fens, but also to an undesired phosphorus mobilisation (Patrick & Khalid, 1974). In addition, it may also lead to high sulphide and ammonium concentrations (Lamers et al., 1998). Cusell et al. (2013), for example, found that sulphate-enriched inundated mesocosms resulted in sulphide concentrations up to 200 µM that had a toxic effect on *S. scorpioides* and more importantly resulted in a vegetation change from a dominance of *Carex* species to *Juncus* and grass species. Lamers et al. (1998) also found this vegetation shift but at significantly lower concentrations with sulphide levels around 20 uM.
Groundwater flow can be an important control on oxidation–reduction potentials. Studies have shown that stagnant wetlands often have lower redox potentials than wetlands with moving water (Armstrong & Boatman, 1967; Shaw & Wheeler, 1991; Wheeler et al., 2009). Higher redox potentials may cause a lower availability of phytotoxins due to the redox-related solubilities of elements such as Fe^{2+}, Mn^{2+}, S^{-} and some species such as *Molinia caerulea* can grow better under these conditions (Armstrong & Boatman, 1967). In contrast, flowing groundwater can sometimes have lower redox potentials compared to more stagnant areas of wetlands. This may happen in

areas where groundwater seepage is strongly reducing which is sometimes seen in the oxidation of ferric iron (Fe^{2+}) upon outflow (Wheeler et al., 2009).

Changes in soil chemistry are caused by draining and subsequent rewetting. Aggenbach et al. (2013) noticed a greater pool of iron at restored as opposed to reference sites, attributing it partly as a result of a concentration effect due to organic matter loss during the drainage period. However, this cannot explain the parallel loss of Ca, Mg and S. Hence, it was hypothesized that this is caused by recurring oxidation and reduction of iron (and sulphides) during the draining of the fen, together with a continuous supply via the groundwater of ferrous iron. When the water table is low during summer the topsoil is aerated and ferrous iron and sulphides are oxidized.

iii. Phytotoxic metals

Ions such as aluminium and iron concentrations also play important roles in fens due to their toxicity under certain circumstances. These ions when exposed to acidic conditions may become bioavailable in concentrations that limit productivity or are directly toxic to plants (McBride et al., 2010). This phenomenon is seen more in base-poor fens. However, the vegetation in base-rich fens may also become threatened when groundwater inputs are limited due to changes in hydrology and land use in and around the fen.

Under circumstances where the fen is inundated for a longer period of time, iron might become reduced which can results in increased phosphorus mobilization and availability which can promote eutrophication (Cusell et al., 2013). However, iron has also been seen to bind sulphide and prevent its toxic effects under inundated conditions where sulphate has become reduced (Smolders & Roelofs, 2009).

iv. Fertility

Nitrogen (N), phosphorus (P) and potassium (K), which can all be referred to as ‘plant macronutrients’, all act as the most important enrichment media since they are the major plant nutrients that typically limit plant growth in a fen. The nutrient regime of a fen is affected by its interactions with the surrounding landscape by factors such as geology, geomorphology, catchment hydrology and land use which establish water nutrient content and its rate and direction of flow into the fen (McBride et al., 2010). This balance may be very delicate; shifts in the origin of water supply, for example, affect the biogeochemical processes. Equally, at an internal level accumulation of phosphorus and nitrogen are highly dependent on the oxygen availability in peat soil. Therefore making release and holding rates very dependent on water level fluctuations, soil permeability, temperature and the most important factor; the quality of water travelling through the peat surface (Koerselman et al, 1990).

Phosphorus

In fens, phosphorus adsorption process occurs when dissolved phosphorus interacts with and develops a strong bond to sediments (McBride et al., 2010). However, certain situations allow this chemical binding to become weaker. When redox potentials fall to a low level the bound phosphorus may be released and become bioavailable to plants. This can lead to a flush of the nutrient through internal nutrient cycling into the fen or its adjacent habitats and may affect typical fen vegetation (Shaw & Wheeler, 1991). Furthermore, some plants can excrete the enzyme phosphatase at the root surface to release phosphate from organic stores (McBride et al., 2010). Lucassen et al., (2004) found that phosphorus can be released from peat soil into groundwater by mobilisation and reduction of sediment causing a reduction of ferric oxyhydroxides due to water level changes. Some fens however can also have high concentrations of iron that bind phosphorus (Bedford & Godwin, 2003).

Phosphorus mobility is also strongly pH dependent. Wetlands that have high internal concentrations of calcium, iron, or aluminium have a high potential to retain phosphorus but these processes are directly influenced by their pH (Bedford & Godwin, 2003). Aluminium and iron play a more important role at low pH when they adsorb plant available orthophosphate. However, when acidic conditions prevail and the pH rises above 6.5 the process reverses and phosphorus, aluminium and iron are released back into the system (McBride et al., 2010). In contrast, when pH values above 7 are found, high calcium levels may cause the formation of insoluble calcium phosphates which results in immobilized phosphorus ions that are unavailable for uptake by fen vegetation (Bedford & Godwin, 2003; McBride et al., 2010). Furthermore, several humic metals may also be responsible for the formation or release of phosphorus complexes. Iron in the form of ferric iron Fe^{3+} for example may release phosphorus to the free water table by more oxygen rich water with a higher redox potential. Work done by the Environment Agency and Natural Resources Wales on Cors Bodeilio (Schlumberger Water Services 2009) suggested that even if phosphate concentrations are low, excess nitrates may still cause damage to wetland vegetation (see following section).

A more complicated relationship between nitrate, sulphate and phosphate concentrations was found by Lucassen et al., (2004) induced by sulphate ions. Reactions between SO_4^{2-} with $Fe\text{-}PO_4^{3-}$ complexes caused mobilisation of PO_4^{3-} and SO_4^{2-} and reductions in reactions with $Fe\text{-}PO_4^{3-}$ during which FeS_x is produced and PO_4^{3-} is freed. This reaction suggests that high phosphate concentrations may be found in scenarios where high sulphate concentrations are reduced to sulphides.

Nitrogen

Plant growth in many ecosystems may be supported or limited by the nutrient nitrogen (N). In peatlands, the largest proportion of the soil's nitrogen occurs as organic N. However, this form can be converted to ammonia and nitrate by micro-organisms via a process known as mineralisation (McBride et al., 2010).

Fens have a high potential for denitrification because of their anaerobic peat soil and while generally denitrification rates have found to be low in fens, such low rates are attributed to a lack of nitrate supply in undisturbed peatlands (Bedford & Godwin, 2003). Peat contains a large amount of soluble organic carbon, which is the food source for the heterotrophic bacteria that carry out denitrification, and so the process is unlikely to be energy limited in such wetland environments. For example, in Pollardstown fen in Ireland, Kuczyńska (2008) found very low concentrations of nitrates in the peat in contrast with higher concentrations found in gravel layers underlying the peat layers, which suggested significant levels of denitrification for the water moving up through the peat, as there were relatively high flows in discreet pathways through the peat.

In contrast, when fens are subjected to neutral pH under aerobic conditions, nitrification can occur, causing ammonia (NH_3) to convert to nitrate (NO_3^-). Both nutrient forms can move freely in solution and are available for uptake by micro-organisms and plants (McBride et al., 2010). Furthermore, dry and wet atmospheric deposition of nitrate and ammonia add nitrogen to the surface of fens which may then be taken up by wetland plants. GWDTEs in England and Wales were reported to receiving nitrogen deposition above the critical loads in 64 % of the cases (Farr and Hall, 2014). Farr et al. (2019) used the results of isotope analysis and age dating techniques in addition to standard geochemical evidence to calibrate a model (Farmscoper) that calculates the amount of nitrate leaching in a site. The model may also be used to add mitigation measures such as changing to a land use with lower terrestrial nitrate in order to meet proposed 'threshold' values for nitrate. Changing the land use, however, did not result in meeting the proposed groundwater 'threshold' values for nitrate even if terrestrial nitrate were reduced. It is possible that dry deposition is responsible for this as it was not reported that emission from its biggest sources (coal-burning power plants, factories, and automobiles) was reduced.

Nutrient limitation and nutrient loss

Utilisation of some nutrients may be limited by the availability of others (McBride et al., 2010). These limiting factors are often the key to maintaining species-rich fen vegetation, since they prohibit the growth of the more nutrient-responsive and often aggressive plants.

The limitation most commonly seen in peatlands is where phosphorus is the limiting factor (van Diggelen et al., 2006; Koerselman, et al., 2010). The low phosphate restricts a relative surplus of nitrogen, although again the study on Cors Bodeilio suggested that even where phosphate concentrations are low, excess nitrates may still cause damage to the wetland vegetation. An increase in readily bioavailable phosphate without a change to the concentration of nitrogen could quickly result in enrichment and eutrophication if these concentrations persist. This may result in an increased growth of some plants and the loss of others less able to respond to the new source of nutrient (McBride et al., 2010). Therefore, fens can be extremely vulnerable to seemingly small increased in limiting nutrient concentrations.

Loss of nutrients through drainage may also affect the peat fertility. De Mars et al (1996) found that drainage causes aeration of topsoil as well as accelerated decomposition and an increased nutrient release which resulted in phosphorus and potassium limitations.

2.4.2.2. Hydrochemical thresholds for calcareous fens

Guidelines for threshold values of substances in Irish groundwater were established in 2010 (Government of Ireland, 2010) and are shown in Table 2.5. The values were generally based on drinking water standards, one of the main uses of groundwater in Ireland. Values for ammonium and molybdate reactive phosphorus were established from surface water Environmental Quality Standards (EQS). Since threshold values tailored to calcareous fens in Ireland have not been established, the following sections attempt to define general threshold levels found in calcareous fens with high ecological condition. It should also be noted that Kimberley and Coxon (2013, 2015) have looked at different options on setting threshold values, finding for example, that the nitrate range of in fens in good condition was 0.56 to 2.26 mg-N/l NO_3 – i.e. much lower than the groundwater threshold value in Table 2.5 based on drinking water standards. It should be stated that this range is much lower to the reported nitrate threshold values of 15 mg-NO_3/l in rich fens with high ecological value (UKTAG 2012), Further work by Kimberley (2013) has proposed a threshold value of 2.26 mg-N/l for Irish calcareous fens that seems to be being used informally by the Irish EPA now. It should be noted that in England there is now a move towards restoring natural hydrological functioning rather than focussing on achieving specific threshold values.

Table 2.5. Groundwater threshold values of substances in Ireland (Government of Ireland, 2010) and the UK (EC, 2006)

Element	***Threshold Values (mg/l)***	
	Ireland	***United Kingdom***
Molybdate Reactive Phosphorus (as P):	0.035	0.054
Ammonium (as N):	0.175	0.29

Nitrite (as N)	0.114	
Nitrate (as N)	8.47	8 – 42
Chloride	187.5	188
Sulphate	187.5	188

i. Base richness

Measuring pH is used as a method for assessing acidity or alkalinity in a wetland. Waters with a pH greater than 7.0 are considered to be alkaline, and less than 7.0 acidic. Base-poor fens receive their main input from rainwater, which is slightly acidic by absorption of a small amount of carbon dioxide (CO^2) from the atmosphere, resulting in weak carbonic acid (H_2CO_3), whereas fens that receive their bulk from ground water tend to be more alkaline. The groundwater will be more alkaline when it has come into contact with calcareous rocks such as limestone from which it has picked up base ions such as Ca^{2+} and Mg^{2+} and their associated bicarbonate ions (HCO^{-3}).

According to Wheeler and Proctor (2000) the subdivision of wetlands by the base richness gradient is a pH lower than 5 and higher than 6 for bogs and fens respectively. Furthermore, fens should have high concentrations of calcium and high bicarbonate acids as opposed to bogs. A more complex division was created by Johnson (2000) who split fens into poor, moderate, extreme and extremely rich classes. Poor fens are recognised by a pH of 4.0-5.5 and calcium concentration of 2-7 mg/l. The pH of moderate fens is 5.5-7.1 and has a calcium concentration ranging between 10-50 mg/l and rich fens have a pH of 6.0-7.5 and calcium between 25-80 mg/l. And last an extremely rich fen pH is 6.5-8.5 and calcium greater than 30 mg/l.

Electrical conductivity (EC) can also be used to assess fen water and soil chemistry. The EC of drinking water would be expected to lie around is 0.005 S/m, whereas seawater reaches about 5.0 S/m. Conductivity provides a useful guide to base enrichment but not macronutrient concentrations (McBride et al., 2010). The background levels of EC in pure limestone aquifers in Ireland (that usually feed calcareous fens) ranges between 179 to 601 (median 417) µS/m (Tedd et al., 2017).

ii. Redox

Redox potential can give an estimate as to whether groundwater or soils are aerobic or anaerobic. This information is important since certain processes that can degrade peat soil (such as denitrification, phosphorus release and toxification by Fe^{2+}, Mn^{2+} and S^{-}) occur under anaerobic conditions. Aerobic soils have redox potentials of about 0.6 V and anaerobic soils have redox potentials between 0.4 and -0.2 V (McBride et al., 2010). Redox potential measurements are made using redox electrodes (usually made of platinum) and are measured in volts (V), millivolts (mV) or Eh (where 1 Eh = 1mV).

iii. Phytotoxic compounds

Observed apparent threshold values for sulphide concentration in mesocosm experiments were reported by (Geurts et al., 2009) to be between 1.6 to 22 mg/L when measuring sulphide toxicity on vegetation development in fens in a mesocosm experiment. They also found that the toxic effect of sulphide was more acute under nutrient-poor conditions, where almost all species disappeared completely.

As for total iron (Fe), the threshold limit was reported to be between 10–25 mg/L (Snowden & Wheeler, 1993). Anything above this limit is likely to be toxic for many fen species. Aggenbach et al., (2013) found that rich fen bryophytes, e.g. *Hamatocaulis vernicosus* only thrived under Fe concentrations at sites where the content was in the range of 0.73–8 mg/l Fe. The species *Calliergonella cuspidate* seemed to withstand iron-rich conditions, but disappears at levels higher than 170 mg/L. However, vascular plants like *Equisetum fluviatile* and *Carex rostrate* seemed to thrive in iron rich sites.

iv. Fertility

The UK Technical Advisory Group (UKTAG) on the Water Framework Directive has published threshold values for fens of good and poor quality divided by oligotrophic and mesotrophic fens which respond to base poor and base rich fen respectively. These values were derived from an extensive literature study and are shown in Table 2.6. Note, a *good* or *poor* designation means there is a 75% or greater likelihood that site condition will be good or poor if this value is not reached or exceeded respectively (see UKTAG, 2012).

Table 2.6. Threshold values for UK fens (UKTAG, 2012)

	Good: oligotrophic	**Good: mesotrophic**	**Poor: oligotrophic**	**Poor: mesotrophic**
Nitrate (mg-N/l)	2.9	3.4	7.2	7.1
Phosphate (mg-P/l)	0.021	0.033	0.064	0.034

Nutrient loads

Diffuse enrichment is caused by both rural and urban land-based activities within the fen catchment. DEFRA (2008) reported that 60% of nitrates, 25% of phosphorus and 70% of sediments entering UK water bodies are contributed by agriculture. In Ireland, a Source Load Apportionment

Model developed by the EPA (Mockler et al., 2017) has shown that the main sources of phosphorus in surface water bodies are from municipal wastewater treatment plants and agriculture, with wide variations across the country related to local anthropogenic pressures and the hydrogeological setting, whilst the main source of nitrogen is from agriculture.

Phosphorus

Median phosphate concentrations in Scottish fens were found to be 0.10 mg/l PO_4-P in groundwater which is very high considering the threshold values in the UK were reported at 0.053 mg/l PO_4-P. Groundwater results were however skewed by the analytical level of detection of 0.20 mg/l used in laboratory test for some of the samples. Approximately 40% of concentrations were at or below this level of detection (Schutten, 2019).

Duval (2010) found phosphate to be very low in Canadian riparian fens (mean ±1 SD= 0.001 ± 0.0004 mg/l PO4-P). Between 15 and 40 % of the total dissolved phosphate was unavailable to vegetation due to high calcium and magnesium carbonate levels which he explained was due to the saturation index of calcite. This suggests that calcareous fen plant species not only tolerate low P levels, but also are particularly well adapted to compete for this scarce resource (Duval et al., 2012).

Nitrogen

According to UKTAG (2012), nitrate values in rich (mesotrophic) fen with high ecological value have a threshold of 3.4 mg-N/l (i.e. there is a 75% or greater likelihood that site condition will be in a good condition if this value is not exceeded). Concentrations found in Scotland were significantly lower with an average of 0.25 mg-N/l, whereas total nitrogen had a median concentration of 3.0 mg-N/l measured across 20 sites (Schutten,2019).

Ammonium concentrations are expected to be high in peatlands, due to the anaerobic nature of peat. However, concentrations vary between sites. Kuczyńska (2008) found values ranging from as low as 0.08 up to 0.34 NH_4 -N mg/l in Pollardstown Fen with higher concentrations on the northern site of the fen. Here, generally high nitrogen loads were found and this was likely to reflect organic pollution rather than purely chemical characteristics of peat. Duval (2010) found higher averages in Canadian riparian fens and also bigger ranges with ammonium concentrations of 0.52 ±0.97 mg-N/l.

Bobbink et al. (2002) have described a critical load threshold for both base poor and rich fens through atmospheric deposition which is expressed as kilograms of nitrogen per hectare per year (kg-N/ha). This critical load is used to identify the threshold deposition rate of nitrogen to fens, which upon exceeding will result adverse effect to fen vegetation (see Table 2.7).

Table 2.7. Critical loads for base poor and rich fens (Achermann & Bobbink, 2003).

Ecosystem Type	Critical Load kg N ha^{-1} yr^{-1}	Signs that critical load has been exceeded i.e. observable impact in the fen
Base-poor fens	5 – 10	Increased sedges and other vascular plants. Negative effects on peat mosses
Base-rich fens	15 – 35	Increased tall graminoids (grasses, sedges) Decreased species diversity

Critical load values of poor and rich fens used in atmospheric deposition modelling by APIS (2013) in the UK are shown in Table 2.8. Valley mires, poor fens and transitions mires stand out most when compared to the values described by Bobbink et al. (2002) which show a lower range of 5-10 (kg-N/ha), whereas APIS shows a range of 10-15 (kg-N/ha). The ranges for rich fens are higher, however, and more comparable between Tables 2.7 and 2.8.

Table 2.8. Indicative values within nutrient nitrogen critical load ranges for use in air pollution impact assessments for EUNIS habitat types

Mire, bog and fen habitats	***Critical load (CL) range (kgN/ha/yr)***	***Recommended value to use at screening stage of assessment (kgN/ha/yr)***	***Recommended value to use at detailed assessment stage (kgN/ha/yr)***
Valley mires, poor fens and transition mires (D2)	10-15	10	10
Rich fens (D4.1)	15-30	15	15

v. Relationship of vegetation to environmental variables

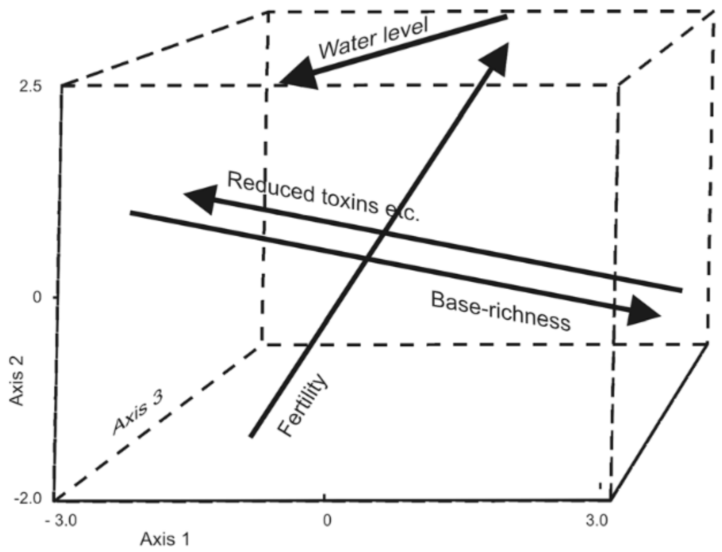
Duval (2010) found that in general it proved less useful to explain the reaction of vegetation species when solely looking at the effects of nutrient gradients itself. The study found a great variability in plant species distribution and hydrological and biogeochemical gradients. This variability shows that heterogeneous environmental variables have numerous different influences on the response and range of numerous vascular plant species. Nevertheless, the author was able to make some general predictions. For example, most grasses weren't affected to nitrogen levels above 0.5 mg/l, while sedges showed a decreased cover response to increased nitrogen presence.

In the presence of increased phosphate concentrations to level above > 25 mg/l P, many species cover, as well as *Carex flava, Carex lacustris*, and *Carex lasiocarpa* responded with a drastically decreasing cover. However, in contrast both *Calamagrostis canadensis* and Typha spp. increased cover with increasing P-availability. Species such as *Menyanthes trifoliata* and *Rubus pubescens* all displayed local maximum cover around 20-30 mg/L P. In contrast, fen vegetation responded to only a single peak (unimodal) in the lower range of p values. Like stated before this suggests that calcareous fen plant species not only tolerate low P levels, but also are particularly well adapted to compete for this scarce resource (Duval et al., 2012).

For potassium many species showed no or a very gentle response to increasing concentrations. Duval (2010) concluded that of all the environmental variables tested, the duration of initial saturation to start the growing season as well as the peat organic matter content proved most useful in determining individual species ranges or tolerances.

Wheeler and Shaw (1995) developed a conceptual model that tries to take into account all numerous environmental variables (see Figure 2.6). The level of water relative to the ground surface can have striking effects upon the composition of wetland vegetation. Furthermore, the response of plants to water regimes can be strongly influenced by other environmental conditions. The water level ranges occupied by plant species can be modified inter alia by oxidation–reduction potentials, water flow, concentrations of reduced toxins (especially Fe^{2+}, Mn^{2+} and S^{-}), availability of nutrients (NPK), competition with other plant species, and facilitative oxygenation of the rooting zone by companion species.

Linear regression relationships between species-richness terms and selected environmental variables are shown in Table 2.9. These show that, on average, the number of plant species in wetland vegetation decreases with a decrease in pH and with an increase in soil fertility or in the concentration of potentially phytotoxic metals (Al and Fe). Interestingly, the number of wetland species and rare wetland species per unit area shows no significant trend in relation to variation in summer water level and oxidation–reduction potential (Eh), but there is significant tendency for the total number of species to be greater in the drier (lower water level, higher Eh) samples.



This is a conceptual diagram based on a Canonical Correspondence Analysis (see Appendix 2) of floristic and environmental data collected from fens throughout Britain. Axes 1, 2 and 3 correspond to the first, second and third most important gradients of variation in the species composition of fen vegetation. The direction of the arrows indicates the degree to which they correspond to the floristic gradients. The principal axis of vegetation variation corresponds quite closely to variation in base richness of the water (pH, alkalinity) and (inversely) to variation in the availability of phytotoxins such as Al^{+++}, Fe^{++} and Mn^{++}. The gradient of fertility (estimated phytometrically) is almost orthogonal to the base-rich gradient and corresponds well with the second main component of variation in the vegetation. The third axis of vegetation variation corresponds well to variation in summer water level, which is again almost orthogonal to both the base-richness and fertility gradients.

Figure 2.6. Conceptual model of the controls environmental data on vegetation collected from British fens (Wheeler and Shaw, 1995).

Table 2.9. Linear regression relationships between three species-richness terms (y) and selected environmental variables (x) from samples of wetland vegetation (Wheeler et al., 2009).

The species-richness terms refer to the total number of each category of plant species per unit area (4 m^2).

y: x:	All species	Wetland species	Rare wetland species
pH	$y = 2.9x + 4.4$ $p < 0.0001$	$y = 1.7x + 6.9$ $p < 0.0001$	$y = 0.6x – 1.7$ $p < 0.0001$
Fertility	$y = –0.2x + 23.1$ $p < 0.0001$	$y = –0.2x + 18.7$ $p < 0.0001$	$y = –0.03x + 2.0$ $p < 0.001$
Water level	$y = –0.06x + 20.9$ $p < 0.001$	not significant ($p = 0.68$)	not significant ($p = 0.22$)
Eh	$y = 0.009x + 18.9$ $p < 0.01$	not significant ($p = 0.19$)	not significant ($p = 0.47$)
Fe	$y = –0.004x + 21.7$ $p < 0.0001$	$y = –0.003x + 17.0$ $p < 0.0001$	$y = –0.001x + 1.7$ $p < 0.0001$
Al	$y = –0.019x + 21.6$ $p < 0.0001$	$y = –0.013x + 16.9$ $p < 0.0001$	$y = –0.004x + 1.6$ $p < 0.0001$

2.5. Fens in Ireland

2.5.1. Current state of knowledge

A national study into the nutrient threshold values (TV) relevant to calcareous fens was conducted by Kimberley (2013). The study at large provides a range of options for incorporating nitrate threshold values relevant to Irish calcareous fens into the GWB classification process. A few options for nitrate threshold values were given ranging from 7 to 11.1 mg/L as NO3.

However, currently available data were insufficient to develop a phosphate threshold values for Irish calcareous fens and the present focus on developing a nitrate threshold values should not imply that Irish calcareous fens are at a greater risk from N than P. Research is needed to determine the nature of nutrient limitation within alkaline fens (7230) and species-rich Cladium fens (7210).

Other national studies exist in the form of habitat and conservations status reports for the NPWS (Foss, 2007) (Foss & Crushell, 2008). The total number recognised fen sites in Ireland is 808 with a total extent of 10,307 ha of Habitats Directive Annex 1 fen types. If all fen types are included (also fen types without a Habitats Directive Annex 1 designation) this area increases to a total 22.180 (Foss, 2007). Foss & Crushell (2008) attributes different key species, species richness as well as pH ranges to the main fen types in the Habitats Directive Annex 1 (7140, 7230 and 7210).

2.5.2. Implications for the future

Calcareous fens, as wetlands, are predominantly fed by carbonate rich groundwater. As such they are host to a wide variety of characteristic vegetation. The physiognomy and ecology of a fen are thus strongly correlated with the prevailing hydrological regime and its related water quality. However, a quantitative understanding of that relationship has only generated a relatively sparse literature to date, and particularly in Ireland. Such literature as there is tends to be site specific.

While calcareous fen is a listed habitat in the EU Habitats Directive, what actually constitutes such a fen is not well defined and subject to interpretation among member states. The literature indicates that the wide diversity in underlying hydrogeology and climate conditions across Europe give rise to an equally diverse range in vegetation and ecology. Nevertheless, attempts have been made to identify common threads in order to define a type 'calcareous fen' against which the condition of any given fen can be compared.

What constitutes a calcareous fen has fallen into two approaches, one based partly on hydrology and the other based only on the ecological description. The hydrological approach is more of a classification method, relying on the different hydromorphologies driving the calcareous water supply to a fen, represented by the WETMEX typology (Wheeler et al, 2009). Developed in the UK, its range of forms of fen is influenced by the frequent occurrence of fens on chalk bedrock. However, it does address the hydrological requirements for those fens, in terms of water level. While a useful approach to classifying the hydrology of fens, it does not fully encompass the range in underlying hydrogeological conditions in Ireland, particularly with reference to karst. Although addressing the nature of the water supply mechanisms for a fen, WETMEX, however, does not directly address the overall ecological condition of a fen. Other studies (e.g. Wassen, 1995) have taken a site-specific approach to defining the controlling hydrological conditions.

The second approach to defining a calcareous fen is based on its ecological physiognomy – that is, its characteristic vegetation. There is a divergence in this approach between the habitat as described in the EU Directive and that used by the UK in its guidance documents under the EU WFD. This divergence is described well in Kimberley and Coxon (2015) and the national fen studies for the NPWS (Foss, 2007) but the Habitats Directive merely mentions the presence of Cladium mariscus as the key species for calcareous fen. The UK approach encompasses a wider range of vegetation species under the NVC (National Vegetation Classification). However, both approaches are qualitative making a quantitative approach to defining a calcareous fen in 'good condition' difficult. Such a determination ultimately relies on 'professional judgement' (Kimberley and Coxon, 2015) (Foss, 2007) (Foss & Crushell, 2008).

Thus, it follows that the environmental supporting conditions (hydrological regime and water quality) for a fen in 'good condition' may be difficult to determine and a somewhat subjective process, as the literature indicates.

In the light of the sparsity of data, one alternative approach to defining criteria for determining conservation status or potential impact on a fen (for designated areas) has been to identify 'keystone species' which are deemed to be the most sensitive to that potential impact and then to determine the environmental conditions required to sustain that species. That species may not be typical of a calcareous fen in general, but it serves to provide criteria for mitigating potential impact on a particular wetland. Determining the environmental conditions for sustaining the whorl snail (Vertigo geyeri) in Pollardstown Fen in Ireland is an example of such an approach (Kuczyńska, 2008).

2.6. Conclusions

The need remains to develop appropriate criteria for environmental supporting conditions for calcareous fens which occur in characteristic hydrogeological regimes in Ireland. Kimberley and Coxon (2015) have highlighted the shortcomings in Irish data. The need for appropriate Threshold Values for water quality in fens is clear. In common with other jurisdictions, however, the assumption is usually made that steady state conditions prevail, so that means or medians of hydrological/chemical data are used as measures of condition. In the light of climate change and of the relatively dynamic nature of Irish hydrogeology, attention needs to be paid to characterizing both the temporal and spatial variation in the data – and what effects, if any, they have on the ecology of the fen. In short, frequency- duration of a particular hydrological condition can have an ecological impact as shown in the recent studies of turlough fens. Moreover, given the current interest in rewetting degraded fen peatlands with a view to increasing carbon sequestration, the need for understanding the necessary conditions for sustaining a fen in good condition are ever more acute (Emsens et al., 2020).

The literature, underpinned by the sparseness of quantitative data required to assist in determining the functionality of fens, reveals the fundamental gap in their hydro-ecological understanding. Given the lack of quantitative measures to determine the ecological condition of a fen in Ireland, the review of the literature suggests that a first approach should be to make detailed measurements of the hydrology and water quality of selected fens in order to determine what correlation exists between the observed vegetation and the underlying hydrochemical and hydrological conditions. The outcome will assist in determining how to establish appropriate Threshold Values for water quality and for water quantity/level for those fens deemed to be in good, sustainable condition.

3. Study Sites

Before the methodology was finalised research sites were chosen by screening them according to criteria outlined in Section 3.1. The selected sites then undergo a desk study to further define their characteristics in terms of geology, hydrogeology, ecology, land use as well as past changes made to the landscape. Every study site section is closed with a display of the conceptual models that were made using data acquired from desk study in order to understand the basic functioning of the site. Furthermore, the conceptual models were used to make instrumental decisions based on the threats and pressures that were thought to occur.

3.1. Site selection process

3.1.1. Site criteria

Criteria for the overall site selection process included identifying four sites that span a wide range of different fen conditions ranging from what is consisted to be a relatively pristine site through to sites impacted by water quality and water quantity issues (Figure 3.1). At least one site was required that fell under the quantity pressure category. This site could be either damaged or under pressure by drainage or abstraction. Similarly, a site that fell under the quality pressure was required, preferably one that is under pressure from nutrient pollution and thus threatened to become eutrophied. Finally, at least one site had to be selected that is known to be still in a relative pristine (intact) state.

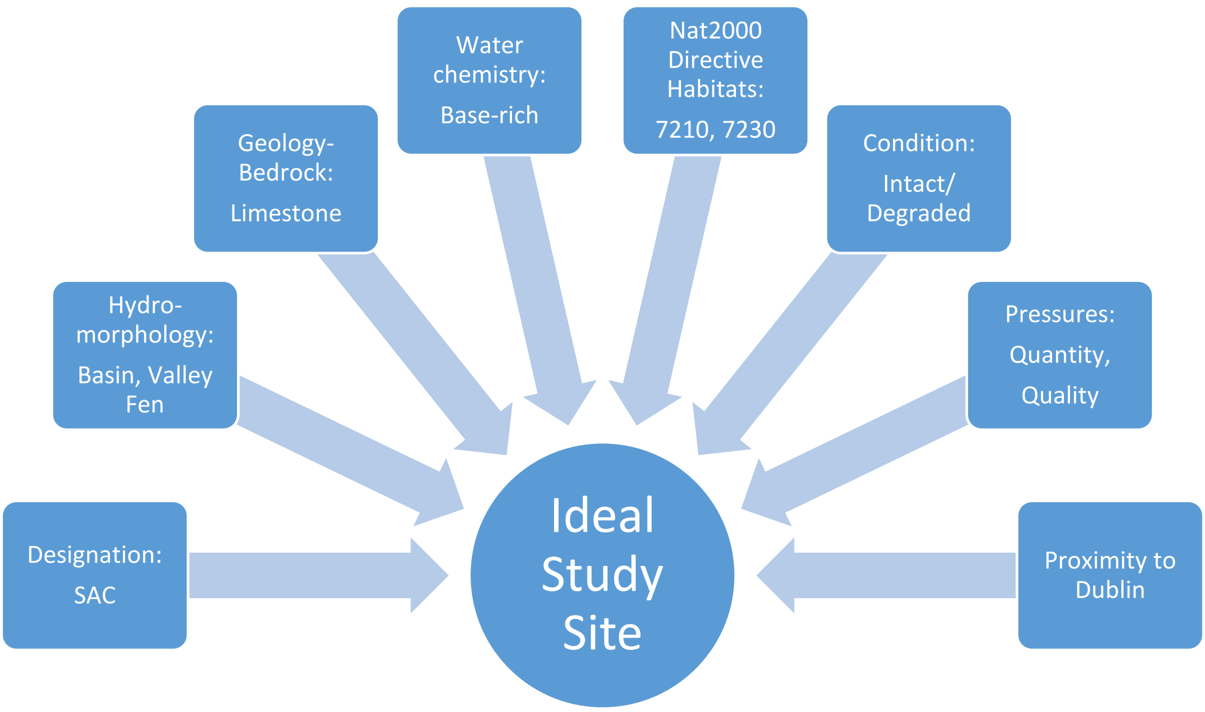


Figure 3.1. Criteria for the fen site selection process

Other numerous criteria were identified that needed to be met by each of these research sites. This was done in order to show clear boundaries between the fens in this research since this type of wetland in Ireland can have many different characteristics as discussed in Chapter 2. These criteria are displayed in Figure 3.1. This figure shows that not only the fen condition was important in the site selection process, but that sites were also prioritised if they were SAC designated and had Habitats Directive Annex 1 priority fen habitats: 7210 - Calcareous fens with Cladium mariscus and species of the Caricion davallianae and 7230 - Alkaline fens. In the habitat maps and data analysis of this research the designation according to Fossitt is used; a habitat scheme adopted by the ecologists that surveyed the fen sites. The Annex 1 habitats equivalent to the Fossitt scheme is presented in Table 3.1.

Table 3.1. Habitats Directive Annex 1 habitats and their Fossitt habitat equivalent

NFS Fen Classification Scheme	**EU Habitats Directive Habitat**	**Fossitt Habitat Scheme**
Alkaline fen	7230 Alkaline fens	PF1 Rich fen and flush PF2 Poor fen and flush
Calcareous fens with *Cladium mariscus*	7210 *Calcareous fens with Cladium mariscus and species of the Caricion davallianae	PF1 Rich fen and flush PF2 Poor fen and flush
Transition Mire	7140 Transition mires and quaking bogs	PF3 Transition mire and quaking bog

Furthermore, calcareous fen sites were required which meant that underlying bedrock aquifer needed to be limestone with resultant base-rich water chemistry. Because of the heterogeneous nature of fens in the Irish landscape a ranking was made taking into account these criteria. The ranking (Table 3.2) indicates the importance of study site satisfying each criterion as well as any criteria that could be compromised if necessary. These ranking produced a scoring list which was could be completed from desk study research. Each criterion was subdivided into a series of sub categories, each with a given score between 2 and -3.

Then, information on fens in Ireland was gathered from the IPCC's peatland sites database (IPCC, 2009), the SAC database from the NWPS (NWPS, 2016) and wetland survey reports for Kildare, Louth, Monaghan and Wicklow prepared by Peter Foss (Foss, 2007). The sites receiving the highest score were identified to be most suitable for the field studies.

Table 3.2. Criteria scoring for the selection of suitable fen research sites.

Catchment Criteria	Criteria Categories	Score
Designation	SAC	2
	cNHA	0.5
	none	-1
Annex 1 Habitat	7210, 7230	2
	7220	1
	Any other	-3
Geology	Carboniferous Limestone	1
	Marine shelf facies, Courceyan limestone	0
	Any other	-3
Geomorphology	Valley, Basin	1
	Open water	0
	Floodplain, Flush, Spring	-1
Water chemistry	Base-rich	1
	Base-Poor	-1
Damage, Treats and Pressures	None	2
	Quantity: drainage, abstraction	2
	Quality: nutrients, pollution	2
	Both quantity and quality	0
< 3 hour drive from Dublin	Yes	1
	No	-1

3.1.2. Selected study sites

From the list with the highest scoring fen sites, and following subsequent site visits, four sites were chosen. These fen sites where chosen with a spread in different fen conditions in mind: 1) Ballymore – intact, 2) Tory Hill – Quantity pressures, 3) Scragh Bog – Quality pressures, 4) Pollardstown Fen – Quality/Quantity pressures (Table 3.3). The scores of these sites according to criteria in Table 3.2 can also be found in Table 3.3. The location of these fens are shown in Figure 3.2. It should be noted that two standalone research sites in Pollardstown where chosen to be studied: Site A. an area which is relatively still intact and supports three rare sub-aquatic invertebrate species: *Vertigo geyeri*, *V. angustior* and *V. moulinsiana* (NWPS, 2014) and Site D. an area in the north which is under pressure in both quality and quantity aspects.

Table 3.3. Site specifics as reported in Natura 2000 – standard data form

Name	Pollardstown	Tory Hill	Scragh Bog (fen)	Ballymore
County	Kildare	Limerick	Westmeath	Westmeath
Area (ha)	266.1	76.9	23.9	43.1
Designation	SAC NHA	SAC, pNHA	SAC	SAC
Condition	Degraded	Degraded	Near intact	Intact
Damage, Threats and Pressures	Drainage Grazing Dumping Gravel quarry	Drainage Infilling Grazing	Fertilisation Roads Diffuse Pollution	Diffuse Pollution
Criteria scoring	8	10	10	10

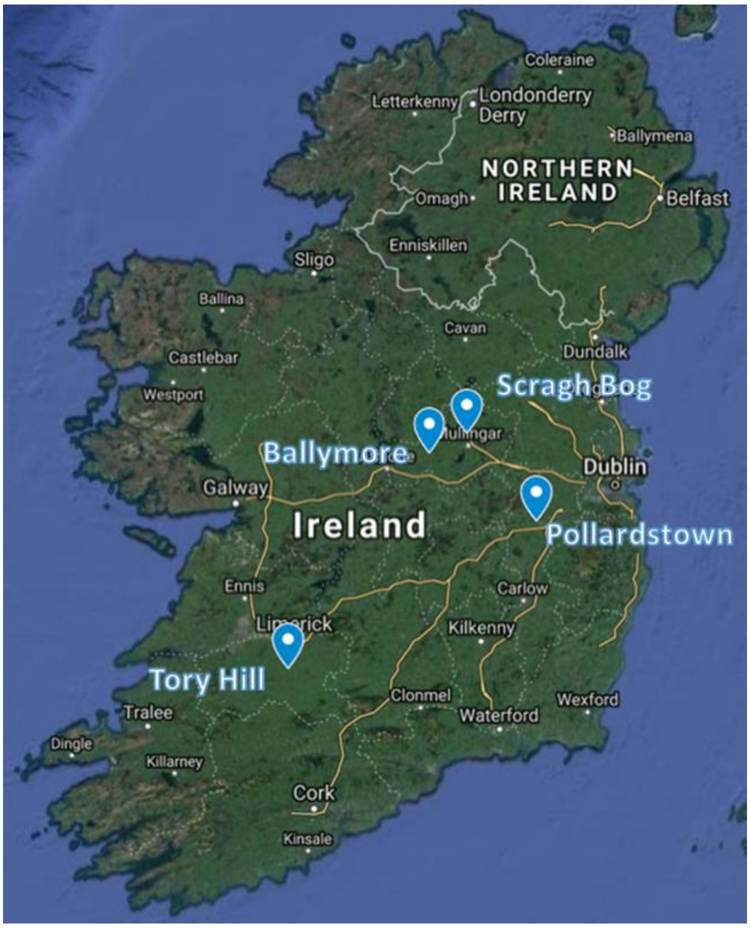


Figure 3.2. Fen research sites

3.1.3. Conceptual models

Before any instrumentation was installed, information on geology, hydrology, water chemistry and any existing instrumentation in the chosen fens was used to create preliminary conceptual models of the research sites. The model can give preliminary information on the flow regime in the fen and therefore show areas that could be sensitive to pressures, be it from either drainage or nutrient inputs. These models were then used to identify the areas where piezometer transects should be installed and adapted utilising the data in results Chapters 5 to 9. The preliminary conceptual models and suggested piezometer locations are shown and discussed per site at the end of Section 3.2 Ballymore, 3.3 Pollardstown Fen. 3.4 Scragh Bog and 3.5 Tory Hill. Data used for these conceptual models are described and referenced throughout those sections.

3.2. Ballymore Fen

3.2.1. Study area

The site is of considerable conservation significance because of its overall variety of habitats and species in a relatively small area and the occurrence of locally rare and protected plant and amphibian species.

3.2.2. Ecology

Ballymore Fen supports an excellent example of transition mire in association with calcareous fen and developing raised bog and supports a great diversity of flora and fauna on a relatively small area some of which are legally protected (NWPS, 2014a). A habitat map of the site is shown in Figure 3.3.

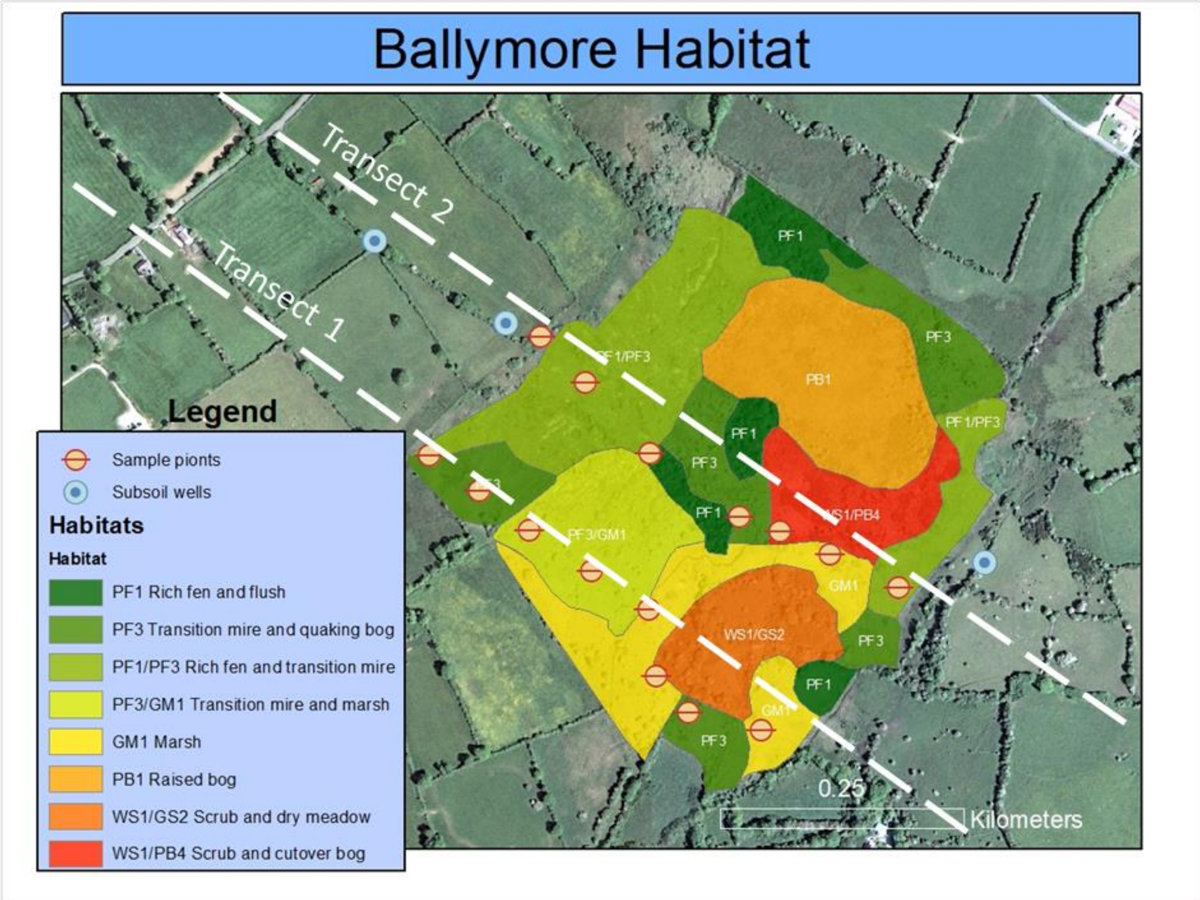


Figure 3.3. Habitat map of Ballymore (Adapted from Regan & Connaghan (2017)).

Ballymore is designated as a Special Area of Conservation (SAC) (NWPS, 2015a).The site has been selected for an area with transition mire habitat which falls under the E.U. Habitats Directive. This habitat type has a good degree of representativity in the area. Two other habitats, reported as alkaline fen and degraded raised bog (still capable of natural regeneration) were found on site as well (NWPS, 2015a). However, the presence of these habitat types were said to be of no significance. Still, these fen habitat types account for 16% of the total area. The area of all Annex I habitat types cover around 40% of the total protected area (NWPS, 2015a).

No Annex II species were reported at the site. However, the fen supports the locally rare sedge species (*Carex limosa*) and has an excellent diversity of bryophytes (NWPS, 2015a). Furthermore, the round-leaved wintergreen (*Pyrola rotundifolia*) which is included in the Red Data Book is found between hummocks. Two legally protected amphibian species, the common frog (*Rana temporaria*) and the Smooth Newt (*Triturus vulgaris*) were found in the flowing streams and drains (NWPS, 2015a) (NWPS, 2016b). The overall aim of the Habitats Directive is to achieve favourable conservation status of the mentioned habitat which is of community interest (NWPS, 2016b).

3.2.3. Geology and structure

The site expands over an area of 43.1 ha and lies approximately 17 km west of Mullingar in county Westmeath (NWPS, 2014a). The fen is surrounded by a calcareous drift and occupies a wide and deep depression which may have been a lake in the past. The majority of the basin consist of a peat layer underlain by Carboniferous Limestone. The peat is separated from the bedrock by sandy clay and marl (NWPS, 2014a). The site also contains a mineral mound in the south remnants of active and cutover bog in the north.

From soil logs recorded during installation and soils logs recorded by Regan & Connaghan (2017) geology transects were drawn of areas that were investigated (see Figure 3.5 and 3.6). The legend of the different soil types can be found in Figure 3.4.

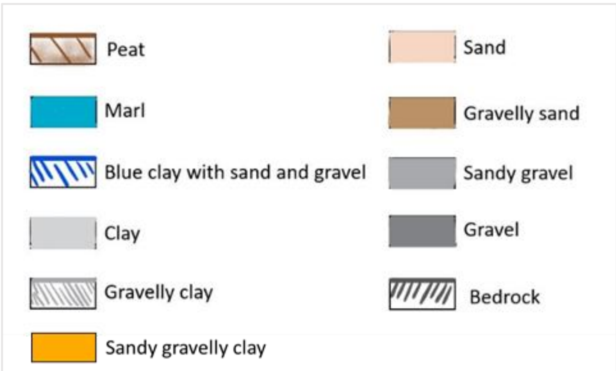


Figure 3.4. Soil type legend of Ballymore fen.

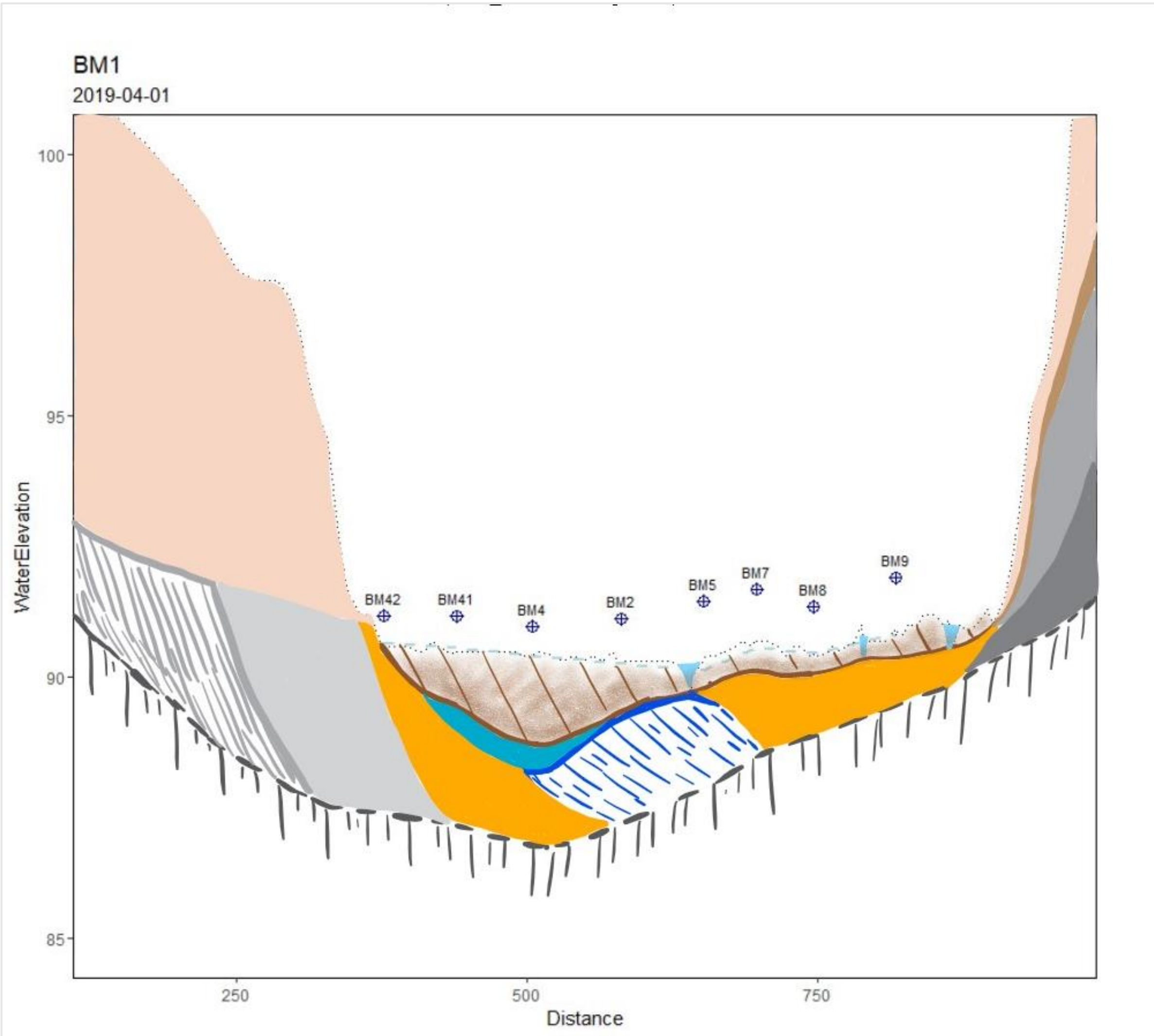


Figure 3.5. Geology transect 1, Ballymore. The transect location can be found in Figure 3.3 and moves from the northwest to the southeast. Figure 3.4 contains the legend which explains the soils shown on the transect.

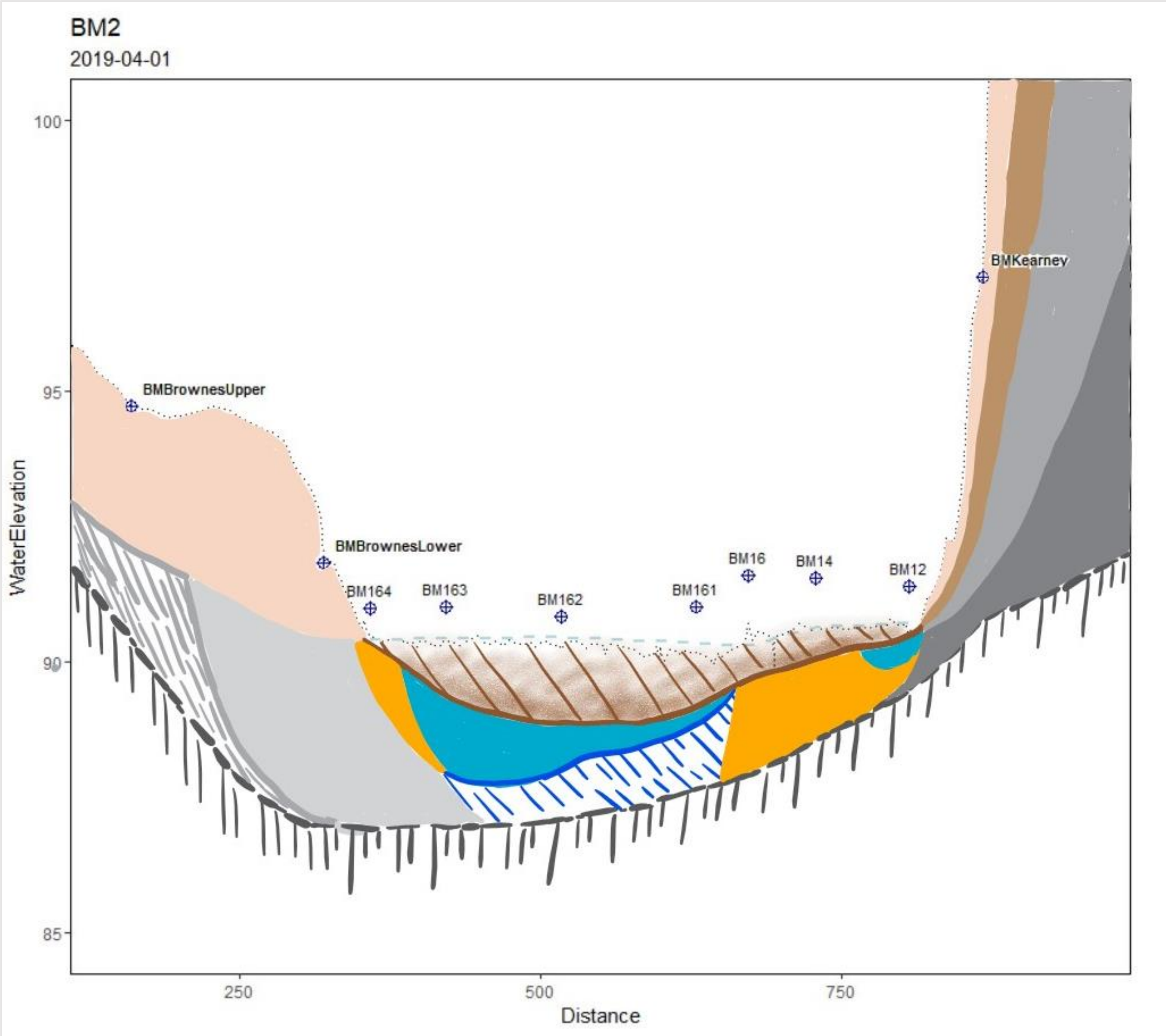


Figure 3.6. Geology transect 2, Ballymore. The transect location can be found in Figure 3.3 and moves from the northwest to the southeast. Figure 3.4 contains the legend which explains the soils shown on the transect.

3.2.4. Hydrogeology

The site is fed by springs from both east and west. Small streams are running through from the north-east and south and wetter fen areas are located towards the centre and south of Ballymore (NWPS, 2015a). A more extensive hydrological investigation was executed by Regan & Connaghan (2017) for National Parks & Wildlife Service. They found that a consistently high groundwater table has resulted in the presence of excellent expanses of alkaline fen and transition mire habitat. This is due to a relatively large groundwater catchment area as well as minimal drainage by the fen's drain. Furthermore, daily precipitation and ET were obtained from Met Eireann station Mullingar.

3.2.5. Land use

On the slopes surrounding the fen contains a mosaic of improved and semi-improved species-rich calcareous grasslands which is lightly grazed by cattle. Some heather covered ridges and banks on the site suggest that the fen has been cut for turf in the past (NWPS, 2014a). According to a local landowner peat cutting by hand was practiced probably for a few hundred years. In the 19th century peat cutting was predominantly executed from the west using the still existent 'boreen' to haul the peat to the main road. The cutting has seized since 1988. Currently, fen vegetation is regenerating in these area and the ground below is very wet and soft. Furthermore, an OPW arterial drain (C8/7/3) which is connected to the natural drain of the site (Figure 3.3) was widened and deepened 1960, as part of the River Inny arterial drainage scheme. The drain in the fen was left relatively untouched.

Pressures with medium negative impact are point source pollution to surface water and the occurrence of problematic native species. Occurring pressures with low negative impact are fertilisation and lack of grazing on pastoral systems (NWPS, 2015a).

3.2.6. Preliminary conceptual model

Preliminary conceptual models of Ballymore fen are presented in Figure 3.7 and 3.8 utilising the information gathered during the desk study. From this possible impacts such as drainage and nutrient pollution are identified and placed on the model. Finally, piezometer and phreatic tube locations are chosen in places that encompass a wide array of fen ecology. They are also placed along hypothesised direction of flowlines. This way the piezometers should identify where the water enters the fen and how it moves through its sediments.

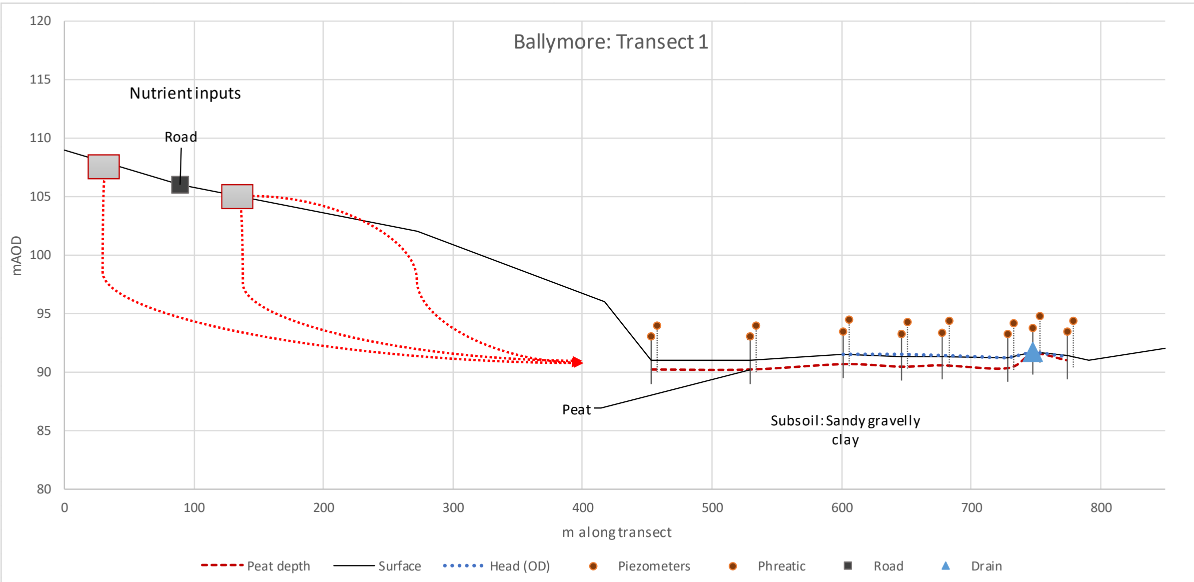


Figure 3.7. Preliminary conceptual model of Ballymore fen transect 1. The transect location can be found in Figure 3.3 and moves from the northwest to the southeast. Created utilising data from Regan & Connaghan (2017). mOAD is in meters Above Ordnance Datum (AOD).

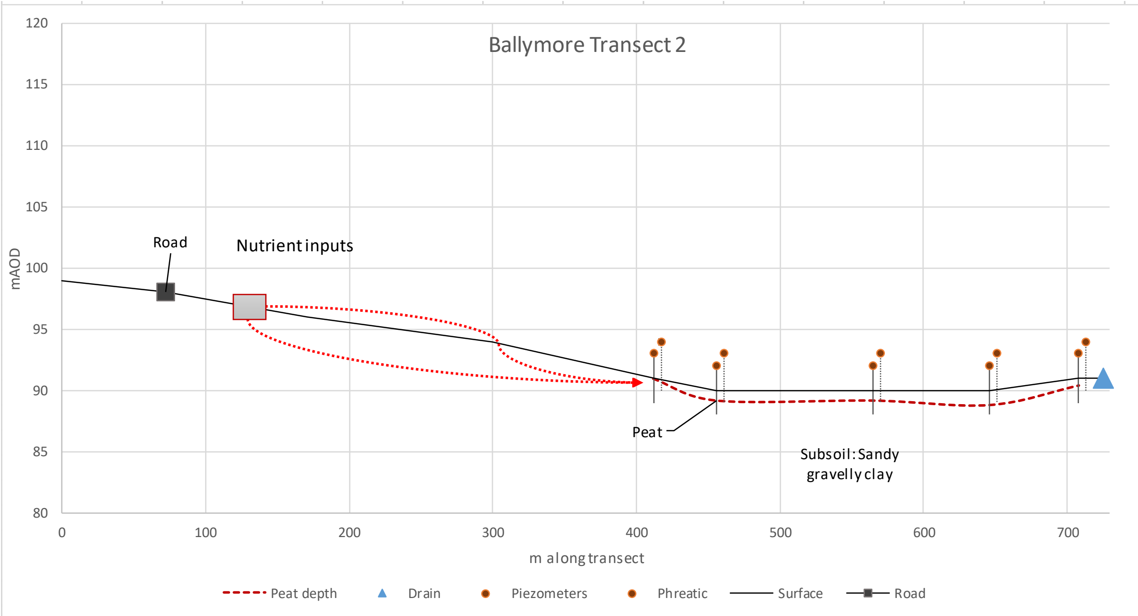


Figure 3.8. Preliminary conceptual model of Ballymore fen transect 2. The transect location can be found in Figure 3.3 and moves from the northwest to the southeast. Created utilising data from Regan & Connaghan (2017). mOAD is in meters Above Ordnance Datum (AOD).

3.3. Pollardstown Fen

3.3.1. Study area

Pollardstown Fen is the largest remaining calcareous spring-fed fen in Ireland. It contains an ecosystem of international importance because of its unique and endangered plant communities (NPWS, 2013).

Pollardstown Fen is a well-studied site. Ecological research was executed on invertebrates, insects, and spiders (Helsdingen, van, 1997) (Kuczynska & Moorkens, 2010) as well as hydrological and geological research (Mistear, Brown, & Johnston, 2009) (Kuczynska A. , 2008) (Daly, 1981). This site is of considerable conservation value because of its good representation of the three Annex I habitats and in particular the rarity and numbers of flora and fauna species in these habitats.

Since Pollardstown fen is of considerable size two research sites (Site A and site D) were selected based on research conducted before. Figure 3.9 shows a map of the fen with the locations of those sites as well as the main springs and outlets.

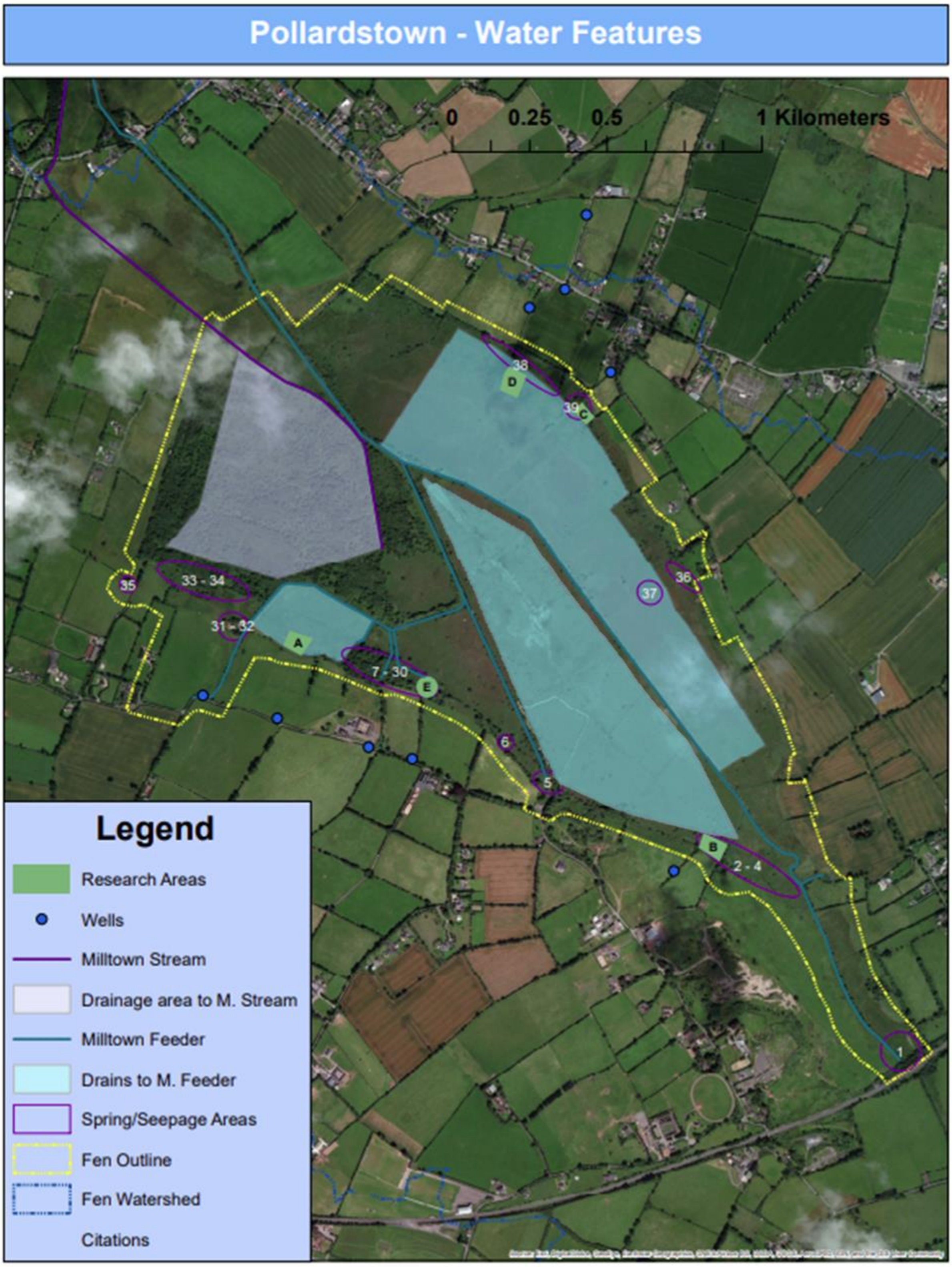


Figure 3.9. General water features and research sites of Pollardstown Fen (Adapted from Kuczyńska (2008))

3.3.2. Ecology

The site is designated as a Special Area of Conservation (SAC) and Natural Heritage Area (NHA). Furthermore, much of the site with fen vegetation is now owned by the Office of Public Works and is a Statutory Nature Reserve since 15 December 1986 (Parkes & Sheehan-Clarke, 2005) (NPWS, 2013).

The site has been selected for a number of habitats and species under the E.U. Habitats Directive such as Cladium fen, petrifying fen and alkaline fen listed in Annex I (NPWS, 2013). These habitat types have a high degree of representativity and cover around 30% of the total area (NWPS, 2014). Other habitats found on the site are humid grassland, artificial monoculture forest and standing water bodies. Habitat maps of the sites is shown in Figure 3.10 and 3.11.

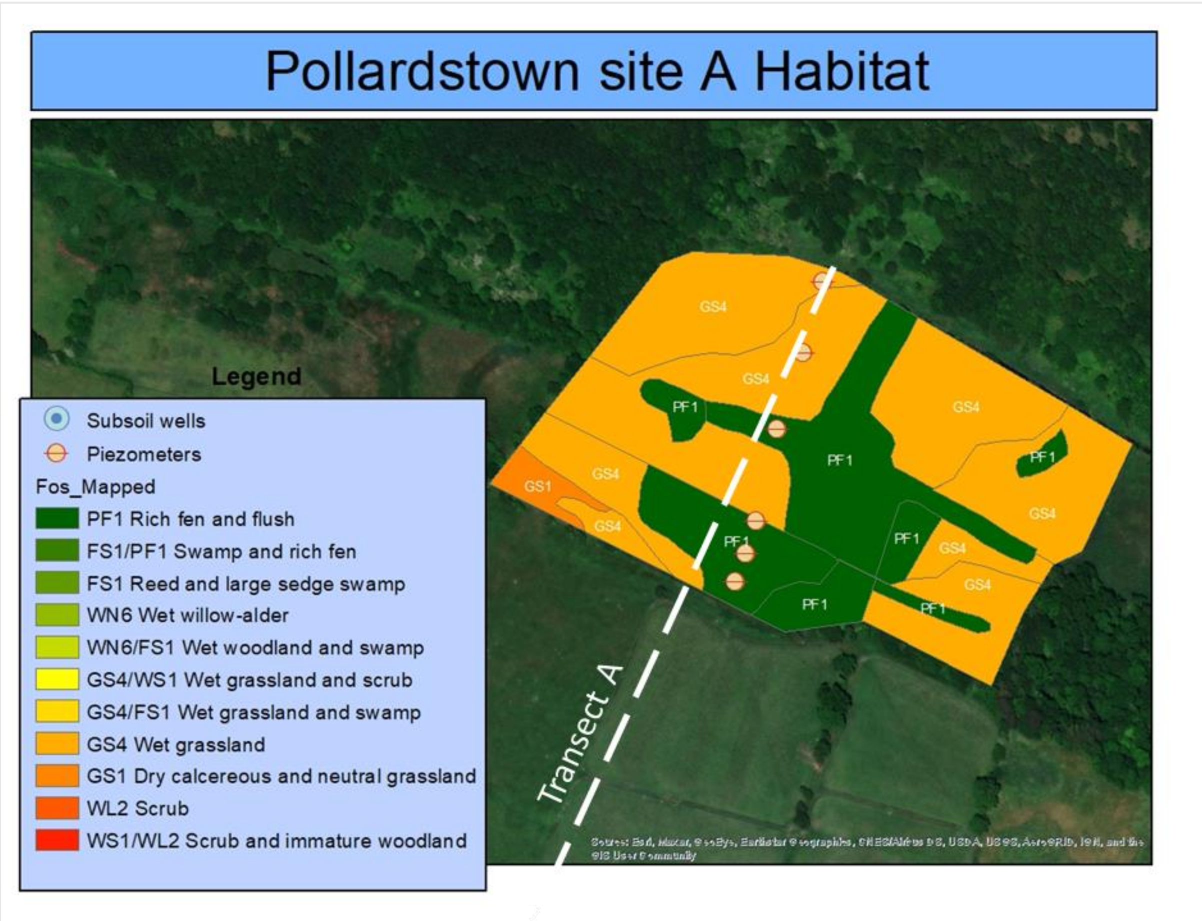


Figure 3.10. Habitat map of Pollardstown site A (Provided by BEC).

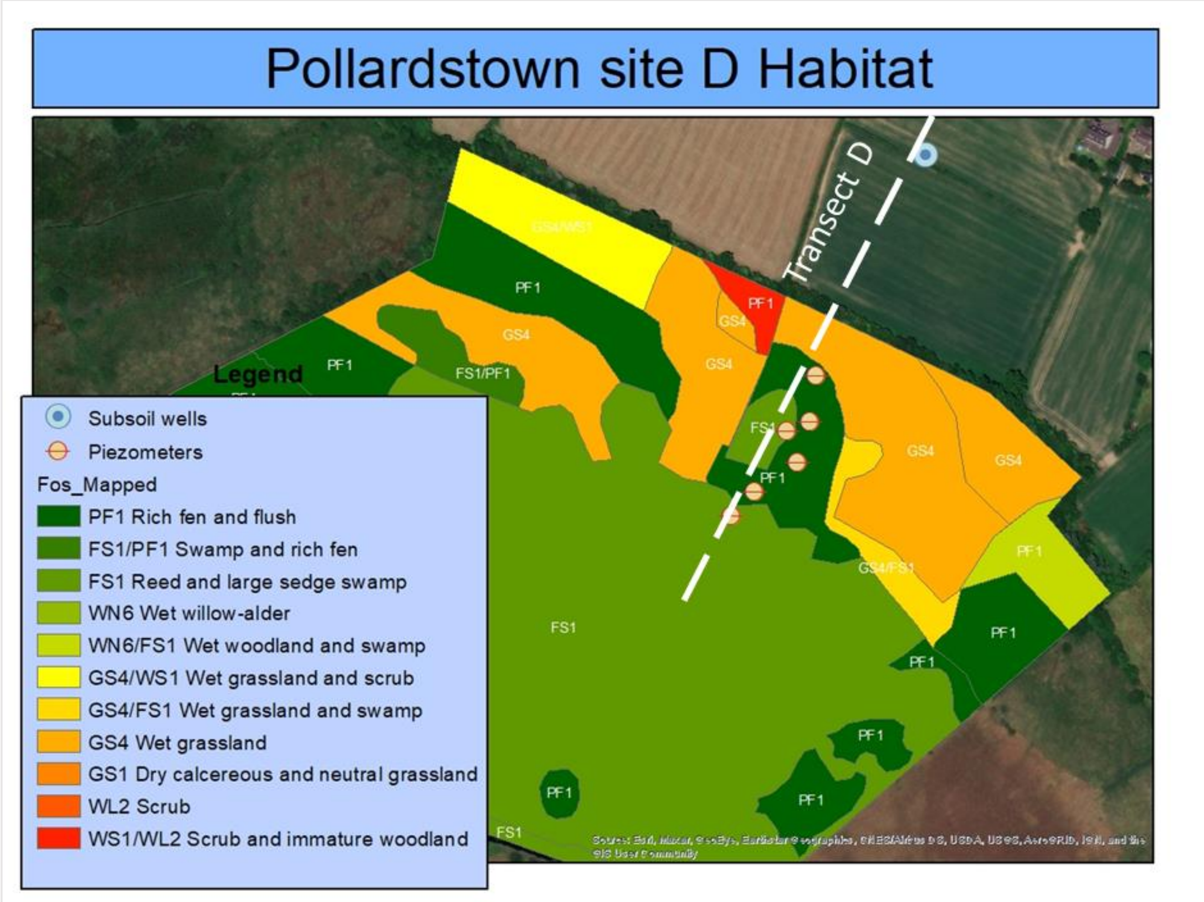


Figure 3.11. Habitat map of Pollardstown site A (Provided by BEC).

Three species of Annex II Whorl snails (*Vertigo geyeri*, *V. angustior* and *V. moulinsiana*) are found in the site (NWPS, 2014). Pollardstown is the only known site in Ireland which supports all three of these rare sub-aquatic invertebrate species (NPWS, 2013). Other Annex II species include birds such as the Teal, Mallard, Coot, Snipe, Grasshopper Warbler, Sedge Warbler and Whinchat that use the fen vegetation for breeding and wintering grounds (NWPS, 2014). Vertebrates species found in Pollardstown and included in Annex II species are the Otter, Brook Lamprey and White-clawed Crayfish (Kuczynska A. , 2008) (NPWS, 2013). The overall aim of the Habitats Directive is to achieve favourable conservation status of the mentioned species and habitats that are of community interest (NWPS, 2016).

3.3.3. Geology and structure

The fen is situated on the northern margin of the Curragh aquiver in county Kildare (NPWS, 2013) (Mistear, Brown, & Johnston, 2009). The site is 266 ha in area and lies in a shallow basin consisting of peat-marl deposit. The peat layered with calcareous marl reaches a maximum depth of 6 meters and is underlain by clay (Mistear, Brown, & Johnston, 2009) (NWPS, 2014) (Parkes & Sheehan-Clarke, 2005).

From soil logs recorded during installation and soils logs recorded in Kuczynska, A. (2008) geology transects were drawn of areas that were investigated (see Figure 3.13 and 3.14)

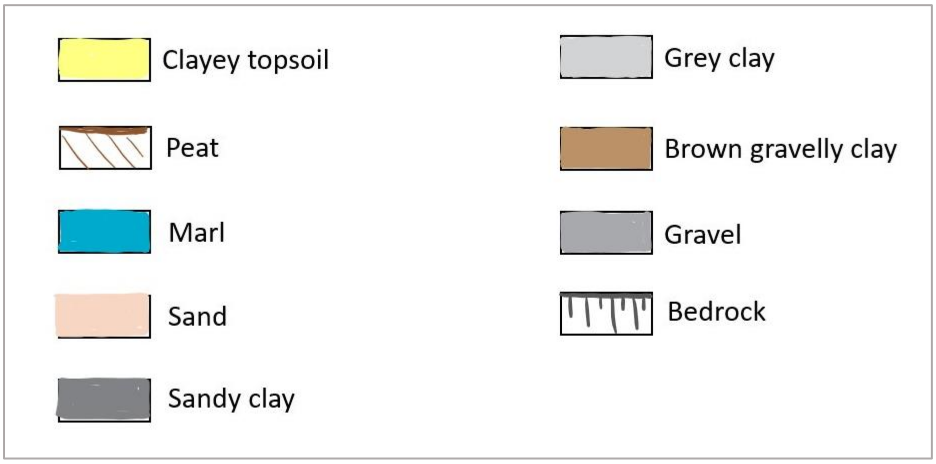


Figure 3.12. Soil type legend of Pollardstown fen.

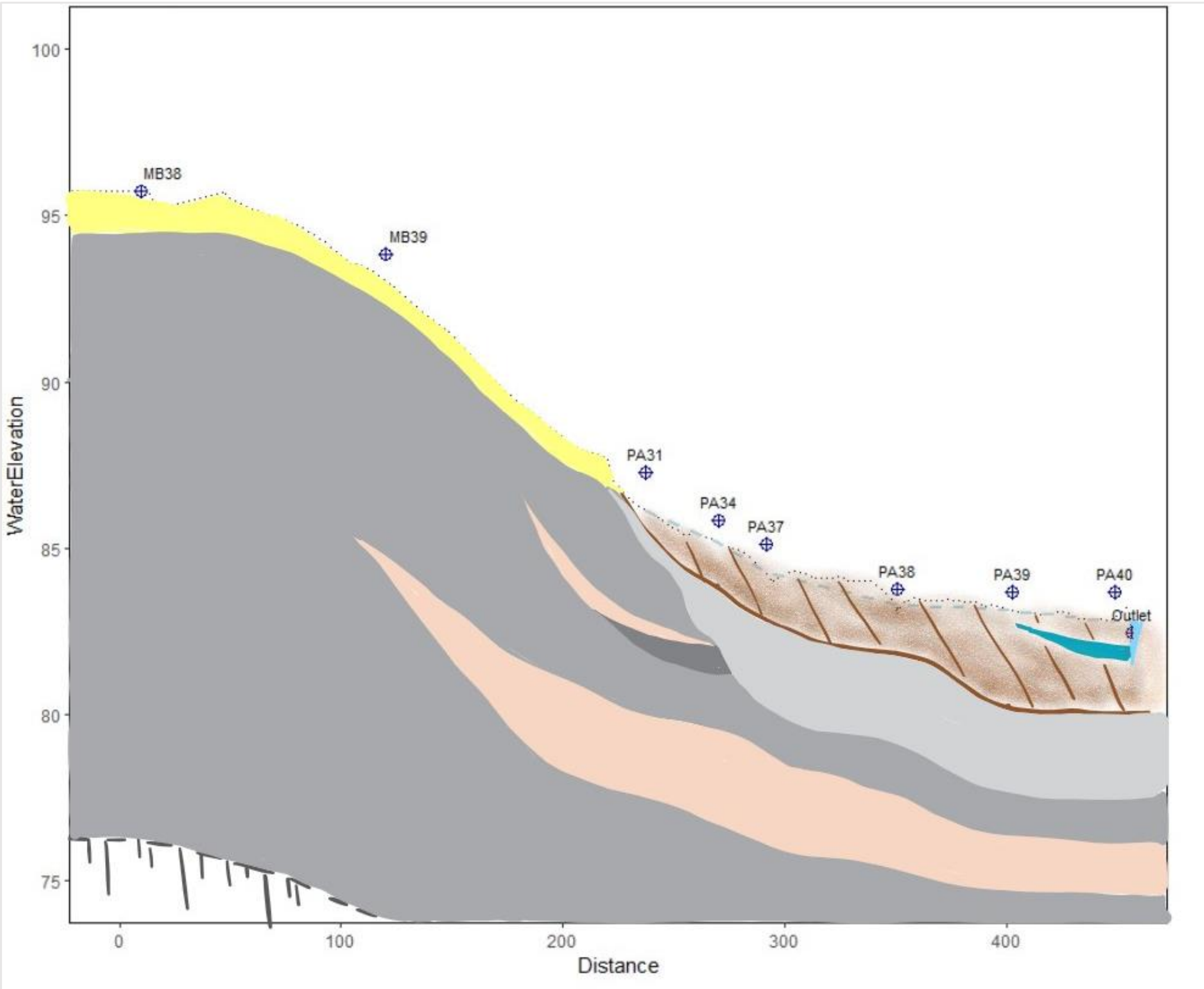


Figure 3.13. Geology transect Pollardstown site A. Figure 3.12 contains the legend which explains the soils shown in the transect. The transect location can be found in Figure 3.10 and moves from the south to the north.

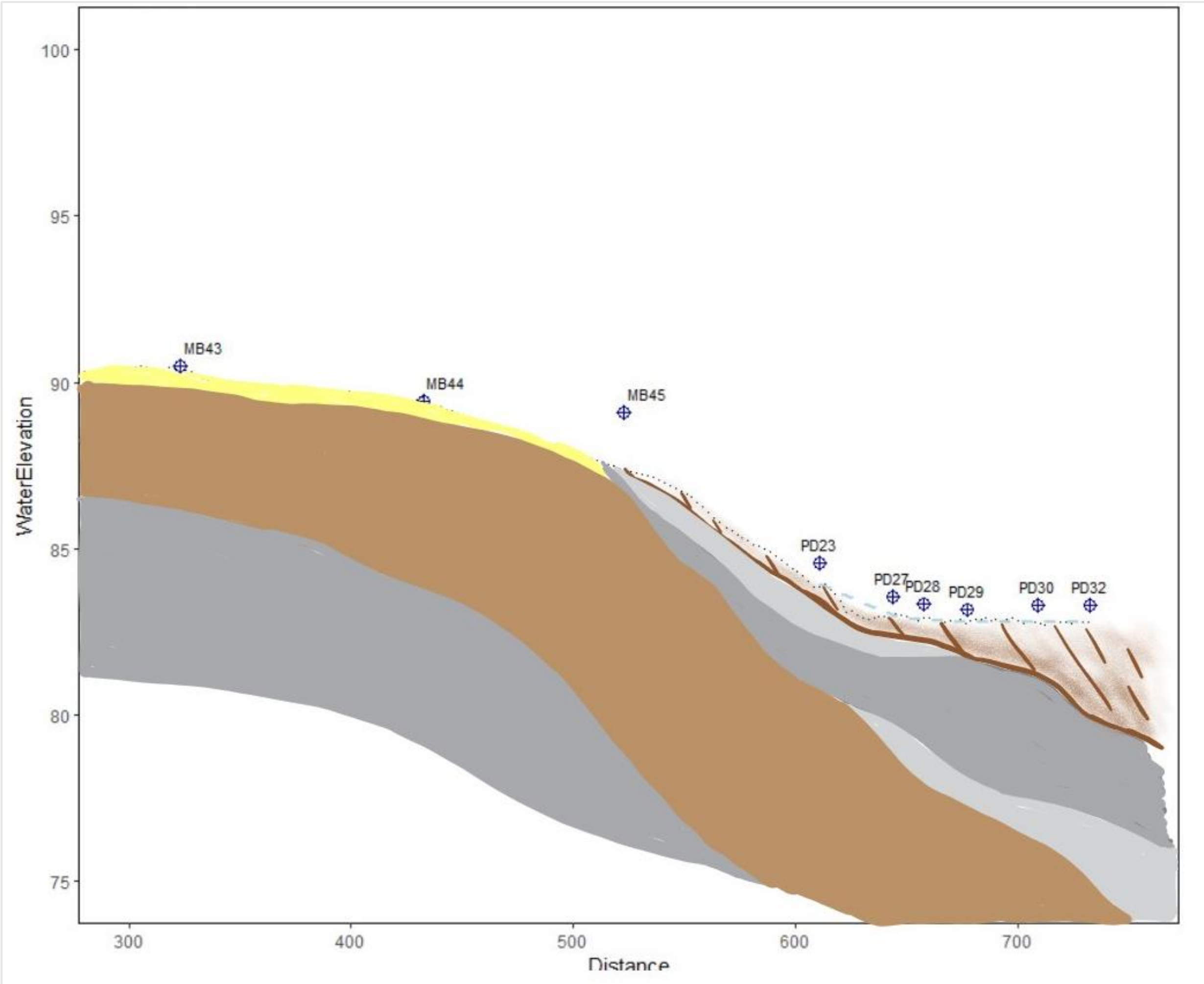


Figure 3.14. Geology transect Pollardstown site D. The transect location can be found in Figure 3.12 and moves from the north to the south. Figure 3.11 contains the legend which explains the soils shown on the transect.

3.3.4. Hydrogeology

Around forty springs provide a constant supply of mineral-rich water from the aquifer and the lime-stone ground in the north. Because of the glacial deposits and the mineral rich ground water flow the formation of Pollardstown Fen began after the Midlandian glaciation (Parkes & Sheehan-Clarke, 2005). Peat accumulated under waterlogged conditions and formed a wetland fed by groundwater known as a fen (Mistear, Brown, & Johnston, 2009) (NPWS, 2013) (Kuczynska A. , 2008). Furthermore, daily rainfall data was obtained from Naas (Osberstown) weather station and evaporation and

evapotranspiration from Mullingar weather station.

3.3.5. Land use

Restoration has been conducted in 1983 following partial reclamation executed in 1979. This was done by re-flooding of the central fen area to allow re-establishment and expansion of aquatic and reed swamp vegetation and their associated fauna (NWPS, 2014) (NPWS, 2013). However, a number of activities, treats and pressures remain that have an impact on the site. Current pressures occurring at Pollardstown were updated by the NWPS (2014), these include: high impact from grazing and medium impact from dispersed habitation, disposal of inert materials and sand and gravel extraction. Some low impact activities on the site such as hunting, fishing and fire suppression were reported as well.

3.3.6. Preliminary conceptual model

Preliminary conceptual models of Pollardstown fen with possible negative impacts and phreatic tube and piezometer locations are presented in Figure 3.15 and 3.16 utilising the information gathered during the desk study.

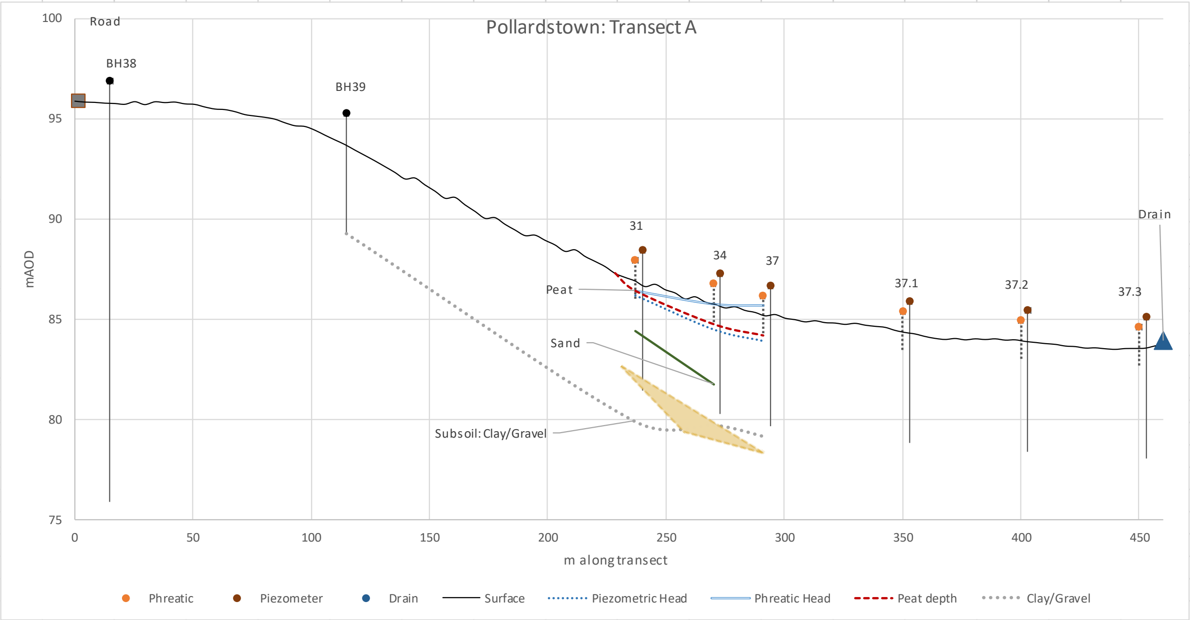


Figure 3.15. Preliminary conceptual model of Pollardstown transect A. The transect location can be found in Figure 3.10 and moves from the south to the north. Created utilising data from Mistear, Brown, & Johnston, 2009; Kuczynska, 2008. mOAD is in meters Above Ordnance Datum (AOD).

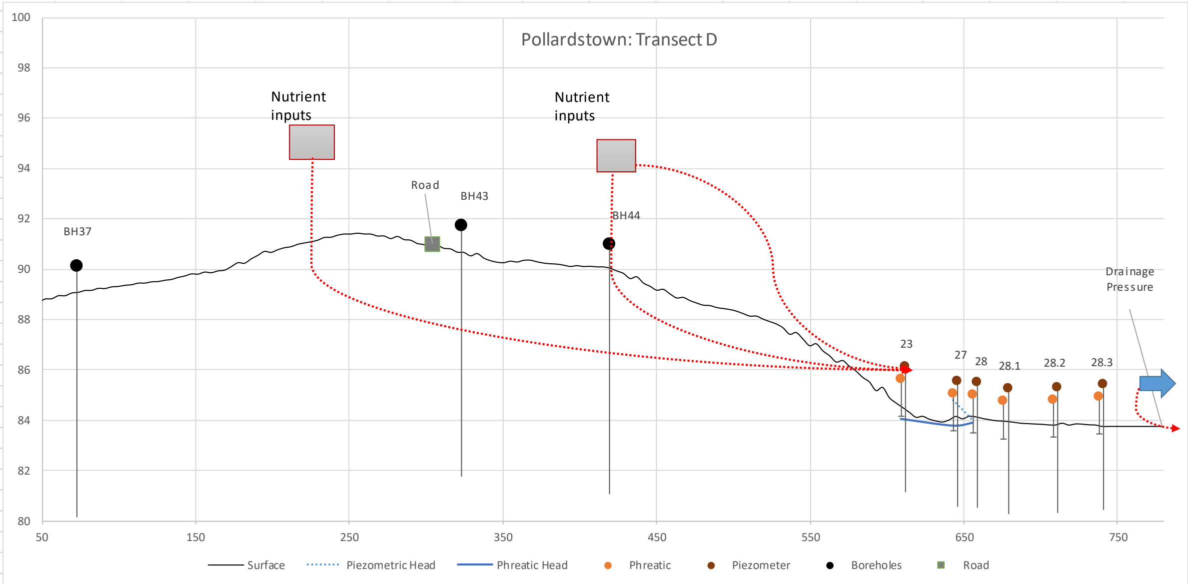


Figure 3.16. Preliminary conceptual model of Pollardstown transect D. The transect location can be found in Figure 3.11 and moves from the north to the south. Created utilising data from Mistear, Brown, & Johnston, 2009; Kuczynska, 2008. mOAD is in meters Above Ordnance Datum (AOD).

3.4. Scragh Bog

3.4.1. Study area

Scragh Bog is a fen with a floating vegetation raft and with typical plants including sedges *Carex spp*, black bogrush *Schoenus nigricans*, bog bean *Menyanthes trifoliata* and brown mosses *Scorpidium scorpioides* (NPWS, 2019) The Irish term 'bogach', meaning soft boggy ground, is used for peatlands in general. Therefore, this name does not distinguish the bogs and fens. Ecological and phytosociological research as well as sediment analytical studies were conducted by O'Connell (1980 and 1981). Additionally, a more recent study analysed vegetation, water and soil samples and found that the N and P plant tissue concentrations were higher in the buffer zone of the fen and the vegetation composition indicated eutrophic species (Paullissen et al., 2016).This site is of considerable conservation value because of its good representation of two Annex I habitats and in particular the rarity of flora and fauna species in these habitats.

3.4.2. Ecology

Scragh Bog is designated as a Special Area of Conservation (SAC) since 2018. The site also has been selected for a number of habitats and species under the E.U. Habitats Directive such as alkaline fen and transition mires listed in Annex I (NPWS, 2019). These habitat types have a high degree of representativity and cover around 45% of the total area. Other habitats found on the site are bog woodland, swamp, scrub and standing water bodies. A habitat maps of the site is shown in Figure 3.17.

Furthermore an Annex II protected plant species was found in the fen; the moss *Hamatocaulis vernicosus. Furthermore the* Annex II species also included the Marsh Fritillary. The overall aim of the Habitats Directive is to achieve favourable conservation status of the mentioned species and habitats that are of community interest.

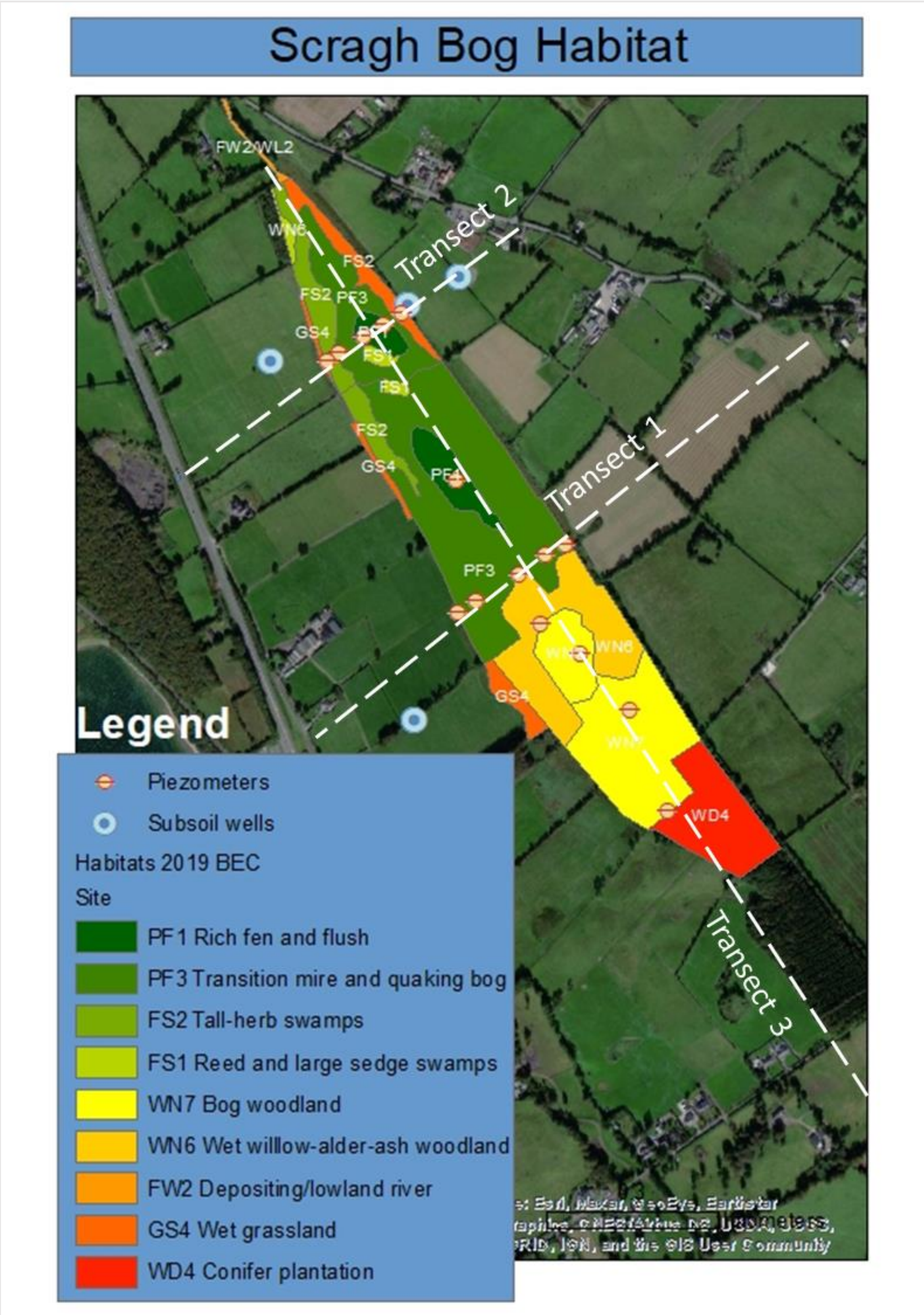


Figure 3.17. Habitat map of Scragh Bog (Provided by BEC).

3.4.3. Geology and structure

The fen is located in an oval-shaped depression within gently undulating countryside which geologically is part of the Carboniferous central plain (O'Connell, 1981). The glacially deposited ridges, consisting of sand, gravel and clay, are used as meadows (O'Connell 1980). The fen has an area of 23.8 ha (NPWS, 2019). From soil logs recorded during installation geology transects were drawn of areas that were investigated (see Figure 3.18 and 3.19).

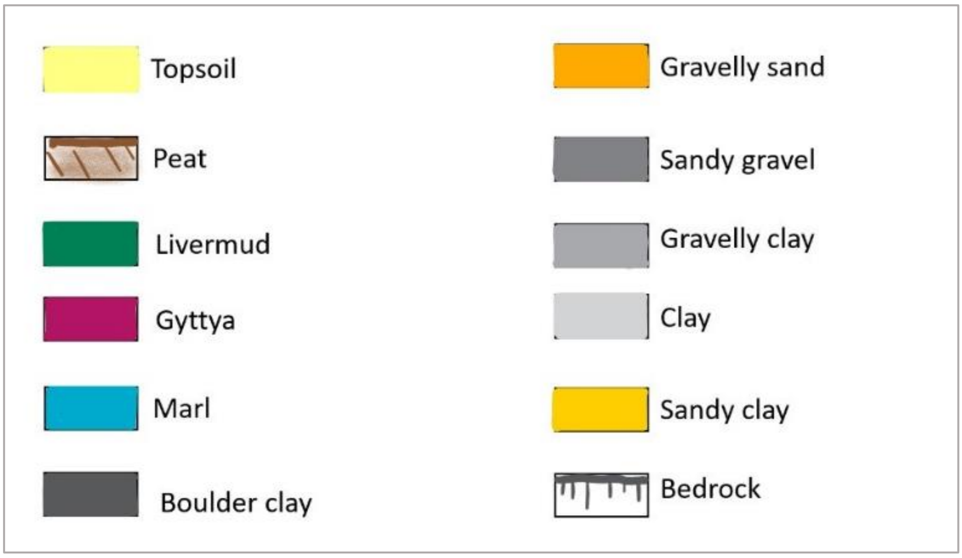


Figure 3.18. Soil type legend of Scragh Bog.

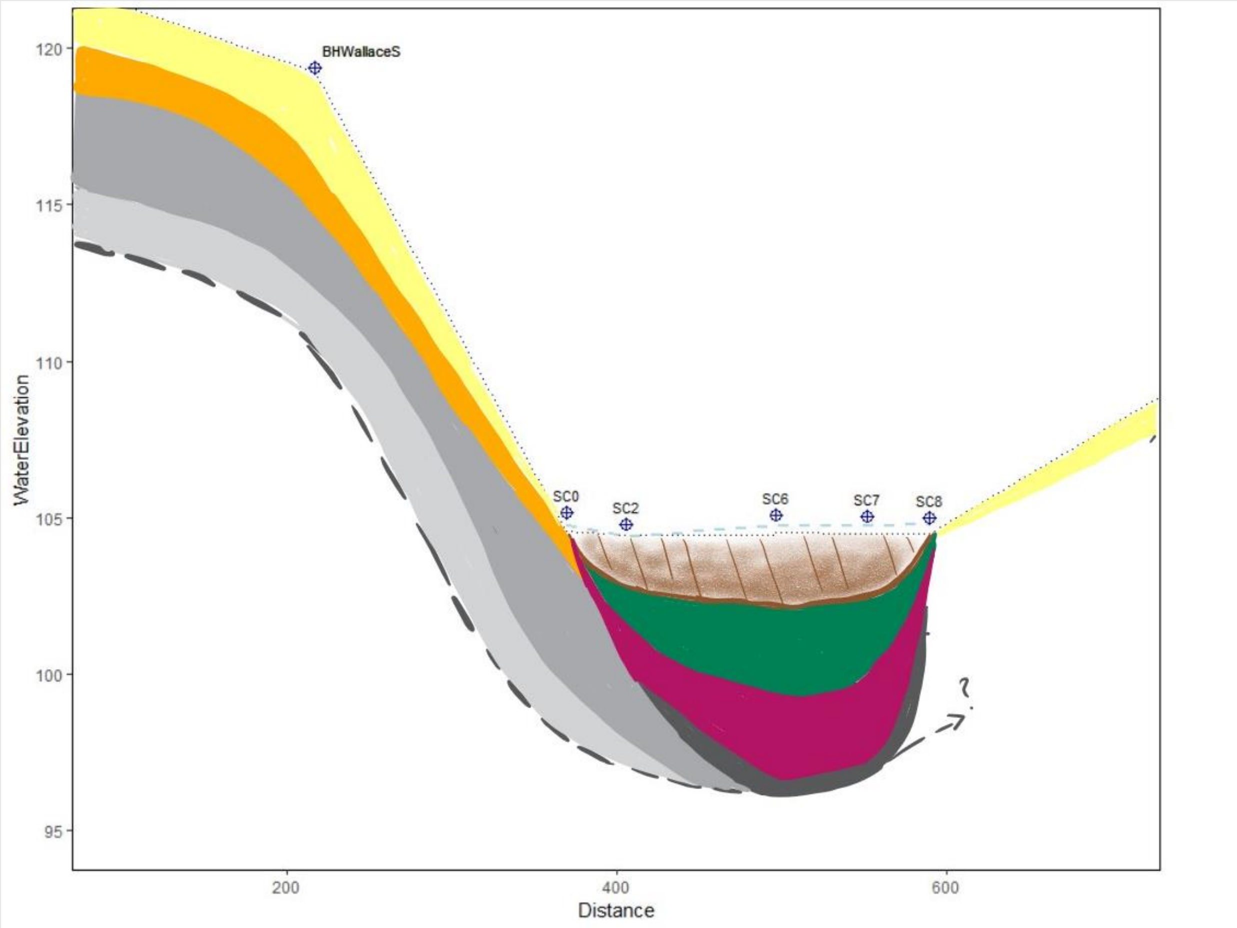


Figure 3.19. Geology transect Scragh Bog transect 1. The transect location can be found in Figure 3.17 and moves from the south west to the north east. Figure 3.18 contains the legend which explains the soils shown in the transect.

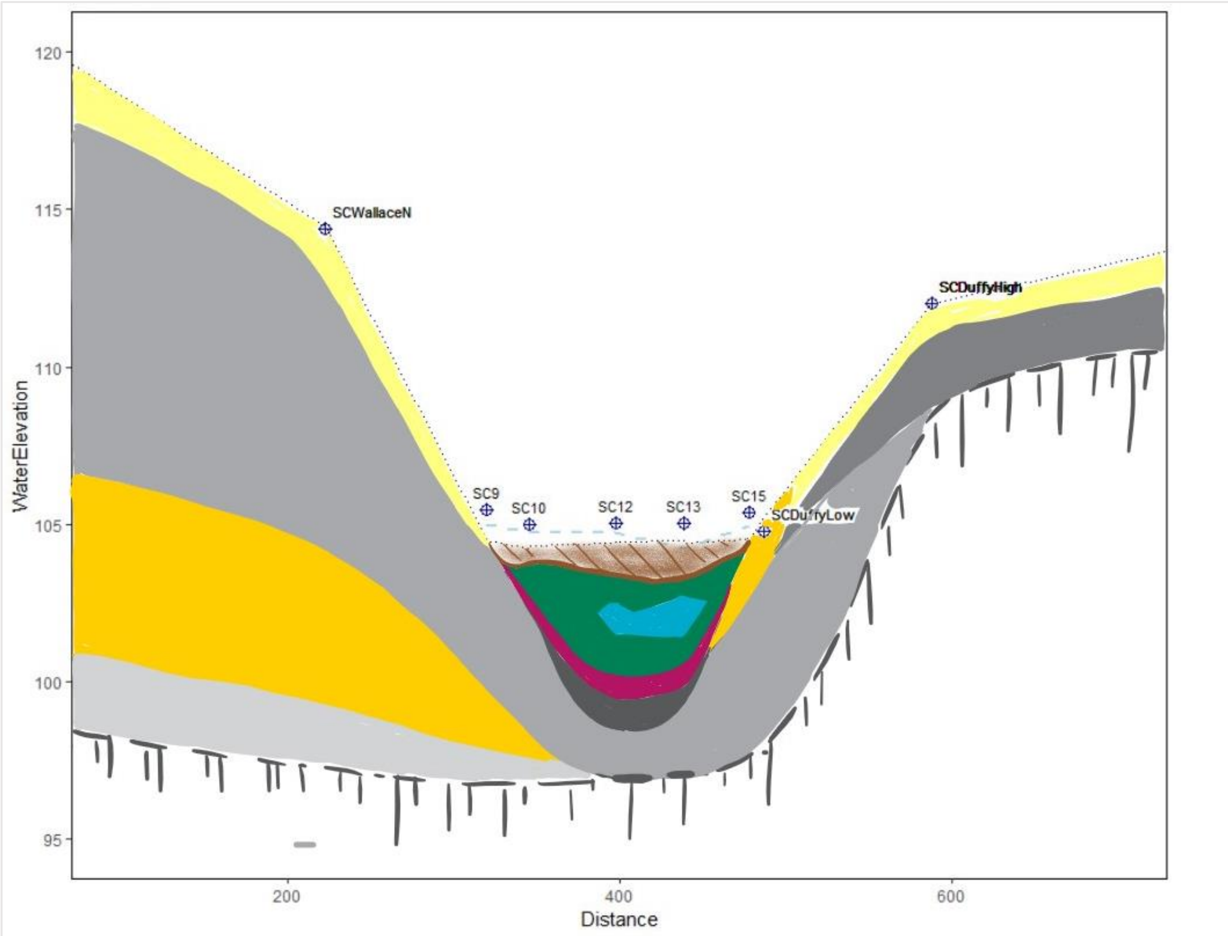


Figure 3.20. Geology transect Scragh Bog transect 2. The transect location can be found in Figure 3.17 and moves from the south west to the north east. Figure 3.18 contains the legend which explains the soils shown in the transect.

3.4.4. Hydrogeology

Scragh Bog is drained by a single artificially deepened outlet located in the northwest, which is maintained by the OPW. There are no inflowing streams, but some weak surface springs were noted near the south-eastern end by O'Connell (1981). To the west of the fen lies Lough Owel, which is separated by a low ridge. Rainwater that falls on the adjacent eskers flows via runoff, or after infiltration as shallow groundwater to the area. These eskers are grazed and fertilized with manure and artificial fertilizers and this might form a source for eutrophication. Furthermore, daily precipitation and ET were obtained from Met Eireann station Mullingar.

3.4.5. Land use

At the south-eastern end, the fen vegetation is replaced by carr which in turn has been partly displaced by *Pimis sylvestris* which was planted in 1870. Afforestation was resumed in this part in 1949 and continued in 1951 with the planting of *Picea abies, Pinus sylvestris* and some *Abies alba* (O'Connell, 1981). There have been reported a number of activities, treats and pressures by the NWPS (2019) that may have an impact on the site, these include: medium impact from fertilisation, diffuse pollution to surface waters due to household sewage and waste waters as well as agricultural roadways close to the fen. Agricultural activities were reported to have low impact on the site.

Interestingly these pressures may have caused some changed to happen in Scragh Bog fen, however they also may have been caused by natural succession. In Figure 3.21 an aerial photograph made in 1972 was compared to a snap shot made by drone survey conducted by GeoAerospace in 2019. From this it seems like a significant amount of tree and scrub has encroached (A) onto the fen which could be proof of nutrient pollution where more eutrophic plant species outcompete the fen vegetation. This may also be apparent in the scrub growth (B) disperse through the fen. It further seem like the *Schoenus-dominated* fen community area (D) has decreased over time. A more positive change is also observed where pool were filled in by fen vegetation (C) which could be a clear sign of natural succession.

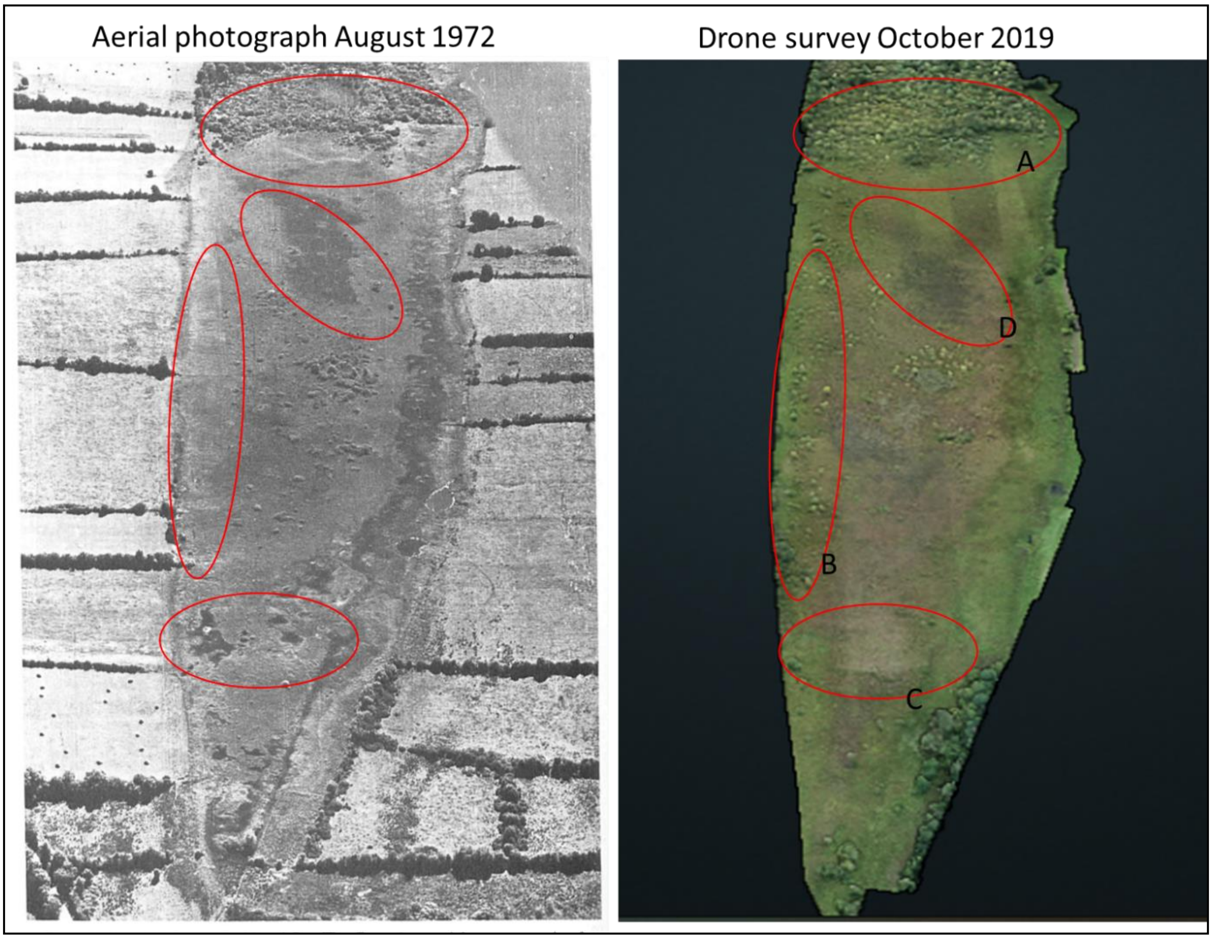


Figure 3.21. Comparison of an aerial photograph taken by L. Swann in 1972 (left) and a drone survey conducted by GeoAerospace in 2019 (right). The aerial photograph was presented by O'Connell (1981) and has the following description: *"Oblique aerial photograph of Scragh Bog. The afforested area and the carr community of the south-eastern end lie at the top of the photograph. The Schoenus-dominated fen community appears as a dark patch approximately 1cm from the carr. In the centre Salix and Betula trees constitute a prominent feature. Water bodies appear as dark areas along the right-hand side and in the bottom part of the photograph. Two lighter patches with white borders at the right-hand side denote drinking pools used by cattle."* The figures are compared at certain areas by red circles. Changes

are circled in red and have the following captions, A) Tree and scrub encroachment, B) Scrub growth, C) Filling in of waterbodies, D) Schoenus-dominated fen community area seems to decrease.

3.4.6. Preliminary conceptual model

Preliminary conceptual models of Scragh bog with possible negative impacts and phreatic tube and piezometer locations are presented in Figure 22, 23 and 24 utilising the information gathered during the desk study.

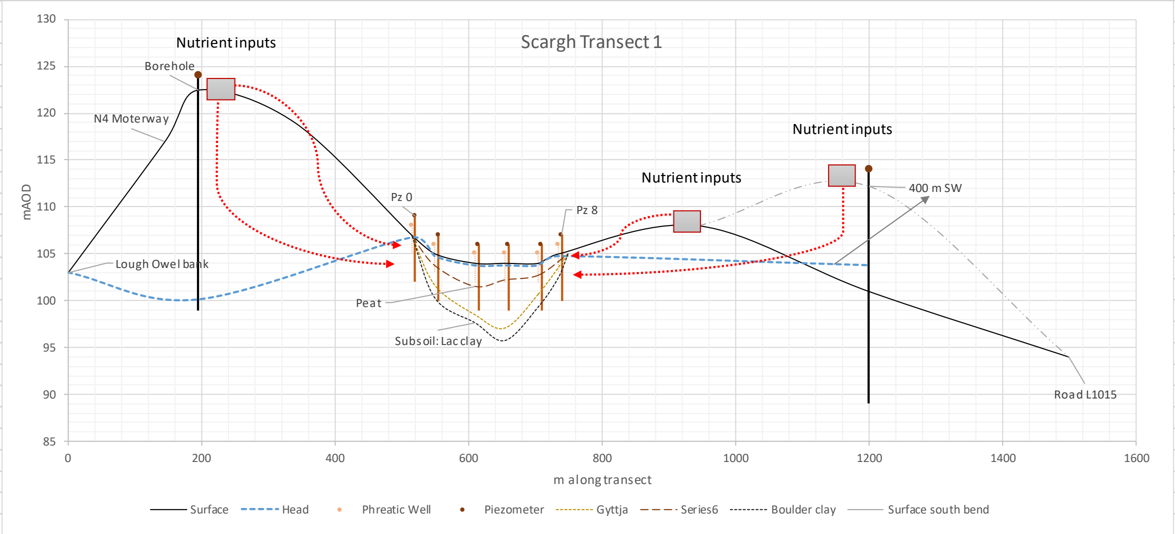


Figure 3.22. Conceptual model of Scragh bog transect 1. The transect location can be found in Figure 3.17 and moves from the south west to the north east. Created utilising data from O'Connell 1980. mOAD is in meters Above Ordnance Datum (AOD).

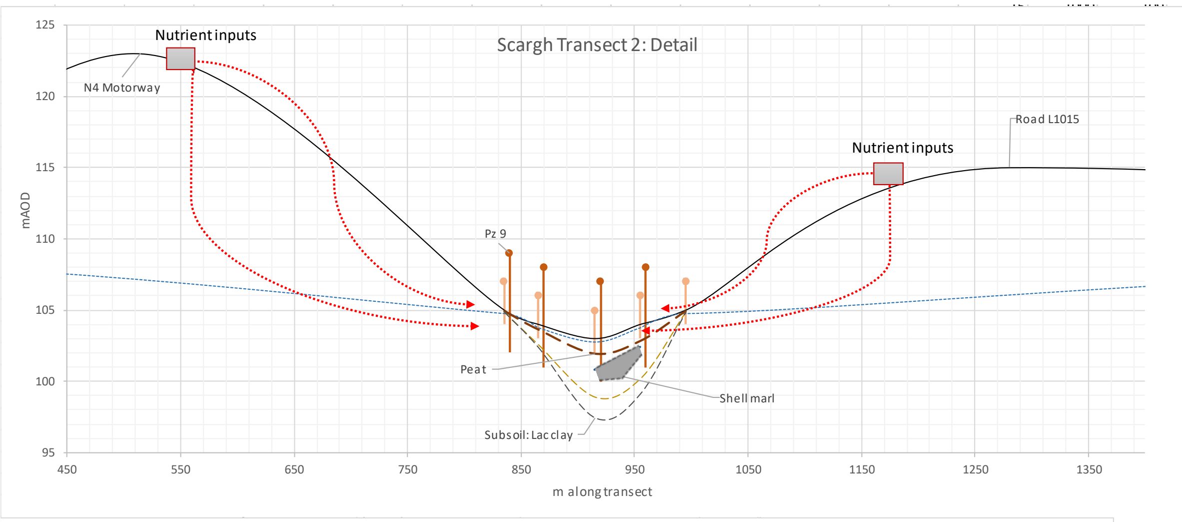


Figure 3.23. Preliminary conceptual model of Scragh bog transect 2. The transect location can be found in Figure 3.17 and moves from the south west to the north east. Created utilising data from O'Connell 1980. mOAD is in meters Above Ordnance Datum (AOD).

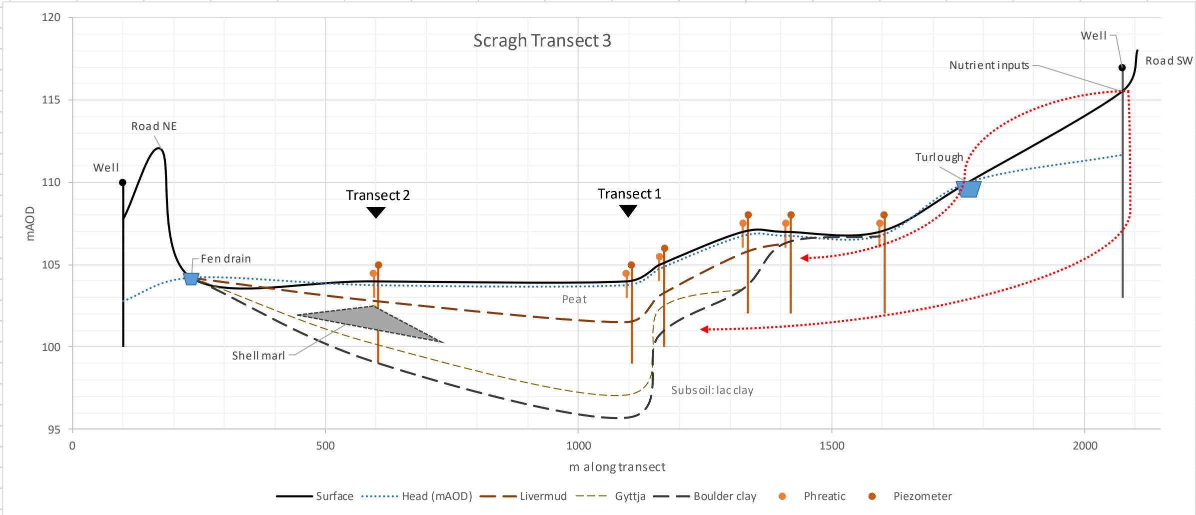


Figure 3.24. Preliminary conceptual model of Scragh bog transect 3. The transect location can be found in Figure 3.17 and moves from the north to the south. Created utilising data from O'Connell 1980. mOAD is in meters Above Ordnance Datum (AOD).

3.5. Tory Hill

3.5.1. Study area

Tory Hill Fen is of great geomorphological interest because of its excellent example of an end-moraine amongst a region of volcanic intrusions (NWPS, 2013a). Furthermore, the site has an excellent diversity of habitats and species on relative small area (NWPS, 2015).

Multidisciplinary palaeoecological research has been executed on the site in form of pollen analysis and analysis of stable isotopes of late-glacial sediments (O'Connell, Huang, & Eicher, 1999). Other research done in Tory Hill included an eco-hydrological study (Regan & Connaghan, 2016) as well as a conservation study assessed under the Habitats Directive and Water Framework Directive (Regan et al, 2016). The latter research found the site to have an unfavourable ecological condition due to degradation caused by groundwater pressures. Tory Hill is of considerable conservation value because of its habitat diversity and in particular the good representation of the three Annex I habitats.

3.5.2. Ecology

Tory Hill is designated as a Special Area of Conservation (SAC) and proposed as a Natural Heritage Area (pNHA) (Perrin, O'Hanrahan, & Barron, 2009). The site has been selected for a number of habitats under the E.U. Habitats Directive such as alkaline fens, Cladium fens and Orchid-rich Calcareous Grassland listed in Annex I (NWPS, 2015). The fen habitat types have a good degree of representativity whereas the Orchid-rich Calcareous Grassland is an excellent representation.

Together, the Annex I habitats cover around 9% of the total area (NWPS, 2015). Other habitats found in the site are humid grassland, deciduous woodland, heath and scrubs, dry grassland, standing water bodies and inland rock. A habitat map of the site is shown in Figure 3.25.

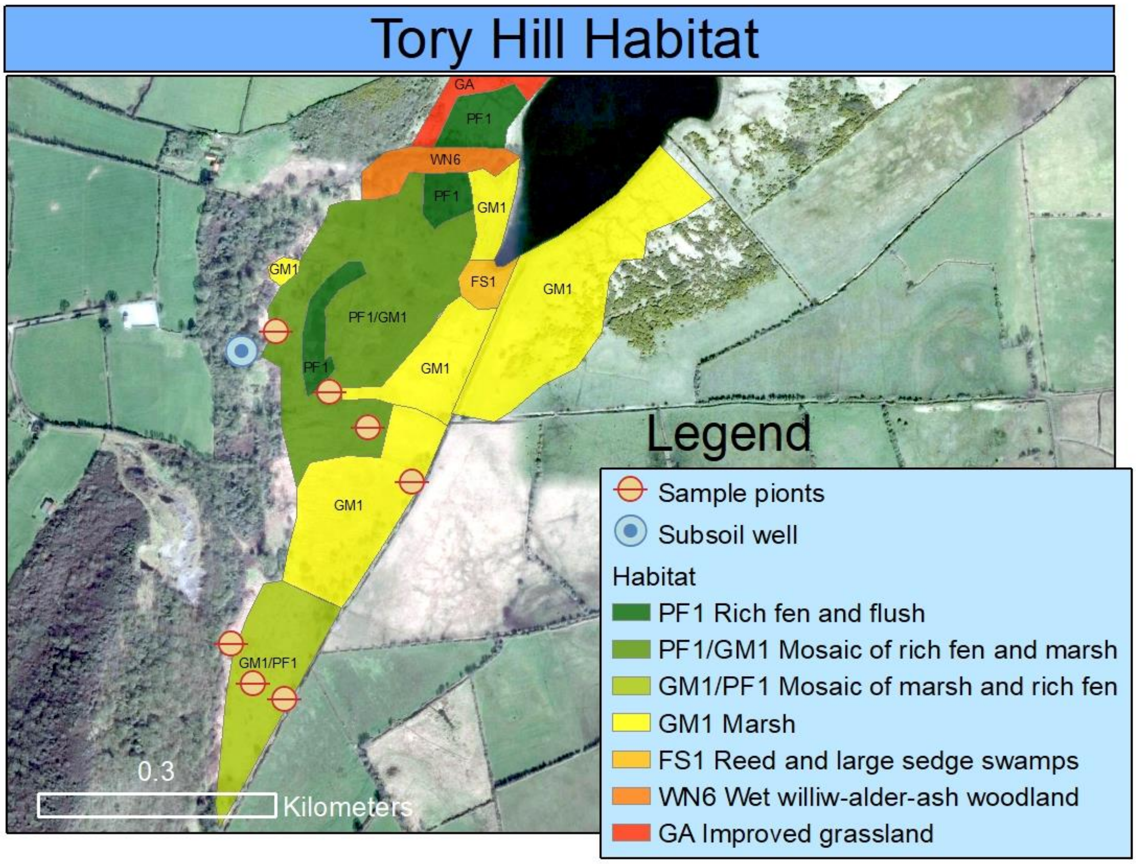


Figure 3.25. Habitat map of Tory Hill

No Annex II species were reported at the site. However, a great diversity of flora was reported on the site among which were four orchid species (*Ophrys apifera, Anacamptis pyramidalis, Orchis mascula* and *Dactylorhiza fuchsii*) found in the designated Annex I habitat. The fen habitats are representred by the great fen-sedge (*Cladium mariscus*) and small fen-sedge (*Caricion davallianae*). The overall aim of the Habitats Directive is to achieve favourable conservation status of the mentioned habitats that are of community interest (NWPS, 2016a).

3.5.3. Geology and structure

The site, which expands over an area of 76.9 ha, can be found 2 km north-east of Croom in county Limerick and includes an isolated wooded limestone hill having elevations rising to 112 m (NWPS, 2013a). A lake, Lough Nagirra, and its adjacent fen and wet grassland vegetation is located to the north and north-east of Tory Hill. The fen lies between the hill and the lake (Regan et al, 2016) (NWPS, 2013a). The bedrock on the site consist mainly of Carboniferous limestone and the soil is specified as coarse and calcareous (O'Connell, Huang, & Eicher, 1999). The shallow peat layer is underlain lake clay, separating the peat from its source groundwater body, a karstified aquifer (Regan et al, 2016). The mayor part of Tory Hill supports fen and reed swamp vegetation which was formed after the Midlandian glaciation (O'Connell, Huang, & Eicher, 1999).

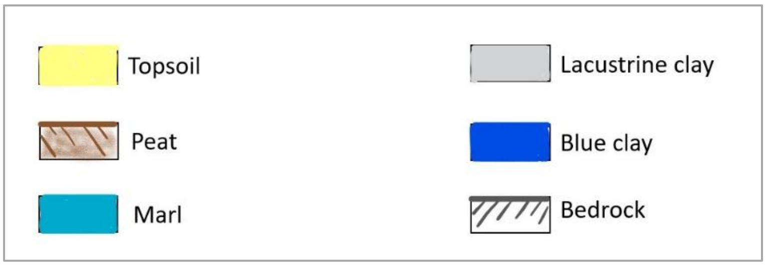


Figure 3.26. Soil type legend of Tory Hill.

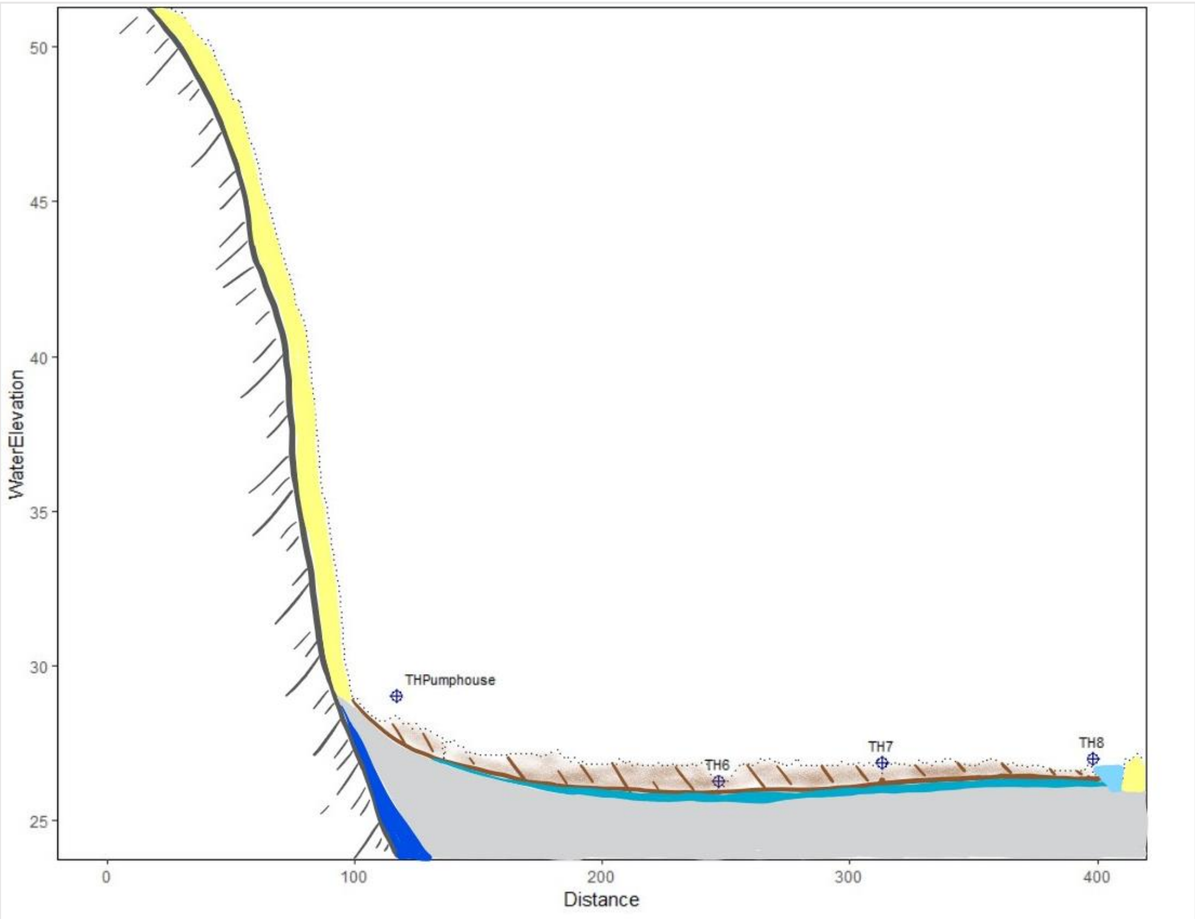


Figure 3.27. Geology transect Tory Hill transect 2. The transect location can be found in Figure 3.25 and moves from the east to the west. Figure 3.26 contains the legend which explains the soils shown in the transect.

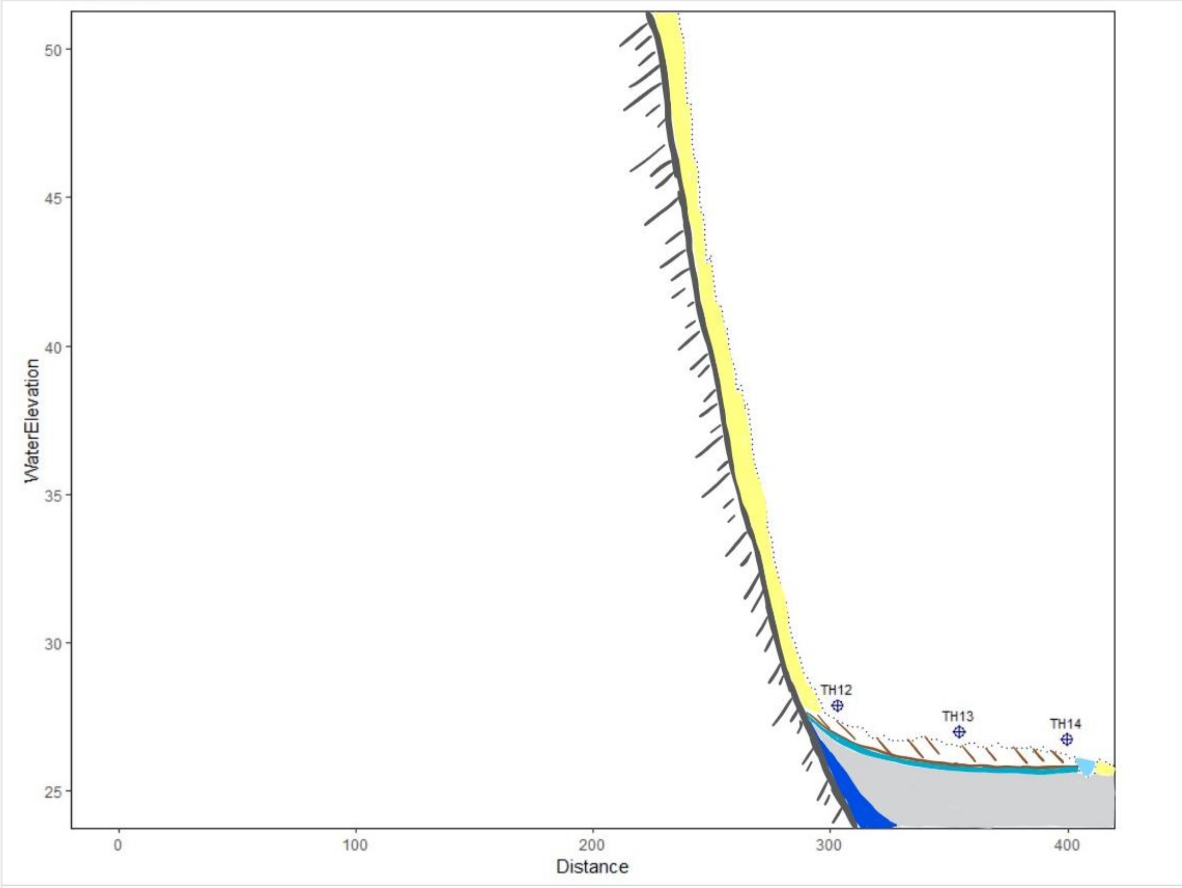


Figure 3.28. Geology transect Tory Hill transect 1. The transect location can be found in Figure 3.25 and moves from the east to the west. Figure 3.26 contains the legend which explains the soils shown in the transect.

3.5.4. Hydrogeology

Regan & Connaghan (2016) found that the main drain has a great impact on Tory Hill fen level. This was most apparent at low flow during the summer period. Groundwater levels are deep, at > 0.5m below the ground surface for most areas of the fen. This has a detrimental effect on the fen vegetation and this is apparent from the habitat survey where the general absence of a brown moss layer is coupled with low cover of the majority of the vascular indicator species, such as sedges (Carex spp.). Furthermore, daily precipitation and ET were obtained from Met Eireann station Shannon Airport.

3.5.5. Land use

A number of activities, treats and pressures have a negative impact on Tory Hill. Hydraulic conditions were altered through anthropogenic activities such as drainage works. This has a high negative impact on the site. Additional negative impact on the site is caused by infilling, drainage and non-intensive grazing (NWPS, 2015). Regan & Connaghan (2016) found that the main drain of Tory Hill has been in existence since at least the late 1830's. On the early 1900's Ordnance Survey map it is obvious that the outflow stream/drain was straightened and probably substantially deepened at some stage between the 1830's and the 1900's. After this century there has been further cleaning/deepening of the drain most recently by Office of Public Works drainage division in 1970's (Regan & Connaghan, 2016).

3.5.6. Preliminary conceptual model

Preliminary conceptual models of Tory Hill with possible negative impacts and phreatic tube and piezometer locations are presented in Figure 3.29 and 3.30 utilising the information gathered during the desk study.

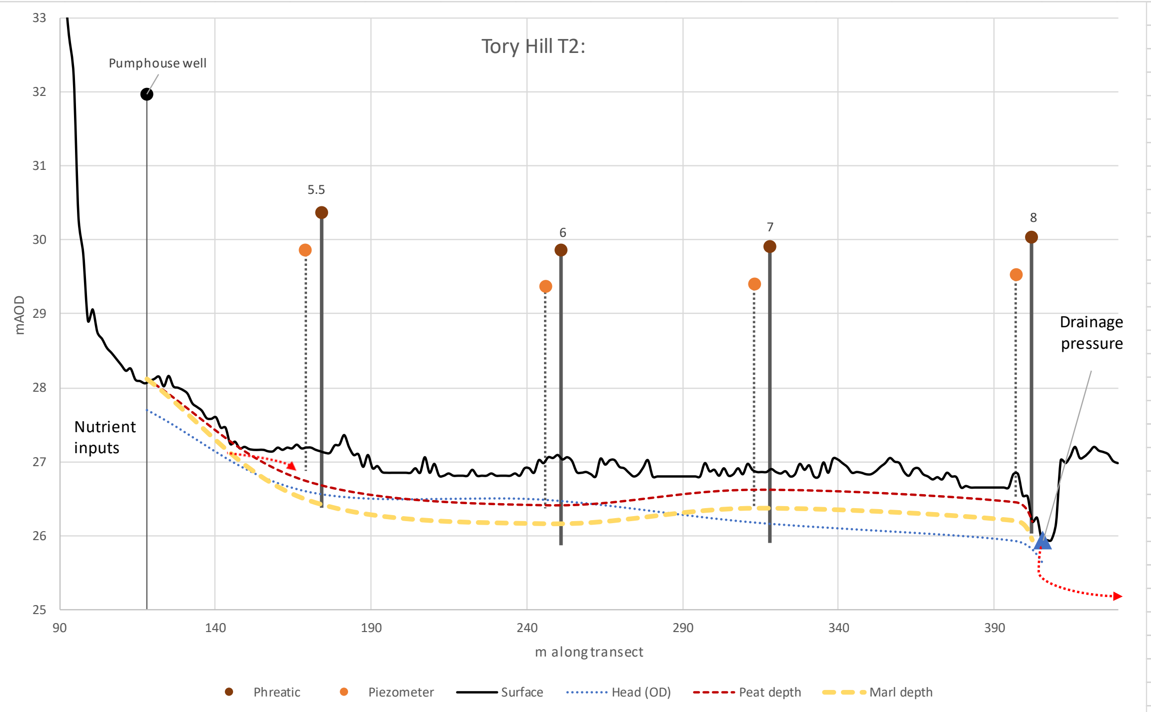


Figure 3.29. Preliminary conceptual model of Tory Hill transect 1. The transect location can be found in Figure 3.25 and moves from the east to the west. Created utilising data from Regan & Connaghan, 2016. mOAD is in meters Above Ordnance Datum (AOD).

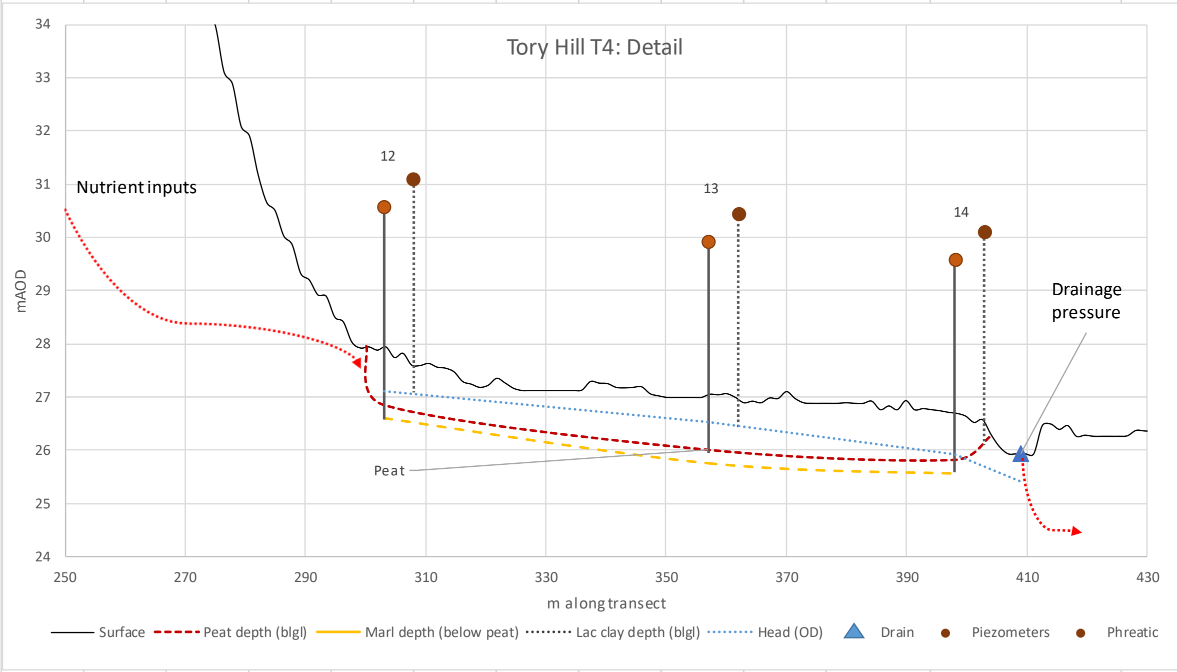


Figure 3.30. Preliminary conceptual model of Tory Hill transect 2. The transect location can be found in Figure 3.25 and moves from the east to the west. Created utilising data from Regan & Connaghan, 2016. mOAD is in meters Above Ordnance Datum (AOD).

4. Materials and Methods

4.1. Site installations

On the basis of preliminary conceptual models of each fen mentioned in Chapter 3, locations for water level measurements as well as water samples were identified. Piezometers and phreatic tubes were installed in the peat layer of the fen itself as well as the subsoil of land adjacent to the fen. In total nine transects where installed spread over four field sites resulting in 125 sample points.

4.1.1. Fen instrumentation

Installation of instrumentation in the fens was carried out in June and July 2018. Piezometers where installed at the locations depicted in Figures 4.3 to 4.7 in order to measure the groundwater level. Holes were predrilled using a hand auger with extensions up to 8 m. Holes were drilled to the depth just before the transition from peat layer to subsoil, while recording the soil logs of the different layers which are reported in Appendix A. The installation specifics of the piezometers are also reported here. Furthermore, phreatic tubes were installed at those locations at a depth of 1m in order to measure the free water table. This creates, together with the piezometers, a piezometer nest (Figure 4.1). By doing so, information can be gathered on downward or upward hydraulic gradients within the peat which will help to give subsequent insights concerning the source of typical chemical compositions of the fen pore water.

The piezometers (Stuart Well Services Ltd) were constructed out of PVC and had a filter tip of 45 cm installed. The filter tip was made of HDPE and had an engaged filter with an average pore diameter of approximately 60 microns and a permeability of approximately 3 x10-4m/sec. An example of an installed piezometer and phreatic tube can be observe in Figure 4.2. The phreatic tubes (Brooks building supplies) were made of PVC and had horizontal slots of 3 mm every 2.5 cm on the part that was installed in the fen. The part of the tube that was slotted was encased in geosock material with a mesh size of 0.5 mm to make sure no sediment would enter the tube.

A levelling survey was conducted of the ground surface as well as the top of the phreatic wells and piezometers with a Trimble® R6 GPS System (correct to 2mm on hard surfaces). Location coordinates of the piezometers and phreatic tubes as well as other information such as ground level and depth can be found in Appendix B.

4.1.2. Subsoil instrumentation

Piezometers were also installed outside of the fens down into the subsoil within the catchment / recharge area of the research sites in July 2018 with the help of Geological Survey Ireland. Their rotary drilling rig was able to reach depths up to 17 meters in the subsoil. Piezometers where installed on locations depicted in Figures 4.3 to 4.7. The soil logs of the different layers in the subsoil as well as the installation specifics of the piezometers are reported in Appendix A. The location and height of the installed subsoil piezometers were recorded during the aforementioned levelling survey. Other information such as location coordinates, depth and ground level elevation can be found in Appendix B.

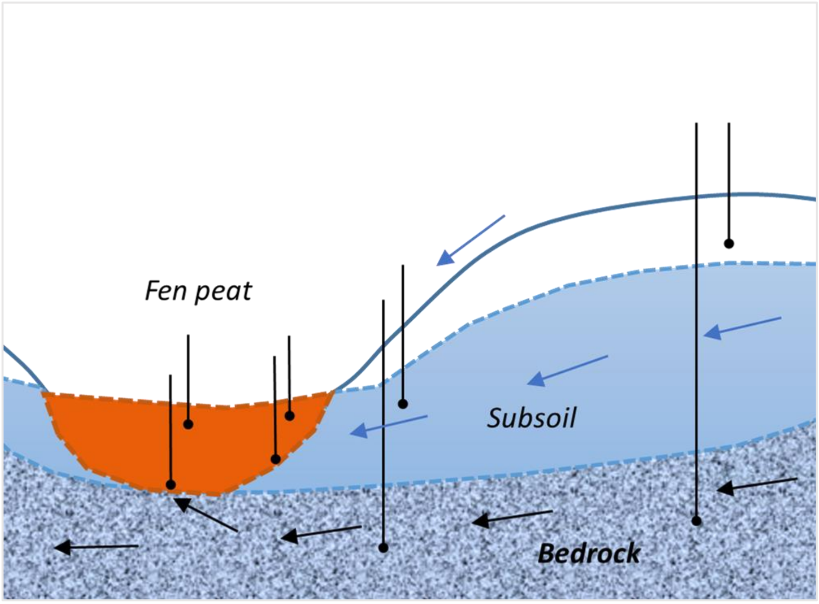


Figure 4.1. Depiction of piezometer nests in the fen and in the surrounding catchment

Figure 4.2. A phreatic tube (left) and piezometer (right) installed in Scragh Bog.

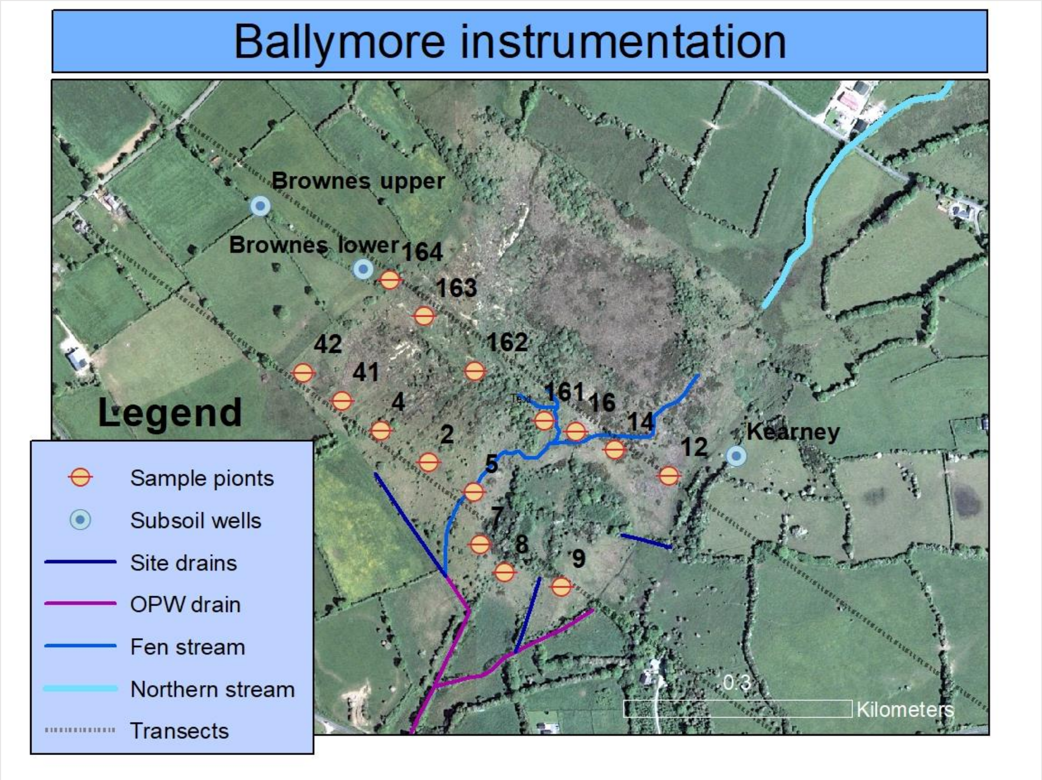


Figure 4.3. Ballymore instrumentation map showing fen piezometer and phreatic tube locations, subsoil well locations and the main site drains.

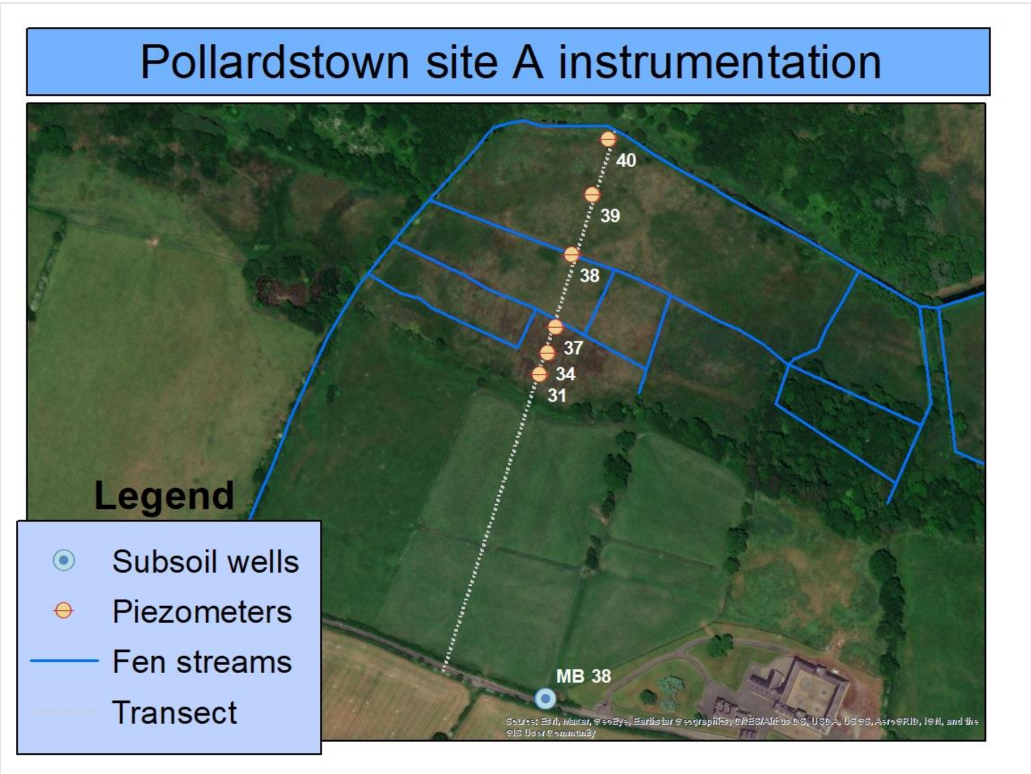


Figure 4.4. Pollardstown site A instrumentation map showing fen piezometer and phreatic tube locations, subsoil well locations and the main site drains.

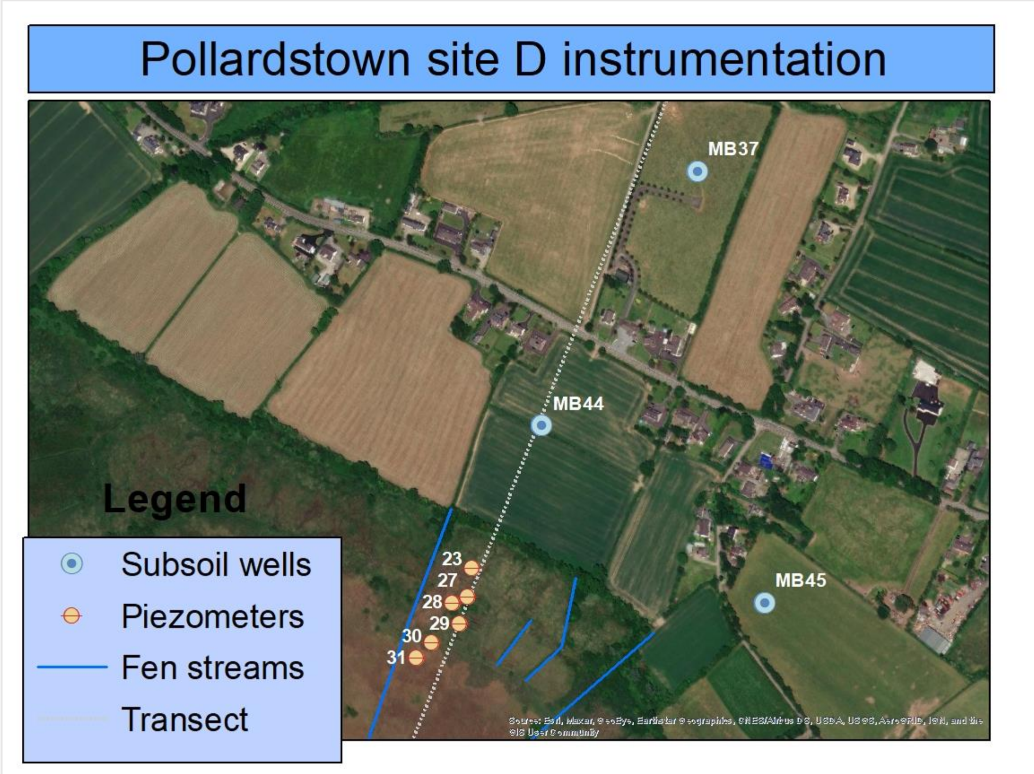


Figure 4.5. Pollardstown site D instrumentation map showing fen piezometer and phreatic tube locations, subsoil well locations and the main site drains.

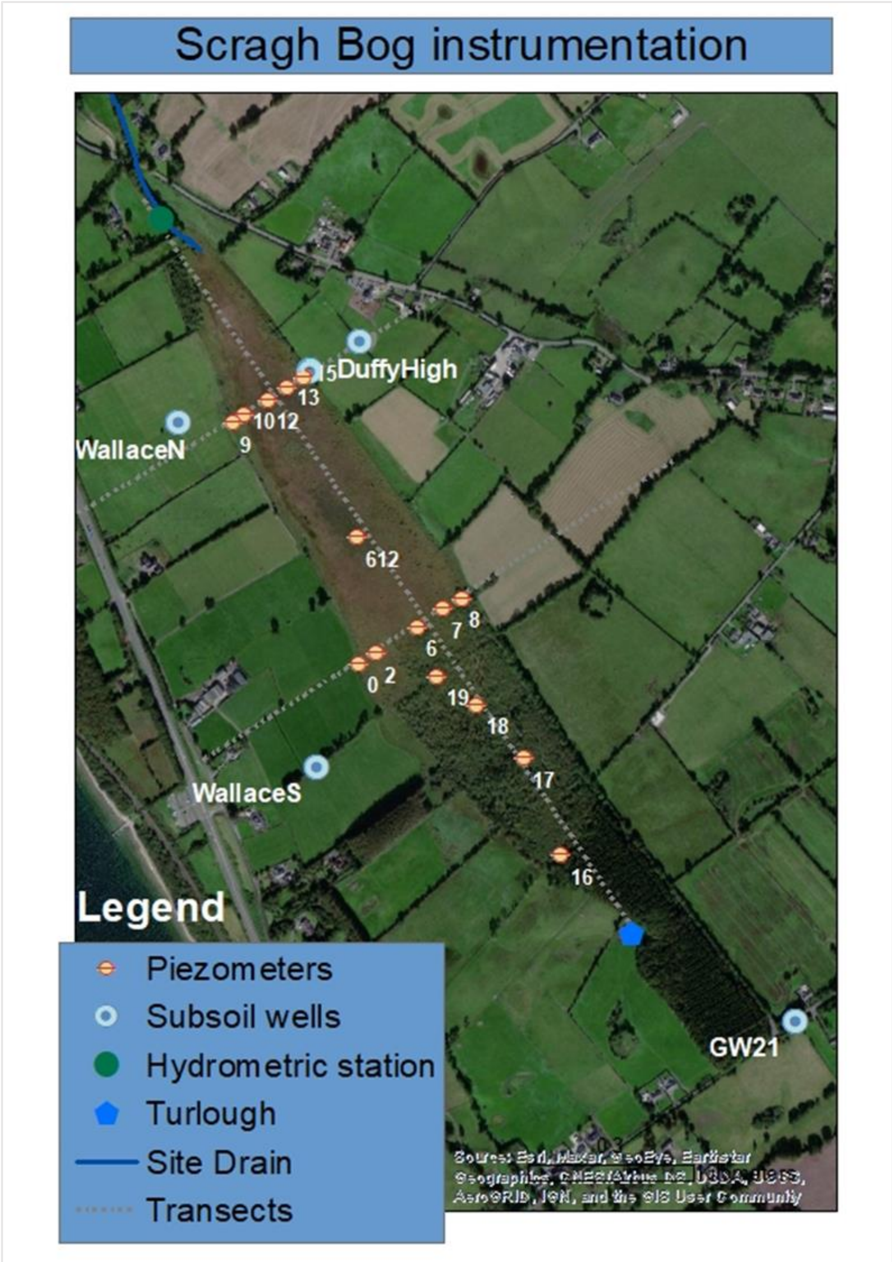


Figure 4.6. Scragh Bog instrumentation map showing fen piezometer and phreatic tube locations, subsoil well locations and the main site drains.

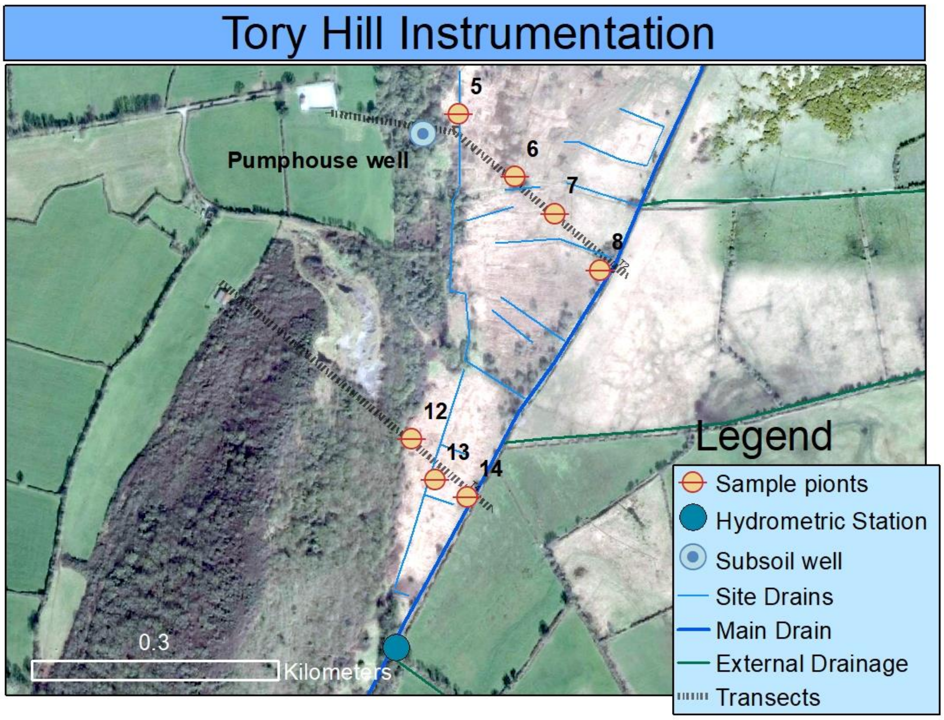


Figure 4.7. Tory Hill instrumentation map showing fen piezometer and phreatic tube locations, subsoil well locations and the main site drains.

4.1.3. Habitat surveys

Habitats were surveyed by different ecologists in order to find out what habitats were supported within the fens. The habitats of Scragh Bog and Pollardstown were surveyed by ecologists from BEC consultants in 2019. Tory Hill and Ballymore were surveyed by John Conaghan in 2015 and 2017 respectively (Regan & Conaghan, 2016) (Regan & Conaghan, 2017). Relevés with vegetation species percentages recorded next to the piezometer nest can be found in Appendix C.

Additionally BEC also drafted assessment criteria for fen habitats that aimed to categorise the fen habitats into 'good' or poor'. These criteria were used on the relevés surveyed in the research sites for the purpose of assigning environmental conditions for good quality fen vegetation. The assessment criteria and fen quality assessments can be found in Appendix D.

4.1.4. Flow measurement instrumentation

A flume was installed in the natural outlet of Ballymore (Figure 4.8), making discharge measurements more reliable. A stilling well fitted with an OTT Orpheus Mini was installed one meter upstream from the flume in October 2018, replacing the stilling well installed by Regan & Connaghan (2017). A stilling well was already in place in the outlet of Scragh Bog (Figure 4.9). Here the head is measured by the OPW. The stage data was downloaded from waterlevel.ie (OPW, 2020). Tory Hill fen had a OTT Orpheus Minis (integrated pressure sensors and data loggers for water level measurement) installed by Regan & Connaghan (2016) (Figure 4.10).The discharge of the main outlet in Pollardstown (Hanged man's Arch, see Figure 4.11) was also downloaded from waterlevel.ie (OPW, 2020).

Figure 4.8. Ballymore hydrometric station and flume.

Figure 4.9. Scragh Bog hydrometric station and staff gauge.

Figure 4.10. Tory Hill hydrometric station and staff gauge.

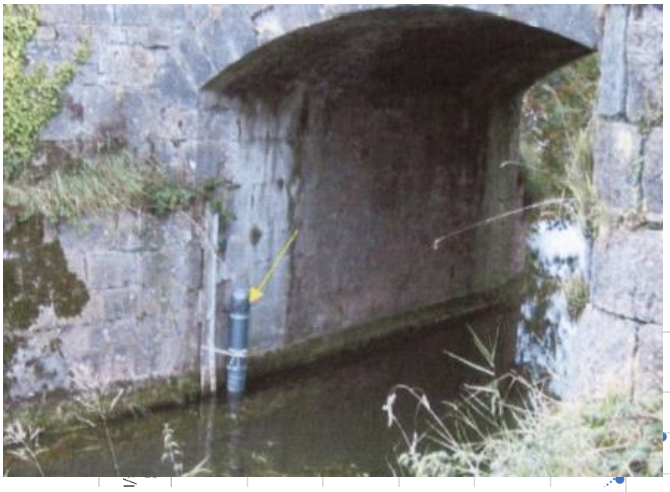
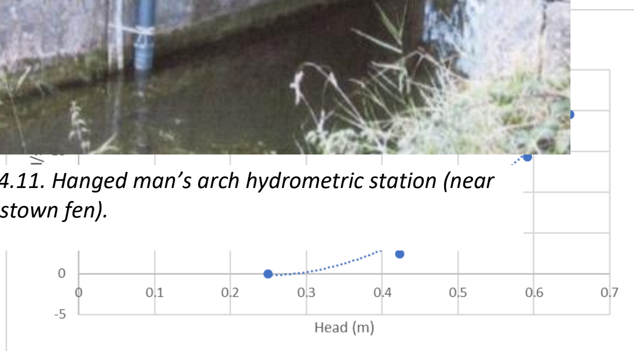


Figure 4.11. Hanged man's arch hydrometric station (near Pollardstown fen).

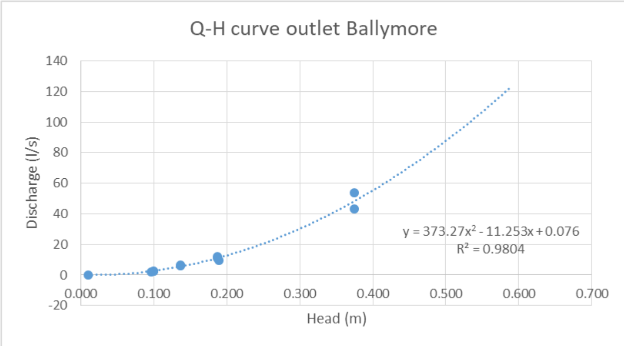


The discharge rates were calculated by performing dilution gauging at different heads. The dilution gauging is executed by adding a known quantity of a tracer (in this case salt) to a stream and observe its concentration in the stream at a point where it is fully mixed with the flow (Hudson et al., 2008). The tracer is more diluted at higher recorded heads. Therefore the higher the head, the higher the flow. The curves of the

Figure 4.12. Discharge rating of the outlet in Ballymore

Figure 4.13. Discharge rating of the outlet in Scragh Bog

discharge rates measured in the outlets of Ballymore, Scragh Bog and Tory Hill are shown in Figures 4.12, 4.13 and 4.14.



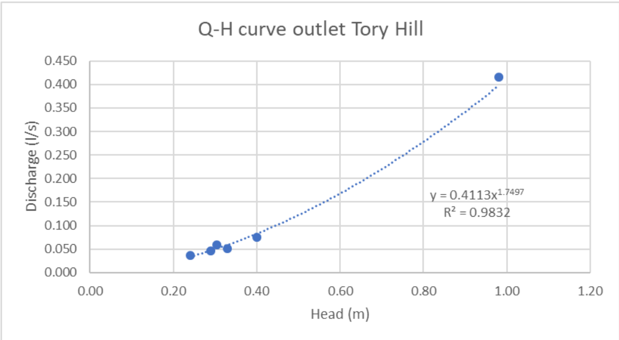


Figure 4.14. Discharge rating of the outlet in Tory Hill

4.1.5. Waterlevel logger instrumentation

A total of eight OTT Orpheus mini, two Ecologs 500, 10 TD Divers and 15 Micro Divers were installed across all of the research sites. Finally, each site was equipped with a Baro-Diver to account for barometric compensation for the pressure measured with the TD and Micro Divers. The TD and Micro divers were compensated using the software DiverOffice.

This data was subsequently used for the hydrographs and contour line maps. Hydrographs were generated using excel. Contour line maps were made in ArcGIS (ArcMap 10.8.1) with the 3D analyst tool. First, measured surface water points in the fen as well as points for the groundwater table around the fen (in boreholes or wells) were interpolated with the 'topo to raster' tool. This raster file was then changed into a contour file with an interval of 0.1 m and interpreted.

4.1.6. Meteological data instrumentation and collection

Meteological data was collected from the weather station database of Met Éireann (2020). The specific weather station data utilised in calculating the water balances in listed below:

- Pollardstown: rainfall data from Naas (Osberstown) weather station, evaporation and evapotranspiration from Mullingar weather station
- Scragh Bog and Ballymore: all data from Mullingar weather station

- Tory Hill: all data from Shannon Airport.

4.1.7. Water balance calculation

For each fen, water balances were estimated for each hydrological year (defined as beginning of October to end of September each year in the Irish climate) as follows:

$$Q = P_f + P_c - E_f - E_c \ \pm S$$

where,

Q = net flow from fen (units) as measured by continuous monitoring at the flume

P_f = rainfall directly onto fen surface

P_c = rainfall onto rest of supporting catchment area

E_f = evapotranspiration from fen

E_c = evapotranspiration from rest of supporting catchment area

S = change in water storage in fen between start and end of the year

Q was measured by continuous monitoring at the flume, as specified in Section 4.1.4. The rainfall and evapotranspiration data (PET) was collected from Met Eireann weather stations. These were used for P_f, P_c, E_c and E_f. To E_f an arbitrary crop factor of 1.2 was applied to total evapotranspiration as wetlands typically have evapotranspiration fluxes that are greater than grasslands (which is where Met Eireann corrects their measurements of actual evapotranspiration). The change in water storage (S) between the start and end of the hydrological year was calculated by taking the average surface water levels that were recorded by water level loggers in different areas throughout the fen. Furthermore, the net surface water and groundwater inputs of the fen are contained in the water balance by subtracting the evapotranspiration from the fen (P_f) and the surrounding catchment (P_c) from the precipitation on the fen (E_c) and surrounding catchment (E_f).

4.2. Water sampling

Water sampling from phreatic tubes and piezometers was conducted following the sampling procedures and protocols mentioned by the USGS (USGS, 2005) from July 2018 to February 2020. The sampling dates and days between samples are shown in Tables 4.1 and 4.2 respectively.

Table 4.1. Field sampling dates

Field sampling dates						
	Jul-18	**Aug-18**	**Sep-18**	**Oct-18**	**Nov-18**	**Dec-18**
Ballymore	**19/07/2018**	**30/08/2018**	**27/09/2018**	**25/10/2018**	**25/11/2018**	
Pollardstown	18/07/2018	21/08/2018	03/10/2018	31/10/2018	21/11/2018	
Scragh Bog	24/07/2018	29/08/2018	26/09/2018	24/10/2018	26/11/2018	
Tory Hill	26/07/2018	22/08/2018	02/10/2018	30/10/2018	20/11/2018	
	Jan-19	**Feb-19**	**Mar-19**	**Apr-19**	**May-19**	**Jun-19**
Ballymore	27/01/2019			01/04/2019		10/06/2019
Pollardstown	29/01/2019			15/04/2019		16/06/2019
Scragh Bog	23/01/2019		25/03/2019			03/06/2019
Tory Hill	22/01/2019			09/04/2019		17/06/2019
	Jul-19	**Aug-19**	**Sep-19**	**Oct-19**	**Nov-19**	**Dec-19**
Ballymore		13/08/2019		15/10/2019		03/12/2019
Pollardstown		20/08/2019		22/10/2019		16/12/2019
Scragh Bog		05/08/2019		07/10/2019		09/12/2019
Tory Hill		21/08/2019		23/10/2019		17/12/2019

Table 4.2. Time between sampling dates

Days between samplings	Jul-Aug 2018	Aug-Sep 2018	Sep-Oct 2018	Oct-Nov 2018	Nov 2018-Jan 2019	Jan-Apr 2019	Apr-June 2019	Jun-Aug 2019	Aug-Oct 2019	Oct-Dec 2019
Ballymore	42	28	28	31	63	64	70	64	63	49
Pollardstown	34	43	28	21	69	76	62	65	63	55
Scragh Bog	36	28	28	33	58	61	70	63	63	63
Tory Hill	27	41	28	21	63	77	69	65	63	55

Prior to sampling the water level is measured and recorded using a Van Walt V025 dip meter with a range of 0-60 meters. Then water samples are taken from the well, piezometer or phreatic tube that as a representative sample of the underlying groundwater in that area.

To ensure this is achieved each well is purged prior to sampling. This is done in order to remove standing water from the sample points thus making sure that the collected sample is ”fresh” groundwater instead of stagnant water that may have been remained within the sampler for a long period which might compromise the water quality results. The (USGS, 2005) sampling procedure recommends purging three or more well volumes. However, in this study as piezometers are installed in the peat layer, consideration has to be given to instances where the peat pore water is only slowly percolating into the piezometer, since active peat is known to slowly release water. Therefore, at least one well volume was purged from such sampling locations and more when possible. Purging was done fifteen minutes before sampling and sometimes longer as the fens substrate was less permeable.

After purging, the sample was taken using a CV 12 mm OD bailer and was measured for pH, oxygen (mg/l), temperature (C°) and electrical conductivity. Samples to be used for parameters except total phosphorus (Figure 4.3) were filtered with a 0.45 µm Minisart filter into a 30 ml sterile plastic sample bottles. Samples to be used for phosphate analysis were filtered 15 minutes after collection (USGS, 2005). The unfiltered sample collected for total phosphorus was stored into sterilised 100 ml amber glass bottles which were also used as a backup for additional analysis when needed. Some of the sampling materials are shown in Figure 4.15.

Figure 4.15. Water sampling and filtering in Pollardstown fen.

Samples were stored in a icebox keeping the samples in the dark and cool (4˚C) as much as possible during the transfer to the lab. Replicate (duplicate) samples and transport/trip blanks are used as quality control (QC) measures to ensure reliability of the sampling and laboratory protocols.

4.3. Laboratory analysis

Eighteen water quality parameters were analysed from the water samples collected. Every parameter except for Total Phosphorus was measured in its dissolved form. The instruments and corresponding methods are discussed in the following sections. A summary of the water quality analysis is given in Table 4.3.

Table 4.3. Summary water quality analysis

Parameter	Measured form	Soluble ?	Analysis Form	Instruments	Additional steps
Sodium	Na	Yes	Atomic emission spectroscopy	ICP-EAS	
Magnesium	Mg	Yes	Atomic emission spectroscopy	ICP-EAS	
Calcium	Ca	Yes	Atomic emission spectroscopy	ICP-EAS	
Potassium	K	Yes	Atomic emission spectroscopy	ICP-EAS	
Manganese	Mn	Yes	Atomic emission spectroscopy	ICP-EAS	
Iron	Fe	Yes	Atomic emission spectroscopy	ICP-EAS	
Ferrous Iron	Fe	Yes	Spectrophotometric	Konelab	
Silica	Si	Yes	Spectrophotometric	Konelab	
Chloride	CL_2 - Cl	Yes	Spectrophotometric	Konelab	
Sulphate	as SO_4	Yes	Spectrophotometric	Konelab	
Ammonium	NH_4+ as N	Yes	Spectrophotometric	Konelab	
Nitrite	NO_2- as N	Yes	Spectrophotometric	Konelab	
Total Oxidised Nitrogen	TOxN as N	Yes	Spectrophotometric	Konelab	
Nitrate	NO_3- as N	Yes	Spectrophotometric	Konelab (indirect: TON - NO_2 = NO_3)	
Alkalinity	$CaCO_3$-	Yes	Spectrophotometric	Konelab	
Dissolved Reactive Phosphorus	$PO_4$3- as P	Yes	Spectrophotometric	Konelab + Hach	
Total Phosphorus	as P	No	Spectrophotometric	Hach	Digestion
Total Nitrogen	as N	No	Catalytic thermal decomposition/chemiluminescence	TOC-L	Digestion

4.3.1. Spectrophotometry

4.3.1.1. Hach spectrophotometer

In order to reach the lowest possible method limit detections for dissolved reactive phosphorus (mg/l as P) and total phosphorus (mg/P) analysis for these parameters was conducted in the laboratory of the Centre of Environment using a Hach spectrophotometer (TYPE). For this analysis a Molybdate blue reagent as well as standards were made up freshly every time and the spectrophotometer is calibrated before each use. Then the analysis is conducted using a 5 cm cell providing very sensitive detection. Furthermore, the analysis was done in triplicate according to the protocol minimising the chance of unreliable results. The expected MLD for both DRP and TP are calculated to be approximately 1 ug/l. Samples were analysed within 48 hours after collection as is specified by (APHA, 1998).

4.3.1.2. Lachat flow injection analysis

Flow injection analysis, or FIA, is a continuous flow method for rapidly processing samples (Hach Company, 2009). The peristaltic pump draws sample from the sampler into the injection valve. Simultaneously, reagents are continuously pumped through the system. The sample is loaded into the sample loop of one or more injection valves. The sample and reagents then merge in the manifold (reaction module) where the sample can be diluted, dialyzed, extracted, incubated and derivatized. Mixing occurs in the narrow bore tubing under laminar flow conditions. For each method, the operating parameters are optimized to address high sample throughput, high precision and high accuracy.

The Lachat 3-channel flow injection analyser can analyse ammonia and total oxidised nitrogen simultaneously. Nitrite has to be analysed on a separate run. Reagents are made up fresh from stock solutions and the instrument is calibrated with every run. This was done according to the method described in APHA (1996). The analysis was done in duplicate minimizing the chance of unreliable results. The MLDs for these are calculated to be approximately 1 µg/l as limit for both NH3-N, TON-N and NO2-N, but it is advised to report anything below 5 µg/l as < 5µg/l. Samples were also analysed within 48 hours after collection as is specified by (APHA, 1998).

4.3.1.3. Konelab

Nitrite (mg/l N), sulphate (mg/l SO4), chloride (mg/l Cl), ferrous Iron (mg/l Fe), silica (mg/l SiO2) and alkalinity (mg/l CaCO3) were analysed using an automated spectrophotometric instrument: a Konelab 20i (Thermo Scientific, 2003) in the TCD environmental engineering lab. This spectrophotometric analysis has much less sampling handling compared to the method described in 4.3.1.1. and 4.3.1.2. The system has software with features supporting different applications such as pre-dilution of a sample, or in the case of a re-run sample, post-dilution in which high and low secondary dilution are handled automatically. When performing a calibration for certain parameters, a series of calibrator samples can be diluted automatically. Hence, this instrument saves a lot of time compared to what would be an otherwise time consuming analysis using a spectrophotometer with a manual method.

Precision Reagents and Standards (Thermo Scientific, 2015) are manufactured under ISO 9001 : 2008 and are in compliance with the ISO 15923-1 standard for the determination of water pollutants using an automated photometric procedure.

Two millilitres was extracted from the samples and placed into wells that were then mixed with the appropriate volume of reagent, incubated and measured upon entering the Konelab. Samples were analysed at least 48 hours after collection as specified by (APHA, 1998) for elements such as ammonia, nitrate and ferrous iron (Fe^{2+}).

4.3.1.4. Shimadzhu TOC-L.

A part of each sample was acidified to a pH below 2 with 2M hydrochloric acid. These samples were then analysed for total dissolved nitrogen and dissolved organic carbon using the Shimadzhu TOC-L. This instrument contains an automated TN analyser (TN measurement unit TNM-1). It measured Total Nitrogen through catalytic thermal decomposition/chemiluminescence 720 °C. The TOC-L has a lower detection limit of 4 µg/L and an automatic dilution function which enables measurements up to 30,000 mg/L.

4.4. Data processing

4.4.1. Limit of detection

Before any of the statistical tests were executed, the hydrochemical data was transformed to calculated limit of detection. For chemical analysis conducted with the Konelab the method in

Table 1 is used. This method is also used by Precision Serosep (Thermo Scientific, 2015) to measure the method limit detection of the reagents they supply for use with the Konelab.

The same reagents were used in the laboratory to measure surface water samples, however due to different environments the instruments may be in at the time of measuring limit detections samples a separate test was conducted for samples analysed in the Environmental Engineering laboratory. For this so called blank samples were run; distilled samples. From these samples the standard deviation as well as the average were used to calculate method limit detection (MLD) as seen in Table 4.4.

Samples that were analysed in the Laboratory of Environmental Sciences use the limit of detection (LOD) in the reported test standard. This test standard was also checked during sample analysis. The limit of detection for these methods can be found in Table 4.5.

Table 4.4. Method limit detection calculated on tests performed on the Konelab and Shimadzu TOC-L. The same method was as Thermo Scientific (2015) from the factsheet on the calculating the MLD on their reagents.

Method limit detection			
		From Thermo Scientific Factsheet (mg/l)	**Measured from Konelab and TOC-L blanks (mg/l)**
Parameter	Unit measured	MDL = 3.14 x SD (blank sample, n = 7)	*MDL = 3.14 x SD (blank sample, n = 30)
Dissolved reactive phosphorus	as P	0.0004	0.05
Total ammonia	as N	0.0005	0.093
Total oxidised Nitrogen	as N	0.0006	0.054
Nitrite	as N	0.0004	0.066
Alkalinity	as CaCO3	3.4	1.981
Chloride	as Cl	0.035	0.12
Ferrous Iron	as Fe2+	0.260	0.009
Silica	as SiO2	0.01	0.055
Sulphate	as SO4	0.26	2.134
Total dissolved nitrogen	as N		0.089
Dissolved organic carbon	as DOC		1.349

Once the limit of detection is specified, calculations could be made (as specified by Verbovsek (2011)) to limit the amount of hydrochemistry data that falls below the LOD and would be deemed unusable. Verbovsek, (2011) specifies that a substitution method can be used whereby LODV2 is calculated. This method proved to produce the smallest amount of error using censored geochemical data. This substitution was also done on the fen dataset with the outcome presented in Table 2.

Any data that fell below this calculation was transformed to the outcome of LOD√2 as to still be able to use that data in statistical analysis.

Table 4.5. Limit of detection applied on dataset – Wattslab analysis

LOD - Wattslab		
Parameter	**LOD (mg/l)**	**LOD/√2 (mg/l)**
Nitrite	0.0659	0.0466
Total dissolved nitrogen	0.0890	1.4004
Alkalinity	1.9805	0.0852
Chloride	0.1204	0.9539
Dissolved organic carbon	1.3490	0.0061
Ferrous iron	0.0086	0.0390
Silica	0.0552	1.5088
Sulphate	2.1337	0.0352

Table 4.6. Limit of detection applied on dataset – Centre of environment analysis

LOD - Centre of environment		
Parameter	**LOD (mg/l)**	**LOD/√2 (mg/l)**
Dissolved reactive phosphorus	0.0010	0.0007
Total phosphorus	0.0010	0.0007
Total ammonia	0.0050	0.0035
Total oxidised nitrogen	0.0050	0.0035

4.4.2. Statistical analysis

Different forms of analysis were performed using statistical packages in R-Studio. The codes used to execute these tests on the data frame of this research can be found in Appendix E. This program was used especially for the transect plots described in Section 4.4.2.3 since the figures were build up from scratch and ggplot (a plotting program in R) was easiest to use for this purpose.

4.4.2.1. Boxplots

For Spring/Summer boxplots was data gathered between April 1th and September 30th and Autumn/Winter boxplots was gathered between October 1th and March 31th. This division is based on the hydrological start and end of the year mentioned throughout the results chapters.

Statistical analysis

A Welch t-test was used to calculate significant differences between data groups of the hydrochemistry and hydrology between different mediums (such as comparing between sites and different sampled objects). This test was chosen since the data frame had unequal variances between groups. Furthermore, the statistical package in R-studio estimates the degrees of freedom based on the data group inputs. It also calculates the confidence interval of the mean at 95%. The significance was determined from p-values associated with the relationships. A p value < 0.05 indicates a significant difference between groups. The test was either performed on the hypothesis that two data groups are significantly different (two-sided) or that the first data group is either significantly greater or less than the second data group.

4.4.2.2. Nonmetric Multidimensional Scaling

A Nonmetric Multidimensional Scaling ordination (NMDS) using ecological and hydrochemical data was plotted in order to find environmental vectors that have some form of correlation with the sampled locations and their specific habitat including the vegetation species that were surveyed. For this two data sets were used. The IVC data set contained the recorded species percentage at each surveyed relevé. The environmental set (ENV) consisted out of vegetation type cover (%), Fossitt habitat codes and the hydrochemistry results (mg/l).

The first NMDS plot was generated with the environmental variables vegetation cover (%) and the presence of the Fossitt habitats in a biplot. They are displayed as vectors (max p-value = 0.2) and are plotted on top of a scatterplot with the surveyed relevés and species with the highest abundances (10%). The data is plotted with 100 randomised runs using the Bray distance. Similar approaches of NMDS were performed by Ahmad et al. (2020) for fens and Waldren (2015) for turloughs.

Another NMDS was run with hydrochemistry results from April and June 2019 as environmental variables since this time is deemed to be the growing season of the fen, the presumption being that the nutrients and minerals present during this season would have the most impact on the fen vegetation.

4.4.2.3. Transect plots

R studio was also used to build figures with the piezometer nest transects shown on top of the geology. This base image was then used to display hydrology and hydrochemistry data.

5. Results – Ballymore

5.1. Hydrology

5.1.1. Annual water balance

The size of the topographical catchment area of Ballymore, which was determined by drawing a polygon over the highest points surrounding the fen, was measured to be 0.88 km^2 (Figure 5.1). The fen area is 0.23 km^2.

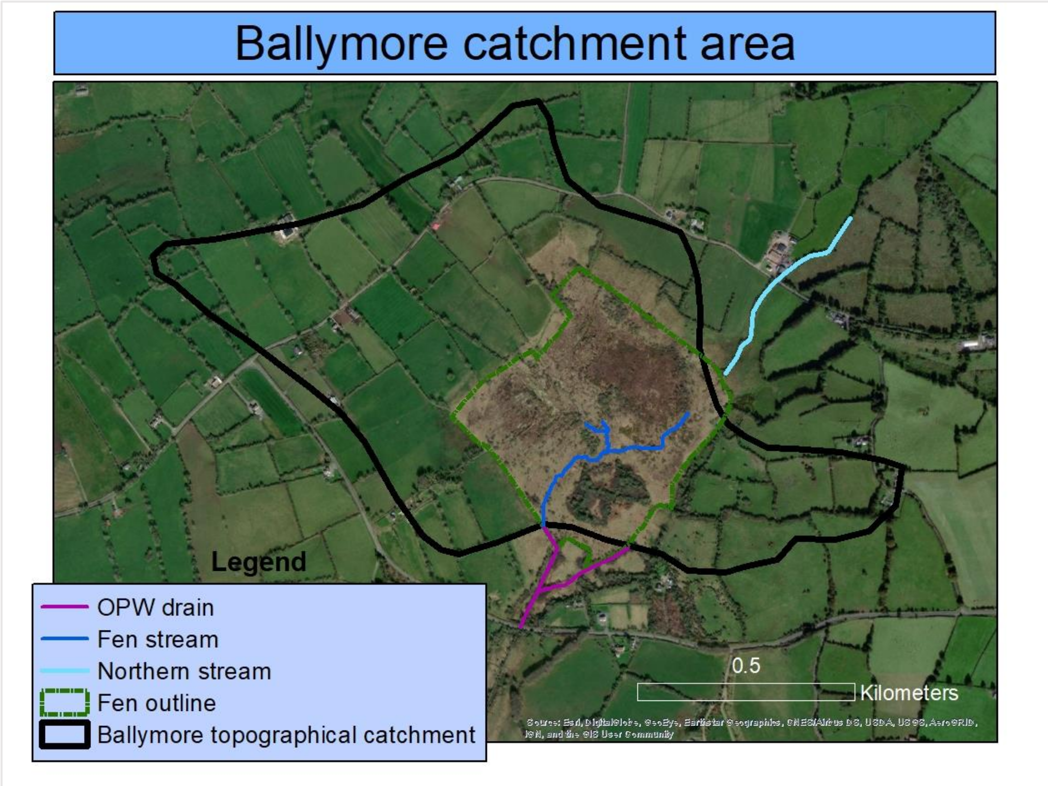


Figure 5.1. Topographic catchment area of Ballymore (Adapted from Regan & Connaghan (2017)

For each fen, water balances were estimated across each hydrological year (defined as beginning of October to end of September each year in the Irish climate) as specified in the methodology Section 4.1.7.

To determine the change in storage at the beginning and end of each hydrological year the water level change was calculated from phreatic tubes by taking the mean water level difference measured by divers installed across the fen (Figure 5.8). From this a phreatic (or open) water level increase was observed of 0.15 m across hydrological year 2018/19 compared to the 2019/20 hydrological year which saw a small decrease of 0.05 m. This data was then incorporated into the water balance as a change in storage volume by multiplying the water level change by the fen area.

The resultant water balances between the two hydrological years (2018/19 and 2019/20) required slightly different catchment areas in order to close the respective balances (assuming that the catchment area was the parameter with the most uncertainty in the balance). The total catchment area over the period of a hydrological year was adjusted such that the water balance figure is zero. When this was done the apparent optimal catchment area for hydrological year 2018-2019 was 0.80 km^2 whereas the optimal catchment area for hydrological year 2019/20 was 1.17 km^2.
The catchment area needed to close the water balance over the two full hydrological years is 0.99 km^2 which is used for the overall water balance calculations in Table 5.1. It should be noted that the calculated catchment area of the site is somewhat larger than the topographical catchment area of 0.88 km^2.

Table 5.1. Water balance of a hydrological years 2018/19 and 2019/20 in Ballymore fen using catchment area 0.99 km^2.

01-10-2018 to 30-09-2019			
Fen water level change: + 0.15 m			
	Total (m^3)	**Flux (mm/d)**	**Fraction of rainfall**
Rainfall on fen	239969	2.84	
Rainfall on catchment	788047	2.84	
Evapotranspiration from fen	142983	1.70	13.9%
Evapotranspiration from catchment	391291	1.41	38.1%
Runoff from fen	360684	1.02	35.1%
Change in fen storage	34664	0.42	3.4%
Error in water balance	*-98393*	*-0.27*	***-9.6%***
01-10-2019 to 30-09-2020			
Fen water level change: - 0.05 m			
	Total (m^3)	**Flux (mm/d)**	**Fraction of rainfall**
Rainfall on fen	247364	2.92	
Rainfall on catchment	812332	2.92	
Evapotranspiration from fen	140376	1.66	13.2%
Evapotranspiration from catchment	384158	1.38	36.3%
Runoff from fen	648234	1.79	61.2%
Change in fen storage	-11555	0.14	-1.1%
Error in water balance	*101517*	*0.28*	***+9.6%***

It is estimated that of the recorded rainfall between October 1st 2018 and September 30th 2019 approximately 13.9% was lost as evapotranspiration directly from the fen and 38.1% was lost as evapotranspiration from the surrounding catchment. Furthermore, 35.1 % was lost as run off (discharge) via the fen's main outlet (shown in Figure 5.1). For the following year it was estimated that 13.2% and 30.3% was lost as evapotranspiration from the fen and surrounding catchment, respectively. However, a 65.1% was left the fen by surface discharge at the outlet this year which is almost twice as much as the previous year. The calculated change in volume does not affect the overall water balances by much as they account for only 3.4% lost from rainfall in the first and 1.1 % added to the rainfall in the second hydrological year.

Figure 5.2 gives a more schematic overview of the water balance during hydrological years 2018/19 and 2019/20. As shown in Table 5.1 the total quantity of rainfall that fell on the fen and its catchment was roughly the same over both two years. This is also true for the evapotranspiration from the fen and its catchment. However, the runoff from the fen seems to change a lot between years. The fen loses only 360684 m^3 during hydrological year 2018/19 whereas this number increases to 648234 m^3 in the following year.

In order to maintain the water levels measured in the phreatic tubes during those years the fen had presumably lost 100000 m^3 in the first year and gained that same volume back the next hydrological year. It is very likely that these apparent losses and gains are due to the following mechanism; there exist a certain lag between the amount of water that enters the surrounding catchment by precipitation and the amount of water that the fen receives from the surrounding catchment by sub surface flow. This is displayed in Figure 5.3 where a considerable amount of effective rainfall was recorded on the 22nd of September 2019 of which it is likely that a significant amount of water enters the fen after the 1st of October 2019 and is therefore calculated in the water balance of 2019/20. It is therefore hypothesised that the water lost from the fen to the regional groundwater table during hydrological year 2018/19 is due to the lag of water entering the fen via subsurface flow from the surrounding catchment which then shows up as water gained in hydrological year 2019/20.

However, this doesn't take away the fact that an exchange of water to and from the regional groundwater table is impossible since Ballymore fen is underlain by Waulsortian limestone characterised as Locally Important aquifer (Ll vulnerability class) of bedrock that is moderately productive only in local zones. This means that the fen might be able to recharge to the groundwater and in turn be recharged quite rapidly. This is further indicated by GSI vulnerability mapping for the Ballymore fen catchment, which maps the area ranging between High and Extreme vulnerability, indicating shallow subsoils of relatively high permeability. It has to be noted however, that this additional explanation is only of minor importance in water gains and losses as only a small amount is expected to be exchanged between the bedrock and the sediments in the fen as a low permeable clay layer is under laying the peat layer. Unfortunately, it was not possible to determine what proportion of water losses and gains between hydrological years is due to either catchment lag or exchange between the fen and the regional groundwater table with the type of data collected in this study.

Nevertheless, finalized water balances revealed the degree to which the groundwater catchment area could fluctuate from year to year in Ballymore. They further give good insight of the water fluxes over a hydrological year which is important for the fen's management as it shows what quantities of water entering and leaving the site are typical in order to support the site specific fen vegetation.

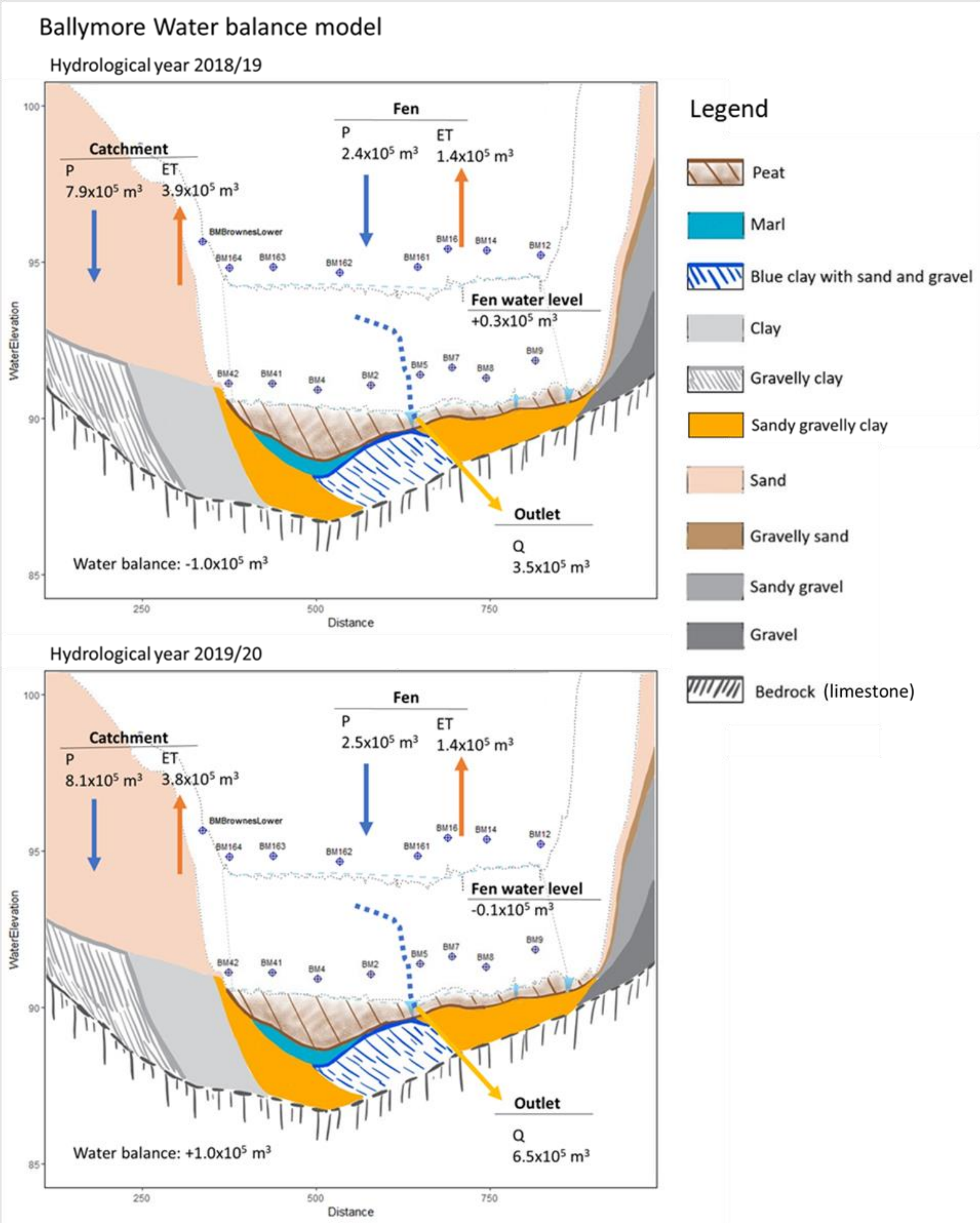


Figure 5.2. Water balance model of hydrological year 2018/19 and 2019/20 in Ballymore

5.1.2. Seasonal water balance

5.1.2.1. Hydrological year 2018-2019

As seen in Table 5.2 the fen loses 111633 m^3 which equates to about 22.8% of the total rainfall from groundwater as well as other surface losses. During the summer of 2018 Ireland experienced a significant drought (Met Éireann, 2020) therefore decreasing aquifer levels in the catchment area of the fen significantly. While the dry months were observed from May to July some of the after effects were still found in subsoil well BRL (with screen depth of 4 mBGL) in October (Figure 5.14). Because of this it is hypothesised that the fen then reacted accordingly by slowly losing water to lowering groundwater level in the surrounding catchment. This process was, however, attenuated because water is held for a much longer time in fen peat due to capillary forces than in the soil in fields surrounding the fen. This may create a delay in the fen groundwater levels dropping after a sustained drought, which is apparent in the winter water balance of 2018/2019. The fen will keep losing water until it is in equilibrium with its surrounding catchment, although it should be noted that this process is transient with changing hydrological inputs and outputs. The amount of shrinkage in a fen depends on the duration of the drought, the size and depth of the fen (and therefore the amount of peat that can store water), the maturity of humification in the peat (which influences the capillary processes) and the geomorphology of the catchment. When the drought has passed and aquifers start to fill up once more the fen hypothetically reacts again by swelling up from the water it receives from the catchment. Surface oscillation in bogs caused by the expansion of the peat body has been observed in several bogs during the wet season (Howie & Hebda, 2018).

This mechanism can be seen in the summer of hydrological year 2018/19 and throughout the hydrological year 2019/20 (see Table 5.3). In the winter of hydrological year 2018/19 (Table 5.2) water levels rose by 0.12 m throughout the season even though the fen lost 22.8% of the rainfall to the regional groundwater table. However, this apparent loss may also largely be explained by the lag of water entering the fen via subsurface flow from the surrounding catchment which is again visible in Figure 5.3 where highly effective rainfall is recorded through much of March 2019, meaning a high portion of water reached the fen during the summer water balance of 2018/19.

The fen also saw a big portion lost via runoff, indeed, 65.2% of the annual runoff was lost during this season. The water level rise in Ballymore fen was equivalent to a storage of 27731 m^3 of water during that time. The hydrological summer saw another small water level rise of 0.03 m which accounted for an additional storage of 6933 m^3. Since the aquifer had recovered during this time, the fen gained 2.5% of its water balance from groundwater.

Table 5.2. Seasonal water balances of hydrological year 2018/19 in Ballymore

01-10-2018 to 31-03-2019 (Winter)			
Fen water level change: + 0.12 m			
	Total (m^3)	**Flux (mm/d)**	**Fraction of rainfall**
Rainfall on fen	114461	2.74	
Rainfall on catchment	375886	2.74	
Evapotranspiration from fen	30921	0.74	6.3%
Evapotranspiration from catchment	84618	0.62	5.7%
Runoff	235444	1.31	48.0%
Change in fen storage	27731	0.66	5.7%
Error in water balance	*-111633*	*-0.62*	***-22.8%***
01-04-2019 to 30-09-2019 (Summer)			
Fen water level change: + 0.03 m			
	Total (m^3)	**Flux (mm/d)**	**Fraction of rainfall**
Rainfall on fen	125508	2.98	
Rainfall on catchment	412161	2.98	
Evapotranspiration from fen	112063	2.66	20.8%
Evapotranspiration from catchment	306674	2.22	57.0%
Runoff	125240	0.70	23.3%
Change in fen storage	6933	0.17	1.3%
Error in water balance	*13240*	*0.07*	***+2.5%***

5.1.2.2. Hydrological year 2019-2020

As seen in Table 5.3, according to the water balance the fen apparently lost 51710 m^3 during the winter and gained 153227 m^3 during the summer of the hydrological year 2019/20 from the groundwater. Again, this is largely due to the lag of water entering the fen from the subsurface of the surrounding catchment as high values of effective rainfall are recorded throughout March 2019.

However an exchange of water between the fen and the regional groundwater table may also partly be the reason for these gains and losses by the following reaction; when aquifer levels were once restored after the drought during the summer months of 2018 it is hypothesised that the fen reacts to this by swelling up from the water it receives from the catchment. The fen keeps swelling in response to rising levels in the aquifer until the fen peat is saturated. After this point has been reached the fen shows a higher drainage rate from its outlet, which can be seen from the runoff of the hydrological year 2019/20 especially during the winter months. Approximately 71.8 % of the annual runoff was lost during this season, which is similar to the previous hydrological year.

Table 5.3. Seasonal water balances of hydrological year 2019/20 in Ballymore

01-10-2019 to 31-03-2020 (Winter)			
Fen water level change: - 0.08 m			
	Total (m^3)	**Flux (mm/d)**	**Fraction of rainfall**
Rainfall on fen	141707	3.37	
Rainfall on catchment	465361	3.37	
Evapotranspiration from fen	29090	0.69	4.8%
Evapotranspiration from catchment	79609	0.58	13.1%
Runoff	465146	2.58	76.6%
Change in fen storage	-18488	-0.44	-3.0%
Error in water balance	*-51710*	*-0.29*	***-8.5%***
01-04-2020 to 30-09-2020 (Summer)			
Fen water level change: + 0.05 m			
	Total (m^3)	**Flux (mm/d)**	**Fraction of rainfall**
Rainfall on fen	105657	2.51	
Rainfall on catchment	346971	2.51	
Evapotranspiration from fen	111286	2.65	24.6%
Evapotranspiration from catchment	304549	2.20	67.3%
Runoff	183088	1.02	40.4%
Change in fen storage	6933	0.17	1.5%
Error in water balance	*153227*	*0.85*	***+33.9%***

5.1.3. Runoff

A time-series of discharge from the fen outlet is presented in Figure 5.3, with discharges varying between 0 and 235 m^3/hr over the two consecutive hydrological years. High discharge events occurred mainly (but were not limited to) the winter and spring between October 1st 2018 and September 30th 2020. The largest events occurred in hydrological year 2019/2020 with the peak discharge of 235 m^3/hr occurring in the winter of 2019. This was followed up by an event with long sustained high discharges in the spring 2020 with a peak discharge of 207 m^3/hr. Another such event occurred in the autumn of 2020 with discharges up to 217 m^3/hr. Hydrological year 2018/2019 saw smaller events, which seemed to occur during the same general time period as the peaks the following year. The spring of 2019 saw the largest event with discharges up to 140 m^3/hr, whereas the winter of 2018 and autumn of 2019 only had discharges up to 198 m^3/hr and 118 m^3/hr respectively.

Also included on the hydrograph are daily rainfall amounts from the weather station, located approximately 18 km east-northeast from the fen and calculated effective rainfall in mm/d. While rainfall occurs throughout the hydrograph, the effective rainfall events only happen during the large discharge events previously mentioned. Furthermore, despite there being little rain during the summer months of 2019 and 2020 there is a slow rate of hydrograph recession which indicates the large amount of water that the fen is able to store. This slow recession could be greatly impacted by hydrological changes to the fens outlet and basin as well as peat extraction. During the driest of the summer months the fen greatly reduces its runoff, sometimes dropping down to 0 m^3/hr. Another observation made is that there is less discharge during the autumn of 2019 in comparison to the autumn of 2020 where the fen outlet already starts discharging water at the end of July.

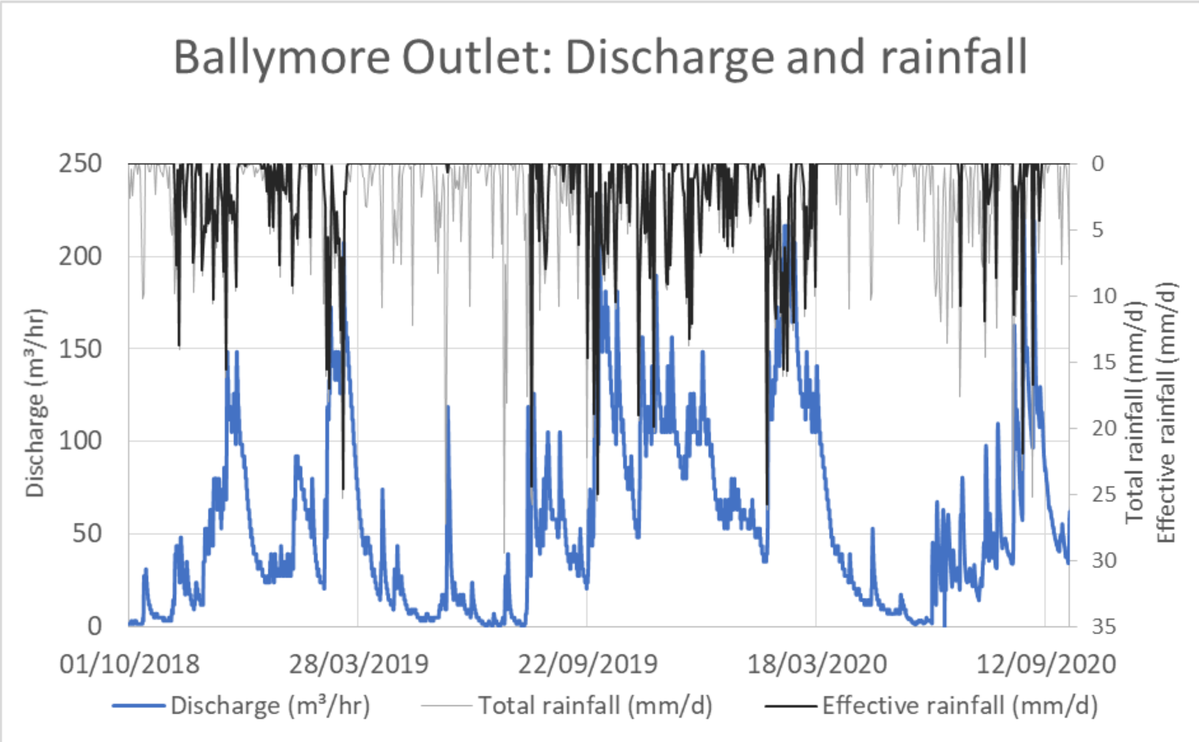


Figure 5.3. Ballymore outlet hydrograph and total/effective rainfall between October 1th 2018 and September 30th 2020.

Figure 5.4 displays the electrical conductivity (EC) recorded against the time-series discharge. The consistently high EC in the drain with an average of 5.5 ms/cm indicates a dominant groundwater contribution to runoff. Using the electrical conductivity, the groundwater contribution of the total discharge can be calculated. For this calculation it was assumed based on EC measurements from boreholes around the fen that the groundwater that supplied the fen had an EC of 8 ms /cm. Rainfall that mixes with the water in the outlet was assumed to have an EC of 0.5 ms/cm based on Figure 5.4. The outcome of this is presented in Figure 5.5 where the groundwater portion was found to be as low as 33% of the total discharge during periods of high effective rainfall compared to periods of little to no effective rainfall during which the largest contribution of groundwater (up to 87%) is calculated.

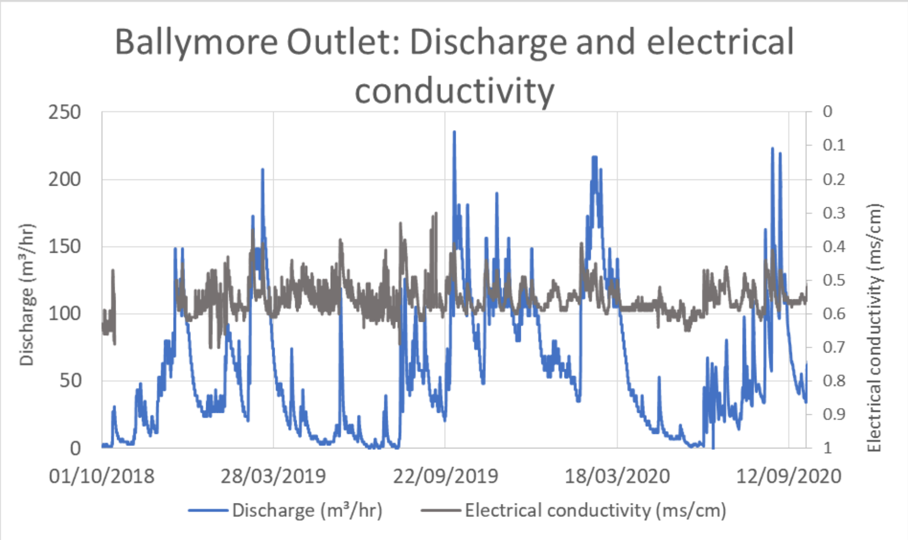


Figure 5.4. Ballymore outlet hydrograph and electrical conductivity between October 1th 2018 and September 30th 2020.

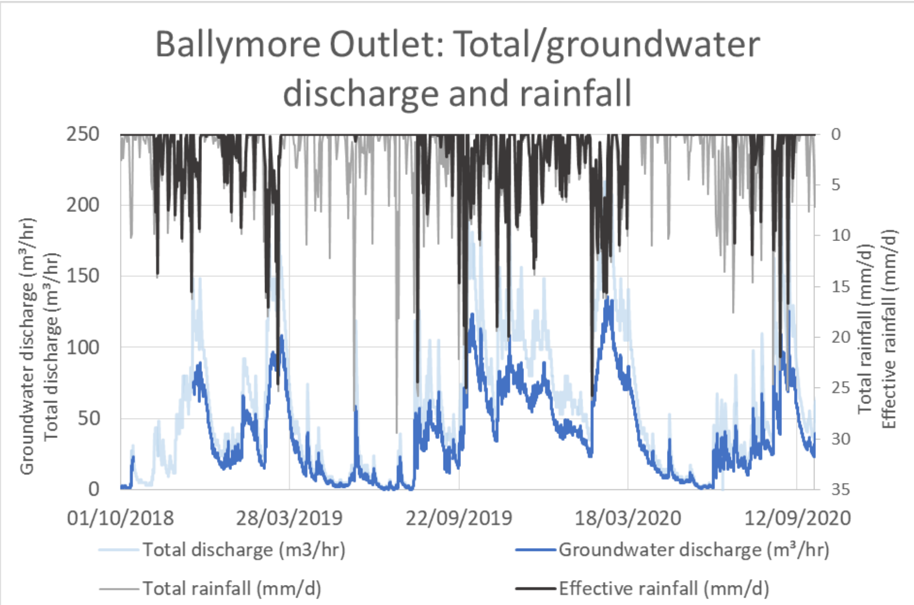


Figure 5.5. Ballymore total and groundwater hydrograph and total/effective rainfall between October 1th 2018 and September 30th 2020. Note: no groundwater contribution was calculated between October and December 2018 due to a lack of readings.

A time-series of the total evapotranspiration against the discharge is displayed in Figure 5.5. The highest numbers were recorded in the summers with a total evapotransipiration of 6.5 mm/d in 2019 and 6.7 mm/d in 2020. The amount of discharge starts to decrease at the end of March in 2019 and 2020 while the evapotranspiration is increasing, as would be expected. This trend seems to continue until there is a minimum of runoff during the summer. Then, when the evapotranspiration starts to decrease at the end of the summer, the discharge increases again. There is less discharge during the autumn of 2019 in comparison to the autumn of 2020 where the fen outlet already starts discharging water at the end of July.

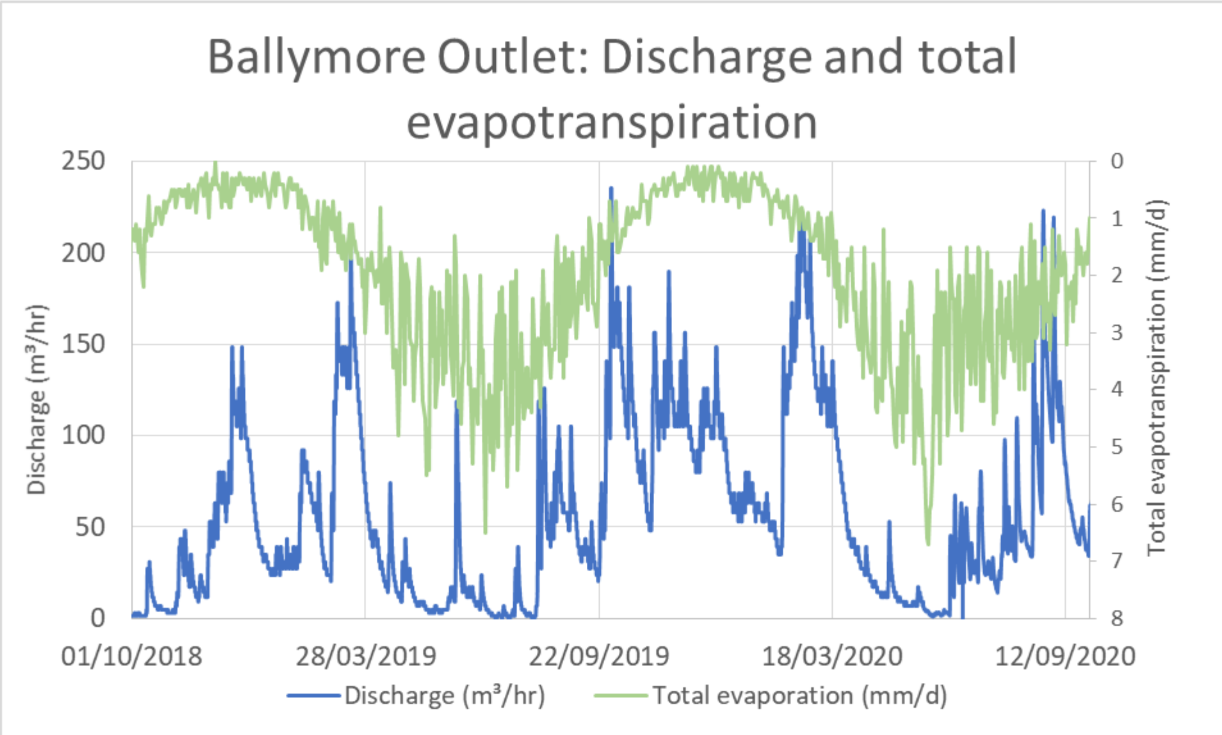


Figure 5.6. Ballymore outlet hydrograph and actual ecapotranspiration between October 1th 2018 and September 30th 2020.

The effect of daily evapotranspiration can be found in the time-series of the waterlevels of the fen as shown in Figure 5.7. Here a hydrograph is presented over the span of 20 days as well as the temperature. Even though the temperature remains constant in the water column, a diurnal water level fluctuation of around 1 cm can be observed, within the more gradual rise and fall of the water table linked to recharge events. The waterlevel drops during the days when the vegetation roots are actively taking up and transpiring water. This level is then more or less restored during the night. However, a small decreasing trend can be observed on days with no rainfall which is due to the continous draining of water from the fen via the outlet. Additionally, the effect of rainfall compared to the piezometric water levels can clearly be seen on Figure 5.12. The rainfall around 15th April 2019 shows that there is no delay between total rainfall on the fen and the water level fluctuation. Furthermore, in this example, the peak of the total rainfall (2 mm) sees a direct equivalent rise in the fens waterlevel. There are numeral different ways in which these diurnal groundwater fluctuations are used in literature. Ahmad et al. (2020) also observed these diurnal groundwater fluctuations and used data collected in different fen types in order to calculate daily ET within different species composition. In two fen habitats labelled 'cleared and mowed fen' the mean evapotranspiration rate was reported as 5.84 (± 1.71) mm/day and 5.24 (± 1.73) mm/day. Frahm et al. (2010) estimated the evapotranspiration rate of two vegetation sites - willow (Salix spp.) and reed (Phragmites australis) in a riverine fen. Using a "Draw Down Recharge" method the maximum daily values of ET_{GW} were estimated at 7.9 mm/day for willow and 5.9 mm/day for reed. These values, however, are much higher than reported in the literature review (see Table 2.3) with values reported not higher than 3.0 mm/day for poor open fen. These differences emphasises the heterogeneity of ET values in different fen habitats and meteorological systems.

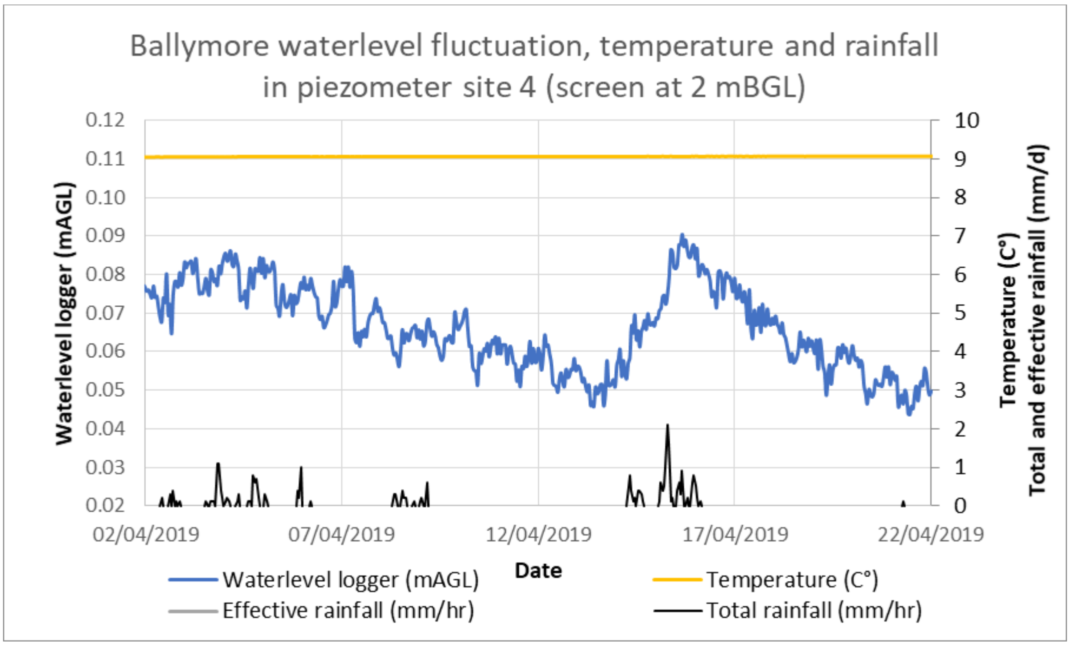


Figure 5.7. Ballymore fen piezometer hydrograph and temperature between April 5th 2019 and April 25th 2019.

Figure 5.8 displays the hydrograph against the recorded temperature of the fen outlet. The temperatures range from a minimum of 0.1 °C in the winter up to a maximum of 22.2 °C in the summer.

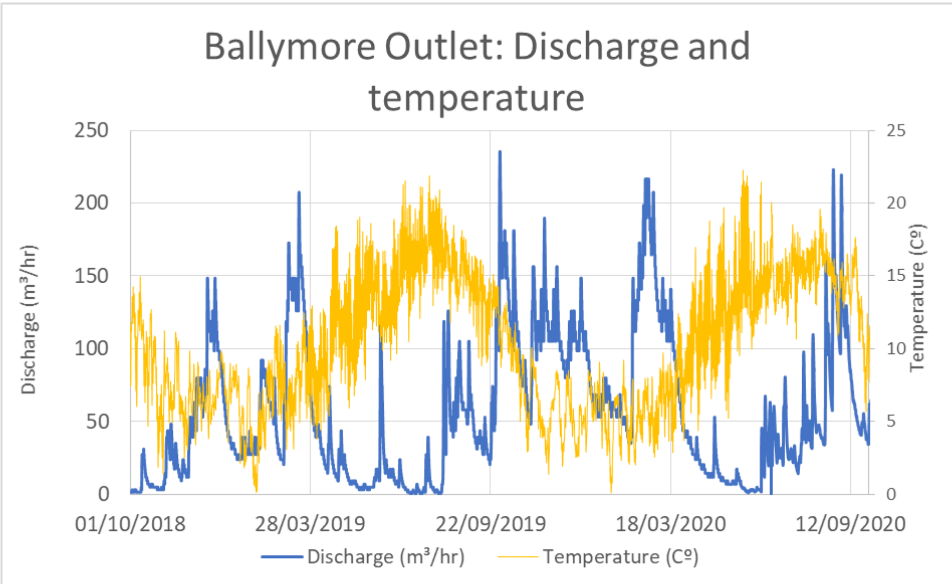


Figure 5.8. Ballymore outlet hydrograph and temperature between October 1th 2018 and September 30th 2020.

5.1.4. Fen piezometer and phreatic tube data

Surface water points in the fen and groundwater table points around the fen were interpolated into contour lines in order to interpret the flow in and out of the fen. This was done with data collected in August 2019 (Figure 5.10) and February 2020 (Figure 5.11) in order to compare seasonal changes.

Figures 5.11 and 5.12 show the phreatic and piezometric water levels between July 2018 and October 2020 of the measured locations, either as spot measurements or hourly logged water levels. The locations of the sites where this data was collected can be found in the Ballymore instrumentation map in Figure 5.9.

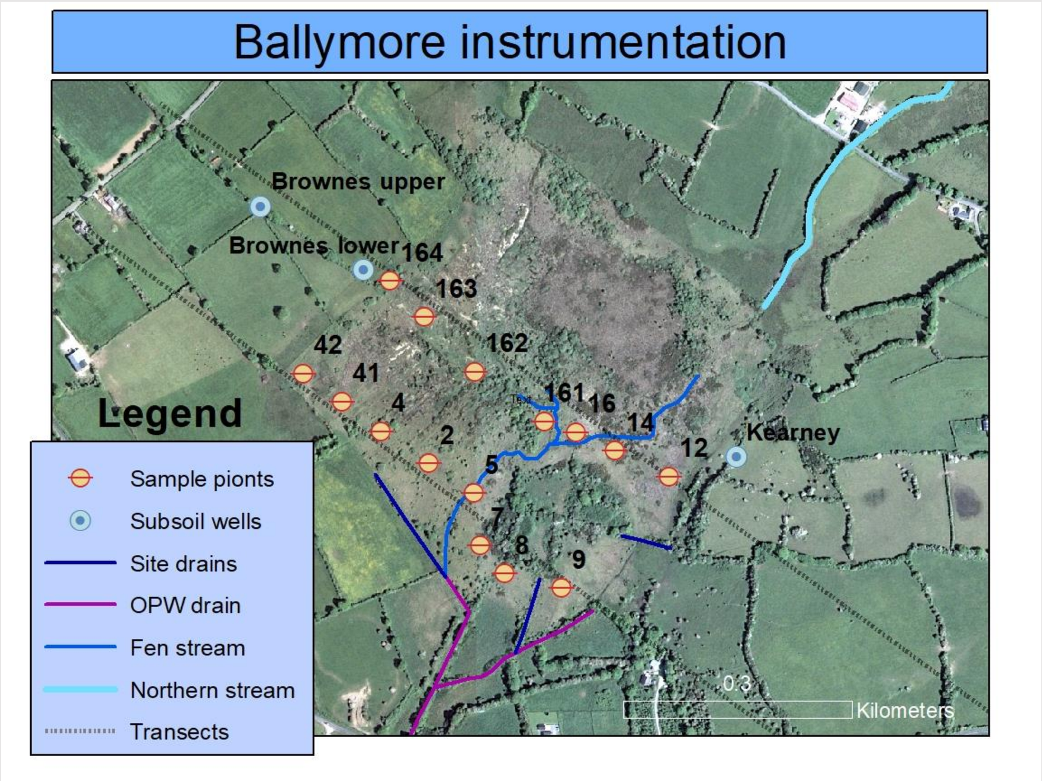


Figure 5.9. Ballymore instrumentation map showing fen piezometer and phreatic tube locations, subsoil well locations and the main site drains.

Water flows from the northwest and the east towards the fens natural fen stream in winter (Figure 5.11) where it discharges into the OPW drain. However, during the summer (Figure 5.10) lowest levels are not found at around the fen stream. Instead the lowest levels are found in the middle of the fen northeast of the fen stream and seem to discharge to the South Eastern border of the fen. It would seem, however that water that flows this way still ends up in the fens natural stream as a small ditch was observed in this location while collecting data. This flowpath is also depicted with a black arrow in Figure 5.10. Other than this shift in the fen, surface water level remain relatively unchanged between seasons which could be evidence for the fens resilience against environmental changes.

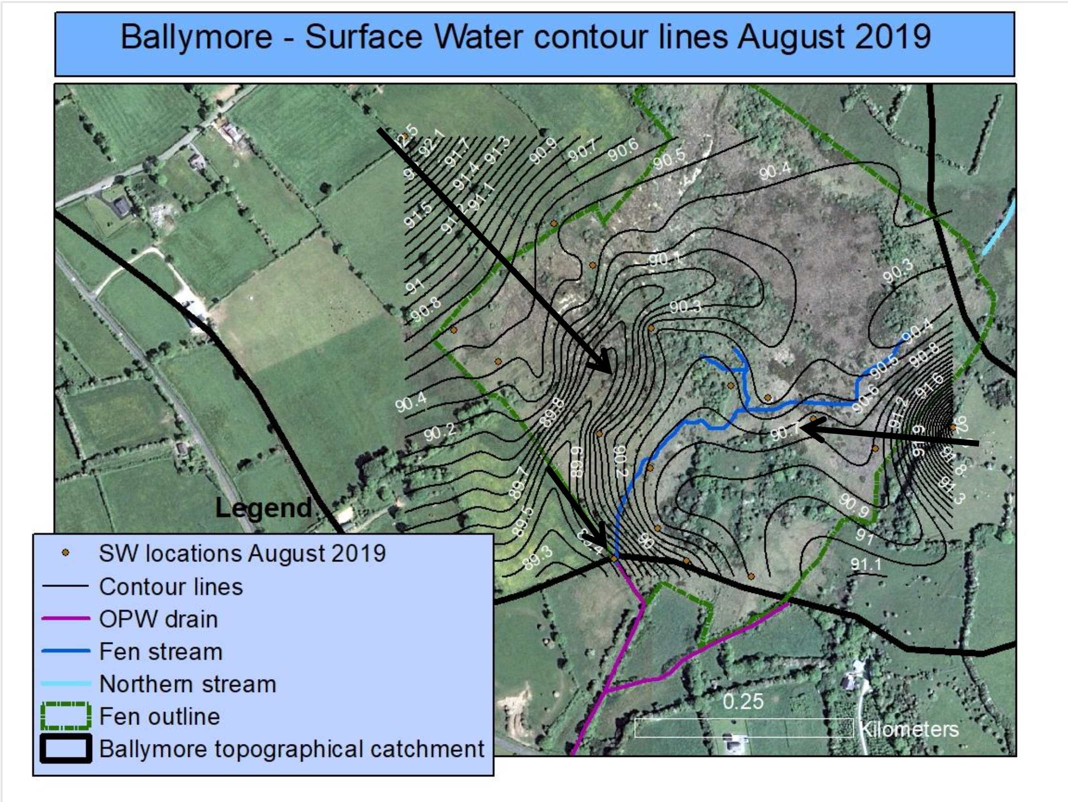


Figure 5.10. Contour lines of fen surface water and surrounding groundwater catchment interpolated using piont measurements in August 2019. Flowlines are presented with black arrows.

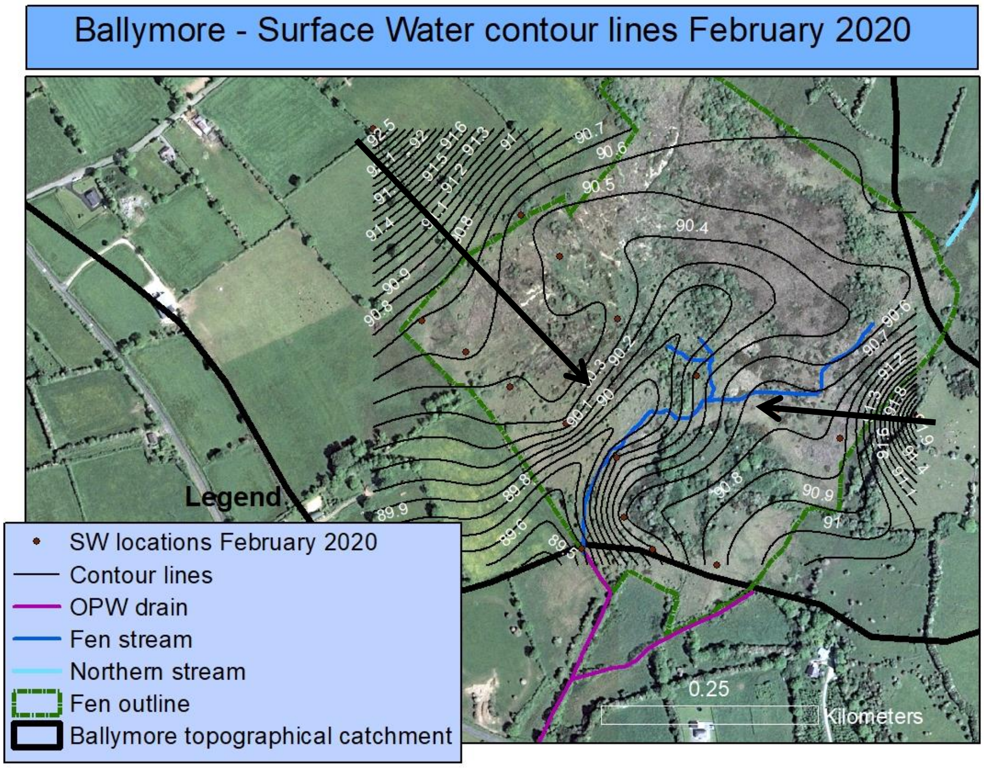


Figure 5.11. Contour lines of fen surface water and surrounding groundwater catchment interpolated using piont measurements in August 2019. Flowlines are presented with black arrows.

The water levels in the phreatic tubes (Figure 5.12) are fairly stable across the two hydrological years but do show a small response to periods of high effective rainfall. However, water levels do not drop significantly during periods when no rainfall was recorded and never drop below the invert of the outlet measured in the area where the water leaves the fen. The difference in water height between all measured points in the fen was as much as 1 m which could be attributed to slightly different surface elevations and the fact that different habitats have different water storage abilities. The water height in the fen can also be affected by location of the measuring point relative to the fen's outlet. BM5 with the lowest recorded water levels is located in an area close to the fen's outlet, reflecting the drawdown at this point with the water level strongly controlled by the stage of the outlet. The same response is also seen in BM16, and MB161 were also low water levels were recorded.

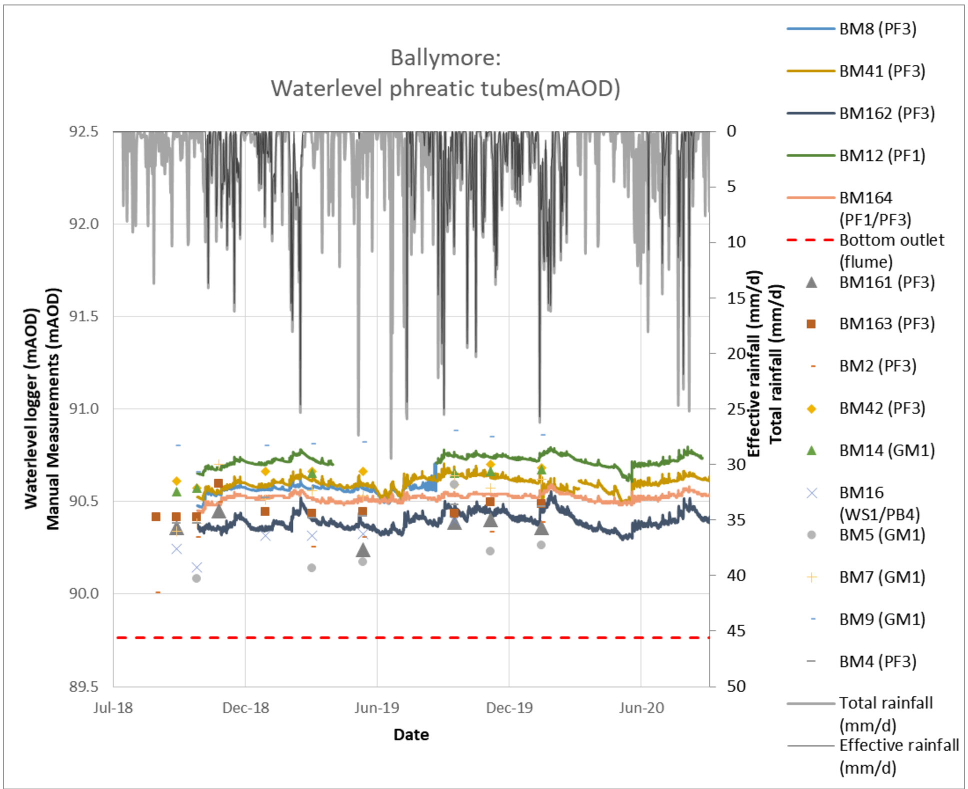


Figure 5.12. Phreatic water level hydrograph of spot measurements and water level loggers and rainfall. The height of the bottom of the outlet (measured at the flume) is presented with a red dashed line.

The outlet seems to have a larger influence on the water level fluctuations measured in the piezometers (Figure 5.13). Here BM5, BM16 and BM161 display considerable water level drops, which are sometimes recorded below the invert of the outlet. This observation could be explained by the fact that there may exist and exchange of water between the fen and the regional groundwater table. During the summer groundwater may leak through the fen base. Another explanation for this is the increase rate of evapotranspiration rates. It is likely that these processes are occurring simultaneously in order for this effect to happen.

Furthermore, the drought in the summer of 2018 did seem to have a considerable effect on the water levels. These levels were however, quickly restored during the following winter. There appears to be strong ground water inflows at the screen depth of piezometer BM162 (3.45 mBGL) in the middle of the fen where the water level is recorded at much higher level than the water level in the phreatic tube.

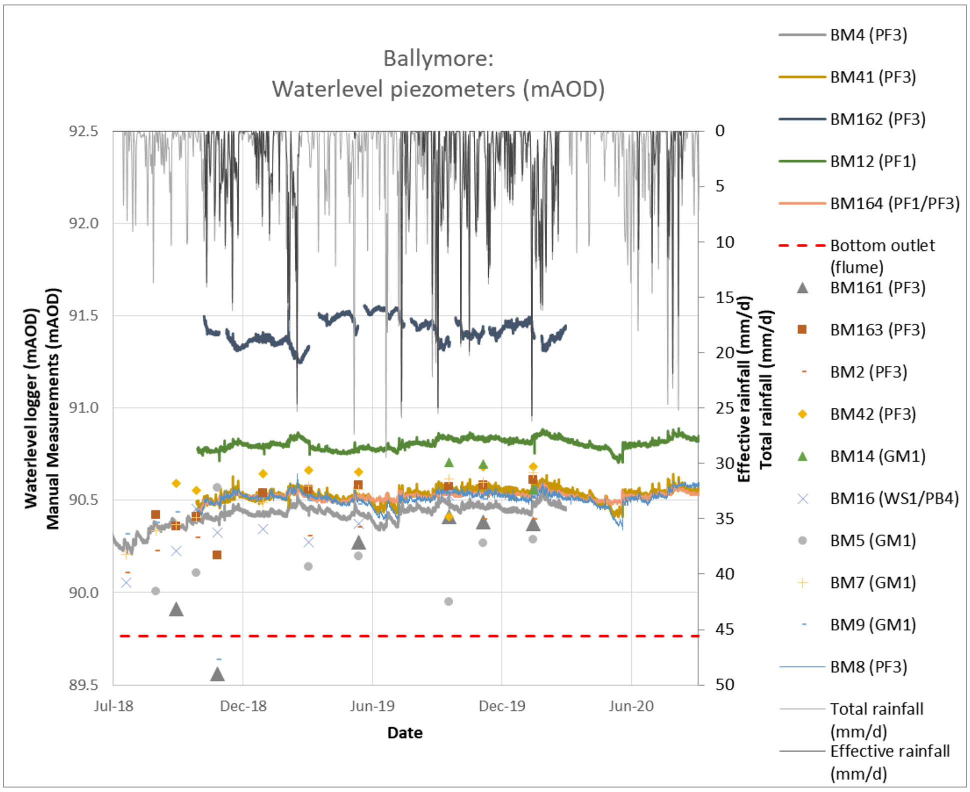


Figure 5.13. Piezometric water level hydrograph of spot measurements and water level loggers and rainfall. The height of the bottom of the outlet (measured at the flume) is presented with a red dashed line.

A time series of water levels in the fen are compared to levels in the surrounding catchment in Figures 5.14 and 5.15. The phreatic and piezometer water levels at site 164 are compared in Figure 5.14. A visible springs feeds the fen close to this site. The piezometer has been installed in the subsoil below the fen peat and has a screen depth of 1 mBGL. The subsoil well BRL in the surrounding catchment was installed approximately 10 m from the fen site and is compared the water levels in the fen.

The figure shows that the water levels measured in both the piezometer and phreatic tube show minimal response to the signal of the piezometric heads in the adjacent subsoil borehole. It has to be noted however that there was almost no water level difference between the phreatic tube and piezometer in site 164 as seen in their overlap. This shows that the piezometer with its screen at 1 m is still measuring the free water table rather than the piezometric head.

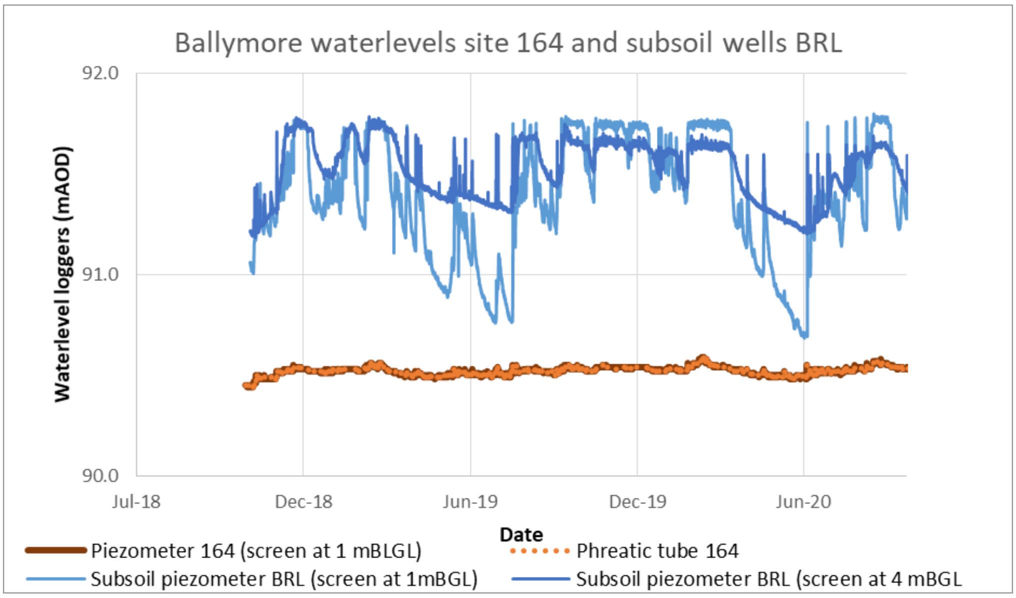


Figure 5.14. Hydrograph of phreatic and piezometric water levels at site 164 and piezometric water levels in subsoil well BRL.

Figure 5.15 shows the phreatic and piezometric head recorded in site 12. The piezometer has been installed in the fen peat and has a screen depth of 1.7 mBGL. The piezometer nest is located at the opposite end of the fen from subsoil well BRL at a distance of approximately 620 m. Again, both the piezometer and phreatic tube show minimal response to the signal of the piezometric heads of BRL. However, it is interesting to note that the water level drop measured in the subsoil piezometer (with screen of 1 mBGL) in May 2020 had a direct effect on the water level of site 12. It therefore seems that this specific fen location has a threshold level where the water table around the fen needs to be above approximately 90.8 mBGL in order not to have drastic draw down effects. Furthermore, this also seems to indicate that water is entering the fen at this elevation especially since the subsoil wells were measuring the water levels at the opposite side of the fen from the piezometer at site 12.

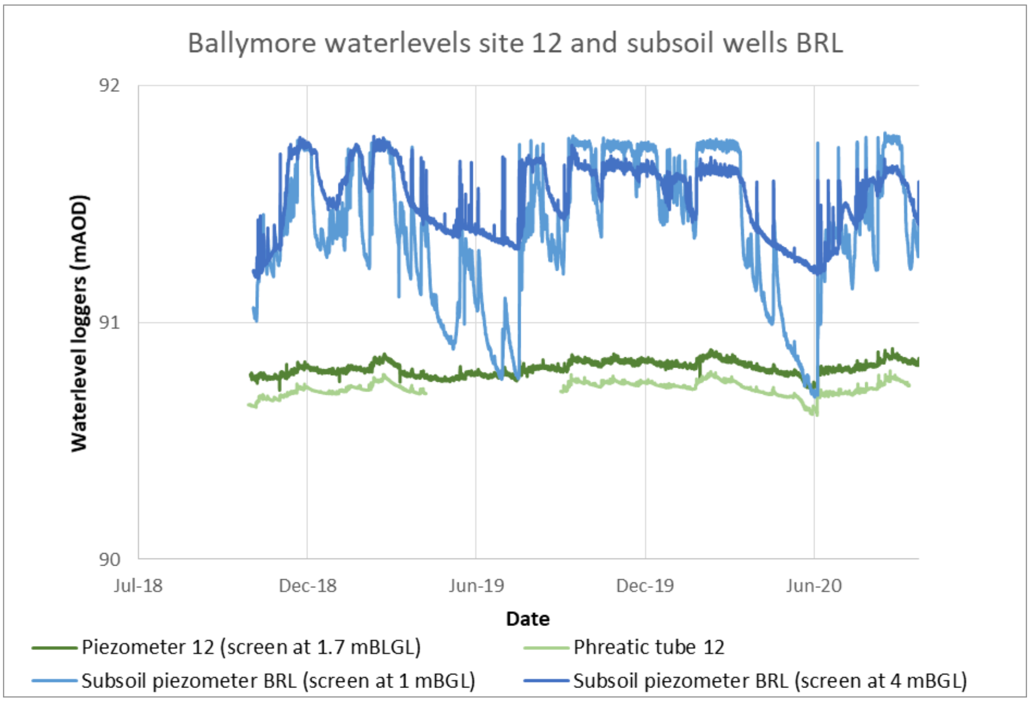


Figure 5.15. Hydrograph of phreatic and piezometric water levels at site 164 and piezometric water levels in subsoil well BRL.

5.1.4.1. Electrical conductivity

By measuring electrical conductivity (EC) an estimation can be made of the groundwater inputs in the fen. This has been done by several authors (Wheeler et al., 2009 and Harvey et al., 2007) in previous research. A time series of EC in the phreatic tubes of Ballymore coupled with rainfall is shown in Figure 5.16. Here it seems that during the winter months when effective rainfall was a lot higher, there is an increasing trend of EC which implies stronger groundwater inputs. It is further possible that this elevation is caused by surface runoff which is expected to be stronger after a dry summer. The lowest EC were recorded in the summer implying that the groundwater inflows during this time are much reduced. This is confirming the lag mechanism between the water entering the fen via subsurface flow from precipitation on the surrounding catchment as the water gained during the summer is of low EC. This means that this water does not enter the fen from the groundwater table but rather from the sub surface of the surrounding catchment.

The phreatic tube with some of the highest values sustained throughout the data collection are locations BM163, BM164 and BM12. As seen in Figure 5.9 these phreatic tubes are located at the north-eastern or south-western edges of the fen. This may indicate that regional groundwater has a stronger hydraulic gradient into the fen at these edges. Indeed, there was a visible spring spotted on the adjacent hill close to BM164.

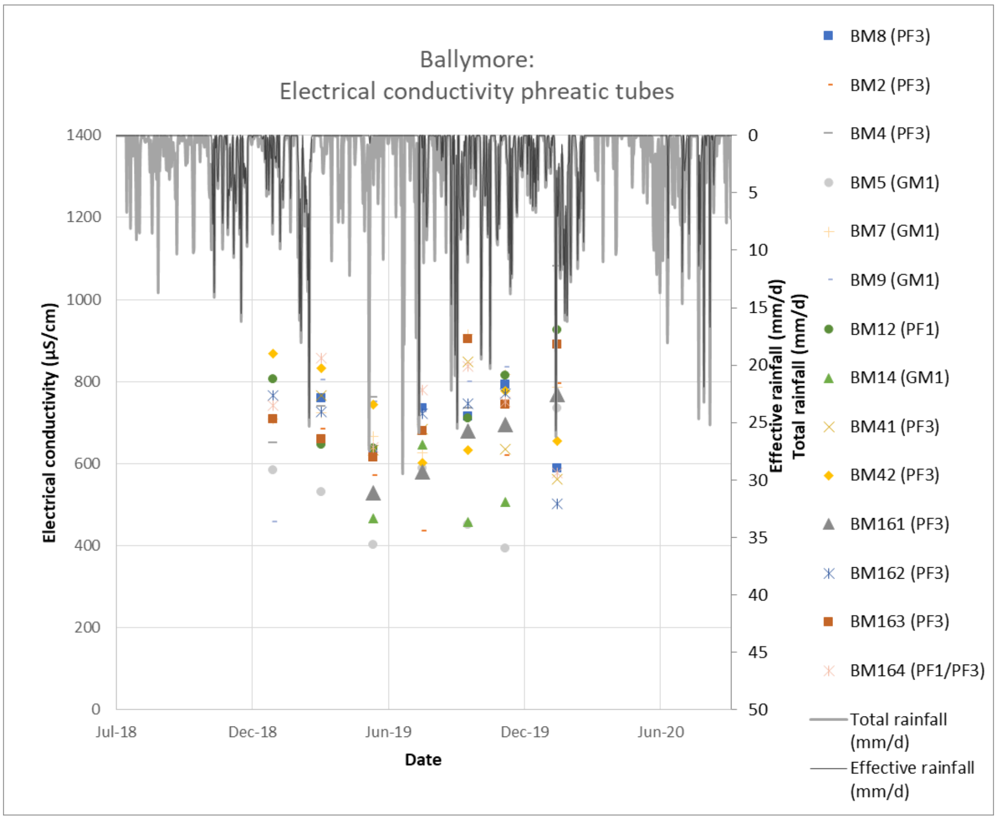


Figure 5.16. Time series of electrical conductivity (µm/cm) in phreatic tubes.

In Figure 5.17 some of the EC measured in piezometers follows the same trend as the phreatic tubes, however not all. At sample locations BM42, BM14 and BM5 there is a decreasing trend in the winter of 2019. This could suggest that in those locations lower groundwater inputs were found at the screen depth of the piezometers. These low measurements only seemed to occur in the winter of 2019/2020 as the winter of 2018/2019 does show an increasing trend. It could be that groundwater flow in these locations and at that depth can change quite significantly in reaction to the regional catchment. Also, it has to be noted that BM14 is a sample location on the cutover bog/fen margin close to the ombrotrophic bog which may therefore have received higher influxes of surface water at times.

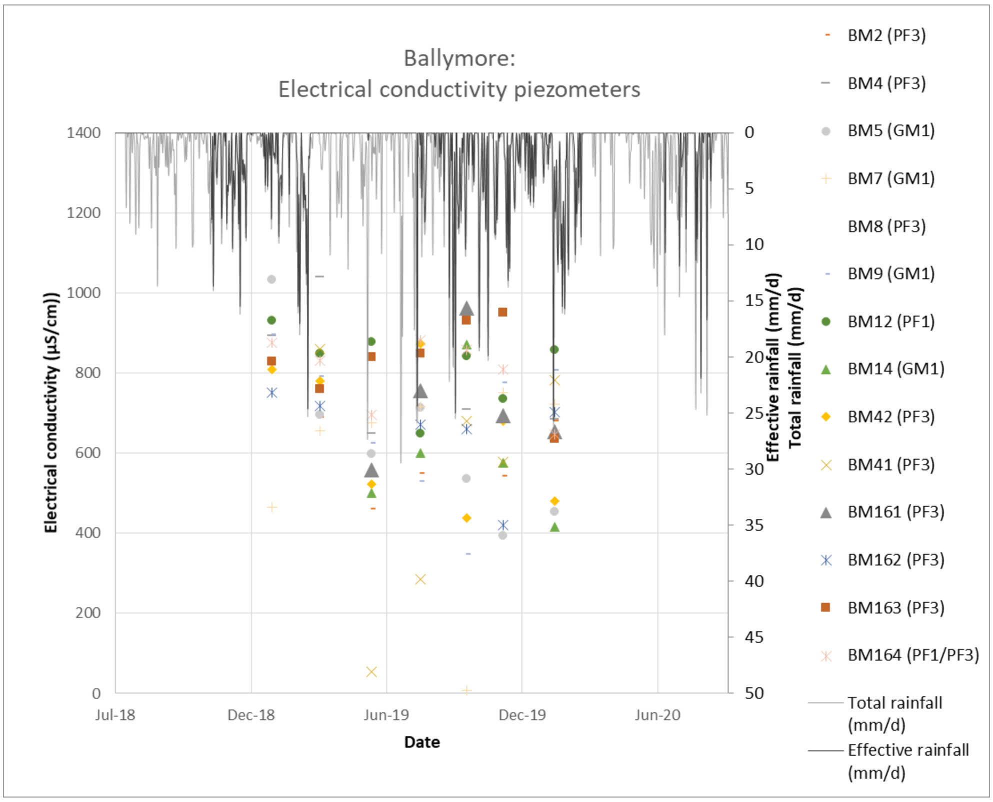


Figure 5.17. Time series of electrical conductivity (µm/cm) in phreatic tubes.

The boxplots in Figure 5.18 show overall and seasonal electrical conductivity from data collected in the fen as well as from boreholes outside the fen. The overall EC in the boreholes and the fen are quite similar although the measurements in the boreholes showed more fluctuation. Indeed, it seems that the EC in the Spring/Summer is higher with a median of 788 µm/cm than the Autumn/Winter with a median of 596 µm/cm. However, when a Welch t-test was conducted a p-value of 0.07 is returned, which means that the values in the Spring/Summer were not significantly greater (statistically) than those in the Autumn/Winter.

The median of the phreatic tubes is lower in the summer (653 µm/cm) than in the winter (735 µm/cm) which supports the observation of the trend made earlier. The lowest EC were recorded in the summer implying that the groundwater inflows during this time are much reduced. This difference was proven to be statistically significant with a p-value of 0.04.

The median of EC in piezometers does not differ a lot between seasons. The Spring/Summer medians are slightly lower than the Autumn/Winter with 730 µm/cm and 683 µm/cm respectively. This could be because of the different behaviour of the sample locations BM42, BM14 and BM5, as mentioned earlier. Another possible explanation for this is different peat structures at those locations. Peat can absorb variable amounts of chloride and from complex anions with phosphorus and sulphur which in turn regulates the amount of EC measured in the water column. Upon further observation of the boxplot it seems that the Autumn/Winter piezometer boxplots exhibit more extreme fluctuations with a first quartile of 567 µm/cm and a third quartile of 860 µm/cm.

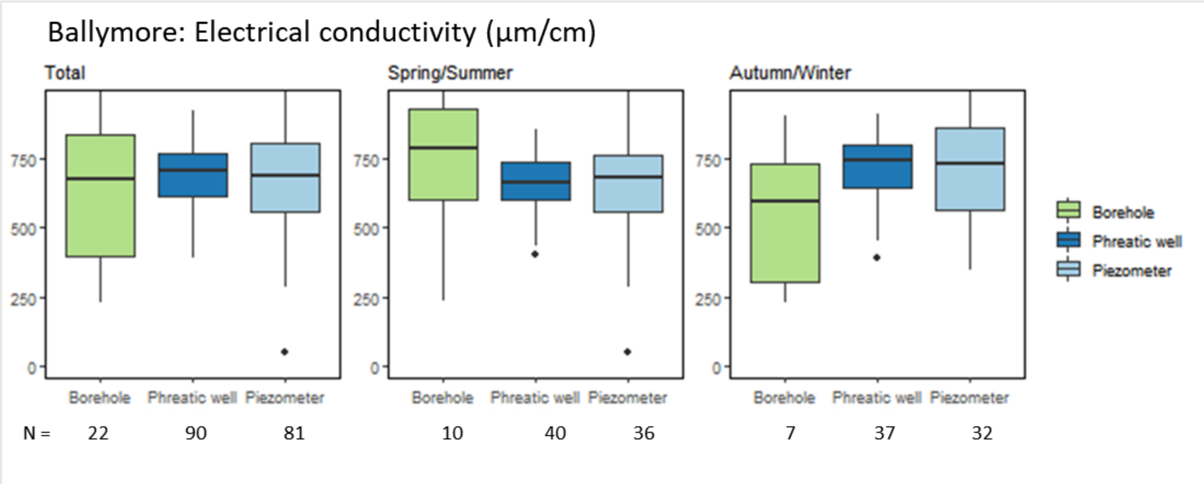


Figure 5.18. Electrical conductivity (µm/cm) in phreatic tubes and piezometers inside and boreholes outside the fen in Ballymore.

5.1.4.2. pH

The pH in Figure 5.19 suggest that values found in the fen are very similar to values found in boreholes outside the fen. Indeed, the median pH of the boreholes is 7.38 compared to the median for the piezometers in the fen of 7.34. The phreatic tubes contain an only slightly lower pH of 7.12. Furthermore, there does not seem to be any seasonal change evident. This may suggest either a considerable buffering capacity for acidity produced by the process of organic matter decomposition in peat as well as quite rapid groundwater flow through the peat in the fen.

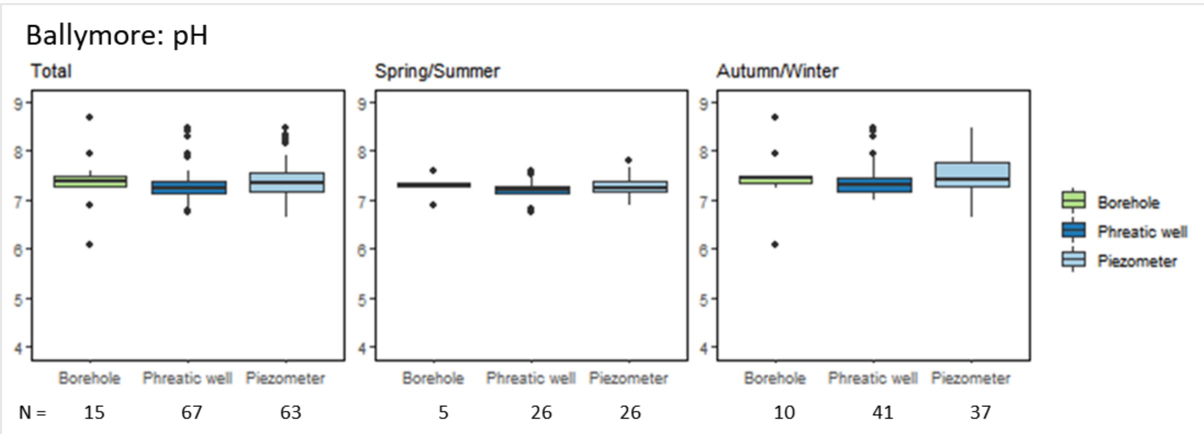


Figure 5.19. pH in phreatic tubes and piezometers inside and boreholes outside the fen in Ballymore.

5.1.4.3. Temperature

The temperature boxplots in Figure 5.20 suggest that the temperature in the water column of the fen and the surrounding catchment are similar. There is, however, a seasonal change in the phreatic tubes. The Spring/Summer median is 14.6 °C whereas the Autumn/Winter median is much lower with 9.6 °C which was proven to be a statistically significant difference with p-value 0.00. This seasonal change is to be expected for the water column at the surface.

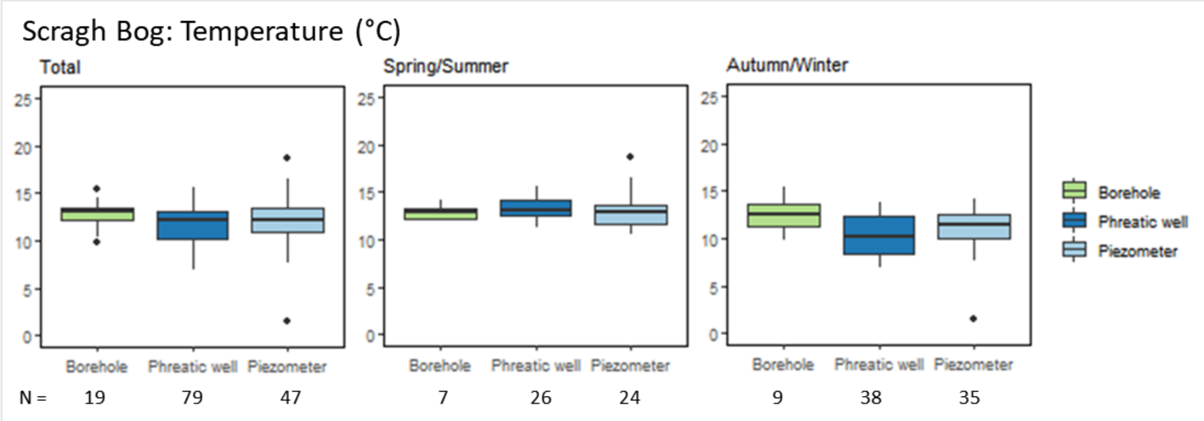


Figure 5.20. Temperature (°C) in phreatic tubes and piezometers inside and boreholes outside the fen in Ballymore.

5.1.5. Conceptual hydrology model

Several findings can be summarised from results presented and discussed in previous sections:

- 0.23 km^2 fen is supported by a surrounding catchment of 0.76 km^2. The ratio of fen to catchment is approximately 1:3.
- Discharge has the largest influence in changing the yearly water balance with big fluctuations. This discharge, however, does not lead to significant drops in phreatic and piezometric water levels and hence the fen surface water levels seem resilient to the typical Irish climate. During drier periods the fen will be sustained by groundwater inputs. For example, during the significant droughts (for Ireland at least) in 2018 the phreatic water levels did not drop below the invert of the outlet during this time.
- Effect of daily evaporation is seen in water levels during the day but the fen recovers during the night.
- The water balance calculations suggest higher groundwater inputs into the fen during the winter and this was confirmed with a rising trend of EC recorded in the sediments of the fen. Relatively though the groundwater contribution portion was actually lower during the winter due to high contribution of surface water to the system (from rainfall).
- Groundwater portion calculations of the fen runoff showed that during periods of high effective rainfall (during the winter) the relative groundwater contribution to the outlet of the fen is at the lowest 33%, whereas during low effective rainfall (summer) the relative contribution is much higher, up to 87%. This means that groundwater is being stored in the fen during periods of high effective rainfall.

5.2. Hydrochemistry

The following section contains a series of boxplots of the hydrochemistry data gathered in and outside the Ballymore. A total of 253 samples were collected from boreholes, phreatic tubes and piezometers and subsequently analysed for phosphorus, nitrogen and other hydrochemical parameters. The number of samples that made up the statistics for each boxplot is reported in each figure separately below the different sampling types. Statistical differences between the distributions shown in the boxplots were tested with a Welch t-test.

5.2.1. Phosphorus

The measured dissolved reactive phosphorus (DRP) (Figure 5.21) in the fen is low with medians of 0.01 mg-P/l in phreatic tubes and 0.02 mg-P/l in piezometers. In contrast to this are the much higher concentrations in the boreholes with a median of 0.1 mg-P/l. The concentrations in the boreholes were significant greater when compared to both the piezometers (p-value 0.04) and the phreatic tubes (p-value 0.00). They also fluctuate a great deal from the first to the third quartile with concentrations of 0.03 to 0.47 mg-P/l respectively. The DRP concentrations in the boreholes are also higher with bigger fluctuations in the Autumn/Winter. The summer shows the lowest concentrations with a median of 0.06 mg-P/l compared to 0.21 mg-P/l in the winter although this difference was not statistically significantly different (p-value of 0.09).

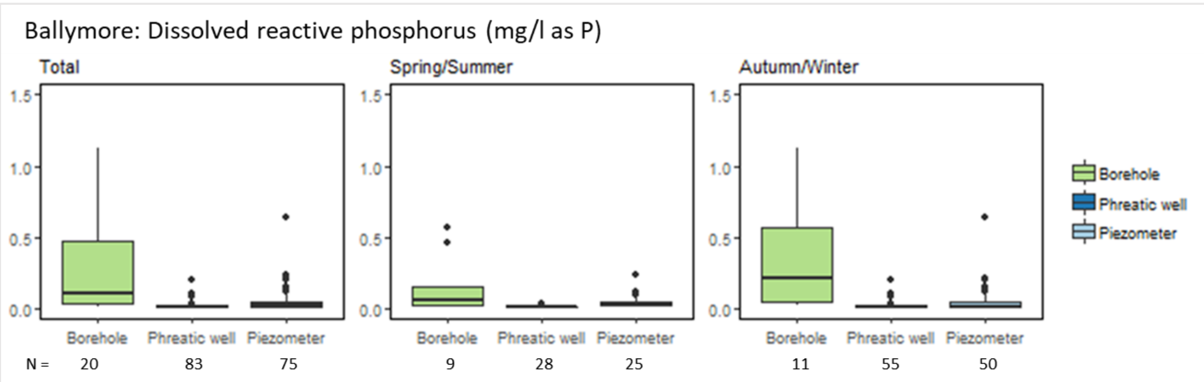


Figure 5.21. Dissolved reactive phosphorus in mg-P/l sampled from phreatic tubes and piezometers inside and boreholes outside the fen in Ballymore.

A similar trend can be seen in Figure 5.22 where high concentrations of total phosphorus (TP) are found in the boreholes outside the fen with a median of 0.85 mg-P/l. This does not seem to be reflected in the fen with medians of 0.12 mg-P/l in phreatic tubes and 0.14 mg-P/l in piezometers. The Welch t-test proved that TP was significantly greater in the boreholes than in the phreatic tubes and the piezometers (both having a p-value of 0.00).

Furthermore, the borehole data shows a seasonal change with the lowest concentrations in the summer (median of 0.51 mg-P/l) compared to increased values in the winter (median of 0.85 mg-P/l). This difference however, was not statistically significant (p-value of 0.49).

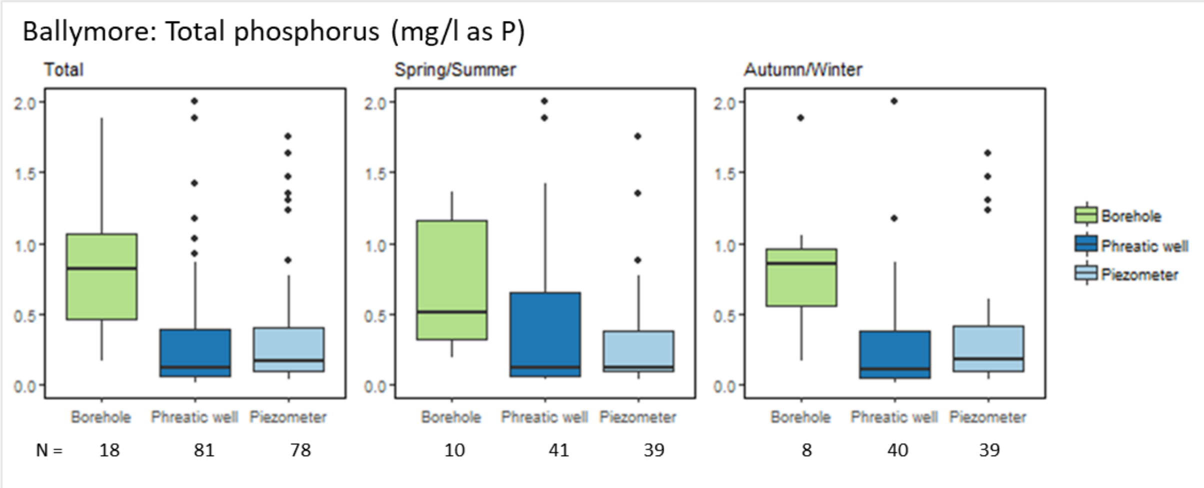


Figure 5.22. Total phosphorus in mg-P/l sampled from phreatic tubes and piezometers inside and boreholes outside the fen in Ballymore.

The ratios of DRP to TP from the full dataset across the year is 1:9 in boreholes, 1.12 in the phreatic tubes and 1:7 in the piezometers.

Finally, the DRP concentrations in the boreholes (median of 0.10 mg/l) substantially exceed the reported groundwater threshold values in Ireland (Government of Ireland, 2010) of 0.035 mg-P/l. mg-P/l. This is not true for water sampled in the fen with medians of 0.01 mg-P/l in phreatic tubes and 0.02 mg-P/l in piezometers.

What these boxplots therefore reveal is that high concentrations of phosphorus in the surrounding regional groundwater catchment was not a reflection of the concentrations in the Ballymore fen. It seems that even though the surrounding catchment contains higher concentrations of phosphorus this does not appear to directly increase the concentrations in Ballymore fen. The phreatic water table remained almost free of DRP which is where the fen vegetation takes its nutrients from. Presumably, the fen vegetation takes up the available phosphorus which is therefore transformed into organic phosphorus. This may then slowly be released to the water table on the ultimate decay of the vegetation but due to anoxic conditions much of it will be stored in the sediments. This release of soluble phosphorus during decay seems to be visible in the phreatic tubes during the spring and summer (Figure 5.19) where total dissolved phosphorus is found with increased concentrations.

The implication of these findings is that the wetland is acting as a sink for the incoming P in the groundwater, thereby acting to effectively treat the water before it leaves the fen as surface water discharge. Crowley et al. (2010) found that mosses influence phosphorus cycling in rich fens by improving vascular plant P acquisition which ultimately provides a mechanism for higher plant species diversity. P storage and nutrient cycling was also reported on numerous occasions by McBride et al., (2010). Conversely, in periods when the fen is losing water to groundwater, it must only be acting to slight dilute groundwater P concentrations.

5.2.2. Nitrogen

The boxplots in Figure 5.23 display low total ammonia concentrations in boreholes outside the fen with a median of 0.16 mg-N/l. The concentrations are comparable with phreatic tubes (median of 0.13 mg-N/l) and piezometers (median of 0.13 mg/l) and were not found to be significantly different to the borehole concentrations (p-values of 0.92 and 0.06 respectively). However, the piezometric median concentrations increased in the Autumn/Winter to 0.50 mg-N/l, although this change was not reflected in the boreholes surrounding the fen.

This might suggest that the ammonia in the piezometers does not originate from direct regional groundwater feed at depth, but rather from processes in the fen peat itself which seem to be activated during the Autumn/Winter such as the annual breakdown of decaying vegetation above. Furthermore, these increases can also be caused by the release of ammonia from the soils under oxidising conditions by ammonification of organic nitrogen. These conditions could be brought about by greater surface water proportions during the winter as was found from EC values in Figure 5.18.

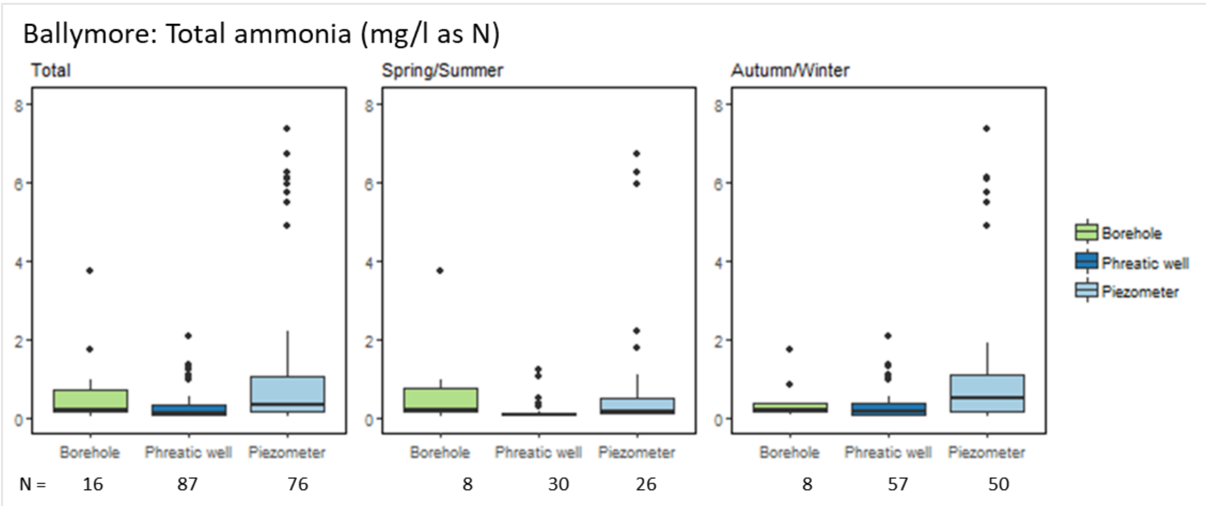


Figure 5.23. Total ammonia in mg-N/l sampled from phreatic tubes and piezometers inside and boreholes outside the fen in Ballymore.

Nitrite was barely found in and around the fen in the boxplots of Figure 5.24, which is not unexpected for this usually transitory form of nitrogen in the environment. Most samples were analysed below the limit of detection which was 0.05 mg-N/l. However, some spikes up to 0.3 mg-N/l were seen in the piezometric data which could indicate some sporadic nitrification processes in the soil at the time of sampling. However, some spikes up to 0.3 mg-N/l were seen in the piezometric data which could indicate some sporadic nitrification processes in the soil at the time of sampling. Most of these spikes were observed within piezometer 162 which takes samples of water from soils at a depth of 3 m. It would be possible that this specific soil type (marl and clays) slows down nitrification which causes these higher measured nitrite values to appear.

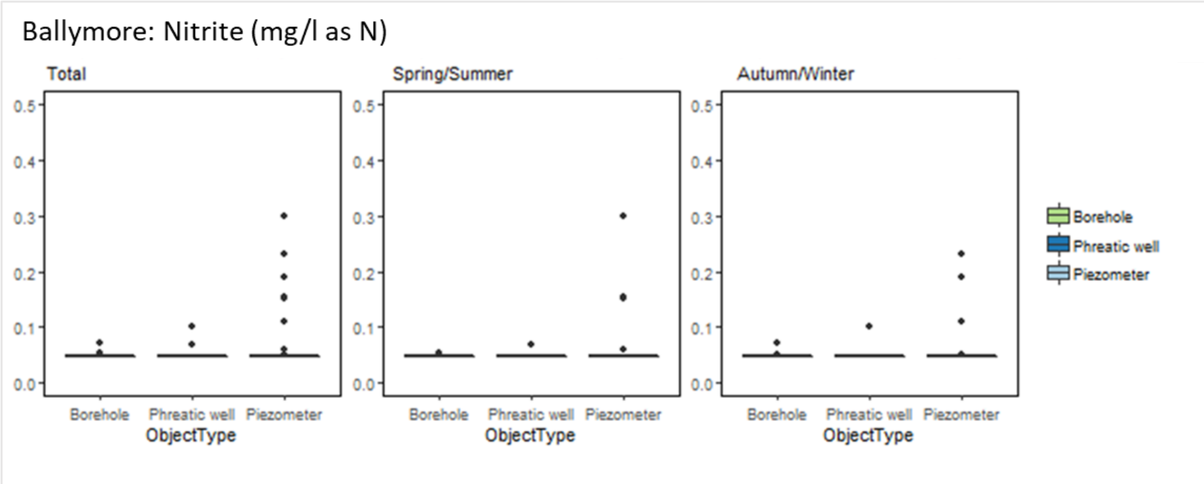


Figure 5.24. Nitrite in mg-N/l sampled from phreatic tubes and piezometers inside and boreholes outside the fen in Ballymore.

Total oxidised nitrogen results reveal higher concentrations in the boreholes, (see Figure 5.25) with the median of 0.15 mg-N/l, compared to the medians of the phreatic tubes and piezometers at 0.01 and 0.04 mg-N/l respectively. The values in the phreatic tubes were proven to be significantly lower than those found in the boreholes (p-value 0.01), however, this was not significant for concentrations in the piezometers (p-value 0.10). The concentration in the boreholes increases significantly (p-value = 0.02) during the Autumn/Winter changing from 0.15 to 0.47 mg-N/l.

The fact that the concentrations of total oxidised nitrogen are much lower in the phreatic water table than in the boreholes around the fen could again be explained that this nutrient is taken up by the vegetation during the growing season, a process also reported by Bedford & Godwin (2003) and Kuczyńska (2008). Indeed, some of the phreatic wells in the Autumn/Winter are reported with higher values whereas the Spring/Summer all displays all concentrations below 0.1 mg/l as N. Furthermore due to high denitrification potential in more reducing conditions deeper in the fen peat (Bedford & Godwin 2003) the nitrite and nitrate might ultimately be getting lost into the air as nitrogen gas. McBride et al., (2010) also reports that especially waterlogged conditions in fen soils can reduce nitrogen availability through denitrification which is the case for most of Ballymore fen throughout the hydrological year with a mean surface water level of 0.195 m above ground level.

Again, as for the P results, the implication that follows out of this is that the relatively high nitrate in the regional aquafer feeding the fen is not reflected inside the fen, thereby implying that the wetland is acting as a sink or an attenuation process for dissolved N.

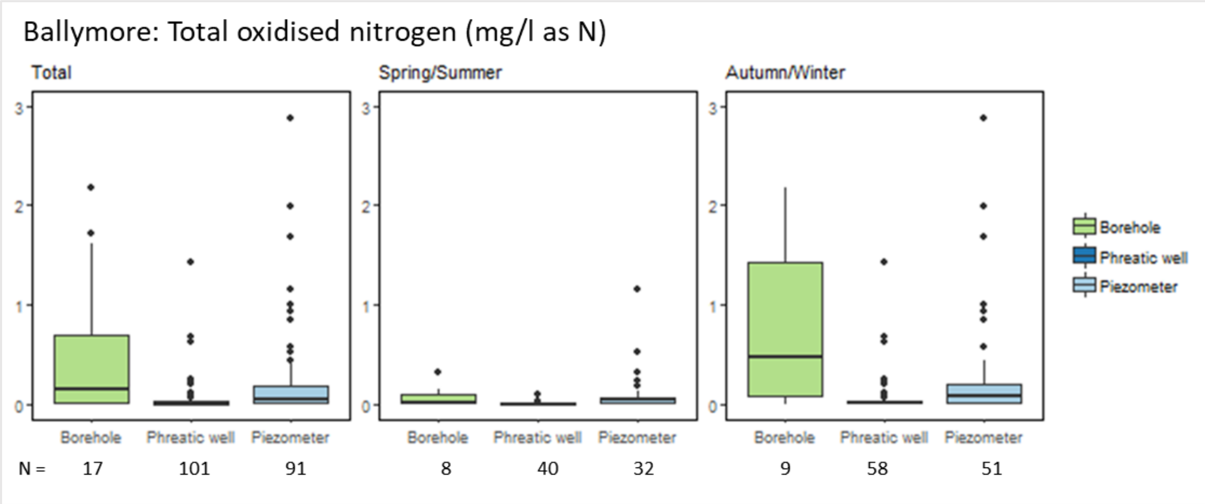


Figure 5.25. Total oxidised nitrogen in mg-N/l sampled from phreatic tubes and piezometers inside and boreholes outside the fen in Ballymore.

The highest proportion of total dissolved nitrogen (TDN) is found in the boreholes around the fen as seen in Figure 5.26. Here the median is 1.95 mg-N/l. This high concentration is also reflected in the piezometric measurements with a median of 1.25 mg-N/l. Indeed, a two-sided Welch T-test did not prove any significant difference with a p-value of 0.39. The concentrations found in the phreatic tubes are significantly lower (p-value = 0.00) with a median of 0.72 mg-N/l.

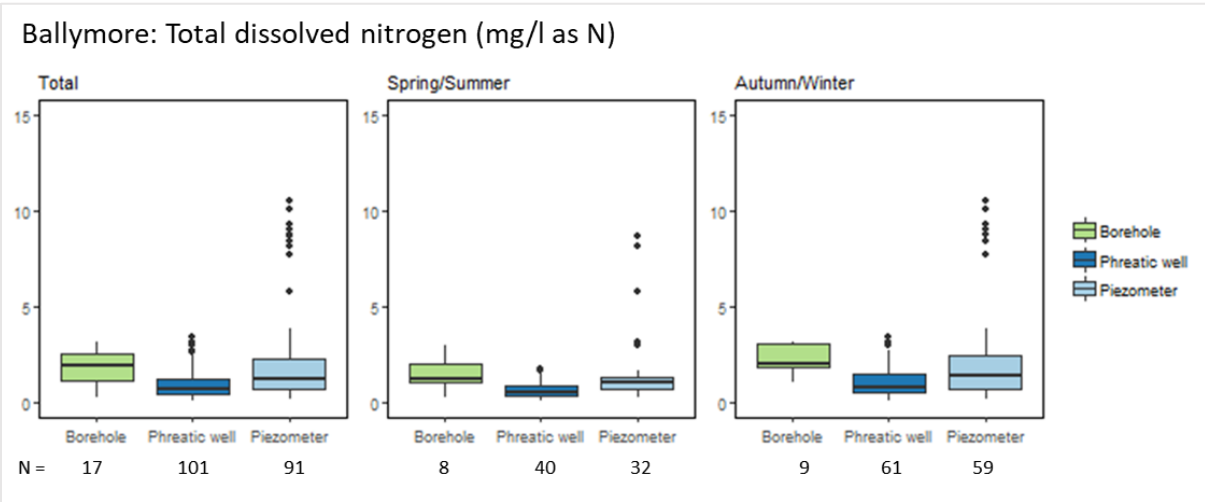


Figure 5.26. Total dissolved nitrogen in mg-N/l sampled from phreatic tubes and piezometers inside and boreholes outside the fen in Ballymore.

The ratios of median total ammonia to total dissolved nitrogen were 1:10 in boreholes and piezometers and in 1:7 in the phreatic tubes. The ratio of median total oxidised nitrogen to total dissolved nitrogen was lowest for the boreholes at 1:13, compared to 1:28 and 1:72 in the piezometers and phreatic tubes which were much higher respectively.

These ratios show that there is quite a significant portion of TDN that is neither ammonia nor nitrate/nitrite but are rather comprised of different forms of organic nitrogen. Davidsson et al. (2002) explained that mineralization of organic material in soil will cause a release of various fractions of nitrogen. During mineralization, dissolved organic nitrogen (DON) may be produced. However, the amount of DON produced in wetlands as well as its bioavailability that has so far been little studied.

The reported medians do not exceed groundwater threshold values for nitrite nor nitrate (Government of Ireland, 2010). None of the medians found in and around the fen were found higher than 0.114 mg-N/l for nitrite. There were, however, a few outliers that did exceed the threshold value. The total oxidised nitrogen concentrations had an insignificant amount of nitrite and a therefore can be regarded as a reflection of nitrate. None of the measured concentrations exceeded the threshold value of 8.47 mg-N/l.

5.2.3. Other chemistry

The overall concentrations of alkalinity were quite similar when comparing data in and around the fen as seen in Figure 5.27. Indeed, a two-sided Welch t-test proved no significant difference between the boreholes and the phreatic tubes (p-value = 0.99) and the piezometers (p-value = 0.61). The median in the boreholes was 215.5 mg/l as $CaCO_3$, whereas the median of the piezometers was somewhat higher at 231.2 mg/l as $CaCO_3$ reflecting high groundwater inputs into the fen. The alkalinity in the phreatic tubes were slightly lower with 195.9 mg/l as $CaCO_3$ which is probably a reflection of the mixed groundwater and surface water portions at the surface of the fen and/or the ability of the vegetation to take up minerals.

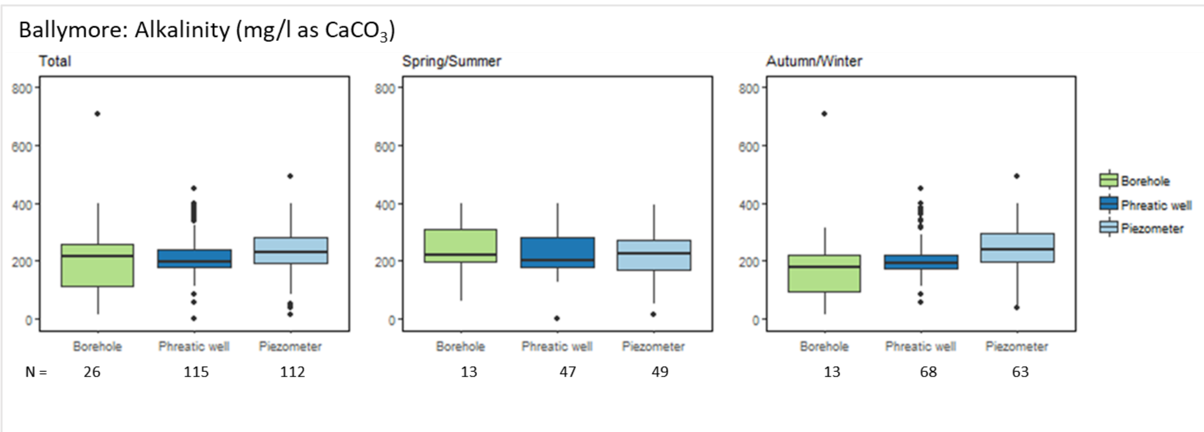


Figure 5.27. Alkalinity in mg/l as $CaCO_3$ sampled from phreatic tubes and piezometers inside and boreholes outside the fen in Ballymore.

The overall concentrations of chloride (Figure 5.28) were also similar between the boreholes around the fen (median of 25.3 mg/l) compared to measurements taken in the phreatic tubes (median of 26.4 mg/l - not significantly with p-value of 0.91) and in the piezometers (median of 22.6 mg/l - not significantly with a p-value of 0.06).

The boreholes seem to have a seasonal change with high concentrations during the Spring/Summer (median of 36.2mg/l) and a lower concentrations during the Autumn/Winter (median of 18.0 mg/l) which mirrors the pattern seen in the EC levels (Figure 5.18). The Spring/Summer concentrations were, however, not significantly greater with a p-value of 0.75. This seasonal shift is not reflected in the phreatic tubes and piezometers. Finally, the reported medians are far below the Irish groundwater threshold values for chloride (Government of Ireland, 2010) which is 187.5 mg/l.

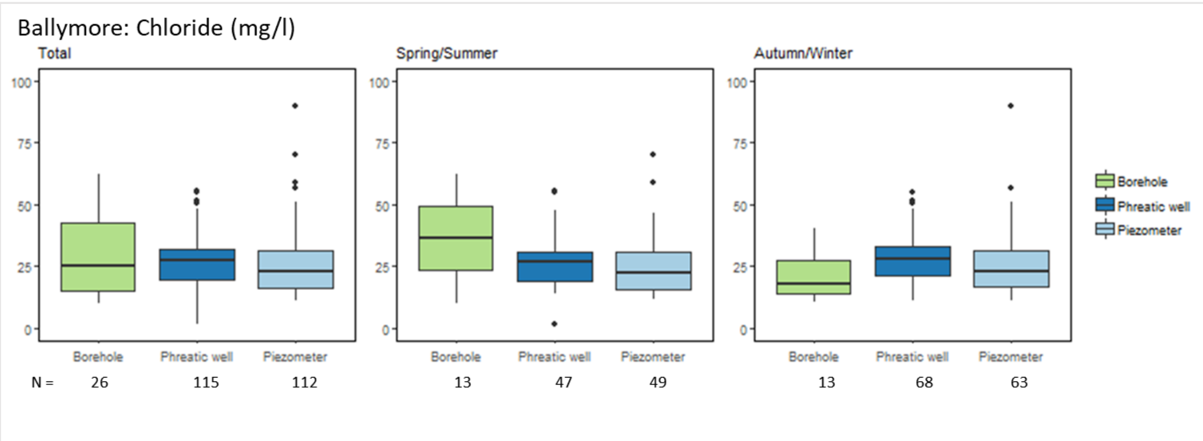


Figure 5.28. Chloride in mg/l sampled from phreatic tubes and piezometers inside and boreholes outside the fen in Ballymore.

Figure 5.29 shows that the boreholes had higher concentrations of silica with a median of 5.1 mg/l as SiO_2 which is to be expected in groundwater. The concentrations found in the fen are significantly lower with a p-values of 0.00 and 0.02 for phreatic tubes and piezometers, respectively, reflecting that water is the fen is a combination of both groundwater and surface water inputs. Median silica concentrations in the phreatic tubes were 4.0 mg/l as SiO_2 compared to the piezometers which were lower with a median of 3.1 mg/l as SiO_2.

Concentrations of sulphate appeared to increase a lot between the Spring/Summer and the Autumn/Winter, shifting from a median of 1.5 mg/l as SO_4^{2-} to 18.0 mg/l as SO_4^{2-} in the phreatic tubes and from 1.5 mg/l as SO_4^{2-} to 16.6 mg/l as SO_4^{2-} in the piezometers. It has to be noted that 1.5 mg/l as SO_4^{2-} was the limit of detection during laboratory analysis. This increase in the phreatic tubes and piezometer were significant with p-value of 0.01 and 0.04 respectively. The increase of sulphate during the Autumn/Winter could either be explained by a release of sulphate from decaying vegetation and/or higher concentrations of oxygen entering the fen via surface water causing sulphide to oxidise into sulphate (Wheeler and Proctor, 2000; Cushell et al., 2013; McBride et al., 2010). Another explanation for this phenomenon is the fact that sulphide oxidises during peak recession time which is presumably caused by the lowering water table in the fen. This introduces oxygen into areas which were previously been anaerobic causing a shift from sulphide into sulphate (Wheeler and Proctor, 2000; Cusell et al., 2013). Despite all these differences, the measured values are still all far below the Irish groundwater threshold values for sulphate (Government of Ireland, 2010) which is 187.5 mg/l. However, it has to be taken into account that fluctuations such as shown in Figure 5.27 may still have an ecological impact even if they are below a certain threshold.

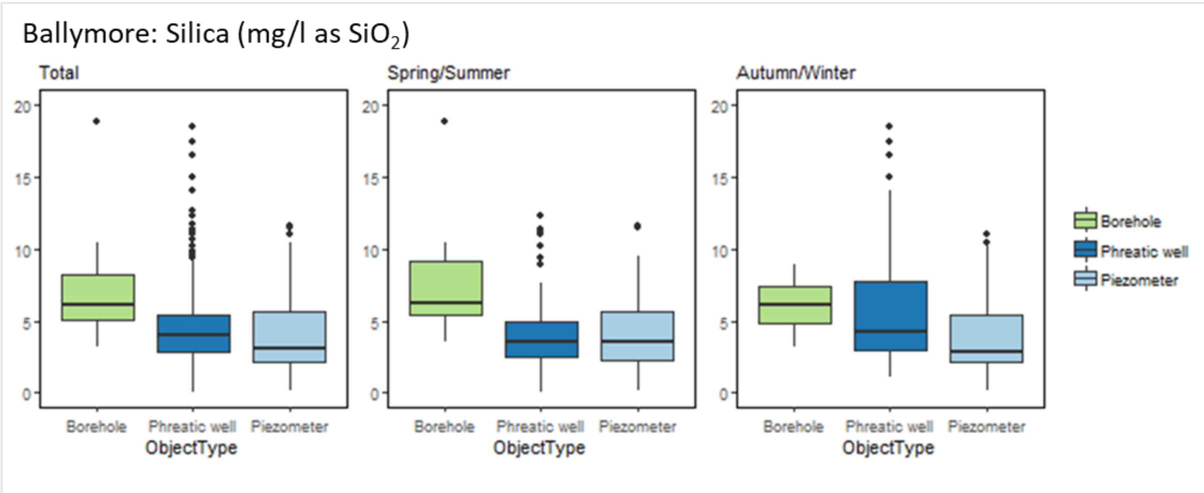


Figure 5.29. Silica in mg/l as SiO_2 sampled from phreatic tubes and piezometers inside and boreholes outside the fen in Ballymore.

Sulphate concentrations were found to be higher in the boreholes around the fen with a median of 31.4 mg/l as SO_4^{2-} compared to much lower concentrations in the fen with medians of 10.4 mg/l as SO_4^{2-} and 3.1 mg/l as SO_4^{2-} in the phreatic tubes and piezometers, respectively (Figure 5.30). These were not, however, significantly lower with p-values of 0.06 and 0.09 for the phreatic tubes and piezometers respectively.

Concentrations of sulphate appeared to increase a lot between the Spring/Summer and the Autumn/Winter, shifting from a median of 1.5 mg/l as SO_4^{2-} to 18.0 mg/l as SO_4^{2-} in the phreatic tubes and from 1.5 mg/l as SO_4^{2-} to 16.6 mg/l as SO_4^{2-} in the piezometers. It has to be noted that 1.5 mg/l as SO_4^{2-} was the limit of detection during laboratory analysis. This increase in the phreatic tubes and piezometer were significant with p-value of 0.01 and 0.04 respectively. The increase of sulphate during the Autumn/Winter could either be explained by a release of sulphate from decaying vegetation and/or higher concentrations of oxygen entering the fen via surface water causing sulphide to oxidise into sulphate. Despite all these differences, the measured values are still all far below the Irish groundwater threshold values for sulphate (Government of Ireland, 2010) which is 187.5 mg/l.

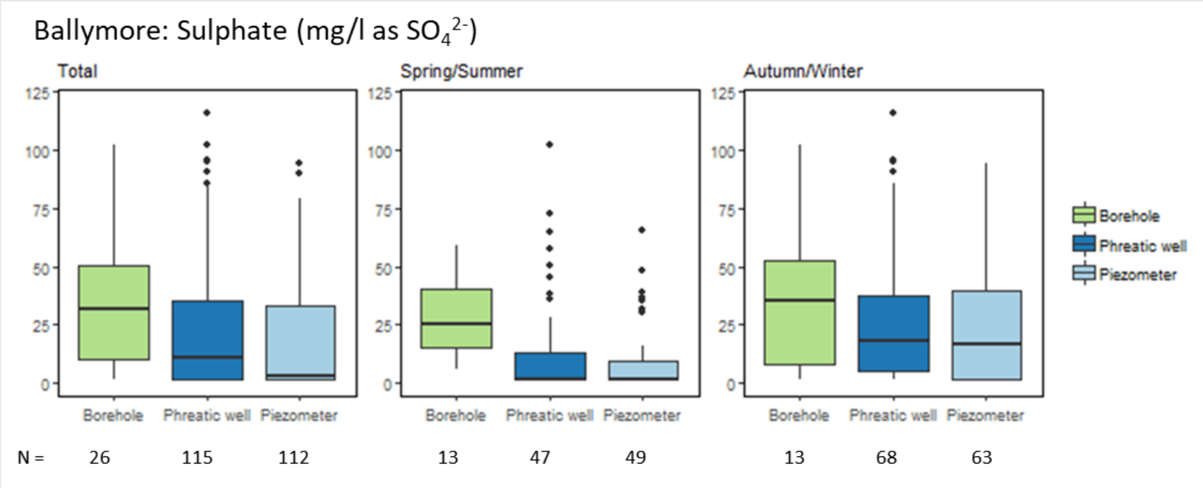


Figure 5.30. Sulphate in mg/l as SO_4^{2-} sampled from phreatic tubes and piezometers inside and boreholes outside the fen in Ballymore.

Dissolved organic carbon (DOC) in and around the fen was found with very comparable concentrations which did not seem to change between seasons (Figure 5.31). This could be due to the boreholes installed quite close to the edge of the fen and thus within the peat basin, thereby receiving leached DOC. The boreholes in the fen catchment had a median of 7.4 mg/l, compared to concentrations in the phreatic tubes and piezometers which both had a median values of 8.4 mg/l. Two sided Welch T-test did not prove any significant difference between the boreholes and the phreatic tubes (p-value = 0.41) as well as the piezometers (p-value = 0.99).

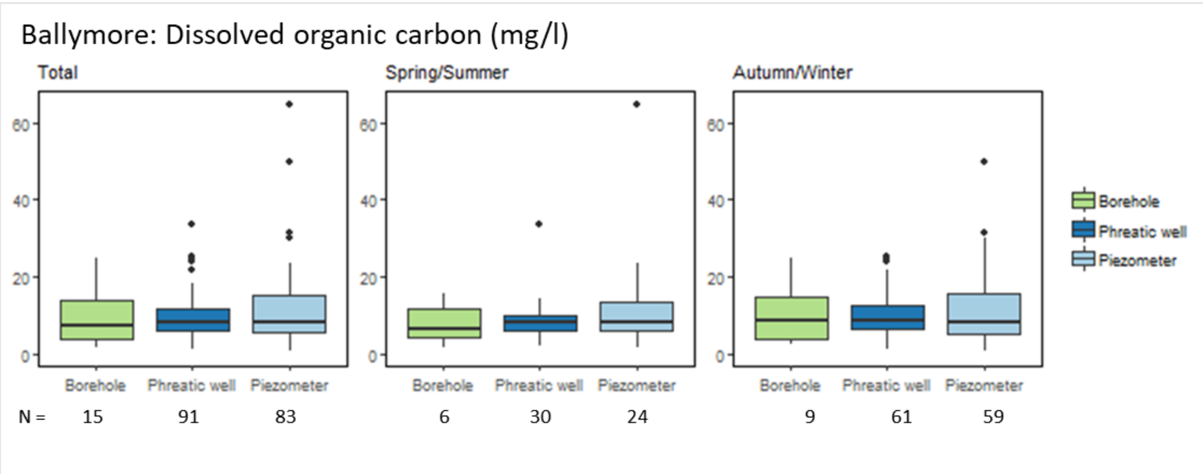


Figure 5.31. Dissolved organic carbon in mg/l sampled from phreatic tubes and piezometers inside and boreholes outside the fen in Ballymore.

Finally, a comparison of the ferrous iron results showed that very low concentrations were detected in the boreholes around the fen (median concentration of 0.06 mg/l as Fe^{2+}) as seen in

Figure 5.32. Concentrations measured in the piezometers were higher (median of 0.12 mg/l as Fe^{2+}) but not significantly so (p-value of 0.17).

The ferrous iron concentrations measured in the fen also seem to be lower during the Autumn/Winter than the Spring/Summer. This could be evidence for more reducing conditions during the summer, caused by higher temperatures leading to more microbial activity. The median in the phreatic tubes decreases from 0.123 to 0.052 mg/l as Fe^{2+} whereas the median in the piezometers decreases from 0.177 to 0.007 mg/l as Fe^{2+}. While this decrease in the phreatic tubes was significant (p-value = 0.02) this was not true for the piezometers (p-value = 0.15). The seasonal change of ferrous iron concentrations in the phreatic water table indicates a fluctuating redox microenvironment. This is interesting since a fluctuating redox environment will also affect other biogeochemical reactions which may include nutrients to release or precipitate out from the free water table. Furthermore, research found that ferric iron and ferrous iron redox reactions plays an important role in the biogeochemical cycles of C, N, S, and P, and seems to be partial driver or involved into their biogeochemical cycles at various scales (Yichun, Shen, Strong , & Hailong, 2012).

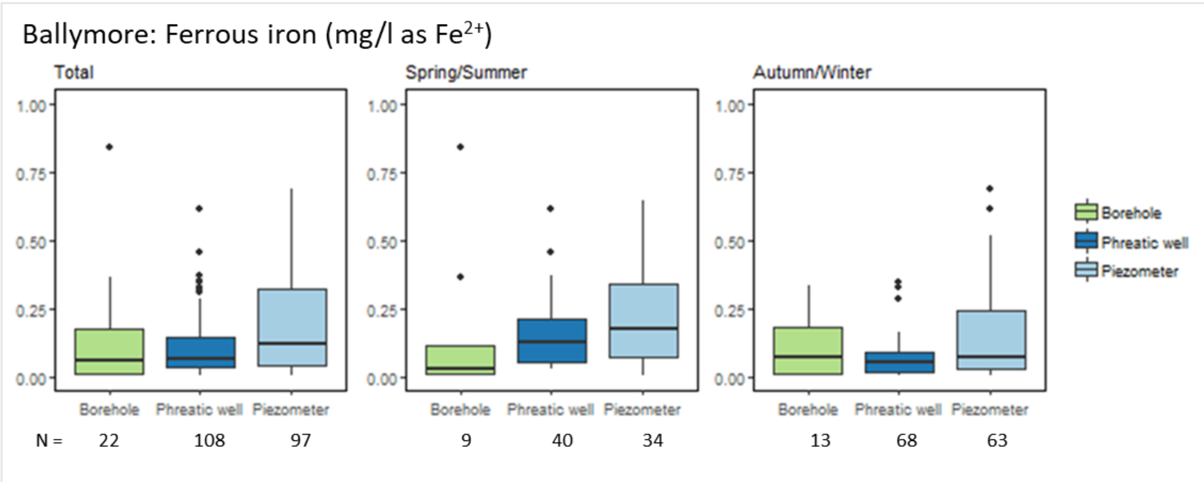


Figure 5.32. Ferrous iron in mg/l as Fe^{2+} sampled from phreatic tubes and piezometers inside and boreholes outside the fen in Ballymore.

5.2.4. Conceptual chemistry model

- The fen seems to acts as a natural treatment system for the incoming nutrients in the groundwater which are then subject to internal nutrient cycling in terms of uptake by fen vegetation and then annual die back and decay in the fen peat sediments.
    - DRP and TP is significantly higher in boreholes than in fen. Groundwater input with higher concentrations of phosphorus do not affect the water column in the fen.

  - Higher concentrations of ammonia found in deeper layers of Ballymore which seem to be present due to nutrient cycling within the fen peat itself rather than being brought in by groundwater.
  - Total dissolved nitrogen is present in piezometers and could be received from groundwater, yet these concentrations are not present in the surface water.
- Both ammonia and sulphate are higher in the winter. These increases can be caused by the release of these chemical from the soils under oxidising conditions caused by higher flow through of aerobic water. Indeed, it was established before that the fraction of surface water into the fen is greater during the winter. This may further be caused by the die-back of seasonal vegetation which releases the nutrients and minerals back into the system.

5.3. Linkage to fen habitat

5.3.1. Hydrology and fen habitat

5.3.1.1. Boxplots water level

The boxplot in Figure 5.33 displays the water level for different Fossitt habitats in the fen for all measurements (taken relative the ground surface). Overall, the water levels in phreatic tubes and the piezometers are very similar. The Scrub and cutover bog (WS1/PB4) has the lowest recorded water levels with a median of -0.38 m (below ground surface). The Marsh (GM1) habitat also has water levels below ground level but not as extreme with a median of -0.06 m.

The fen habitats however, all show median water levels above the surface. The medians of the Transition mire (PF3) and Rich fen and Transition mire mosaic habitat (PF1/PF3) are very similar with water levels of +0.07 and +0.03 mAGL. PF3, however, seemed to have more fluctuations than the mosaic habitat. Here the water fluctuated 0.17 m between the first and third quartile, however this could also be a reflection of the larger measurement number. The Rich fen and flush habitat (PF1) had water levels constantly recorded above ground level with a median of +0.10 mAGL. Here there were almost no fluctuations recorded. It has to be noted, that the PF1 habitat only had one sample site and therefore the numbers of measurements taken was lower than for the other fen habitats.

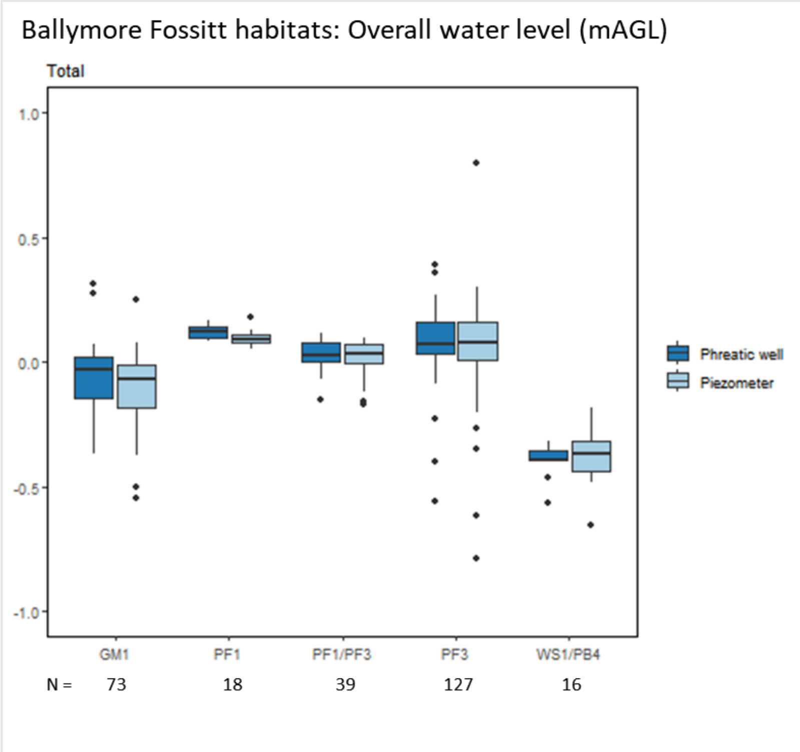


Figure 5.33. Overall water level in meters above ground level in the different habitats of Ballymore fen measured in phreatic tubes and piezometers.

Figure 5.34 shows the seasonal fluctuation in water levels. The fluctuations between the summer and winter are minimal, especially for the fen habitats in which the water levels only change by a few millimetres between seasons. The largest change was recorded in WS1/PB4 with a fluctuation of 0.05 m in the phreatic tube. The same change was also recorded in the phreatic tube of GM1 habitat. What is interesting is that the water levels for both these habitats were higher in the Spring/Summer than in the Autumn/Winter which might be a reflection of stronger groundwater flux into the fen during the spring rather than the winter.

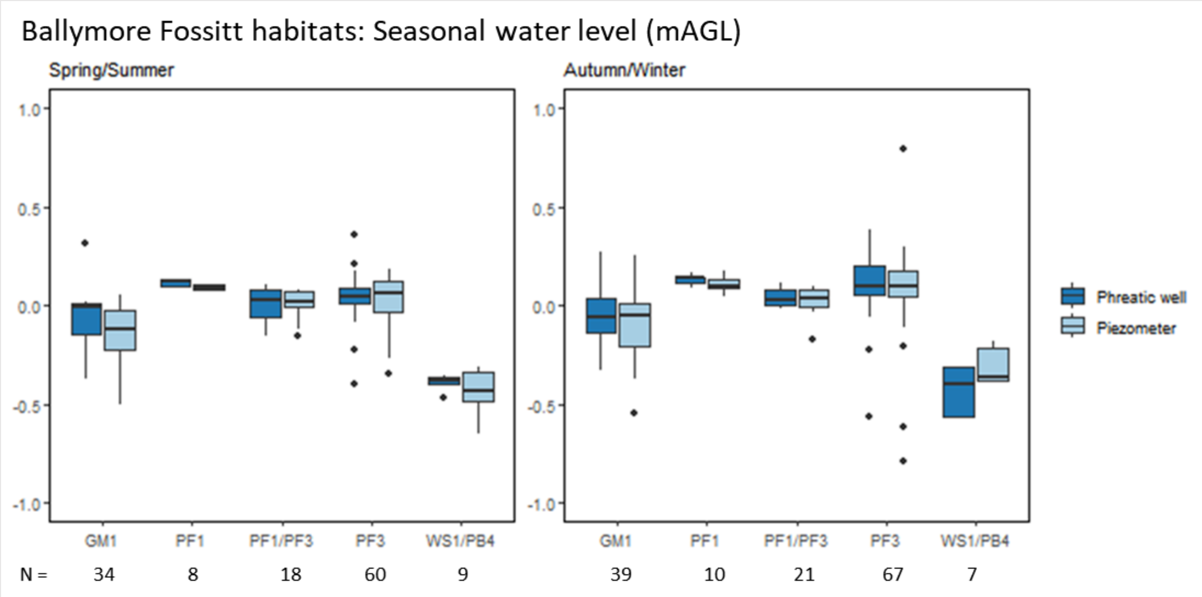


Figure 5.34. Seasonal water level in meters above ground level in the different habitats of Ballymore fen measured in phreatic tubes and piezometers.

5.3.1.2. Frequency duration curves

Frequency duration curves of surface water level, relative to the ground surface, are presented in Figure 5.35. These curves allow a better understanding of surface water level behaviour at each of the measurement points. The curves were made with the water level time series from data collected between October 2018 and October 2020.

Site BM12 with PF1 Rich fen and flush habitat has a very stable surface water table throughout the hydrological year not fluctuating more than 0.10 m and never falling below ground level. This implies that the habitat is supported when water levels are above ground level with minimal fluctuation. A similar pattern can be observed at BM162 although the water levels here are even 0.05 m higher for more than 50% of the time.

The fen habitats PF3 Transition mire and quaking bog and mosaic habitat PF1/PF3 in sites BM8, BM164 and BM41 seem to be supported with surface water levels above the ground surface for > 90% of the year. These levels also do not seem to fluctuate more than 0.1 m which indicates a relatively stable surface water table.

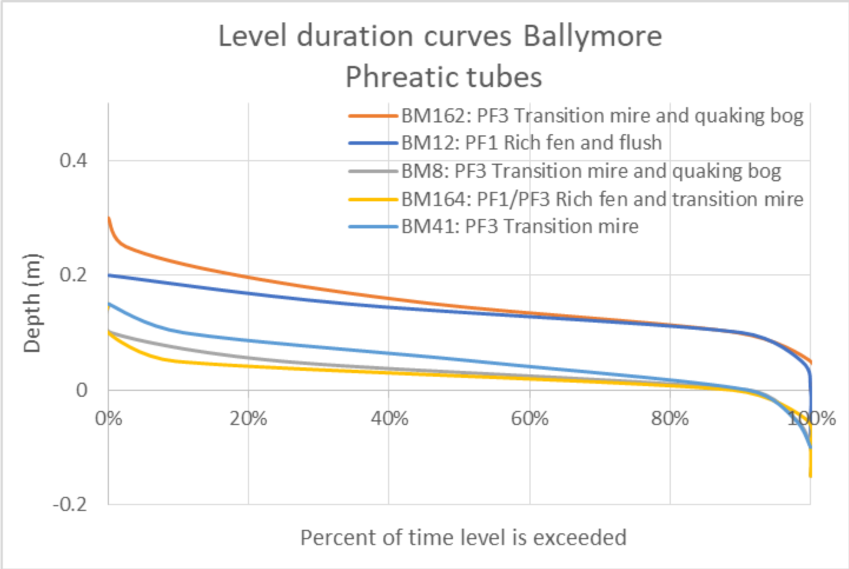


Figure 5.35. Phreatic level duration curves recorded in different habitats in Ballymore. The negative numbers are water levels below groundlevel.

From the phreatic and piezometric water level time series at BM12, BM8, BM41, BM162 and BM164 hydraulic gradients were calculated, as shown on Figure 5.36. Overall, the hydraulic gradients show very little fluctuations at the measured locations, suggesting that seasonal water fluxes (see Section 5.1.2) as well as effective rainfall do not play a big part in controlling the upwards or downwards flows in Ballymore fen. This furthermore implies that the hydrochemical fluctuations between the phreatic wells and the piezometers in the deeper layers of the fen are linked to internal decay processes and cycling of water (and dissolved chemicals) rather being a receptor of groundwater influxes. However, this does not imply that the fen is a closed system as hydrochemical characteristics may still be impacted by groundwater discharging into the fen, of which the flow impacts rates of elemental cycling.

There are, however, some minor fluctuations displayed in the graph. BM164 has a mosaic habitat of rich fen and transition mire and this location has the highest constant upward flows with hydraulic gradients around 0.3. Curiously, the hydraulic gradient increases during the summer of 2019. It could be that this site is receives more groundwater inputs during the summer from the nearby visible spring. Indeed, the fen overall receives a larger percentage of ground water during the summer of 2019 than during the winter, as is seen in the seasonal water balance (Table 5.2). Higher influxes of groundwater during the summer can be caused by high evapotranspiration from fen vegetation, as well as other factors such as higher microbiological activity (Wheeler et al., 2009). Cusell et al. (2013) has also reported higher groundwater inputs characterised by a high infiltration of bicarbonate-rich water.

BM8 with PF3 transition mire and seems to have a somewhat higher hydraulic gradient during the winter of 2018 and spring of 2019, after which time the gradient seems to drop during the summer. BM41 with PF3 Transition mire and quaking bog seem to be somewhat affected by a seasonal water level drop during the summer of 2020. Furthermore, a small decrease can be observed during the autumn of 2019.

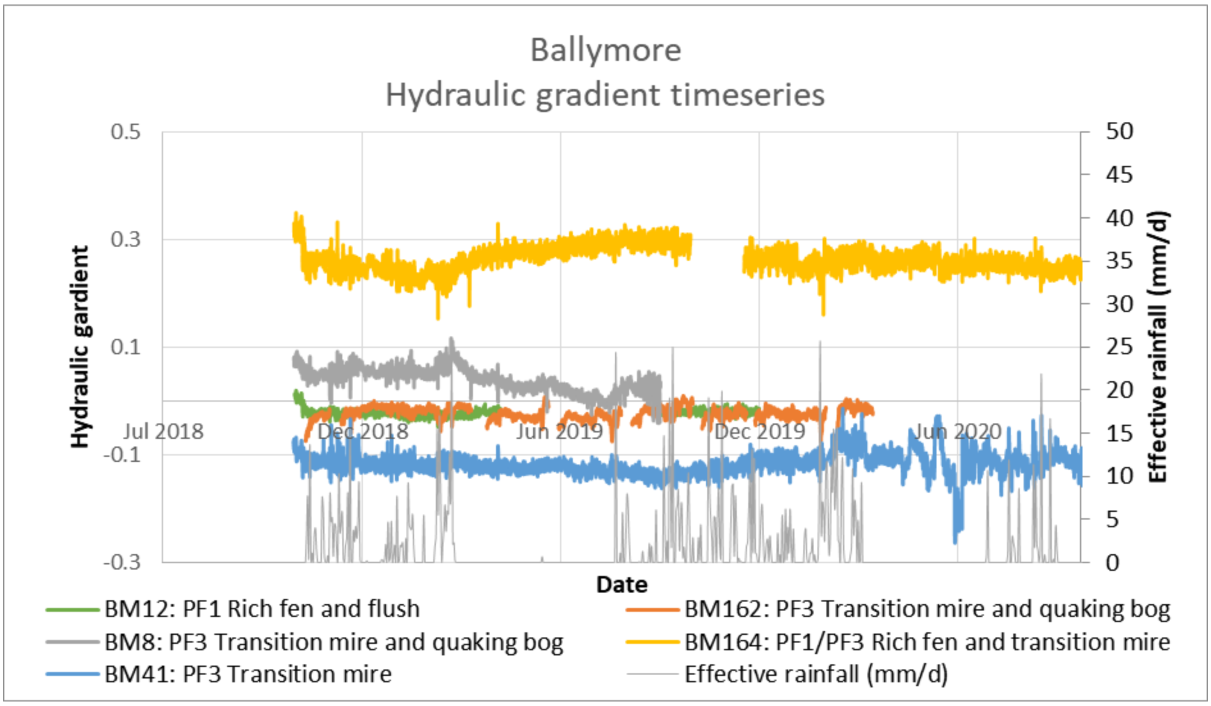


Figure 5.36. Hydraulic gradient timeseries calculated using the phreatic and piezometric water level timeseries in Ballymore. Effective rainfall is displayed here as well.

On the whole, upwards flows seem to occur at the north eastern edge of the fen (BM164) and near the mineral mound in the south (BM8). BM162 located in the middle of the fen and BM12 at the southwestern edge of Ballymore have small downwards flow throughout the hydrological year. Stronger downwards flows seem to occur near the eastern edge of the fen in BM41.

In order to see the overall change in hydraulic gradient the data was also plotted as frequency durations curves as seen in Figure 5.37. Here can be seen that hydraulic gradient levels do not fluctuate more than 0.05 at the time of measuring. BM12 with PF1 Rich fen and flush and BM162 with PF3 Transition mire and quaking bog have the most stable hydraulic gradient.

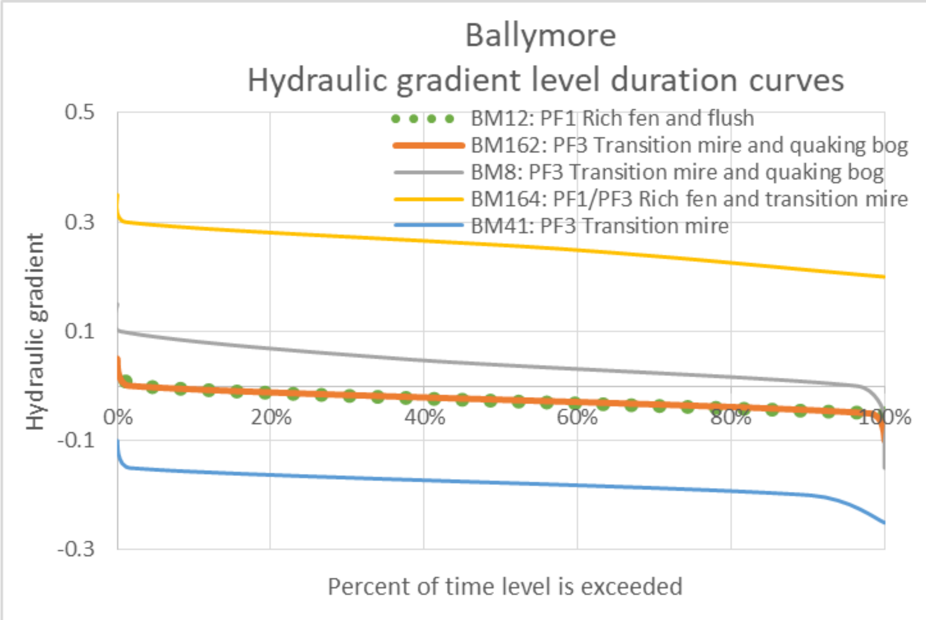


Figure 5.37. Level duration curves of the hydraulic gradients calculated from the water level time series in monitored phreatic tubes and piezometers.

5.3.2. Hydrochemistry and fen habitat

5.3.2.1. Nonmetric Multidimensional Scaling ordination

A Nonmetric Multidimensional Scaling ordination (NMDS) using ecological and hydrochemical data was plotted in order to find environmental vectors that have some form of correlation with the sampled locations and their specific habitat including the vegetation species that were surveyed. Similar approaches of NMDS were performed by Ahmad et al. (2020) for fens and Waldren (2015). For this two data sets were used. The IVC (ecological) data set contained the recorded species percentage at each surveyed relevé. This set contained 11 relevés and 44 species. The environmental set (ENV) consisted out of vegetation type cover (%), Fossitt habitat codes and the hydrochemistry results (mg/l).

The first NMDS plot was generated with the environmental variables vegetation cover (%) and the presence of the Fossitt habitats in a biplot (Figure 5.38). They are displayed as vectors and are plotted on top of a scatterplot with the surveyed relevés and species with the highest abundances. After 100 randomised runs, the reported stress was low (0.092) which is expected of a small dataset.

The surface water cover score was highly negatively correlated with axis NMDS1. This indicates that the clusters towards the negative end of Axis 2 are associated with a higher cover of surface water. Only relevé BM161 seems to be associated with this, however none of the species with the highest abundance seem to be associated with a high percentage of surface water. However, the Fossitt habitat PF3 Transition mire and quaking bog is also highly negative on axis NMDS1 meaning that relevés with that habitat experienced higher coverage of surface water. On the positive end of the NMDS1 and NMDS 2 axes a high correlation is shown with herb cover score and vegetation height. It therefore suggests that relevés recorded with a high cover of herbs and high stands of vegetation are associated with the habitat GM1 Marsh. It also suggests that this habitat is not associated with PF3 and high surface water cover. Furthermore, on the NMDS1 axis, it seems that *Filipendula Ulmaria* is highly positively correlated with GM1.

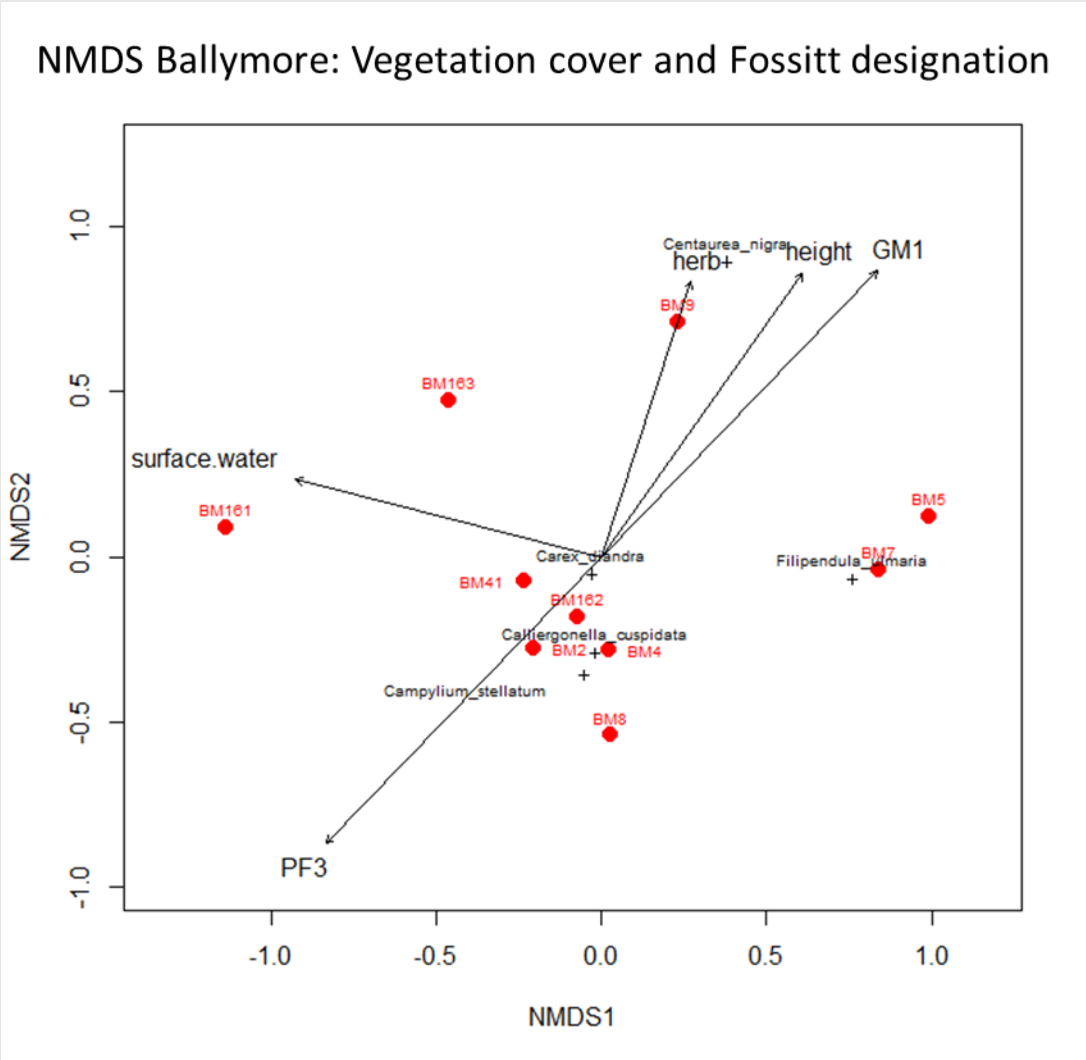


Figure 5.38. Multidimensional Scaling ordination of dimensions 1 and 2 with vegetation cover and Fossitt habitats plotted as vectors (max p-value = 0.2) in Ballymore fen. The phreatic and piezometer nest locations are shown in red and the names of the species with the highest abundances (10%) are also plotted.

Another NMDS was run with hydrochemistry results from April and June 2019 as environmental variables since this time is deemed to be the growing season of the fen, the presumption being that the nutrients and minerals present during this season would have the most impact on the fen vegetation. Unfortunately, it was not possible to plot the data of April 2019 since some data was missing. This resulted in an insufficient amount of permutations, which made it impossible to plot the hydrochemistry data as vectors on the plot. However, the June 2019 plot was generated successfully, as seen in Figure 5.39, with a loss reported stress of 0.093. Observed is a positive correlation of total oxidised nitrogen with habitat PF3 (when compared to Figure 5.36), meaning that higher concentrations are expected in the surface water here. Lower concentrations are expected in GM1 since they were negatively correlated. Furthermore chloride, ferrous iron and total dissolved nitrogen seem correlated with GM1. Higher concentrations seem to be expected here compared to PF3, where no correlation was detected.

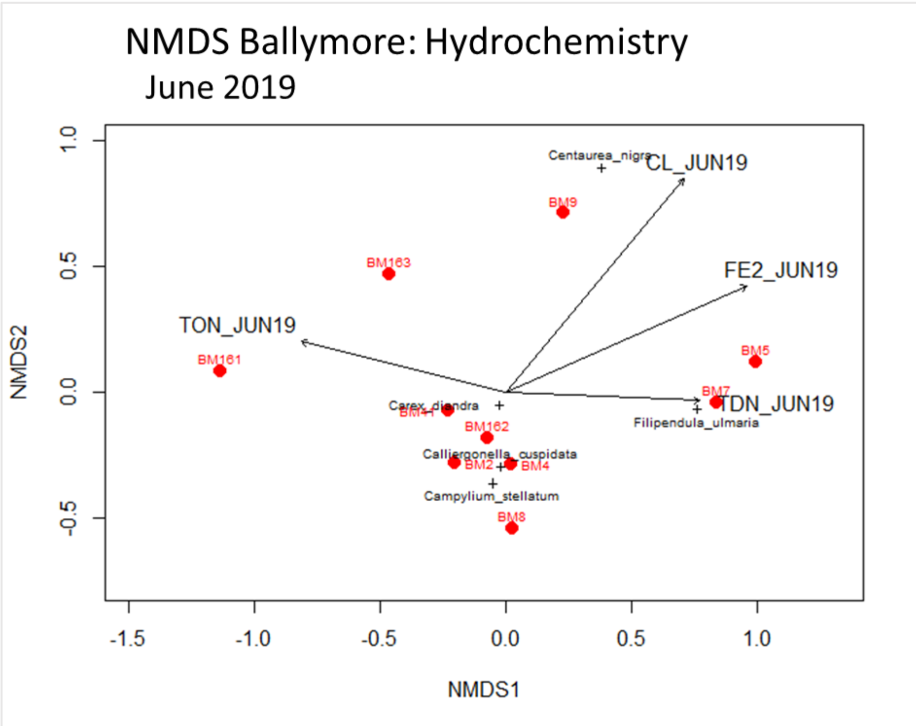


Figure 5.39. Multidimensional Scaling ordination of dimensions 1 and 2 with hydrochemistry concentrations plotted as vectors (max p-value = 0.2) in Ballymore fen. The phreatic and piezometer nest locations are shown in red and the names of the species with the highest abundances (10%) are also plotted.

5.3.2.2. Boxplots hydrochemistry

To investigate the correlations in Figure 5.38 and 5.39 boxplots were generated of the hydrochemistry results with respect to the different habitats. This was done in order to find out if the correlations occurring in the NMDS plots during the growing season can be found in the specified habitats when looking at the complete dataset. The boxplots were further divided by phreatic tubes and piezometers to see if the correlation occur in surface or down in the underlying sediments.

Total oxidised nitrogen was positively correlated with habitat PF3, meaning that higher concentrations are expected in the surface water here. In comparison GM1 was not correlated, as expected with the low values. However, this is not seen in the boxplot in Figure 5.40. Here the

Spring/Summer medians for phreatic tubes are 0.006 mg-N/l in habitat PF3 and 0.016 mg-N/l for GM1 which is opposite to what was expected.

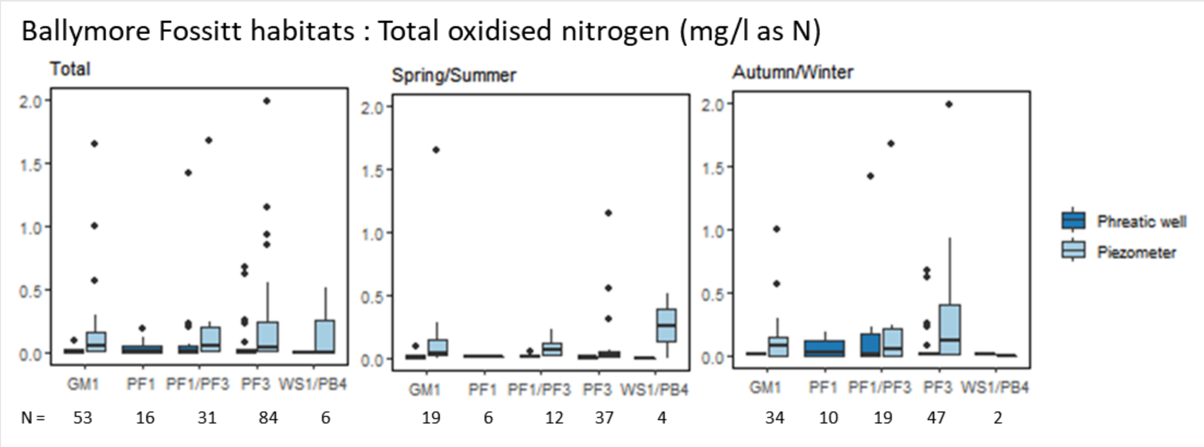


Figure 5.40. Total oxidised nitrogen in mg-N/l in the different habitats of Ballymore sampled from phreatic tubes and piezometers.

The same seems to be happening in Figures 5.41, 5.42 and 5.43, where it was expected that higher concentrations of chloride, ferrous iron and total dissolved nitrogen were correlated with habitat GM1 and lower concentrations with PF3. However, none of these hydrochemical parameters seemed to show this relationship in the boxplots. In fact, the concentrations of these parameters between PF3 and GM1 were found to be very similar.

What this seems to suggest is that spot measurements of hydrochemical parameters during the growing season (used in for generating NMDS plots in Figures 5.36 and 5.37) cannot necessarily be used as a representative for different fen habitats. In this case, this is probably due to the fact that the dataset being used for analysis is too small. Another possible reason for this lack in representability is that the interrelationship between fen habitats and water chemistry is much more complicated that can be revealed by such simple comparisons used to generate these plots. Therefore, more targeted studies are needed at fen sites to really understand the water quality dynamics.

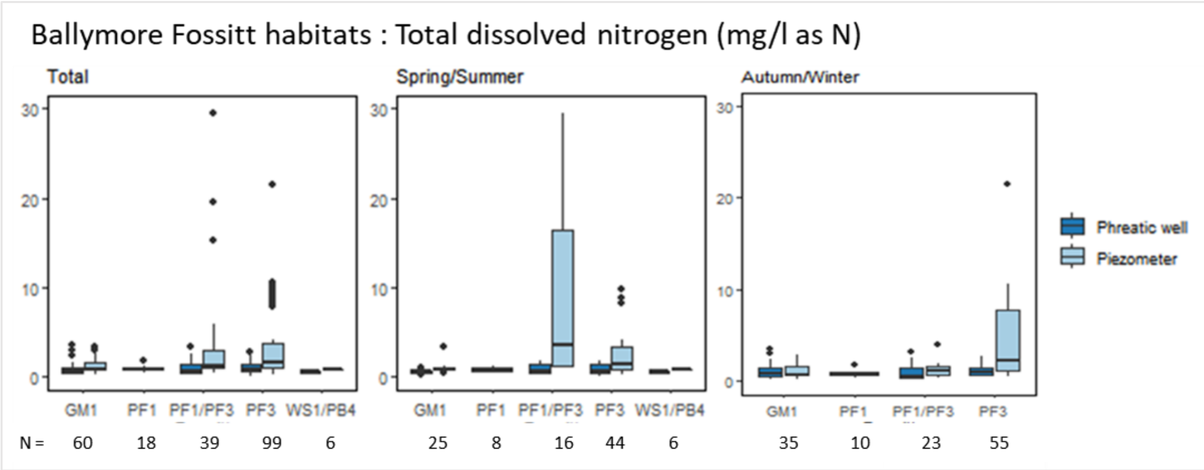


Figure 5.41. Total dissolved nitrogen in mg-N/l in the different habitats of Ballymore sampled from phreatic tubes and piezometers.

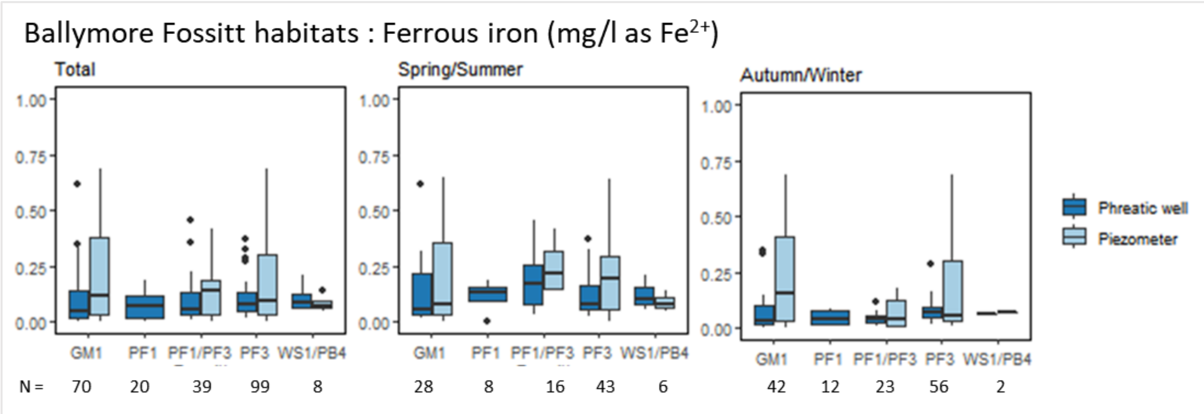


Figure 5.42. Ferrous iron in mg/l as Fe^{2+} in the different habitats of Ballymore sampled from phreatic tubes and piezometers.

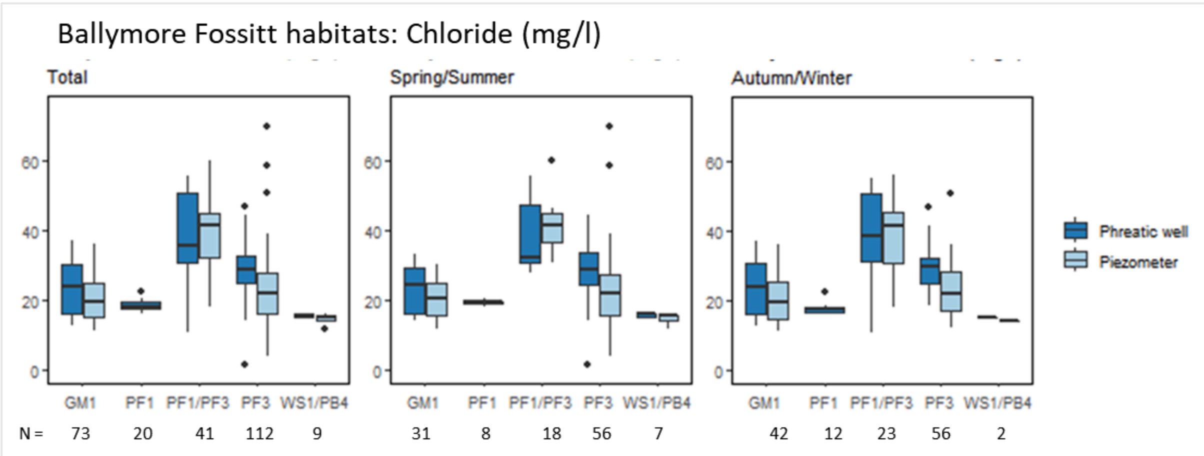


Figure 5.43. Chloride in mg/l in the different habitats of Ballymore sampled from phreatic tubes and piezometers.

5.3.3. Seasonal hydraulic gradients and hydrochemistry

The following sections bring the knowledge of the hydrology and hydrochemistry together on Ballymore transects 1 and 2. By displaying the gathered information in this manner, hydraulic flow paths can be recognised that drive the distribution of the hydrochemistry with respect to its geomorphological setting. Each transect has the soil geology displayed as well. The legend of these soils can found in Figure 5.44.

Figure 5.44. Legend of the different soils found in Ballymore transect 1 and 2

5.3.3.1. Dissolved reactive phosphorus

Figure 5.45 displays Ballymore transect 1 with data collected in August 2019. In summer, almost all hydraulic gradients are slightly downward, which enables the aerobic surface water to move down into the underlying sediments. The effect is the strongest in the middle of the fen. Relatively high dissolved reactive phosphorus is present in both the surface and the underlying sediments.

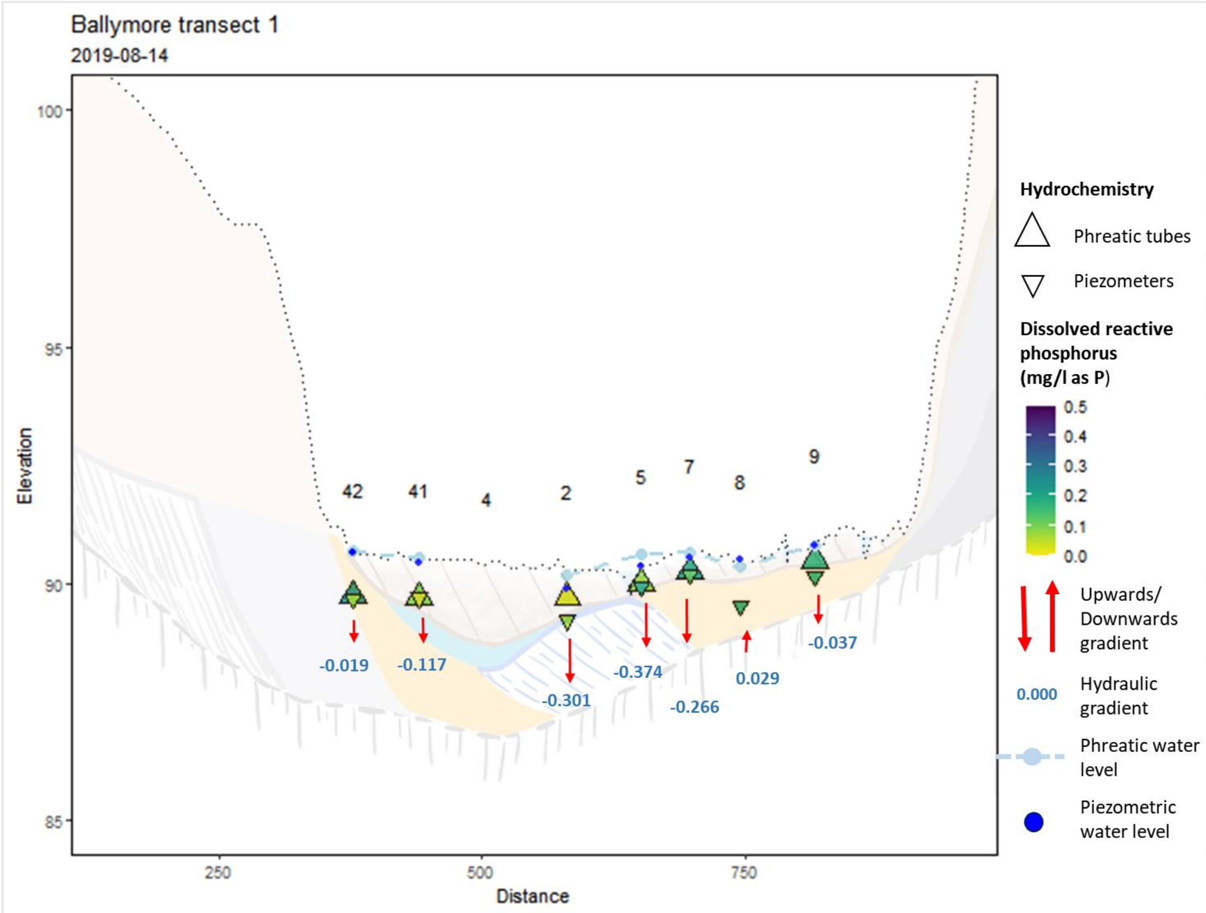


Figure 5.45. Hydrology and dissolved reactive phosphorus (mg-P/l) of Ballymore transect 1 in August 2019. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

A winter version of Ballymore transect 1 with data collected in February 2020 is displayed in Figure 5.46. From this can be observed that the mostly downward flows during August 2019 have changed significantly. Upward flows can be observed in the middle of the fen where the peat has an underlying soil shift from 'blue clay with sand and gravel' to 'sandy gravelly clay' Interestingly, the flows near the edges of the fen are still downwards, although less extreme than in the summer. These localised gradients seem contrary to the overall water balance which implies that the highest input of the regional groundwater was found during the summer. Therefore it seems that there exist localized inputs of upwelling groundwater combined with preferential pathways through peat strata have resulted in corresponding local circulation patterns in the subsurface.

The high DRP concentrations seem to have completely vanished from the free water table which is reflective of the higher amounts surface water contribution to the fen in winter which causes dilution of fen water with surface water.

Another explanation for this could be that the DRP has been rendered immobile by precipitation with metal ions such as Al^{3+} and Fe^{3+} (these ions are dominant in acidic soils) and Ca^{2+} (dominant in calcareous soils). It is possible that the redox chemistry processes that drive this precipitation are activated by the influx of low dissolved oxygen with a low redox potential from the deeper layers in the fen, which is proven by the upwards flow occurring the fen. Furthermore, the free water table in soil of Ballymore proved to contain high concentrations of alkalinity of which the ion Ca^{2+} is part of. This occurrence is likely due to clays at the bottom of the peat basin. Ferrous iron (which is subsequently an indicator for high Fe^{3+}) was also found with elevated concentrations. In contrast, phosphorus may be released to the free water table. This could be enabled by more oxygen rich water with a higher redox potential as seen in Figure 5.46. Indeed the effect of redox potential on the immobilisation and release of phosphorus was affirmed by studies such as Wheeler and Proctor, 2000; Cusell et al., 2013. Ann et al. (1999) also reports that water-table fluctuations and variable hydraulic loading rates in wetlands can alter soil redox conditions, and the solubility of P compounds.

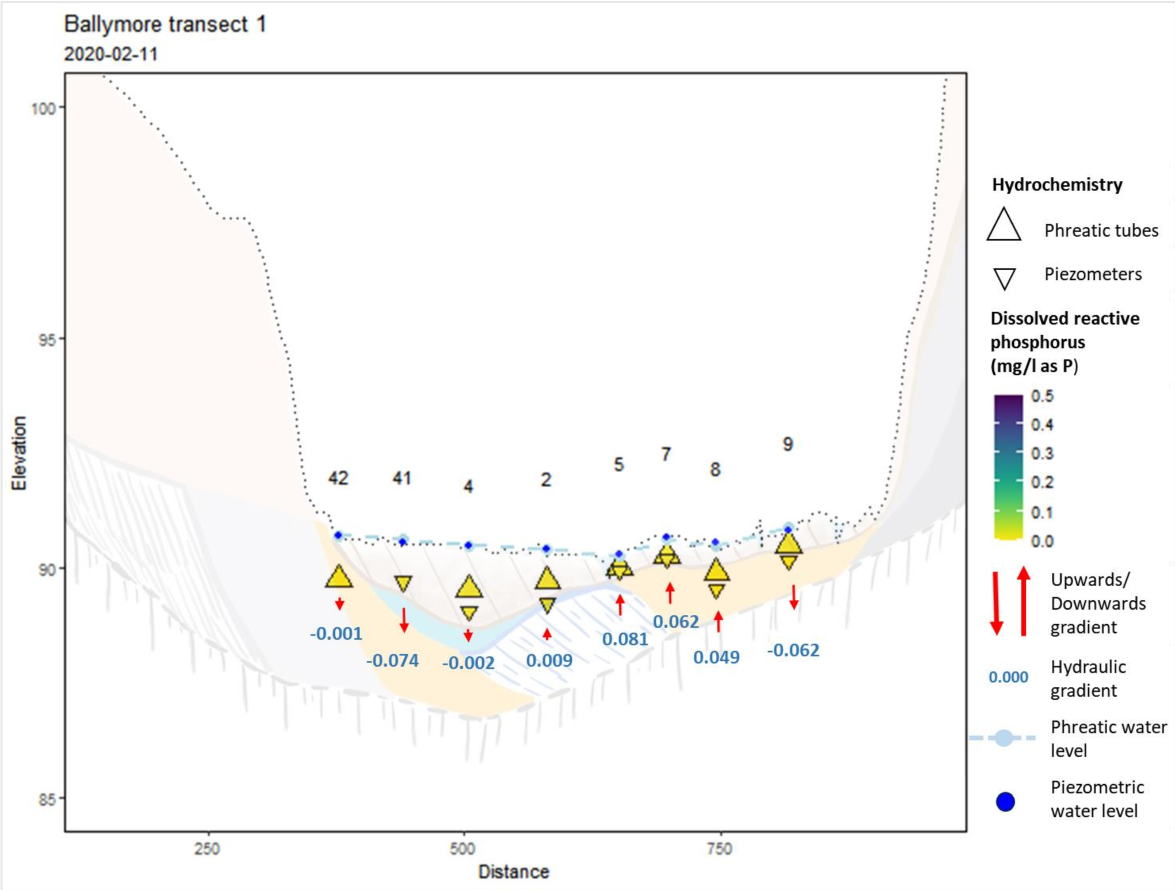


Figure 5.46. Hydrology and dissolved reactive phosphorus (mg-P/l) of Ballymore transect 1 in February 2020. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

This same effect with respect to DRP can be more or less observed in Ballymore transect 2. The difference here is that in some areas along the transect small upward flows can be observed during August 2019 (Figure 5.47). These upward flows are not directly linked to inputs from the groundwater catchment but rather from changes in hydraulic conductivity in the lower sediment layers which can cause small differences in respective heads at different depths. However, at location BM161 a strong downwards gradient was measured in the same geological conditions as transect 1. High DRP concentrations are also present throughout the transect.

Again, in February 2020 the DRP seems to drop down to very low levels in the free water table (Figure 5.48) although some higher concentrations still remain in deeper piezometers from which the screen is located in 'blue clay with sand and gravel'. However, some downwards flows are still present, especially at the right edge of the fen. There is also a very strong downward gradient observed in Brownes Upper; a borehole in the catchment outside the fen.

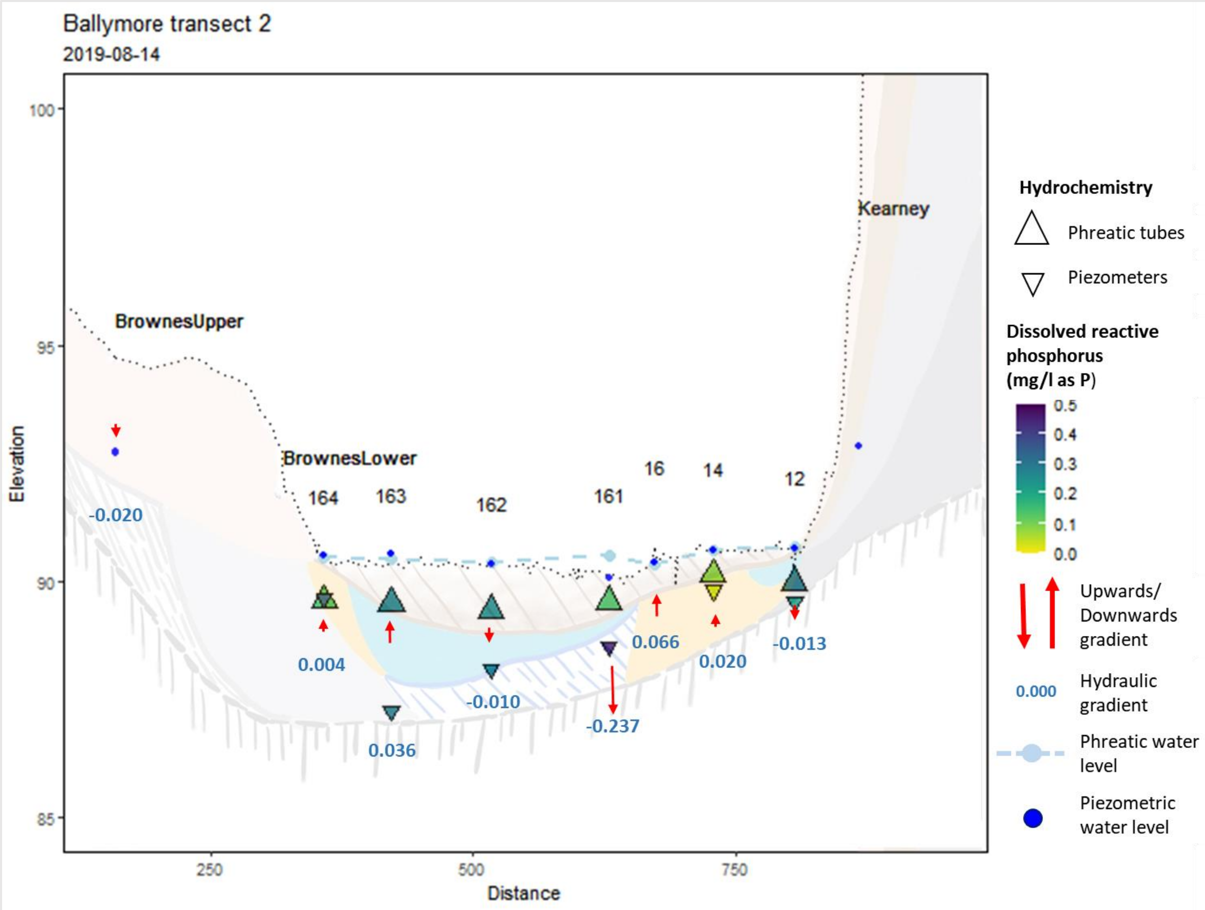


Figure 5.47. Hydrology and dissolved reactive phosphorus (mg-P/l) of Ballymore transect 2 in August 2019. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

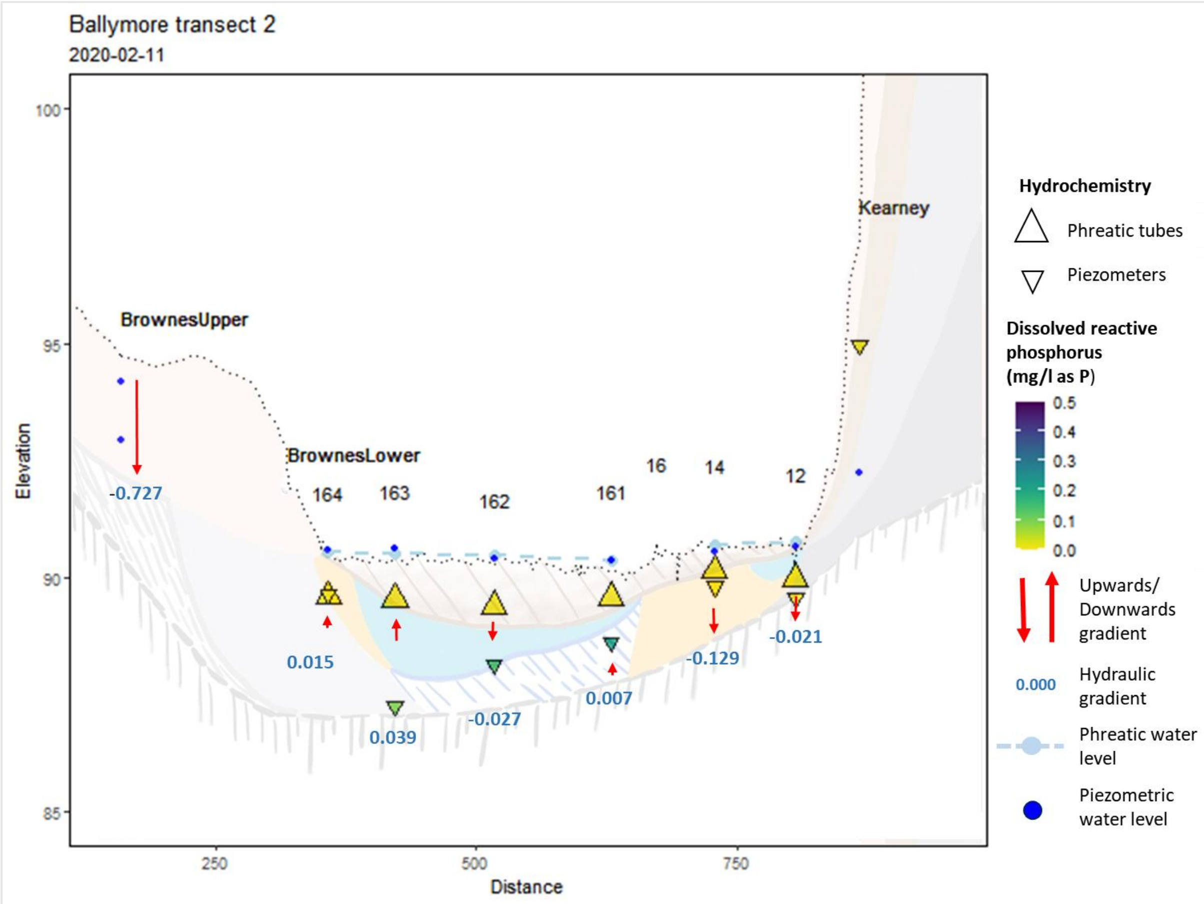


Figure 5.48. Hydrology and dissolved reactive phosphorus (mg-P/l) of Ballymore transect 2 in February 2020. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

5.3.3.2. Total ammonia

Very low levels of total ammonia were found across Ballymore transect 1 (Figure 5.49 and 5.50). The seasonal water level fluctuations as well as the changing hydraulic gradients do not seem to have any influence on the uptake or release of total ammonia in the soil. The locations BM42, BM8 and BM9 seem only slightly higher during August 2019. The same manner of elevation is also observed in locations BM41 BM2 and BM8. These slight concentration fluctuations depend on surrounding land use and furthermore may be caused by organic matter degradation.

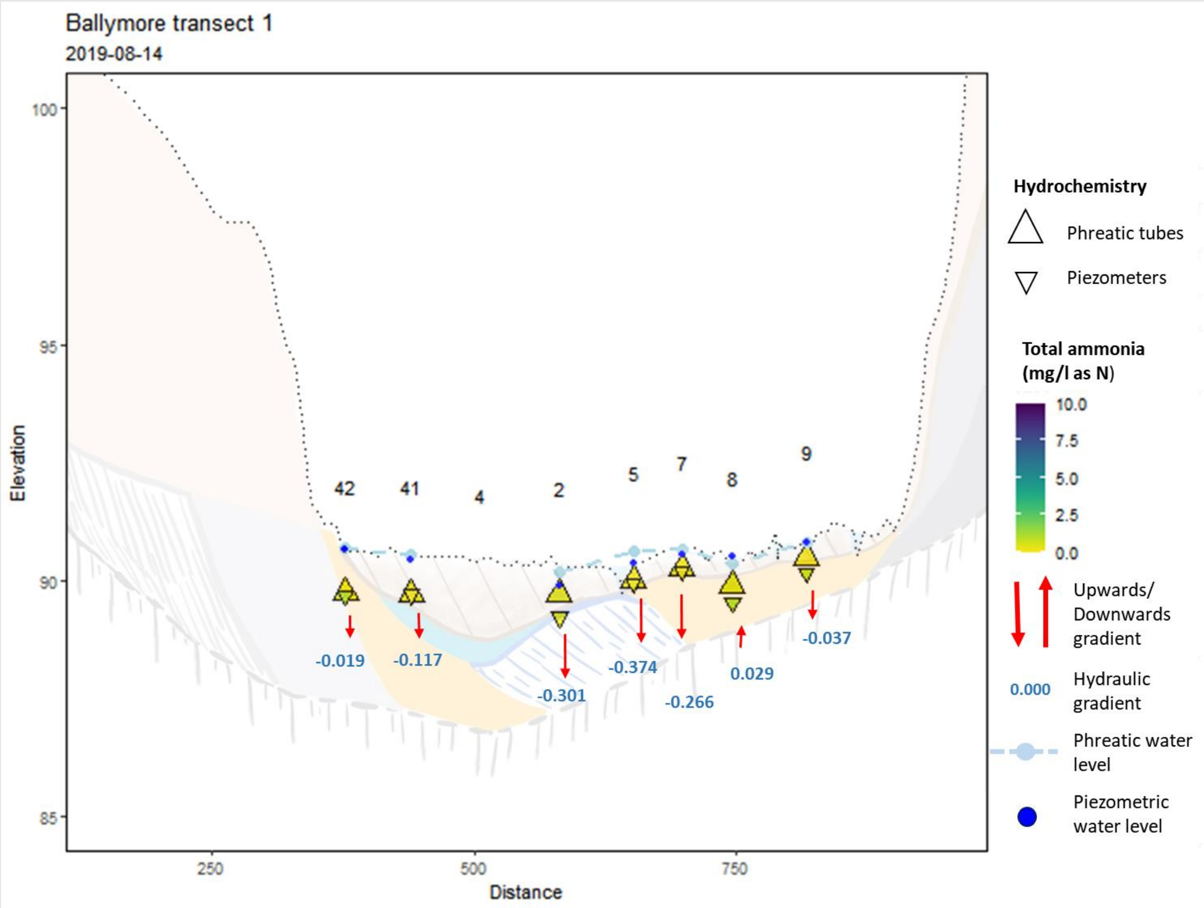


Figure 5.49. Hydrology and total ammonia (mg-N/l) of Ballymore transect 1 in August 2019. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

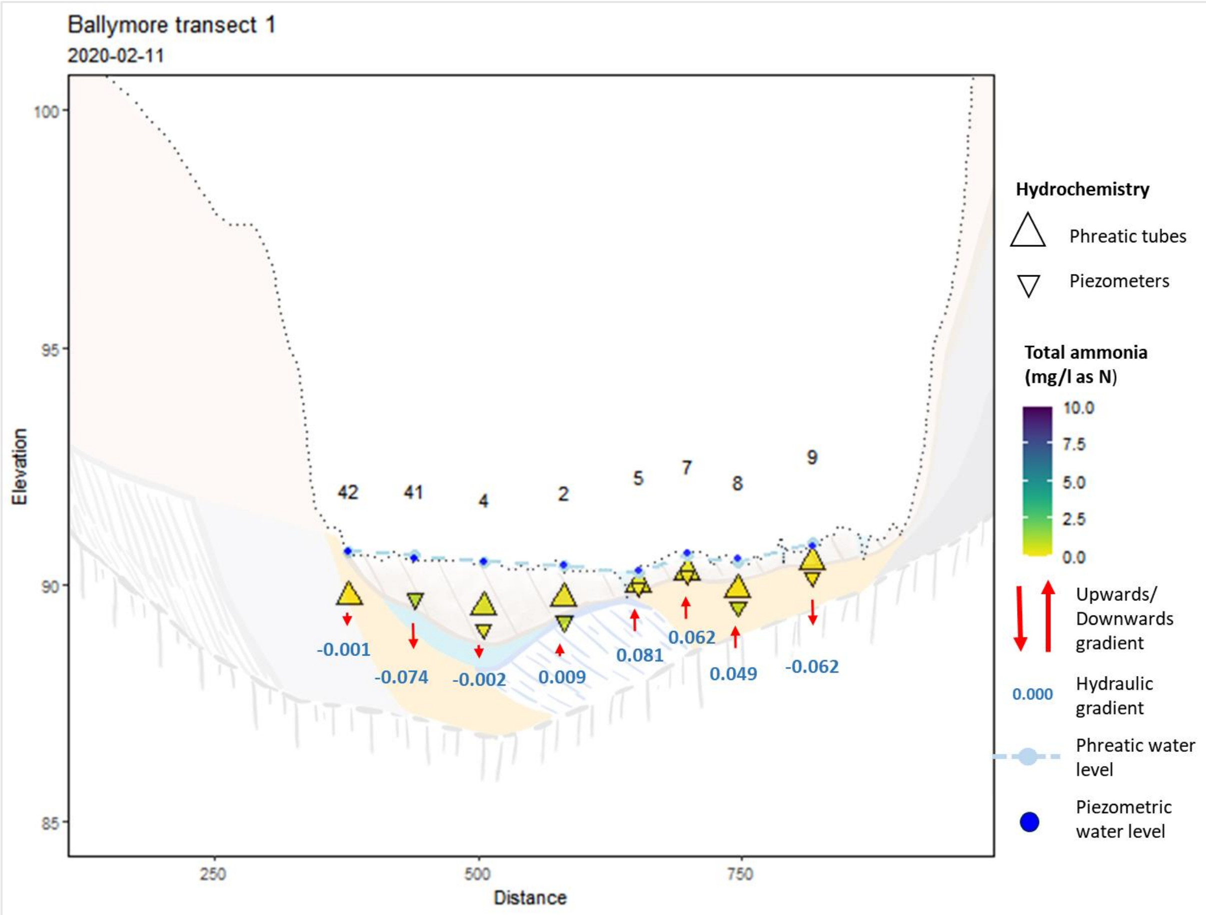


Figure 5.50. Hydrology and total ammonia (mg-N/l) of Ballymore transect 1 in February 2020. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

Figure 5.51 shows elevated total ammonia concentrations in the deeper layers of the fen at the left hand side of Transect 2. This higher ammonia concentration also seems to be reflected in the borehole outside the fen. This could point to groundwater flows with high total ammonia reaching the fen area at the left edge. Indeed, the piezometer at location BM164 did show increased concentrations. This elevated total ammonia flux does not seem to affect the free water table in the fen peat much which is contrast to the DRP concentrations in Figure 5.47 where high concentrations are found throughout the whole fen in the summer. Although N-P ratios could not be determined since phosphorus was measured as Total Phosphorus and nitrogen as TDN/Total Dissolved Nitrogen, it is suggested that while the vegetation might have taken up most of the ammonia in the growing season, much of the phosphorus was left in the soil water. Such an effect would suggest that Ballymore fen is limited by nitrogen rather than phosphorus.

There is a slightly elevated concentration found in location BM163 but it does not seem to travel further to the right as no total ammonia is found in the peat layer there. Instead, elevated total ammonia is rather found in the deeper layers of the fen with soil consisting of 'marl' underlain by 'blue clay with sand and gravel'. The deeper layers of the fen therefore show evidence of ammonification where ammonium is produced from particulate and dissolved organic material (Bedford & Godwin, 2003; Davidson et al., 2002; Kuczyńska 2008)

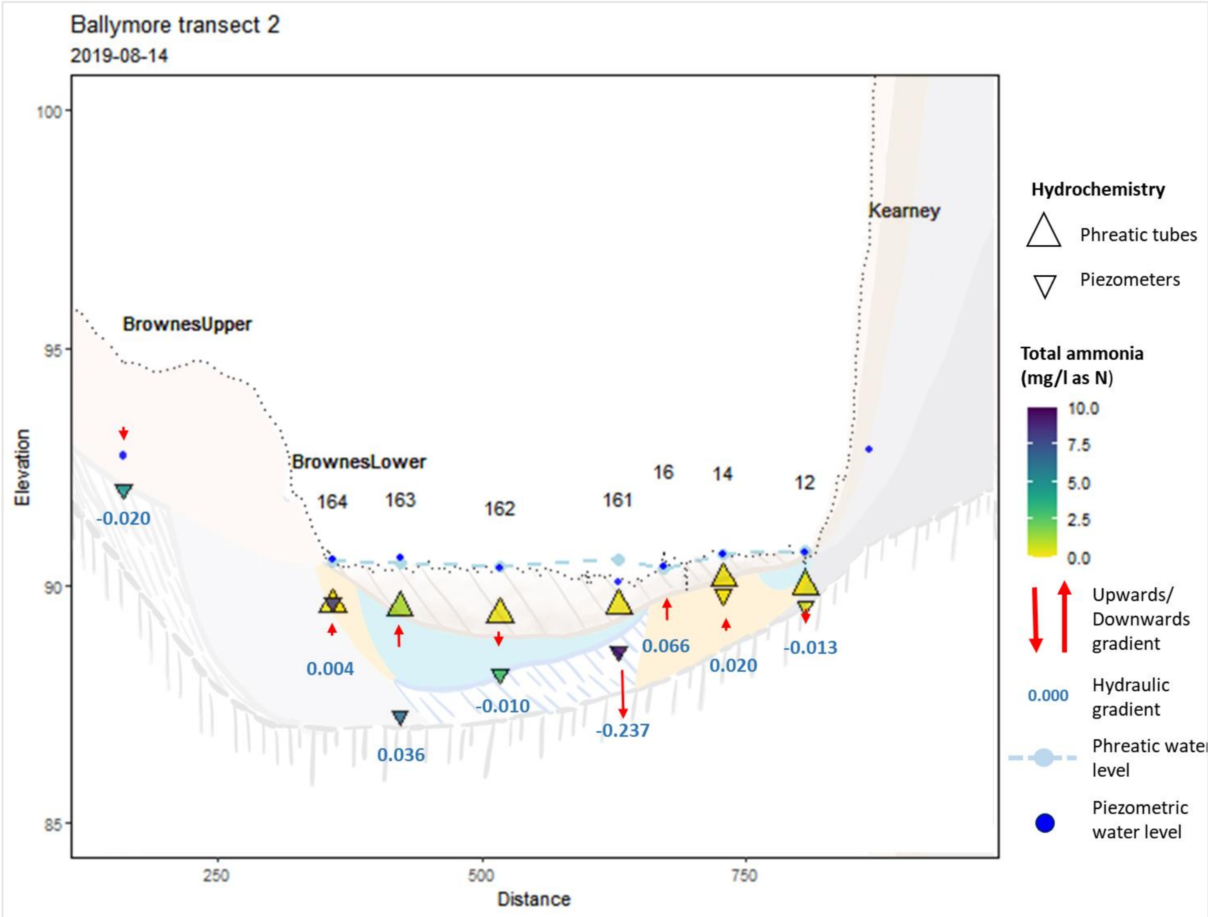


Figure 5.51. Hydrology and total ammonia (mg-N/l) of Ballymore transect 2 in August 2019. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

The locations with elevated total ammonia in August 2019 seem to have decreased during the following winter as seen in the transect in Figure 5.52 displaying data collected in February 2020. Interestingly this is a similar trend as seen in DRP concentrations further confirming increased dilution of fen water with surface water.

Elevated concentrations of ammonia were still found in in the piezometers measured in the soil layer 'blue clay with sand and gravel' which was also the case with the DRP concentrations. It therefore seems that this particular soil layer has quite low permeability which in turn does slow down the exchange of nutrients into soil solution. Long term this means that these types of soil layers are good receptors for nutrient sinks provided water levels do not greatly fluctuate changing the soil from saturated to unsaturated in short periods of time.

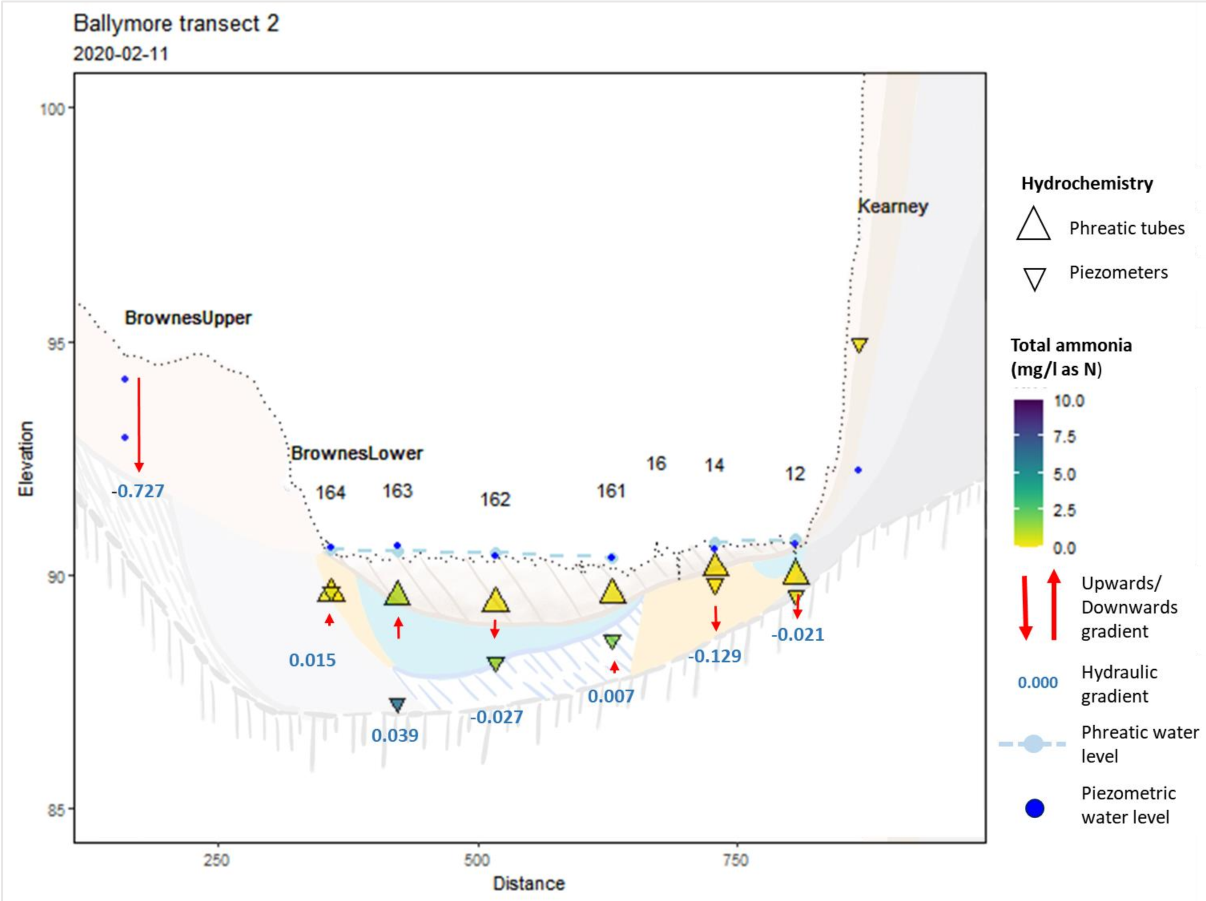


Figure 5.52. Hydrology and total ammonia (mg-N/l) of Ballymore transect 2 in February 2020. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

5.4. Conceptual model

5.4.1. Site summary

Ballymore fen spans 0.23 km^2 and is supported by a surrounding catchment of 0.76 km^2. It contains a wide array from fen and bog habitat to marsh and scrub. By combining designated fen habitats Rich fen and flush (PF1), Poor fen and flush (PF2) and Transition mire and quaking bog (PF3) it can be calculated how much fen the SAC supports. The fen supports a total of 0.12 km^2 of designated fen habitat which is 53% of the entire site. From the seven assessed relevés conducted during the vegetation survey as specified in Section 4.1.3 only one failed the fen assessment criteria in Appendix D, which proves that the site supports overall good quality fen vegetation.

The fen receives both groundwater and surface water in a hydrological year, however the proportions change seasonally. The fen received the greatest relative proportion of groundwater during the summer, while in winter the greatest relative proportion is from surface water. Regional groundwater rates were overall higher in the winter but were relatively lower compared to the surface flow inputs at that time of year.

Even though the discharge from the fen's outlet can change the water balance drastically, this does not seem to have a significant effect on the surface water levels showing the resilience of the fen. Hydraulic gradients in the fen show minimal fluctuations between the phreatic wells and the piezometers in the deeper layers of the fen peat which points to internal cycling of water (and dissolved chemicals) in the fen. There is, however, a clear difference between summer and winter which suggest that the internal cycling is influenced by seasonal changes.

Ballymore fen further seems to act as a treatment system for the incoming nutrients in the groundwater. These nutrients are taken-up by fen vegetation and then subject to internal nutrient cycling in terms of annual die back and decay in the sediments which then results in release of these nutrients, followed by further uptake through which the cycle continues. This is further proven by significantly higher concentrations of DRP, TP and ammonia found in the catchment rather than in the fen. Nutrient concentrations may also be diluted by surface water and flushed out of the fen via its outlet as was found in significant decreased DRP concentrations in the winter (Figure 5.43 to 5.46).

5.4.2. Conceptual model

A conceptual box model is displayed in Figure 5.53, showing the water balance, surface water level fluctuation and median nutrient concentrations in the fen and its catchment.

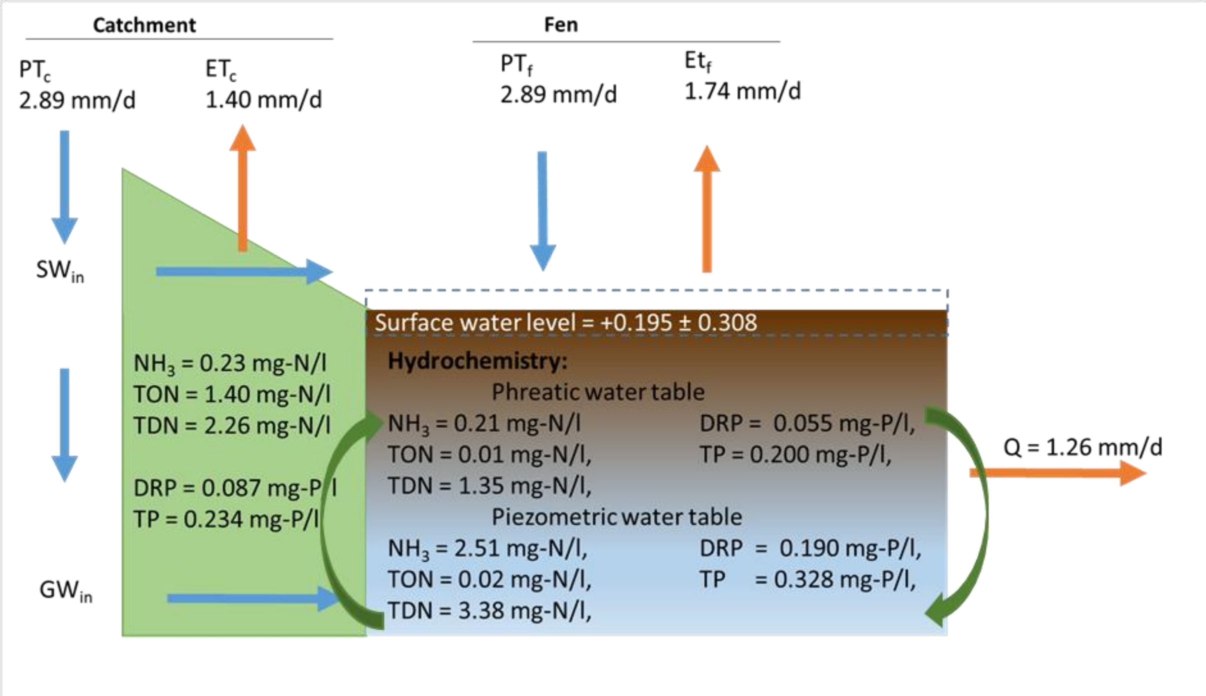


Figure 5.53. Conceptual box model of Ballymore displaying the water balance, surface water level fluctuation and median nutrient concentrations in the fen and its catchment.

6. Results – Pollardstown site A

6.1. Hydrology

Since Pollardstown site A is only a small part of the whole of Pollardstown Fen a previously estimated water balance is presented based on the findings by Kuczyńska (2008). Furthermore, the runoff from the site as well as the piezometric and phreatic water levels will be compared with data previously collected in order to detect any changes. This dataset was made available from the data collected during the research undertaken for the construction of the Kildare bypass (Kildare County Council, 2003).

6.1.1. Pollardstown Fen annual water balance

Kuczyńska (2008) established that the total catchment area of Pollardstown fen is 32.2 km^2 (with a fen area of 2.7 km^2) and can be divided in to two sub catchments based on the runoff to two outlets: Milltown Stream and Milltown Feeder which is the principal source for the Grand Canal. Pollardstown site A drains via the Milltown feeder and site D towards the Milltown stream but there is no clear separation of the runoff pathways. This catchment has a ground water divide that runs parallel to and above the Milltown-Newbridge road R416 and is fed by the major spring outflow at 7 Springs and marginal seepages and springs along the southern and northern fen margin (Kuczyńska 2008). The locations of the catchments and springs in the fen can be found in Figure 3.9).

Kuczyńska (2008) estimated the water balance by calculating the recharge into and discharge from the fen for the hydrological year 2003/04. The recharge feeding springs, seepages with the effective rainfall into the fen was found to be 11.18 x 10^6 m^3/y. The discharge from the fen in the Milltown feeder was estimated to be between 8.6 and 11 x 10^6 m^3/y. According to these values the recharge estimates are higher than the discharge values. However a study by Misstear & Brown (2008) suggests that not all effective rainfall that falls over the Curragh aquifer contributes to the groundwater recharge in the fen catchment but rather that the proposed recharge coefficients range between 72-100%. Moreover, these coefficients account for recharge to regional groundwater flow that does not pass through the Milltown feeder outlet to the fen. The water balance calculation then falls well within this coefficient with a range of 77-98% when taking the lower and higher discharge estimates into account.

A more recent water balance for Pollardstown fen was calculated as specified in the methodology Section 4.1.7. The change in storage of the fen was deemed to be negligible as the water levels in the phreatic tubes didn't show any change (Figure 6.7 and 7.2). Furthermore, since some observations were missing from the discharge of Milltown Feeder measured at Hanged Man's

Arch (Figure 6.1), an average was taken measured between 1 October 2019 and 30 September 2020. The resultant water balance is displayed in Table 6.1.

Table 6.1. Water balance of a hydrological years 2018/19 and 2019/20 in Pollardstown using catchment area 33.2 km².

01-10-2019 to 30-09-2020			
	Total (m^3)	**Flux (mm/d)**	**Fraction of rainfall**
Rainfall on fen	2436560	2.50	
Rainfall on catchment	27058640	2.50	
Evapotranspiration from fen	1615790	1.66	5.5%
Evapotranspiration from catchment	14953148	1.38	50.7%
Runoff from fen	10564560	0.90	35.8%
Error in water balance	*-2361702*	*0.28*	***-8.0%***

By estimation, from the recorded rainfall between October 1th 2018 and September 30th 2019 approximately 5.5% was lost from the fen and 50.7% was lost from the surrounding catchment as evapotranspiration. Furthermore, 35.8 % was lost as run off via the Milltown feeder (shown in Figure 3.9). Evapotranspiration from the catchment sees the largest amount of water lost, in contrast to the water balances from Ballymore (Table 5.1) catchment suppling the fen is much larger in comparison.

To compare this water balance to the one estimated by Kuczyńska (2008) the recharge value was calculated by adding the rainfall of the fen and the catchment (29.50 x 10^6 m^3/y). The discharge was calculated by combining the values of evapotranspiration from fen and catchment as well as the runoff (27.10 x 10^6 m^3/y). Similarly, the recharge values are higher that the discharge values. They are, however, still acceptable as the recharge coefficient is 92%.

6.1.2. Pollardstown Fen runoff

The water in Pollardstown site A runs off via the southern tributary drain which joins other tributaries in the Milltown Feeder system. The combined runoff is measured at a water level monitoring station at Hanged Man's Arch in Milltown.

Unfortunately, the data available from the waterlevel.ie website (OPW, 2020) were not complete, as is seen in Figure 6.1. Here intervals of data are missing in several places between 1th October 2018 and 30th September 2020. Nevertheless, some observations can be made from these data. Discharge varies between 439 and 2372 m^3/hr over the two consecutive hydrological years. The average discharge during this period was 1101 m^3/hr. High sustained discharge events seem to occur during periods of high effective rainfall as seen in the winter of 2019 and spring of 2020. The peak discharge was recorded with 2235 m^3/hr during this time. After this period of high effective rainfall, the discharge decreased quite suddenly from around 1400 to 800 between March and

April 2020. This apparently steep discharge recession may reflect the ability of the drainage to rapidly remove direct fen rainfall, although controlled by the capacity of the canal to accept it. The total rainfall does seem to influence the discharge as is seen in high discharge events during the summer and autumn of 2019 with discharges up to 1440 m^3/hr. The summer of 2020 was even recorded with an extreme high peak discharge of 2372 m^3/hr. During the driest of the summer months, the fen runoff reduces although a minimum of 439 m^3/hr was recorded, probably due to groundwater base flow. Compared to Ballymore, this shows that Pollardstown is fed by a much larger groundwater catchment via an extensive network of springs and seepages in the catchment area allowing for higher discharge rates.

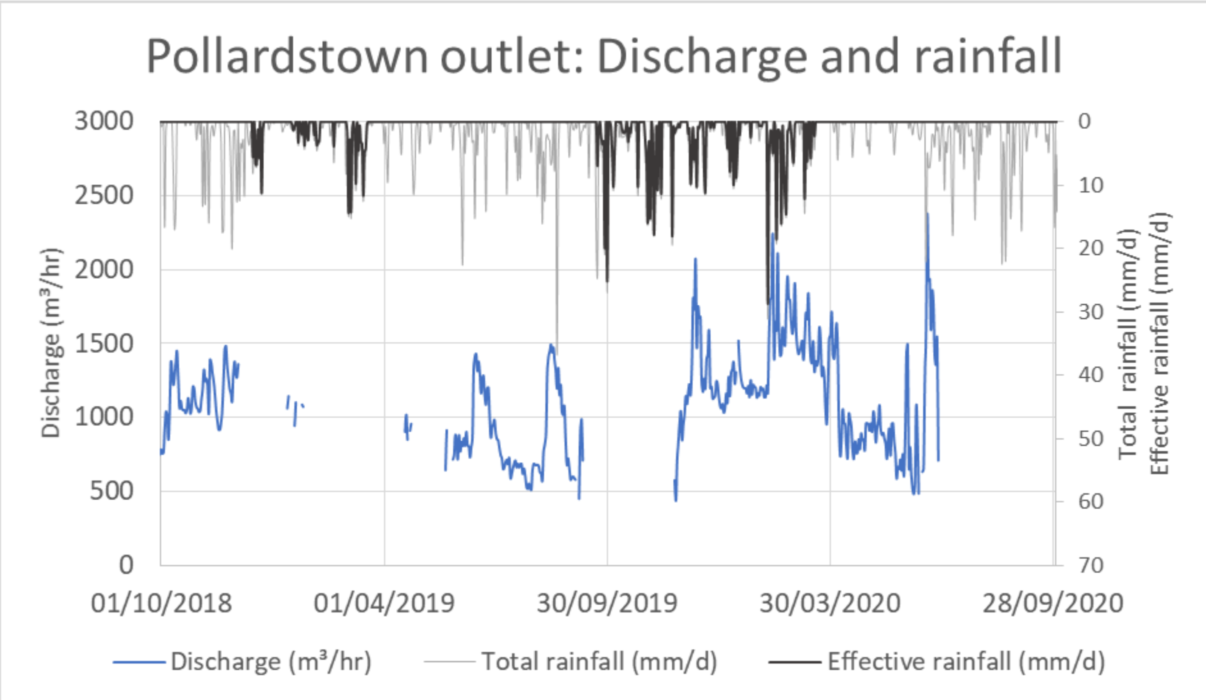


Figure 6.1. Pollardstown fen outlet hydrograph and total/effective rainfall between October 1th 2018 and September 30th 2020.

Kuczyńska (2008) reported discharge values of measurements collected between 1993 and 2005 at a monitoring station on the Milltown Feeder under the Hanged Man's Arch. The average discharge of 1135 is in close agreement to the average discharge measured during the hydrological years of 2018/19 and 2019/20 m^3/hr. Low discharges are also comparable to the current values with 576 m^3/hr. However, discharge peaks were originally recorded at a much lower rate of 1728m^3/hr instead of the more recent 2372m^3/hr. Although it has to be noted that these measurements were obtained by spot measurements in the past rather than the current continuous monitoring by the OPW using an in situ ultrasonic gauge.

A time-series of the total evapotranspiration against the discharge is displayed in Figure 6.2. The amount of discharge starts to decrease quite suddenly at the end of March in 2020 while the evapotranspiration is increasing. This trend seemed to continue until there a minimum runoff is reached during the summer. Then, when the actual evapotranspiration decreases at the end of the summer, the discharge increases again. There seems to be a sudden drop in total evaporation during the high discharge event in the summer of 2020. This phenomenon seems to coincide with a high rainfall event in Figure 6.1.

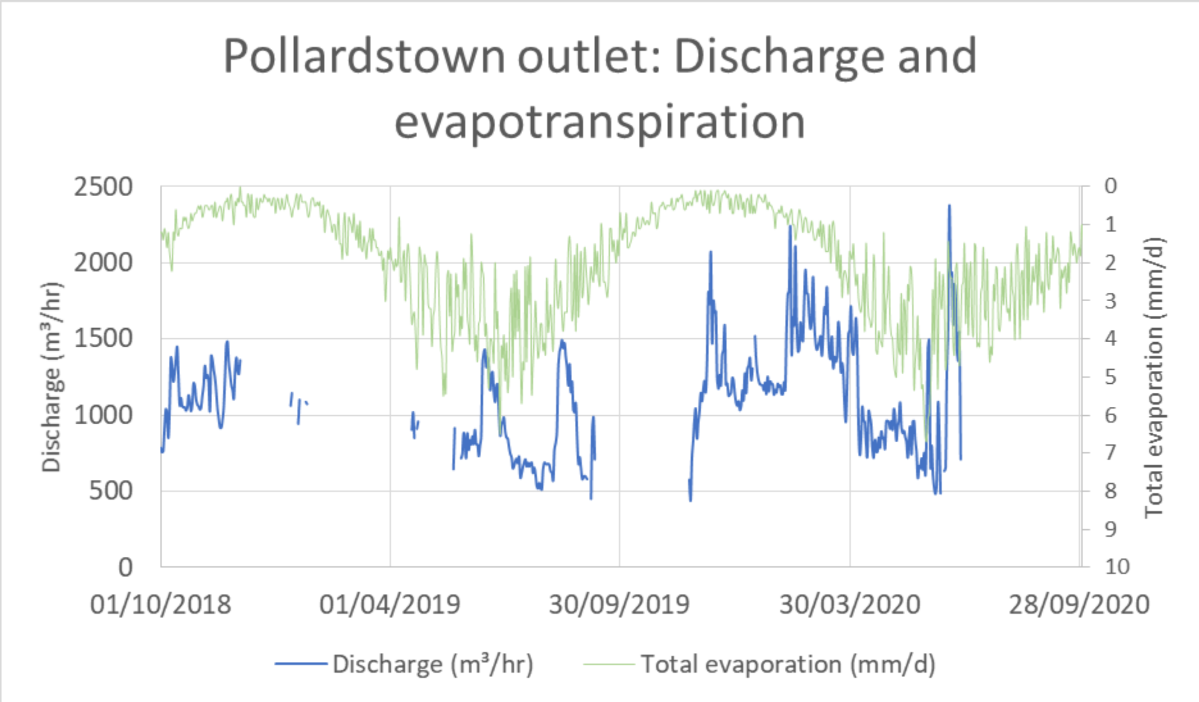


Figure 6.2. Pollardstown outlet hydrograph and actual ecapotranspiration between October 1th 2018 and September 30th 2020.

The effect of daily evapotranspiration on the water levels of the fen is found in Figure 6.3. In the 20 day hydrograph of phreatic tube 37 a diurnal water level fluctuation of around 1 cm can be observed per day. The waterlevel drops during the days when the vegetation roots are actively taking up and transpiring water and then the levels are more or less restored during the night (Ahmad et al. 2020; Frahm et al. 2010). This same diurnal pattern was seen at Ballymore fen but the fluctuation at Pollardstown does not seem to increase and decrease as gradually as it did in Ballymore. This could be the caused by a difference in peat structure as well as diffused overland flow from springs up-hill. Proof of diffused seepage can be seen in water levels after the 10th of April, where no rainfall is recorded yet there is a gradual increase of water levels. Another such occasion is recorded on the 20th of April where there is a rather rapid increase of 5 cm. Additionally, the direct rainfall does not seem to always have the same effect on the water levels. The rain event of the 7th of April has a delayed response whereas the effect of the rainfall on the 15th of April can be observed immideately. This response might be due to the peat becoming completely saturated during the initial rainfall allowing for an immediate water level increase during the second rainfall event.

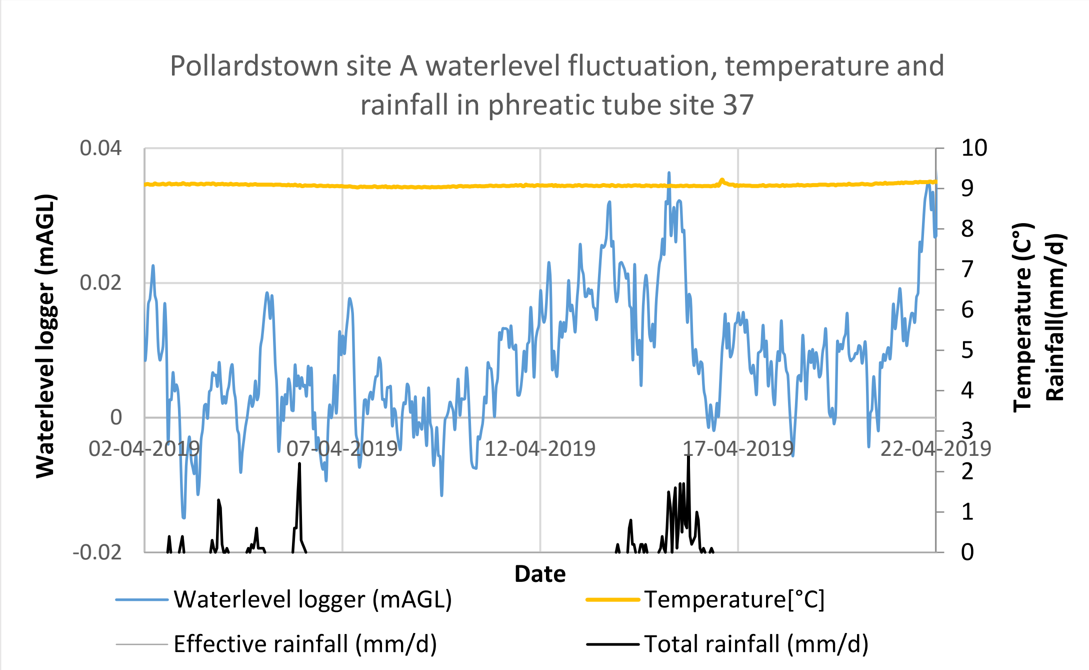


Figure 6.3. Pollardstown fen phreatic tube hydrograph temperature and rainfall between 2th April 2019 and 22th April 2019.

6.1.3. Piezometer and phreatic tube data

Surface water points in the fen and groundwater table points around the fen were interpolated into contour lines in order to interpret the flow in and out of the fen in Figures 6.5 and 6.6. Figures 6.7 and 6.8 show the water levels recorded in the phreatic tubes and piezometers between July 2018 and October 2020. The locations of the sites where this data was collected can be found in the Pollardstown site A instrumentation map in Figure 6.4.

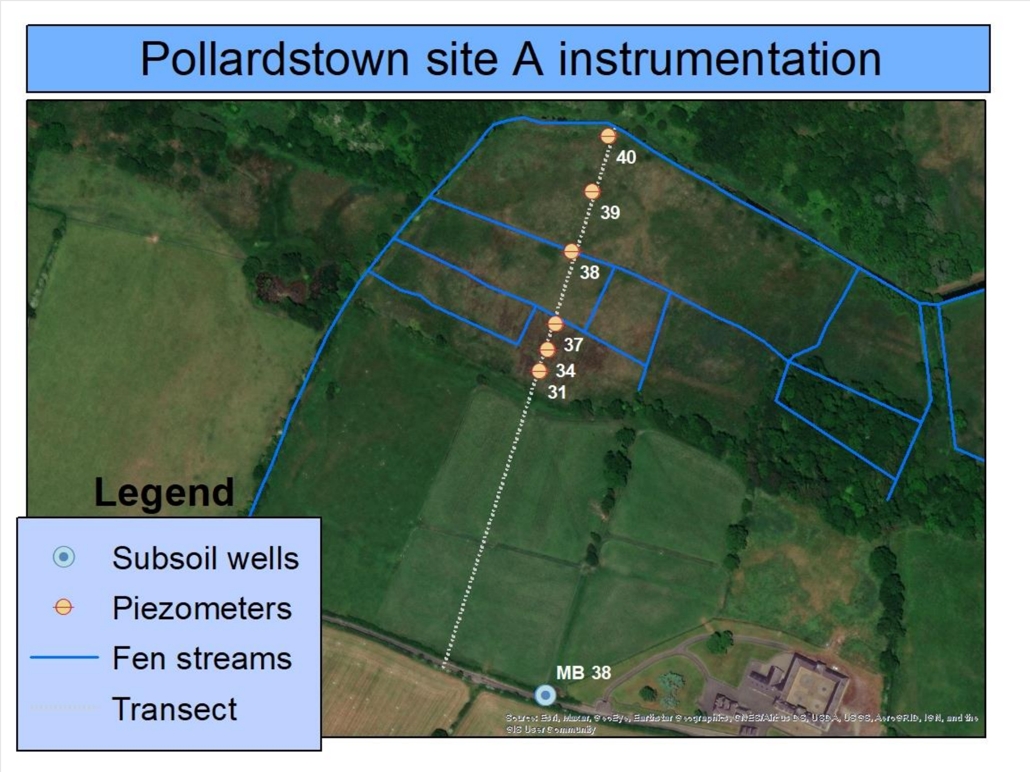


Figure 6.4. Pollardstown site A instrumentation map showing fen piezometer and phreatic tube locations, subsoil well locations and the main site drains.

Flow can be observed from the south to the north discharging into Milltown feeder during the summer (Figure 6.5) and the winter (Figure 6.6). During the winter the flow seems to shift somewhat with additional flows from the southwest. It is possible this happens because the water levels of the feeder are expected to be higher during the winter which saturates the peat from the left side resulting in higher levels in that area. Another more likely cause is that there is groundwater dispersed at the surface of the fen in discrete pathways which then moves laterally over the surface of the fen mixing with rainwater before it is discharged in the outlet. It has to be noted that these contours are a small subset of the the 'catchment' as shown on the map of boreholes in Section 3.3.

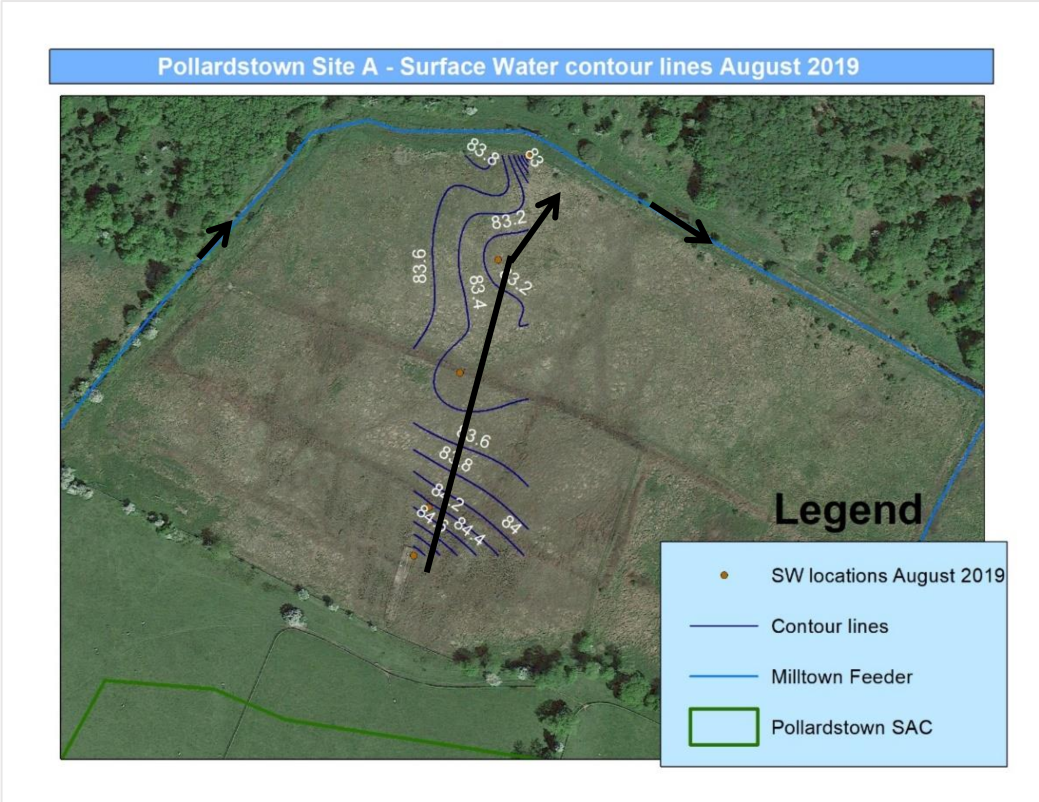


Figure 6.5. Contour lines of fen surface water and surrounding groundwater catchment interpolated using piont measurements in August 2019. Flowlines are presented with black arrows.

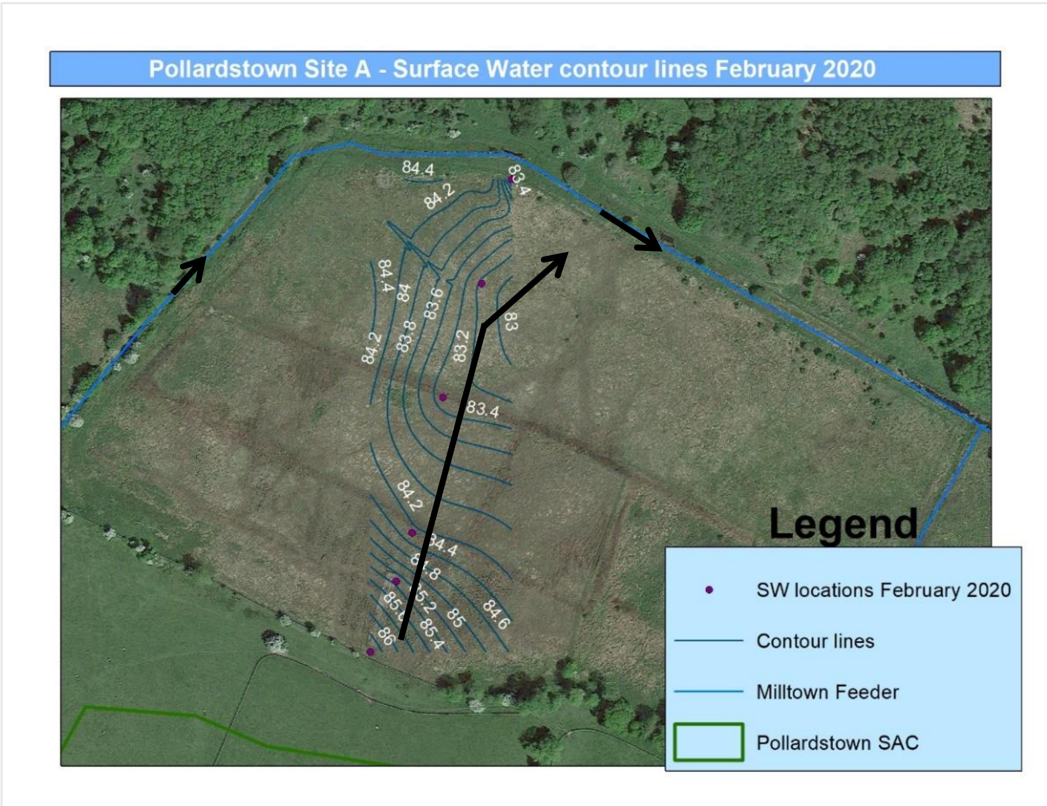


Figure 6.6. Contour lines of fen surface water and surrounding groundwater catchment interpolated using piont measurements in February 2020. Flowlines are presented with black arrows.

The water levels in the phreatic tubes (Figure 6.7) at all locations except PA40 are stable between hydrological years and do not seem affected much by periods of high effective rainfall. They also do not display any decrease during periods when no effective rainfall was recorded. The surface water level in phreatic PA 40 is affected by periods of effective rainfall as well as total rainfall, although this effect is primarily caused by the drawdown from adjacent local drainage. The difference in water level elevation along this transect in the fen is as much as 3 metres. This is because the surface, topographic elevations have a downwards slope from the edge of the fen at site PA31 to the discharge point at site PA40, next to the tributary drain.

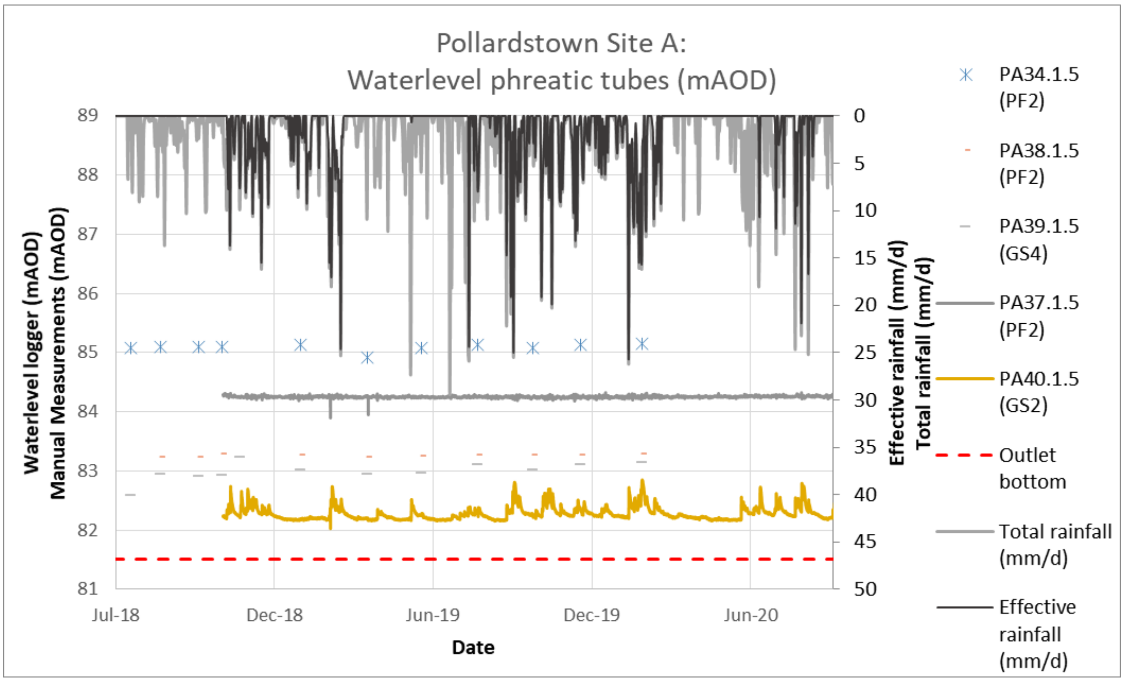


Figure 6.7. Phreatic water level hydrograph of spot measurements and water level loggers and rainfall. The height of the bottom of the outlet is presented with a red dashed line.

The water levels in piezometers PA31.4 (with a screen at 4 mBGL) and PA37.6 (with a screen at 6 m BGL) show a slow seasonal fluctuation and show similar behaviour (Figure 6.8). Piezometers PA34.7 and PA34.4 with screens at 7 and 4 m BGL respectively follow a similar trend exept for two occasions. A drop can be observed relative to the trend of the other piezometers PA31.4 and PA37.6 on April and October 2019. This fluctuation could be caused by different soil retention times reflecting various proximities to localized seepage zones on the margin of the fen. Indeed from the geological transect (Figure 6.48) a sandy and sandy clay layer can be seen in the vicinity of the piezometer screens of PA34.7 and PA34.4.

Again, piezometer 40.3 shows water level fluctuations in response to rainfall. The outlet canal drain has a drawdown effect on the water level. However the water level never decreases to less than 0.5 m above the invert since the water level is ultimately controlled by the main discharge of the fen to the canal. Moreover, as the main drainage channels are dredged every 5 to 6 years, part of the piezometer could be positioned in more permeable soil showing a reponse with more fluctuations.

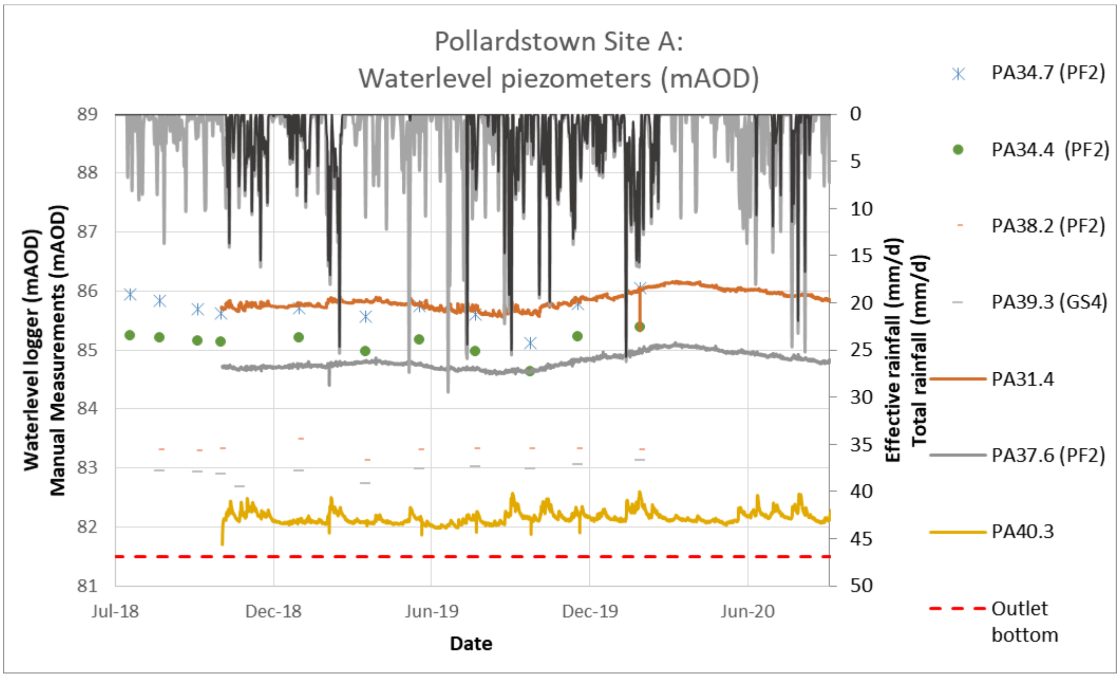


Figure 6.8 Piezometric water level hydrograph of spot measurements and water level loggers and rainfall. The elevation of the bottom of the outlet is presented by red dashed line.

Figure 6.9 shows hydrographs of phreatic tubes and piezometers measured between 2001 and 2007 which can now be compared to the more recent measurements gathered in this research. It has to be noted that the scale of this data is measured to the Poolbeg tidal level instead of the currently used Malin tidal head. The Poolbeg tidal level stands 2.71 m higher than Malin however for an easier comparison the current 2018-2020 data will be converted to the Poolbeg level in the text below.

The piezometric head of PA31.4 shows seasonal lows in 2002/03 of 88.5. These then decrease by about 0.25 m until 2007 which is probably the result of lowering of the regional groundwater table due to the construction of the Kildare Town By-Pass in 2002. From data in Figure 6.6 it seem that these levels have been maintained until now as low levels of PA31.4 were recorded at 88.3 m OD in the summer of 2019. It seemed that the road construction also had a great impact on the phreatic water levels of PA34.1.5 as levels were stable around 88 m OD until 2002. After this the water levels saw seasonal drops of up 0.5 m. From current data (Figure 6.9) it seems that levels have since stabilised however there is a possibility that they have permanently dropped as the levels were measured around 87.7 m OD between 2018 and 2020. PA34.4 shows similar behaviour and levels to PA34.1.5 where a permanent level drop was measured after 2002 and has since then been sustained until 2020. The level changes and current maintenance of PA34.7 are comparable to PA31.4.

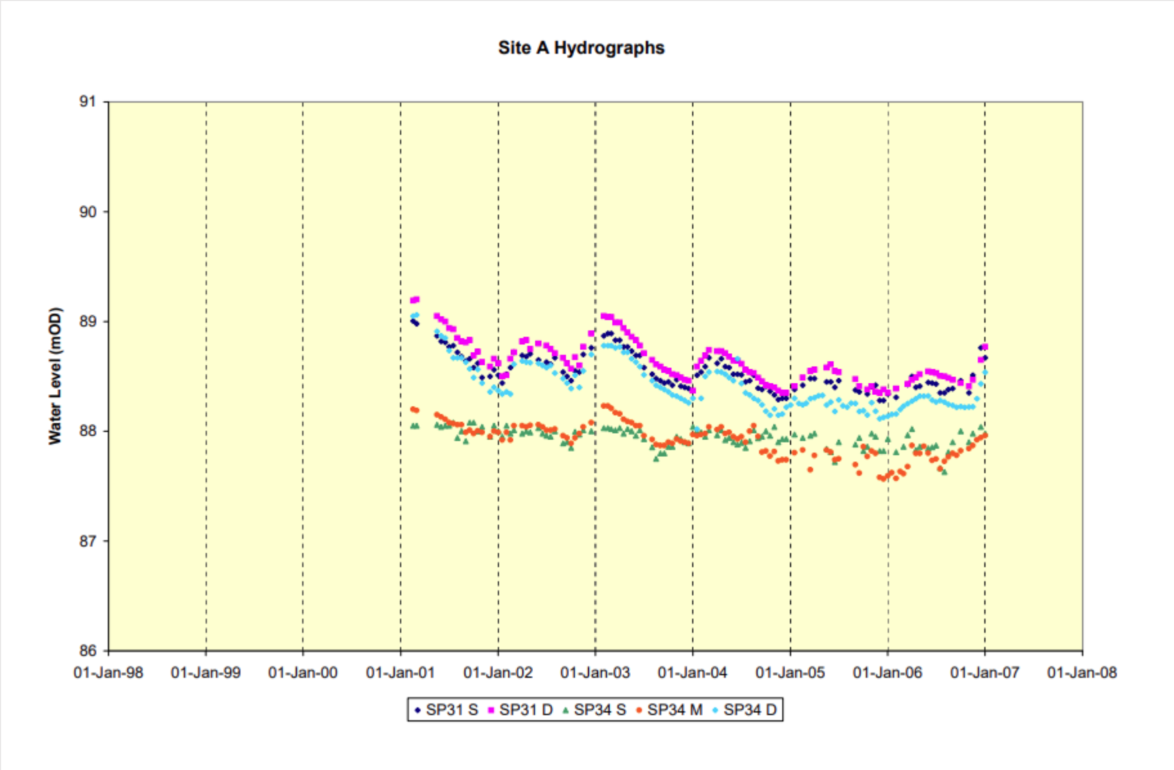


Figure 6.9. Phreatic and piezometric hydrographs measured in Pollardstown site A between 2001 and 2007 (Sholl, 2007). SP stands for standpipe and the number point to the location. The scale of the measurements is in m OD according to the head in Poolbeg. SP31 D can be compared to PA31.4, SP34 S to PA34.1.5, SP34 M to PA34.4 and SP34 D to PA34.7.

Figure 6.10 compares the water levels in subsoil borehole MB38 to the phreatic and piezometer water levels at fen station 37. They are located in the middle of the fen about 50 m from the adjacent topographic rise at the margin. The piezometer has been installed in the subsoil below the fen peat and has a screen depth of 7mBGL. It is believed that multiple discrete springs or seepages feed the fen in this area. The figure shows that while the piezometric head follows along with the signal of the piezometric head in the adjacent subsoil borehole, the phreatic head does not seem to follow. In fact, while the piezometric head in the subsoil well fluctuates by approximately a metre and the piezometric head in the fen fluctuates by about half a metre, the phreatic head does not fluctuate more than a few centimetres in the years 2018-2020. The reason that there was very little change in phreatic water level depths is that at this locations the fen is not particularly controlled by the outlet. The water that enters the fen can move laterally over a wide area fairly easily and therefore there was very little change in depths in relation to flows coming in. The stability of the phreatic level in comparison to the piezometric level at depth and the regional groundwater level is also a reflection of the control imposed on the flow to the fen by the presence of the low permeability peat substrate (Aldous et al, 2015; Kuczyńska, 2008; van Wirdum, 1991). It is this characteristic hydrogeology that gives rise to tufa formation and is able to sustain unique habitats for fauna such as the whorl snail, Vertigo geyeri (Kuczyńska, 2008; Foss, 2007).

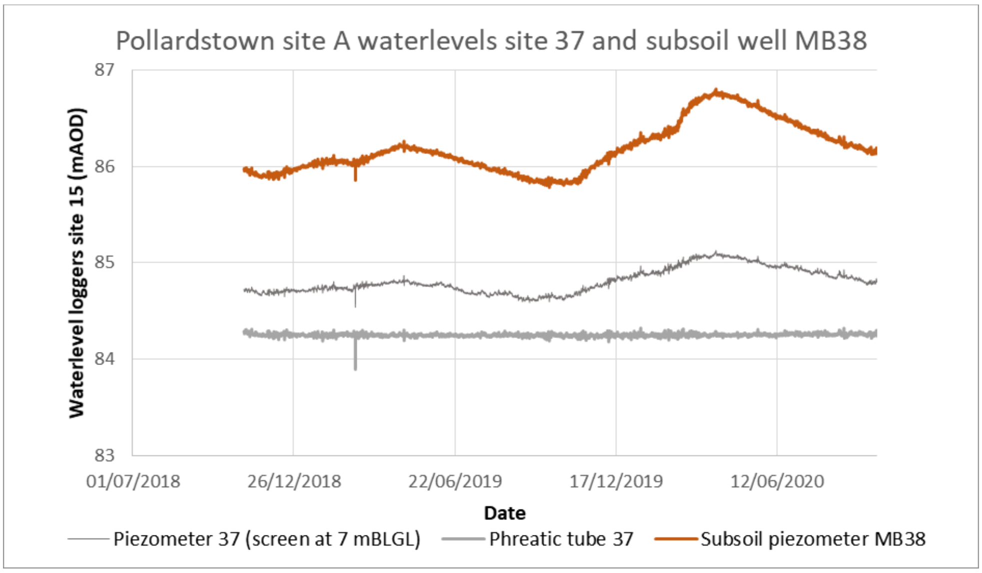


Figure 6.10. Hydrograph of phreatic and piezometric water levels at site 37 and piezometric water levels in subsoil well MB38

Phreatic and piezometric water levels at station 40 are compared in Figure 6.11. They are located next to the outlet channel at Pollardstown site A and about 150 m to the north from station 37. The piezometer was installed in fen peat and has a screen depth of 3 mBGL. The water levels in the fen are then again compared to subsoil borehole MB38.

What stands out is that while the piezometric head at station 37 seemed to follow the piezometric head at the subsoil borehole, this is not the case at site 40 at all. Not only the piezometric head, but also the phreatic head has a very flashy response to rainfall. Furthermore the concentration of the flows here is creating more of a phreatic fluctuation. Since the site is located about 7m from the outlet (Milltown Feeder), it can be suggested that the discharge from this outlet has a statistically significant draw down effect on the water levels in the fen peat. The same more responsive behaviour can also be seen from the discharge of the outlet measured at Hanged Man's Arch (see Figure 6.2)

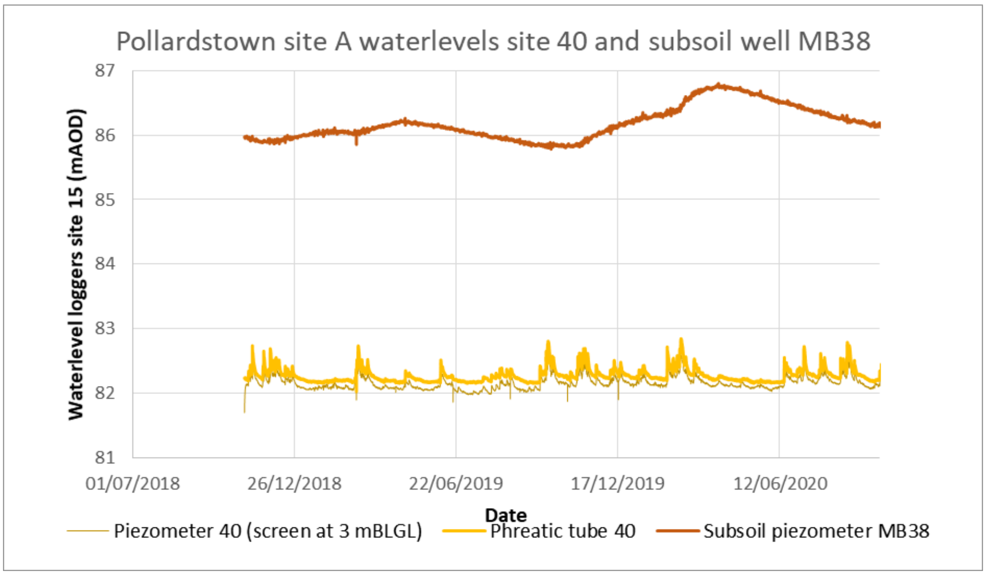


Figure 6.11. Hydrograph of phreatic and piezometric water levels at site 40 and piezometric water levels in subsoil well MB38

The current water levels measured in subsoil well MB38 can be compared to data measured between 2002 and 2007 in order to see if the situation has changed between now and then. Seasonal low measurements were found to be around 88.25 m OD (Poolbeg) in the older data (Figure 6.12). The seasonal lows measured in this research were found around 88.75 m OD which could mean that regional groundwater levels have recovered by about 0.5 m after the completion of Kildare Town By-Pass.

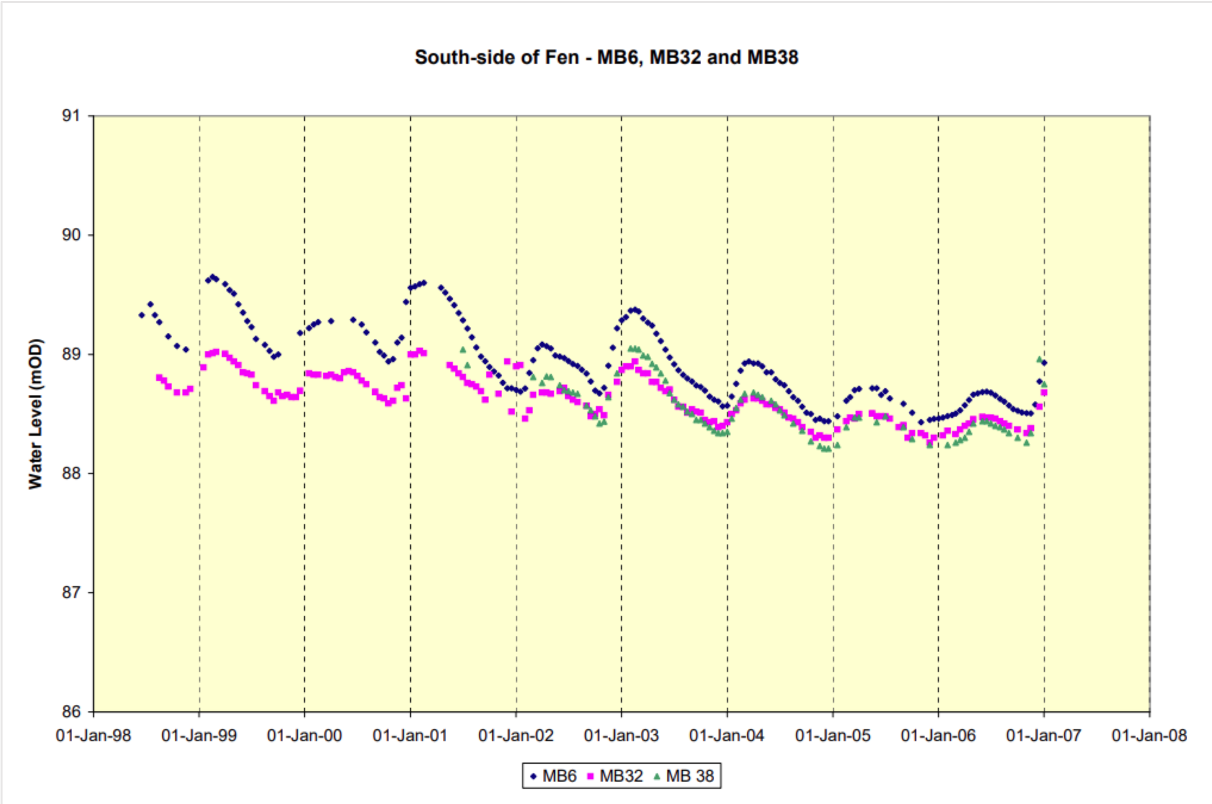


Figure 6.12. Hydrograph of borehole measured on the south-side of Pollardstown fen between 1998 and 2007 (Sholl, 2007). The scale of the measurements is in m OD according to the head in Poolbeg.

6.1.3.1. Electrical conductivity

From a time series of electrical conductivity in the phreatic tubes of Pollardstown coupled with rainfall (Figure 6.13) an estimation can be made of the relative contributions of groundwater inputs throughout 2018-2020. However it does not appear that effective rainfall has an effect on the recorded EC as no strong increasing or decreasing trends can be observed. This implies that the phreatic water table of Pollardstown site A is fed mainly by ground water at a relatively even rate throughout the year.

Location PA39 is an exception to this rule, however, which is believed to be due to its location. The phreatic tube was installed in a small streamlet in the middle of the fen that drains the upper levels of the phreatic fen which would be mixed with rainwater. The stream seems to receive a larger proportion of rainfall during the summer and autumn of 2019. Another location that has strong EC fluctuations is PA40. As mentioned before this phreatic tube shows strong phreatic water level fluctuation because of its proximity to the outlet. This location does show an increase in EC during the autumn of 2019 when high effective rainfall was measured again after the summer.

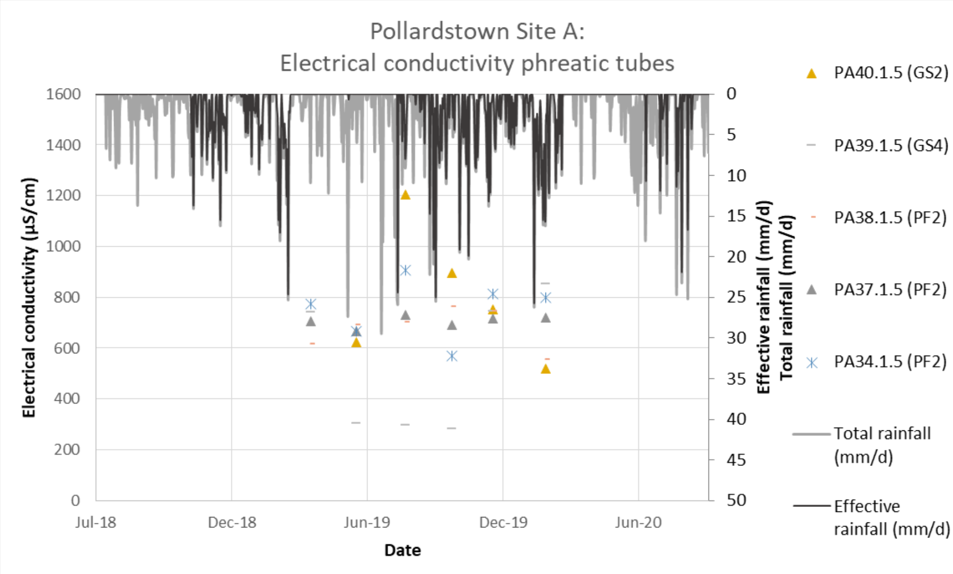


Figure 6.13 Time series of electrical conductivity (µm/cm) in phreatic tubes.

In Figure 6.14 again no major EC fluctuations are measured in most piezometers throughout the year. PA34.4 however does display low values during the spring of 2019 and the winter of 2019/20. This could have something to do with the fact that the screen is located in sandy or sandy clay layers combined with the high effective rainfall during that time.

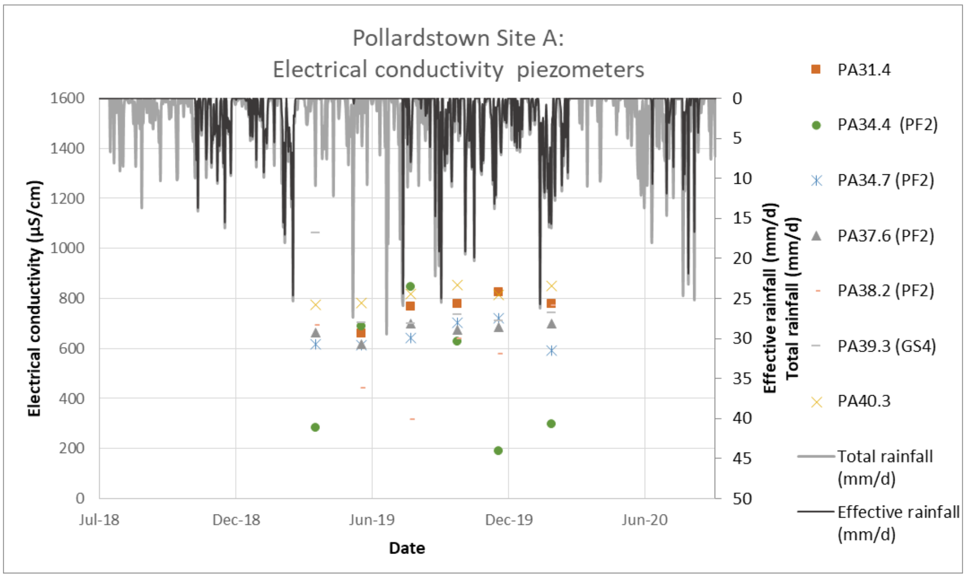


Figure 6.14. Time series of electrical conductivity (µm/cm) in piezometers.

The boxplots in Figure 6.15 show overall and seasonal electrical conductivity from data collected in the fen as well as from boreholes outside the fen. The overall EC in the boreholes is very similar to the fen which implies that the piezometric as well as the phreatic water table receives most of its water from the regional groundwater catchment. A great proportion of the water at the surface in this part of the fen is groundwater and with slow seepage rates through the fen peat explains the presence of the unique tufa habitat.

There is also no seasonal change in the piezometers and phreatic tubes which was also proven by a two sided Welch t-test with p-values of 0.81 and 0.85 respectively. These findings confirm that both the phreatic and piezometric water tables of Pollardstown A are fed by groundwater at a relatively even rate throughout the year, as would be expected from such a large alluvial aquifer (i.e. the Curragh gravel aquifer).

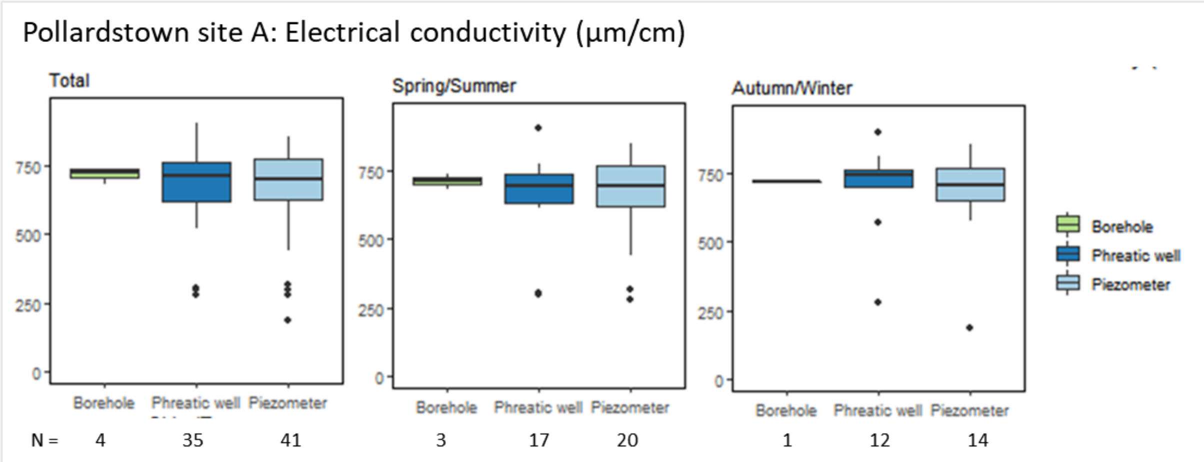


Figure 6.15. Electrical conductivity (µm/cm) in phreatic wells and piezometers inside and boreholes outside the fen in Pollardstown site A.

6.1.3.2. pH

The pH in Figure 6.16 suggest that values found in the fen are very similar to values found in boreholes outside the fen. Indeed, the median of the boreholes is 7.23 and the median for the piezometers in the fen is 7.39. The median of the phreatic tubes is very close to piezometers with a pH of 7.35. There also was no seasonal change recorded.

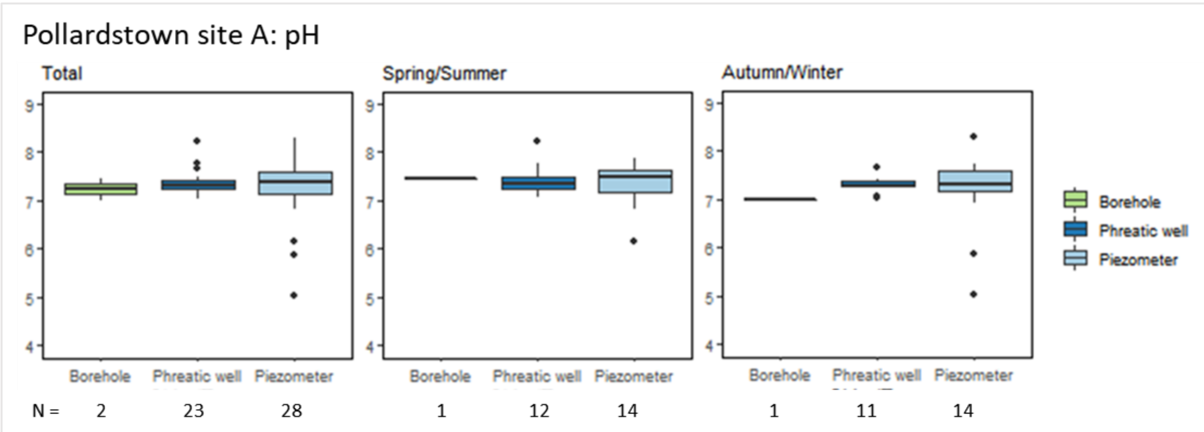


Figure 6.16. pH in phreatic wells and piezometers inside and boreholes outside the fen in Pollardstown site A.

6.1.3.3. Temperature

The temperature boxplots in Figure 6.17 suggest that the temperature in the water column of the fen and the surrounding catchment are similar. There is, however, a seasonal change recorded in the phreatic tubes as well as the piezometers. In the phreatic tubes the Spring/Summer median was at 12.2 °C whereas the Autumn/Winter median was much lower with 9.6 °C. This was proven to be statistically significant (p-value 0.00). This seasonal change is to be expected for the water column at the surface. The piezometers had a seasonal decrease from 12.6 to 9.65 °C. This change was also proved to be statistically significant (p-value 0.00). These temperatures are reflective of expected temperatures for regional groundwater in Ireland (11-12 C°) with some seasonal variation as groundwater emerges in seepages and springs.

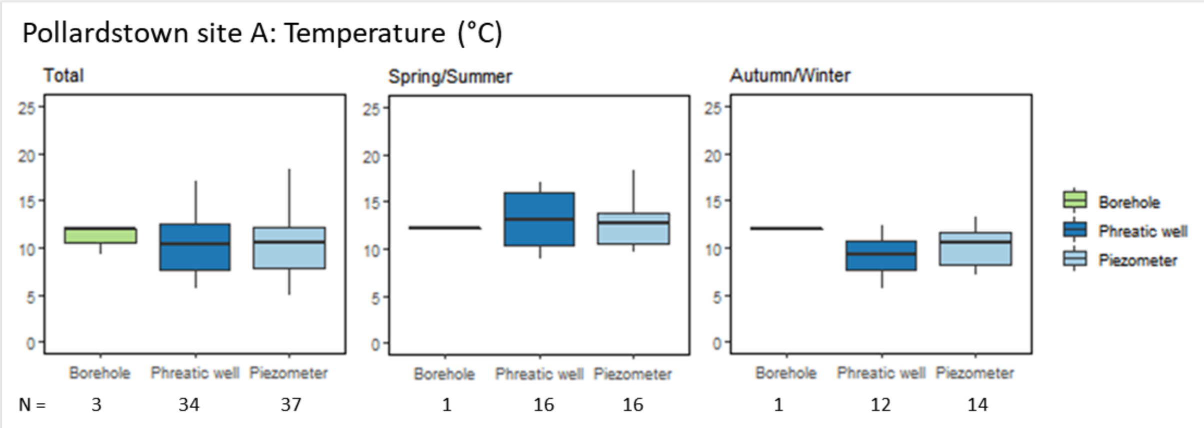


Figure 6.17. Temperature (°C) in phreatic wells and piezometers inside and boreholes outside the fen in Pollardstown site A.

6.1.1. Conceptual hydrology model

- 2.7 km^2 fen is supported by a catchment of 32.2 km^2. The ratio of fen to catchment is 1:12.
- Like Kuczyńska (2008) found with hydrological year 2003/04 the recent calculated water balance of hydrological year saw higher recharge values than discharge values. The recharge coefficient of 92% is however still acceptable as suggested by Misstear & Brown (2008).
- Average discharge rates via Milltown feeder are in close agreement to the discharge rates measured between 1993 and 2005 proving no hydrological changes have occurred between then and now.
- Water levels in most phreatic tubes shows very little change in level and are not affected by meteorological changes. Phreatic tubes that show no control by the outlet have a maintained water level provided by water entering the fen moving laterally of a wide area with ease.
- The similarity of EC in the boreholes to the piezometers as well as the phreatic tubes implies that the fen receives most of its water from the regional groundwater catchment which, together with slow seepage rates, explains the presence of unique tufa habitat.

6.2. Hydrochemistry

Hydrochemical sampling is reported in a series of boxplots of the data collated in and outside Pollardstown site A. A total of 139 samples were collected from boreholes, phreatic tubes and piezometers and subsequently analysed for phosphorus, nitrogen and other hydrochemistry. The total collected data set is displayed as well as the seasonal differences between the spring and summer with samples collected between 1st April and 30th September and the autumn and winter with samples collected between 1st October and 31st March.

6.2.1. Phosphorus

The boreholes around Pollardstown site are found with considerably higher concentrations of dissolved reactive phosphorus (DRP) than in the fen itself (Figure 6.18). The concentrations in the boreholes were statistically significantly greater when compared to the phreatic tubes (p-value of 0.02). This was also the case when compared to concentrations in the phreatic tubes (p-value of 0.04). The median concentration in the boreholes was reported with 0.607 mg-P/l whereas the median in the piezometers and phreatic tubes were found to be statistically significantly lower with 0.039 and 0.014 mg-P/l respectively. While the values from the boreholes seem anomalous, the boreholes themselves were in grassland and finished with steel casing and lockable caps, outside of the adjacent fenced grazing areas.

Additionally, the concentrations in the borehole and phreatic tubes remains stable throughout the seasons. However the concentrations seem to fluctuate in the borehole with lower concentrations during the Spring/Summer (median of 0.030 mg-P/l) which increased to a median concentration of 0.715 mg-P/l during the Autumn/Winter.

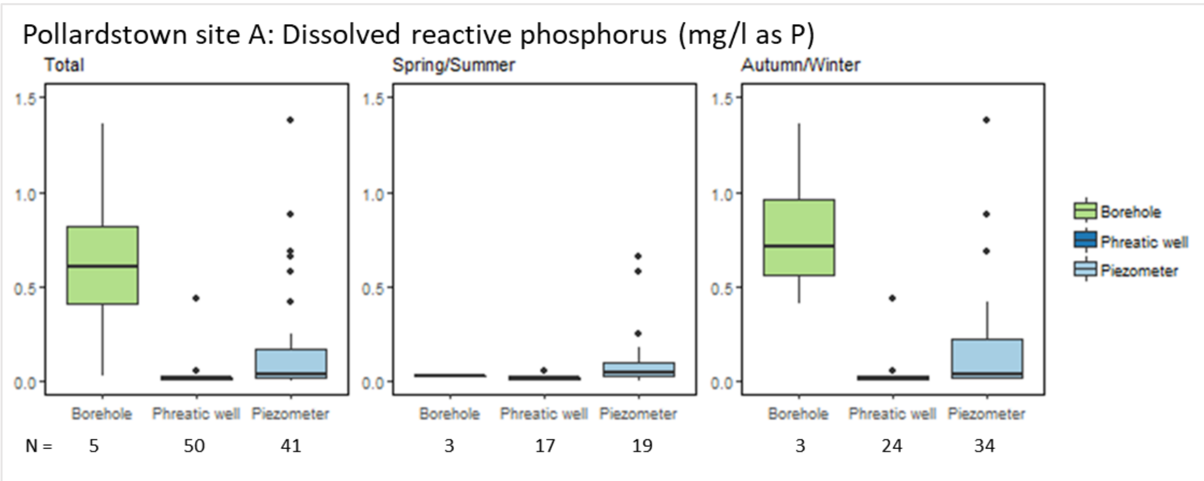


Figure 6.18. Dissolved reactive phosphorus in mg-P/l sampled from phreatic wells and piezometers inside and boreholes outside the fen in Pollardstown site A.

Similar trends are displayed in the boxplots of Figure 6.19. Here the concentrations of total phosphorus (TP) are also found statistically significantly higher than in the supplying aquifer (median of 1.32 mg-P/l) than in the fen itself with a median concentration of 0.11 mg-P/l found in the phreatic tubes and 0.13 mg-P/l in the piezometers. The concentrations were, however, not statistically significantly lower than in the boreholes (p-value of 0.14) when compared to the piezometers and a p-value of 0.12 compared to phreatic tubes.

The data shows a seasonal change in the boreholes around Pollardstown site A which also seems to happen in the boreholes of Ballymore (Section 5.1.2). The summer has the lowest concentrations with a median of 0.08 mg-P/l. This number then increases in the winter to 1.34 mg-P/l. The summer concentrations were tested statistically significant less with a p-value of 0.00. This phenomenon could be explained by processes of calcite precipitation and hydroxyapatite (Emsens et al., 2016; Duval 2010).

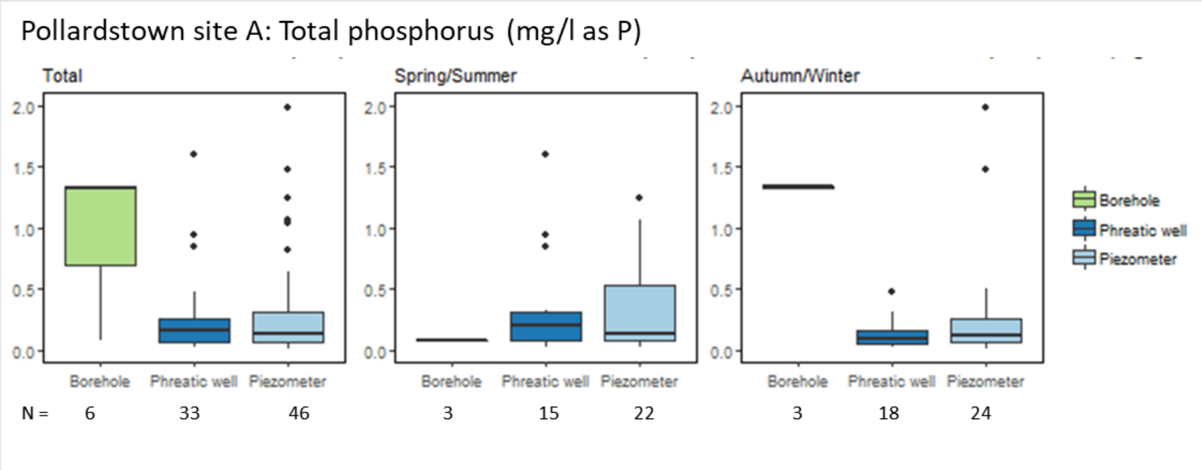


Figure 6.19. Total phosphorus in mg-P/l sampled from phreatic wells and piezometers inside and boreholes outside the fen in Pollardstown site A.

Since both DRP and TP concentrations are found to be statistically significantly higher in the boreholes than in the fen it can be concluded that surrounding regional groundwater catchment is not a reflection for phosphorus values found in the fen. Hence, this points to a similar hypothesis that was developed for Ballymore fen, whereby it seems that the vegetation in Pollardstown site A may be taking up the incoming DRP and ultimately storing it as TP in its accumulating sediments. On top of that, the phosphorus is also likely cycling between fractions in the sediments and shows some seasonal fluxes.

The ratios of DRP and TP from the medians of the 'Total' boxplots is 1:2 for the boreholes, with 1:3 for the piezometers and 1:8 for the phreatic tubes.

The reported groundwater threshold values of DRP in Ireland is 0.035 mg-P/l (Government of Ireland, 2010), which is statistically significantly exceeded in the boreholes around Pollardstown A with a median of 0.607 mg-P/l. This is not the case with the median concentrations found in the phreatic tubes with a median value of 0.014 mg-P/l. The piezometers (in or below the peat substrate) slightly exceeded this threshold with 0.039 mg-P/l.

6.2.2. Nitrogen

In Figure 6.20 the total ammonia concentrations in the boreholes outside the fen had a median of 0.22 mg-N/l. The concentrations are comparable with phreatic tubes (median of 0.14 mg-N/l) and piezometers (median of 0.27 mg/l). Indeed, the concentrations in both the phreatic tubes and the piezometers were not statistically significantly smaller with a p-values of 0.25 and 0.68 respectively. Seasonal fluctuations were not observed.

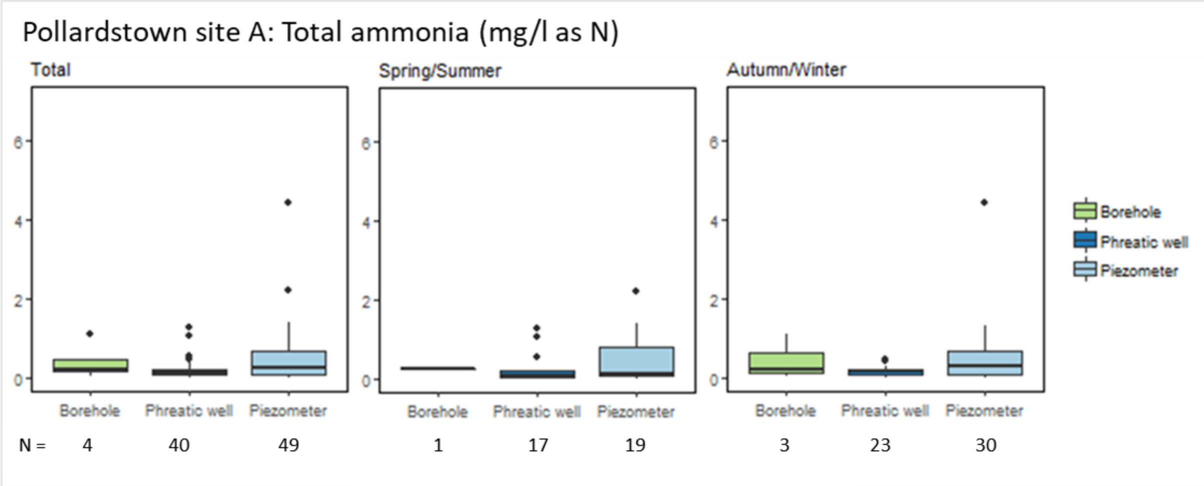


Figure 6.20. Total ammonia in mg-N/l sampled from phreatic wells and piezometers inside and boreholes outside the fen in Pollardstown site A.

Nitrite is barely found (Figure 6.21) in and around Pollardstown site A, with almost all samples were below the 0.05 mg-N/l limit of detection.

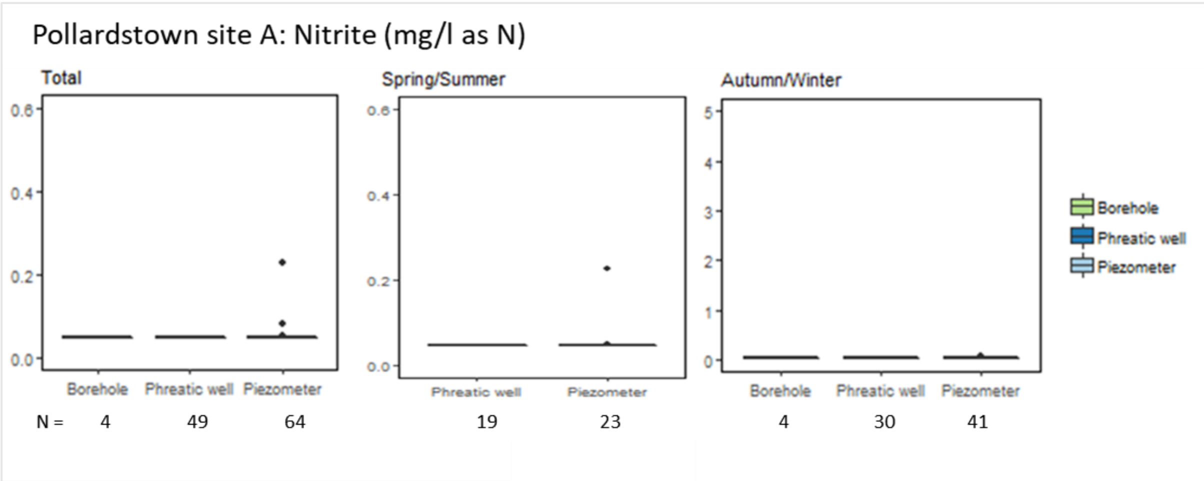


Figure 6.21. Nitrite in mg-N/l sampled from phreatic wells and piezometers inside and boreholes outside the fen in Pollardstown site A.

High concentrations of total oxidised nitrogen in the surrounding catchment seem to be somewhat reflected in the piezometers as seen in Figure 6.22. The median concentration was 1.49 mg-N/l in the boreholes compared to 0.63 mg-N/l in the piezometers. The median concentration of dissolved oxidised nitrogen in the phreatic tubes was much lower at 0.12 mg-N/l. Although the values in the phreatic tubes and piezometers were lower, the differences from the concentrations in the borehole was not found to be statistically significant, the Welch t-test returning p-values of 0.72 and 0.18.

The high concentrations in the aquifer are visible in the boreholes but also in the piezometers since their screen was installed at considerable depth and also displayed artesian conditions. These higher concentrations, however, do not seem to be reflected at the phreatic surface at Pollardstown site A. Again, this may suggest that plant nutrient uptake and/or denitrification ensure the total oxidised nitrogen concentration remains at relative low levels, as seen in Ballymore fen.

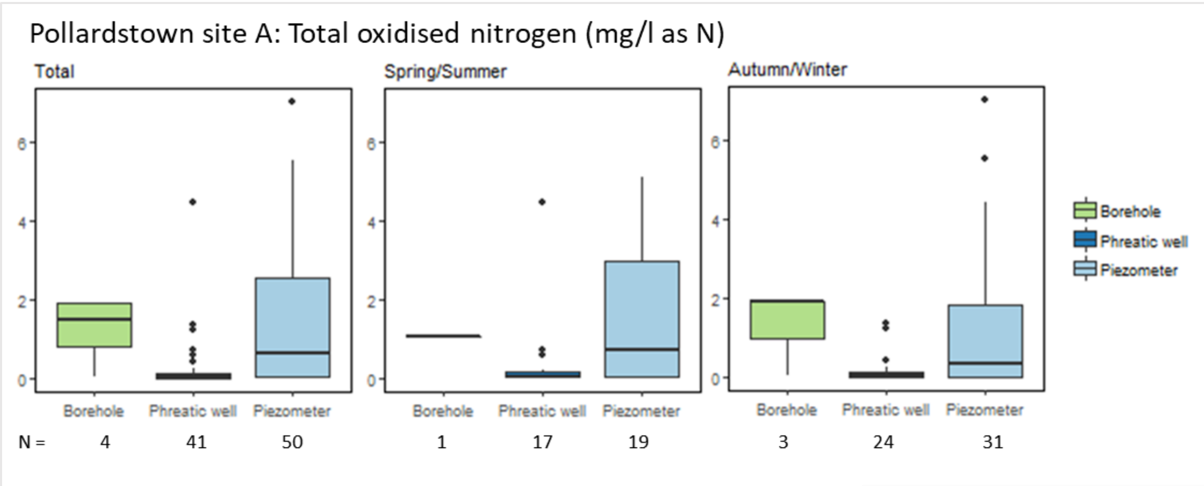


Figure 6.22. Total oxidised nitrogen in mg-N/l sampled from phreatic wells and piezometers inside and boreholes outside the fen in Pollardstown site A.

The concentrations of total dissolved nitrogen are found to be at similar levels and distributed in and around the fen (as was found in Ballymore). The highest proportion of total dissolved nitrogen is found in the boreholes around the fen as seen in Figure 6.23 with a median is 2.52 mg-N/l. This high concentration is also reflected in the piezometric measurements with a median of 2.17 mg-N/l. which was not significantly different (p-value of 0.44). The concentrations found in the phreatic tubes were also not statistically significantly lower (p-value = 0.09) with a median of 1.47 mg-N/l.

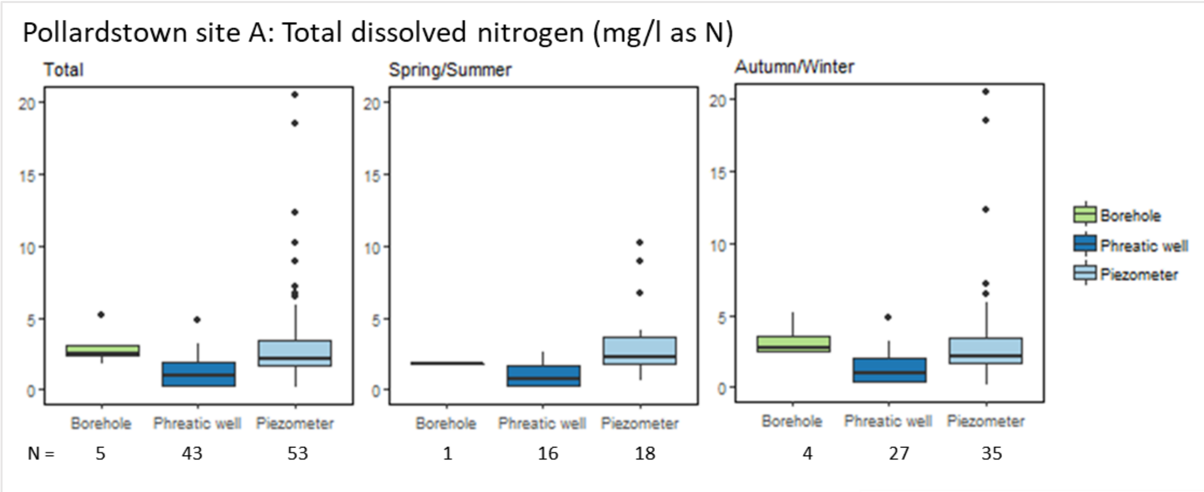


Figure 6.23. Total dissolved nitrogen in mg-N/l sampled from phreatic wells and piezometers inside and boreholes outside the fen in Pollardstown site A.

The ratios of total ammonia to total dissolved nitrogen was similar in boreholes and piezometers with 1:11 and 1:8 respectively. A much higher ratio was found in phreatic tubes with 1:105. The ratio of total oxidised nitrogen to total dissolved nitrogen was lowest for the boreholes with 1:2 as well as the piezometers with 1:3. The ratios of the phreatic tubes again was much higher with 1:12 respectively.

These ratios imply that a significant portion of the TDN in the piezometers is made up of ammonia and nitrate/nitrite. From the reported medians it was found that then 51% are in different forms of organic nitrogen. This is in contrast to the phreatic tubes where 0.82% of TDN was deemed to be in its organic form.

The reported medians do not exceed groundwater threshold values for nitrite or nitrate (Government of Ireland, 2010). None of the medians found in and around the fen were found higher than the threshold value of 0.114 mg-N/l for nitrite. There were, however, a few outliers in the piezometers that did exceed this threshold value. The total oxidised nitrogen concentrations had an insignificant amount of nitrite and a therefore can be regarded as a reflection of nitrate. None of the measured nitrate concentrations exceeded the threshold value of 8.47 mg-N/l.

6.2.3. Other chemistry

The overall concentrations of alkalinity seemed quite similar when comparing data in and around the fen as seen in Figure 6.24. Indeed, a two sided Welch t-test proved no statistically significant difference between the boreholes and the phreatic tubes (p-value = 0.65) or the piezometers (p-value = 0.09). The median in the boreholes was 189.3 mg/l as $CaCO_3$, whereas the median of the piezometers was somewhat higher at 207.5 mg/l as $CaCO_3$ reflecting high groundwater inputs and the retention of alkalinity in the fens groundwater. The alkalinity in the phreatic tubes was also higher at 194.4 mg/l.

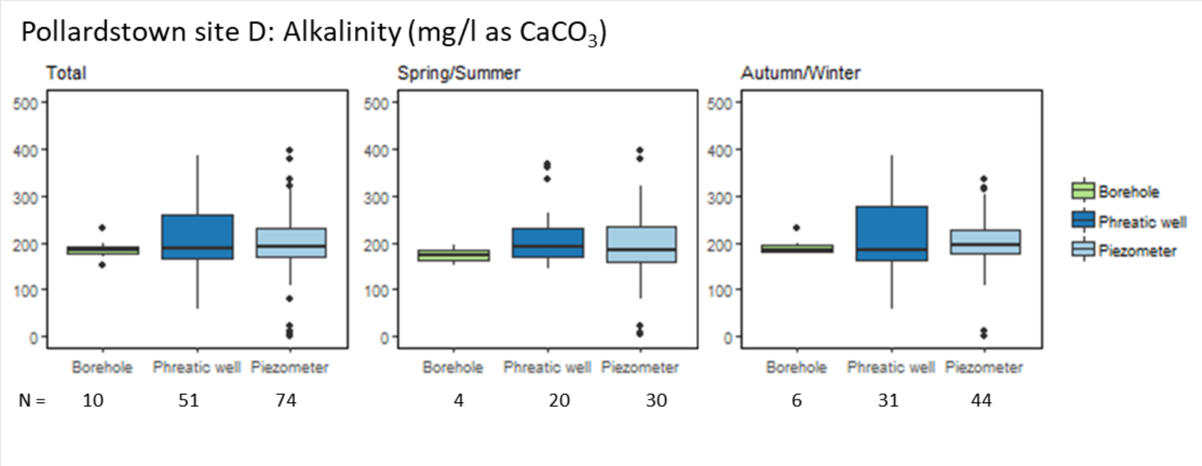


Figure 6.24. Alkalinity in mg/l as $CaCO_3$ sampled from phreatic wells and piezometers inside and boreholes outside the fen in Pollardstown site A.

The overall concentrations of chloride (Figure 6.25) were also similar between boreholes around the fen (median of 16.01 mg/l) compared to measurements taken in the phreatic tubes (median of 14.8 mg/l - not significantly with p-value of 0.72) and in the piezometers (median of 13.9 mg/l - not significantly with a p-value of 0.74). Also, the reported medians are far below the Irish groundwater threshold values for chloride (Government of Ireland, 2010) which is 187.5 mg/l.

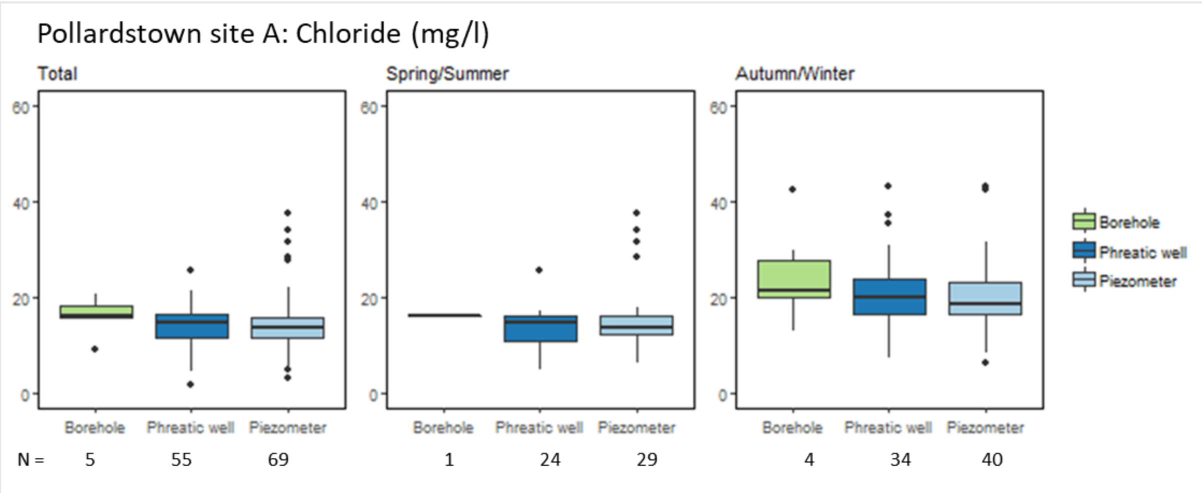


Figure 6.25. Chloride in mg/l sampled from phreatic wells and piezometers inside and boreholes outside the fen in Pollardstown site A.

Figure 6.26 shows the boreholes have the highest concentrations of silica with a median of 7.9 mg/l as SiO_2 as expected for groundwater. The concentrations found in the fen are statistically significantly lower with a p-values of 0.00 for both phreatic tubes and piezometers, respectively, reflecting that water in the fen is a combination of both groundwater and surface water inputs. Median silica concentrations in the phreatic tubes were 6.8 mg/l as SiO_2 compared to the concentrations in the piezometers which were slightly lower with a median of 6.3 mg/l as SiO_2.

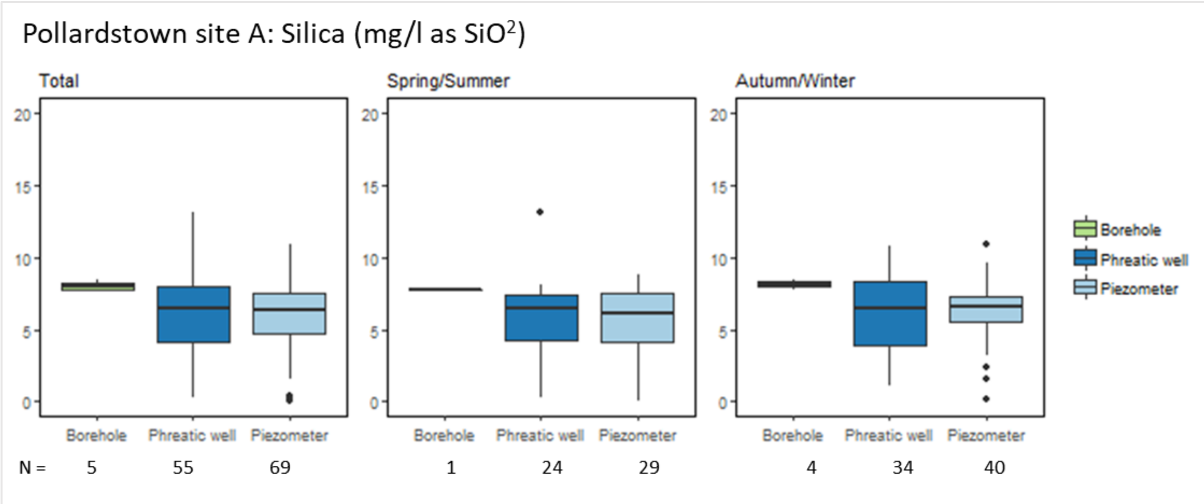


Figure 6.26. Silica in mg/l as SiO^2 sampled from phreatic wells and piezometers inside and boreholes outside the fen in Pollardstown site A.

Sulphate concentrations (Figure 6.27) were found to be higher in the boreholes around the fen with a median of 15.2 mg/l as SO_4^{2-} compared to lower concentrations in the in the piezometers with a median of 8.1 mg/l as SO_4^{2-} as well as the phreatic tubes with a median of 5.4 mg/l as SO_4^{2-}. They were however not statistically significantly less as the p-values were 0.62 and 0.36 for phreatic tubes and piezometers respectively. Again, as for the nutrients, it is possible that this reduction is due to the interaction with the fen vegetation. This would however also require sulphide values in order to find the total S balance in order to further support this statement.
Concentrations of sulphate increased between the Spring/Summer and the Autumn/Winter in the fen, shifting from a median of 3.5 mg/l as SO_4^{2-} to 12.5 mg/l as SO_4^{2-} in the phreatic tubes and from 4.5 mg/l as SO_4^{2-} to 11.5 mg/l as SO_4^{2-} in the piezometers. This decrease in either the phreatic tubes or piezometers was not statistically significant with a p-values of 0.13 and 0.34 respectively. Despite all that, the measured values are still all far below the Irish groundwater threshold values for sulphate (Government of Ireland, 2010) which is 187.5 mg/l.

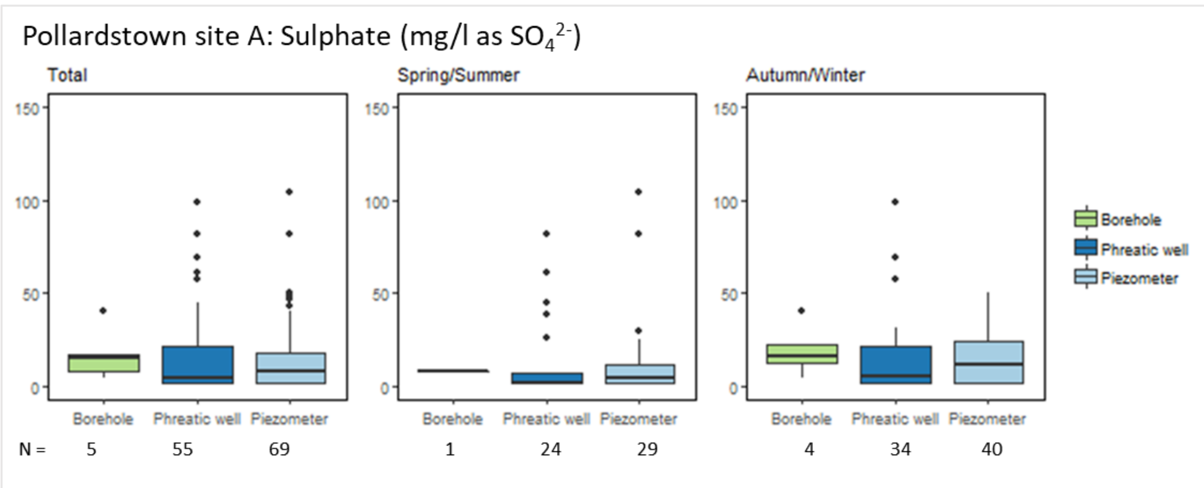


Figure 6.27. Sulphate in mg/l as SO_4^{2-} sampled from phreatic wells and piezometers inside and boreholes outside the fen in Pollardstown site A.

Dissolved organic carbon in and around the fen was found with higher concentrations in the fen than in the surrounding aquifer (Figure 6.28). There is an especially large range in the median to the third quartile of the phreatic tubes. These high values are expected from the annual die-back and breakdown of dead vegetation down into the peat.

The boreholes in the fen catchment had a median of 2.5 mg/l. The median concentrations were found to be only slightly higher in the phreatic tubes and piezometers with a median of 4.8 mg/l and 3.0 mg/l, respectively. Two sided Welch T-test proved that the boreholes had statistically significant lower values than the phreatic tubes (p-value = 0.00) as well as the piezometers (p-value = 0.01).

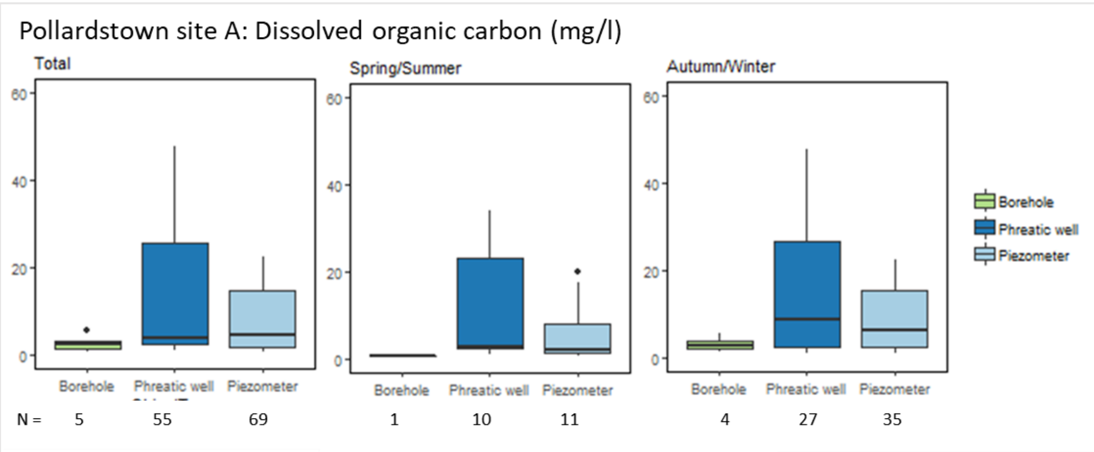


Figure 6.28. Dissolved organic carbon in mg/l sampled from phreatic wells and piezometers inside and boreholes outside the fen in Pollardstown site A.

Almost no ferrous iron was detected in the boreholes around the fen with a median concentration of 0.06 mg/l as Fe^{2+} as seen in Figure 6.29, which would suggest that the groundwater is in an oxic state. Concentrations seem higher in the phreatic tubes with a median of 0.12 mg/l as Fe^{2+} and this was proven to be statistically significant with a p-value of 0.02.

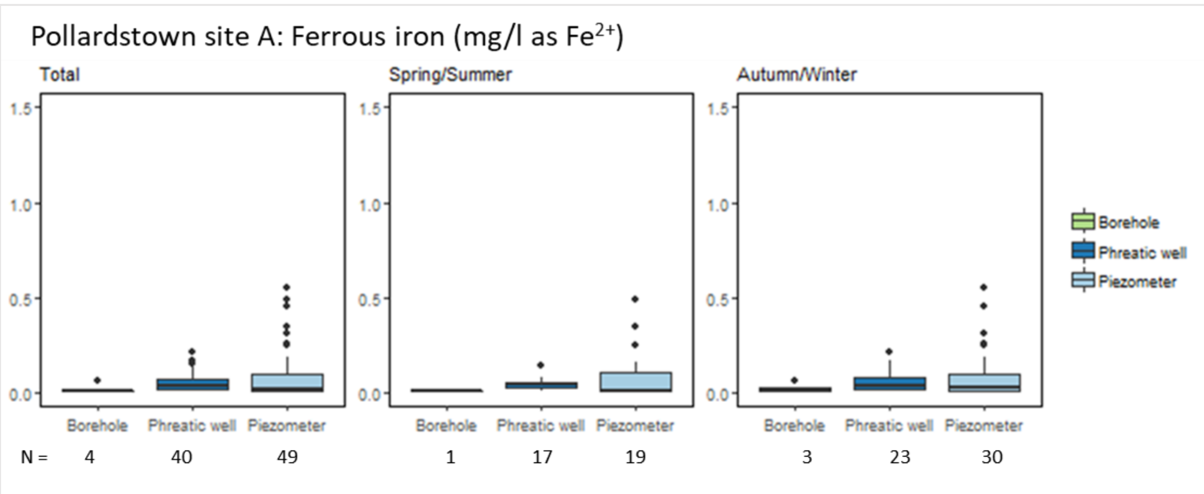


Figure 6.29. Ferrous iron in mg/l as Fe^{2+} sampled from phreatic wells and piezometers inside and boreholes outside the fen in Pollardstown site A.

6.2.4. Conceptual chemistry model

- DRP and TP is statistically significantly higher in boreholes than in the phreatic water table. The piezometers did not have significantly lower concentrations than the boreholes. This has to with the fact that the screen of some of these piezometers is actually located in the underlying clay gravel aquifer. Groundwater input with higher concentrations of phosphorus do not affect the water column in the fen.
- Nitrogen concentrations are found lower in the phreatic water table than in the surrounding catchment due to nutrient recycling.
- Low concentrations of ferrous iron in the boreholes surrounding the fen suggest that the groundwater is in in an oxic state. This compared to higher concentrations in the phreatic tubes suggest that the phreatic water table is in a more reduced state.

6.3. Linkage to fen habitat

6.3.1. Hydrology and fen habitat

6.3.1.1. Boxplots water level

The boxplot in Figure 6.30 displays the water level for different Fossitt habitats in the fen for all measurements (taken relative the ground surface). Only water levels with a median of +0.01 mAGL seem to be able to support the habitat Poor fen and flush (PF2). It also seems to need a positive pressure in the aquifer below, as measured in the piezometers showing artisan conditions (median of +0.46 mAGL). These conditions cause the groundwater to feed the fen via diffuse springs. Not all piezometers of Pollardstown A had artisan conditions, however, which is visible in the high fluctuations of the boxplot.

Overall, the water levels in phreatic tubes and the piezometers are very similar in both the Dry meadows and grassy verges (GS2) and Wet grassland (GS4) habitats. However, the vegetation in GS2 is supported by drier conditions (median = -0.61 mAGL) compared to the wetter conditions for GS4 (median = -0.18 mAGL), as would be expected from their descriptions. It should be noted also that the GS2 habitat is located close to the outlet (Milltown Feeder) which, as discussed, experiences significant draw down effects.

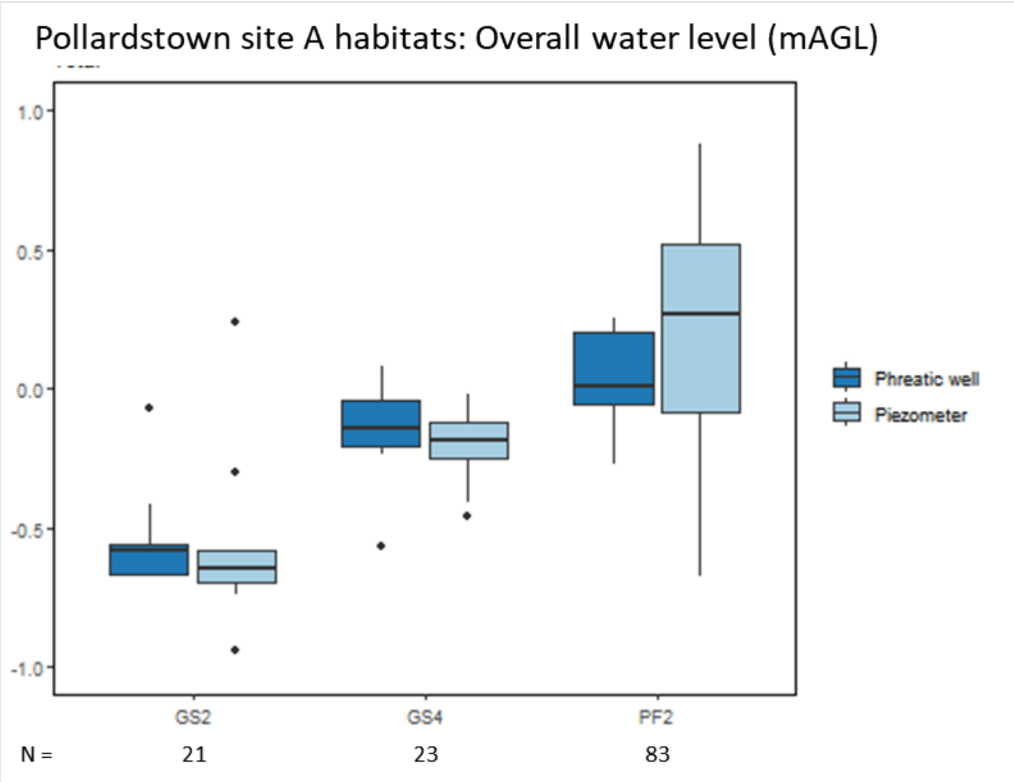


Figure 6.30. Overall water level in meters above ground level in the different habitats of Pollardstown site A measured in phreatic wells and piezometers.

In the boxplots of Figure 6.31 the water levels in PF2 remain stable whereas the other habitats display some seasonal fluctuations. The most significant change was detected in habitat GS2. Here phreatic water levels dropped by 0.05 m and piezometric water levels even dropped by 0.16 m during the Spring/Summer. The phreatic water levels in GS4 displayed a median phreatic water level decrease of 0.14 whereas the piezometers had a water level decrease by 0.10 m during the Autumn/Winter. This could be an indicator that this habitat was receiving a higher proportion of surface flow during that time. It seems this may be a plausible explanation as Bond et al. (2020) found that winter overland flow velocities were significantly higher than in summer in English upland grasslands.

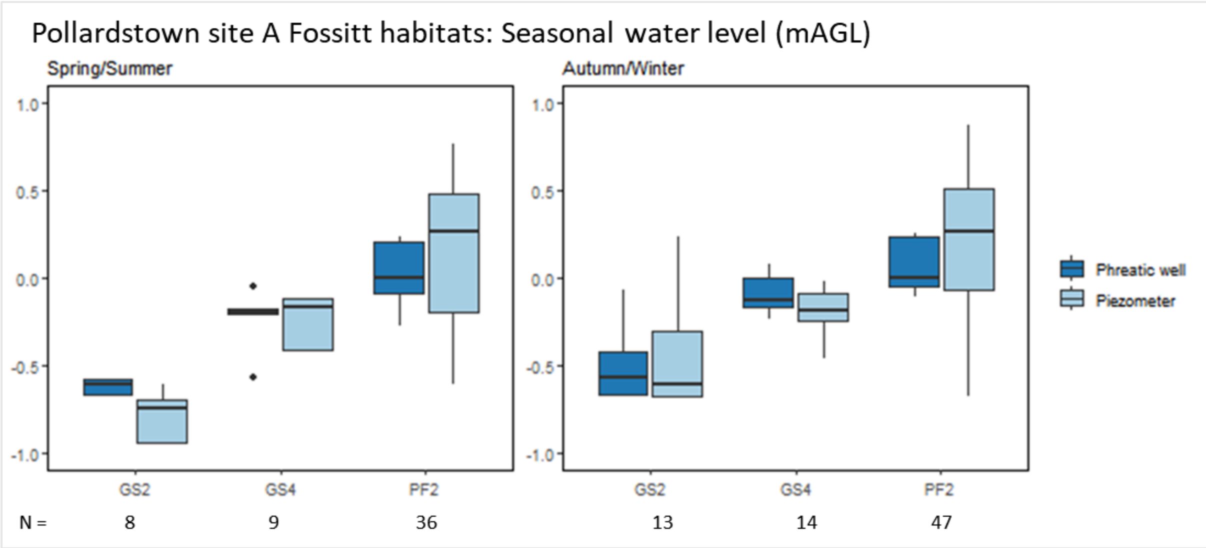


Figure 6.31. Seasonal water level in meters above ground level in the different habitats of Pollardstown site A measured in phreatic wells and piezometers.

6.3.1.2. Frequency duration curves

The curves (Figure 6.30) were made with the water level time series from data collected between October 2018 and October 2020. As mentioned before in Section 6.3.1.1 the PF2 habitat is supported with phreatic water levels above the surface as reflected in the level duration curve of site PA37. The phreatic water level only seems to fall below the surface elevation for <10% of the hydrological year. PA40 located in the GS4 habitat has phreatic water levels at much lower levels with at least 0.3 mBGL measured during the two hydrological years. This was mainly caused by the drawdown of the nearby outlet. The large fluctuations of 0.5 m are due to temporary spikes in water levels linked to rainfall recharge events.

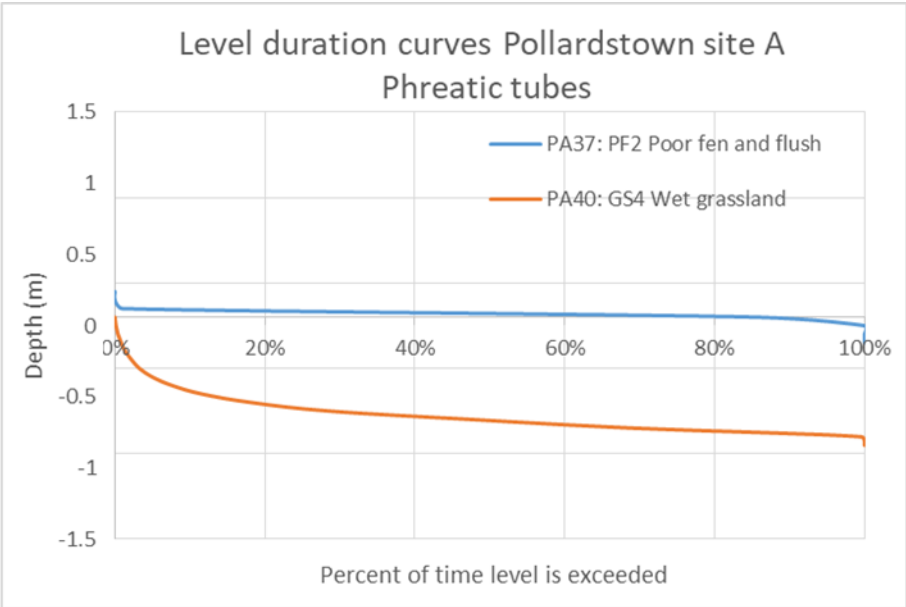


Figure 6.32. Phreatic level duration curves recorded in different habitats in Pollardstown site A. The negative numbers are water levels below groundlevel.

In a research conducted for Kildare County council level durations of PQ10 Shallow were plotted with water level time series of a phreatic tube located between sampling sites PA31 and PA34 in Figure 6.33. The curve was plotted with a series of water level data collected between 1998 and 2007. The habitat in PQ10 seems supported by a water level above 86 m OD for >70% of the time. After this the level have a change of dropping around 0.25 m which seems like a mayor change compared to the levels of PA37 in Figure 6.30. However it is possible that this change is due to changing phreatic water tables caused by construction of the Kildare Town By-Pass in 2002. This is further confirmed in Figure 6.34 where the water levels are at a steady level of around 88.5 m OD measured to Poolbeg tidal level until after 2002 where water levels drop about 0.25 m. It is expected that these levels have now stabilised since comparison to water levels of a phreatic tube located close to PQ10 shows stable levels measured between 2018-2020 (as seen in the phreatic water levels of PA34.1.5 in Figure 6.9). It is unknown if levels have recovered to their original level since the phreatic water levels were not measured in the current research.

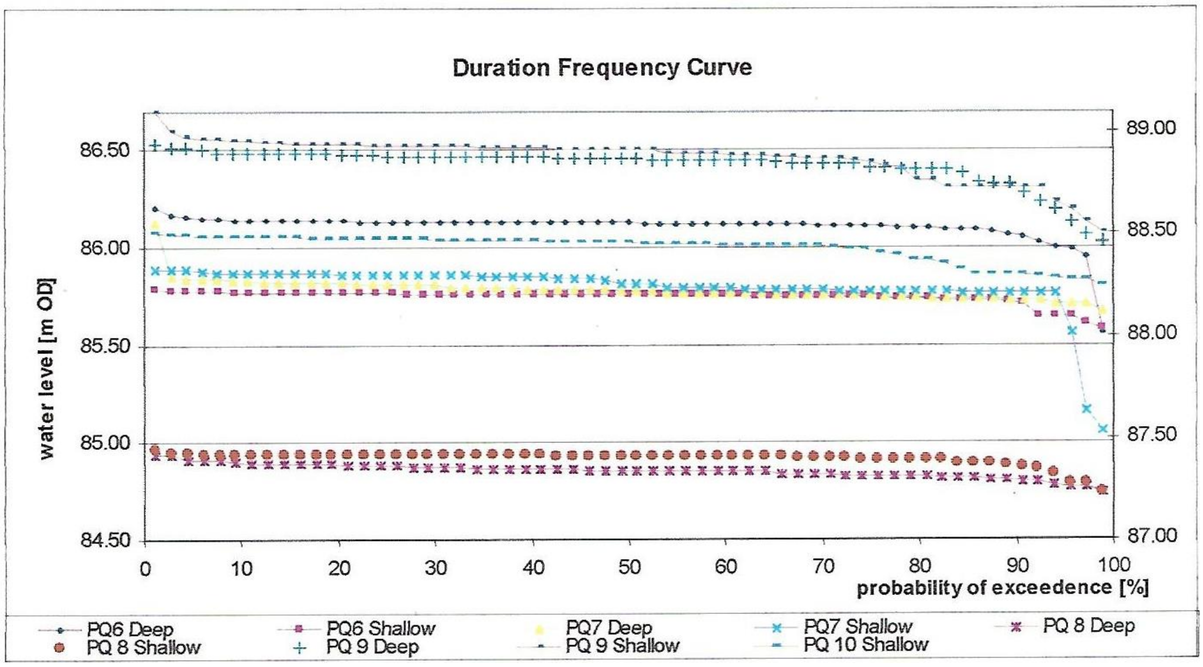


Figure 6.33. Level duration curves measured in the permanent quadrats (PQ) of Pollardstown fen between 1998 and 2007 (Hayes, 2004). The left axis of the graphs shows the scale in m OD measured according to Malin. The right axis is the scale in m OD according to Poolbeg.

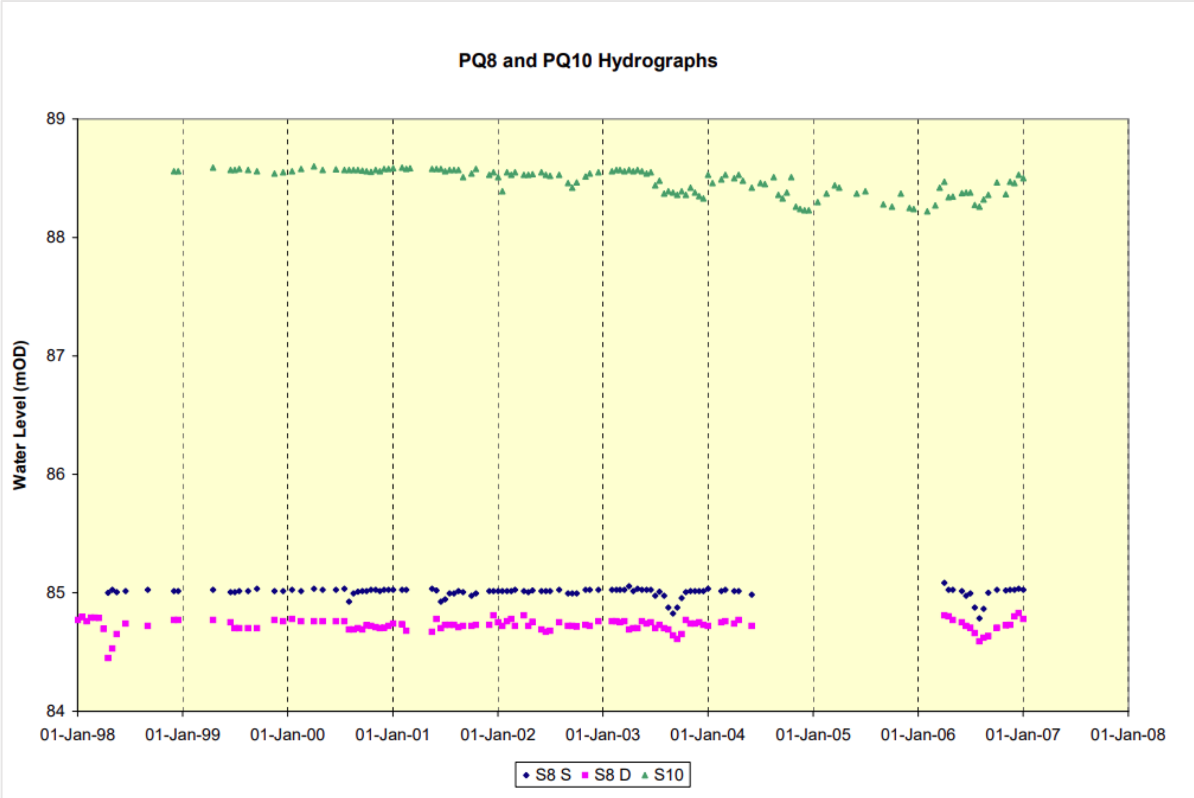


Figure 6.34. Figure 6.35. Phreatic and piezometric hydrographs measured in permanent quadrats between 1998 and 2007 (Sholl, 2007). The scale of the measurements is in m OD according to the head in Poolbeg.

From the phreatic and piezometric water level time series at PA37 and PA40 hydraulic gradients were calculated, as shown in Figure 6.36. PA37 displays a positive hydraulic gradient that is strongly influenced by the fluctuations of water levels in the aquifer. Site PA40 with habitat GS4 had an overall downward gradient. This hydraulic gradient did see temporary stronger downwards fluctuations during periods where effective rainfall was absent. During this time the outlet seems to cause the hydraulic gradient to become even more downward. PA40 was further affected by a seasonal water level drop during the summer of 2019 which resulted in an even lower hydraulic gradient.

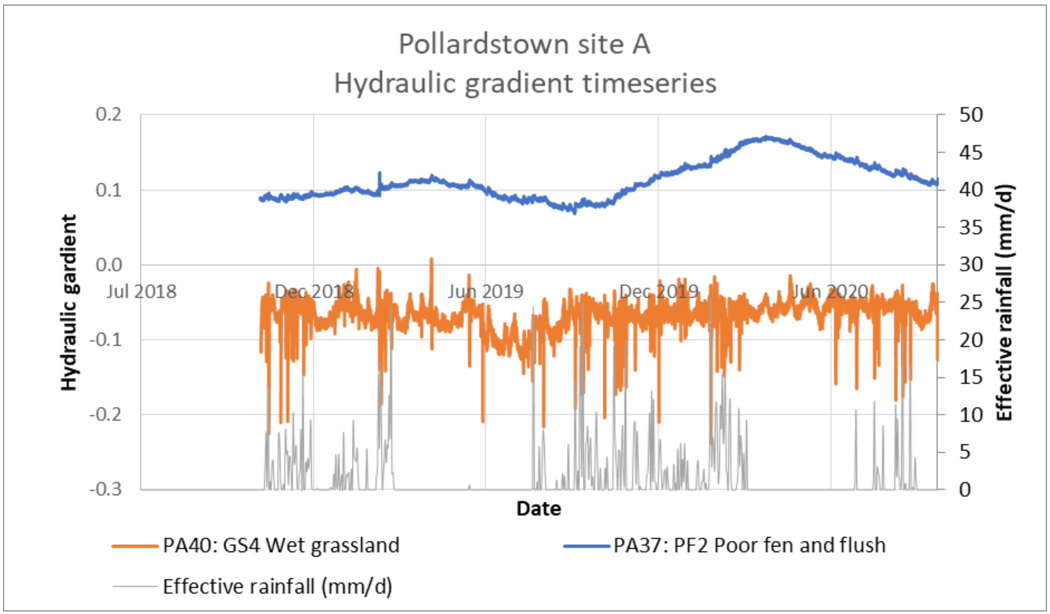


Figure 6.36. Hydraulic gradient timeseries calculated using the phreatic and piezometric water level timeseries in Pollardstown site D. Effective rainfall is displayed here as well.

Overall, upwards flows seem to occur at the southern edge of Pollardstown site A at the bottom of the hill. Indeed it as was found earlier, the catchment feeds groundwater into the fen by diffuse springs in this area. This hydraulic gradient slowly changes to downwards flow when travelling further north along the transect, as the influence of the drawdown at the outlet becomes stronger (as seen in PA40).

In order to see the overall change in hydraulic gradient the data was also plotted as frequency durations curves as seen in Figure 6.37. PA37 displays a level change of 0.1 m although this spread out evenly throughout the hydrological years. The level change in PA40 seems more abrupt. The sudden drying conditions as was seen in the summer of 2019 had a duration of less than 5%.

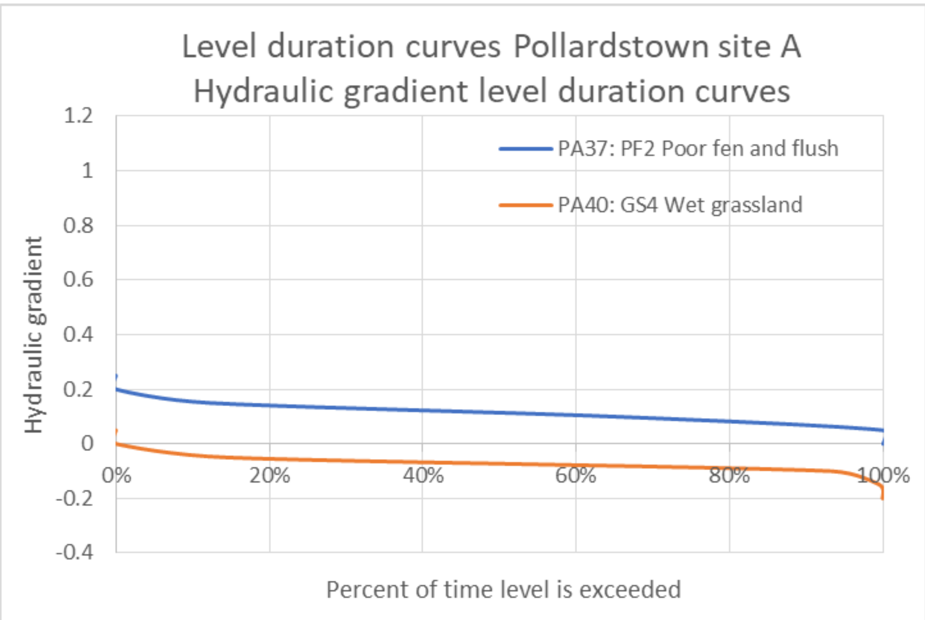


Figure 6.37. Level duration curves of the hydraulic gradients calculated from the water level time series in monitored phreatic tubes and piezometers.

6.3.2. Hydrochemistry and fen habitat

6.1.3.4. Nonmetric Multidimensional Scaling ordination

A Nonmetric Multidimensional Scaling ordination (NMDS) using ecological and hydrochemical data (collected in both Pollardstown site A and D) was plotted in order to find environmental vectors that have some form of correlation with the sampled locations and their specific habitat. The IVC data set containing the recorded species percentage at each surveyed relevé contained 12 relevés and 89 species. The environmental set (ENV) consisted out of vegetation type cover (%), Fossitt habitat codes and the hydrochemistry results (mg/l).

From the NMDS plot in Figure 6.38 can be concluded that surface water cover score was highly negatively correlated with axis NMDS1. This indicates that the clusters towards the negative end of Axis 2 are associated with a higher cover of surface water. Relevé PA38 seems particularly associated with this as well as the abundance of bryophytes. Species *Calliergoriella cuspidata* also has a high association with this environmental vector. Furthermore, habitat PF2 (Poor fen and flush) seems associated with the species *Schoenus nigrigans* and *Juncus Subnodulosus* as they are both negatively correlated on the NMD1 and NMD2 axis. Habitat GS2 (Dry meadows and grassy verges) find high abundances of grass which is not surprising regarding the habitat description. Species *Molinia cearulea* stands out in this habitat. Finally, habitat FS1 (Reed and large sedge swamps) is dominated by *Cladium mariscus* as both are positively correlated along the NMDS2 axis.

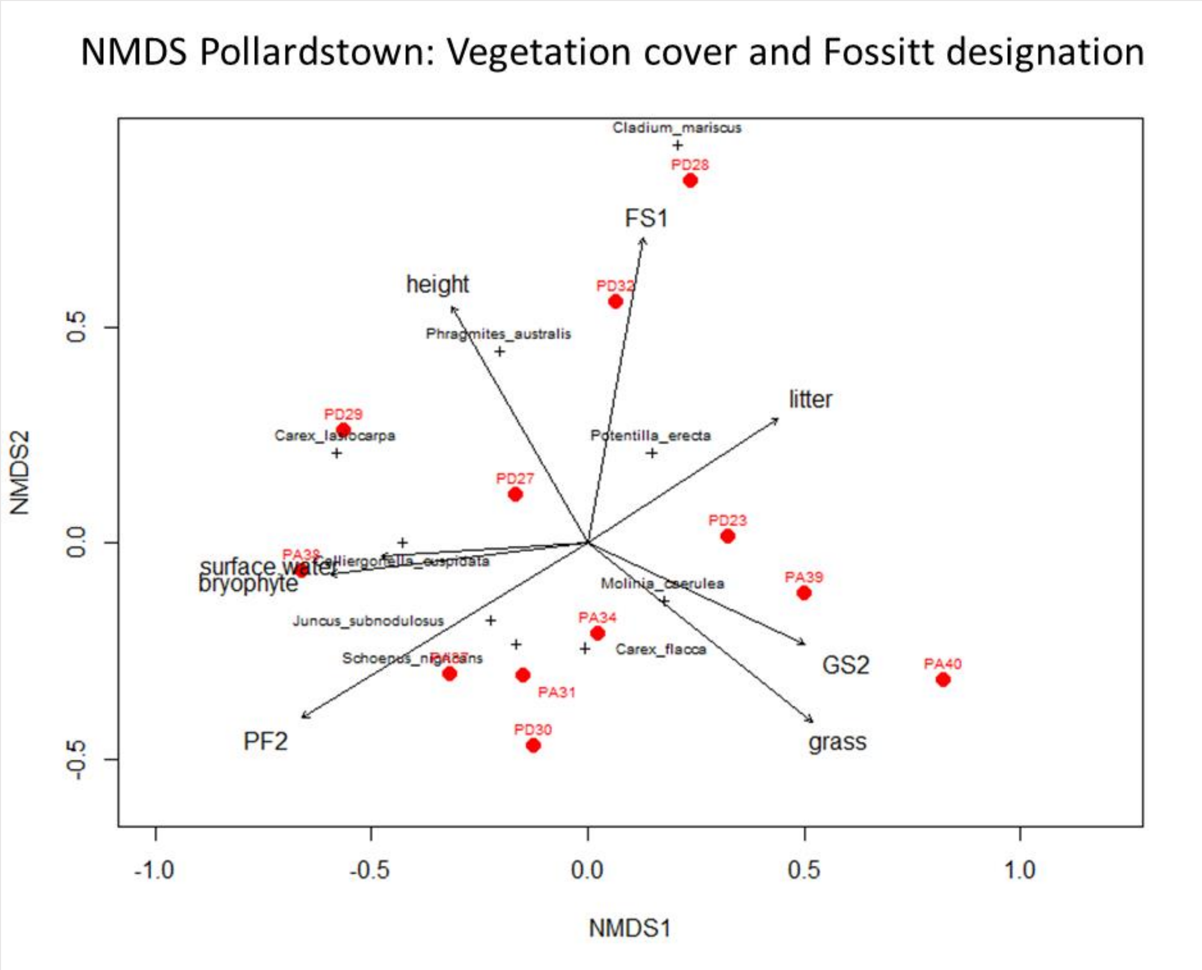


Figure 6.38. Multidimensional Scaling ordination of dimensions 1 and 2 with vegetation cover and Fossitt habitats plotted as vectors (max p-value = 0.2) in Pollardstown sites A and D. The phreatic and piezometer nest locations are shown in red and the names of the species with the highest abundances (10%) are also plotted.

A follow up NMDS was run with hydrochemistry results from April and June 2019 as environmental variables (Figure 6.39). From the plot can be concluded that in PF2 habitats higher concentrations of TP, total oxidised nitrogen, total dissolved nitrogen, silica and sulphate are expected as these are all highly negatively correlated to the NMDS1 and NMDS2 axis. Furthermore, positively correlated along the NMDS2 axis, higher concentrations of DRP, TP and ammonia are expected at sites with a FS1 habitat.

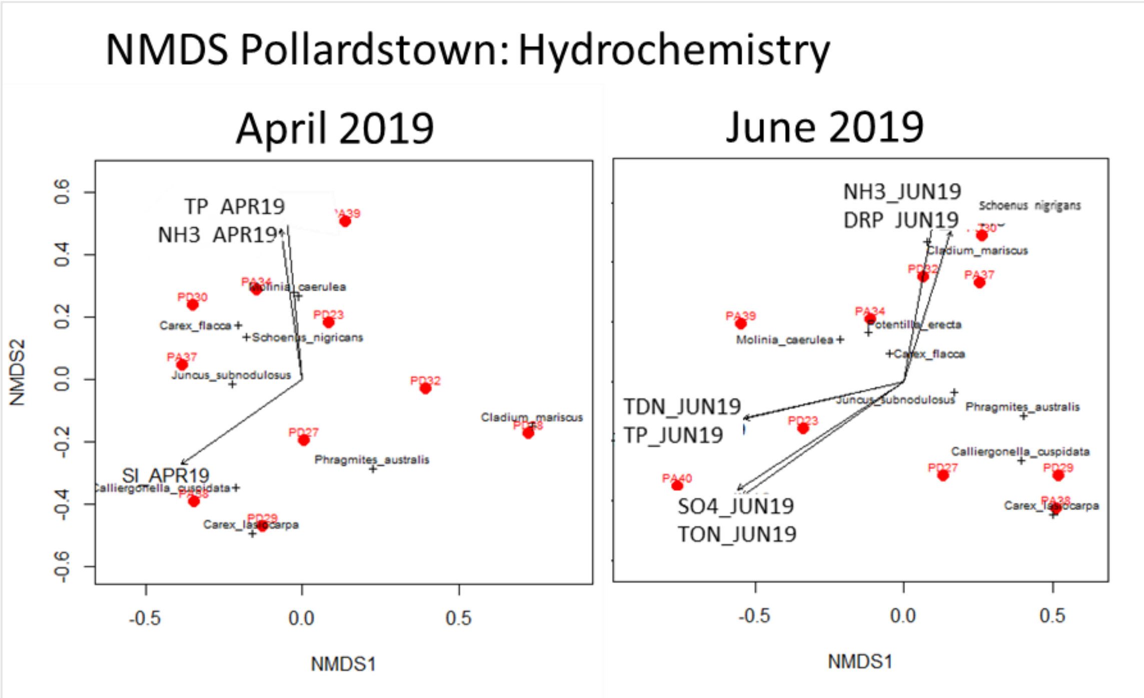


Figure 6.39. Multidimensional Scaling ordination of dimensions 1 and 2 with hydrochemistry concentrations plotted as vectors (max p-value = 0.2) in Pollardstown sites A and D. The phreatic and piezometer nest locations are shown in red and the names of the species with the highest abundances (10%) are also plotted.

6.1.3.5. Boxplots hydrochemistry

The boxplots of Figure 6.40 show that almost no DRP is found in Pollardstown site A. However, higher values are reported in the piezometers of habitat PF2 but since the screens of these piezometers are located in the gravel aquifer beneath the peat it seems likely that this value is more representative for the catchment than for the fen itself.

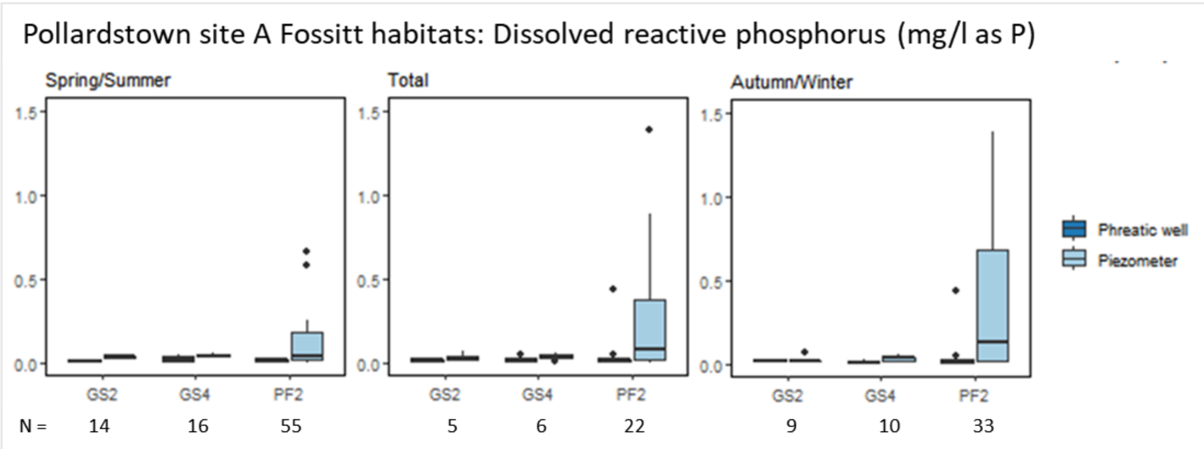


Figure 6.40. Dissolved reactive phosphorus (mg-P/l) in the different habitats of Pollardstown site A sampled from phreatic wells and piezometers.

NMDS plot (Figure 6.41) suggested that PF2 would have higher TP values. This is, however, not the case for the phreatic water table according to Figure 6.35 as both habitats GS2 (Dry meadows and grassy verges) and GS4 (Wet grassland) both show much higher concentrations. The piezometers do show more elevated concentrations in PF2 compared to the other habitats. Interesting to note is that GS2 is has much higher concentrations in the Spring/Summer that in the Autumn/Winter. This could be due to the fact that a typical GS2 habitat has much more herbaceous vegetation which is able take up more phosphorus. This could be true if the habitat consists mainly out of annual vegetation that requires large amounts phosphorus, especially during the growing season (CFF, 2009). Another possibility is that the relatively dry soil in the area holds high TP concentrations which gets flushed out by higher surface water flows during the winter.

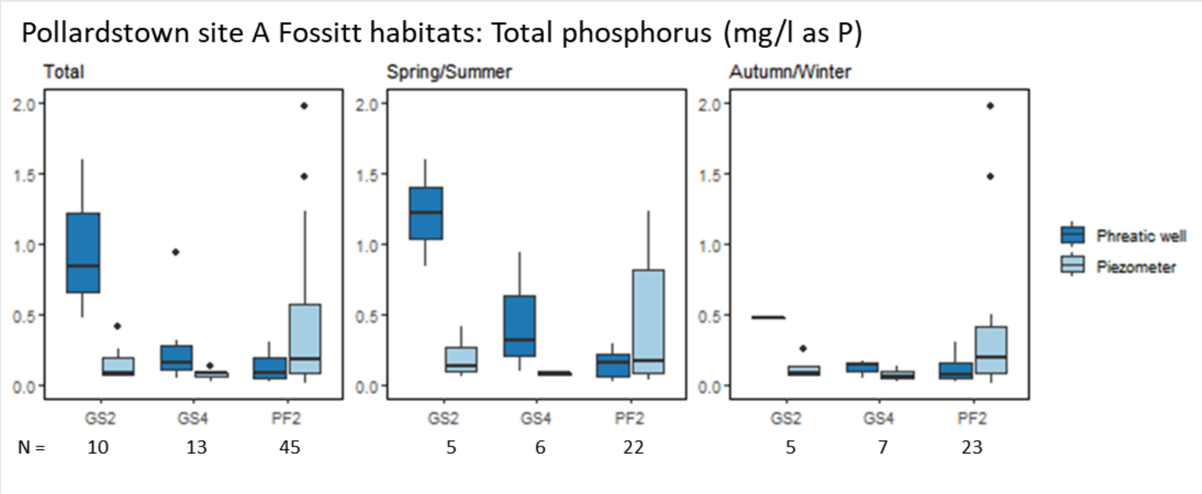


Figure 6.41. Total phosphorus (mg/l as P) in the different habitats of Pollardstown site A sampled from phreatic wells and piezometers.

From Figure 6.42 can be concluded that both the more grassy habitats (GS2 and GS4) contain more ammonia than the fen habitat (PF2). Furthermore ammonia concentrations in the Spring/Summer are higher compared to the Autumn/Winter especially in GS2 and GS4 which could signify either ammonia oxidation in an oxic environment or ammonia oxidation occurring under anoxic conditions (anammox).

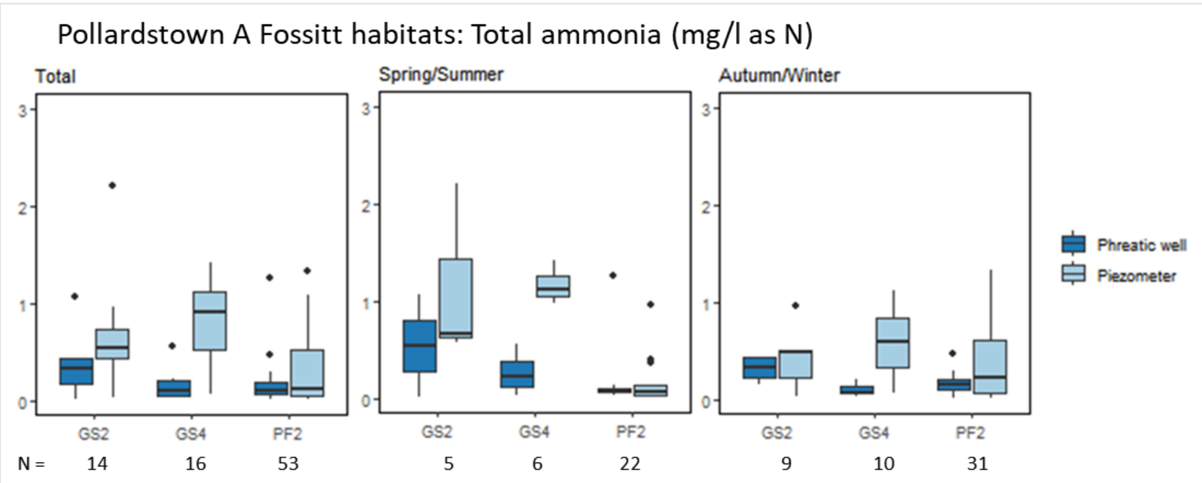


Figure 6.42. Total ammonia (mg-N/l) in the different habitats of Pollardstown site A sampled from phreatic wells and piezometers.

TON was expected to show higher concentrations in habitat PF2 in the NMDS plot. This is however not true for the reported values in Figure 6.43 in the phreatic water table. However, the piezometers show higher values but again it is believed that this is more representative of the catchment rather than the fen especially since there is not much seasonal change. Higher values are however found in the phreatic water table of GS2 with much higher values during the Spring/Summer. Since the water table was seen dropping during this season in Figure 6.29 this higher TON could reflect more oxic conditions in which nitrification occurs.

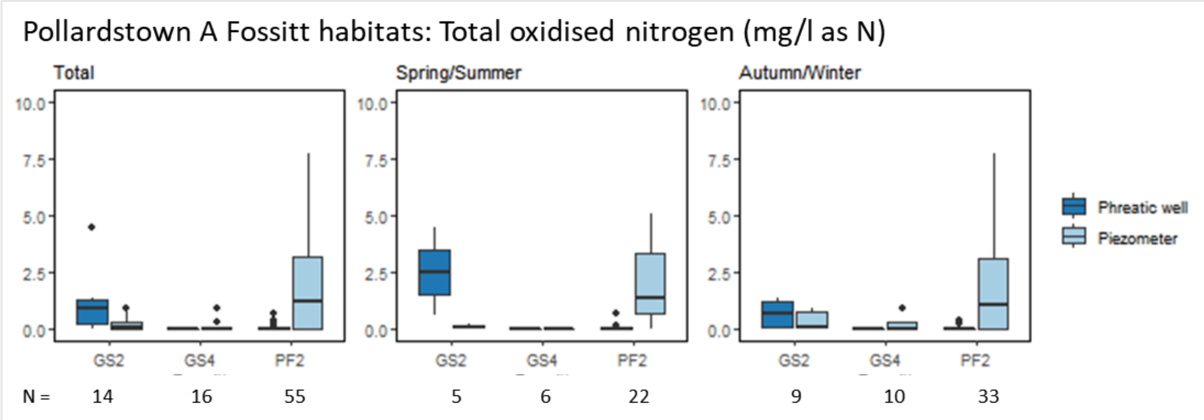


Figure 6.43. Total oxidised nitrogen (mg-N/l) in the different habitats of Pollardstown site A sampled from phreatic wells and piezometers.

From the boxplots in Figure 6.44 the concentrations of total dissolved nitrogen (TDN) in the phreatic tubes are higher in habitats GS2 and GS4 than in PF2 with no clear seasonal change. This is different from what was expected according to the NMDS plot. In contrast, the concentrations of in the piezometers are higher for PF2. These elevated concentrations are often found in gravel aquifers which is where the screens of these piezometers were located.

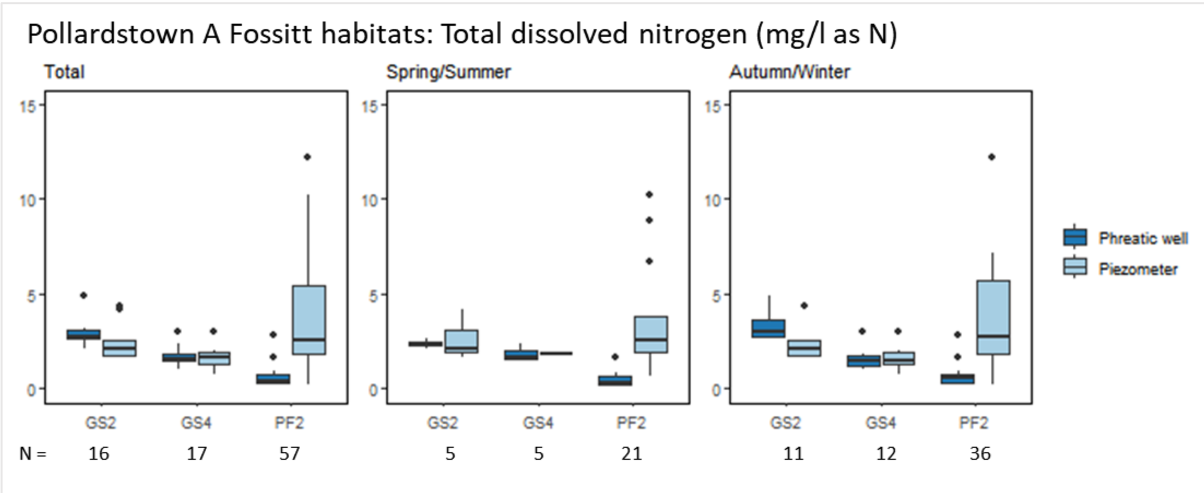


Figure 6.44. Total dissolved nitrogen (mg-N/l) in the different habitats of Pollardstown site A sampled from phreatic wells and piezometers.

According to the NMDS plot, the expectation was that higher silica concentrations would be found in PF2. While this seems true in Figure 6.45, piezometers in habitat GS4 also show the same level. It further seems that there is a lower contribution of groundwater to the phreatic water table in this habitat, according to the low silica values. This seems furthermore likely since the habitat where these concentrations were measured are located closer to the outlet which has drawdown effect on the water table.

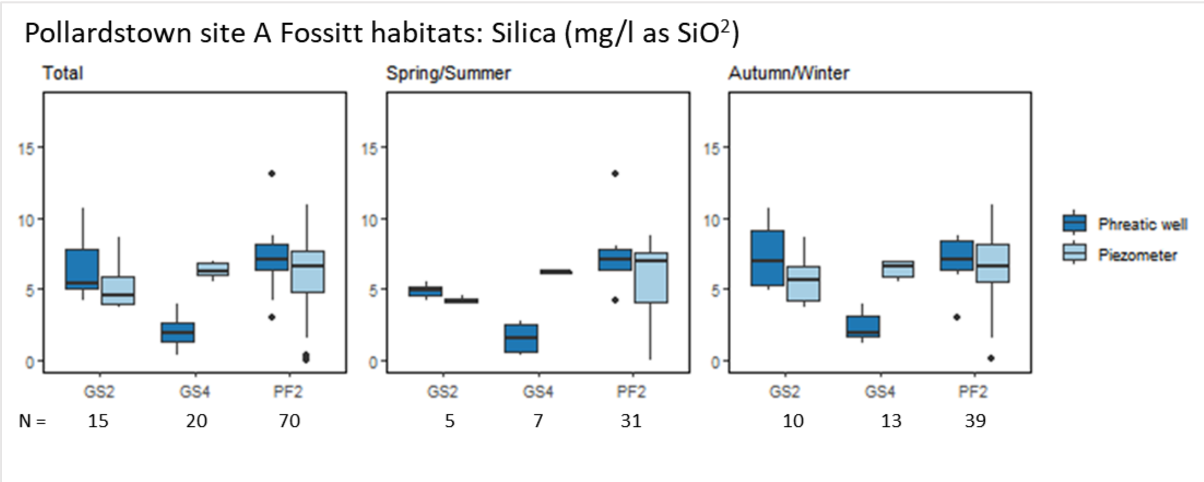


Figure 6.45. Silica (mg/l as SiO^2) in the different habitats of Pollardstown site A sampled from phreatic wells and piezometers.

Again, sulphate concentrations were expected to be elevated in PF2. While they are in comparison to GS4, habitat GS2 report much higher values, especially in the phreatic tubes. The concentrations do not seem to display a particular significant seasonal shift.

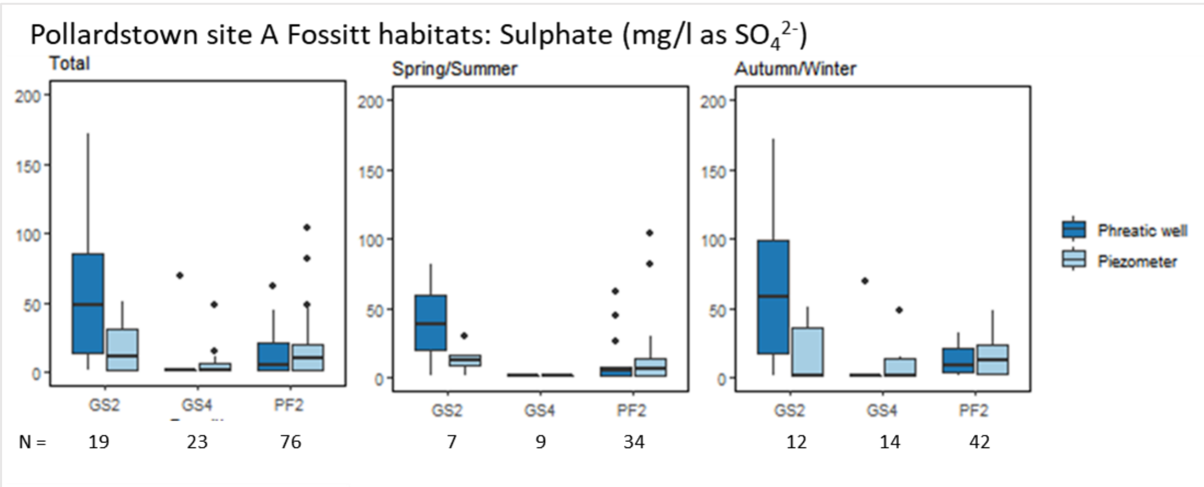


Figure 6.46. Sulphate (mg/l as SO_4^{2-}) in the different habitats of Pollardstown site A sampled from phreatic wells and piezometers.

6.3.3. Mean seasonal hydraulic gradients and hydrochemistry

The following sections bring the knowledge of the hydrology and hydrochemistry together on Pollardstown transect site A. The legend of the soil geology displayed on the transects can be found in Figure 6.47.

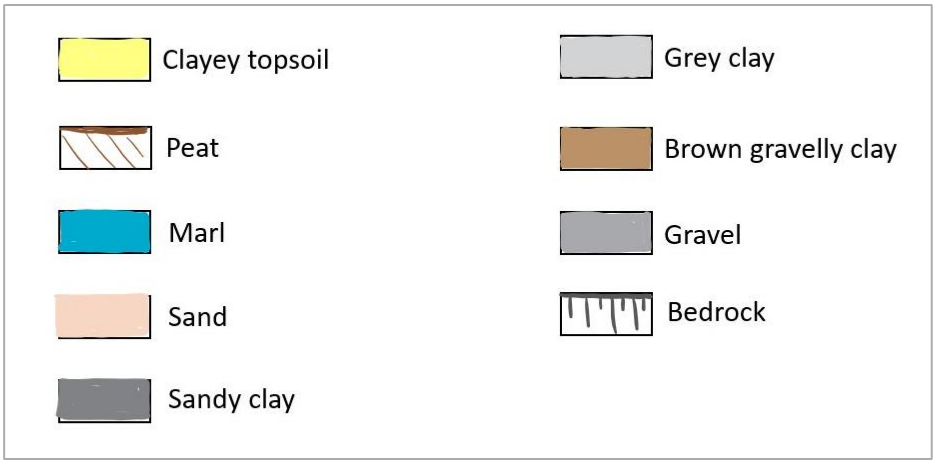


Figure 6.47. Legend explaining the different soil types in the Pollardstown geology transects.

6.3.3.1. Dissolved reactive phosphorus

Figure 6.48 shows the Pollardstown A with data collected in August 2020. The hydraulic gradients show predominant upward gradients on the left were the water is entering the fen. Closer to the outlet more downward gradients are found. It is however more likely that the phreatic water table has a downward gradient because of the surface water leaving the fen via the outlet rather than recharging to the deeper sediment layers.

Furthermore there is an elevated DRP concentration present in the piezometer PA34 with screen of 11 mBGL. This is, however, essentially the aquifer and not reflective of conditions in the fen. Another elevated concentration is found in the piezometer of PA38. Here the upwards conditions imply that this could be leaching in from the grey clay underlying the peat.

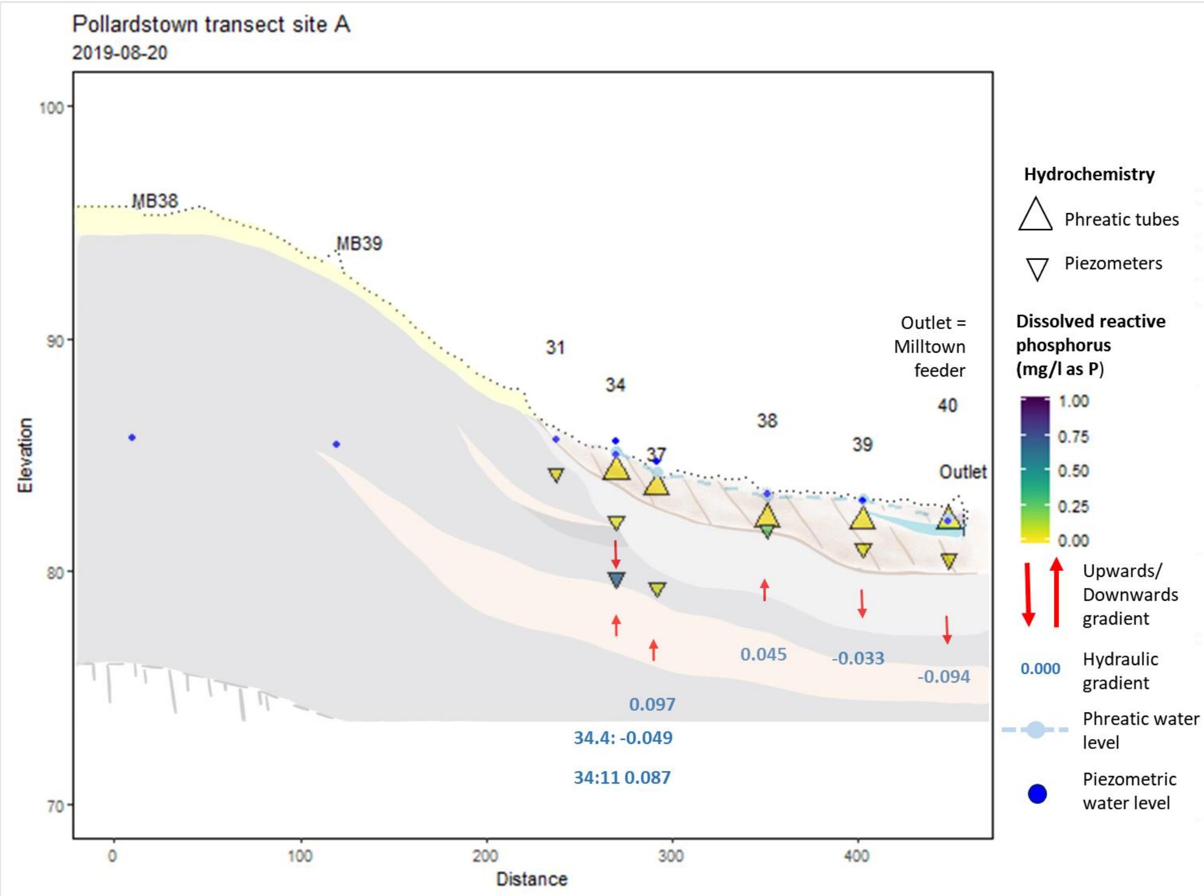


Figure 6.48. Hydrology and dissolved reactive phosphorus (mg-P/l) of Pollardstown site A in August 2019. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

The hydraulic gradient in February 2020 (Figure 6.49) still shows upward gradients on the left with downward gradients on the right with some minor changing. The slightly changing upward and downward gradients are probably caused by the difference in typical conductivities of the lower sediment layers of the fen rather than changes in regional groundwater inputs. Furthermore the elevated DRP concentrations are still found in the same locations and their values have not changed over the winter of 2019/20. Also, this time an elevated value was measured in the catchment (MB38).

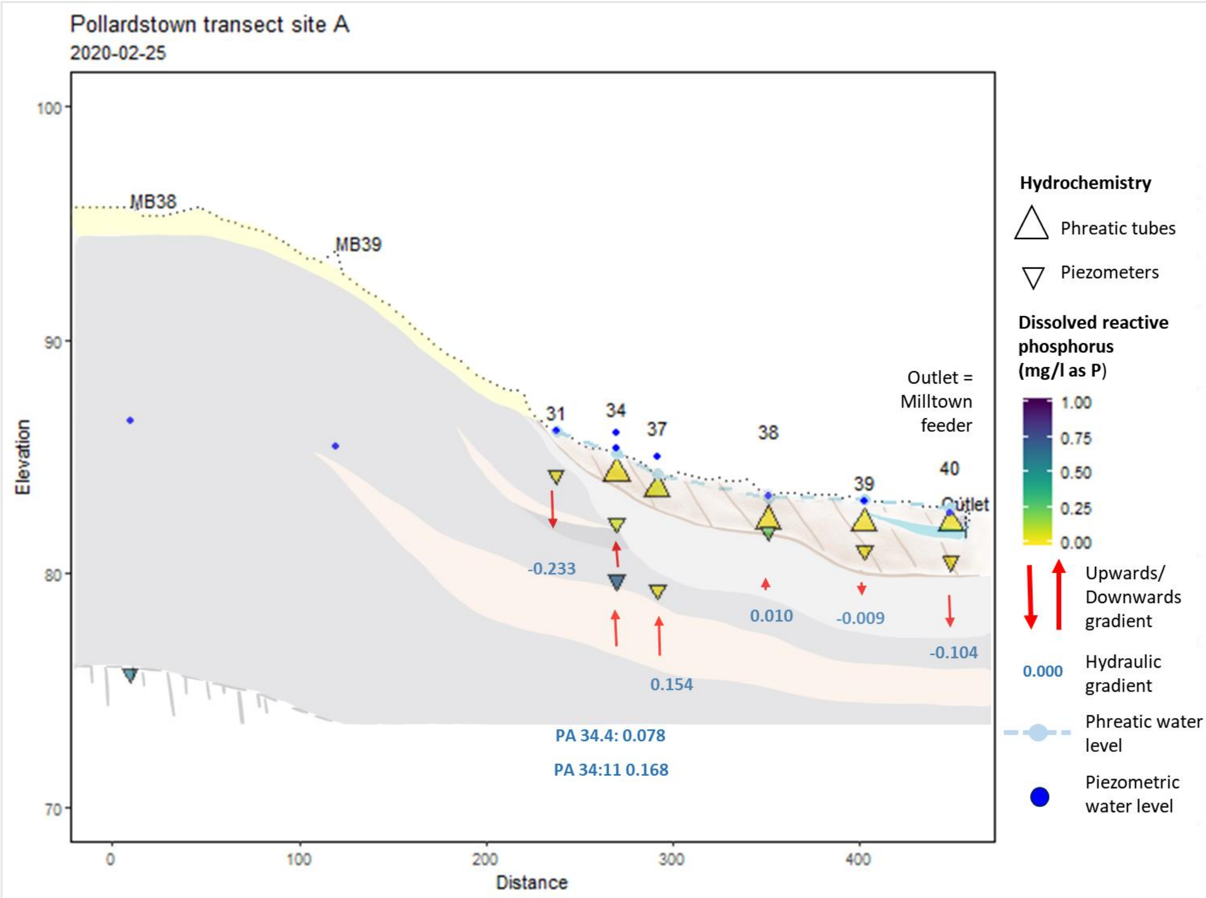


Figure 6.49. Hydrology and dissolved reactive phosphorus (mg-P/l) of Pollardstown site A in February 2020. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

6.3.3.2. Total ammonia

In Figure 6.50 elevated concentrations of total ammonia found in the peat layer in August 2019 which might indicate some denitrifying proccesses under reduced conditions. Furthermore, there are some minor elevated concentrations found in the groundwater under PA31 and PA34. This might indicate a pathway for some of the ammonia flows into Pollardstown site A as the fen is fed by diffuse springs at these locations. It further suggests that nitrate is converted to ammonia within the fen peat (Davidsson et al., 2002; McBride et al., 2010).

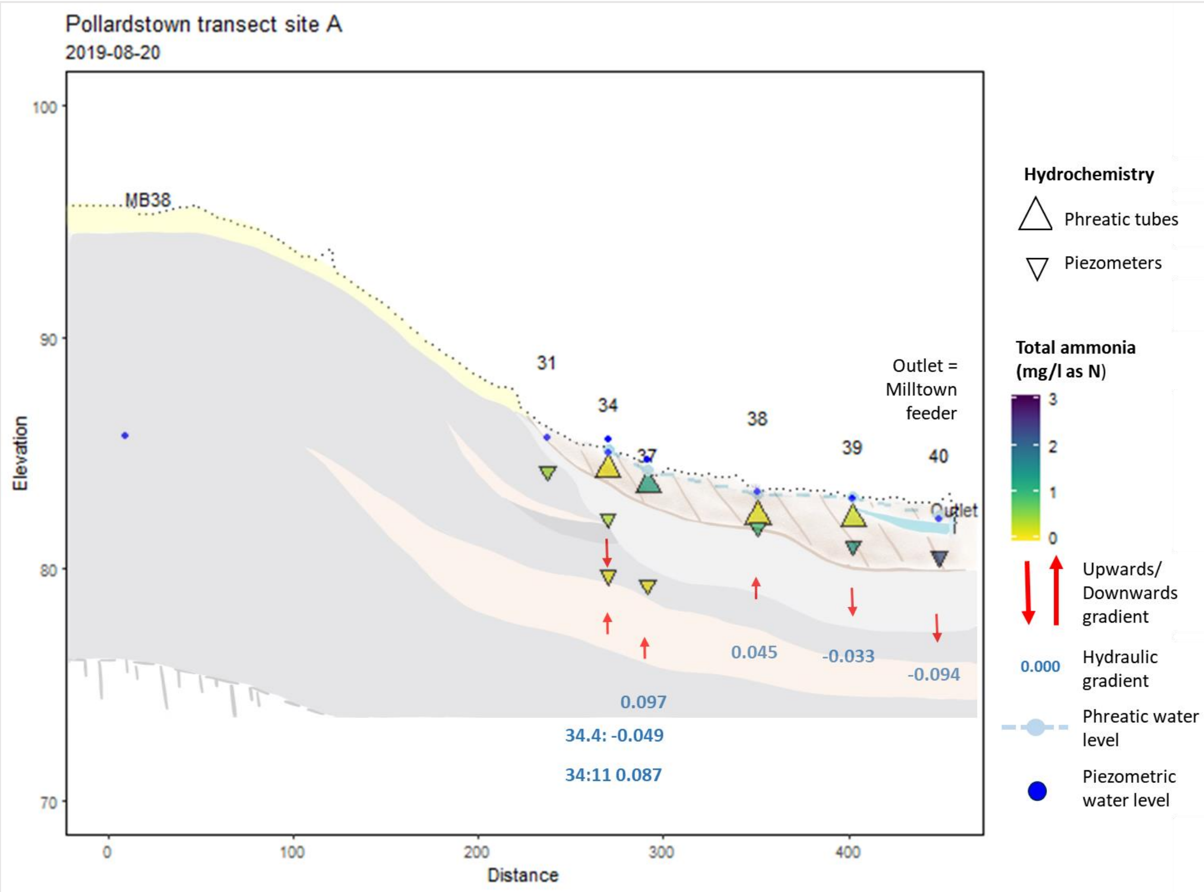


Figure 6.50. Hydrology and total ammonia of Pollardstown site A in August 2019. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

In February 2020 (Figure 6.51) most ammonia seems to be removed from the previous locations with high concentrations. It is likely that that this happened due to higher surface water inputs that dilute ammonia which also creates an oxic environment that promotes nitrification. Some ammonia is still present in the phreatic water table at location PA34. Furthermore a low value was reported in the borehole.

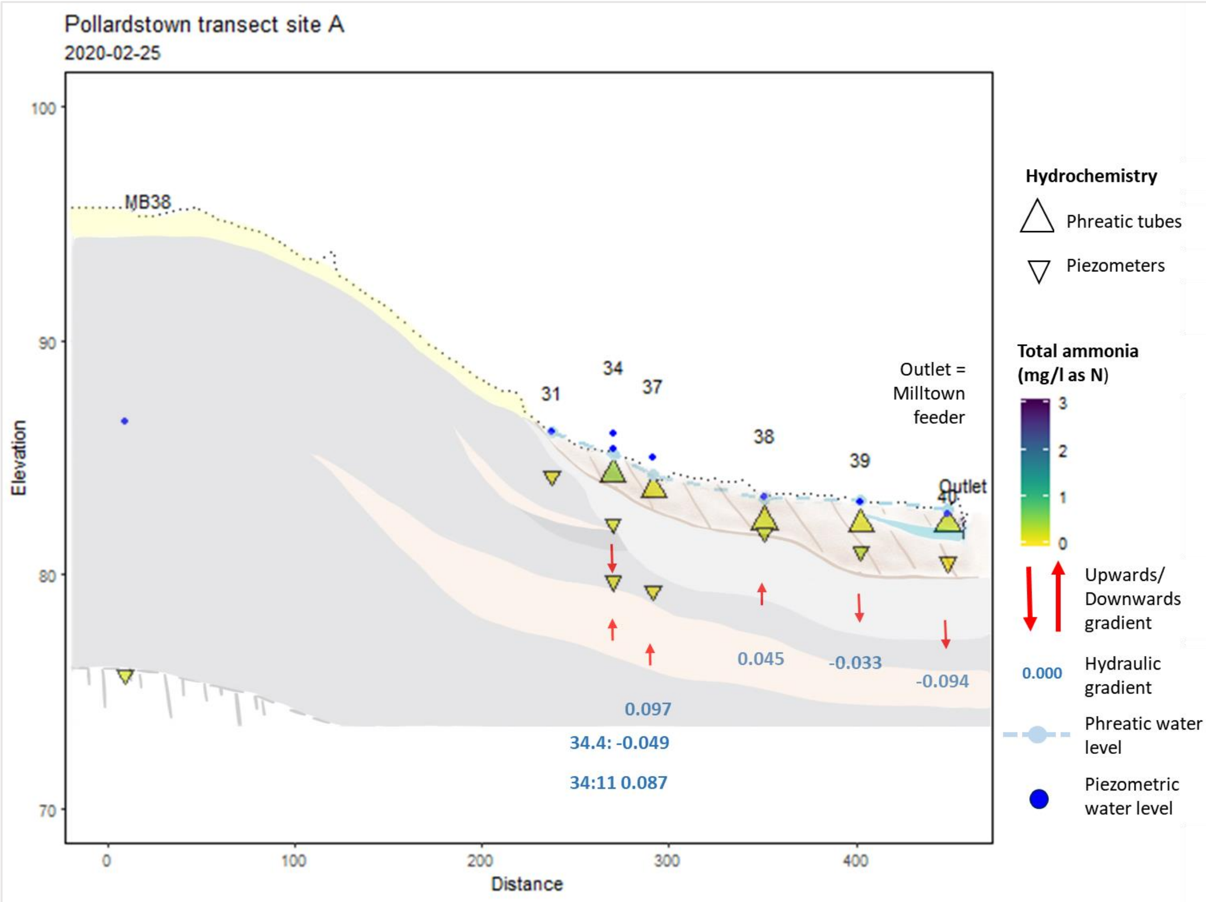


Figure 6.51. Hydrology and total ammonia of Pollardstown site A in February 2020. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

6.2. Conceptual model

6.2.1. Site summary

Pollardstown fen is a 2.7 km^2 site which is supported by a catchment of 32.2 km^2. Pollardstown site A is a small part of this fen (0.07 km^2) and contains poor fen and grassland habitats. Furthermore the site supports a total of 0.02 km^2 which is 31% of the entire site. From the four assessed relevés conducted during the vegetation survey as specified in Section 4.1.3 two failed the fen assessment criteria in Appendix D.

The water balance and discharge rate of the Milltown feeder does not show any changed compared to research conducted in 2003/04 (Kuczyńska 2008) which shows no major hydrological changes have occurred between then and 2019/20. The fen seems to receive most of its groundwater from springs next to the adjacent field which move laterally over the fen with ease which is apparent from the very stagnant levels in the phreatic tubes.

Just like Ballymore, Pollardstown also seems to act a treatment system as apparent from lower nutrient concentrations especially in the phreatic water table. However, it should be taken into consideration that environmental fluctuations may cause the fen to periodically and quickly turn into a source of nutrients to the pore water. This may then in turn be flushed out of the fen via overland flow and outlet(s).

6.2.2. Conceptual model

A conceptual box model is displayed in Figure 6.52, showing the water balance, surface water level fluctuation and median nutrient concentrations in the fen and its catchment.

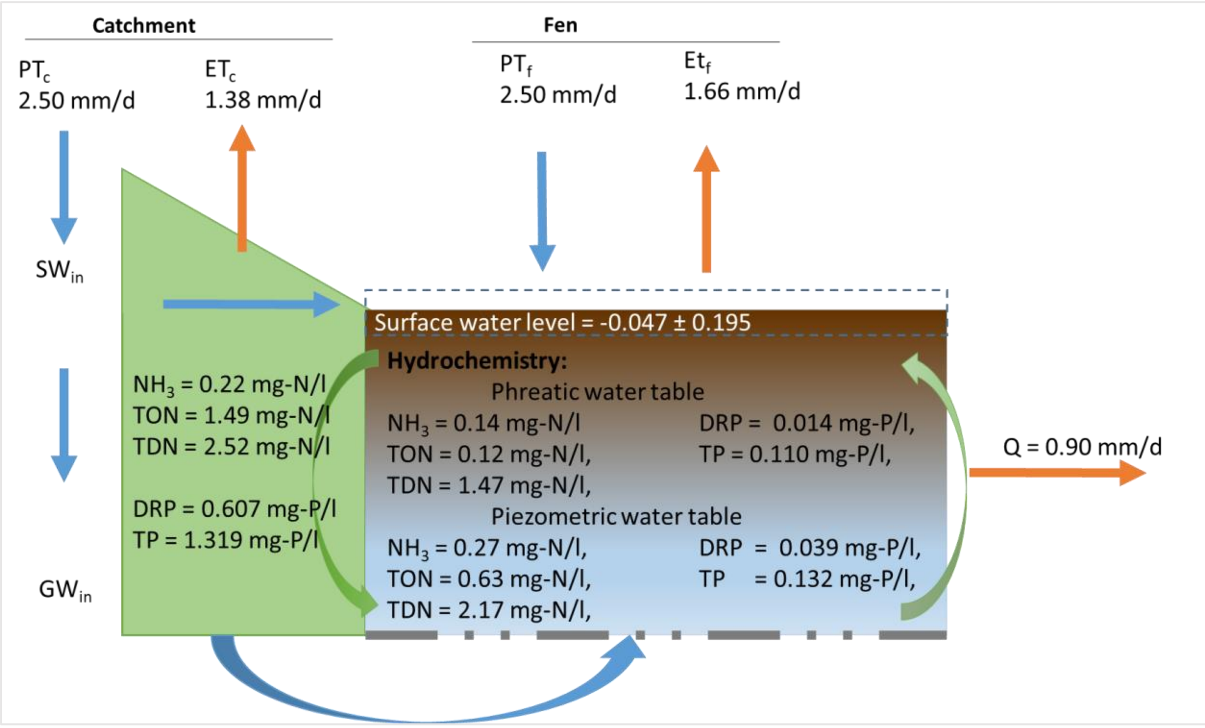


Figure 6.52. Conceptual box model of Pollardstown site A displaying the water balance, surface water level fluctuation and median nutrient concentrations in the fen and its catchment.

7. Results – Pollardstown site D

7.1. Hydrology

The size of Pollardstown site D is only a fraction of the whole of Pollardstown fen. Therefore a water balance of Pollardstown was presented in Section 6.1.1. The current runoff from the fen and its comparison the data previously collected can be found in Section 6.1.2. Again, the phreatic and piezometric water levels will be compared with data from a previously collected dataset (Kildare County Council, 2003) in order to find any changes that may have occurred over time.

7.1.1. Piezometer and phreatic tube data

Surface water points in the fen and groundwater table points around the fen were interpolated into contour lines in order to interpret the flow in and out of the fen in Figures 7.2 and 7.3. Figures 7.4 and 7.5 show the water levels recorded in the phreatic tubes and piezometers between July 2018 and October 2020. The locations of the sites where this data was collected can be found in the Pollardstown site D instrumentation map in Figure 7.1.

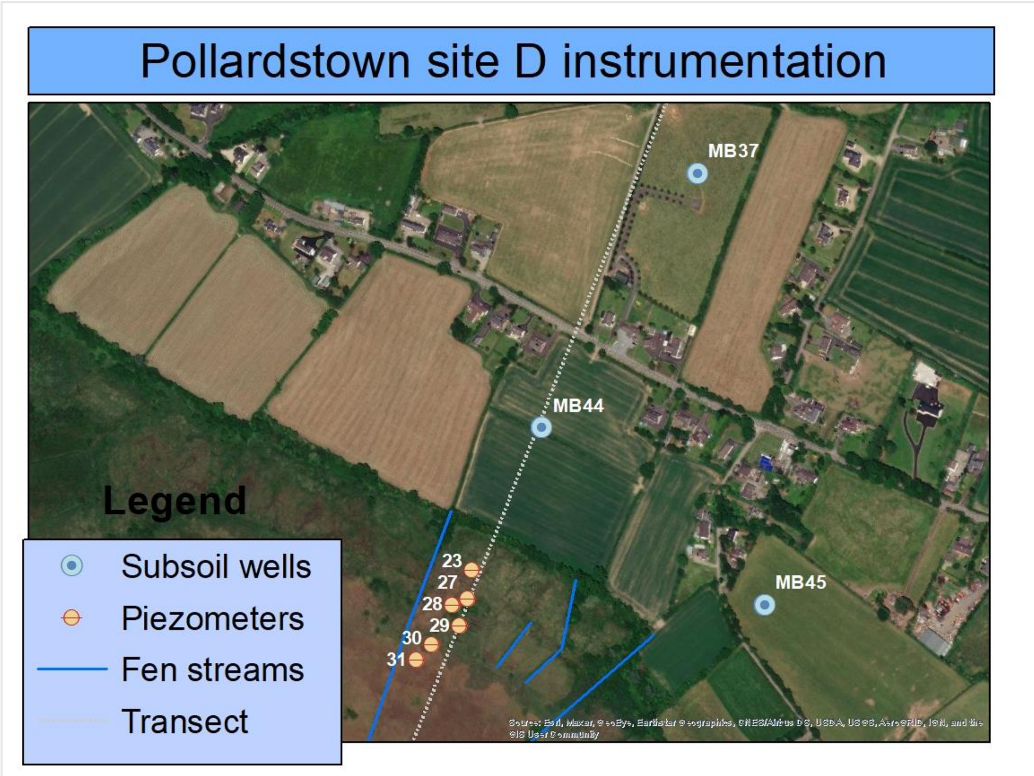


Figure 7.1. Pollardstown site D instrumentation map showing fen piezometer and phreatic tube locations, subsoil well locations and the main site drains.

Lateral flows are observed from north to south eventually ending up in the Milltown feeder (Figures 7.2 and 7.3). There seems to be a small shift between the winter and summer, where higher levels are observed during the winter further into the the fen (south end of the map) as opposed to the summer. This relative small change shows the resilience of the fen during seasonal groundwater changes. The contours also indicate that groundwater seems to be dispersed evenly after flowing upwards to the surface of the fen where it mixes with rainwater before discharging in the southern outlet. It has to be noted that these contours, however, are a small subset of the the 'catchment' as shown on the map of boreholes in Section 3.3.

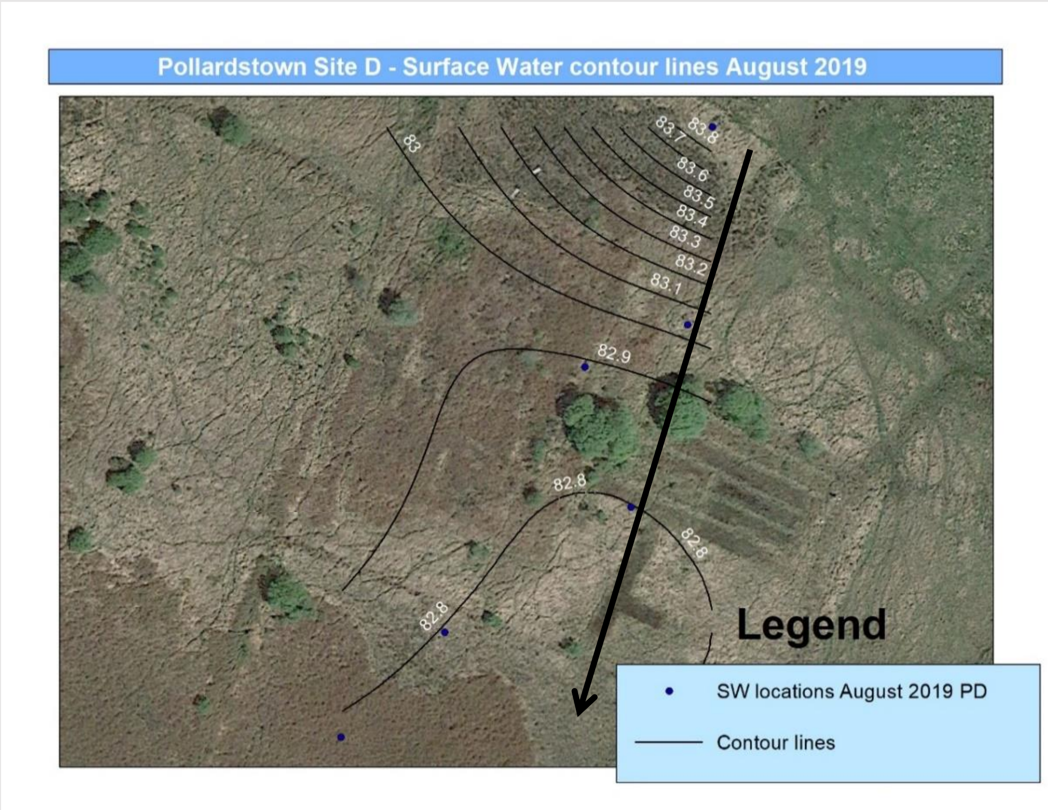


Figure 7.2. Contour lines of fen surface water and surrounding groundwater catchment interpolated using piont measurements in August 2019. Flowlines are presented with black arrows.

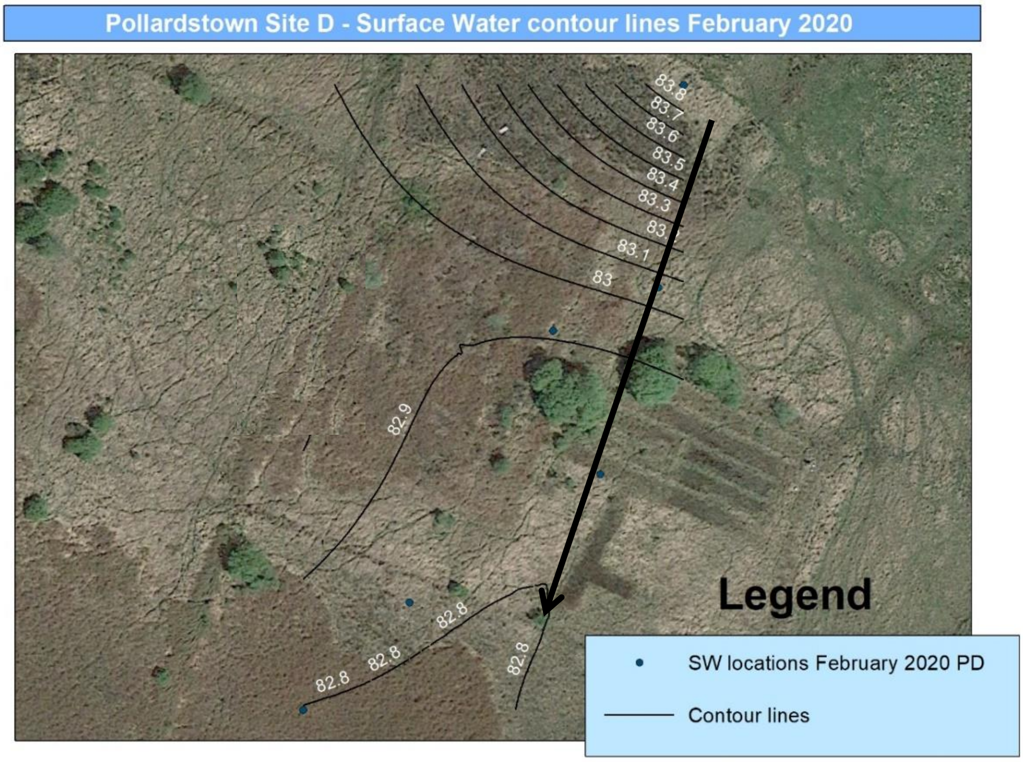


Figure 7.3. Contour lines of fen surface water and surrounding groundwater catchment interpolated using piont measurements in February 2020. Flowlines are presented with black arrows.

The water levels in the phreatic tubes (Figure 7.4) of all sites are very stable throughout the hydrological years and do not seem affected much by periods of high effective rainfall. Futhermore, no drawdown effect was observed since there weren't any drains close to the measured locations. However, the dry summer of 2018 did seem to have an effect on the water levels in PD27 and PD29. Both phreatic water table can be found recovering from lower levels recorded in the summer of 2018. In PD27 this took until November until the water levels were stable again. During this time the water level rose by approximately 0.5 m. The water levels in PD29 took longer to recover; between August 2018 and April 2019 the water levels rose by 0.13 m. PD23 is located on the downward sloping hill along the transect and its surface elevation is one metre higher here than in the other locations. Even so the phreatic surface water level is found at approximately 0.1 m above the surface throughout both hydrological years (see Figure 7.30).

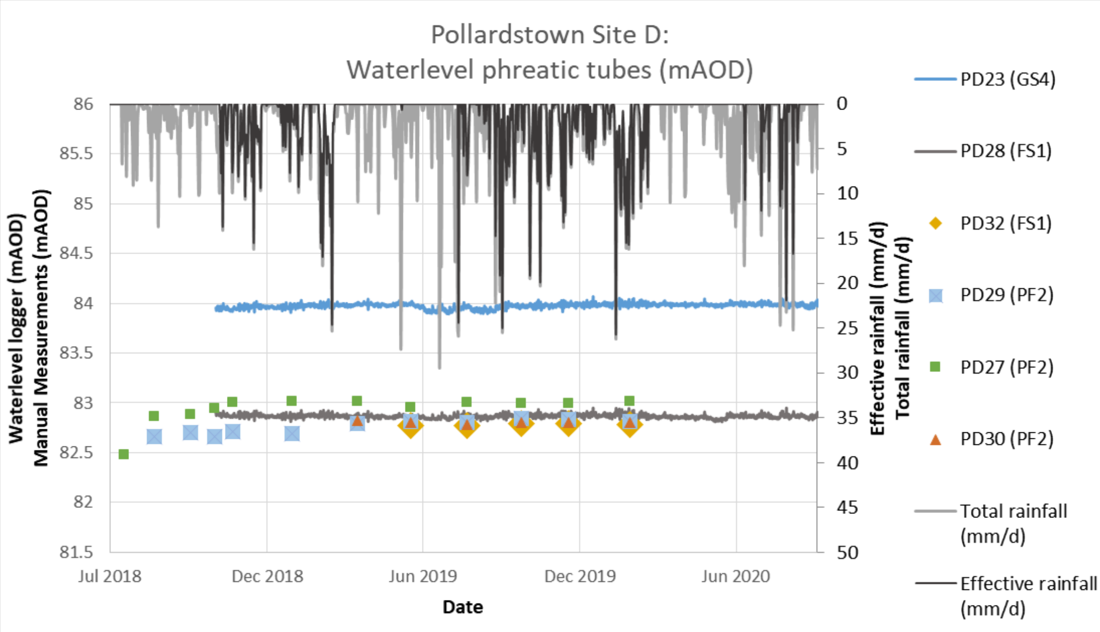


Figure 7.4. Phreatic water level hydrograph of spot measurements and water level loggers and rainfall.

The piezometers of PD27.4, PD30 and PD29 also showed a water level recovery after the dry summer of 2018 as described before (Figure 7.5). An increase of approximately 0.4 m between July and November 2018 is observed amongst these piezometers. A clear reaction to effective rainfall on the piezometric water table is seen in PD23.4 (screen depth of 4 m) and PD28.11 (screen depth of 12 m) with water levels increasing during wet seasons.

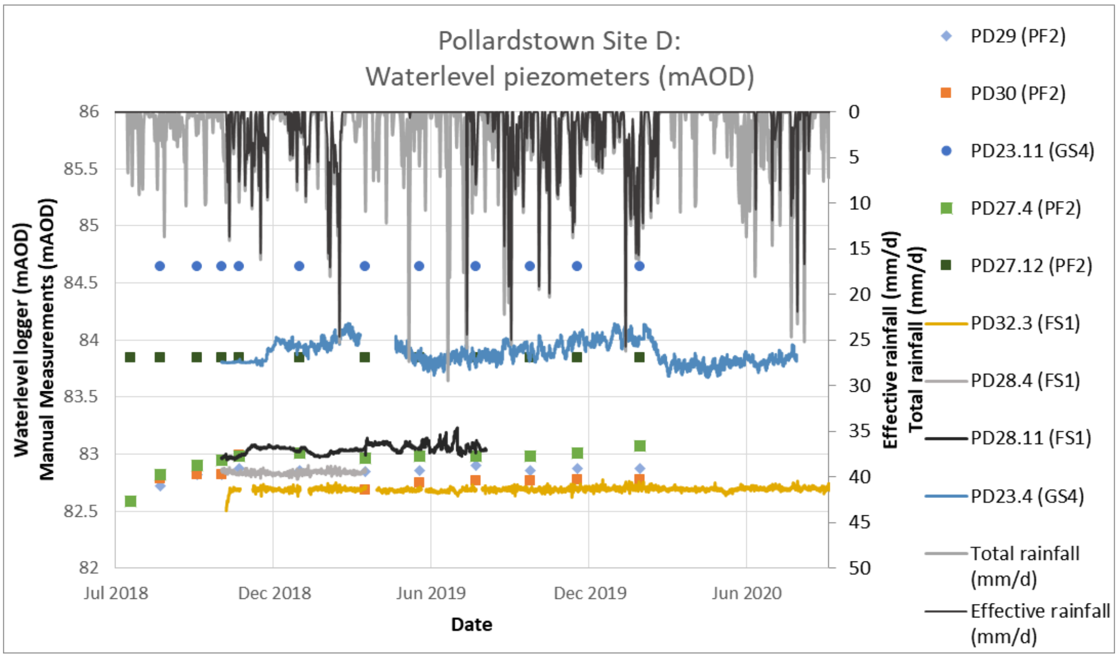


Figure 7.5. Piezometric water level hydrograph of spot measurements and water level loggers and rainfall.

Additionally, some artesian conditions are observed in piezometer PD28 as the piezometric head is much greater that the phreatic head at this location. This explains why there is no decreasing trend found during the months with no effective rainfall. Both PD23.11 (screen depth of 11 m) and PD27.12 (screen depth of 12 m) are showing artesian flow conditions. Here the water levels were observed at levels of approximately 0.8 m above the surface elevation. The screens of the piezometers are located in the confined aquifer containing groundwater under positive pressure, which caused this extreme observed upwards flow. Since the piezometers were overflowing at all times, as observed in Figure 7.6, the waterlevels were measured at a stagnant level. It is expected that these levels will follow the regional groundwater table provided the piezometer is long enough to reach the aquifer.

Figure 7.6. Artesian flow conditions in Pollardstown D. The mounds around the piezometer is tufa which is created by precipitation of carbonate minerals in the groundwater.

The current levels in Figures 7.4 and 7.5 show that PA23.4 seems to follow the seasonal change recorded in Figure 7.7 in the same magnitude. This is however not true for PA23.1.5 where the current phreatic water table changes only by a few centimetres. Furthermore, the current levels seems to have increased somewhat as in both phreatic well and piezometers as the seasonal high was around 86.4 between 2001 and 2007 and stands now at 86.7. From comparison of older data in location PA28 it seems that levels have roughly stayed the same since 2002-2007. Only fluctuations up to 0.1 m were observed.

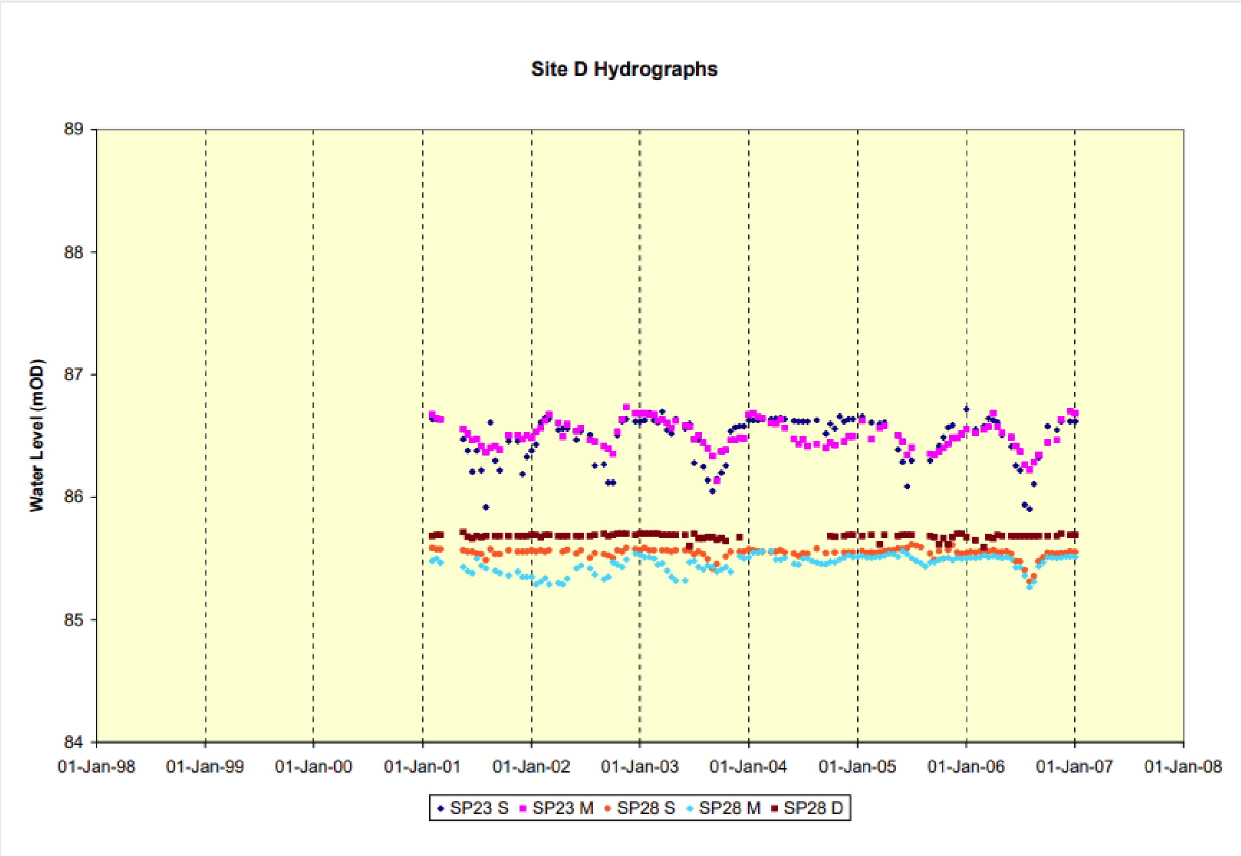


Figure 7.7. Phreatic and piezometric hydrographs measured in Pollardstown site D between 2001 and 2007 (Sholl, 2007). SP stands for standpipe and the number point to the location. The scale of the measurements is in m OD according to the head in Poolbeg. SP23 S can be compared to PD23.1.5, SP23M to PD23.4, SP28 S to PA34.4, SP28 M to PA28.4 and SP28 D to PA28.11.

A time series of water levels in the fen are compared to levels in the surrounding catchment in Figures 7.8 and 7.9. The phreatic and piezometer water levels at site 23 are compared in Figure 7.8. They are located in the middle of the fen about 60 m south from the adjacent field. Multiple discrete springs feed the fen close to this site which are visible on the surface. One piezometer has been installed in the subsoil below the fen peat and has a screen depth of 4 mBGL. The other is an even deeper piezometer with a screen depth of 11 mBGL which is tapping into the underlying aquifer with artesian conditions. The subsoil well MB45 in the surrounding catchment is located 368 m from the fen site and is compared the water levels in the fen.

It seems that the water level is very stable in the phreatic tube whereas piezometer 23 with its screen at 4 m depth follows the ground water trend observed in MB45. Piezometer 23 always displays artesian conditions since the screen (11 m BGL) is located into the aquifer of gravelly clay at this point.

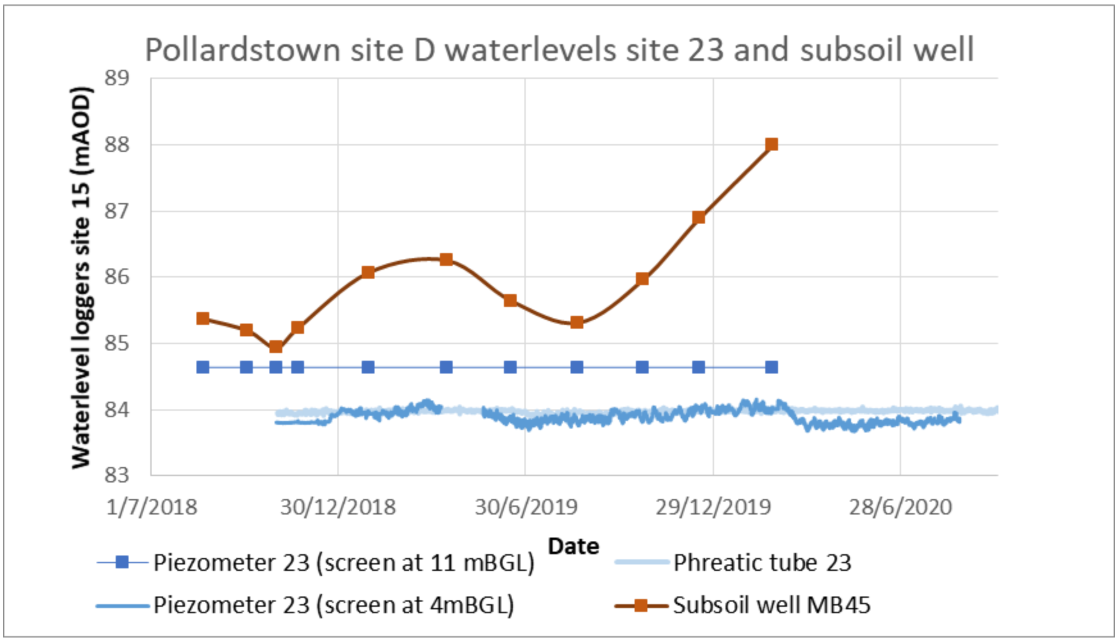


Figure 7.8. Hydrograph of phreatic and piezometric water levels at site 23 and piezometric water levels in subsoil well MB45

Figure 7.9 shows the phreatic and piezometric head recorded in site 29. This piezometer has been installed in the fen peat and has a screen depth of 2.45 mBGL. The site is located about 150 m down from the upper fields. The subsoil well MB45 is located 416 m from the piezometer nest. Neither the water levels in the phreatic well nor the piezometer is controlled by the fluctuations of the groundwater table. However, both do seem to display an increase of water levels after the dry summer of 2018. While both piezometer screens are installed in the peat layer, their water levels take a lot longer to recover than the phreatic tube.

Both phreatic wells of PD23 and PD29 display fluctuations of only a few centimetres throughout the years (2018-2020). This indicates that water recharging the fen via diffuse springs is then able to move relatively easily over and through the upper layers of the fen at this Pollardstown site D.

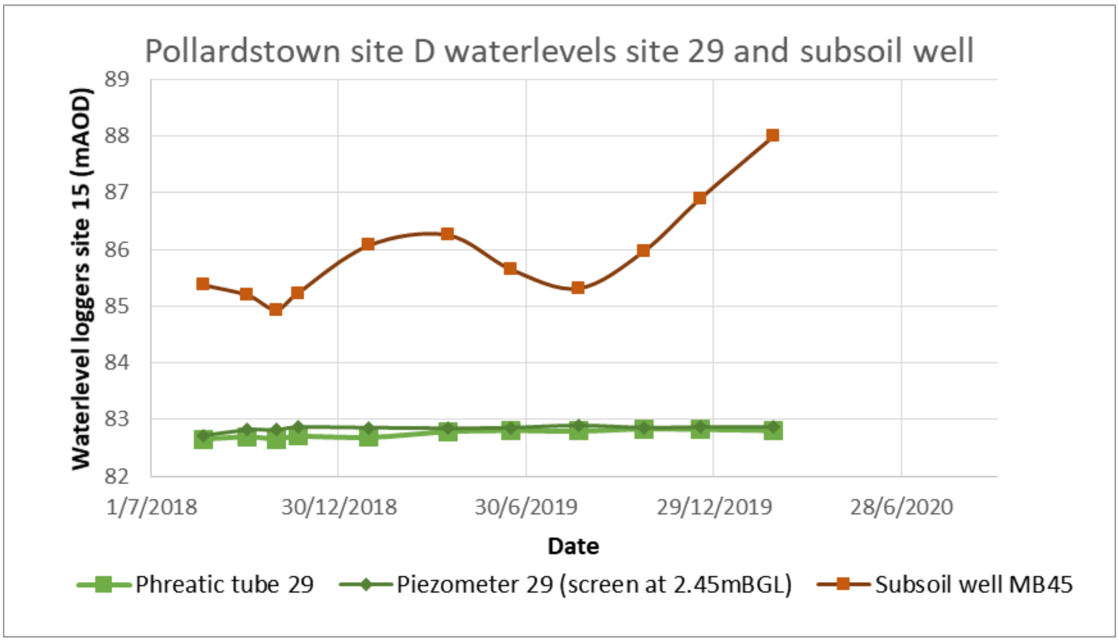


Figure 7.9. Hydrograph of phreatic and piezometric water levels at site 29 and piezometric water levels in subsoil well MB45

The current water levels measured in subsoil well MB38 were compared to data measured between 2002 and 2007 (Figure 7.10). Seasonal low measurements were found to be around 87.9 m OD (Poolbeg) in the older data. These levels are very comparable to the seasonal lows measured in 2018 and 2019 (87.8 m OD) which means no hydrological change has occurred between 2002 and 2019.

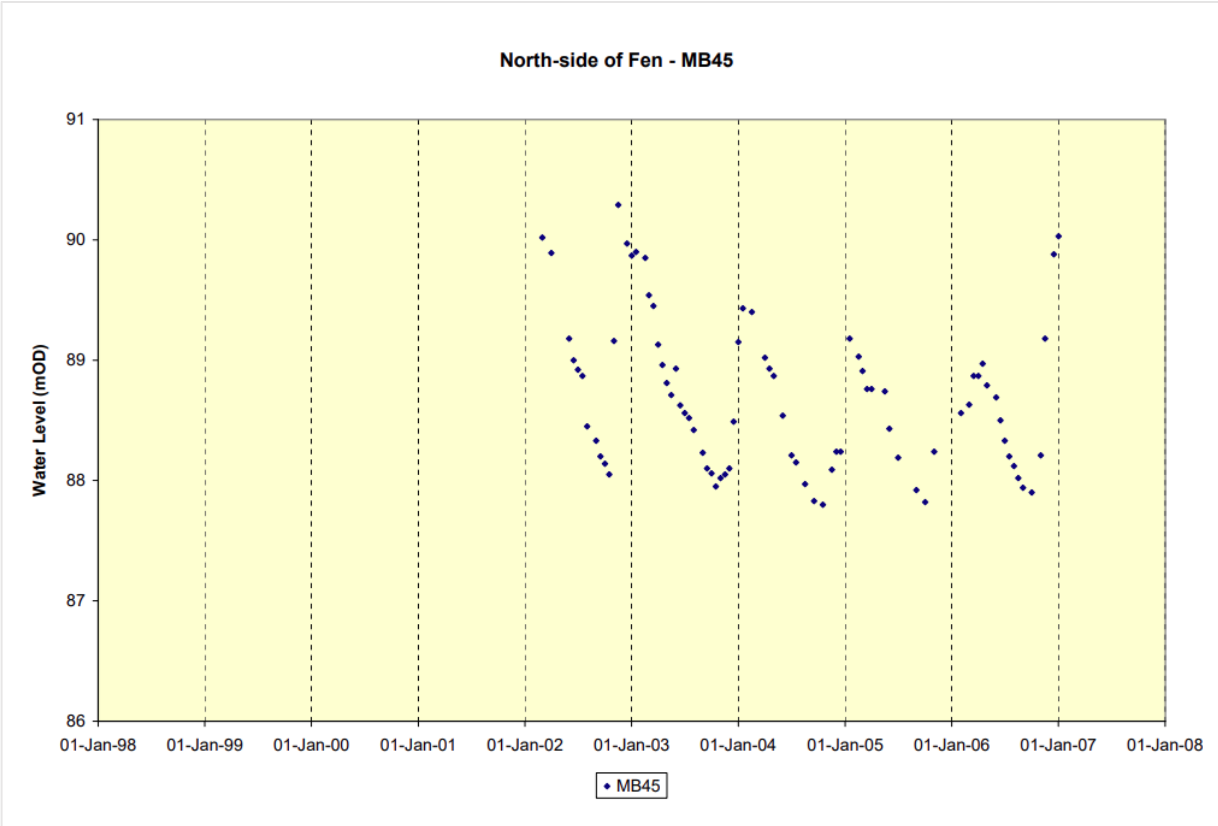


Figure 7.10. Hydrograph of a borehole measured on the north-side from Pollardstown fen between 1998 and 2007 (Sholl, 2007).

7.1.1.1. Electrical conductivity

The time series in Figure 7.11 displays the electrical conductivity in the phreatic tubes of Pollardstown coupled with rainfall. It does not seem that effective rainfall has a direct effect on the recorded EC as no strong increasing or decreasing trends can be observed during periods of high effective rainfall. Therefore it can be assumed that the phreatic water table of Pollardstown site D is mostly fed by groundwater at a reasonably even rate throughout the year.

There are a few phreatic tubes that show a decreasing trend of EC in 2019. Both PD27 and PD29 seemed to have a more surface water flows mixed in with groundwater inflows during the winter of 2019/20. It could be that excess rainfall pools on the fen's surface and flows overland at these locations. In fact both of these locations are quite close to old peat cuttings, as reported by Kuczyńska (2008) in an older habitat map, which could explain the rain water pooling.

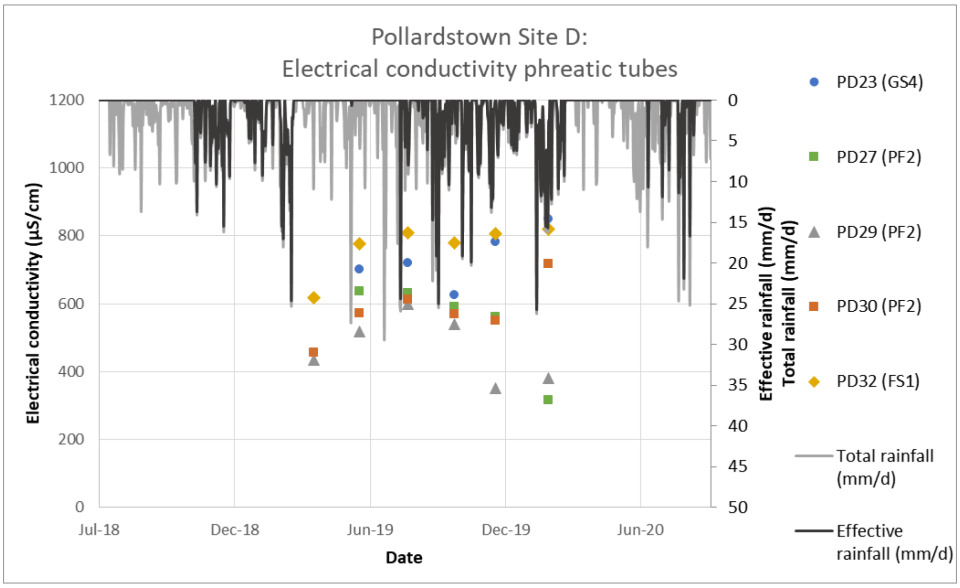


Figure 7.11. Time series of electrical conductivity (μm/cm) in phreatic tubes.

In Figure 7.12 again no significant fluctuations are measured in the piezometers in relation to effective rainfall. Piezometers PD29 and PD27.4 with a 2.45 and 4.0 m screen respectively seem to hold water with a mixed source of groundwater and surface water as the EC is much decreased in comparison to the other piezometers. Interestingly, the water in the screen of piezometer PD30 seems to receive a lot of its water from the surface. It could be that the deeper sediment layers are still affected by the old peat cuttings (as reported in a 2003 habitat map of Pollardstown Fen (Kuczyńska 2008)) a few meters north from this location. Additionally, there may exist preferential flow channels or cracks in the peat which were caused by the dry summer of 2018.

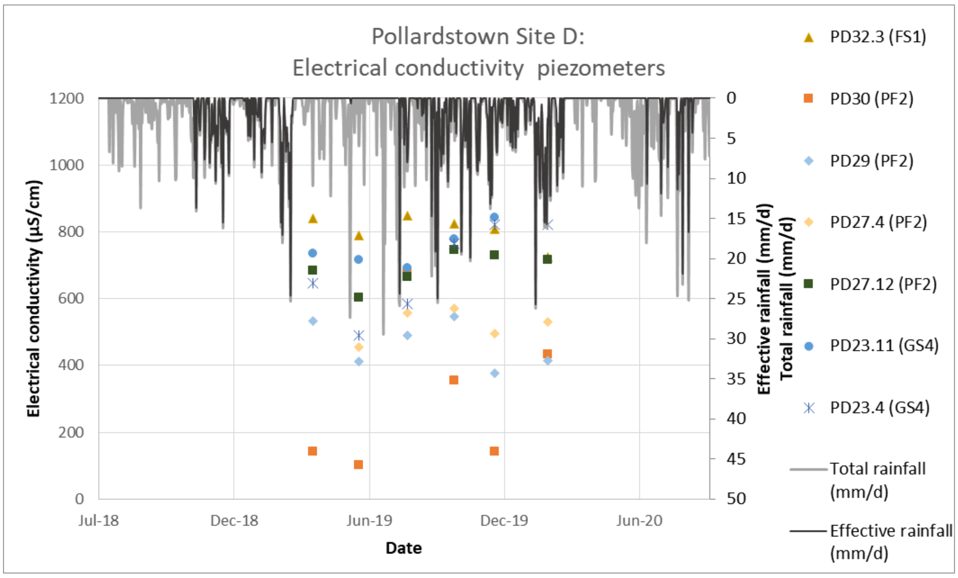


Figure 7.12. Time series of electrical conductivity (μm/cm) in piezometers.

The boxplots in Figure 7.13 show overall and seasonal electrical conductivity from data collected in the fen as well as from boreholes outside the fen. The boxplots show clear contrast between the boreholes with very little fluctuation in values, compared to the the piezometers and phreatic points in the fen which exhibit much more fluctuation. However, the overall EC in the boreholes is measured statistically significantly higher than in the fen itself. This was proven with a Welch test with p-values of 0.00 for both phreatic tubes and piezometers.

These values are different from the EC found in Pollardstown site A where it was implied that this part of the fen is largely fed by groundwater. The difference in magnitude in suggests that Pollardstown site D also receiving water also from another source. It is very likely that the fen receives a portion of surface water runoff. Indeed constant lower values are observed in phreatic water tables of PD27 and PD29. It seems therefore that the surface water runoff may be quite localised and habitat specific. These different observations could be explained by the effect of a rainwater lens. It is possible that the rainwater pools in the upper layer of the peat in some areas because of the low permeability of the peat itself or the underlying sediments.

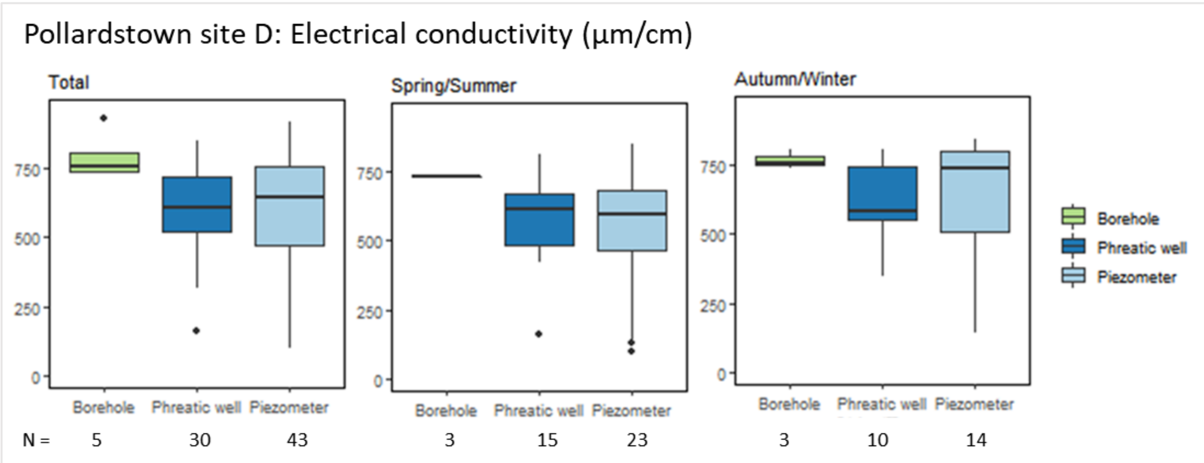


Figure 7.13. Electrical conductivity (µm/cm) in phreatic wells and piezometers inside and boreholes outside the fen in Pollardstown site D.

7.1.1.2. pH

The pH in Figure 7.14 suggest that while values found in the piezometers are found to be very similar to values found in boreholes this is not the case for the phreatic tubes Indeed the median of the boreholes is 7.58 and the median for piezometers in the fen is 7.19. The median of the phreatic tubes was lower with a pH of 7.02. These values were however not found significantly different with a two side Welch test with p-values of 0.53 and 0.63 for phreatic tubes and piezometers respectively.

The pH didn't seem to change in between seasons in the boreholes and the piezometers. The pH in the phreatic wells during the Spring/Summer show some differences between the 1st and 3rd quartile which may suggest that some biochemical reactions are happening here, such as nitrification which acts to reduce the pH. The pH in the phreatic wells during the Spring/Summer show some differences between the 1st and 3rd quartile which may suggest that some biogeochemical reactions are happening here, such as nitrification which acts to reduce the pH (McBride et al., 2010; CPW, 2021).

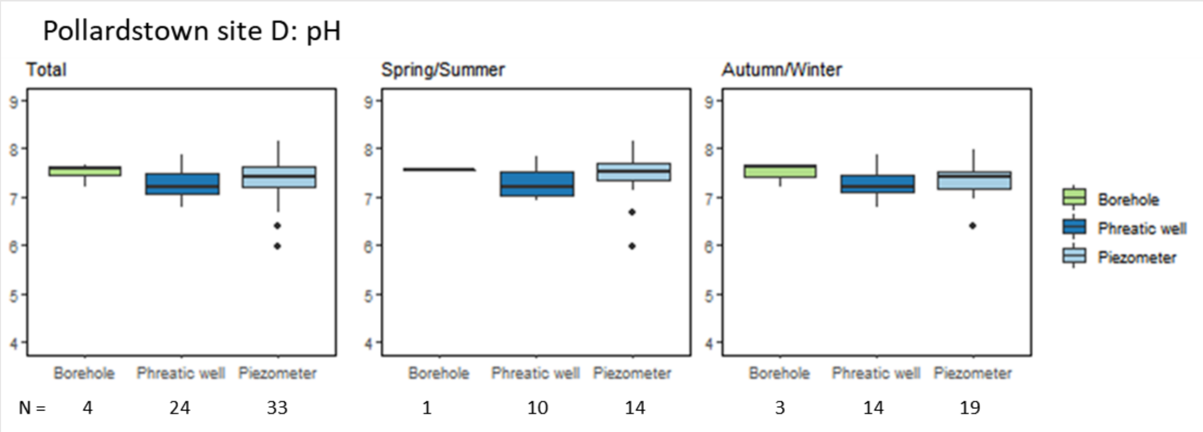


Figure 7.14. pH in phreatic wells and piezometers inside and boreholes outside the fen in Pollardstown site D.

7.1.1.3. Temperature

The temperature boxplots in Figure 7.15 suggest that the temperature in the water column of the fen is lower than the surrounding catchment. The median of the boreholes was 12.7 °C whereas the median for the phreatic tubes and piezometers were reported as 9.3 °C and 9.4 °C respectively. This is again implying that Pollardstown site is also receiving water from the atmosphere rather than solely being fed by the aquifer.

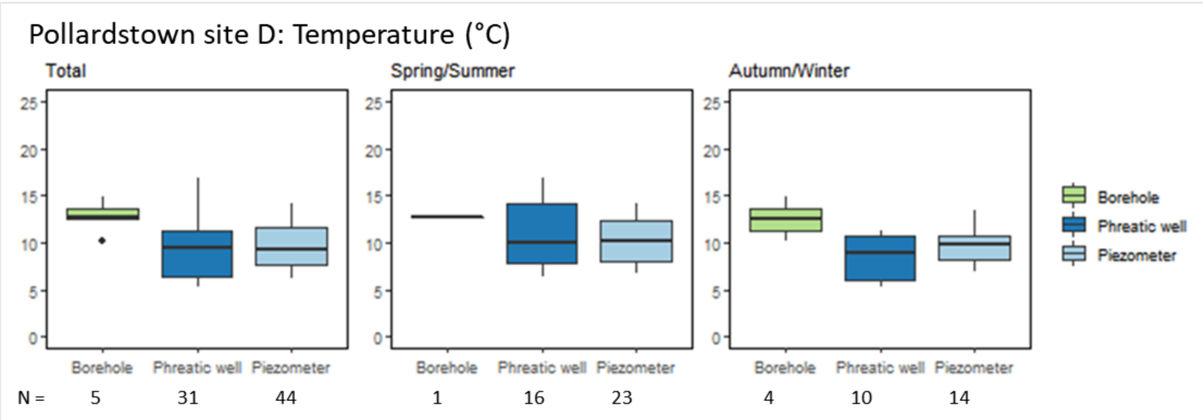


Figure 7.15. Temperature (°C) in phreatic wells and piezometers inside and boreholes outside the fen in Pollardstown site D.

7.1.2. Conceptual hydrology model

- Effective rainfall had no direct effect on the phreatic water levels. Furthermore there were no seasonal changes of EC in the phreatic as well as the piezometric water table implying that the phreatic water table of Pollardstown site D is mostly fed by groundwater at a reasonably even rate throughout the year.
- However some piezometer locations (which are located close to old peat cutting) seem to be fed by a significant portion of surface water as well.

7.2. Hydrochemistry

A total of 134 samples were collected from boreholes, phreatic tubes and piezometers and subsequently analysed for phosphorus, nitrogen and other hydrochemistry. The total collected data set are displayed as well as the seasonal differences between the spring and summer with samples collected between April 1st and September 30th and the autumn and winter with samples collected between October 1st and March 31st .

7.2.1. Phosphorus

Unlike the boreholes around Pollardstown site A the boreholes in site D have a considerable lower concentration of total dissolved phosphorus (DRP) as seen in Figure 7.16. Even so the values are still found higher around the fen with a median of 0.068 mg-P/l. The concentrations in the phreatic tubes and the piezometers were lower with a median of 0.013 mg-P/l and 0.010 mg-P/l, respectively. The concentrations in the boreholes were significantly greater with a p-value of 0.02 and 0.04 when compared to piezometers and phreatic tubes.

While the concentrations in the fen seemed fluctuating more during the Spring/Summer this was not proven to be a significantly different from the Autumn/Winter with a p-value of 0.62 in phreatic tubes with a p-value of 0.57 in the piezometers.

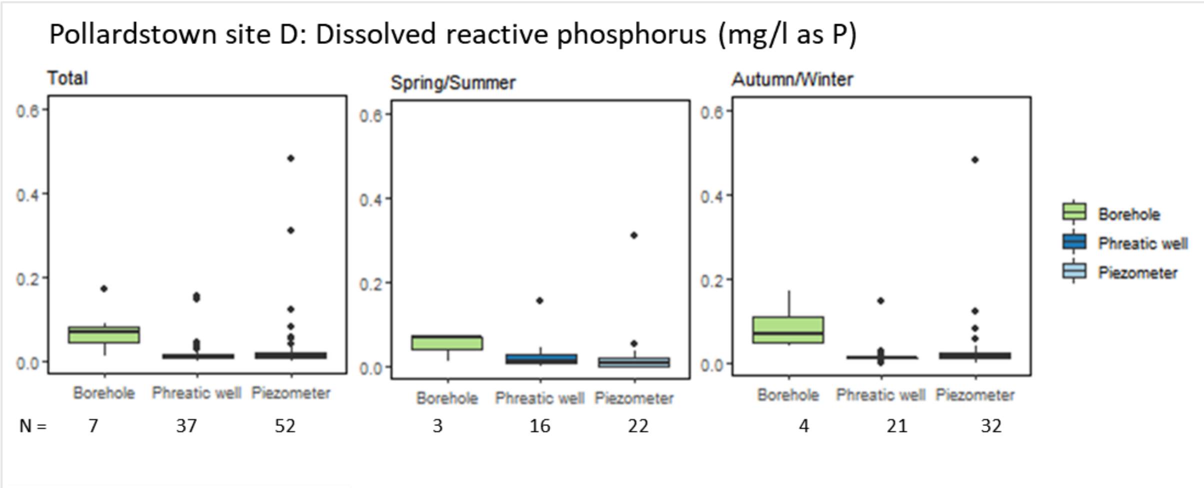


Figure 7.16. Dissolved reactive phosphorus in mg-P/l sampled from phreatic wells and piezometers inside and boreholes outside the fen in Pollardstown site D.

The concentrations of total phosphorus (TP) in the boreholes were reported higher in Ballymore and Pollardstown site A with 0.85 and 1.32 mg/L respectively. This is in contrast to Pollardstown site D were also the values in the regional groundwater catchment were not found higher than in the fen itself (Figure 7.17). It seems that the phreatic water (median of 0.13 mg-P/l) reflects the values found in the boreholes (median of 0.10 mg-P/l). Indeed a two sided Welch test returned with a p-value of 0.19. These values, however, are still relatively high in total phosphorus compared to the other sites. The piezometers are found to have lower concentrations with a median of 0.03 mg-P/l. This was, however, not statistically significantly lower with p-value = 0.29. The higher TP in the phreatic water table is due to phosphorus taken up by vegetation and/or stored in the upper peat layer of Pollardstown site D. This might pose a threat to the vegetation in the future if the fen would experience more extreme water level fluctuations. These fluctuations may bring the phreatic water column in a more aerobic state by mixing the reduced fen water column with oxidising surface water which expedites the release of DRP back into the system. Of course it depends on the vegetation's ability to take up this nutrient whether these phosphorus fluxes have a damaging quality to the fen.

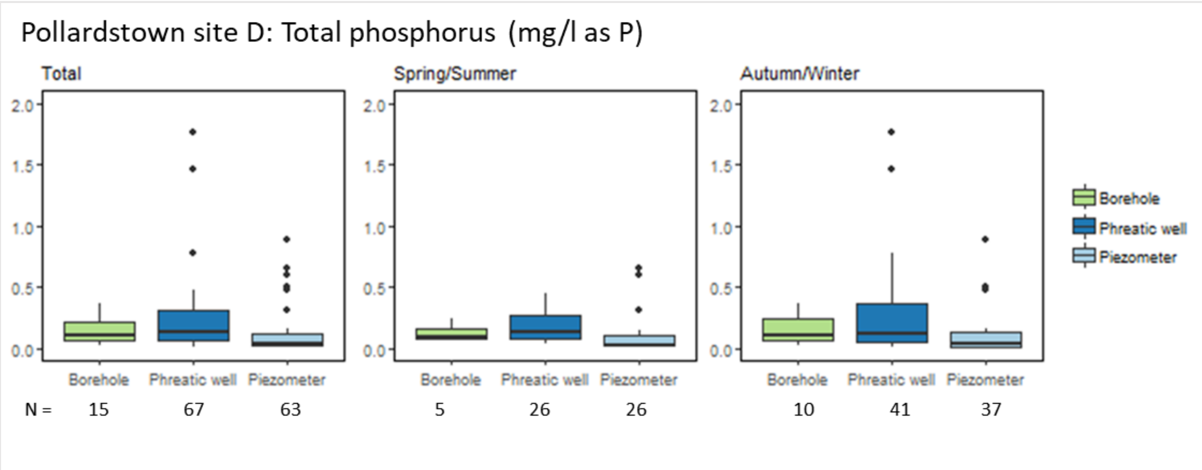


Figure 7.17. Total phosphorus in mg-P/l sampled from phreatic wells and piezometers inside and boreholes outside the fen in Pollardstown site D.

As previously concluded for both Ballymore and Pollardstown site A, the surrounding regional groundwater catchment here in Pollardstown site B cannot be used as a reflection for DRP values found in the fen. This didn't seem the case for TP however as values found in the boreholes surrounding Pollardstown D were reported close to those in the fen. It seems the TP concentrations that flow in from groundwater are stored in the upper fen peat layer. Furthermore, the process of nutrient recycling by vegetation growth in the summer and degradation during the winter may have caused this concentration to appear higher in the phreatic water table than in the ground water table. It could be that the fen vegetation in Pollardstown D (PF2 Poor fen and flush) acts different that the fen vegetation found at other sites (PF2 Rich fen and flush). There may exist a form of quasi-equilibrium whereby the phosphorus coming in and taken up by the plants is being released at the same rates as the decay. This means that there is no net accumulation of phosphorus in particulate form occurring in the fen's peat. Furthermore, the peat is able to sorb and form complexes with phosphorus aiding to this system cycling (Hill et al., 2016, McBride et al., 2010).

The ratios of DRP to TP from the full dataset across the year is 1:1 in boreholes, 1:3 in the phreatic tubes and 1:13 in the piezometers.

When comparing the values collected in and around the fen to the groundwater threshold values of DRP in Ireland (0.035 mg-P/l) it can be concluded that the concentrations are somewhat exceeded in the boreholes around Pollardstown D with a median of 0.068 mg-P/l. This is not the case with the median concentrations found in the fen where the phreatic tubes were found with 0.010 mg-P/l and the piezometers with 0.013 mg-P/l.

7.2.2. Nitrogen

In Figure 7.18 the total ammonia concentrations were found with a median of 0.05 mg-N/l outside the fen. The concentrations are comparable with phreatic tubes (median of 0.08 mg-N/l) and piezometers (median of 0.12 mg-N/l). Indeed, the concentrations in both the phreatic tubes and the piezometers were not statistically significant smaller with a p-values of 0.22 and 0.36 respectively. Some seasonal fluctuations seemed to occur in the boreholes between the median and the third quartile however the Spring/Summer values were not statistically significant less than the Autumn/Winter values (p value = 0.29).

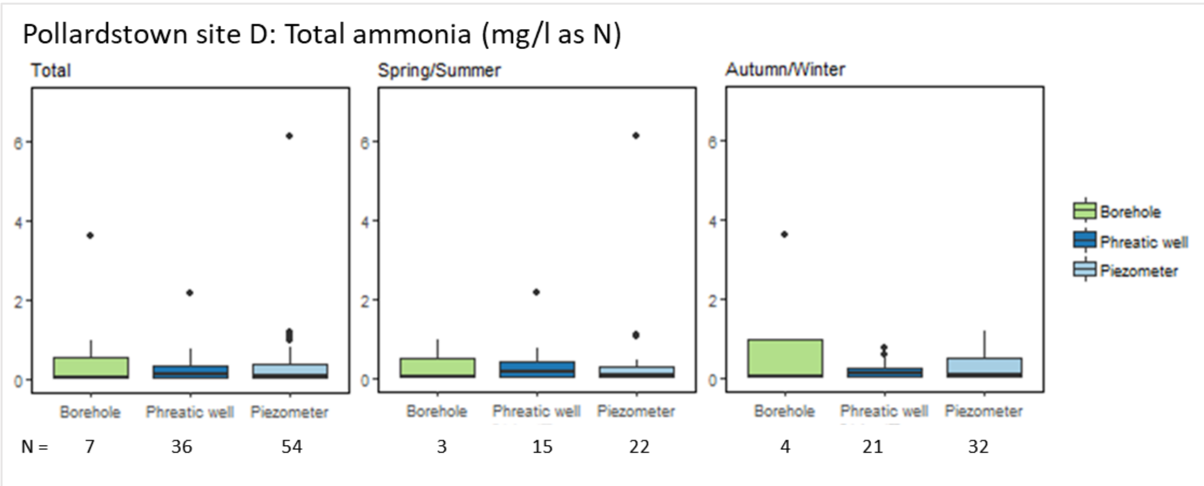


Figure 7.18. Total ammonia in mg-N/l sampled from phreatic wells and piezometers inside and boreholes outside the fen in Pollardstown site D.

What is interesting from Figure 7.19 is that some elevated nitrite values were found in the boreholes (third quartile of 0.51 mg-N/l in the Autumn/Winter) around the fen of Pollardstown site D since most of the values found in Ballymore and Pollardstown were reported below the limit of detection. These values were, however not statistically significantly less with p-value = 0.29. But this does show evidence of some temporary denitrification occurring (possibly linked to some organic pollution source to provide the carbon for the microbial reaction) give the high nitrate values in the groundwater.

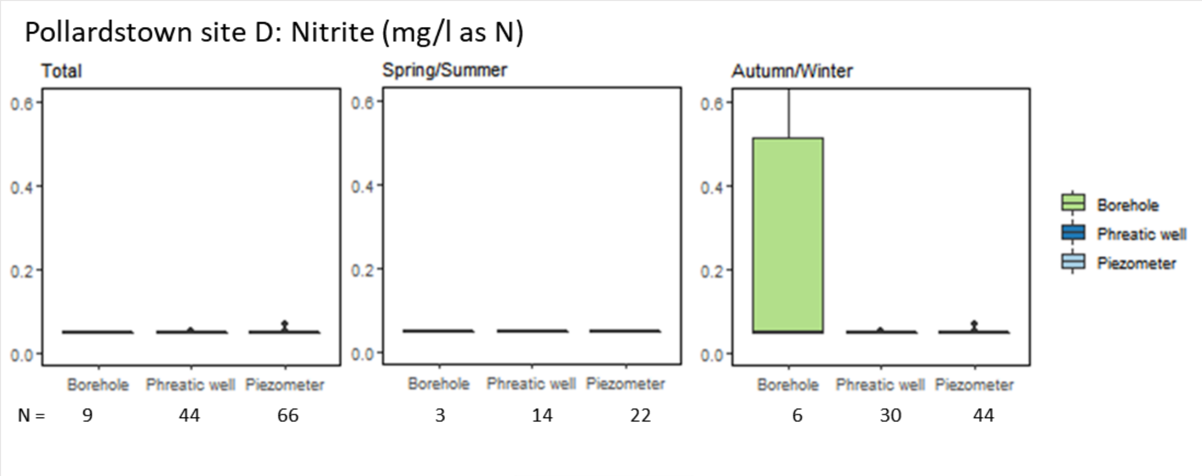


Figure 7.19. Nitrite in mg-N/l sampled from phreatic wells and piezometers inside and boreholes outside the fen in Pollardstown site D.

In Pollardstown site D the already familiar pattern of total oxidised nitrogen concentration is recognised where values in the boreholes are statistically significant higher than in the piezometers and phreatic tubes (Figure 7.20). The median in the boreholes was found with 1.49 mg-N/l and 0.19 mg-N/l in the piezometers. The values in the boreholes were found statistically significantly higher than in the piezometers (p = 0.047).

The median concentration of dissolved oxidised nitrogen in the phreatic tubes was 0.02 mg-N/l. The values in the phreatic tubes were found statistically significant lower than in the boreholes (p-value of 0.02).

Some of the piezometers had their screen installed at the depth of the aquifer as is proven by the artesian conditions. The elevated concentrations in the piezometers and the boreholes do not seem to pose a major threat to phreatic water table of Pollardstown site A. Overall the concentrations here are found much lower. There does seem to be a seasonal increase of total oxidised nitrogen in the Spring/Summer, however, as seen in the elevation of the third quartile, although this was not statistically higher (p-value of 0.24).

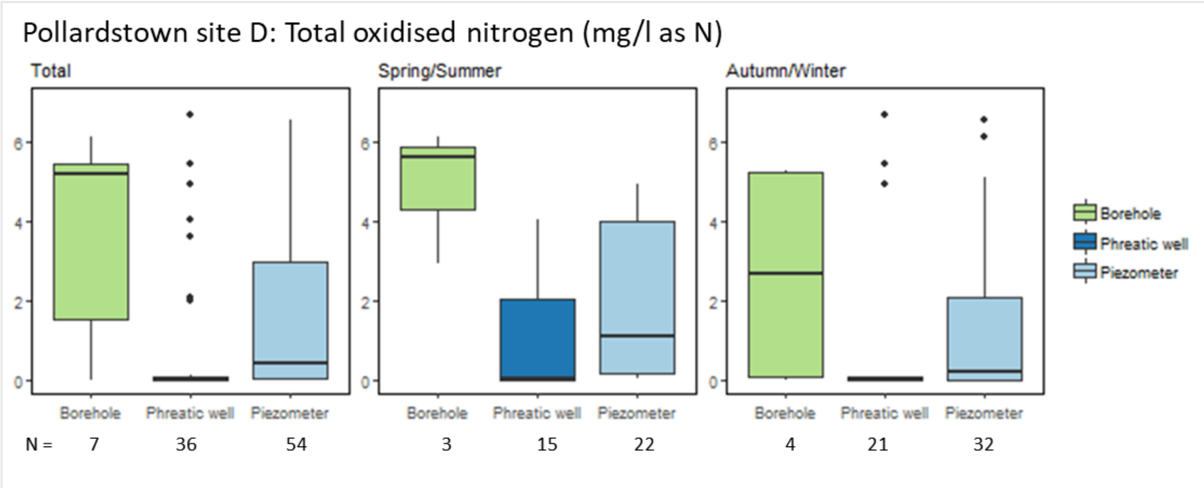


Figure 7.20. Total oxidised nitrogen in mg-N/l sampled from phreatic wells and piezometers inside and boreholes outside the fen in Pollardstown site D.

The total dissolved nitrogen concentrations in Figure 7.21 shows similar patterns in and around the fen as those found in the concentrations of total oxidised nitrogen. The highest proportion of total dissolved nitrogen is found in the boreholes around the fen. Here the median is 7.08 mg-N/l. The concentration in the piezometric measurements was significantly lower with a median of 2.52 mg-N/l (p-value of 0.00). The concentrations found in the phreatic tubes are also statistically significantly lower (p-value = 0.00) with a median of 1.19 mg-N/l. The lower values in the piezometers and phreatic tubes are again evidence that the fen is actively taking up and assimilating nitrogen. Such assimilation was also explained by authors such as Hill et al. (2016) and Paullissen et al. (2016).

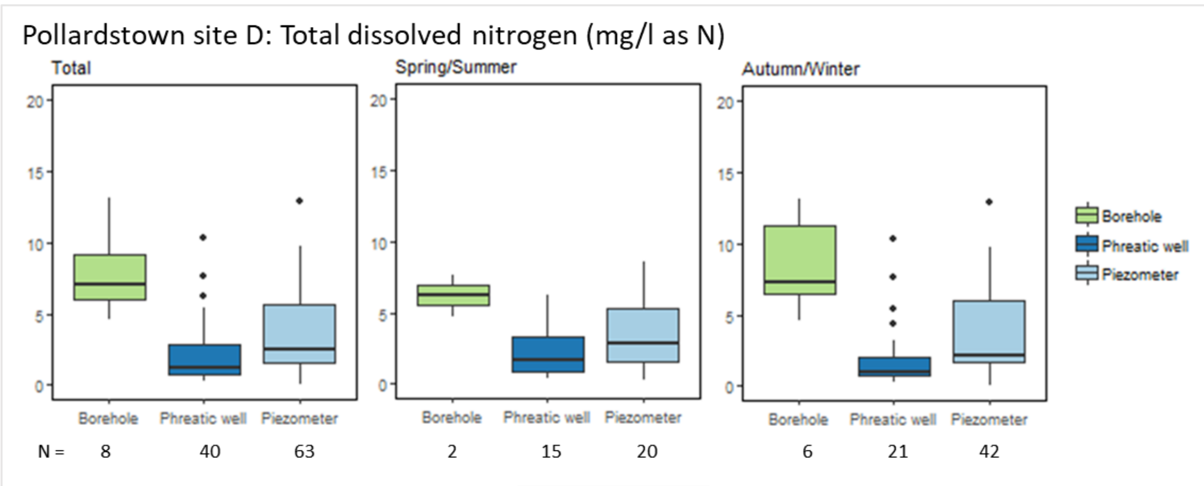


Figure 7.21. Total dissolved nitrogen in mg-N/l sampled from phreatic wells and piezometers inside and boreholes outside the fen in Pollardstown site D.

The ratios of total ammonia to total dissolved nitrogen is very high for the boreholes with 1:151. The piezometers and the phreatic tubes had lower ratios of 1:34 and 1:10 respectively. The ratio of total oxidised nitrogen to total dissolved nitrogen was lowest for the boreholes with 1:5. The very low ammonia concentrations and high nitrate values signify more oxic concentrations in the aquifer (Bedford & Godwin 2003; McBride et al., 2010). The ratios in the piezometers and phreatic tubes were much higher with 1:21 and 1:52 respectively.

The reported medians do not exceed groundwater threshold values for nitrite or nitrate (Government of Ireland, 2010), although there were however a few outliers in the aquifer that did exceed the nitrite threshold value of 0.114 mg-N/l, as discussed. Such occasions were mainly observed during the spring and summer within soils underlying the peat.

7.2.3. Other chemistry

The overall concentrations of alkalinity seemed quite similar when comparing data in and around the fen as seen in Figure 7.22 with no significant difference between the boreholes and the piezometers (p-value = 0.42). The phreatic tubes, however, were statistically significant different to the boreholes with a p-value of 0.01. The median in the boreholes was 182.5 mg/l as $CaCO_3$, whereas the median concentration in the piezometers and phreatic tubes were somewhat higher with 192.0 and 188.4 mg/l as $CaCO_3$, respectively. This may reflect retention of alkalinity in the fens groundwater. This may reflect retention of alkalinity in the fens groundwater. Higher concentrations of HCO_3^- are expected here as the groundwater inflows are assumed to have higher pH (±8), however it is important that the groundwater table stays stable for this retention to maintain (McLaughlin et al. 2008).

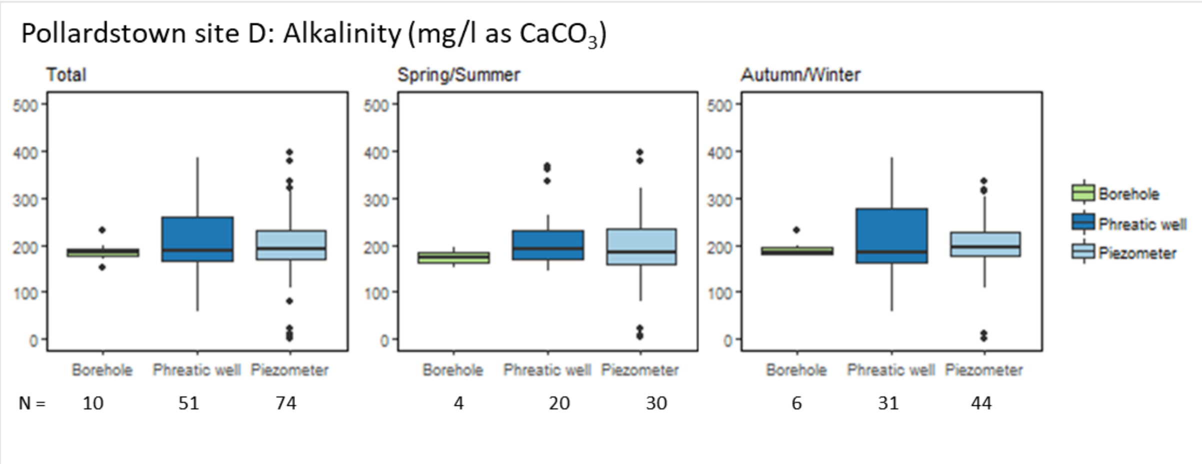


Figure 7.22. Alkalinity in mg/l as $CaCO_3$ sampled from phreatic wells and piezometers inside and boreholes outside the fen in Pollardstown site D.

The overall concentrations of chloride were slightly higher in the boreholes the fen with a median of 21.7 mg/l (Figure 7.23). The median of the chloride concentration in the phreatic tubes was only slightly lower with 19.6 mg/l but not significantly (p-value of 0.91). The median of the piezometers was also lower with 19.6 mg/l but again not significantly (p-value of 0.90).

Since chloride is usually found with higher concentrations in the groundwater the implication can be made that the fen is receiving water from an additional source rather than mostly groundwater.

This corroborates the argument in Section 7.1.1.1 from the EC results that suggests that Pollardstown receives part of its water from the surface.

In any case, the reported medians are far below the Irish groundwater threshold values for chloride (Government of Ireland, 2010) which is 187.5 mg/l.

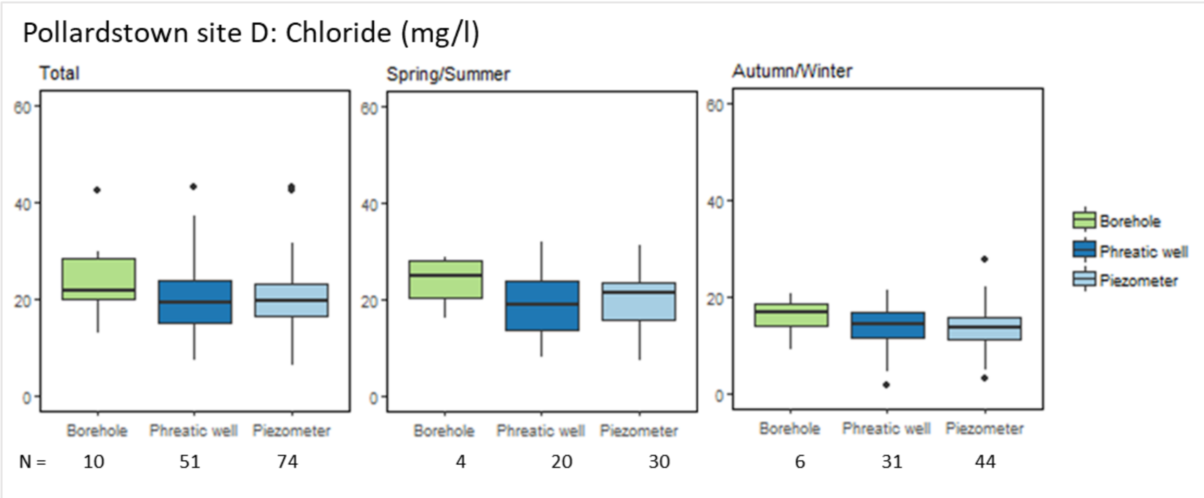


Figure 7.23. Chloride in mg/l sampled from phreatic wells and piezometers inside and boreholes outside the fen in Pollardstown site D.

In Figure 7.24 boreholes are found were observed with higher concentrations of silica (median of 9.5 mg/l as SiO^2) which is to be expected in groundwater. The concentrations found in the fen are statistically significant lower with a p-values of 0.00 for both phreatic tubes and piezometers. The concentration in the phreatic water table was especially low with a median of 5.1 mg/l as SiO^2. The concentrations in the piezometers were higher than that with a median of 7.8 mg/l as SiO^2. Again, this shows that Pollardstown site D has an additional source of surface water entering the fen, where just like EC, silica can be used as a tracer of groundwater.

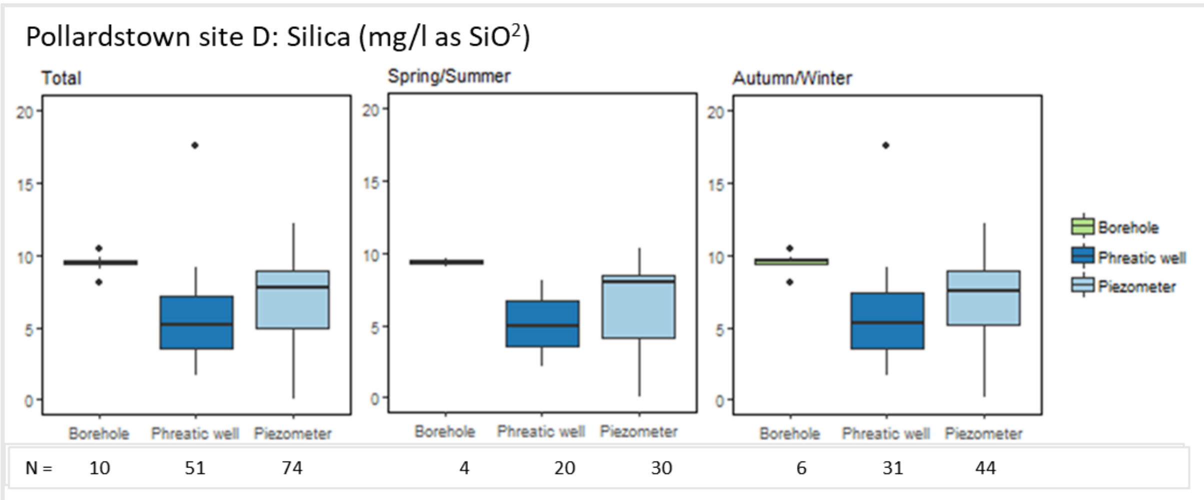


Figure 7.24. Silica in mg/l as SiO^2 sampled from phreatic wells and piezometers inside and boreholes outside the fen in Pollardstown site D.

Sulphate concentrations (Figure 7.25) were found to be slightly higher in the boreholes around the fen with a median of 18.3 mg/l as SO_4^{2-} compared to the concentrations in the fen with medians of 14.5 mg/l as SO_4^{2-} and 14.1 mg/l as SO_4^{2-} in the phreatic tubes and piezometers, respectively. It therefore seems that sulphate does not have much interaction with the fen vegetation in the phreatic water table. The phreatic tubes and piezometers were not statistically significantly lower with p-values of 0.46 and 0.60 respectively.

However, concentrations of sulphate did seem to decrease in the phreatic tubes during the Spring/Summer, the growing season of the fen, signifying that the vegetation did take up some portion of the concentration in the phreatic layer. There could also exist some more oxygen depleted pore waters within the peat acting to precipitate sulphide. The decrease in the phreatic tubes was however not significant with a p-value of 0.35.

The measured values are all far below the Irish groundwater threshold values for sulphate (Government of Ireland, 2010) which is 187.5 mg/l.

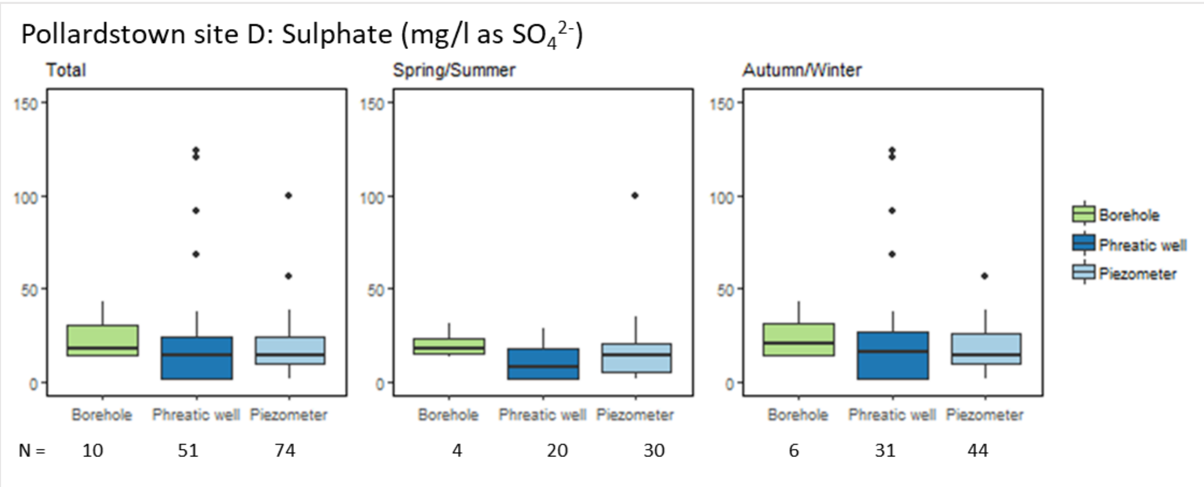


Figure 7.25. Sulphate in mg/l as SO_4^{2-} sampled from phreatic wells and piezometers inside and boreholes outside the fen in Pollardstown site D.

Dissolved organic carbon in and around the fen was found with higher concentrations in the fen than in the surrounding aquifer (Figure 7.26), as found in the other fens. The median in the vegetative phreatic tubes layer was especially high at 12.7 mg/l. These high values are expected were the life cycle of vegetation causes organic build up on the surface of the fen. This organic matter is then slowly decomposing in the upper layer of the peat. The boreholes in the fen catchment had a median of 3.3 mg/l. The concentrations were found lower in the piezometers with a median of 1.6 mg/l. Two sided Welch T-test proved that the boreholes had significant different values than the phreatic tubes (higher with p-value = 0.00) as well as the piezometers (lower with p-value = 0.03).

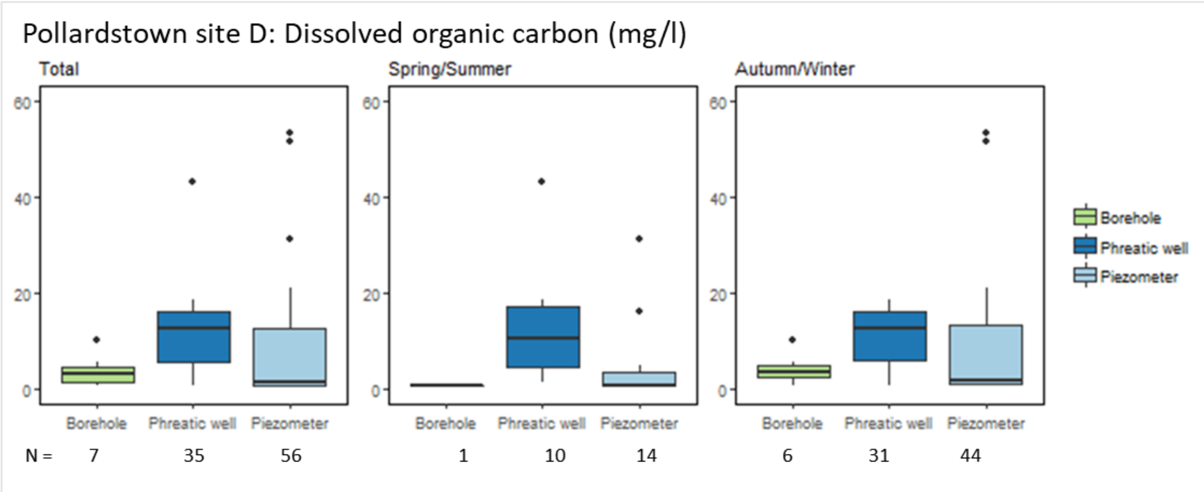


Figure 7.26. Dissolved organic carbon in mg/l sampled from phreatic wells and piezometers inside and boreholes outside the fen in Pollardstown site D.

A lot of ferrous iron was detected in the phreatic water table as median of 0.16 mg/l as Fe^{2+} was reported (Figure 7.27) which points to the argument that the fen is fed a significant groundwater proportion. Iron is normally stored as ferrous iron under reducing conditions. When oxygen rich water enters the phreatic water table ferrous iron oxidises to ferric (3^+) iron. This process may also cause the release of DRP from ferric iron which liberates the bioavailable phosphate for potential uptake by the fen vegetation (Bedford & Godwin, 2003; McBride et al., 2010).

Some higher values were also reported in the piezometers although the median was reported below the limit of detection as 0.006 mg/l as Fe^{2+}. The ferrous iron was also detected below the limit of detection in the boreholes around the fen, which confirms the more oxic conditions in the groundwater as mentioned before. Rather than utilising the measured oxygen values during this research, oxic/anoxic conditions were concluded from the specific chemistry in the water column as water purged from piezometers was done with bailers instead of flow cells allowing oxygen results to become inconclusive.

The concentrations in the phreatic tubes and piezometers were significant higher p-values of 0.00 for both than in the surrounding catchment.

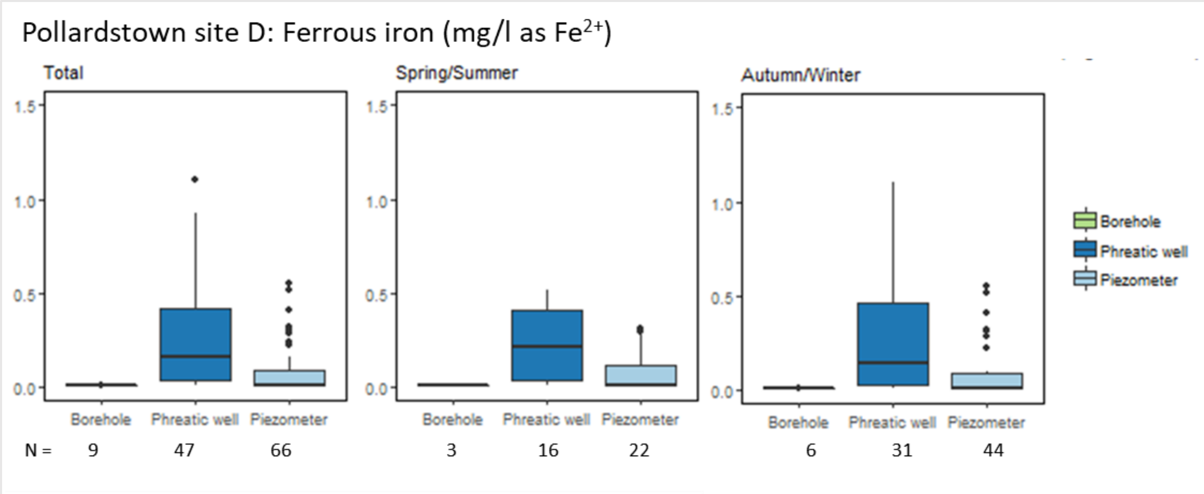


Figure 7.27. Ferrous iron in mg/l as Fe^{2+} sampled from phreatic wells and piezometers inside and boreholes outside the fen in Pollardstown site D.

7.2.4. Conceptual chemistry model

- DRP statistically significant higher in boreholes than in fen. Groundwater input with higher concentrations of phosphorus do not affect the water column in the fen.
- TP is stored in the phreatic water table and its release from the peat is sensitive to ground water fluctuations.
- Nitrogen is largely present as total oxidised nitrogen in the underlying sediments which may be received from the aquifer. The portion in the phreatic water layers however seem to be part of the peat nutrient recycling scheme.
- The site is further proven to receive part of its water from surface water by comparing the concentrations of chloride, silica and ferrous iron in the aquifer to the fen.

7.3. Linkage to fen habitat

7.3.1. Hydrology and fen habitat

7.3.1.1. Boxplots water level

All water levels recorded in the different Fossitt habitats of Pollardstown site D are presented in Figure 7.28. Again, as in Pollardstown site A, positive pressure from the aquifer resulting in artesian conditions are visible in the piezometers. These conditions seem to support habitats Poor fen and flush (PF2) as well as Wet grassland (GS4). Interestingly the phreatic water table in is higher in GS4 (median of +0.09 mAGL) than in PF2 (median of -0.05 mAGL). This was not the case in the PF2 vegetation of Pollardstown A where the habitat was supported by significantly higher water levels.

It would be interesting to see if the water level height has an influence on the quality of PF2 vegetation (see Chapter 10). Habitat Reed and large sedge swamps (FS1) is not supported by artesian condition and yet the phreatic water levels are close to the surface with a median of -0.03 mAGL.

It seems that since the elevations are very stable over time the water that enters the fen moves relatively evenly over its surface, as mentioned in Section X. This is reflected in the phreatic water levels of all habitats.

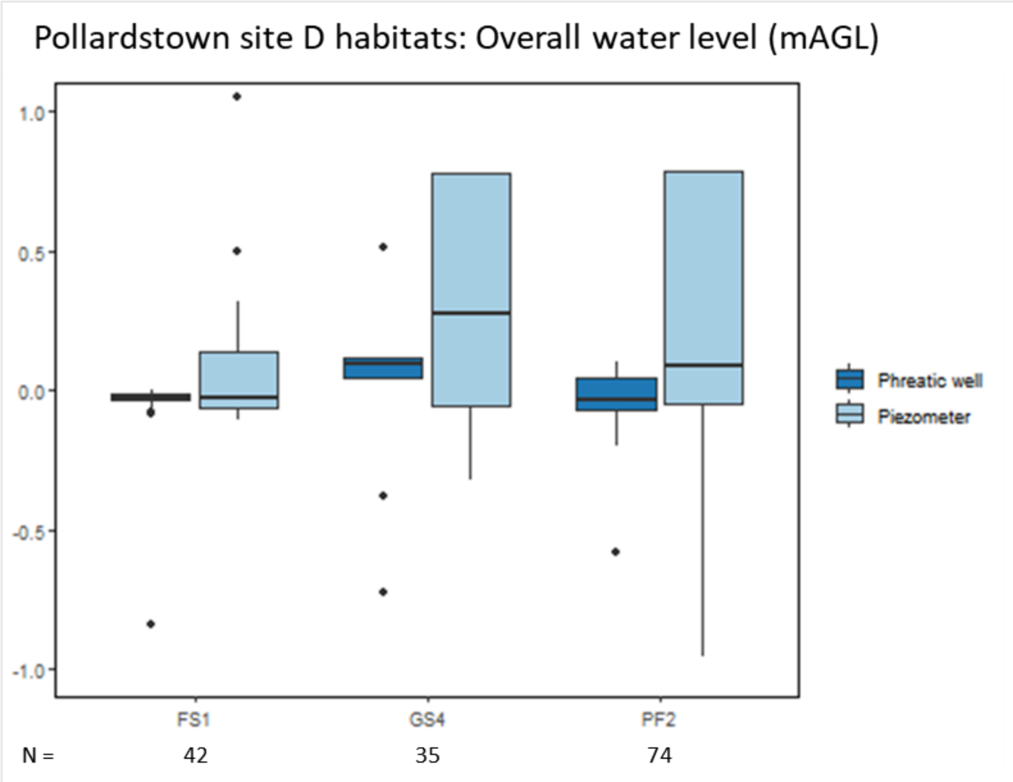


Figure 7.28. Overall water level in meters above ground level in the different habitats of Pollardstown site D measured in phreatic wells and piezometers.

Both habitats FS1 and PF2 do not display any seasonal water level fluctuations in Figure 7.29. Some of the phreatic tubes in GS4 do show decreased water levels during the Spring/Summer as the first quartile is reported with -0.38 mAGL during this time instead of 0.06 mAGL during the Autumn/Winter.

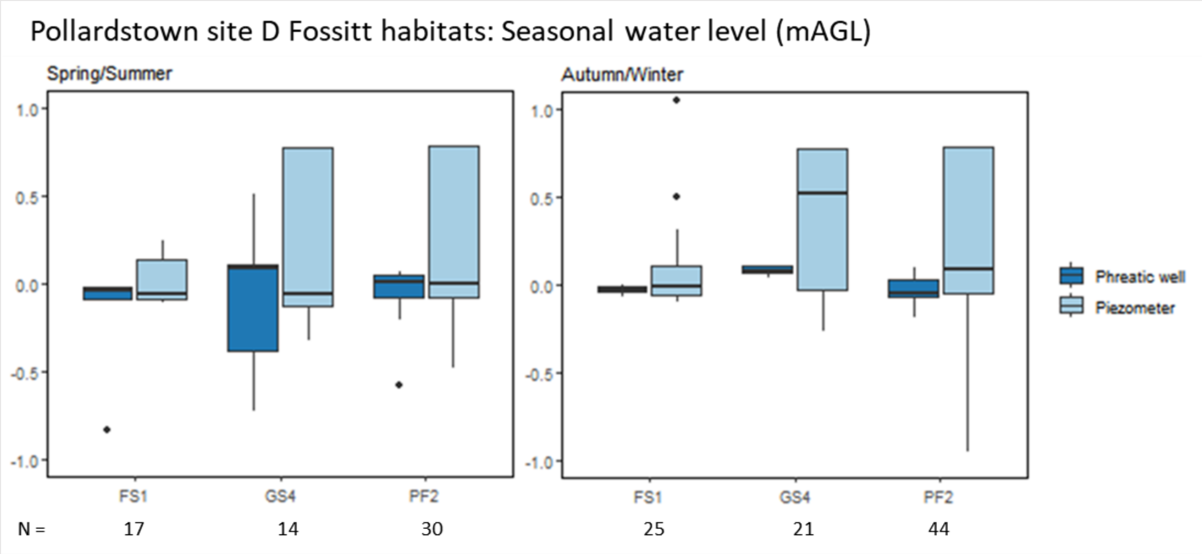


Figure 7.29. Seasonal water level in meters above ground level in the different habitats of Pollardstown site D measured in phreatic wells and piezometers.

7.3.1.1. Frequency duration curves

Frequency duration curves of surface water levels between October 2018 and October 2020 are presented in Figure 7.30. Unfortunately, no time series of habitat PF2 was available as the water level logger malfunctioned for most of the time.

Site PD23 with GS4 habitat seem to be supported with continuous water levels above the surface. This water level was measured above +0.1 mAGL for 60% of the time. The phreatic water levels in PD28 with FS1 were found to be lower than the surface elevation. A fluctuation range of just 0.1 m is observed evenly throughout.

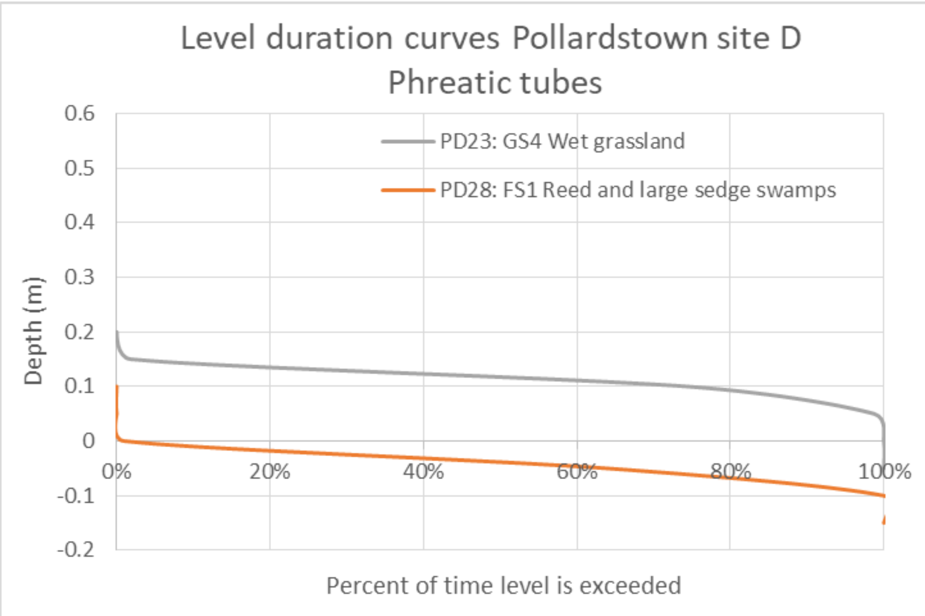


Figure 7.30. Phreatic level duration curves recorded in different habitats in Pollardstown site D. The negative numbers are water levels below groundlevel.

Level duration curves of permanent quadrat PQ6 were plotted with water level time series of a phreatic tube and a piezometer located 100 m west from PD28 between 1998 and 2007 (Figure 7.31). The habitat in PQ10 seems supported by a stable piezometer water level (PQ6 Deep) above 86.2 m OD for <90% of the time. The phreatic water table of that location (PQ6 Shallow) showed very stable levels around 85.75 m OD also for <90% of the time. After this the water level fluctuated 0.2 m. These fluctuations were more extreme in the level duration curves of 2018/20 where fluctuation ranges of just 0.1 m are observed evenly throughout. However, the phreatic conditions of PQ6 are also more stable than seen in PA23 and PA28.

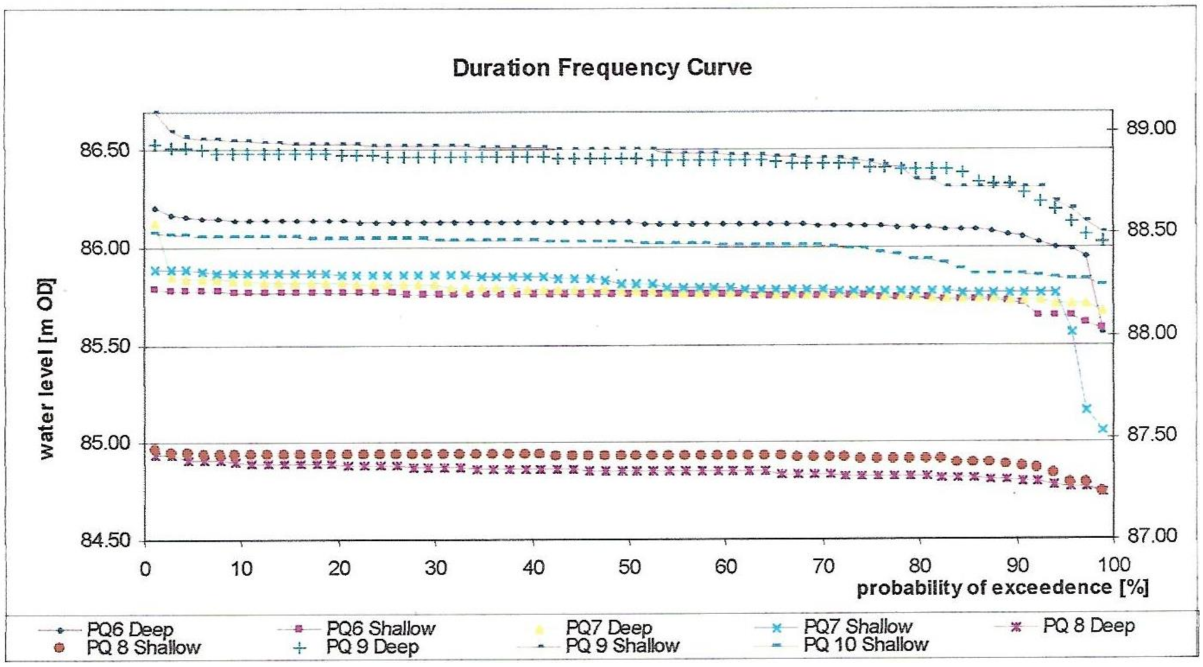


Figure 7.31. Level duration curves measured in the permanent quadrats (PQ) of Pollardstown fen between 1998 and 2007 (Hayes, 2004). The left axis of the graphs shows the scale in m OD measured according to Malin. The right axis is the scale in m OD according to Poolbeg.

From the phreatic and piezometric water level time series at PD23 and PD28 hydraulic gradients were calculated, as shown in Figure 7.32. PA28 displayed upwards flows with a hydraulic gradient of around 0.05 for most of the time. A sudden increase was recorded in the winter of 2019.The hydraulic gradient of PD23 seems to be seasonally affected by effective rainfall with upwards flows during periods of high effective rainfall.

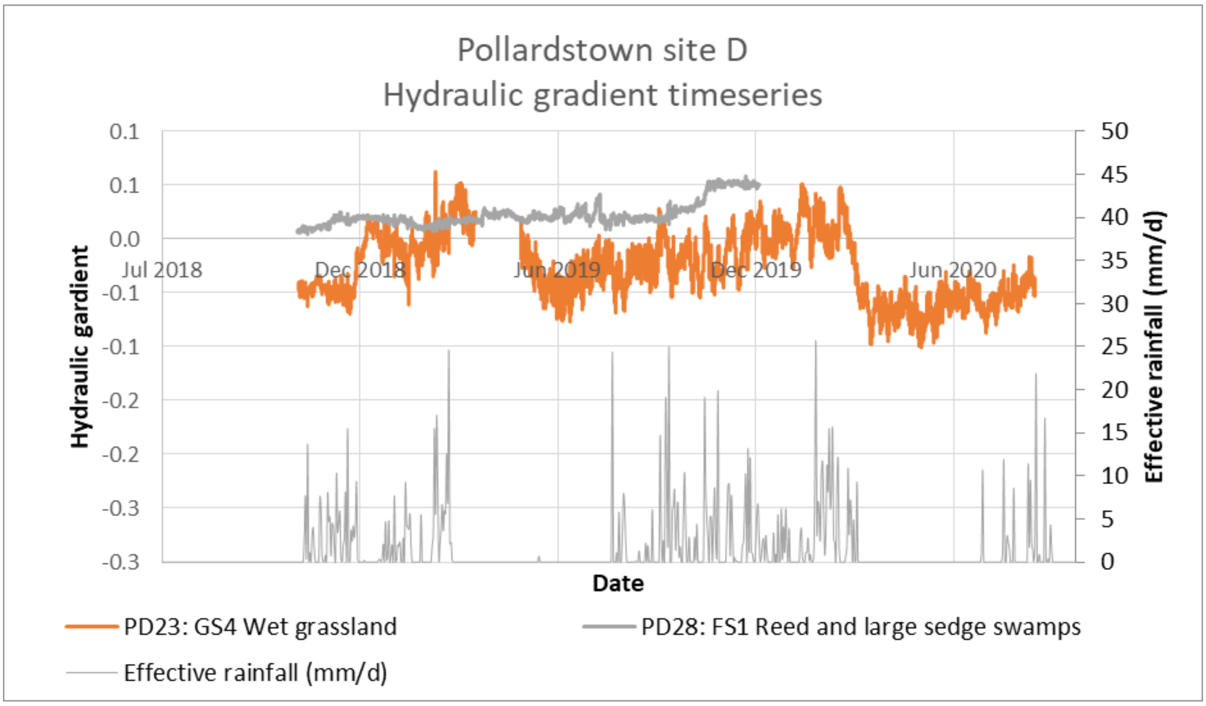


Figure 7.32. Hydraulic gradient timeseries calculated using the phreatic and piezometric water level timeseries in Pollardstown site D. Effective rainfall is displayed here as well.

In contrast to Pollardstown site A, upwards flow seems to occur further in the fen rather than close to the bottom of the hill of the adjacent fields north from the fen (PD23). It has to be noted that the surface elevation of this location is at a higher surface elevation. It may therefore be possible that the groundwater travels below the surface at certain times of the year and that water mainly flows in via diffuse springs further along the transect. Indeed, the hydraulic gradient of PD28 always displayed an upward flow which supports this argument. It is possible that the Milltown Feeder (one of the principal sources for the Grand Canal), which is located approximately 400 m from this location will have considerable effect on the hydraulic gradient travelling even further along the transect.

The hydraulic gradient time series was also plotted as frequency duration curves as seen in Figure 7.33. PA28 displays a minimal level change of 0.05 spread out evenly for 90% of the time. PD23 has a small upwards gradient for 20% of the year, seemingly supported by high effective rainfall as seen in Figure 7.2; this site then has a downward gradient between 0 and 0.05 for most of the time. A stronger downward gradient of more than 0.05 was observed for > 0.80%.

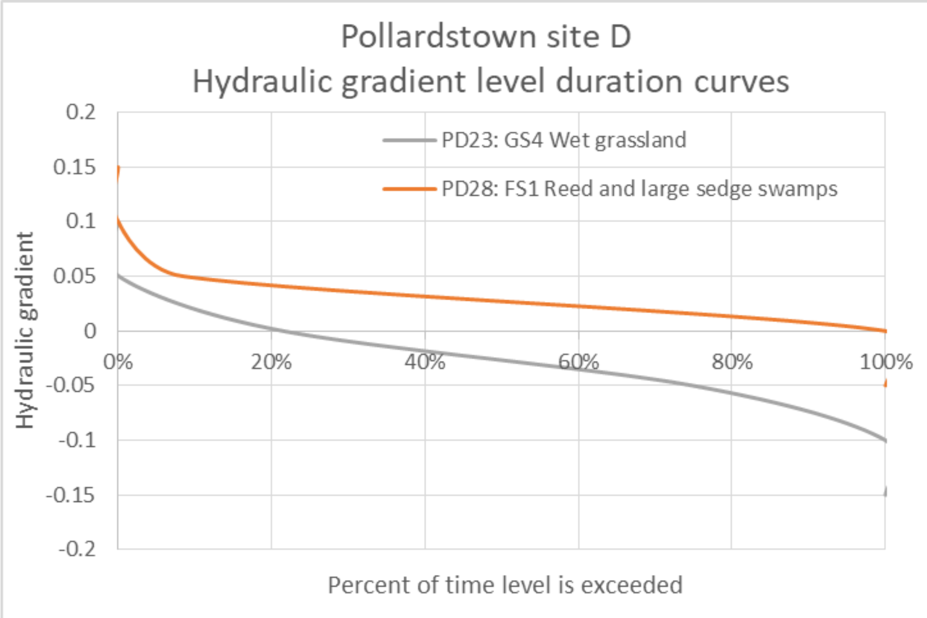


Figure 7.33. Level duration curves of the hydraulic gradients calculated from the water level time series in monitored phreatic tubes and piezometers.

7.3.2. Hydrochemistry and fen habitat

7.3.2.1. Nonmetric Multidimensional Scaling ordination

The NMDS plots were displayed earlier in Section 5.3.2.1 executed with data from both Pollardstown site A and D. The plot with the environmental variables vegetation cover (%) and the presence of the Fossitt habitats in a biplot (Figure 6.39) habitats PF2, FS1 and GS2 were found with significant correlations. Furthermore hydrochemical parameters such as DRP, TP, total ammonia, total oxidised nitrogen, total dissolved nitrogen, silica and sulphate were found to have correlations with the vegetation observed in the fen.

From the NMDS plots of Figures 6.36 and 6.37, it can be observed that higher concentrations of TP, total oxidised nitrogen, total dissolved nitrogen, silica and sulphate were more correlated to habitats with Poor fen and flush (PF2) vegetation than other habitats. Furthermore higher concentrations of total ammonia and DRP were correlated to sites with a FS1 habitat. These correlations of the habitats and the hydrochemistry will be further explored here with results found specifically in Pollardstown site D (Section 7.3.2.2).

7.3.2.2. Boxplots hydrochemistry

The NMDS correlation suggested that habitat PF2 would show higher concentrations of DRP, however Figure 7.34 shows that concentrations of DRP seems higher in the FS1 habitat overall. Some higher elevation of DRP seems to be present in the piezometers of PF2 though.

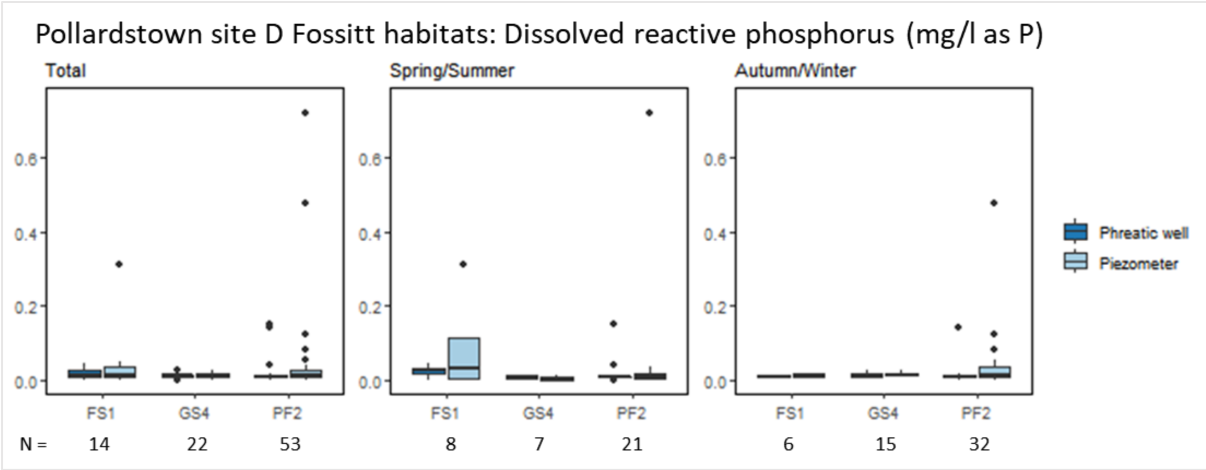


Figure 7.34. Dissolved reactive phosphorus (mg-P/l) in the different habitats of Pollardstown site D sampled from phreatic wells and piezometers.

Higher concentrations do now seem to appear when the habitats are looked at separately (Figure 7.35), especially for FS1 and GS4. This was not visible before in where all the phreatic tubes and piezometers data were lumped together. It may be that the decomposition rate of decaying vegetation is higher in GS4 and PF2 which results in higher TP concentrations. It was also stated before that TP seems to be stored in the upper peat layers of Pollardstown site D which may be release under the influence of water fluctuations (Section 7.2.1). Interestingly GS4 showed larger seasonal surface water fluctuations which now seems to further imply that this has caused TP to be present in the water column at a higher rate than the other habitats.

Furthermore, when looking for evidence to confirm the correlations found in the NMDS plots it seemed that TP had comparable concentrations in habitat PF2 to the other habitats overall. However, higher concentrations were found in the phreatic tubes that were sampled in the Spring/Summer.

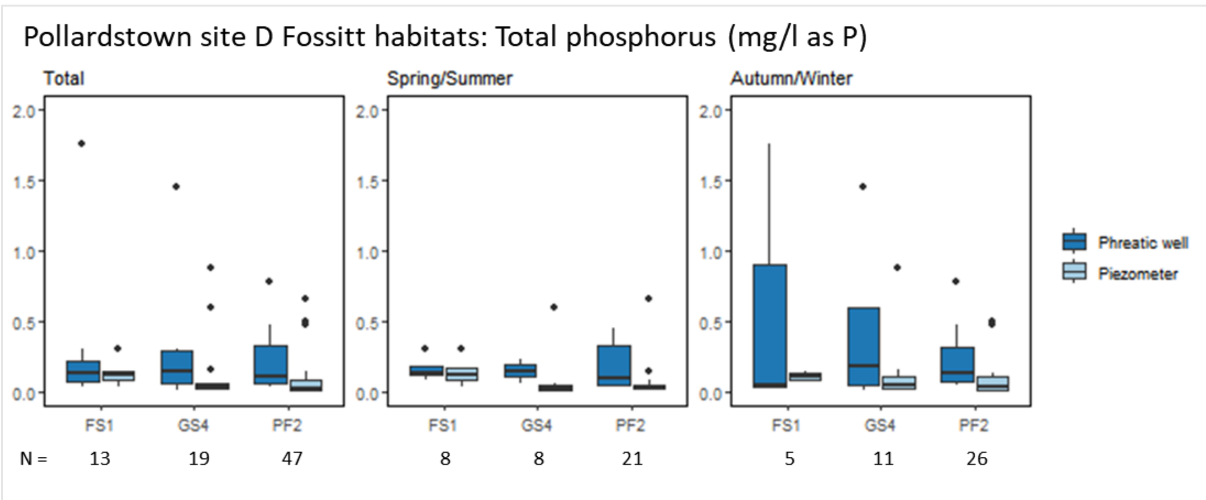


Figure 7.35. Total phosphorus (mg-P/l) in the different habitats of Pollardstown site D sampled from phreatic wells and piezometers.

Total ammonia showed a NMDS correlation for the PF2 habitats. Indeed concentrations are somewhat elevated the Spring/Summer phreatic tubes, however, habitat FS1 displayed much higher values of ammonia (Figure 7.36). Interestingly, total ammonia is mainly found the phreatic tubes during the Spring/Summer in this habitat. In the Autumn/Winter this changes drastically with high concentrations now coming from the piezometers.

The high values could be explained by the possibly low nutrient requirement of this habitat. Another possible reason is that the aquifer supplies a lot of nutrient rich groundwater in FS1 especially in the winter which cannot be taken up by the vegetation or stored in the sediments as seem to happen in other areas of the fen. Furthermore, the high concentrations could be explained by high decomposition rates in the habitats of FS1.

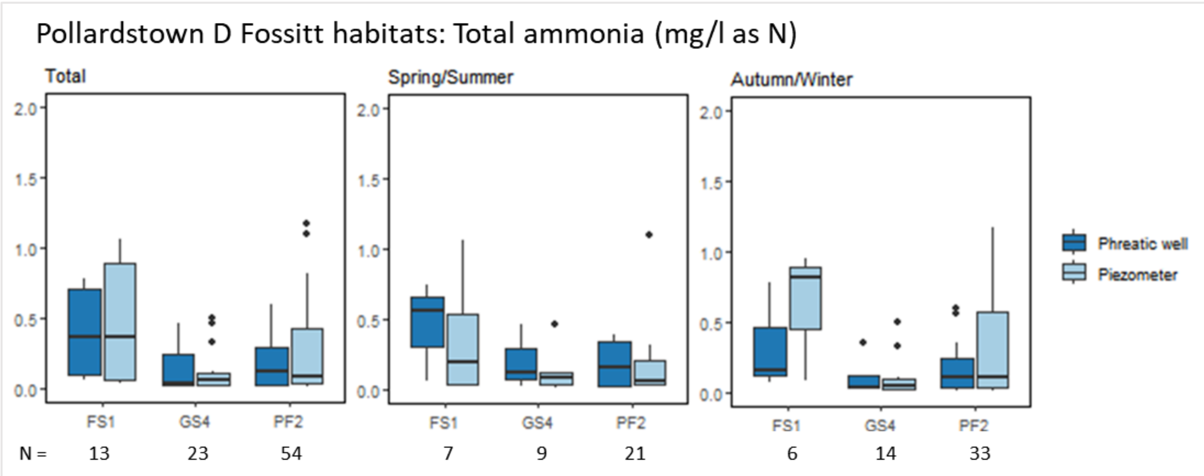


Figure 7.36. Total ammonia (mg-N/l) in the different habitats of Pollardstown site D sampled from phreatic wells and piezometers.

The total oxidised N showed a NMDS correlation with FS1 habitats. While the concentrations are elevated in the phreatic tubes measured in the Spring/Summer, GS4 is the habitat that shows extremely elevated concentrations (Figure 7.37).

The high values could be explained by the possibly low nutrient requirement of this habitat. Another possible reason is that the groundwater supplies a lot of nutrient rich ground water in GS4 which cannot be taken up by the vegetation or stored in the sediments as seem to happen in other areas of the fen.

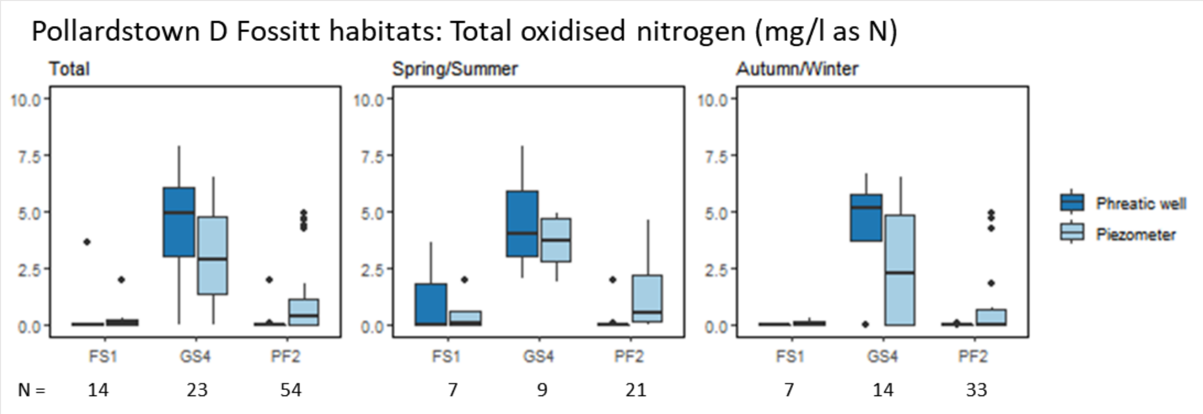


Figure 7.37. Total oxidised nitrogen (mg-N/l) in the different habitats of Pollardstown site D sampled from phreatic wells and piezometers.

Elevated concentrations of total dissolved nitrogen were expected in FS1, according to the NMDS correlations. However, again it is actually habitat GS4 which is found with significantly elevated concentrations as seen in Figure 7.38 which probably originate from groundwater. It is unsure however, if these high nitrate levels caused GS4 to develop as a measure against these high values or if this specific vegetation is located there despite these conditions and have a resilience against high nitrogen values. The piezometers in FS1 had higher concentrations in the Autumn/Winter than in the Spring/Summer.

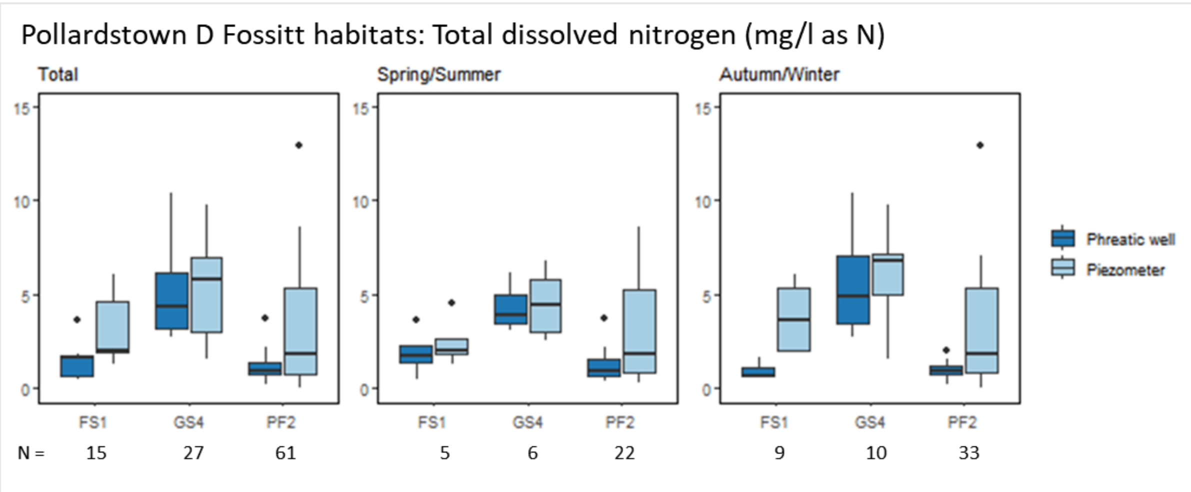


Figure 7.38. Total dissolved nitrogen (mg-N/l) in the different habitats of Pollardstown site D sampled from phreatic wells and piezometers.

According to the NMDS plot, silica was expected with elevated values in PF2. While this is true, GS4 also displays high values, making it predominantly groundwater fed. Both the values in FS1 and well as in the phreatic tubes of PF2 are found with much lower values implying that these areas are mix of ground and surface water.

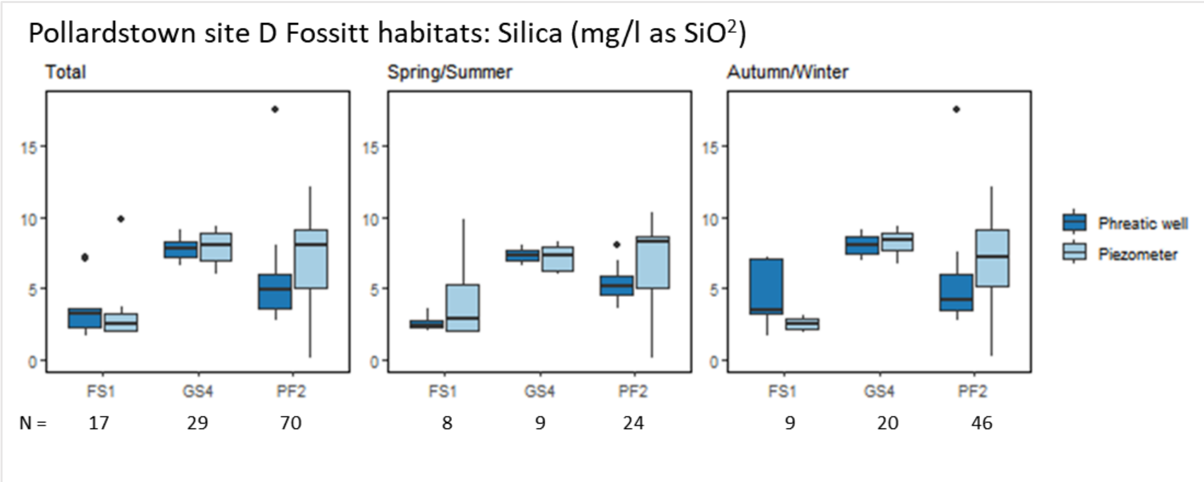


Figure 7.39. Silica (mg/l as SiO^2) in the different habitats of Pollardstown site D sampled from phreatic wells and piezometers.

Again sulphate levels we expected with elevated values in habitat PF2, however the concentrations displayed in Figure 7.40 show no particular difference between the habitats in Pollardstown site D.

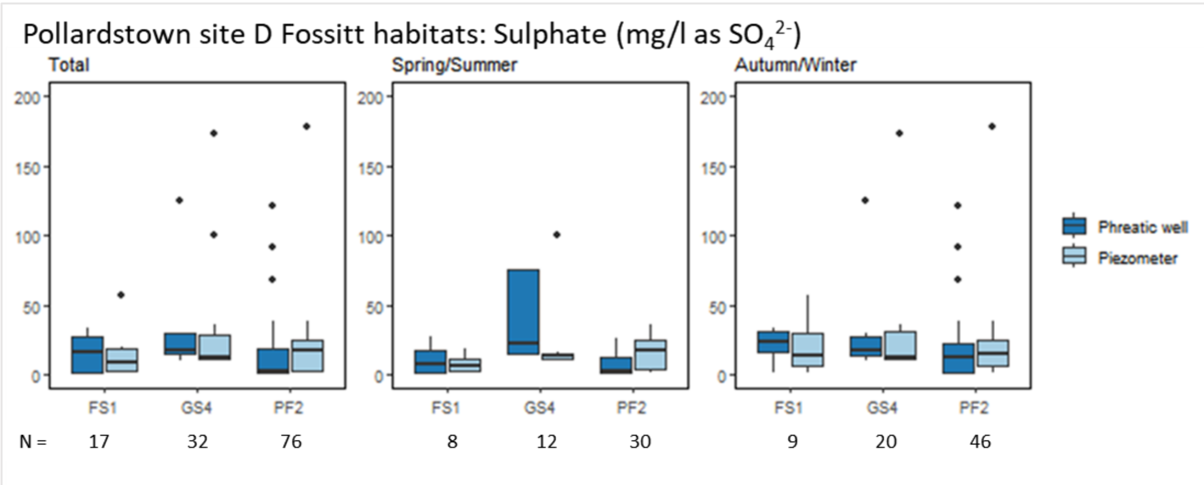


Figure 7.40. Sulphate (mg/l as SO_4^{2-}) in the different habitats of Pollardstown site D sampled from phreatic wells and piezometers.

7.3.3. Mean seasonal hydraulic gradients and hydrochemistry

The following sections bring the knowledge of the hydrology and hydrochemistry together on Pollardstown transect site D. The legend of the soil geology displayed on the transects can be found in Figure 6.41.

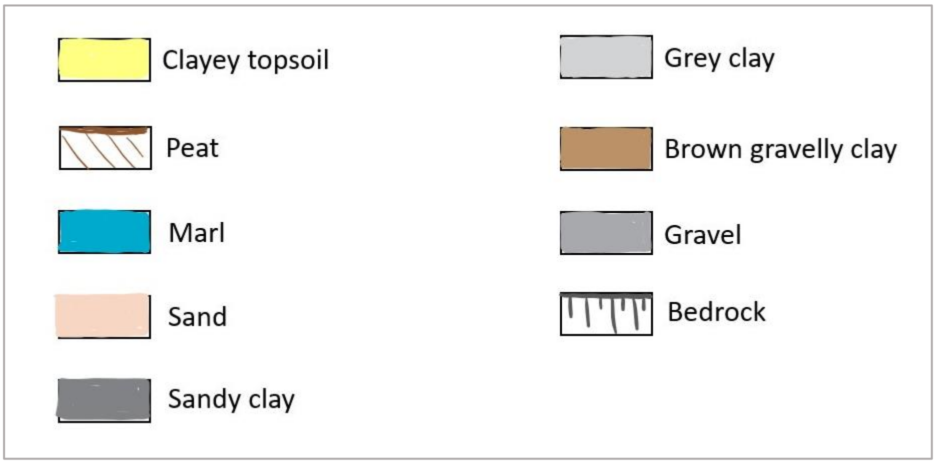


Figure 7.41. Legend explaining the different soil types in the Pollardstown geology transects.

7.3.3.1. Dissolved reactive phosphorus

Figure 7.42 shows the Pollardstown D with data collected in August 2020. The transect displays some upwards flow from the gravel aquifer under the peat layers. Furthermore, further into the transect some downwards flows are observed which could be due to drawdown of the outlet (Milltown feeder) 300 m away. PD29 is the only location where a spike of DRP is found; this area also seems to have some upward flow conditions. This elevation is not completely strange as this location is right next to old peat cuttings. It could be that the damage done to this habitat is causing DRP to leach in to the water column. Interestingly it seems that this elevated DRP concentration has travelled to the phreatic water table in February 2020 (Figure 7.43). The hydraulic gradients have not changed much compared to the summer values. The slight differences seem to be caused by the difference in typical conductivities of the lower sediment layers of the fen. This further supports the theory that the fen utilises some form of inner seasonal water cycling which also moves the containing chemistry around in the different layers. This mechanism may be governed by both water flow and oxygen fluctuations.

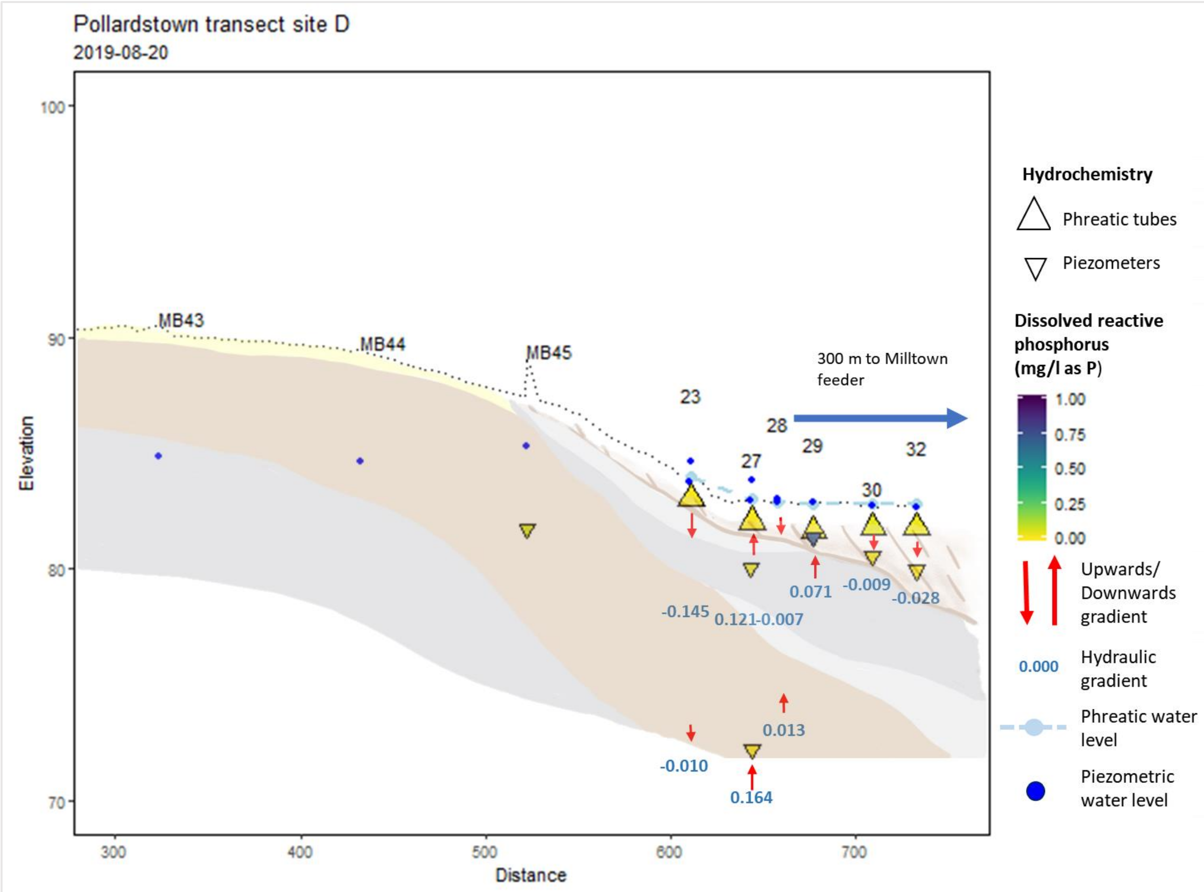


Figure 7.42. Hydrology and dissolved reactive phosphorus (mg-P/l) of Pollardstown site D in August 2019. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

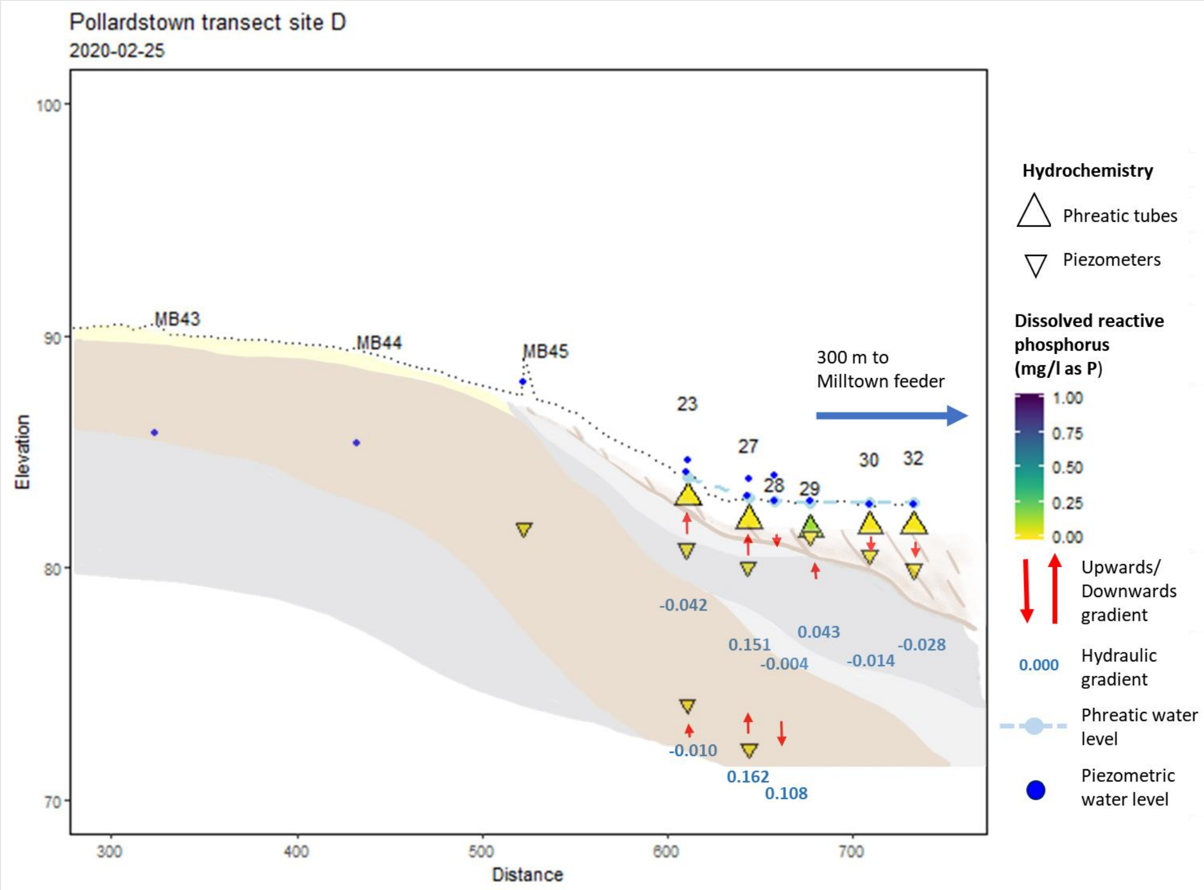


Figure 7.43. Hydrology and dissolved reactive phosphorus (mg-P/l) of Pollardstown site D in February 2020. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

7.3.3.2. Total ammonia

The total ammonia measured during August 2019 (Figure 7.44) displays slightly elevated concentrations in the catchment (MB45). Also some higher concentrations are found in the deeper fen peat layers at locations PA30 and PA32. Again, it seems that ammonia leaches from the old peat cuttings at site PA29. As found in Pollardstown site A most of the elevated ammonia concentrations seems to have disappeared from the fen peat. Ammonia may have been diluted by rainfall in the phreatic layer of the fen or nitrified under oxidised circumstances. However, some elevated values remain in the deeper peat layers.

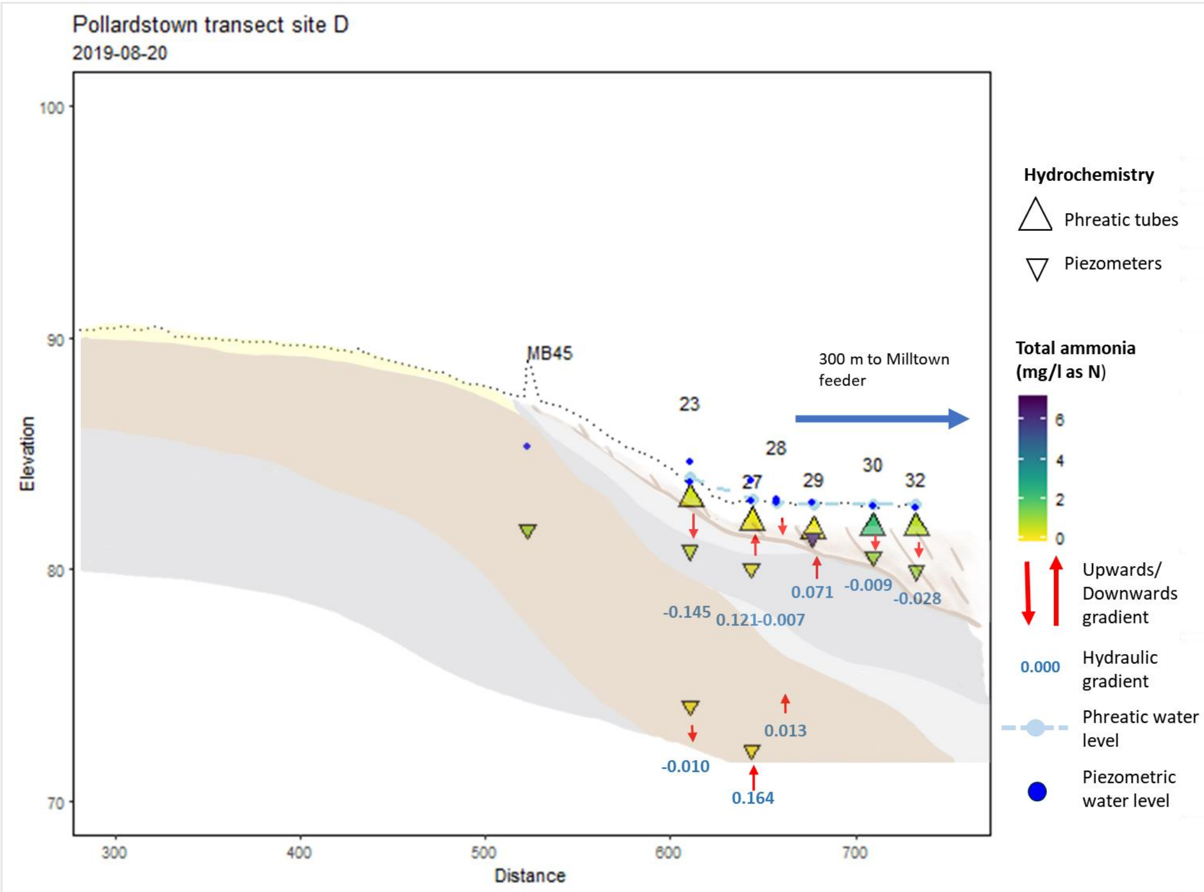


Figure 7.44. Hydrology and total ammonia of Pollardstown site D in August 2019. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

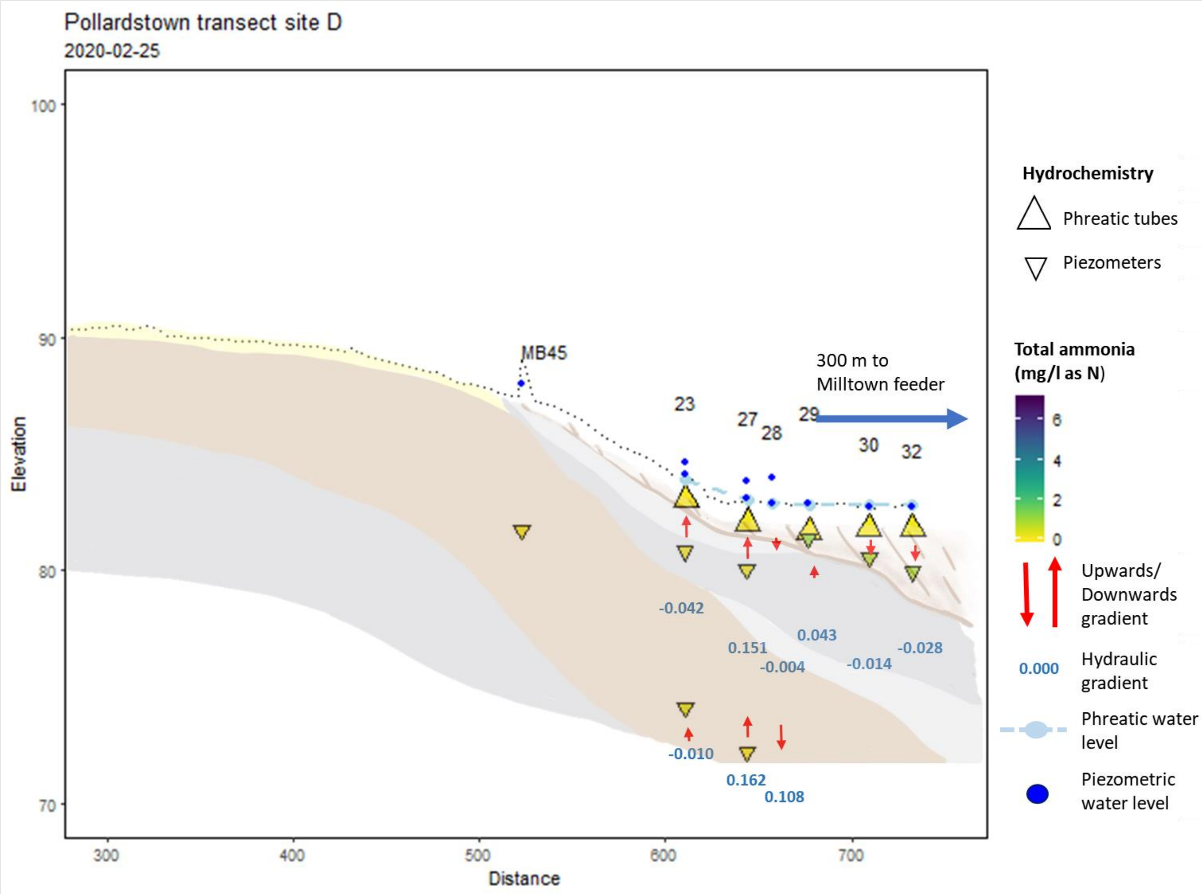


Figure 7.45. Hydrology and total ammonia of Pollardstown site D in February 2020. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

7.4. Conceptual model

7.4.1. Site summary

Pollardstown site D is a small part of Pollardstown fen as a whole fen (0.27 km^2) and contains poor fen swamp and grassland habitats. The site supports a total of 0.05 km^2 which is 20% of the entire site. From the three assessed relevés conducted during the vegetation survey as specified in Section 4.1.3 only one passed the fen assessment criteria in Appendix D.

Effective rainfall did not have an effect on the phreatic water levels and it seems that most of Pollardstown site A is fed by groundwater as found in EC values. This is not true for some locations however, when it seemed that some localised higher surface water inflows were found in locations close to old peat cuttings.

Portions in the phreatic water layers in the fen seem to be part of the peat nutrient recycling scheme when comparing phosphorus and nitrogen in the fen to the catchment. The site is further proven to receive part of its water from surface water as seen in concentrations of chloride, silica and ferrous iron.

7.4.2. Conceptual model

A conceptual box model is displayed in Figure 7.46, showing the water balance, surface water level fluctuation and median nutrient concentrations in the fen and its catchment.

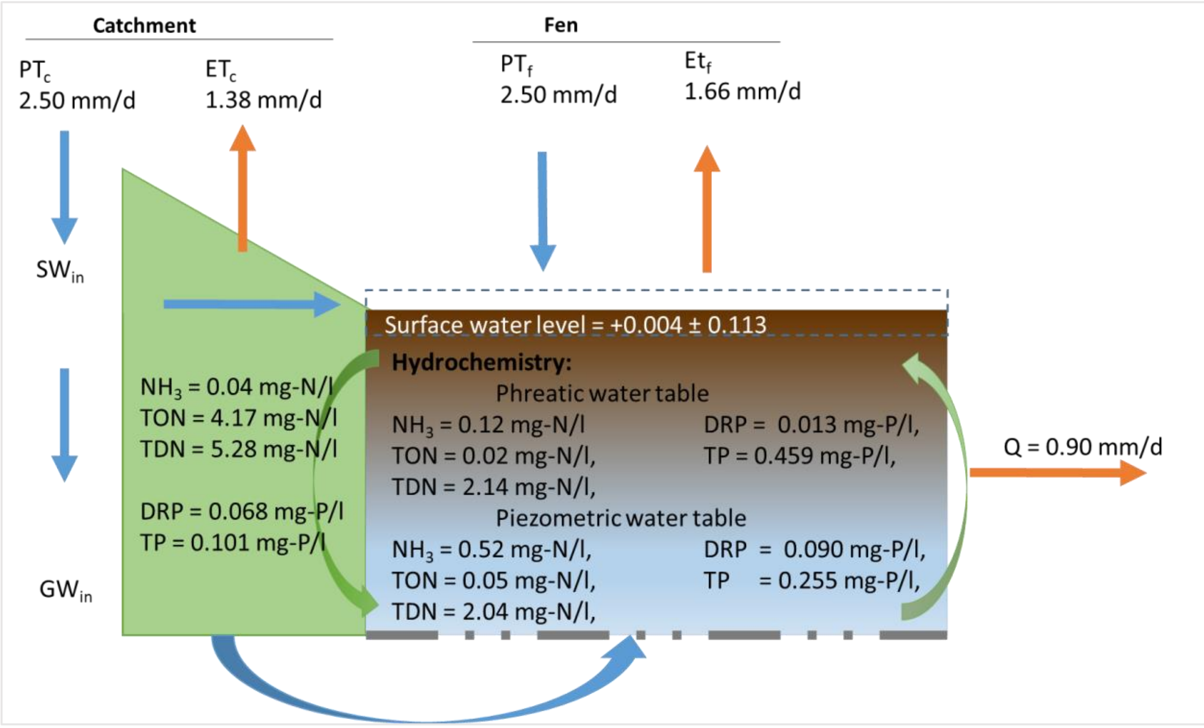


Figure 7.46. Conceptual box model of Pollardstown site D displaying the water balance, surface water level fluctuation and median nutrient concentrations in the fen and its catchment.

8. Results – Scragh Bog

8.1. Hydrology

8.1.1. Annual water balance

The size of the topographical catchment area of Scragh Bog, which was determined by drawing a polygon over the highest points surrounding the fen, was measured to be 1.1 km^2 (Figure 8.1). The fen area is 0.24 km^2.

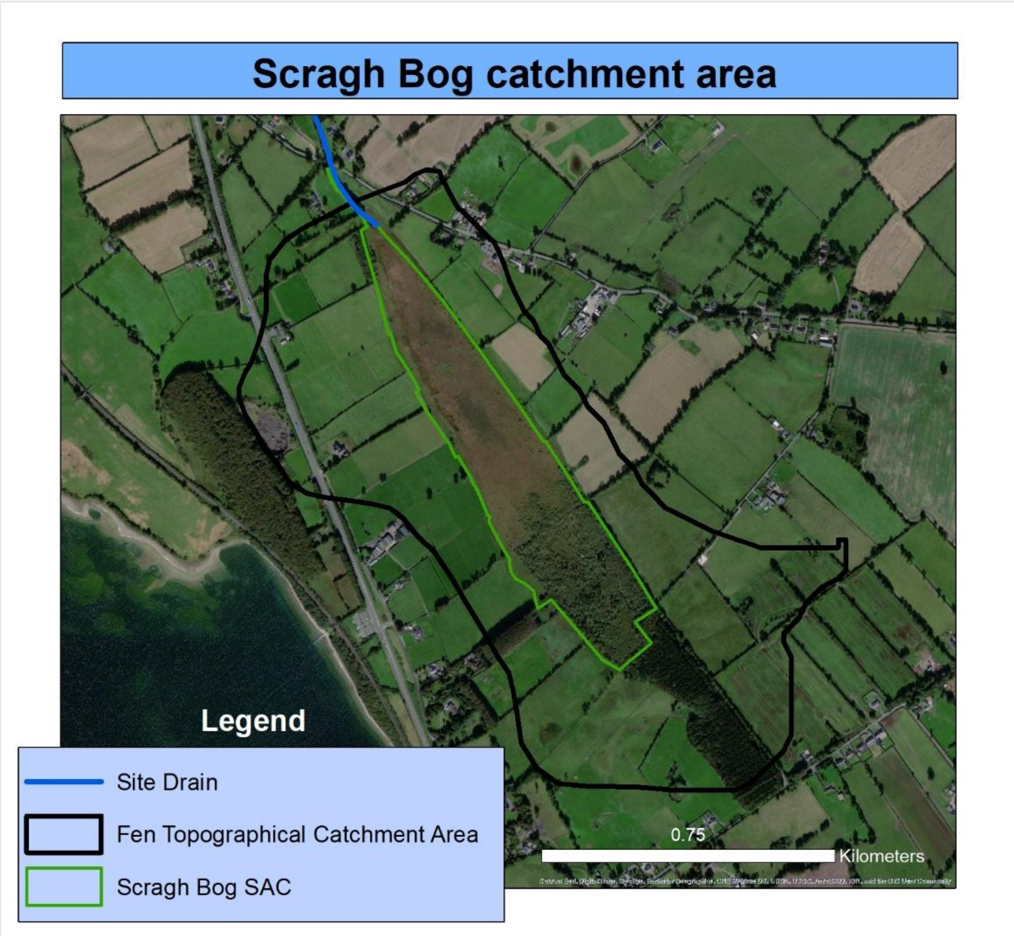


Figure 8.1 Topographic catchment area of Scragh Bog

During the calculation of the water balances there seemed to be significant difference between hydrological years 2018-2019 and 2019-2020. To even out the two years an investigation was done to account for the water level change within the fen. Total average water level changes from phreatic tubes were compared at the start of each hydrological year by taking the mean water level difference measured by divers installed across the fen. From this a water level increase was observed of 0.32 m in hydrological year 2018-2019. The hydrological year 2019-2020 saw a small decrease of 0.05 m. This data was then added to the water balance as an equivalent change in volume (i.e. water level change multiplied by fen area). This decreased the overall change between the two consecutive water balances to an almost insignificant difference.

However, in order to even out the water balances for both years the total catchment was adjusted such that the water balance equates to zero. When this was done the optimal catchment area for hydrological year 2018-2019 was 0.67 km^2 and the optimal catchment area for hydrological year 2019-2020 was 0.72 km^2 which is an 8.3% difference.

Nevertheless, for the water balance calculations in Table 8.1 a total catchment area of 0.70 km^2 was used which gives an even water balance when combining the two hydrological years. The calculated catchment area of the site is about 38.3% smaller than the topographical catchment area of 0.91 km^2 which is quite a large difference. This nonfixed water catchment type is thought to be attributed to the karstic geology of the catchment, where rainfall falling in the topographical catchment may not eventually end up in the fen. This mechanism called interbasin groundwater flow was described by Le Mesnil et al. (2021). Karst influences certain hydrological run-off procceses such as increased flood times and lateral outflow which can make it more likely for interbasin groundwater flow to occur. This may in turn also affect the discharge of a catchment, which was found in high alpine karst environment where discharge showed a distinct plateau from August to December followed by a recession until March (Krainer et al., 2021). Further evidence for the nonfixed nature of the catchment is discussed in Section 8.1.4, were contour lines are showing complex flowlines in Figures 8.7 and 8.8.

By estimation, from the recorded rainfall between October 1th 2018 and September 30th 2019 approximately 21.2% was lost from the fen and 32.6% was lost from the surrounding catchment as evapotranspiration. Furthermore, 33.9 % was lost as run off by the Scragh Bog's outlet (shown in Figure 8.5). The successive hydrological year saw 20.2% lost as evapotranspiration from the fen and 31.0% lost from the surrounding catchment which is not too different from the previous year. The daily flux of rainfall and evapotranspiration is also comparable to the previous year. However in contrast to the previous year a much larger percentage was lost to runoff (52.6%).

Table 8.2. Water balance of a hydrological years 2018/19 and 2019/20 in in Scragh Bog

01-10-2018 to 30-09-2019			
Fen water level change: + 0.32 m			
	Total (m^3)	**Flux (mm/d)**	**Fraction of rainfall**
Rainfall on fen	248441	2.84	
Rainfall on catchment	474150	2.84	
Evapotranspiration from fen	152966	1.75	21.2%
Evapotranspiration from catchment	235431	1.41	32.6%
Runoff from fen	242092	0.98	33.5%
Change in fen storage	76561	0.90	10.6%
Error in water balance	-15541	-0.06	**-2.2%**
01-10-2019 to 30-09-2020			
Fen water level change: - 0.05 m			
	Total (m^3)	**Flux (mm/d)**	**Fraction of rainfall**
Rainfall on fen	256097	2.93	
Rainfall on catchment	488762	2.93	
Evapotranspiration from fen	150177	1.72	20.2%
Evapotranspiration from catchment	231139	1.39	31.0%
Runoff from fen	391532	1.54	52.6%
Change in fen storage	-11963	-0.14	-1.6%
Error in water balance	16026	0.06	**+2.2%**

Figure 8.2 gives a more schematic overview of the water balance during hydrological years 2018-2019 and 2019-2020. As specified earlier in and seen in Tables 8.1 the rainfall on the fen and its surrounding catchment is roughly the same over the two years. This is also true for the evapotranspiration from the fen and its catchment. However, the runoff from the fen seems to change a lot between years. The fen loses only 242092 m^3 during hydrological year 2018-2019 compared to 391532 m^3 in the following year – an increase of roughly 62%.

Even taking into account the water level fluctuations in the fen during the two hydrological years, there is an apparent loss of approximately 16000 m^3 from the first year, a volume that is then subsequently gained back again in the next hydrological year in order to maintain the recorded phreatic water levels during that time. This apparent loss and gain can be associated with increases and decreases of storage in the aquifer, however also with the lags between the of water entering the fen from the subsurface of the surrounding catchment (as explained in Section 5.1). Still, Scragh Bog seems successful in storing water it receives from the catchment during periods of high recharge which is favourable for preventing floods downstream from the fen by attenuating runoff. Scragh Bog was able to store 76561 m^3 when regional aquifer levels were rising again after severely dry spells during May to July 2018 (Met Éireann, 2020). Moreover, when a certain water level height is reached relative to the height of Scragh Bog's outlet, the excess water will be discharged at a much higher rate. Higher fen water levels relative to the outlet show an increased discharge as seen at the start of hydrological year 2019-2020. At that time the fen water level is around 0.32 m higher than the start of the previous year which significantly affected the discharge between October 2019 and March 2020 (see Figure 8.2)

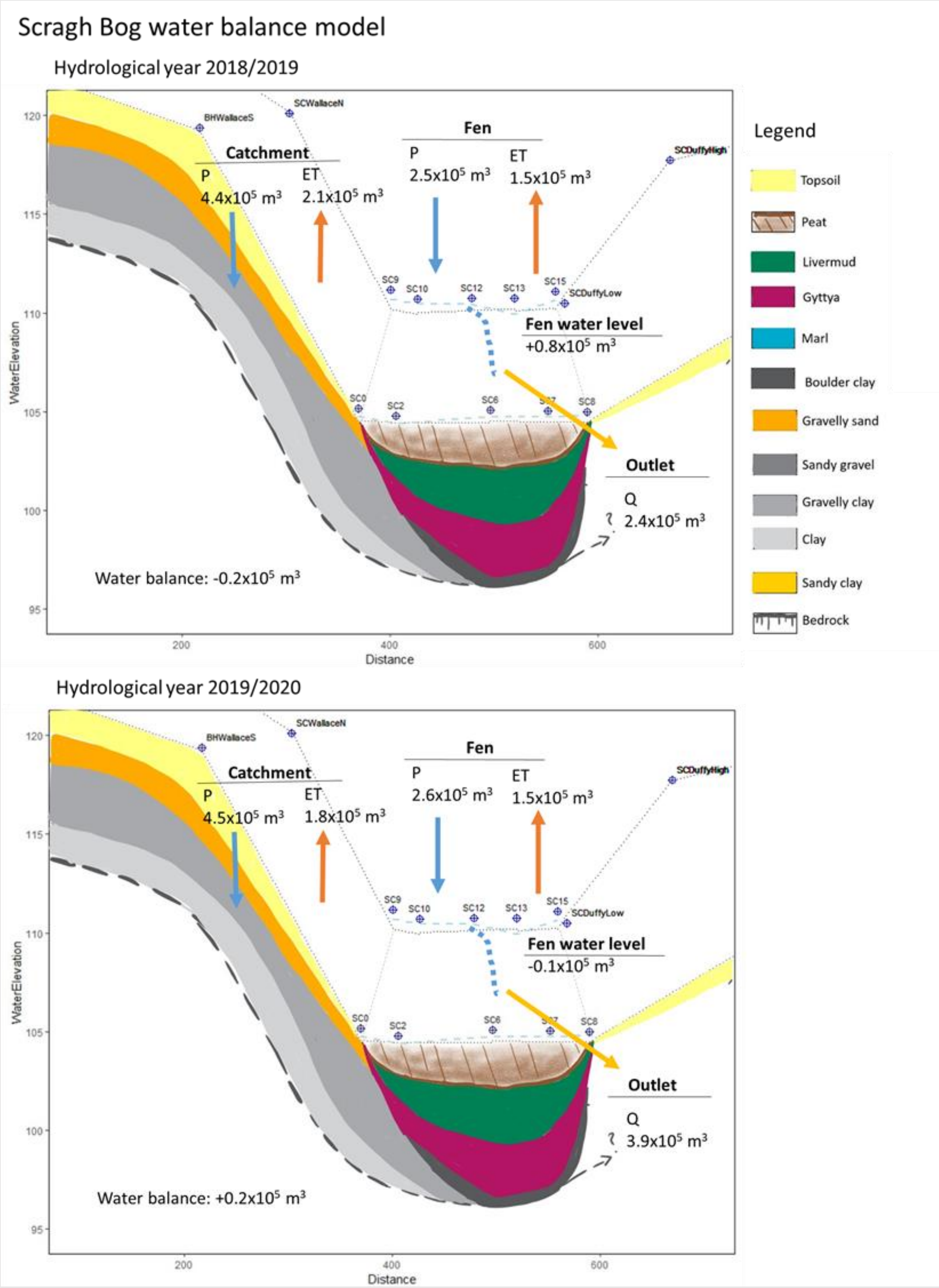


Figure 8.2. Water balance model of the hydrological year 2018/19 and 2019/20 in Scragh Bog

8.1.2. Seasonal water balance

8.1.2.1. Hydrological year 2018-2019

As seen in Table 8.2 there is a 75756 m^3 net volume loss from the fen into storage in the regional aquifer system which equates to a 22.0% of the total rainfall on the derived catchment. During the summer of 2018 Ireland experienced a significant drought (Met Éireann, 2020) whereby aquifer levels in the fen's catchment dropped significantly. The fen net groundwater input to the fen would therefore slowly reduce as storage in the regional groundwater table in the surrounding catchment reduces. A significant water level drop was reported by residents around the fen. Unfortunately, water levels were not measured during the drought so there is no certainty what the severity of the water level loss was. Furthermore, this loss could also be explained by partly be explained by the lag of water entering the fen via subsurface flow from the surrounding catchment as highly effective rainfall is recorded through much of March 2019 (in Figure 8.3), meaning a portion of water reached the fen during the summer water balance of 2018/19.

Nevertheless, the Scragh Bog did seem to recover well once the aquifer levels started to increase. This demonstrates the resilience against of the fen after a sustained drought since water levels did not drop beyond a base threshold and then recovered quickly. This is apparent in the winter water balance of 2018-2019 where the fen actually stored an additional 54071 m^3 water. This was found by calculating the phreatic water level at the start and the end of the hydrological winter season by taking the mean water level difference measured by divers installed across the fen. Even though the 19.2% of the seasonal rainfall was "lost" to storage in the regional groundwater, the water levels rose by 0.23 m in the winter. The fen also saw a big portion lost via runoff. Indeed, 65.2% of the annual runoff was lost during this season.

The hydrological summer saw another water level rise of 0.09 m which accounts for an additional storage of 22490 m^3. Since the aquifer levels had recovered during this time, the fen gained 15.9% of its water balance from groundwater recharge. The seasonal water loss through runoff is equally divided over the winter and summer with 54.0% and 46.0% respectively.

Table 8.3. Seasonal water balances of hydrological year 2018/19 in Scragh bog

01-10-2018 to 31-03-2019 (Winter)			
Fen water level change: + 0.23 m			
	Total (m^3)	**Flux (mm/d)**	**Fraction of rainfall**
Rainfall on fen	118503	2.74	
Rainfall on catchment	226162	2.74	
Evapotranspiration from fen	33079	0.76	9.6%
Evapotranspiration from catchment	50913	0.62	14.8%
Runoff	130845	1.04	38.0%
Change in fen storage	54071	1.25	15.7%
Error in water balance	-75756	-0.60	**-22.0%**
01-04-2019 to 30-09-2019 (Summer)			
Fen water level change: + 0.09 m			
	Total (m^3)	**Flux (mm/d)**	**Fraction of rainfall**
Rainfall on fen	129939	2.98	
Rainfall on catchment	247988	2.98	
Evapotranspiration from fen	119886	2.75	31.7%
Evapotranspiration from catchment	184519	2.22	48.8%
Runoff	111248	0.88	29.4%
Change in fen storage	22490	0.52	6.0%
Error in water balance	60215	0.48	**15.9%**

8.1.2.2. Hydrological year 2019-2020

The winter of hydrological year 2019-2020 (Table 8.3) revealed a bigger loss to the water balance (102943 m^3) than the previous year. However, all this water and more seemed to be gained back from the chages in the regional groundwater storage during the summer. As mentioned before fen water levels increased during 2018-2019. This lead to higher discharge rates in the following year as more of the recharge ended up in the fen (compared to going into to temporary storage in the aquifer). This loss may however also largely be due to the lag of water entering the fen from the subsurface of the surrounding catchment as high values of effective rainfall are recorded throughout March 2019. Approximately 70.5 % of the annual runoff was lost during this season, which an increase from the previous hydrological year. This higher discharge rate also be found in the outlet hydrograph (Figure 8.2). The loss to runoff and groundwater could have caused the phreatic water levels to drop by 0.13 m during the winter. These levels, however, were restored by 0.08 m in the summer.

Table 8.4. Seasonal water balances of hydrological year 2019/20 in Scragh bog

01-10-2019 to 31-03-2020 (Winter) Fen water level change: - 0.13 m			
	Total (m^3)	**Flux (mm/d)**	**Fraction of rainfall**
Rainfall on fen	146711	3.39	
Rainfall on catchment	279997	3.39	
Evapotranspiration from fen	31121	0.72	7.3%
Evapotranspiration from catchment	47899	0.58	11.2%
Runoff	276086	2.19	64.7%
Change in fen storage	-31342	-0.72	-7.3%
Water balance	-102943	-0.82	**-24.1%**
01-04-2020 to 30-09-2020 (Summer) Fen water level change: + 0.08 m			
	Total (m^3)	**Flux (mm/d)**	**Fraction of rainfall**
Rainfall on fen	109387	2.51	
Rainfall on catchment	208765	2.51	
Evapotranspiration from fen	119056	2.73	37.4%
Evapotranspiration from catchment	183240	2.20	57.6%
Runoff	115446	0.91	36.3%
Change in fen storage	19380	0.45	6.1%
Water balance	118969	0.94	**+37.4%**

8.1.3. Runoff

The discharge time-series of Scragh Bog's outlet is presented in Figure 8.3. The total daily rainfall amounts are also shown on the hydrograph. These were collected from the Met Eireann weather station in Mullingar approximately 5 km south from Scragh Bog. The calculated effective rainfall in mm/d is also plotted.

Discharges between May 5th 2018 and September 30th 2020 vary from 0 to 115 m^3/hr. There are clear differences in discharge rate between the winter and summer months. The outlet started the hydrological year of 2018-2019 with no discharge as the fen water levels were still recovering from the drought in the foregoing summer. In fact, the outlet did not show any runoff for 100 days until the 13th October 2018. After this the outlet flow increases in the following months with a peak discharge of 87 m^3/hr in March 2019. The runoff rate of hydrological year 2019-2020 was much higher with peaks up to 113 m^3/hr in the winter and spring.

The water level recovery is made apparent in the minimum discharge rates in 2019 and 2020. Even though the summer months have decreased runoff, the discharge never drops below 10 m^3/hr. Furthermore, despite there being little rain during the summer months of 2019 and 2020 there is a slow rate of hydrograph recession which suggest that Scragh Bog is able to store a large amount of water. This phenomenon may have be caused by the relative low decomposition of the peat in Scragh bog (as is characteristic of fens) as Kellner (2002) found that storage coefficient of peat decreases with further decomposition. Wetlands in general are highly capable of high water retention, but it is important to note that the rate is highly dependent on environmental, geological and hydrological factors (Bedford & Godwin, 2003).

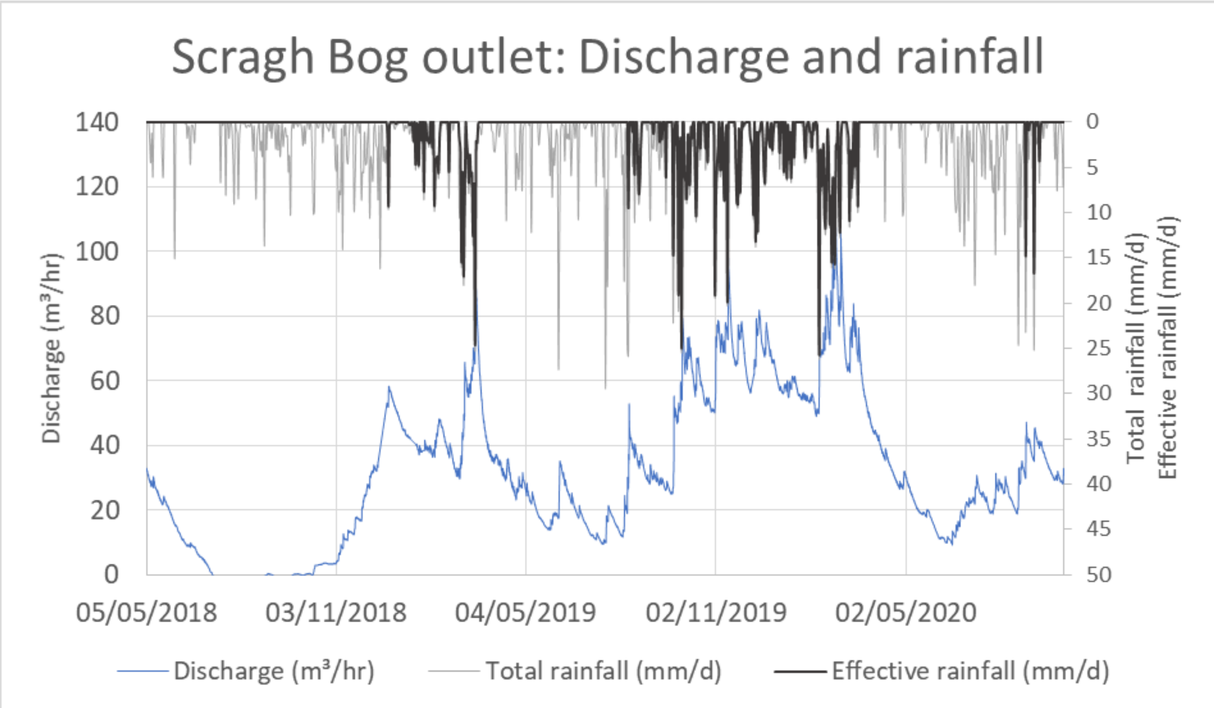


Figure 8.3. Scragh Bog outlet hydrograph and total/effective rainfall between October 1th 2018 and September 30th 2020.

A time-series of the total evapotranspiration against the discharge is displayed in Figure 8.4. The highest numbers were recorded in the summers with a total evapotransipiration of 6.5 mm/d in 2019 and 6.7 mm/d in 2020. The amount of discharge starts to decrease at the end of March in 2019 and 2020 while the evapotranspiration is increasing, as seen in Ballymore and Pollardstown fen. This trend seems to continue until there is a minimum of runoff during the summer. Then, when the evapotranspiration starts to decrease at the end of the summer, the discharge increases again.

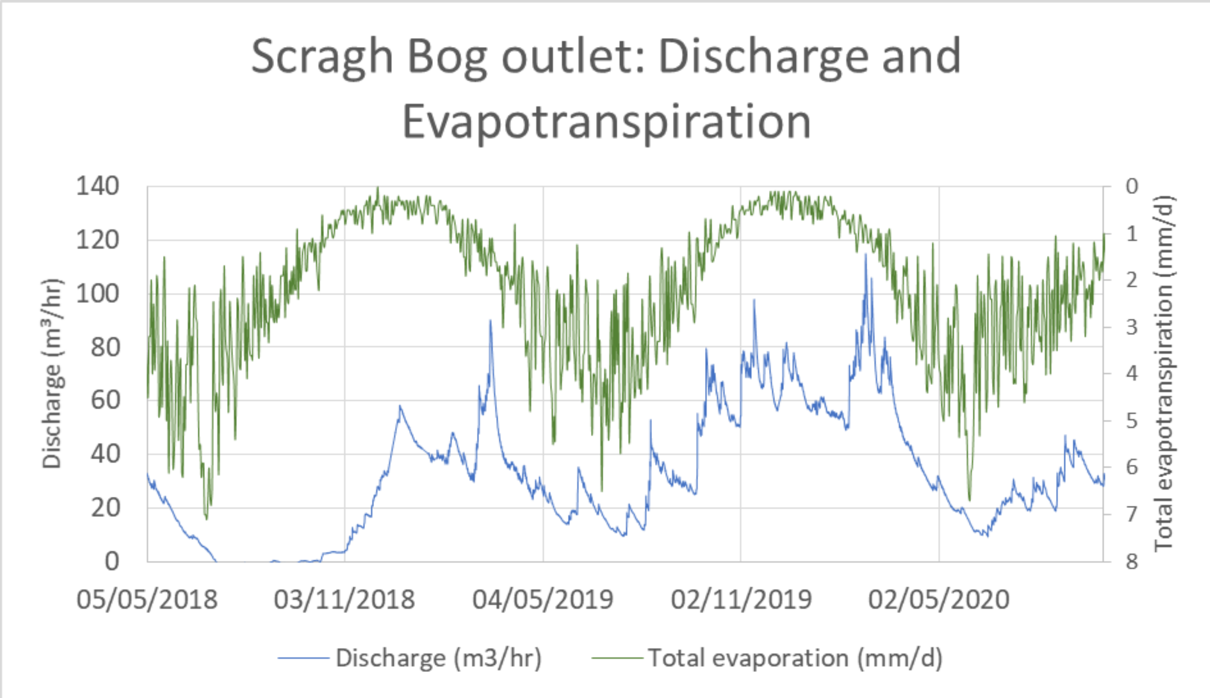


Figure 8.4. *Scragh Bog outlet hydrograph and actual ecapotranspiration between October 1th 2018 and September 30th 2020.*

The effect of daily evapotranspiration on the water levels of the fen is found in Figure 8.5. In the 20 day hydrograph of phreatic tube 17 a diurnal water level fluctuation of around 1 cm can be observed per day. This fluctuation does not show a daily gradual increase and decrease as was seen in Ballymore. The water does seem to decrease daily during the days although more often than not another small increase can be seen during the day. The cause of this could be the existence of diffused seepages in the area of site 17.

Additionally, the rainfall does not seem to always have the same effect on the water levels. The rain event of the 5th of June has a delayed response time and the water level increase in spread out over a couple of days. The discharge increase is delayed by about a day in response to the rainfall on the 10th of June, altough a clear peak can be observed this time instead of a gradual increase.

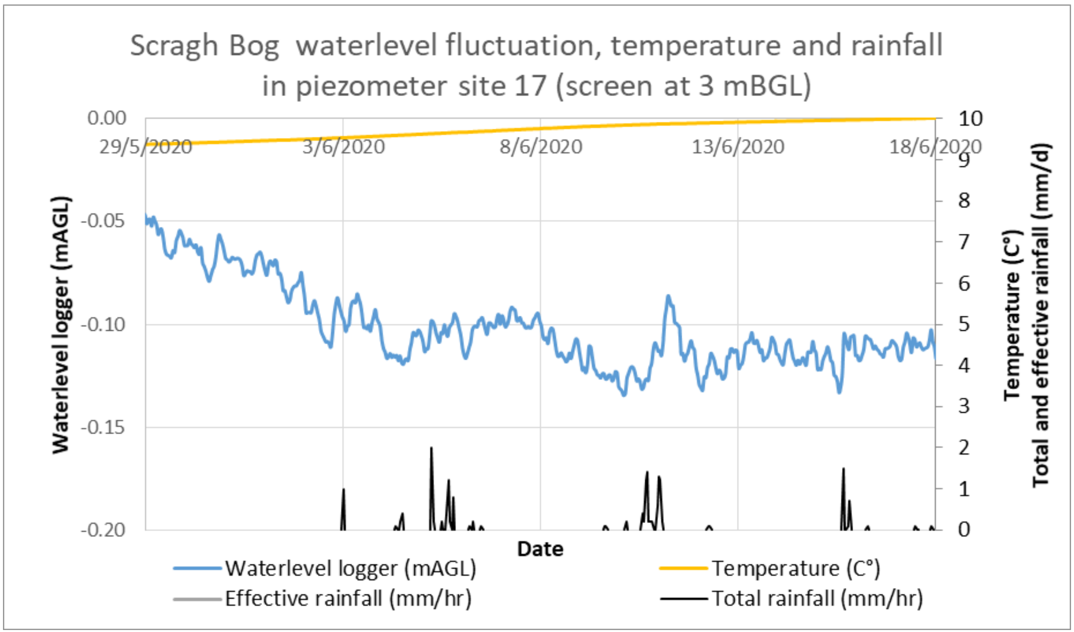


Figure 8.5. Scragh Bog phreatic tube hydrograph and temperature between March 18th 2019 and March 28th 2019.

8.1.4. Fen piezometer and phreatic tube data

Surface water points in the fen and groundwater table points around the fen were interpolated into contour lines in order to interpret the flow in and out of the fen (Figures 8.7 and 8.8).

Figures 8.9 and 8.10 show the water levels recorded in the phreatic tubes and piezometers between July 2018 and October 2020. The locations of the sites where this data was collected can be found in the Scragh Bog instrumentation map in Figure 8.6.

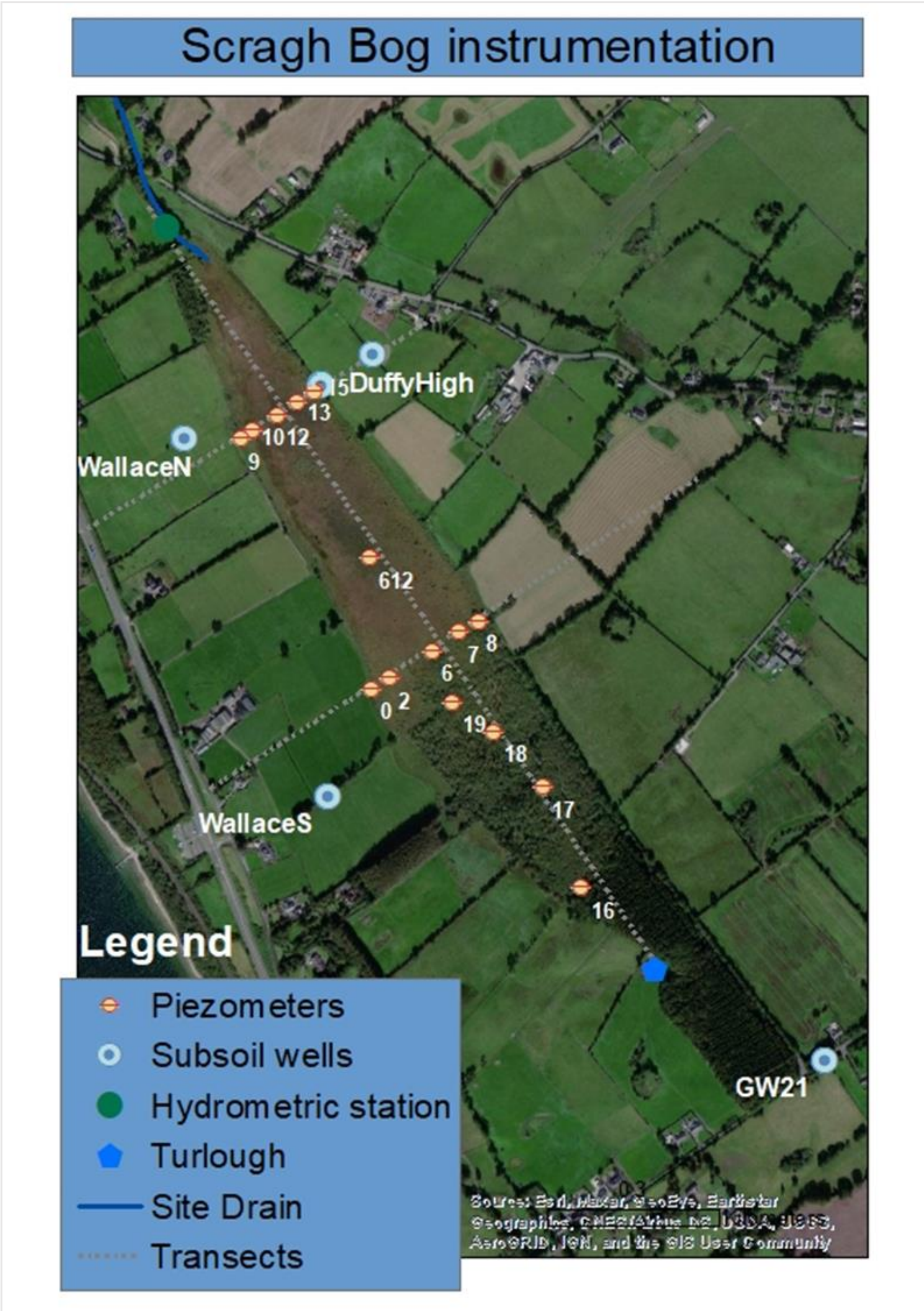


Figure 8.6. Scragh Bog instrumentation map showing fen piezometer and phreatic tube locations, subsoil well locations and the main site drains.

Flows are visible from the forest (southeast) into the fen in the contour maps of Figures 8.7 and 8.8. Flows are also observed entering the fen from adjacent fields in the Northeast. However, the fens also seems to discharge water to the catchment in the Southeast were it ultimately seems to end up in the eastern lake (Lough Owel). Levels around the Scragh Bog's outlet are slightly lower than surface water levels within the fen, which indicates that groundwater entering the fen is discharged into the outlet in the South.

In contrast to Ballymore and Pollardstown, however, not all water within the topographical catchment seems to end up in the outlet as is evident from flows from the fen into Lough Owel. Due to the karstic nature of the geology precipitation in the depicted catchment may not enter the fen but end up in the lake instead. This explains the error in closing the water balances in Tables 8.1 and shows how challenging it is to calculate this in a wetland that is underlain by karst. Other test such as dye tracer test would have to be performed in order to find out these discreet groundwater pathways.

However, if a fen is groundwater fed, which is the case for Scragh Bog, then the contours inevitably will reflect the regional water table. In this case that interactions is complex, given the interaction with karst.

Additionally the contours suggest there seems to be an underlying crossflow the south as well as a possible upwelling near the centre as is indicated in Figure 8.8.

Finally, while groundwater levels around the fen are much lower in the summer than in the winter, this does not seem to have a significant effect on the surface water levels in the fen as these remain relatively the same.

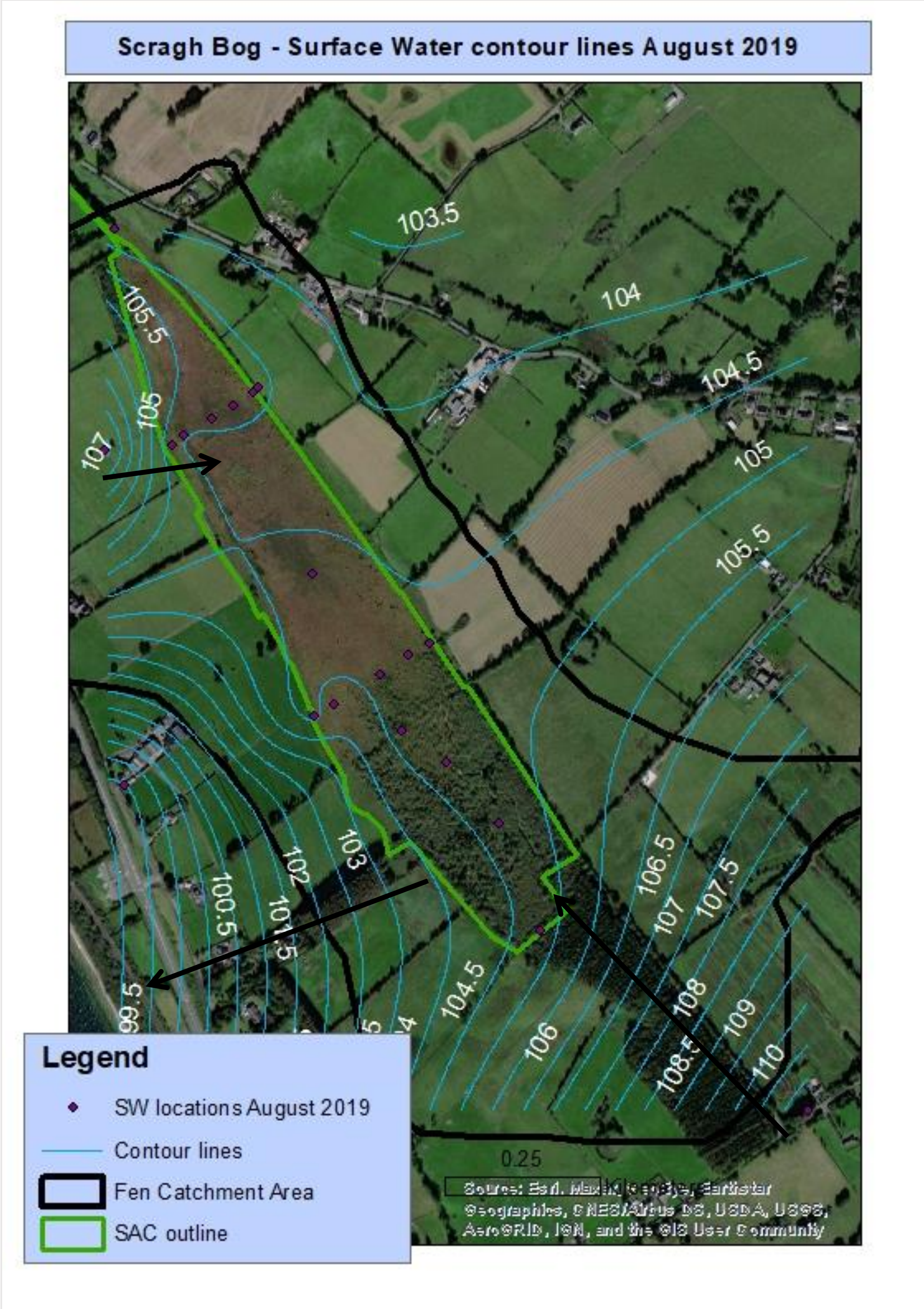


Figure 8.7. Contour lines of fen surface water and surrounding groundwater catchment interpolated using point measurements in August 2019. Flowlines are presented with black arrows.

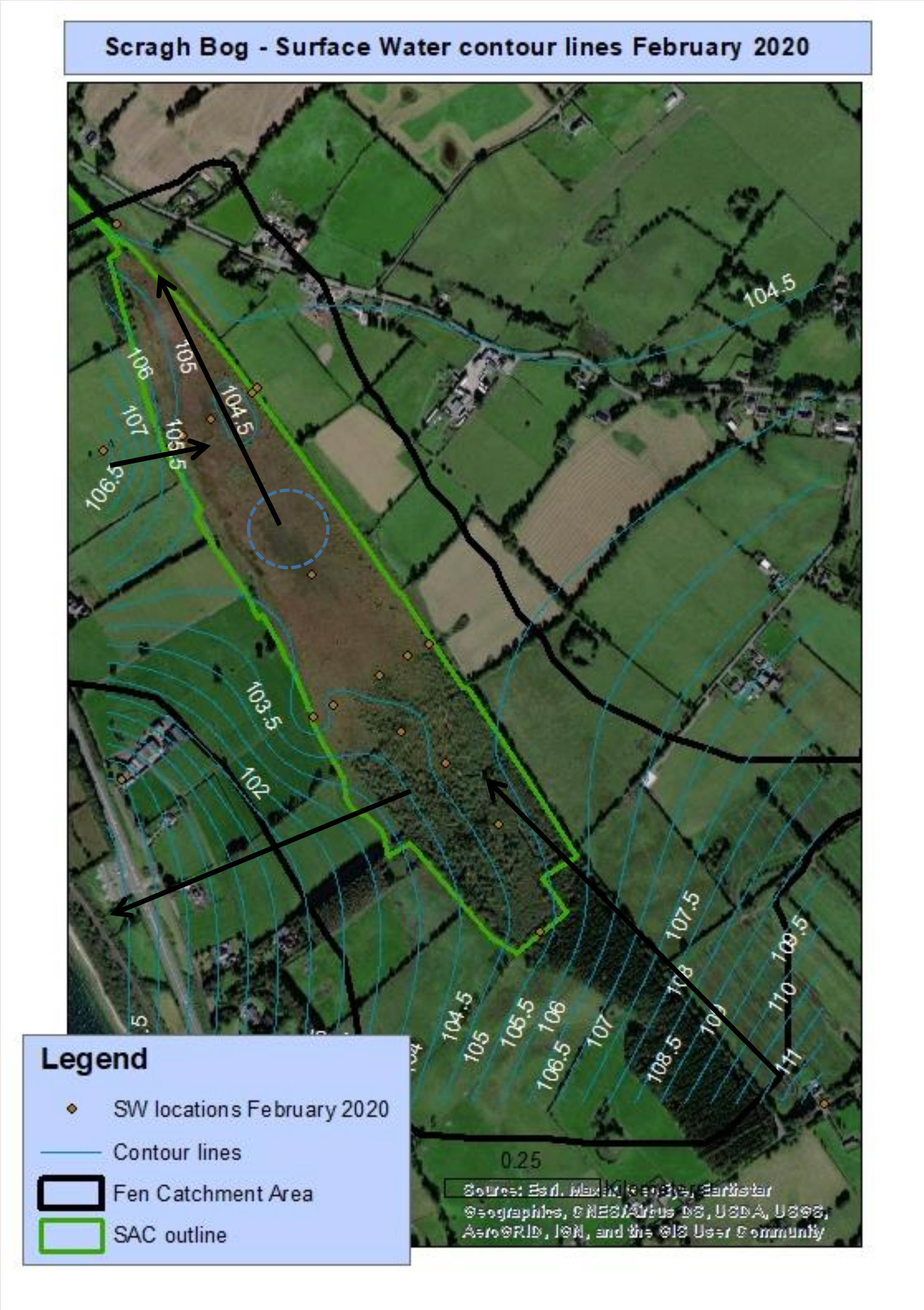


Figure 8.8. Contour lines of fen surface water and surrounding groundwater catchment interpolated using point measurements in August 2019. Flowlines are presented with black arrows.

The water levels in the phreatic tubes (Figure 8.9) for all sites are quite stable between even though they do show some response to both effective and total rainfall. However, the effective rainfall does seem to have a greater effect on the phreatic water levels than the total rainfall, reflected in higher water level increases during periods of high effective rainfall.

Even though the water levels decrease during periods of no effective rainfall they do not drop significantly and generally do not drop below the invert of the outlet measured in the area where the water leaves the fen. That being said, some particularly low water levels were recorded during the summer of 2018 in particular in phreatic tube SC9 located near the western edge of the fen. SC16 also decreased significantly during the following autumn. After this time the water level still saw decreases during the summer although the lowest recorded water level was still 0.5 m above the invert of the outlet.

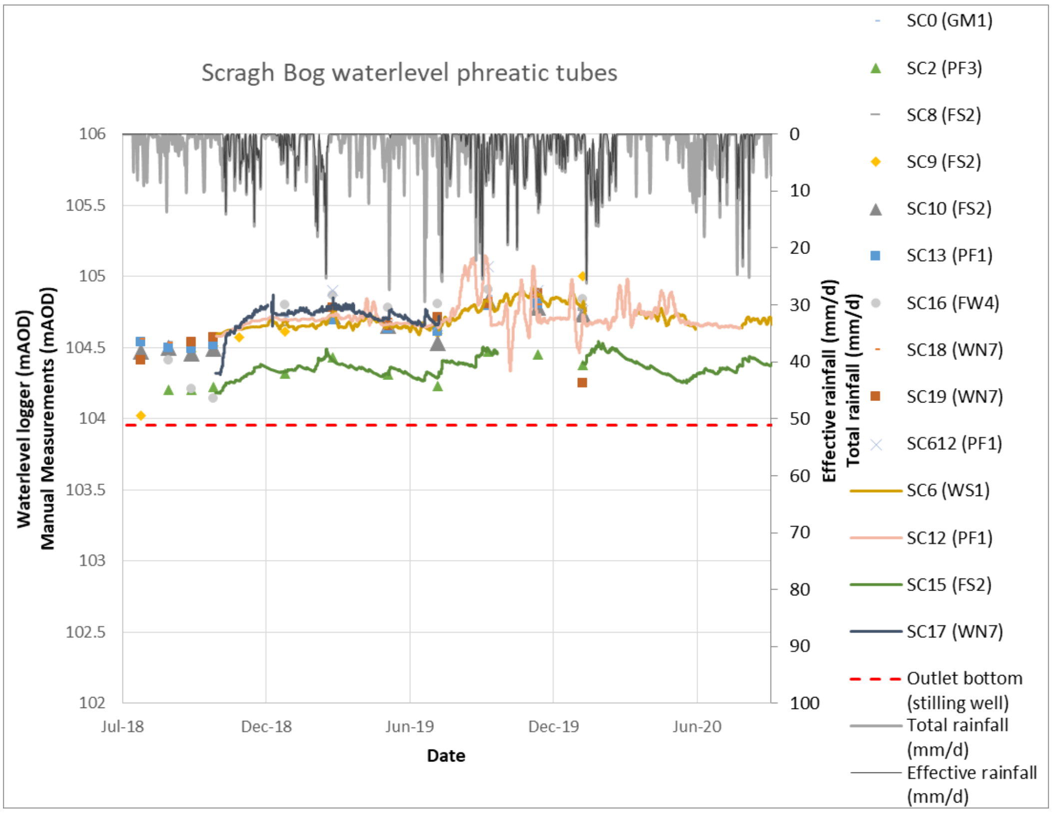


Figure 8.9. Phreatic water level hydrograph of spot measurements and water level loggers and rainfall. The height of the bottom of the outlet (measured at the flume) is presented with a red dashed line.

The piezometric water levels are shown in Figure 8.10. Some data had to be removed from the time series recorded by water level loggers since they were compromised due to using the same tubes for water sampling. After purging the piezometers sometimes showed increases due to water recovery instead of the actual water level fluctuations.

The summer drought of 2018 seemed to have a larger influence on the water level fluctuations measured in the piezometers. Here sites SC2, SC8, SC13, SC17 and SC19 were all recorded below the invert of the outlet even though there was no discharge at the time. Furthermore the hydraulic gradient was showing significant downward flows in those areas. This implies that there exist a downward loss from the fen in general when the head of the regional groundwater table reduces to such low levels.

The water levels of both the piezometers SC13 and SC19 approach the invert of outlet in summer 2019. These also show significant upward hydraulic gradients in the different fen soil layers during the follow winter. This implies that the internal water cycling in these locations are strongly influenced by the seasonal water inputs. There are permanent high flows recorded in SC18 which is located on a woodland bog where it seems that ombrotrophic conditions cause the water to be held higher by the peat.

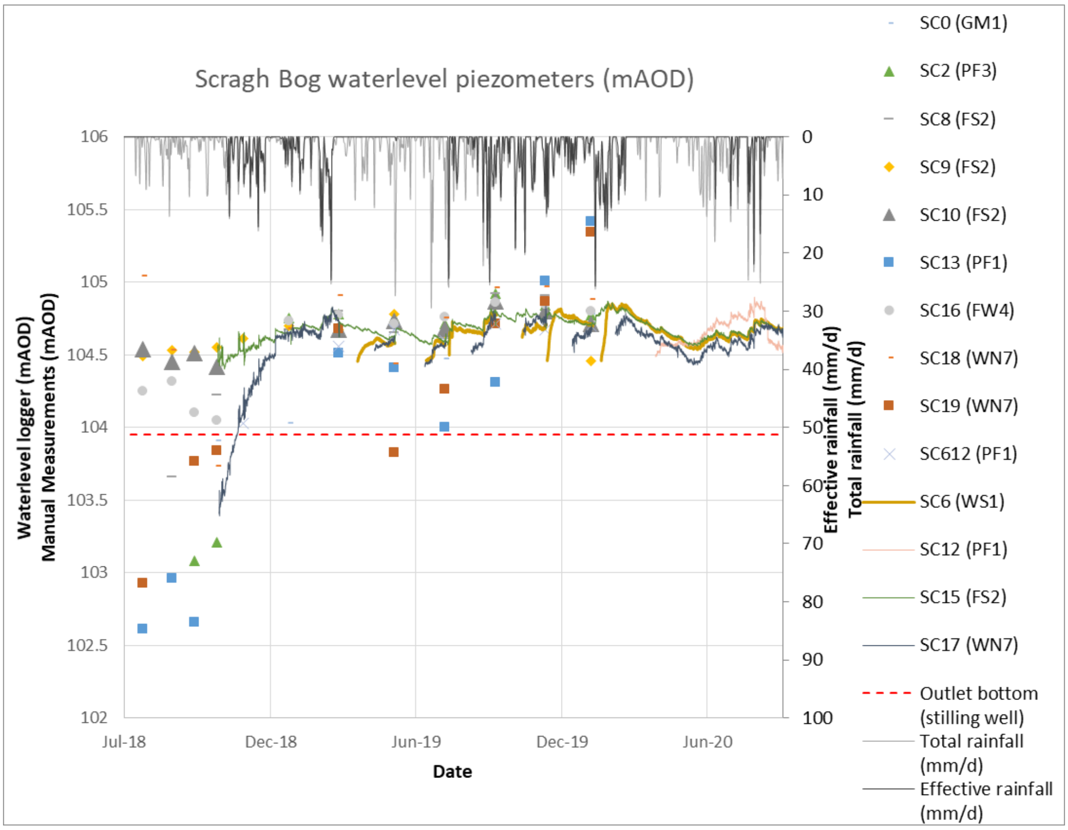


Figure 8.10. Piezometric water level hydrograph of spot measurements and water level loggers and rainfall. The height of the bottom of the outlet (measured at the flume) is presented with a red dashed line.

Figure 8.11 displays the phreatic and piezometer water level comparison of SC17 located in the southern bog woodland area of Scragh Bog. It is believed that multiple discrete springs feed the fen further up north (O'Connell 1981). The water levels are then again compared to subsoil well GW21 and subsoil piezometer WALN15

Groundwater levels are lot higher in the south of the fen (GW21) than in the north (WALN15) even though the ground surface elevations are comparable at 115.5 and 114.5 mAOD respectively. Hence, this would indicate that the groundwater moves in a south to north direction, although the obvious outlet for the groundwater is more across to the west. This outlet drains the fen water into the nearby lake (Lough Owel) which is located approximately 500 m west from Scragh Bog. Piezometer and phreatic tube SC17 are located about 700 m north from GW21 but show much lower water levels. The surface elevation decreases significantly further into the fen to 104.6 mAOD at SC17 which causes this stark contrast. The piezometers showed some particular water fluctuations in the dry summer 2018 which may be linked to the changes in the regional groundwater table. However, the following seasons show minimal reactions to the groundwater fluctuations measured in the subsoil piezometers which implies that under normal circumstances the groundwater table has very little influence on the fen water levels as the outlet offsets this response by the free discharge from the fen once it is above the outlet level.

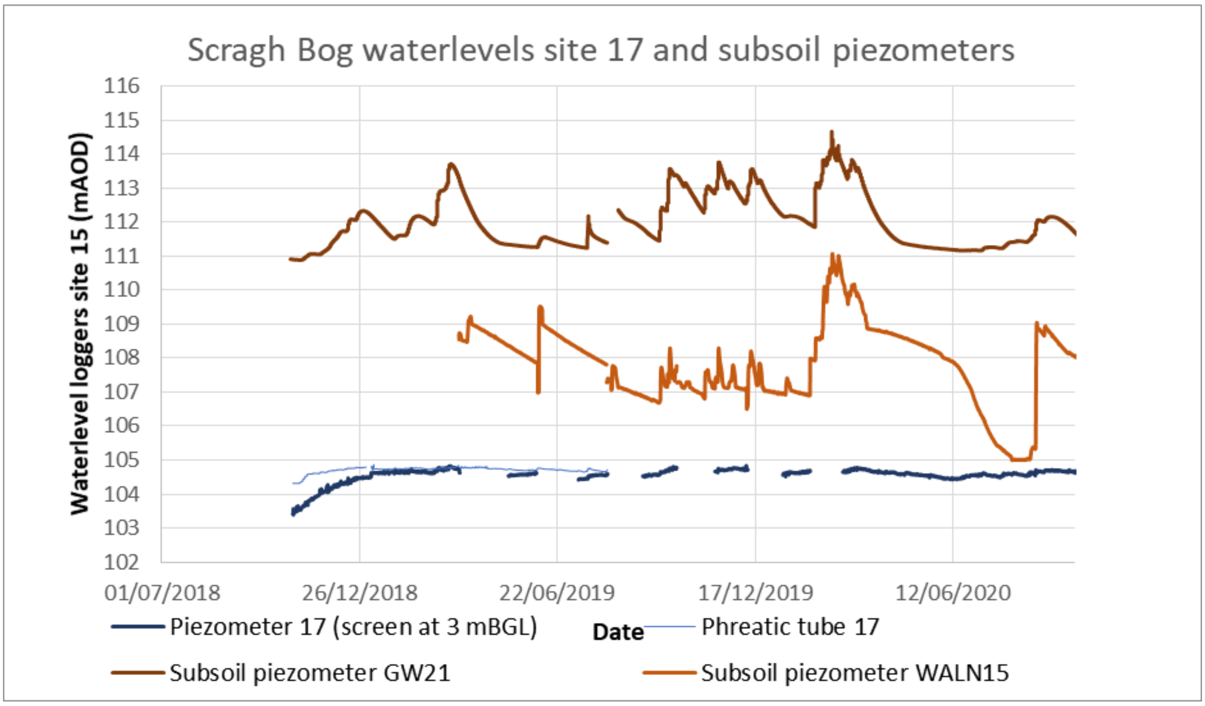


Figure 8.11. Hydrograph of phreatic and piezometric water levels at site 17 and piezometric water levels in subsoil well piezometers GW21 and WALN15.

Figure 8.12 further confirms this where both the phreatic and piezometric water tables in SC15 show minimal response to the groundwater fluctuations. It seems that Scragh Bog is controlled by both the local groundwater table held in the immediate hilly area surrounding the fen and by the aquifer under the fen and even though these areas are interlinked, they may at times show different upward or downward responses. SC15 is located at the north eastern edge of the fen next to adjacent hill. And it indeed seems that this location has an upward hydraulic gradient while downward a downward gradients are recorded in piezometers in the adjacent hill (see also Section 8.3.3). According to the water balance only a catchment of 0.70 km^2 is needed to close the hydrological year which was a lot smaller than its apparent topographical catchment. This further confirms that the fen can maintain its water table fairly level with only a small fraction of the recharge on the regional catchment.

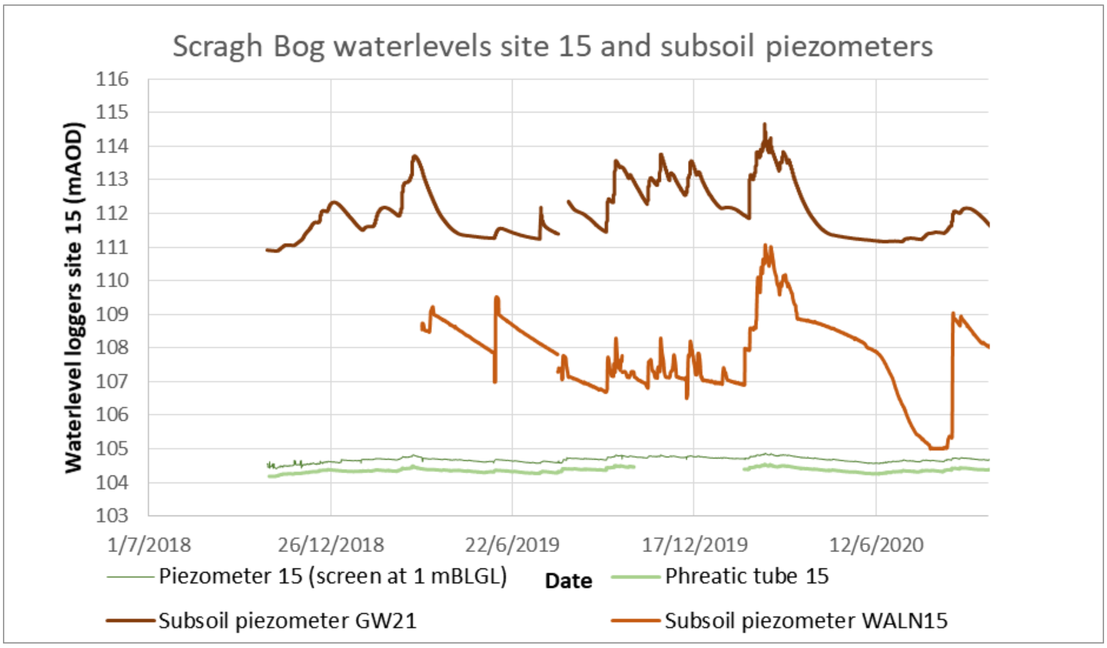


Figure 8.12. Hydrograph of phreatic and piezometric water levels at site 6 and piezometric water levels in subsoil well piezometers GW21 and WALN15.

8.1.4.1. Electrical conductivity

The time series of electrical conductivity (EC) in the phreatic tubes of Scragh Bog against rainfall (Figure 8.13) allows an estimation of the relative strength of groundwater inputs throughout hydrological years 2018/19 and 2019/20.

Overall, there does not seem to be any sort of correlation between effective rainfall and the recorded EC which implies that the balance between groundwater and surface water feeding the phreatic water table of Scragh Bog remains relatively even throughout the year. However, some different locations seem to receive different groundwater proportions. SC10 and SC17 were both recorded with a high EC between 600 and 800 µm/cm implying that these locations receive large groundwater proportions. It further seems that SC19 and SC7 are receiving consistently lower proportions of groundwater with and EC around 400 µm/cm. SC18 displays very constant low EC of just 100 µm/cm, which confirms the previous statement that this site is in fact an ombrotrophic bog.

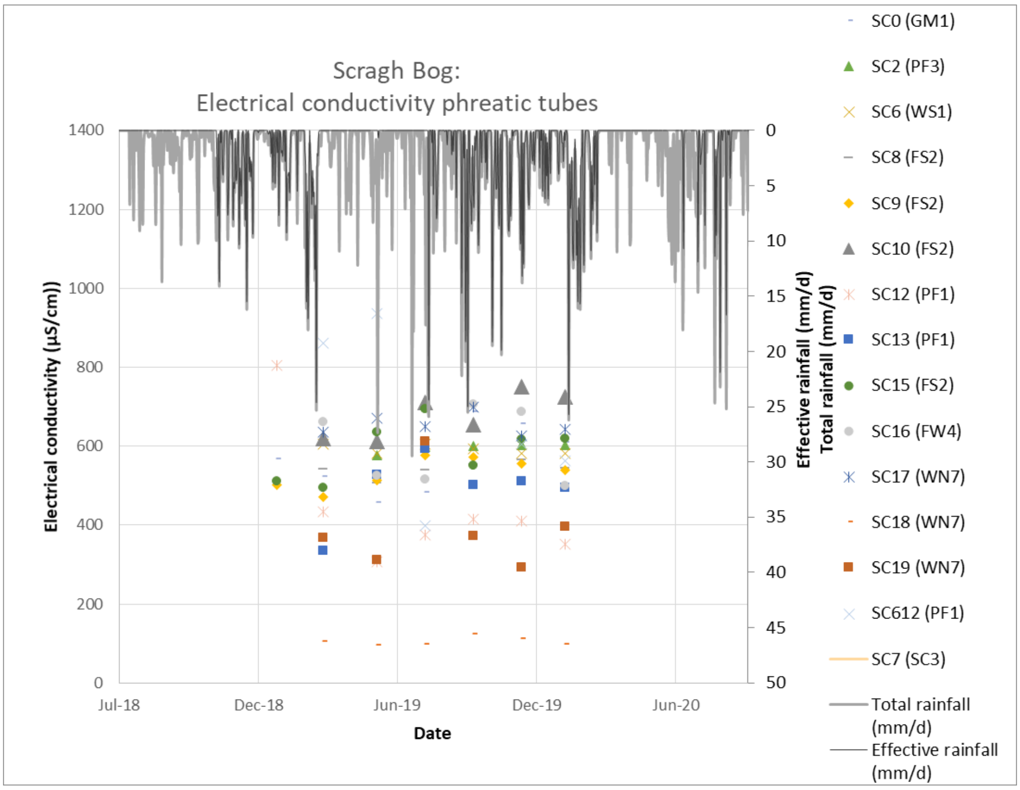


Figure 8.13. Time series of electrical conductivity (µm/cm) in phreatic tubes.

Figure 8.14 again shows no major EC fluctuations in most piezometers throughout the year which implies that also the piezometric water table is fed by groundwater / surface water mix at a relatively even rate throughout the year.

There is, however, some significant differences in the apparent relative groundwater proportions between different locations. Piezometer SC12 receives the highest proportions of groundwater with EC values of around 900 µm/cm. These values are generally higher than the EC found in the boreholes in the catchment of Scragh Bog with a median of 753 implying that this location is could be fed directly by groundwater from the limestone aquifer. It is however also possible that the lower sediments of the fen are acting to increase EC by chemical reactions between these layers and the limestone till.

Most other locations also receive moderately high groundwater inputs with an EC between 600 and 800 µm/cm. SC15 and SC17 appear to be fed by a mixture of groundwater and surface water and the proportions seem to fluctuate seasonally. SC0 located at the north eastern edge of the fen near the bottom of the eastern hill seems to fluctuate significantly from 200 to 600 µm/cm which implies that this site receives different groundwater proportions depending on the seasonal conditions.

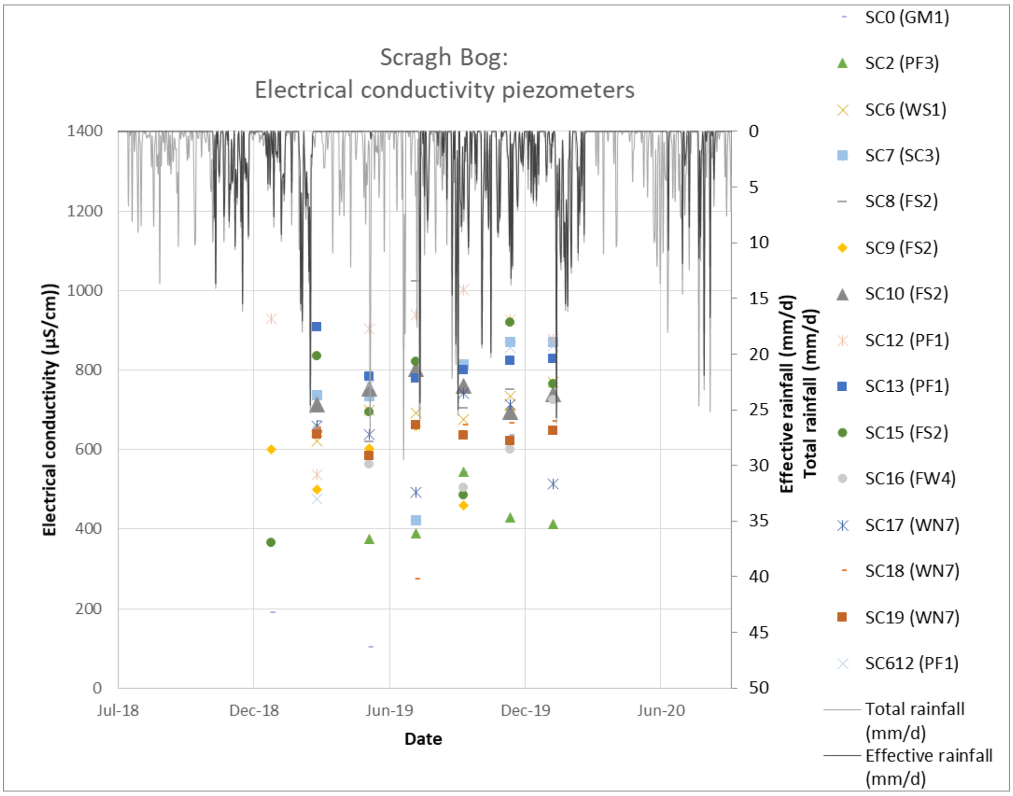


Figure 8.14. *Time series of eelectrical conductivity (µm/cm) in phreatic tubes.*

The boxplots in Figure 8.15 shows the overall and seasonal electrical conductivity from data collected in the fen as well as from boreholes outside the fen. It seems that while the EC in the piezometers is comparable to the boreholes, implying that they either mainly receive groundwater or that the fen soil layers react with the till, the phreatic tubes display much lower values implying the surface water table is fed by a mixture of groundwater and surface water.
Indeed according to a Welch t-test the boreholes (median of 753 µm/cm) were not significantly different from the piezometers (median of 672 µm/cm) with a p-value of 0.24.
Furthermore, it is interesting to note that the boreholes of Scragh Bog show bigger fluctuations of EC than Pollardstown. This is reflective of higher groundwater velocities through fractures and fissured limestone in the catchment of Scragh Bog compared to the boreholes around Pollardstown fen which are down into the large Curragh gravel aquifer with a more homogenous EC.

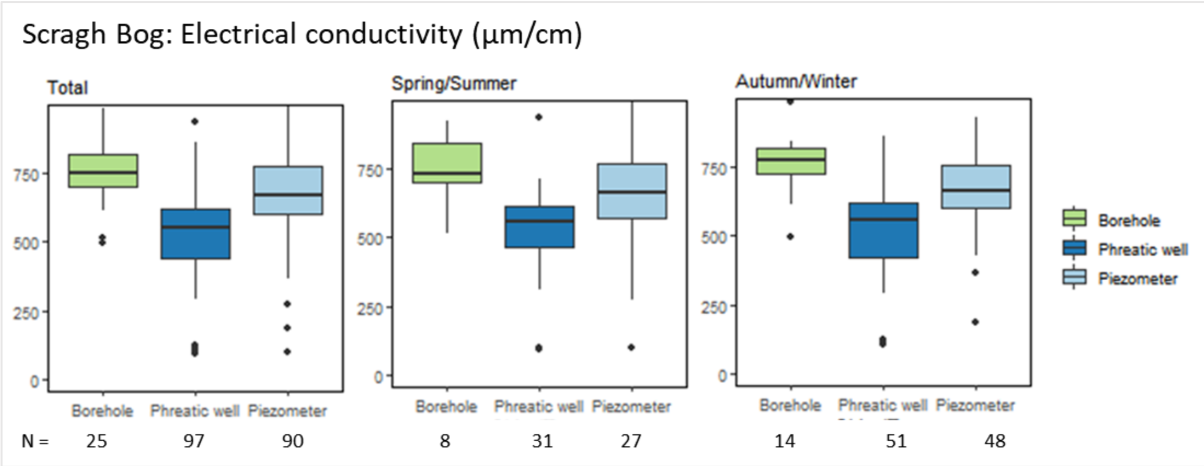


Figure 8.15. Electrical conductivity (µm/cm) in phreatic wells and piezometers inside and boreholes outside the fen in Scragh Bog.

8.1.4.2. pH

The pH results in Figure 8.16 suggest that values found in the fen are fairly similar to values found in boreholes outside the fen. However, it seems that the pH in the phreatic tubes is somewhat lower than the boreholes which further confirms that the phreatic water table is a mixture of (basic) ground water and more acidic surface water. Kooijman et al., (2016) found that decreasing inputs of base rich water to the surface of a fen may cause an increase Sphagnum species, and these mosses actively act to acidify wetlands. This procces was found in a Dutch fen where more isolated acidic areas were dominated by Sphagnum communities, whereas the base-rich areas was found with higher counts of brown moss (van Wirdum, 1991).

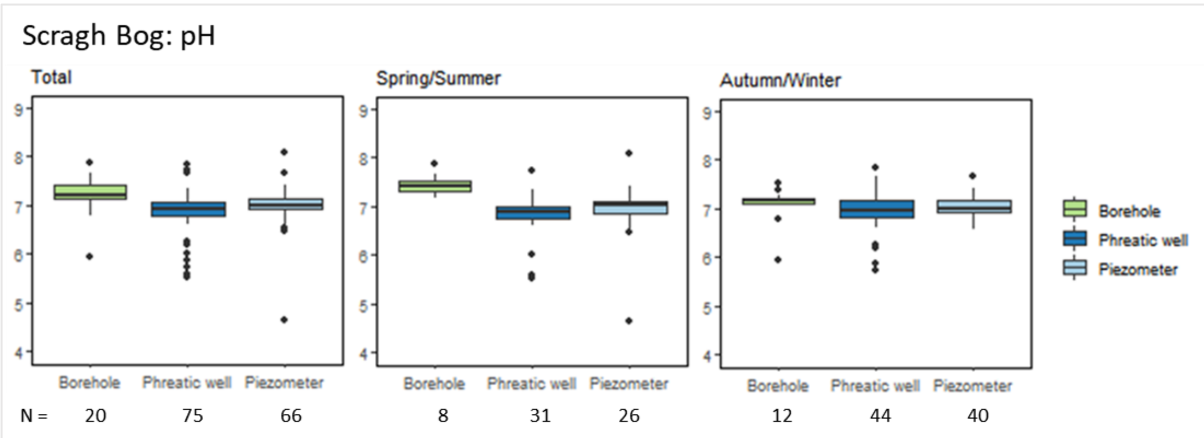


Figure 8.16. pH in phreatic wells and piezometers inside and boreholes outside the fen in Scragh Bog.

8.1.4.3. Temperature

The temperature boxplots in Figure 8.17 suggest that the temperature in the water column of the fen and the surrounding catchment are similar. Again a seasonal change was observed in the phreatic tubes, as expected for the water column at the surface . The Spring/Summer median is 13.1 °C whereas the Autumn/Winter median is much lower with 10.5 °C. .

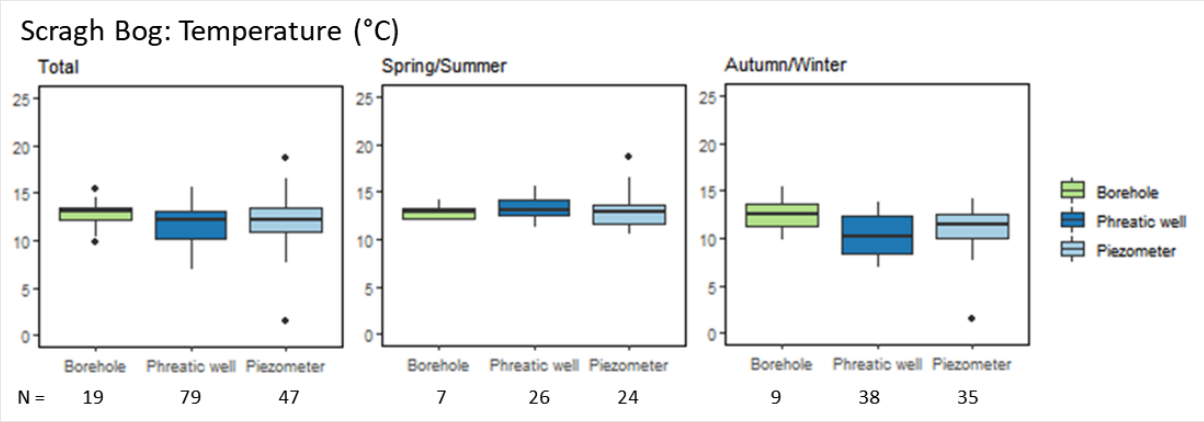


Figure 8.17. Temperature (°C) in phreatic wells and piezometers inside and boreholes outside the fen in Scragh Bog.

8.1.5. Conceptual hydrology model

Several findings can be summarised from results presented and discussed in previous sections:

- 0.24 km^2 fen is supported by a surrounding catchment of 0.46 km^2. The ratio of fen to catchment is approximately 1:2.
- As seen in Ballymore, discharge has the largest influence in changing the yearly water balance and may have significant fluctuations. During the significant droughts in 2018 the phreatic water levels did not drop below the invert of the outlet during this time. This is evidence of the resilience of the fen to such extreme climates as Scragh Bog is still fed by groundwater during this time, all be it at a reduced rate. This flux is acting acting to offset the loss from evapotranspiration in such a manner that the water level depths do not drop too low.
- The effect of daily evaporation was recorded in diurnal water level fluctuations.
- The lack of seasonal EC fluctuations in the phreatic tubes as well as the piezometers implies that the fen is fed by groundwater/surface water mix at a relatively even rate throughout the year.

8.2. Hydrochemistry

The following section contains a series of boxplots of the hydrochemistry data gathered in and outside Scragh Bog. A total of 334 samples were collected from boreholes, phreatic tubes and piezometers.

8.2.1. Phosphorus

Contrary to the other sites the total dissolved phosphorus (DRP) concentration in the surrounding catchment (median of 0.087 mg-P/l) is comparable to the concentrations found in the phreatic tubes (median of 0.055 mg-P/l) as seen in Figure 8.18. Indeed a two sided Welch test resulted in a p-value of 0.66 confirming the they are not statistically significantly different. It therefore seems that the DRP in the catchment is al least partly responsible (together with internal chemical cycling) for the higher concentrations in the phreatic water table of the fen. Even more curious is that the concentration of DPR in the piezometers (median of 0.190 mg-P/l) is statistically significantly greater than the concentrations outside the fen with a p-value of 0.00 . This may corroborate the theory that the small net downward gradients (see Section 8.3.1.2) transport the DRP into the sediments below from the phreatic water table to the underlying substrate. Phosphorus directly from the catchment as well as from the internal breakdown cycle in the vegetation does not seem to build up in the phreatic water table but is rather internally cycled by the wetland itself and then either lost as discharge from the fen, or some it is dispersed into he underying substrate layers.

The quality of the water in the phreatic tubes also seems to correspond to seasonal DRP fluctuations in the boreholes which could relate to the fen is a receptor of this nutrient from the catchment. Another possibility for these fluctuations is the sorption and/or precipitation during seasonal groundwater fluxes which influences the amount of oxygen in the water column (McBride et al., 2010)

The medians in the Spring/Summer were found closely related with 0.098 mg-P/l for the boreholes and 0.103 mg-P/l for the phreatic tubes (and were not significantly different - p-value of 0.54). The Autumn/Winter medians were 0.070 and 0.050 mg-P/l respectively and also not found to be significantly different (p-value of 0.25). There also seems to be higher DRP concentrations found in the piezometers during the Spring/Summer (median of 0.31 mg/L) and and lower during the Ayutumn/Winter (median of 0.21 mg/L) which may also be responsible for the seasonal DRP fluctuations in the phreatic tubes by internal cycling between the different sediment layers of the fen.

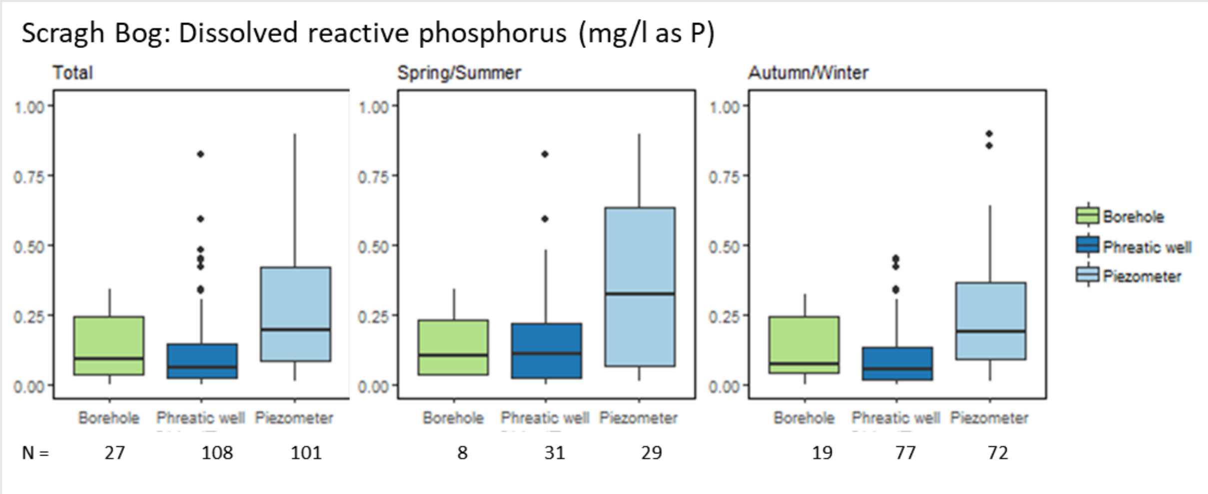


Figure 8.18. Dissolved reactive phosphorus in mg/l as P sampled from phreatic wells and piezometers inside and boreholes outside the fen in Scragh Bog.

The total phosphorus in the boxplots of Figure 8.19 display a similar trend where the concentrations of the boreholes (median of 0.234 mg-P/l) are comparable to the phreatic tubes (median of 0.200 mg-P/l). A two-sided Welch test returned with a p-value of 0.12 confirming the they are not statistically significantly different.

Additionally, it seems that more TP is stored in the phreatic layer than DRP which suggest a large portion of organic phosphorus is also present. This further suggests that downward gradients transport the TP into the underlying substrate which is reflected in the high concentrations of the piezometers (median of 0.328 mg-P/l). The piezometers were tested with significantly greater concentrations that the boreholes with p = 0.00.

Higher concentrations of TP were measured in the piezometers during the Spring/Summer than in the Autumn/Winter which might be due to more downward fluxes during the summer (see also Section 8.3.3) speeding up the downwards flux of the nutrients into lower layers of the fen. It is, however, more likely that some reductive dissolutions of ferric iron bound to phosphorus ore more complexe degradation is happening. This difference was, however, not significantly greater with a p-value of 0.45.

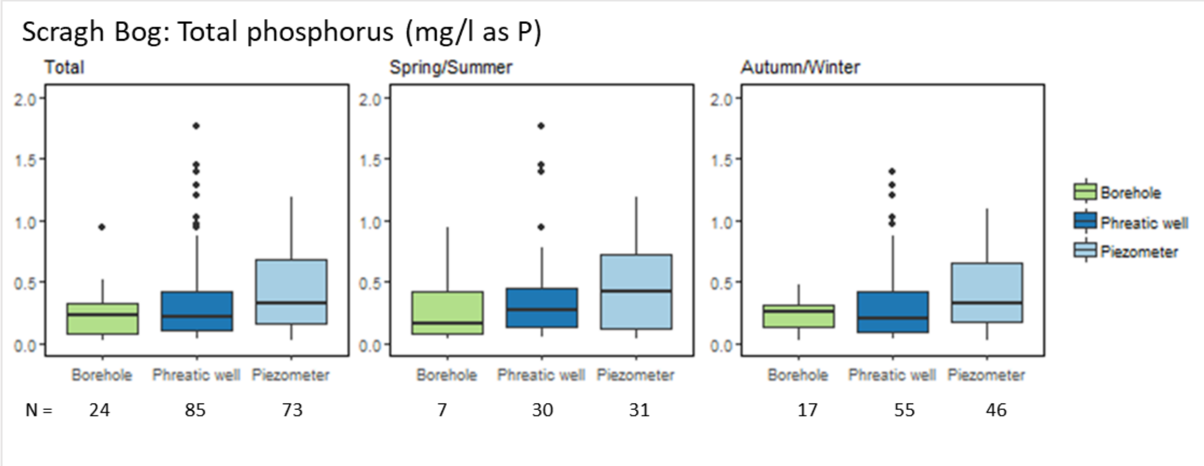


Figure 8.19. Total phosphorus in mg/l as P sampled from phreatic wells and piezometers inside and boreholes outside the fen in Scragh Bog.

The DRP and TP concentrations of the catchments are somewhat reflective of concentrations found in the phreatic water table of the fen. However, the substrate reveals statistically significantly higher concentrations as observed in the piezometers. The overall ratios of median DRP and TP were 1:3 in boreholes and 1:2 for piezometers, compared to 1:4 for the phreatic tubes. Finally, the DRP concentrations in the piezometers (median of 0.190 mg/l) substantially exceed the reported groundwater threshold values in Ireland (Government of Ireland, 2010) of 0.035 mg-P/l. mg-P/l. The threshold value is also exceeded in water sampled in the fen with medians of 0.056 mg-P/l in phreatic tubes and in the catchment with a median of 0.087 mg-P/l in boreholes. It is possible that there exists an overload of phosphorus from groundwater which cannot be effectively 'cleaned' by the fen vegetation and cycled internally. This overload may be caused by the presence of livestock and dairy farms which was found to be more intense around Scragh Bog than in other sites such as Ballymore and Tory Hill.

8.2.2. Nitrogen

The boxplots in Figure 8.20 displays relatively low total ammonia concentrations in boreholes outside the fen (median of 0.23 mg-N/l) and the phreatic tubes (median of 0.21 mg-N/l) in comparison to the piezometers which had a median concentration of 2.51 mg-N/l. The boreholes and phreatic tubes concentrations were statistically significantly less than the piezometers with p-values of 0.00 and 0.00 respectively. The concentrations found in the cathment may be reflective of the values found in the phreatic water table as they were found not significantly different using a two sided Welch test (p-value of 0.53). However the ammonia lower sediments of the fen is not reflective of concentrations found the regional catchment. This is due to due to the break down processes of vegetation at the surface from which the nutrients are the moved to the underlying substrate through downward hydraulic gradients. This decay process will causes organic nitrogen and ammonia to build up over time in the piezometric layer, especially due to the anoxic conditions. The fact that there is no seasonal difference in the concentrations between Spring/Summer and Autumn/Winter may suggest very little change in surface/groundwater proportional flow in the piezometric layer. It is also possible that the internal nutrient cycling is dominating in the production of ammonia under strong reducing conditions (Bedford & Godwin 2003; McBride et al., 2010), as is supported by lower pH level in the phreatic tubes, and is under little influence from relative change in feed to the fen

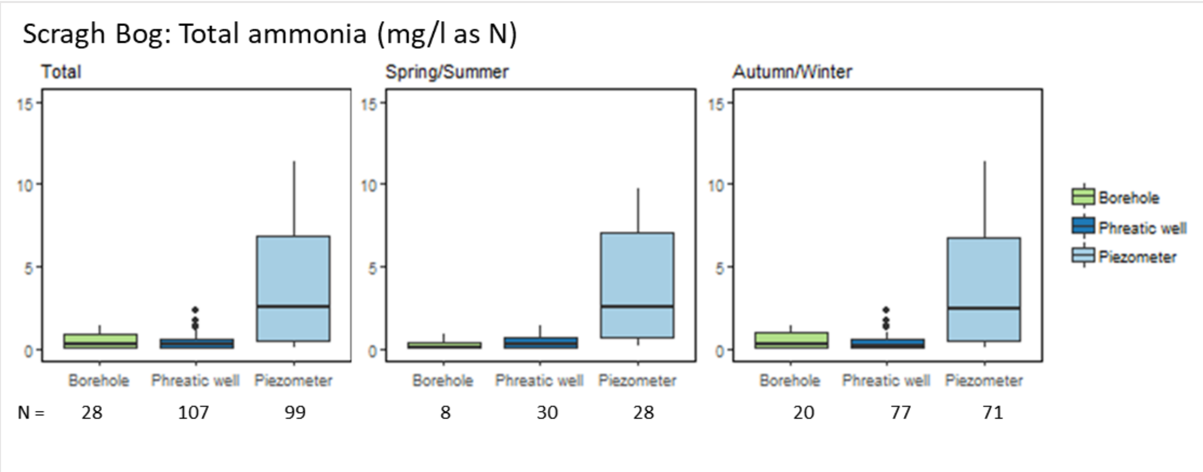


Figure 8.20. Total ammonia in mg/l as N sampled from phreatic wells and piezometers inside and boreholes outside the fen in Scragh Bog.

Nitrite was barely found in and around the fen, as shown in the boxplots of Figure 8.21. Most samples were analysed below the limit of detection which was 0.05 mg-N/l. However, some spikes up to 0.17 mg-N/l were seen in the piezometric data, which could indicate some sporadic nitrification processes in the soil at the time of sampling. Such occasions were mainly observed during the spring and summer within soils underlying the peat.

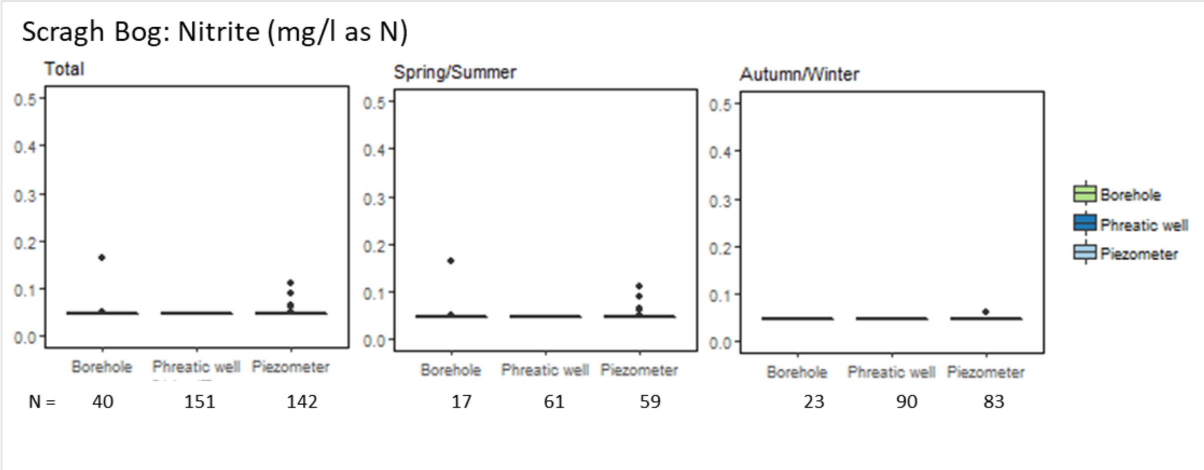


Figure 8.21. Nitrite in mg/l as N sampled from phreatic wells and piezometers inside and boreholes outside the fen in Scragh Bog.

Total oxidised nitrogen results reveal higher concentrations in the boreholes, (see Figure 8.22) with a median of 1.40 mg-N/l, compared to the medians of the phreatic tubes and piezometers at 0.01 and 0.02 mg-N/l respectively. The values in the phreatic tubes and the piezometers were proven to be significantly lower than those found in the boreholes with p-values of 0.00 returned by both tests.

The fact that the fen is found with such low conditions corroborates the previous suggestion of anaerobic conditions in the piezometric layer. The high concentrations of total oxidised nitrogen are filtered out by the vegetation in the phreatic water table and is transformed into organic nitrogen. The nitrogen is then put back into the water column due to the seasonal vegetation break down and again transformed in to total ammonia which ultimately travels into the sediments of the underlying substrate.

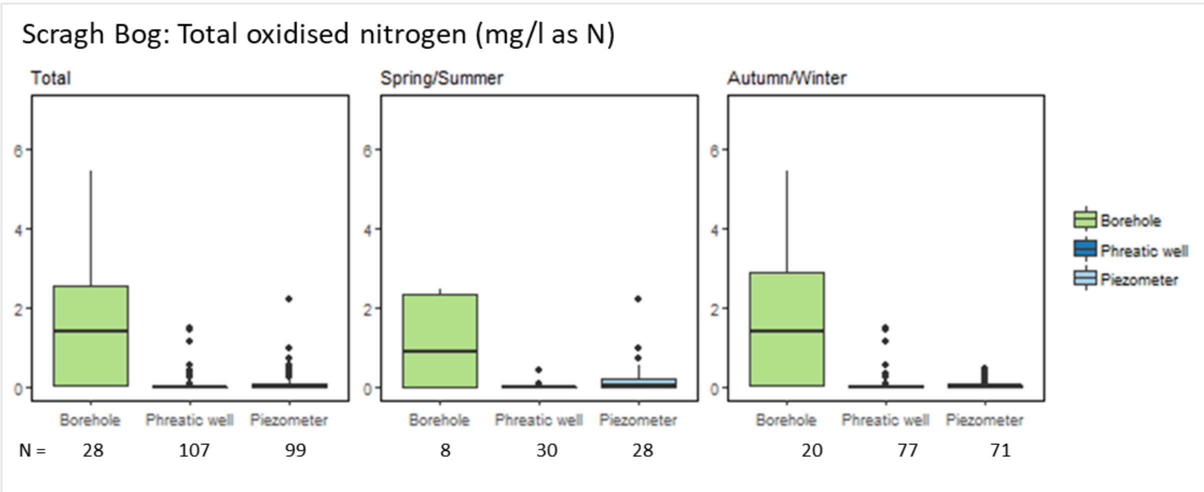


Figure 8.22. Total oxidised nitrogen in mg/l as N sampled from phreatic wells and piezometers inside and boreholes outside the fen in Scragh Bog.

The total dissolved nitrogen is also a reflection of this process with high values found in the piezometers of Figure 8.23. The nitrogen fractions that are not filtered out by the vegetation in Scragh Bog are transferred to the sediments in the piezometric layer the downward gradients.

Total dissolved nitrogen results in the piezometers display a median of 3.38 mg-N/l, higher compared to the medians of the phreatic tubes and boreholes at 1.35 and 2.26 mg-N/l respectively. The values in the phreatic tubes and the boreholes were proven to be significantly lower than those found in the piezometers with p-values of 0.00 returned by both tests. This also indcates that the nitrogen entering the fen with the groundwater is mainly in nitrate form which is indicative of aerobic conditions in the incoming feed water.

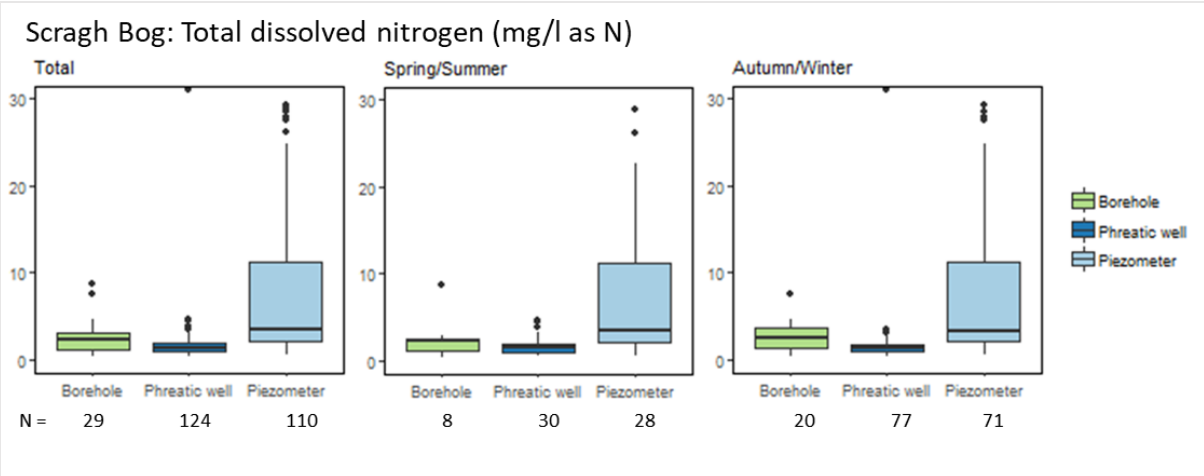


Figure 8.23. Total dissolved nitrogen in mg/l as N sampled from phreatic wells and piezometers inside and boreholes outside the fen in Scragh Bog.

The ratios of total ammonia to total dissolved nitrogen were 1:10 in boreholes and smaller piezometers with 1:1 and the phreatic tubes with 1:6. The ratio of total oxidised nitrogen to total dissolved nitrogen was much lower for the boreholes at 1:2, compared to 1:169 and 1:135 in the piezometers and phreatic tubes respectively. These ratios imply that a significant portion of the TDN in the piezometers and phreatic tubes is made up of ammonia and only a very small proportion of total oxidised nitrogen. From the reported medians it was found that of TDN 25% was found made up of different forms of organic nitrogen in the piezometers. The phreatic water table holds a much greater proportion of TDN in organic form with 83%.

None of the medians found in and around the fen were found higher than 0.114 mg-N/l for nitrite which is the groundwater threshold values in Ireland (Government of Ireland, 2010). There were, however, a few outliers that did exceed the threshold value. The total oxidised nitrogen concentrations had an insignificant amount of nitrite and a therefore can be regarded as a reflection of nitrate. In and around Scragh Bog, none of the measured concentrations exceeded the threshold value of 8.47 mg-N/l.

8.2.3. Other chemistry

The overall concentrations of alkalinity were quite similar when comparing data in the fen and in its bedrock aquifer, as seen in Figure 8.24. Indeed, a two-sided Welch t-test proved no significant difference between the boreholes and the phreatic tubes (p-value = 0.59) and the piezometers (p-value = 0.14). The median in the boreholes was 205.1 mg/l as $CaCO_3$, whereas the median of the piezometers was somewhat higher at 225.4 mg/l as $CaCO_3$ reflecting high groundwater inputs into the fen. Again the alkalinity in the phreatic tubes is lower with a median of 184.9 mg/l as $CaCO_3$ which is a reflection of the mixed groundwater and surface water portions at the surface of the fen and/or the ability of the vegetation to take up minerals.

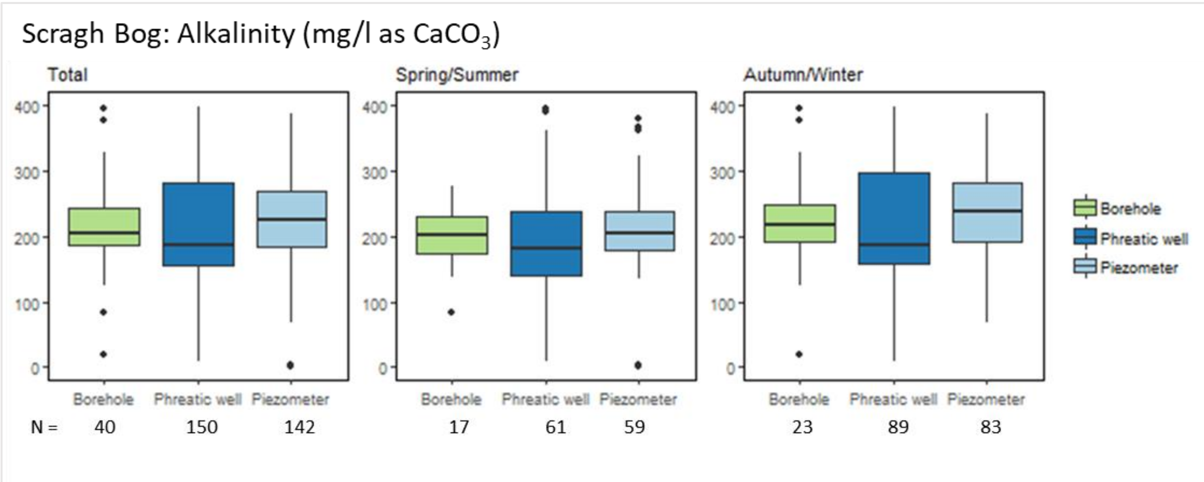


Figure 8.24. Alkalinity in mg/l as $CaCO_3$ sampled from phreatic wells and piezometers inside and boreholes outside the fen in Scragh Bog.

The overall concentrations of chloride (Figure 8.25) were somewhat higher in the boreholes around the fen (median of 24.2 mg/l) compared to measurements taken in the phreatic tubes (median of 17.8 mg/l) and in the piezometers (median of 17.4 mg/l). The concentrations in the boreholes were tested to be significantly greater than in the phreatic tubes and piezometers with p-values of 0.00 returned by both tests. The reported medians are far below the Irish groundwater threshold values for chloride (Government of Ireland, 2010) which is 187.5 mg/l.

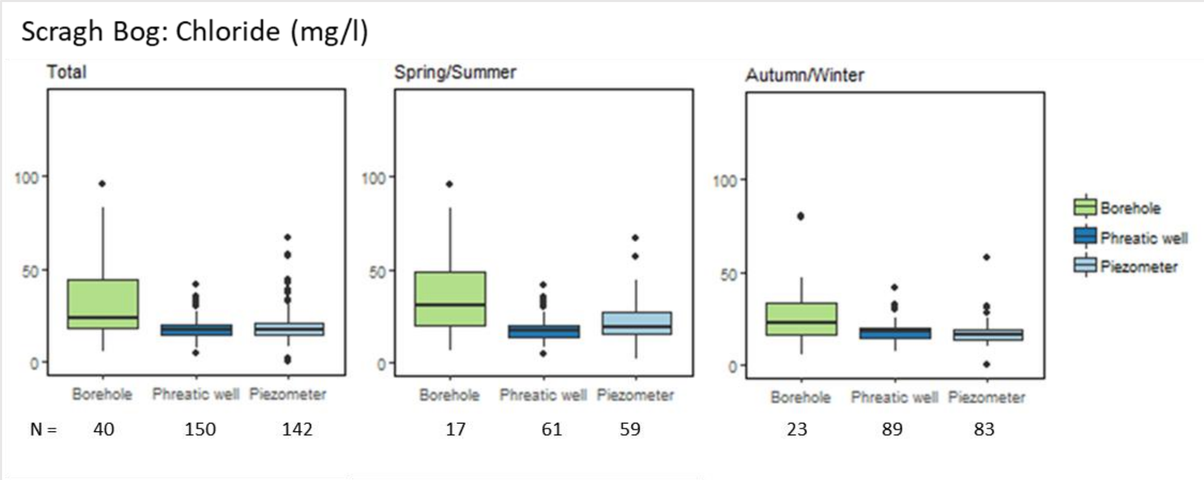


Figure 8.25. Chloride in mg/l sampled from phreatic wells and piezometers inside and boreholes outside the fen in Scragh Bog.

The concentrations of silica in the piezometers (median of as 6.8 mg/l as SiO_2) are comparable to the concentrations found in the catchment (median of 9.2 mg/l as SiO_2) as seen in Figure 8.26. This implies that the piezometric layer is either heavily infuenced by groundwater inputs, or contains the same sort of calcareous sediments as the bedrock aquifer. Indeed a two-tailed Welch t-test proved that they were not significantly different with a p-value of 0.56. Contrary to this are the concentrations found in the phreatic tubes which were statistically significantly (p-value = 0.00) much lower with a median of 3.2 mg/l as SiO_2. This again implies that phreatic layer of Scragh Bog is fed by a mixture of surface water and groundwater.

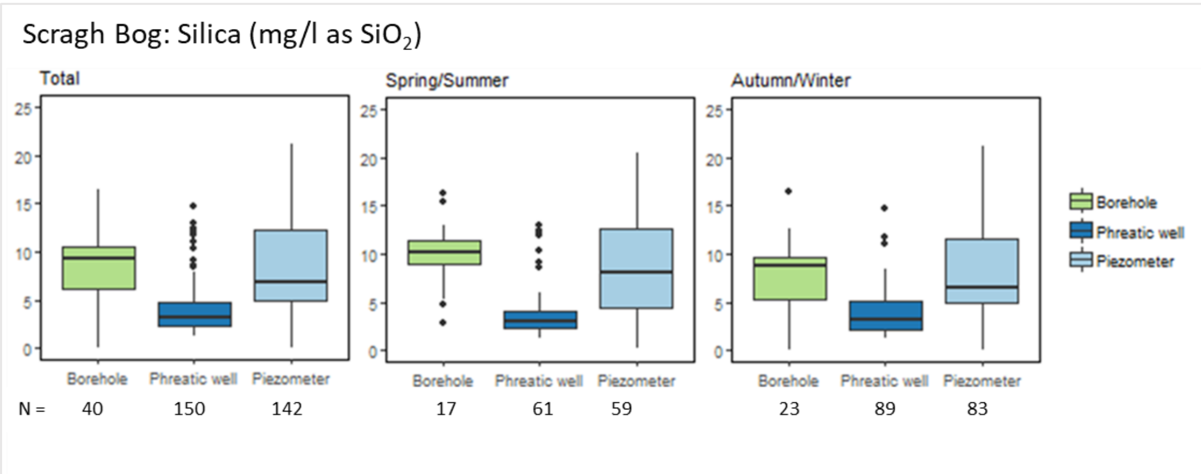


Figure 8.26. Silica in mg/l as SiO^2 sampled from phreatic wells and piezometers inside and boreholes outside the fen in Scragh Bog.

Figure 8.28 shows that the high sulphate concentrations in boreholes (median of 27.7 mg/l as SO_4^{2-}) are not reflective the concentrations found in the fen with a median concentration of 1.5 mg/l as SO_4^{2-} for both phreatic tubes and piezometers. The sulphate values in the fen were significantly lower with p-values of 0.00 in both phreatic tubes and piezometers.

Concentrations of sulphate also increase a lot between the Spring/Summer and the Autumn/Winter, as observed in the other sites as well, shifting from a median of 1.5 to 5.0 mg/l as SO_4^{2-} in the piezometers. The increase in the phreatic tubes was however not statistically significant with a p-value of 0.28. This increase may be due to anaerobic sulphides in the sediments becoming oxidised in higher flow conditions in the winter.

The measured values are still below the Irish groundwater threshold values for sulphate (Government of Ireland, 2010) which is 187.5 mg/l although it has to be noted that some outliers in the fen were found with quite high with concentrations up to 147.3 mg/l.

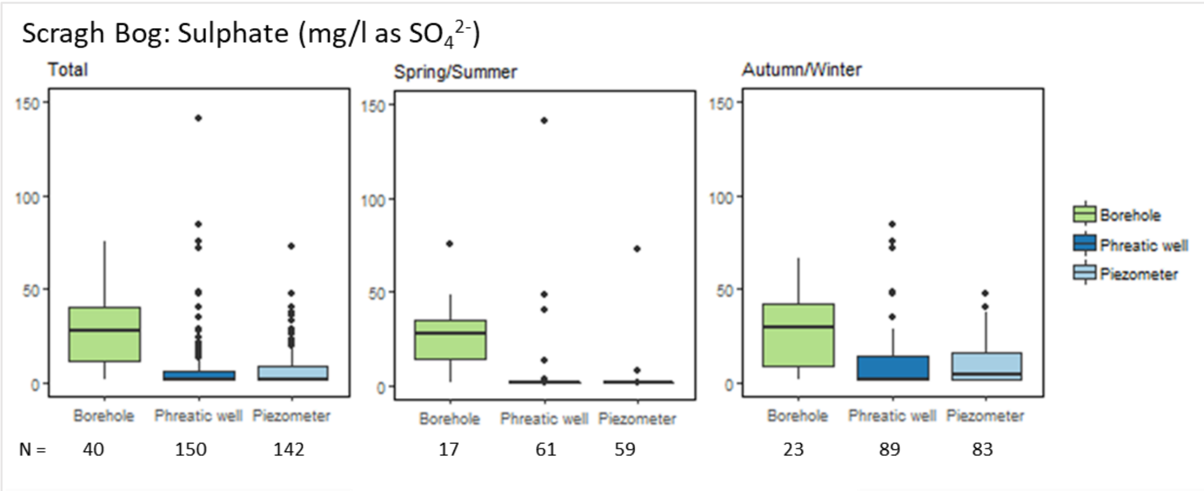


Figure 8.27. Sulphate in mg/l as SO_4^{2-} sampled from phreatic wells and piezometers inside and boreholes outside the fen in Scragh Bog.

Dissolved organic carbon was found with higher concentrations in the fen than in the surrounding aquifer (Figure 8.28). The boreholes in the fen catchment had a median of 7.2 mg/l. The median concentrations were found higher in the phreatic tubes and piezometers with a median of 19.6 mg/l and 13.1 mg/l, respectively. Two sided Welch T-test proved that the boreholes had statistically significant lower values than the phreatic tubes (p-value = 0.00) as well as the piezometers (p-value = 0.00).

These high concentrations in the fen are a reflection of the high rate of decomposition in the upper layer of the peat. Either this decomposition is also present in the substrate of the fen (as seen in the elevated concentration of the piezometers) or the DOC rich surface water is moved there by the downwards hydraulic gradients.

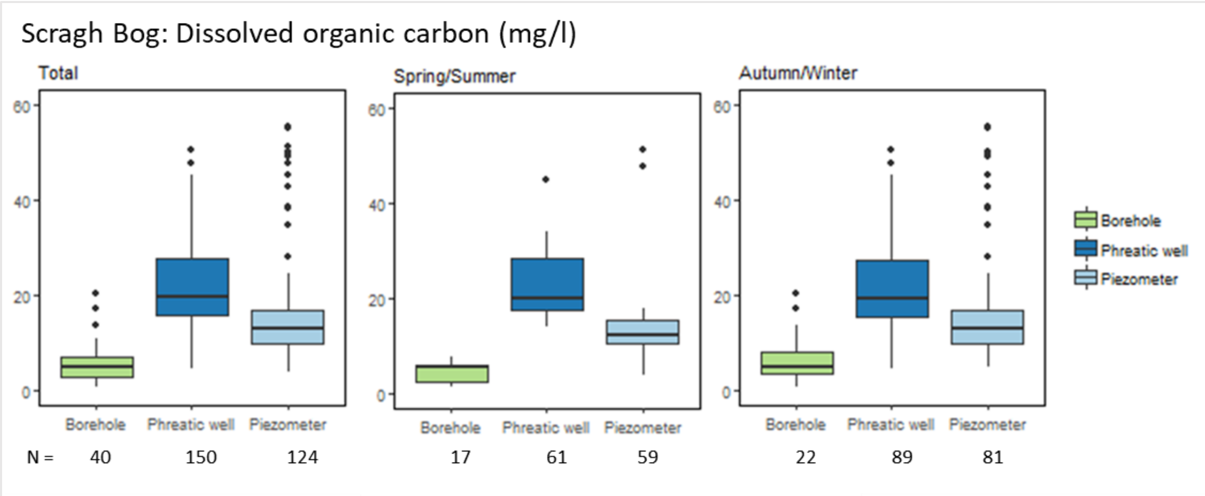


Figure 8.28. Dissolved organic carbon in mg/l sampled from phreatic wells and piezometers inside and boreholes outside the fen in Scragh Bog.

Finally, Figure 8.29 shows that no ferrous iron was detected in the boreholes around the fen with most concentrations measured below the limit of detection (0.006 mg/l as Fe^{2+}). This corroboteras the nitrogen results which suggest predominantly oxic codntions in the groundwater aquifer. Higher values found in the piezometers (median of 0.373 mg/l as Fe^{2+}) and the phreatic tubes (median of 0.278 mg/l as Fe^{2+}) suggest more reducting conditions in the fen (as also suggested by other chemical parameters such as ammonia). The concentrations in the piezometers were significantly higher with a p-values of 0.04 than the surrounding catchment. This was however not true for the concetrations in the phreatic tubes with a p-value of 0.21.

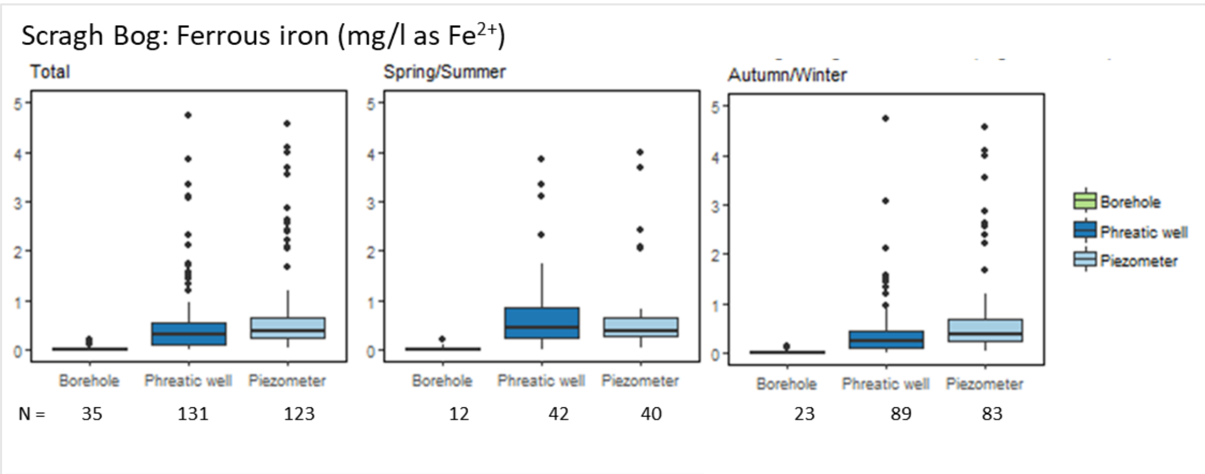


Figure 8.29. Ferrous iron in mg/l as Fe^{2+} sampled from phreatic wells and piezometers inside and boreholes outside the fen in Scragh Bog.

8.2.4. Conceptual hydrochemistry model

- Both phosphorus and nitrogen are part of the fen nutrient recycling scheme where a large amount of certain fractions (DRP and total ammonia) are found in the sediment layer of the substrate underlying the peat layers of Scragh Bog.
- The regional groundwater catchment may be responsible for some of the nutrients found in the phreatic water table coupled with the fact that the internal nutrient cycling by fen vegetation may not be sufficient enough to effectively clean the surface water layer. It therefore seems that Scragh Bog may be under pressure of nutrient pollution from the surrounding catchment.
- Reducing conditions in the piezometric as well as the phreatic layers of the fen is proven by relatively high ferrous iron, ammonia and sulphate measurements – especially in the winter.

8.3. Linkage to fen habitat

8.3.1. Hydrology and fen habitat

8.3.1.1. Boxplots water level

The boxplot in Figure 8.30 displays the overall water level distributions for different Fossitt habitats in the fen. After the piezometers in Scragh Bog were purged, the water level would sometimes take weeks or more to go back to the level it originally stood at during the time of sampling. Hence, the low piezometer waterlevels in Rich fen and flush (PF1) and Transition mire and quaking bog (PF3) are more a reflection of the weak hydraulic gradients at the screen depth of the piezometers than of the actual water levels.

Nevertheless, it seems that phreatic water levels in the PF1 habitat were overall high with a median of 0.231 mAGL. The Tall herb swamp (FS2) was flooded for most of the time with high water levels in the phreatic tubes (median of 0.214 mAGL) as well as the piezometers (median of 0.205 mAGL).

PF3 has a somewhat lower phreatic water level (median of 0.053 mAGL) with larger fluctuations which might be reflective of stronger hydraulic gradients. Indeed locations with PF3 habitats (SC2 and SC7) were displaying strong downward flows of down to -0.346 and as well as strong upwards flows up to 0.246. The Bog woodland habitat (WN7) shows similar patterns with a high phreatic water level (median of 0.076 mAGL) and large fluctuations. Again, large fluctuations in the hydraulic gradients were observed here as well (see Section 8.3.3).

The habitat Drainage ditch (FW4) has the lowest recorded water levels with a median of -0.113 mAGL and -0.127 mAGL in phreatic tubes and piezometers respectively. The 1st quartile is also quite low with medians of -0.321 mAGL and -0.356 mAGL respectively. From this it seems that this habitat is indeed accountable for draining the fen in this area.

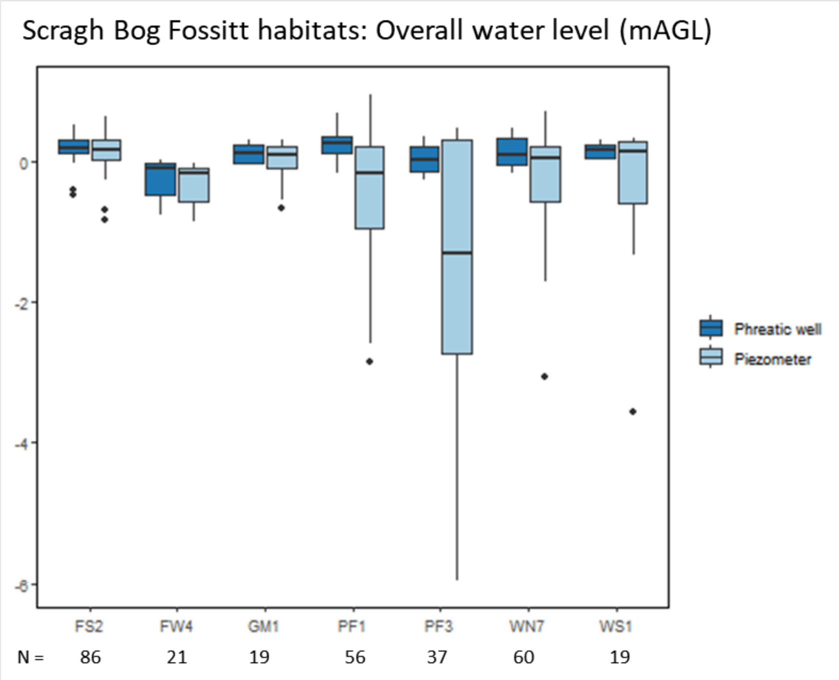


Figure 8.30. Overall water level in meters above ground level in the different habitats of Scragh Bog measured in phreatic wells and piezometers.

Figure 8.31 shows seasonal water level recorded in the Fossitt habitats. The piezometric waterlevels of FW4 were lower in the Spring/Summer (median of -0.269) than in the Autumn/Winter (median of -0.115) proving that in this habitat area there may be a net downward movement of water to the substrate below the phreatic layer in the summer. This did not seem to affect the the water levels in the phreaic tubes however.

A similar trend can be seen in WN7 where piezometric water levels were lower in the Spring/Summer (median of -0.214) than in the Autumn/Winter (median of 0.023), but again, this did not seem to cause any flucutations in the phreatic water table.

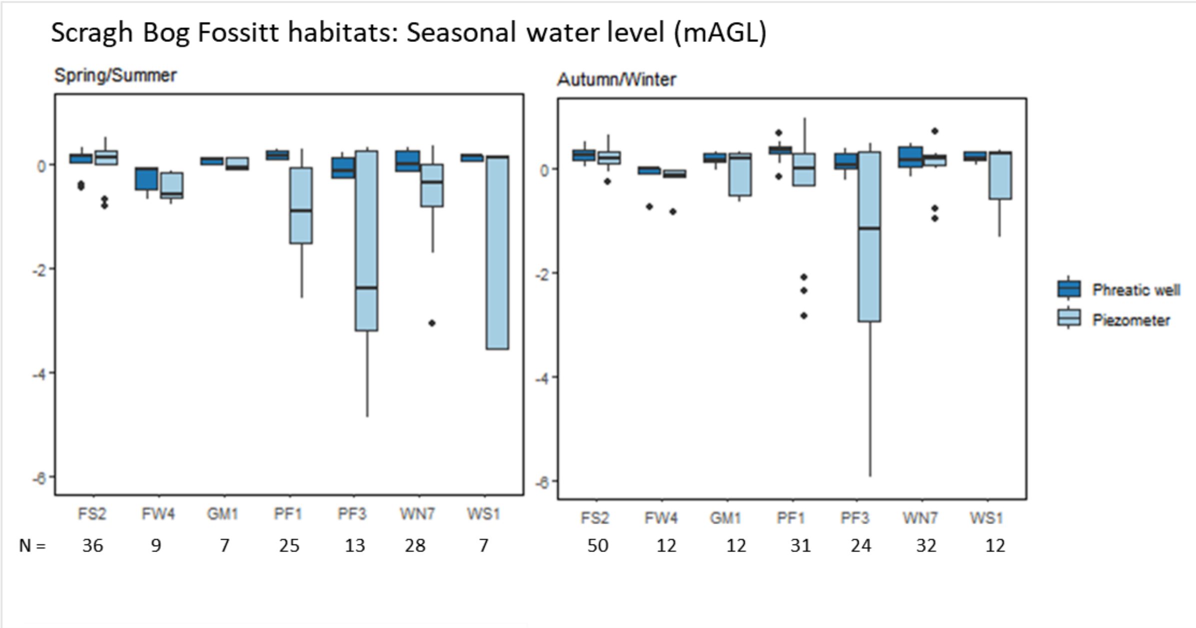


Figure 8.31. Seasonal water level in meters above ground level in the different habitats of Scragh Bog measured in phreatic wells and piezometers.

8.3.1.2. Frequency duration curves

Frequency duration curves of surface water levels made with the water level time series from data collected between October 2018 and October 2020 are presented in Figure 8.32.

Locations SC6, SC12 and SC17 all are supported by waterlevels above the surface elevation. SC15 mainly displays water table levels lower than the surface, however this difference is more related to the location of the measured waterlevels rather than the different habitats. The locations with high water levels are all located in the middle of the fen where vegetation raft is the thickest and is able to hold more water here. SC15 is located at the edge of the fen where the vegetation is rather thin and is more influenced by the water levels of the adjacent fields.

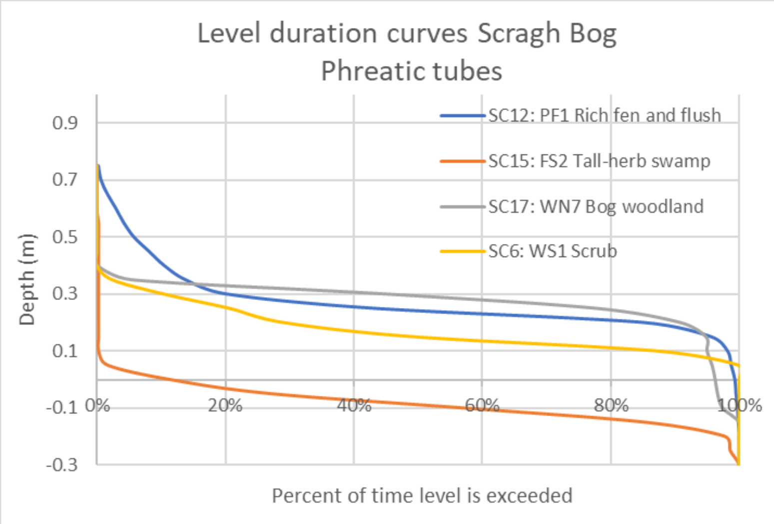


Figure 8.32. Phreatic level duration curves recorded in different habitats in Scragh Bog. The negative numbers are water levels below groundlevel.

Unfortunately the hydraulic gradient of SC17 could only be calculated between October 2018 and March 2019 since the piezometric water level logger broke down (Figure 8.33). However, from the data that was collected it seems that location SC17 took quite a while to recover after summer 2018 drought with downward gradient of increasing from -0.5 to around -0.05.

Both SC6 and SC15 experience small downward gradients throughout the year, however SC15 has slightly more extreme downward gradients during the summer. It therefore seems that the edges of the fen are influenced by seasonal conditions.

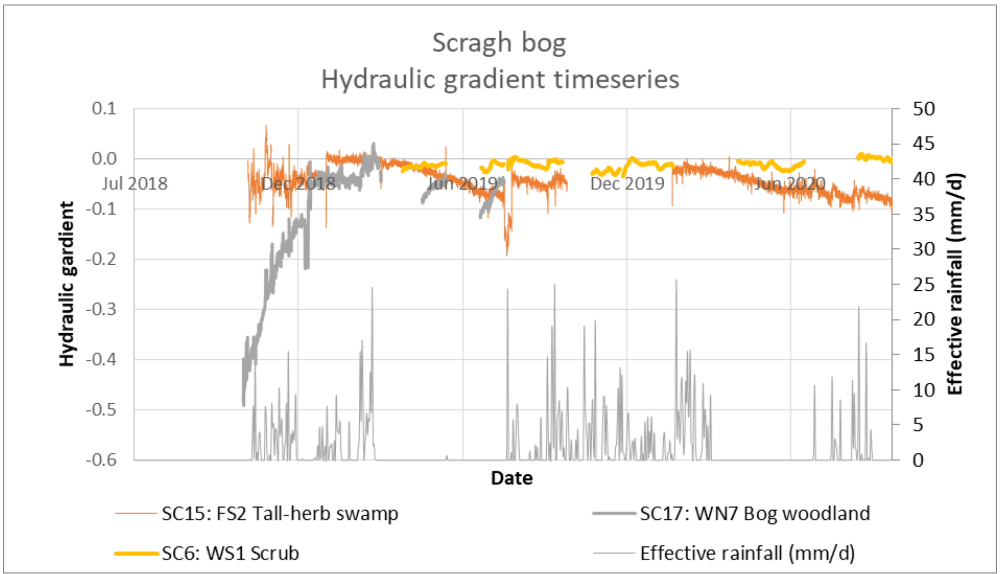


Figure 8.33. Hydraulic gradient timeseries calculated using the phreatic and piezometric water level timeseries in Scragh Bog. Effective rainfall is displayed here as well.

Again SC15 and SC6 both display minor downwards hydraulic gradients in the duration curves of Figure 8.34 which supports the argument that Scragh Bog may be acting as a net sink for incoming nutrients.

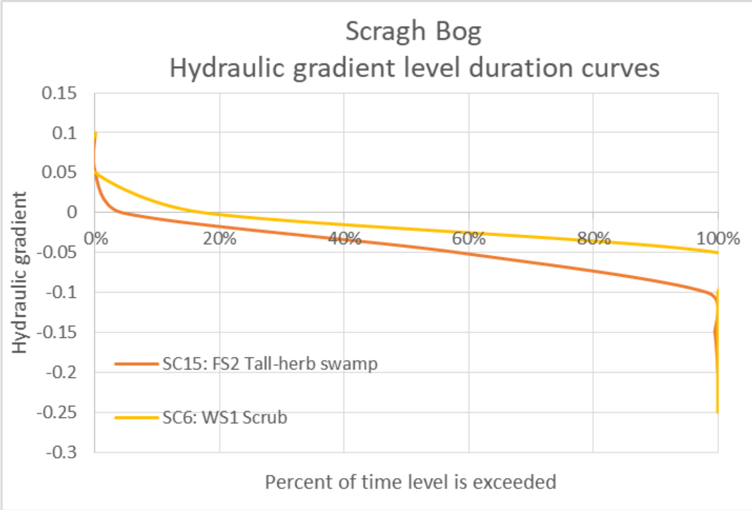


Figure 8.34. Level duration curves of the hydraulic gradients calculated from the water level time series in monitored phreatic tubes and piezometers.

8.3.2. Hydrochemistry

8.3.2.1. Nonmetric Multidimensional Scaling ordination

The NMDS plots were produced using two data sets. The IVC data set contained the recorded species percentage at each surveyed relevé. This set contained 15 relevés and 91 species. The environmental set (ENV) consisted out of vegetation type cover (%), Fossitt habitat codes and the hydrochemistry results (mg/l).

The first NMDS plot was generated with the environmental variables vegetation cover (%) and the presence of the Fossitt habitats in a biplot (Figure 8.35).

Bryrophyte and sedge cover scores were highly negatively correlated on axis NMDS2. Habitat Rich fen and flush (PF1) was also found to be highly negatively correlated on this axis which means that this habitat is associated with high abundances of bryrophyte and sedges. The species fen species *Schoenus nigrigans* is also somewhat associated with these environmental variables. However, this species also seems to be associated with a high rush cover. FW4 seems to be somewhat associated with a high grass cover and the species *Holcus lanatus* and *Salix cinerea* as they are negatively correlated on the NMDS1 axis. A high cover of canopy and litter is found in habitats that are dominated by Betula pubescens as they are all highly negatively correlated on the NMDS1 axis.

Finally, a high cover of herb and surface water is associated with the habitat Tall herb swamps (FS2) as all highly positively correlated on the NMDS2 axis. Species such as *Filipendula ulmaria* and *Equisetum fluviatile* are expected to be somewhat abundant in this environment.

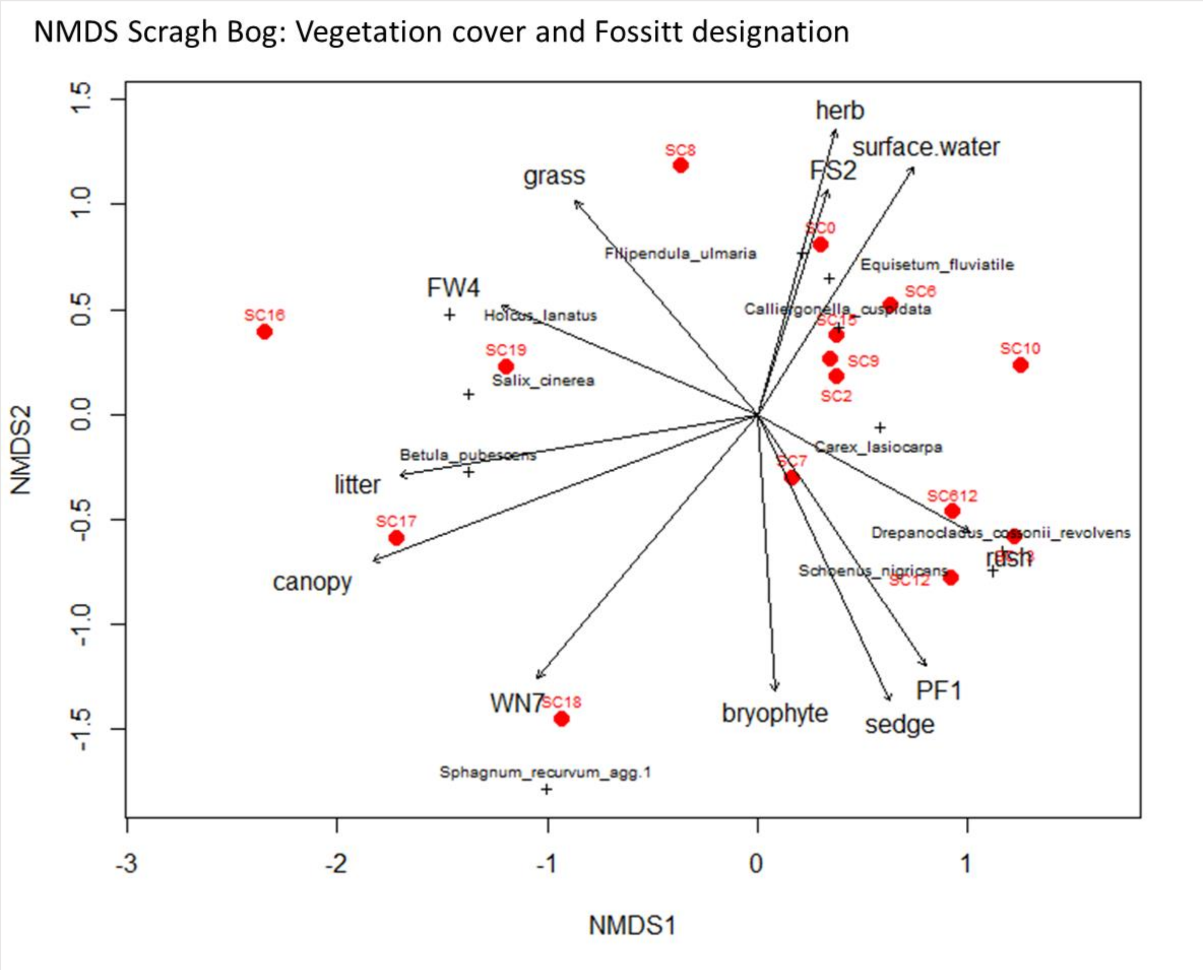


Figure 8.35. Multidimensional Scaling ordination of dimensions 1 and 2 with vegetation cover and Fossitt habitats plotted as vectors (max p-value = 0.2) in Scragh Bog. The phreatic and piezometer nest locations are shown in red and the names of the species with the highest abundances (10%) are also plotted.

The second NMDS shows the hydrochemistry results from April and June 2019 as environmental variables since this time is deemed to be the growing season of the fen. Both the April 2019 and June 2019 were generated successfully with low reported stresses of 0.105 and 0.109 respectively (Figure 8.36). From this a correlation of TP with habitat FS2 is observed in April 2019 meaning that high values of TP would be expected here. Some higher values would also be expected in FW4 according to the June 2019 plot.

Both DRP and ferrous iron seem to be correlated with habitat WN7 all showing high negative correlations on the NMDS2 axis which means high concentrations of those nutrients are expected here. Finally, the correlations on plot June 2019 imply that high ammonia concetrations are expected in habitat PF1.

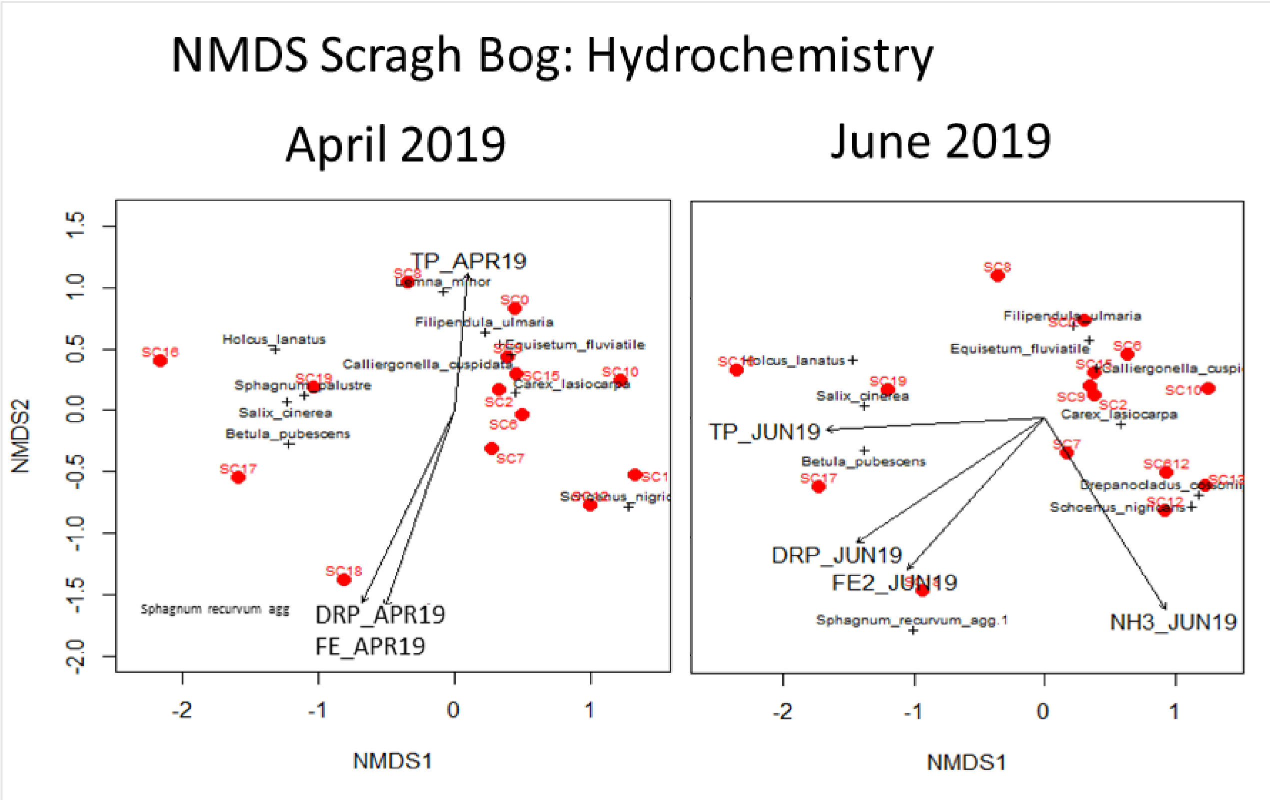


Figure 8.36. Multidimensional Scaling ordination of dimensions 1 and 2 with hydrochemistry concentrations plotted as vectors (max p-value = 0.2) in Scragh Bog. The phreatic and piezometer nest locations are shown in red and the names of the species with the highest abundances (10%) are also plotted.

8.3.2.2. Boxplots hydrochemistry

According to NMDS correlations higher concentrations of DRP were expected in WN7 habitats. However, even though the DRP concentrations in the phreatic tubes of WN7 are somewhat elevated, much higher concentrations are found in phreatic water tables of habitats such as FS2, FW4, GM1 and WS1 (Figure 8.37). In particular, FW4 and GM1 experience much higher concentrations of DRP in the Spring/Summer. Furthermore, in general, higher values were found in the piezometers of PF1, PF3 WN7 and WS1 which implies that the internal nutrient cycling between the upper peat layers and the underlying sediments is occuring within these habitats.

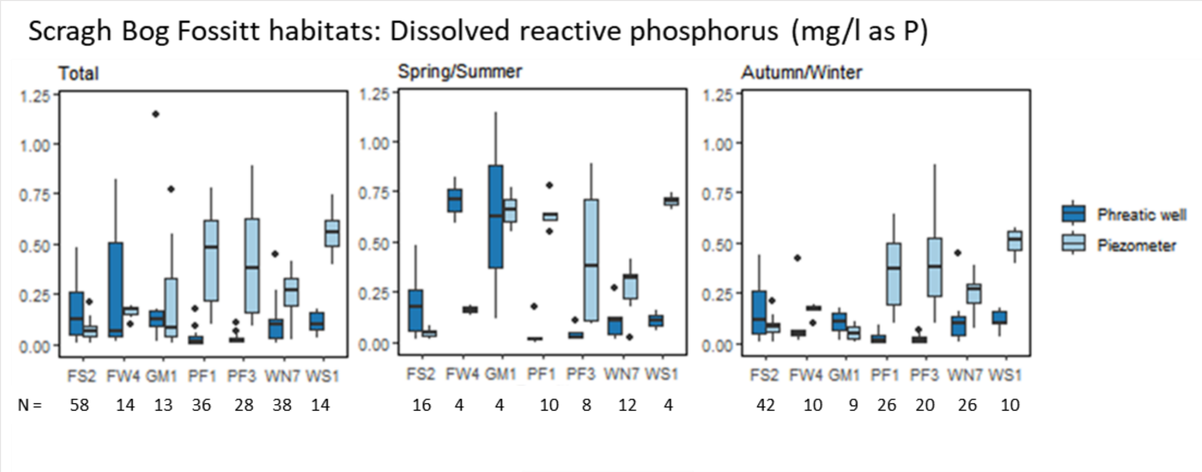


Figure 8.37. Dissolved reactive phosphorus (mg/l as P) in the different habitats of Scragh Bog sampled from phreatic wells and piezometers.

According to NMDS correlations higher concentrations of TP were expected in FW2 and FW4 habitats. While it is true that elevated concentrations are found in these habitats, GM1 display also displays high concentrations of TP in the phreatic water table (Figure 8.38) indicating that a large portion of phosphorus is being converted into organic P by microbial activity.
Furthermore, high concentrations are found in the piezometers of PF1, PF3 WN7 and WS1 which corroborates with the high DRP concentrations in Figure 8.37. This again implies that the nutrient cycling is occurring within these habitats.

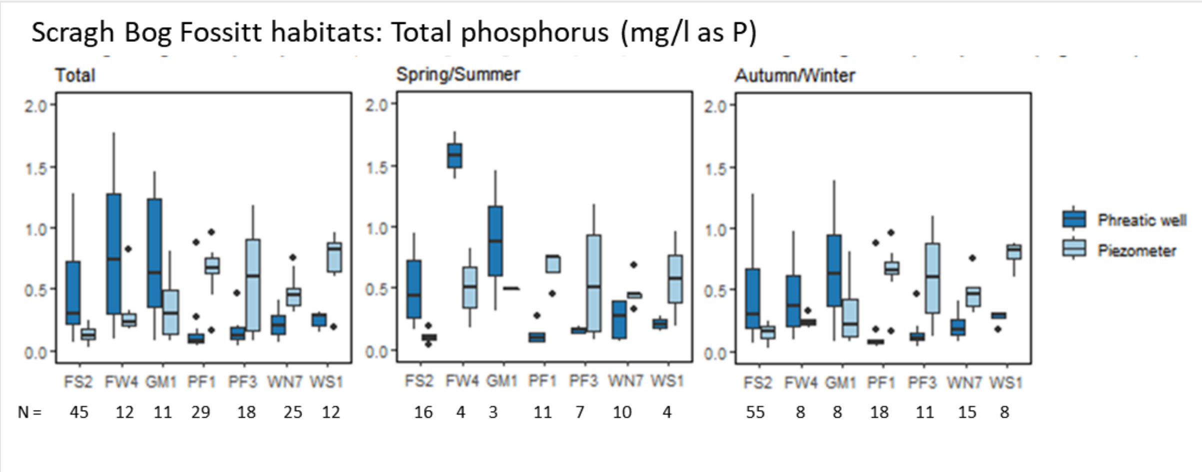


Figure 8.38. Total phosphorus (mg/l as P) in the different habitats of Scragh Bog sampled from phreatic wells and piezometers.

According to NMDS correlations higher concentrations of total ammonia were expected in PF1 habitats. Indeed from the boxplots in Figure 8.39 it seems that the concentrations in the phreatic water tables are elevated, especially in the Spring/Summer. Similar high concentrations are also found in habitats WN7 as WS1. The familiar trend of high concentrations found in the piezometers of PF1, PF3 WN7 and WS1 is also true for total ammonia as was seen with concentrations for DRP and TP.

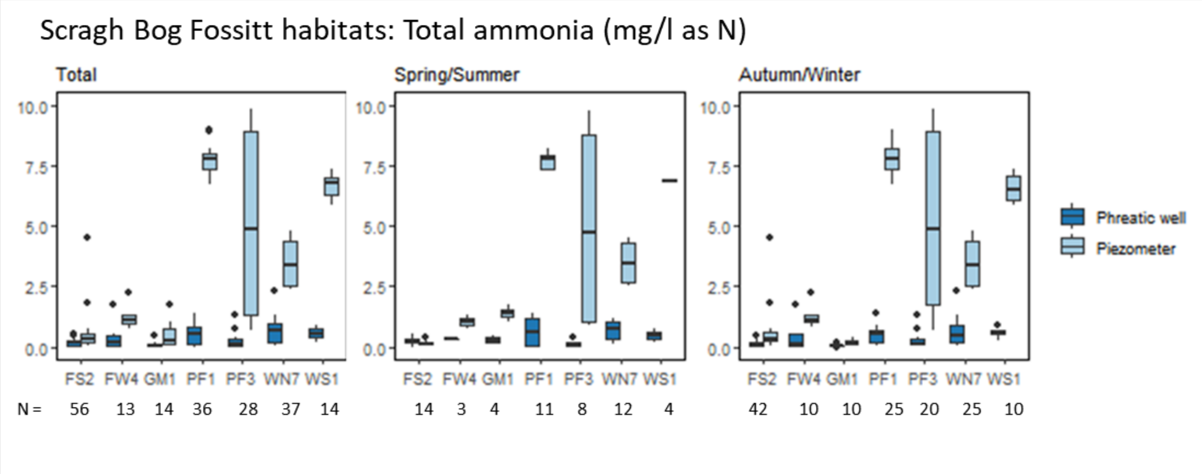


Figure 8.39. Total ammonia (mg/l as N) in the different habitats of Scragh Bog sampled from phreatic wells and piezometers.

According to NMDS correlations higher concentrations of ferrous iron were expected in WN7 habitats. This, however, does not appear to be borne out by the sampling results according to Figure 8.40, where high concentrations are mainly found in habitat FS2.

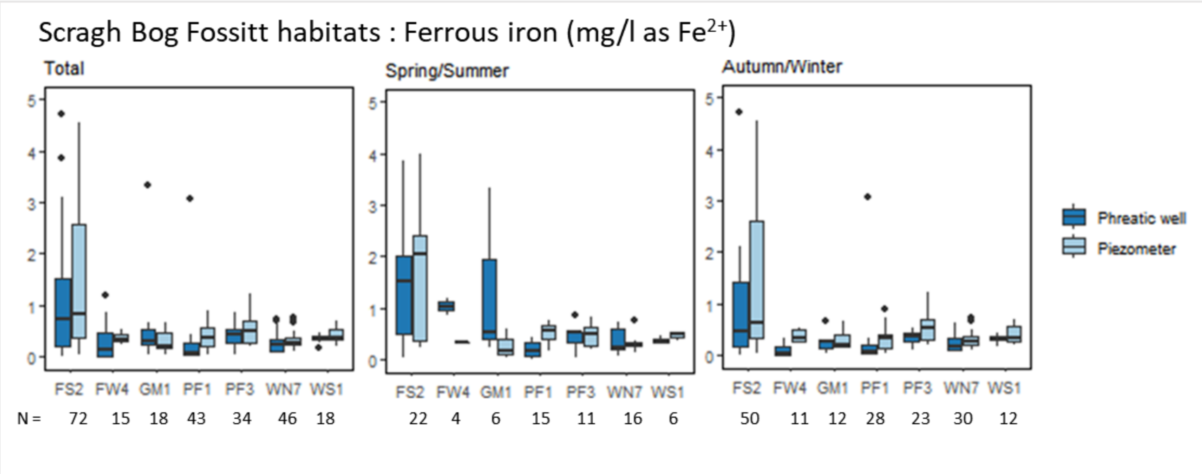


Figure 8.40. Ferrous iron in mg/l as Fe^{2+} in the different habitats of Scragh Bog sampled from phreatic wells and piezometers.

8.3.3. Mean seasonal hydraulic gradients and hydrochemistry

8.3.3.1. Total dissolved phosphorus

Figures 8.41 and 8.42 displays Scragh Bog transect 2 with data collected during August 2019 and February respectively. These results seem to show that Scragh Bog is fed mainly by discrete point springs. An indication of such spring can be seen at the edge of the fen (Duffy low) where a downward hydraulic gradient is measured in the adjacent hill. This water then flows through the ground into the fen directly into the phreatic layers which can be seen in the upwards gradients at SC15.

The different layers of the fen also experience internal nutrient cycling based on upwards and downward fluctuations which again seem to be caused by seasonal water input fluctuations. These downwards gradients cause the DRP in substrate to be high throughout the year (locations SC12 and SC13). However, the DRP experiences fluctuations in the upper peat layers of the fen especially near the edges of the fen. It seems here that the fen receives high concentrations from overland flows over and through the substrate of the adjacent hills in August 2019. These high concentrations are then diluted with high proportions of rainfall (and rainfall runoff) during the winter as seen in the February 2020 plot. High concentrations in the deeper sediment layers are further evidence of nutrient cycling and may prove that Scragh Bog is acting as a net nutrient sink.

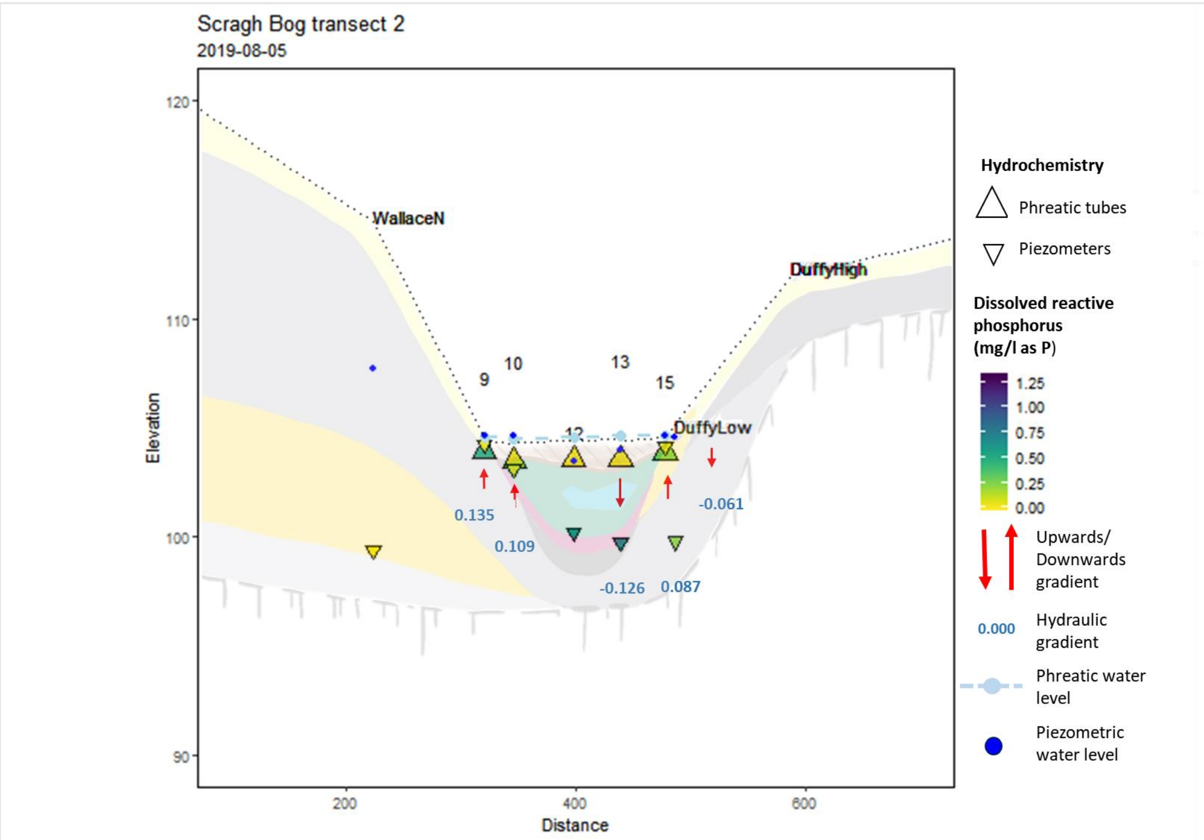


Figure 8.41. Hydrology and dissolved reactive phosphorus (mg/l as P) of Scragh Bog transect 2 in August 2019. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

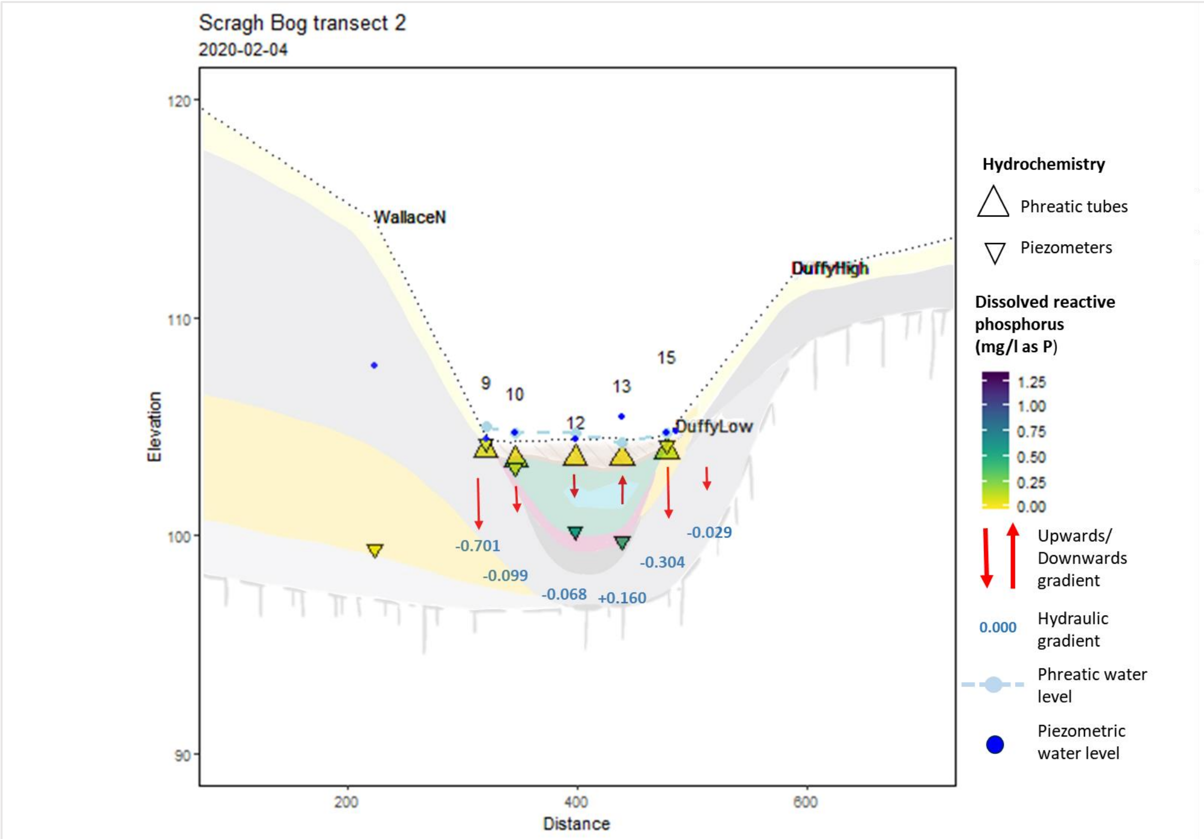


Figure 8.42. Hydrology and dissolved reactive phosphorus (mg/l as P) of Scragh Bog transect 2 in February 2020. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

A longitudinal transect of Scragh Bog is presented for data collected during August and February 2020 in Figures 8.43 and 8.45. Again, the fen experiences internal nutrient cycling based on upwards and downward fluctuations which change seasonally and generally igh concetrations are found in the deeper sediment layers. From the winter transect (Figure 8.43) it seems that the upwards hydraulic gradients at SC19 caused the high concentrations of DRP in the underlying sediments to also be dispersed in the phreatic layer. Elevated concetrations are further found in SC16. It is possible that this area received high concentrations from overland flows or discrete springs.

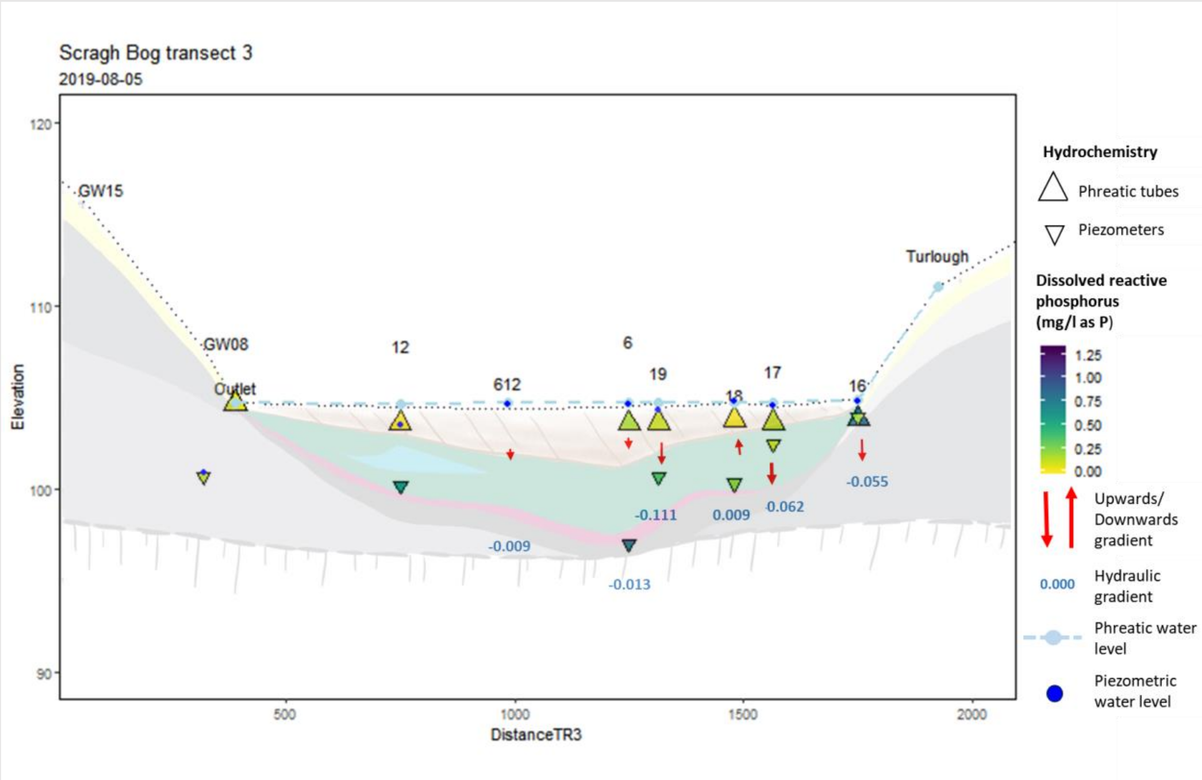


Figure 8.43. Hydrology and dissolved reactive phosphorus (mg/l as P) of Scragh Bog transect 3 in August 2019. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

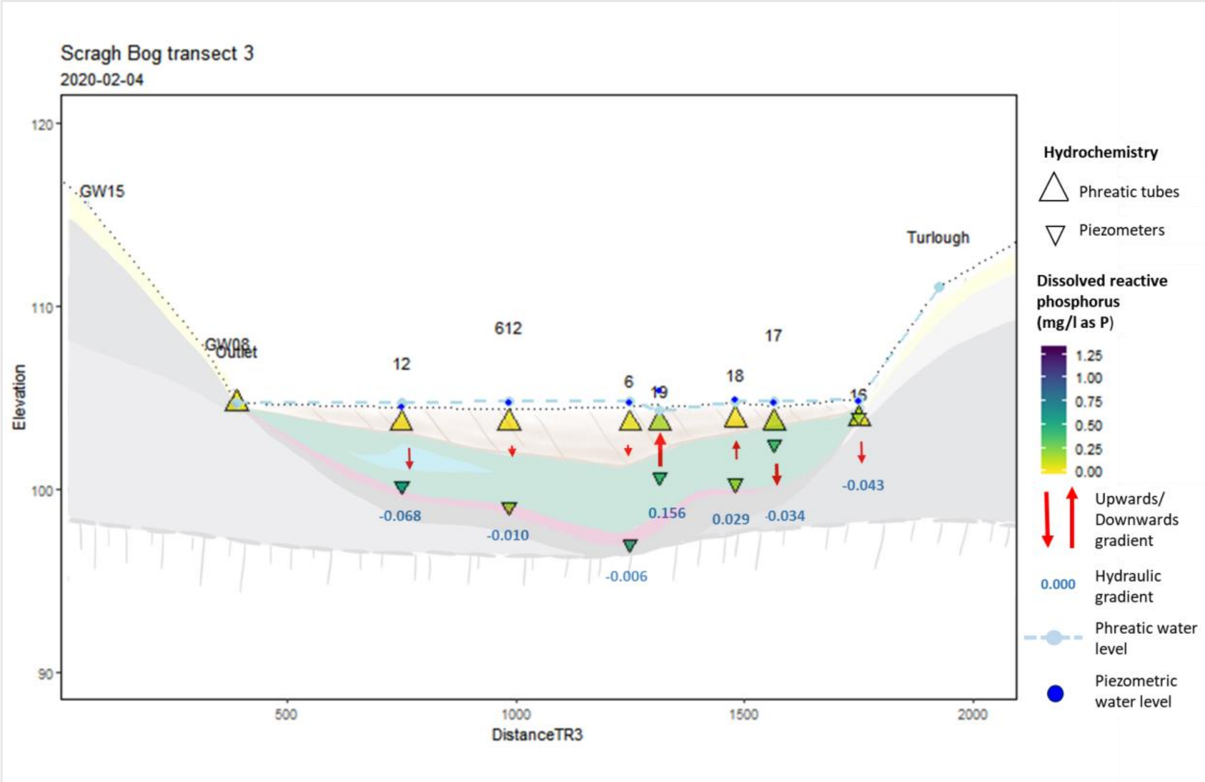


Figure 8.44. Hydrology and dissolved reactive phosphorus (mg/l as P) of Scragh Bog transect 3 in February 2020. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

8.3.3.2. Total ammonia

From both Figure 8.45 and 8.46 it is clear that elevated total ammonia concentrations are dispersed in the lower sediment layers of the peat. This if further confirmed by evidence of downward gradients to these area where the concentrations are the highest. This means these areas are again showing evidence of nutrient sinks created by the breakdown of vegetation at the surface. Lower concentrations are found in the areas of upward hydraulic gradients in Figure 8.45 which means that this is bringing some of the dispersed ammonia is brought back up. This, however, results in lower concentrations due to dilution of nutrients in the fen peat. Nutrients may further be trapped in the form of organic matter in the accumulating peat.

Interest, stronger downward gradients are found in the winter as seen in Figure 8.46 which means that the dispersion rates are probably stronger in this season. Again, the seasonal change proves that the fen cycles water and nutrients containing within in its sediments.

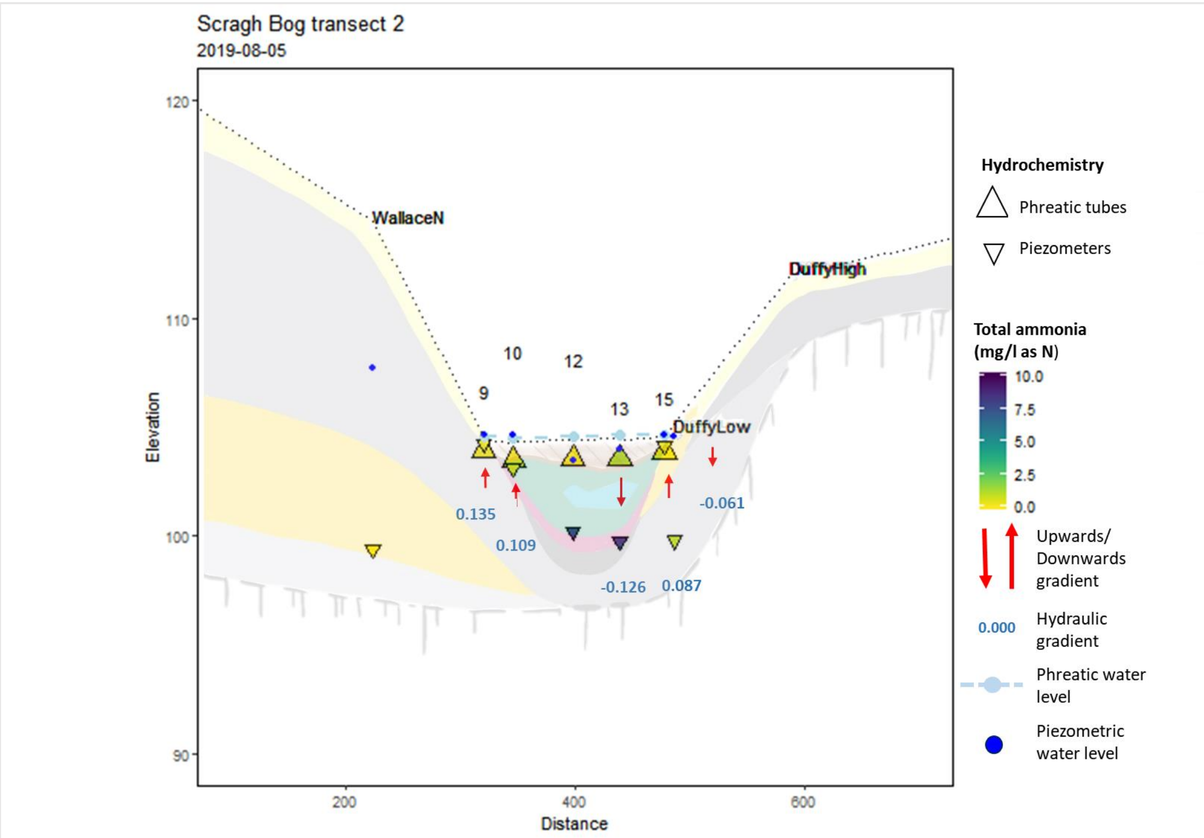


Figure 8.45. Hydrology and total ammonia of Scragh Bog transect 2 in August 2019. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

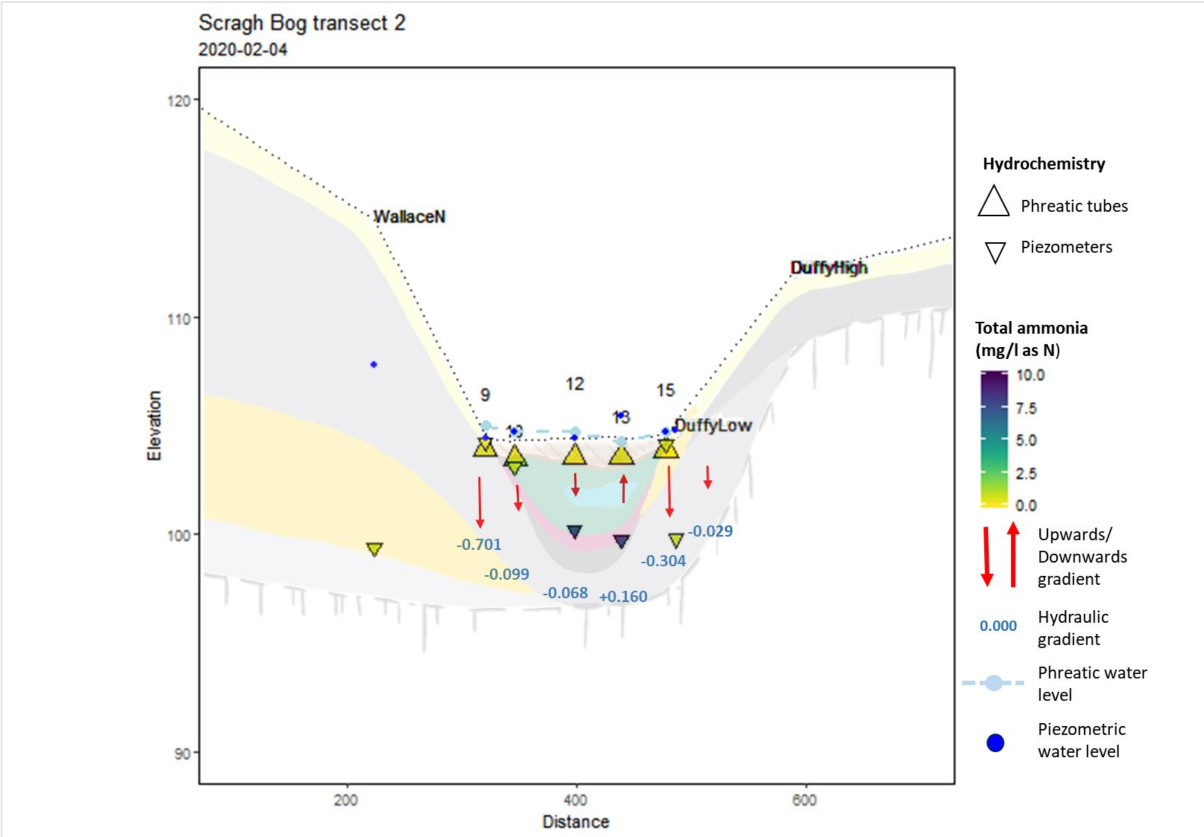


Figure 8.46. Hydrology and total ammonia of Scragh Bog transect 2 in February 2020. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

Figure 8.47 paints a similar picture for the longitudinal transect compared to crosswise transects in figures 8.45 and 8.46 where elevated concentrations in the underlying substrate. Interestingly there seem to exist a seasonal change in the summer as from Figure 8.48 for total ammonia as reported for DRP where it seems that the upwards hydraulic gradients at SC19 caused the high concentrations of total ammonia in the underlying sediments to also be dispersed in the phreatic layer. The phreatic zone of SC17 also seems to receive a higher flux of total ammonia, however this does not seem to be brought about by upwards gradients. Instead, it is possible that this area is fed by discrete spings containing elevated concetrations. The reported downward hydraulic gradient further suggest that the elevated values may be dispersed to lower sediments layers in the fen.

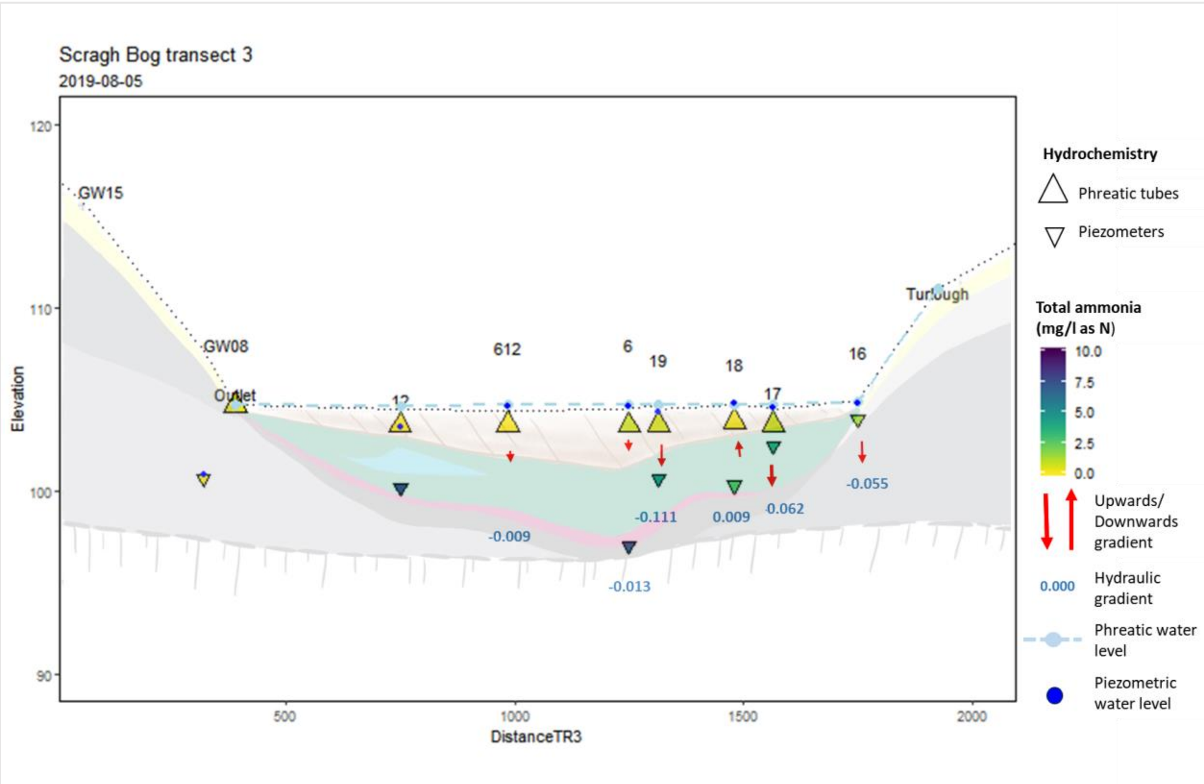


Figure 8.47. Hydrology and total ammonia of Scragh Bog transect 3 in August 2019. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

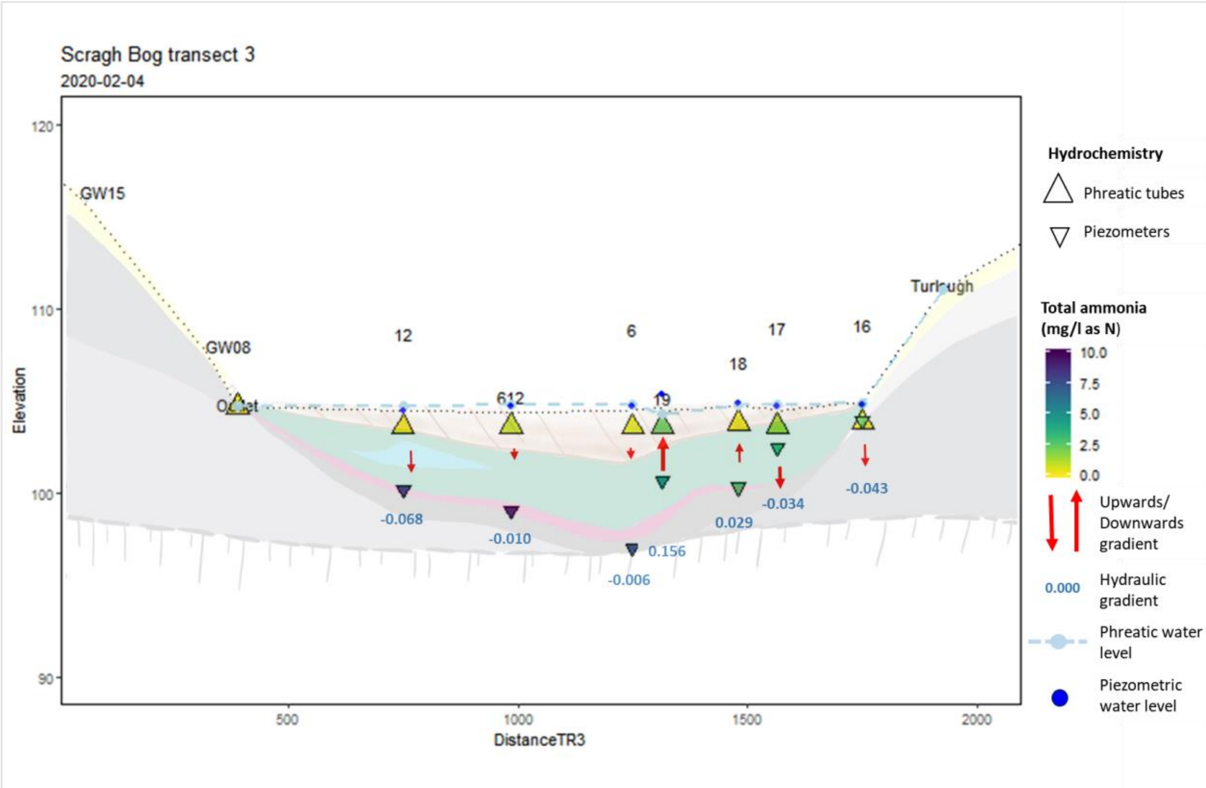


Figure 8.48. Hydrology and total ammonia of Scragh Bog transect 3 in February 2020. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

8.4. Conceptual model

8.4.1. Site summary

Scragh bog spans 0.24 km^2 and is supported by a surrounding catchment of 0.46 km^2. Th site has a wide array of different habitats ranging from fen and bog habitat to scrub and wet woodland. The fen supports a total of 0.11 km^2 of designated fen habitat (consisting of PF1 and PF3) which is 45% of the entire site. From the five assessed relevés conducted during the vegetation survey as specified in Section 4.1.3 only one failed the fen assessment criteria in Appendix D, which proves that the site supports overall good quality fen vegetation.

The fen receives both groundwater and surface water in a hydrological year at a relatively even rate throughout the year. Furthermore, extreme climatic changes such as the reported drought in 2018 seemed to affect the water levels although the fen was found resilient as the phreatic water levels did not drop below the invert of the outlet during this time. Hydraulic gradients fluctuate minimally between the phreatic zone and the deeper sediment layers and supports internal cycling of water (and dissolved chemicals).

Just as reported in Ballymore fen, Scragh Bog seems to act to treat the incoming nutrients by the uptake of fen vegetation. It furthermore seems that the regional groundwater catchment may play a more significant role in existence of the nutrients in the phreatic zone than for other sites. It seems that the water flowing into upper layers of the the fen contains high enough concentrations where the existent fen vegetation is not sufficient enough to effectively clean the surface water layer. This nutrient pollution did not seem to have any direct negative effect on the fen vegetation, however if the inflow of nutrients either persists or increases fen species might be damaged or being outcompeted by more eutrophic species. Additionally, reducing conditions were found in the different sediment layers of the fen as proven by relatively high ferrous iron, ammonia and sulphate measurements.

8.4.2. Conceptual model

A conceptual box model is displayed in Figure 8.49, showing the water balance, surface water level fluctuation and median nutrient concentrations in the fen and its catchment.

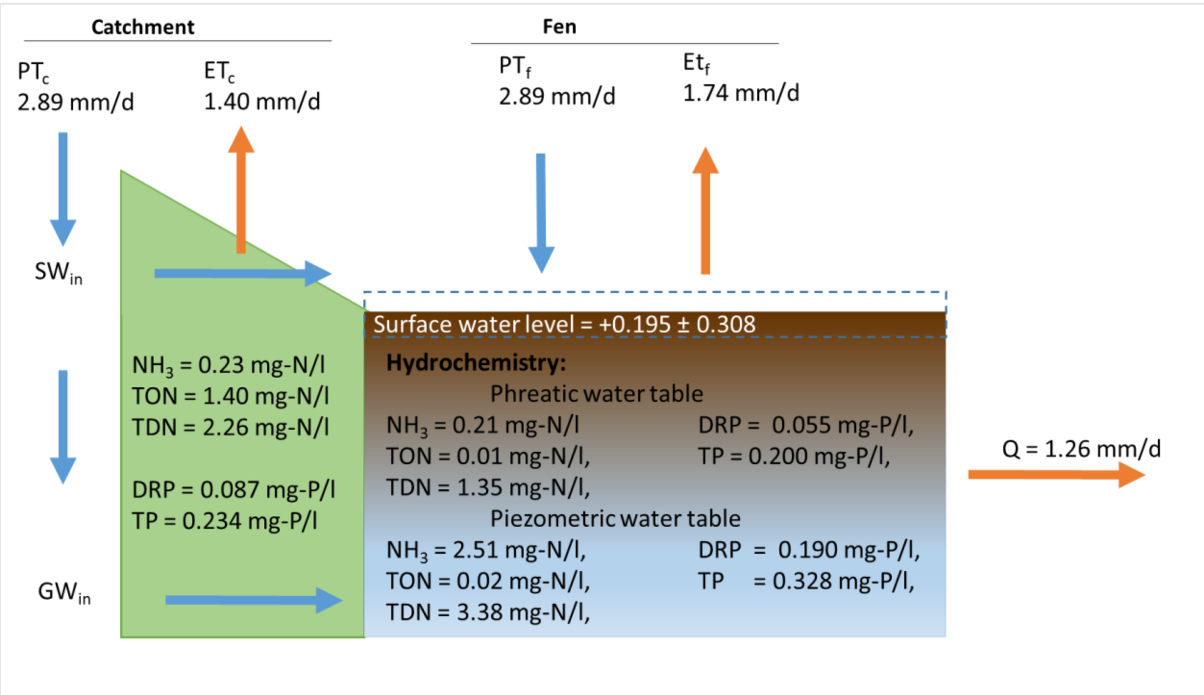


Figure 8.49. Conceptual box model of Scragh Bog displaying the water balance, surface water level fluctuation and median nutrient concentrations in the fen and its catchment.

9. Results – Tory Hill

9.1. Hydrology

9.1.1. Hydrological year water balance

The size of the topographical catchment area discharging to the flow monitoring station just downstream of Tory Hill was measured to be 19.1 km^2 (Figure 9.1). The topographical catchment area discharging to the fen was measured at 0.36 km^2 and the fen area is 0.16 km^2.

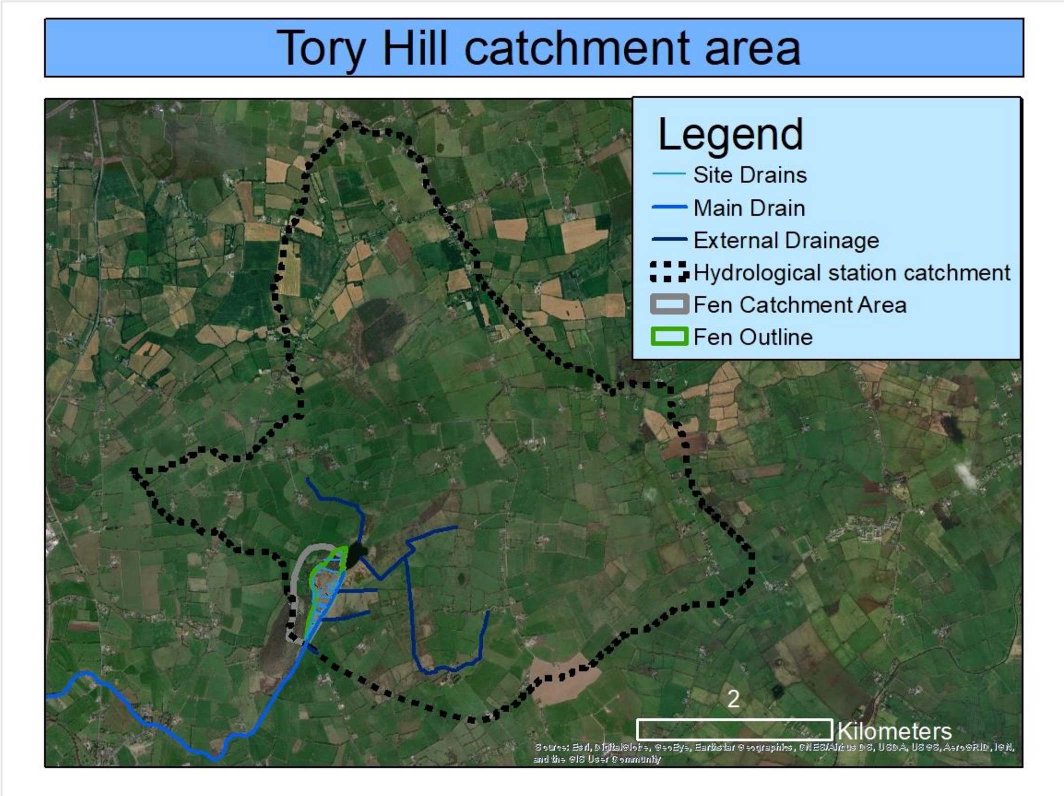


Figure 9.1 Topographic catchment area of Tory Hill (Adapted from Regan & Connaghan, 2016).

To determine the change in water volume storage at the beginning and end of each hydrological year the difference in water levels across the years needed to be calculated. From the logged water levels (Figure 9.1) it was clear, however, that the fen doesn't store water in the same manner as Ballymore, Pollardstown and Scragh Bog. The upper fen peat layer Tory Hill seems to only temporally store water in response to effective rainfall (see also Section 9.3). Where the other sites display slow seasonal water level changes, the water levels in Tory Hill at any point in time (i.e. the start or end of the hydrological year) are merely an indication of recent effective rainfall. During seasons without effective rainfall the phreatic tubes were simply found empty so no real comparison could be made since the minimum phreatic levels could not be recorded. This hydrological behaviour raises the issue of whether Tory Hill can be classified as a fen. While it has some properties of a fen properties of a fen is also has some properties that are not preferable in a healthy fen. For example Tory Hill has a layer of peat but is likely not able to 'grow' peat in the current conditions. The site also has vegetation specific to fens however in the quality assessment designed by BEC it was found all plots failed the criteria. Nevertheless in the spectrum of fen sites investigated, Tory Hill does rank as of poor quality.

The only time series that recorded adequate minimum water levels was piezometer TH8. Therefore, the water levels recorded on this time series were used since the phreatic water level fell to piezometric levels at times which can be seen in Figure 9.8. From this a significant water level increase was observed of 1.21 m across hydrological year 2018/19 compared to hydrological year 2019/20 which saw a decrease of 0.45 m. The latter was incorporated in the water balance of 2019/20 (Figure 9.2). Unfortunately, the data logger for the outlet in Tory Hill malfunctioned across a large part of hydrological year 2018/19. It was therefore impossible to calculate the water balance for that year.

In order to get an even hydrological water balance the catchment was adjusted to 13.35 km^2. The extent of this catchment also includes the lake in the north of Tory Hill. This means that any water level fluctuations here that could influence the fen are also included in the water balance. However, the calculated catchment area of the site is about 30.1% smaller than the topographical catchment area of 19.1 km^2 which is quite a large difference.

By estimation, from the recorded rainfall between October 1th 2019 and September 30th 2020 approximately 0.7% was lost from the fen and 47.3% was lost from the surrounding catchment as evapotranspiration. Furthermore, 52.4 % was lost as discharge via Tory Hill's outlet (shown in Figure 9.5). It has to be noted that water balance is much different from the other fens as the discharge in the outlet is runoff from the whole catchment rather than runoff solely from the fen. Figure 9.1 show the fen catchment are in comparison to catchment that feeds the outlet (measured at the hydrological station. Therefore conclusion made based on the water balance should be in regard to the whole catchment rather than the fen itself.

Table 9.1. Water balance of a hydrological year 2019/20 in Tory Hill.

01-10-2018 to 30-09-2019 Fen water level change: - 0.45 m			
	Total (m^3)	**Flux (mm/d)**	**Fraction of rainfall**
Rainfall on fen	196888	3.35	
Rainfall on catchment	16188902	3.35	
Evapotranspiration from fen	113128	1.93	0.7%
Evapotranspiration from catchment	7751521	1.61	47.3%
Runoff from catchment	8589776	1.76	52.4%
Change in fen storage	-72185	-1.26	-0.4%
Error in water balance	**-3549**	**0.00**	**0.0%**

Figure 9.2 gives a more schematic overview of the water balance of 2019/20. This shows more clearly that Tory Hill is supported by a large catchment as the fen receives very large amounts of rain relative to the discharge. This relative difference is much greater here than in Ballymore and Scragh Bog. This is important since it means that any hydrological changes to the catchment seemingly further away from Tory Hill (such as water abstraction or drainage) may still have an impact on the water levels in the fen.

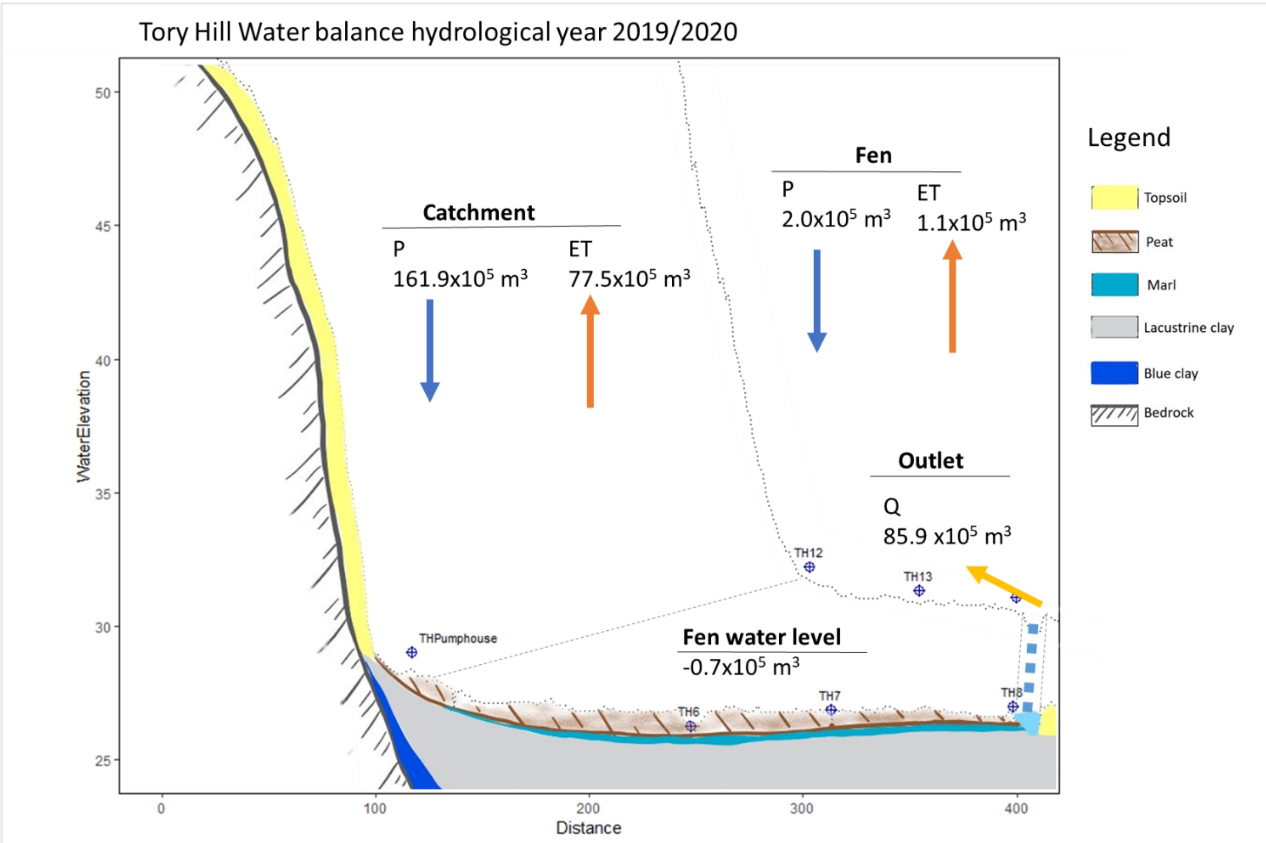


Figure 9.2. Water balance model of the hydrological year 2019-2020 in Tory Hill

9.1.2. Seasonal water balance

9.1.2.1. Hydrological year 2019-2020

As seen in Table 9.2, according to the water balance it seems the catchment feeding the outlet lost 1392807 m^3 during the winter and gained 1409148 m^3 during the summer of the hydrological year 2019/20 from the groundwater aquifer. Instrument failure prevented determination of the exact proportions coming from surface and groundwater but an approximate assessment was made.

However, this seasonal change is more likely to be caused by the delay of rainwater travelling through the soil of surrounding catchment into the fen. A portion of rain water that falls onto the catchment may take a few months before it reaches the outlet which is why the outlet seems to receive such a relatively high percentage of water from the catchment during the summer. This is clearly evident in the higher proportion of discharge to the outlet in the winter (0.78 %).

Table 9.2. Seasonal water balances of hydrological year 2019/20 in Tory Hill

01-10-2019 to 31-03-2020 (Winter) Fen water level change: - 0.43 m			
	Total (m^3)	**Flux (mm/d)**	**Fraction of rainfall**
Rainfall on fen	118800	4.05	
Rainfall on catchment	9768210	4.05	
Evapotranspiration from fen	26314	0.90	0.3%
Evapotranspiration from catchment	1803017	0.75	18.2%
Runoff from catchment	6732245	2.76	68.1%
Change in fen storage	-67373	-2.32	-0.7%
Error in water balance	**-1392807**	**-0.58**	**-14.1%**
01-04-2020 to 30-09-2020 (Summer) Fen water level change: - 0.03 m			
	Total (m^3)	**Flux (mm/d)**	**Fraction of rainfall**
Rainfall on fen	78088	2.66	
Rainfall on catchment	6420692	2.66	
Evapotranspiration from fen	86814	2.96	1.3%
Evapotranspiration from catchment	5948505	2.46	91.5%
Runoff from catchment	1857531	0.76	28.6%
Change in fen storage	15079	-0.16	0.2%
Error in water balance	**1409148**	**0.58**	**+21.7%**

9.1.3. Runoff

A discharge time-series of the outlet is presented in Figure 9.3. The total daily rainfall amounts and the calculated effective rainfall in mm/d are also plotted. With discharges varying between 91 and 2961 m^3/hr and an average of 750 m^3/hr between April 10^{th} 2019 and September 30^{th} 2020 the outlet discharges water at a much greater rate than Ballymore and Scragh Bog. This dynamic directly controlled by the much larger catchment area of which great volumes of water end up bypassing the fen in the deeply dug out artificial outlet along the length of the fen. This is very different scenario to the other fen sites, which all only drain from one point under natural conditions.

There are clear differences in the responses of the discharge rate to rainfall (both total and effective) the highest response showing flows up to 2961 m^3/hr during periods of high effective rainfall. During the summer months no effective rainfall is recorded, yet the outlet still shows discharge peaks up to 564 m^3/hr in response to total rainfall. This is mainly due to overland flow from and around the fen in order to quickly dispose of a surplus of water after a large rainfall events. When the base line of 91 m^3/hr in the graph is approached during dry months, it is assumed that the outlet is mainly discharge groundwater (i.e. baseflow).

Furthermore, even during the wet months of 2019 and 2020 the outlet shows a fast rate of hydrograph recession which suggest that surface water entering the fen runs off very quickly. Tory Hill does therefore seem to have a very small storage and rainfall attenuation capacity.

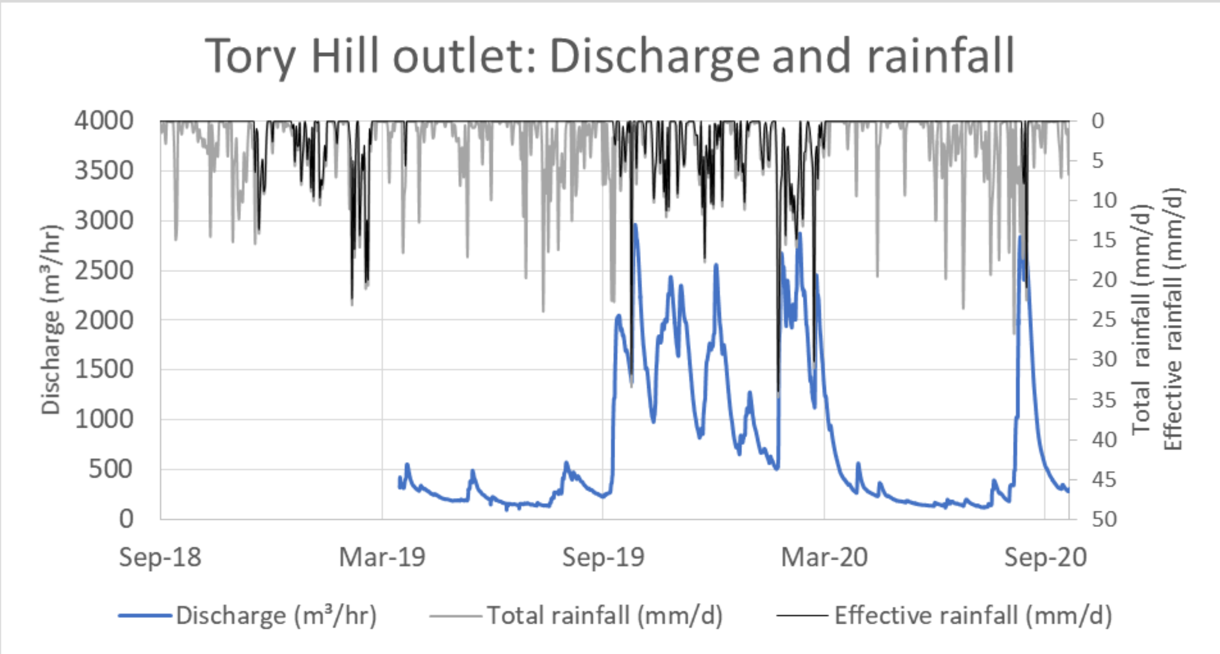


Figure 9.3. Tory Hill outlet hydrograph and total/effective rainfall between April 10th 2019 and September 30th 2020.

A time-series of the total evapotranspiration against the discharge is displayed in Figure 9.4. The highest numbers were recorded during the summers with a total evapotransipiration of 6.9 mm/d in 2019 and 7.4 mm/d in 2020. The amount of discharge starts to increase at the end of September in 2019 and 2020 while the evapotranspiration is around 2 mm/d and still decreasing. This trend seems to continue until there is a maximum runoff during the wet months of 2019/20. Then, when the evapotranspiration starts to increase in spring 2020, the discharge decreases again. One period that seems to countradict this general trend, however, was the high discharge rates during August 2020 when evapotrasnpiration was still at 3 mm/d. The graph shows that this was caused by the high amount of total (and effective) rainfall sustained for a longer period of time. The seasonal discharge fluctuations occur over a much smaller time window than in those observed in Scragh Bog and Ballymore.

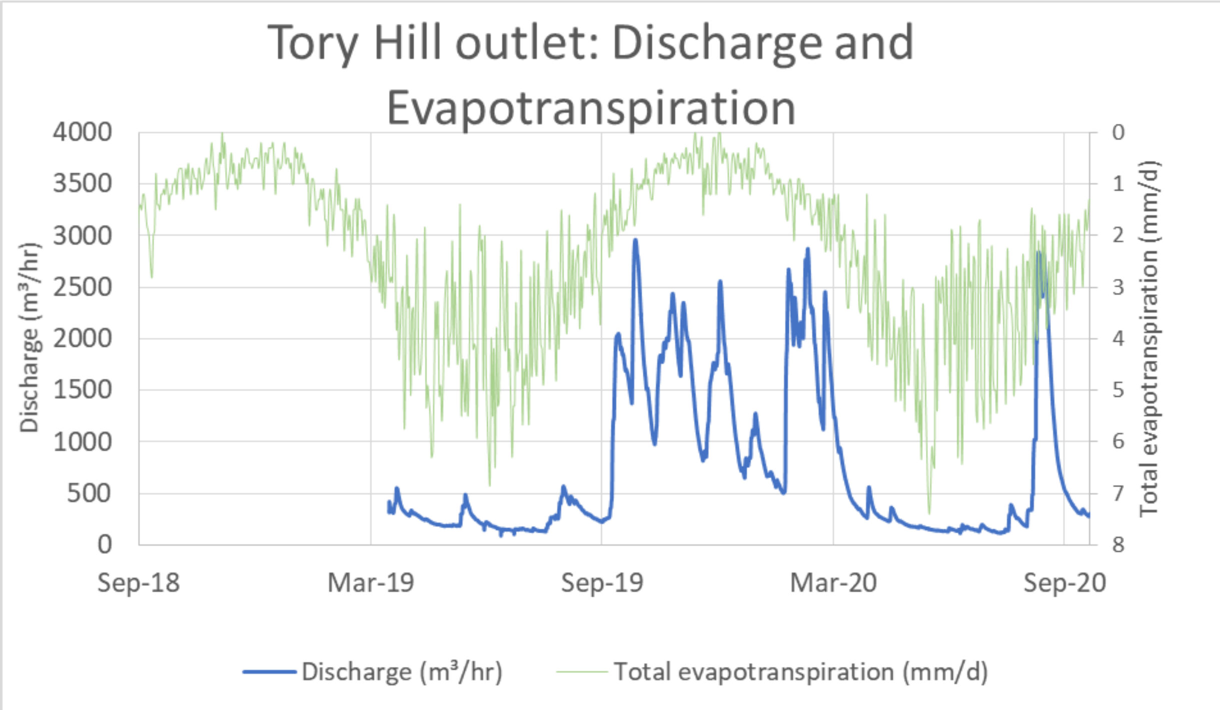


Figure 9.4. Tory Hill outlet hydrograph and actual ecapotranspiration between October 1th 2018 and September 30th 2020.

The effect of daily evapotranspiration can be found in the time-series of the water levels of the fen as shown in Figure 9.5 over the span of 10 days, as well as the temperature. Diurnal evapotranspiration fluctuations were not visible probably since the outlet had a strong draining influence on the phreatic water level: in a mere 10 days the water level dropped by 0.31 m. This a stark contrast to the phreatic fluctuations recorded in the other fens. Phreatic water levels are further discussed in Section 9.1.4.

It is clear that the outlet and/or the lake continues to drain the fen even after prolonged periods without rainfall. The water level in phreatic tube TH8 shows a delay of approximately a day in response to the rainfall event of the 21th of March. It is suggested that the fen is mainly fed by groundwater flow from the bottom of the hill in the north-west and that it took approximately a delay before this water ended up in the phreatic water table of TH8 near the outlet.

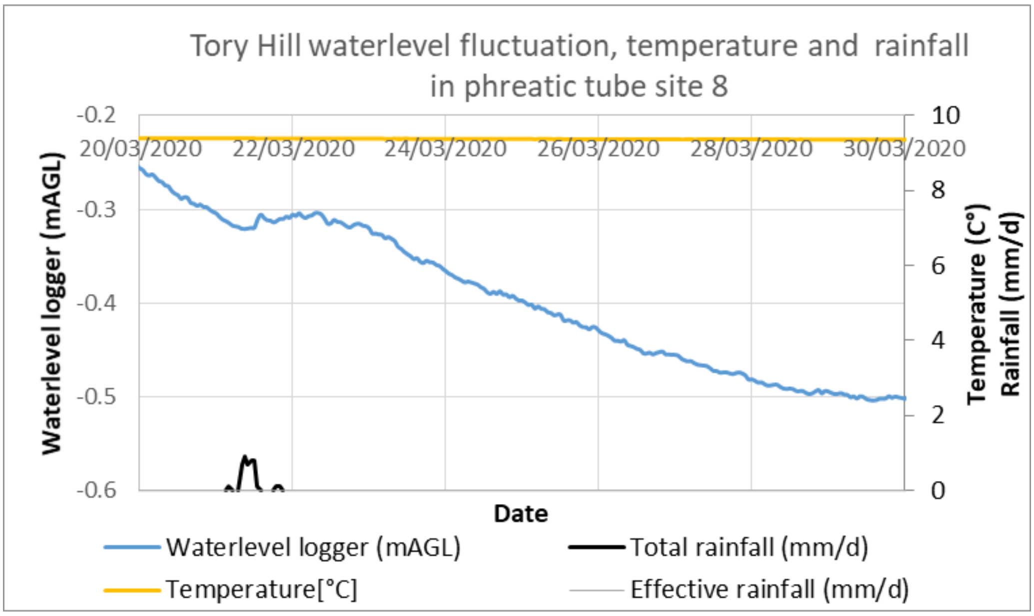


Figure 9.5. Tory Hill fen phreatic tube site 8 hydrograph and temperature between March 18th 2019 and March 28th 2019.

9.1.4. Fen piezometer and phreatic tube data

Surface water points in the fen and groundwater table points around the fen were interpolated into contour lines in order to interpret the flow in and out of the fen (Figures 9.7 and 9.8). Figures 9.9 and 9.10 show the water levels recorded in the phreatic tubes and piezometers between July 2018 and October 2020. The locations of the sites where this data was collected can be found in the Tory Hill instrumentation map of Figure 9.6.

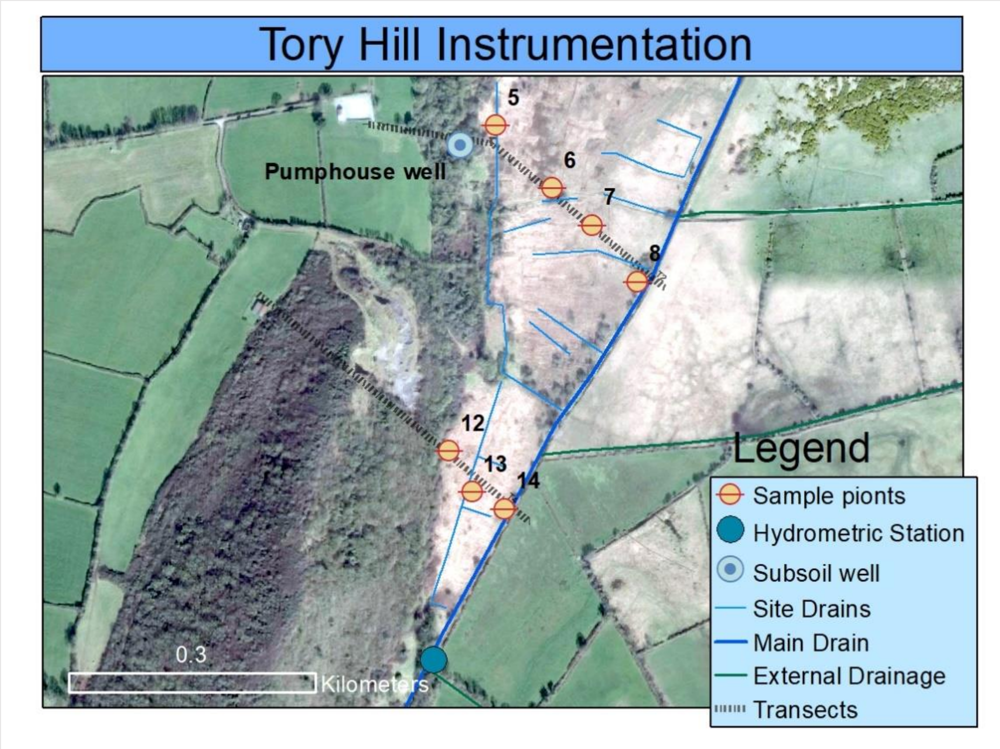


Figure 9.6. Tory Hill instrumentation map showing fen piezometer and phreatic tube locations, subsoil well locations and the main site drains.

Flow lines in Figures 9.7 and 9.8 are found moving from the hill in the west towards the outlet in the east. This show that the drain is affecting the waterlevels of the entire fen. Additionally to those, lateral flow over the surface from north to south is observed during the winter with much higher levels than compared to the summer. Possibly because the drain levels have also risen during the winter.

Contour lines that were generated right from the main drain were manually removed as it seemed that water was discharging from the fen into the fields to the west, while in reality the water level in the main drain is lower than those fields. More levels would have needed to be measured along the drain in order to showcase this effect.

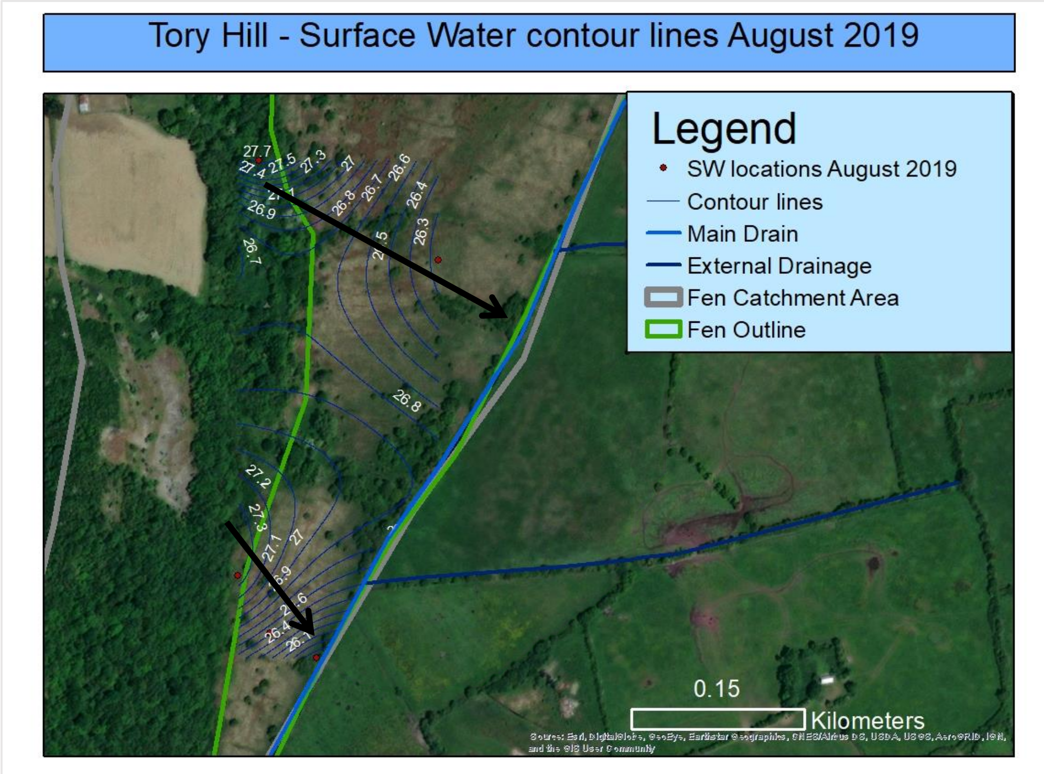


Figure 9.7. Contour lines of fen surface water and surrounding groundwater catchment interpolated using point measurements in August 2019. Flowlines are presented with black arrows.

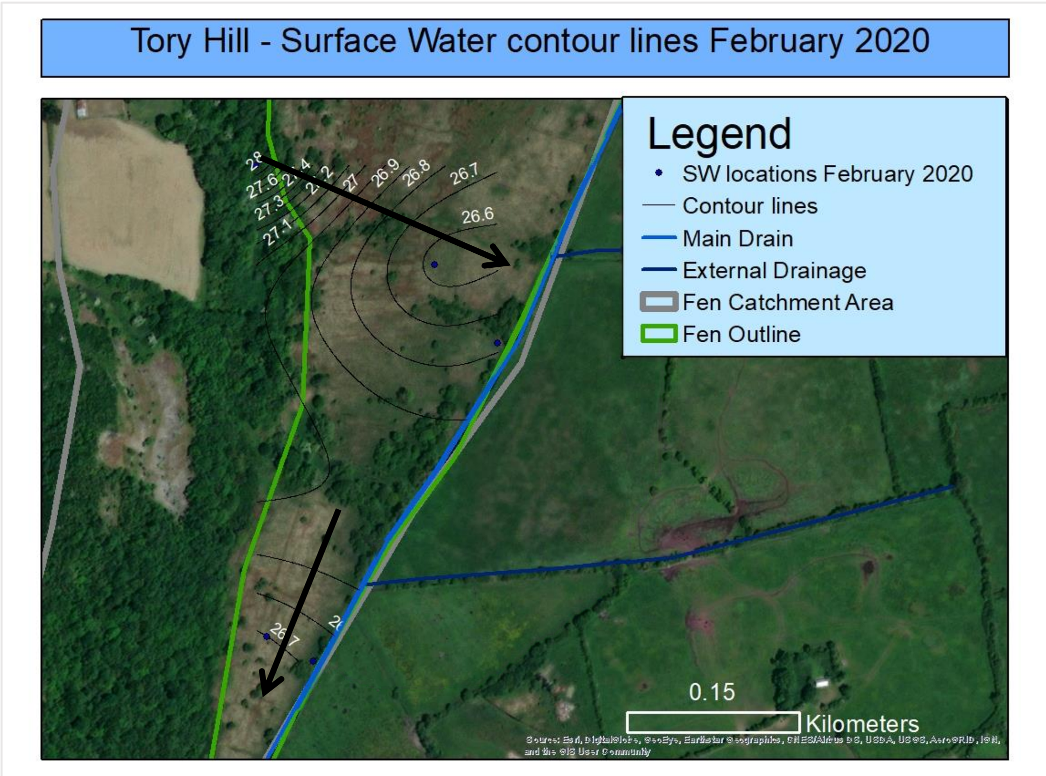


Figure 9.8. Contour lines of fen surface water and surrounding groundwater catchment interpolated using point measurements in February 2020. Flowlines are presented with black arrows.

The phreatic water levels show large fluctuations in response to effective rainfall (Figure 9.9). These fluctuations are a stark contrast to the phreatic fluctuations recorded in the other fens.

As mentioned previously, the phreatic layer only seems to store water temporarily throughout the wet months. During the summer most of the phreatic tubes stood dry - the water level time series can be observed flat lining during the dry seasons. This level is not the actual water level but rather the remainder of moisture at the bottom of the phreatic tube. It is not possible to know whether the phreatic wells dropped below the invert of the outlet since the water levels fell below the end of the phreatic tube, but the fact that water kept discharging from the fen via the outlet and presumably the lake over dry periods in the summer would suggest that the groundwater feed would maintain the water table to some head above the outlet channel.

Higher water levels are recorded in TH12 since it is located at the base of the hill where the fen receives its water from and its surface elevation (26.9 mAOD) is higher than the other locations as well. Both TH8 and TH14 have the lowest observed water levels reflecting the drawdown at these point with the water level strongly controlled by the stage of the outlet. TH6 also displays some low water levels although is due to lower ground surface elevations in contrast to the other locations (25.3 mAOD).

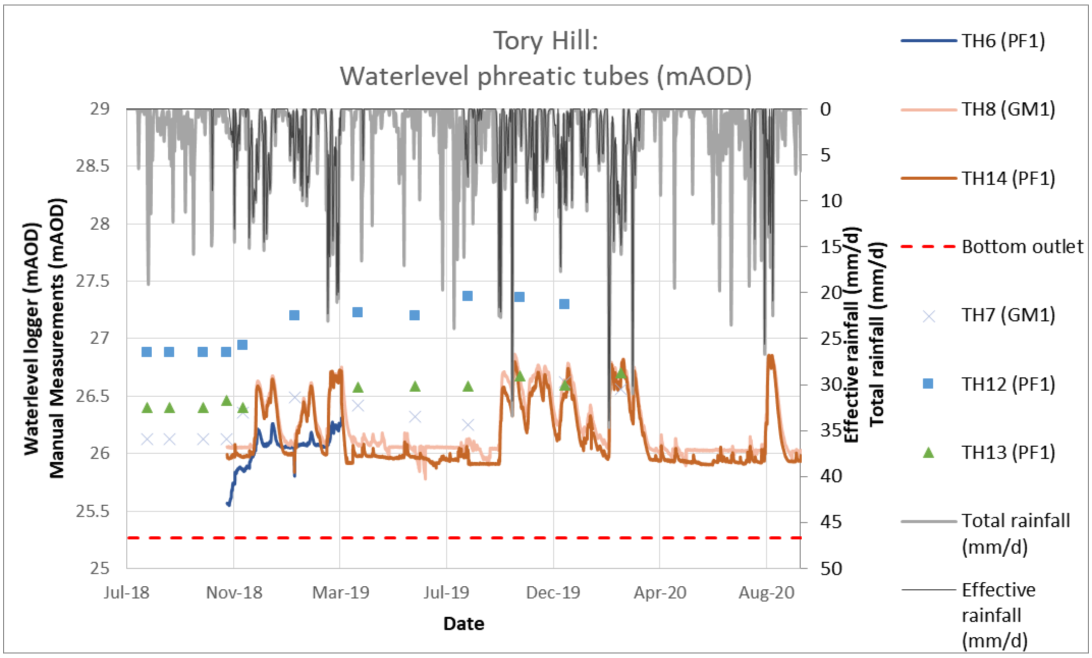


Figure 9.9. Phreatic water level hydrograph of spot measurements and water level loggers and rainfall. The height of the bottom of the outlet (measured at the flume) is presented with a red dashed line.

TH14 again shows that the piezometer was empty during the dry seasons (Figure 9.10). Only TH8 was able to record the true water levels since it had a screen depth of 2 mBGL, which reached below the invert of the outlet. From this time-series it can be observed that the piezometric water level was recorded below the invert of the outlet in the summer of 2018. This water level then recovers over the following months, however the dry seasons of 2019 and 2020 still cause the water level to drop to only 0.1 m above the bottom of the outlet. The total seasonal water fluctuations show a large 1.16 m range.

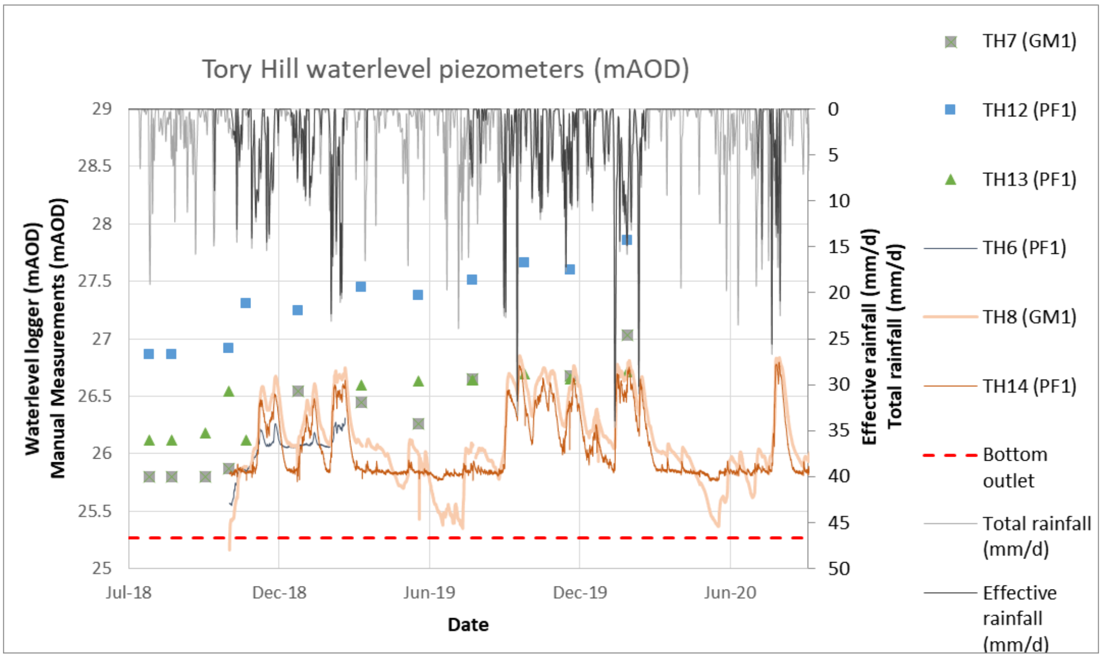


Figure 9.10. Piezometric water level hydrograph of spot measurements and water level loggers and rainfall. The height of the bottom of the outlet (measured at the flume) is presented with a red dashed line.

Figure 9.11 displays the phreatic and piezometer water level comparison of TH8 located next to the artificial outlet upstream from the hydrometric station. The piezometer has a screen depth of 2 mBGL.

The water levels are compared to the Pumphouse well located 335 m northwest from TH8 and the outlet stage.

Both the phreatic tube and the piezometer display water level fluctuations that follow the trend of the outlet stage showing that the drain has a strong control. They are recorded slightly elevated above the outlet stage since the elevation of the invert of the drain near TH8 is higher than the invert elevation recorded approximately 600 m downstream. However this slight head difference is expected in the outlet due to head losses en route.

It further seems that the water levels in the fen show some correlation with the groundwater fluctuations measured in the Pumphouse well which implies that the groundwater table is influencing the fen water levels and/or the groundwater has a input from the catchment has a rapid response to rainfall recharge (which also acts on the fen).

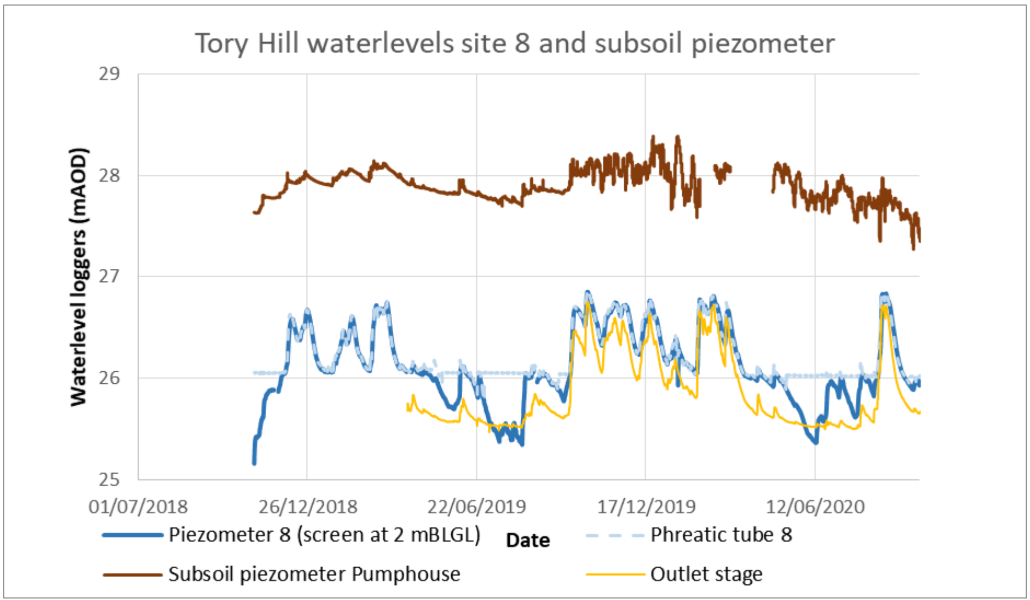


Figure 9.11. Hydrograph of phreatic and piezometric water levels at site 8 and piezometric water levels in subsoil well Pumphouse

Figure 9.12 displays the phreatic and piezometer water level comparison of TH14 located next to the artificial outlet upstream from the hydrometric station and downstream from TH8. The piezometer has a screen depth of 1 mBGL. Both the phreatic tube and the piezometer have the exact same fluctuations as the stage of the outlet meaning that TH14 is directly controlled by the outlet and that the peat apparently has no storage capabilities in this location. Again, the water levels show a correlation with the groundwater inflows from the east as the fluctuations follow the water level signal recorded in the Pumphouse well.

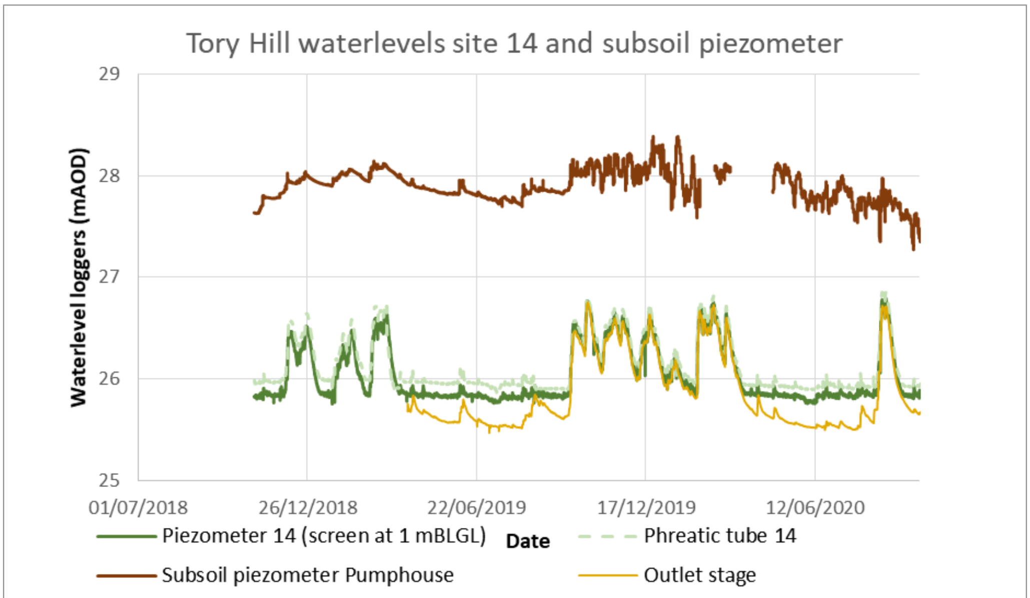


Figure 9.12. Hydrograph of phreatic and piezometric water levels at site 14 and piezometric water levels in subsoil well Pumphouse

9.1.4.1. Electrical conductivity

Figure 9.13 displays the EC results with higher EC recorded during the summer than in the winter which implies that the phreatic water table is mainly fed by groundwater during the dry season, thereby corroborating the findings from the previous water level data. TH12 displays the greatest fluctuations which implies that the fen seems to receive most of its groundwater at the bottom of the limestone hill during the summer. This water then is progressively mixed with more surface runoff during the wetter months.

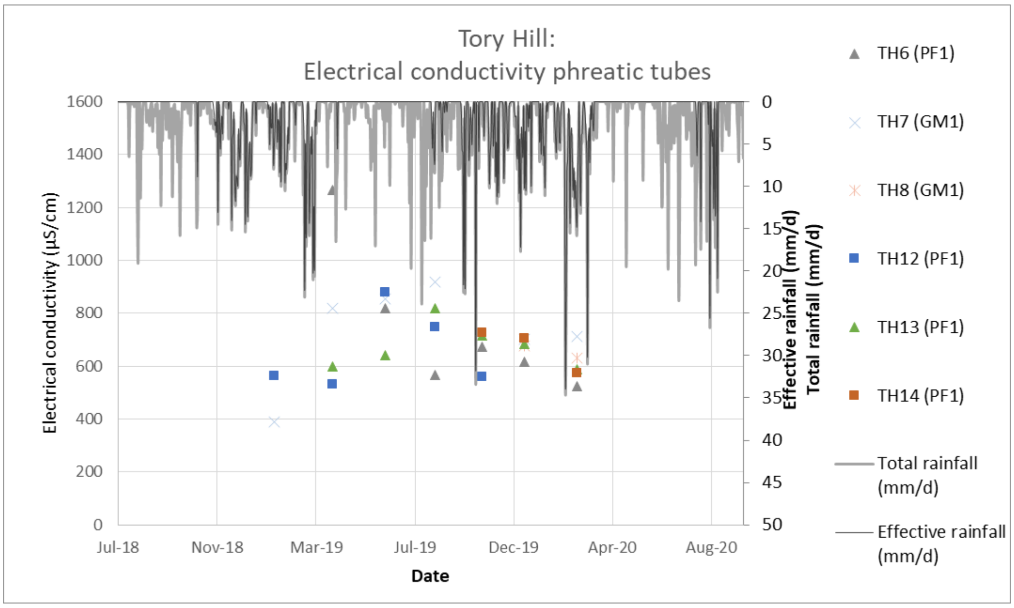


Figure 9.13. Time series of electrical conductivity (µm/cm) in phreatic tubes.

Most of other piezometers in Tory Hill seem to receive higher proportions of surface water mixed in with groundwater during the dry season, according to their lower values of EC (Figure 9.14). TH8, however, always receives high groundwater proportions with an EC between 800 and 1000 µm/cm.

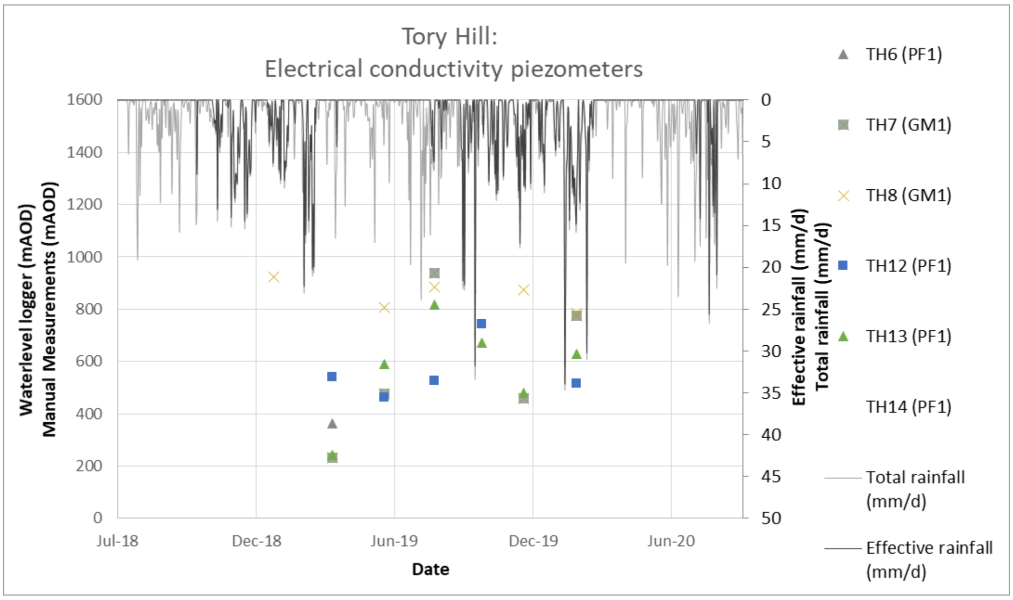


Figure 9.14. Time series of eelectrical conductivity (µm/cm) in phreatic tubes.

The boxplots in Figure 9.15 show the overall and seasonal EC data distributions collected in the fen as well as from boreholes outside the fen. Both the phreatic tubes and piezometers seem to have much lower EC with medians of 615 and 609 µm/cm respectively, implying that water table in Tory Hill is fed by a mixture of groundwater and surface water. Indeed according to a Welch t-test the boreholes (median of 825 µm/cm) were significantly different to the phreatic tubes (p-value = 0.01) and the piezometers (p-value = 0.00).

Furthermore, it seems that the phreatic water table is mainly fed by groundwater during the Spring/Summer whereas the Autumn/Winter seems to be fed by a greater proportions of surface water, as deduced earlier. This is possibly runoff from the lake in the north, overflowing into the fen, which has been observed during fieldwork trips in the winter period.

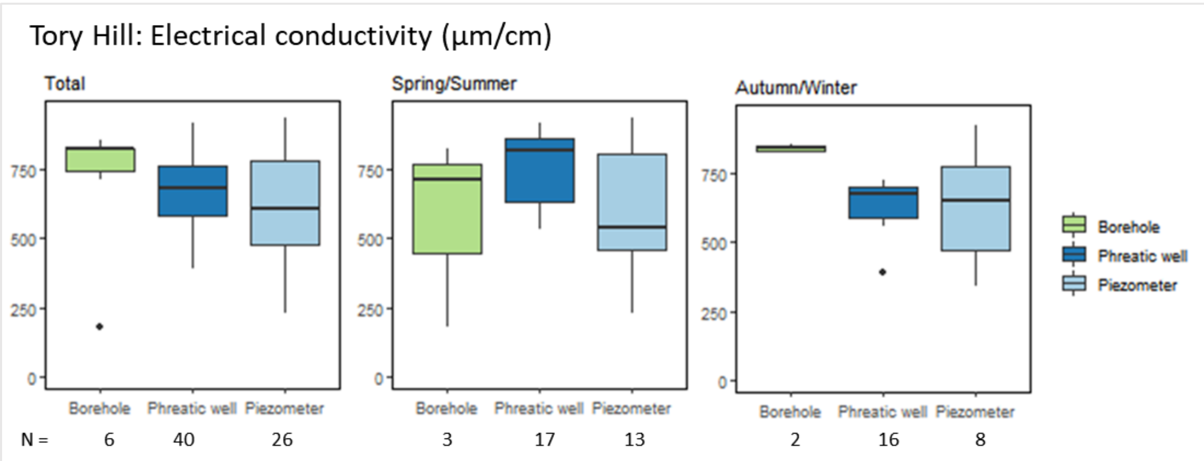


Figure 9.15. Electrical conductivity (µm/cm) in phreatic wells and piezometers inside and boreholes outside the fen in Tory Hill.

9.1.4.2. pH

The pH data Figure 9.16 suggest that values found in the fen are very similar to values found in boreholes outside the fen. Indeed, the median pH of the boreholes is 7.44 compared to the median for the piezometers in the fen of 7.29. The phreatic tubes contain an only slightly lower pH of 7.20.

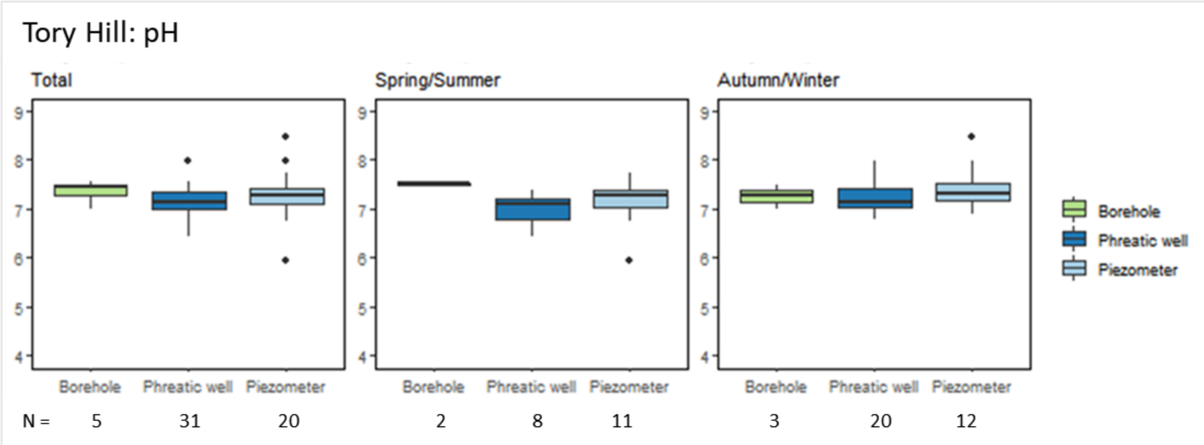


Figure 9.16. pH in phreatic wells and piezometers inside and boreholes outside the fen in Tory Hill.

9.1.4.3. Temperature

The temperature data boxplots in Figure 9.17 suggest that the temperature in the water column of the fen and the surrounding catchment are fairly similar. There is, however, a seasonal change in the boreholes as well as the phreatic tubes and the piezometers. The largest seasonal change is recorded in the phreatic tubes with a Spring/Summer median of 13.5 °C compared to lower the Autumn/Winter median of only 7.5 °C. This seasonal change is to be expected for the water column at the surface but also might signify that a greater proportion of surface water is present here.

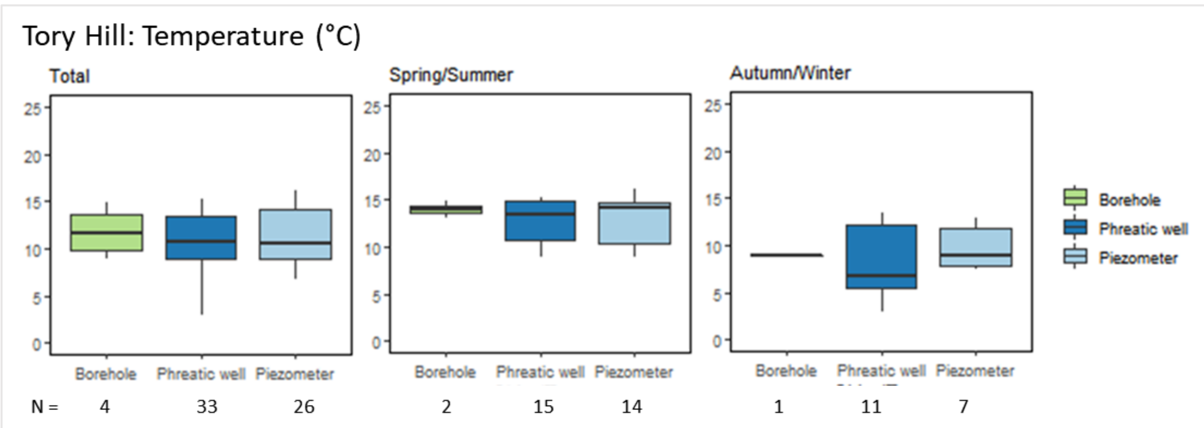


Figure 9.17. Temperature (°C) in phreatic wells and piezometers inside and boreholes outside the fen in Tory Hill.

9.1.5. Conceptual hydrology model

Several findings can be summarised from results presented and discussed in previous sections:

- The outlet adjacent to Tory Hill is sustained by a large catchment (13.35 km^2) relative to the size of the fen (0.16 km^2). This outlet is responsible to keep the fields around Tory Hill from flooding and has a significant control on the fen water levels. The fen itself is presumably fed by a (topographical) catchment of 0.36 km^2.
- Diurnal evapotranspiration fluctuations were not visible since the outlet had a very strong draining influence on the phreatic water level.
- Groundwater seems to be entering the fen at the bottom of the steep limestone escapement in the west with higher proportions during the summer than the winter. During the winter the fen seems fed by a large proportions of surface water which is possibly overland flow from the lake.

9.2. Hydrochemistry

The following section contains a series of boxplots of the hydrochemistry data gathered in and outside Tory Hill. A total of 102 samples were collected from boreholes, phreatic tubes and piezometers and subsequently analysed for phosphorus, nitrogen and other hydrochemistry. Considerably fewer samples could be gathered at this fen compared to the other sites due to empty phreatic wells and piezometers during dry periods. The total collected data set are displayed as well as the seasonal differences between the spring and summer with samples collected between April 1th and September 30th and the autumn and winter with samples collected between October 1th and March 31th.

9.2.1. Phosphorus

The boreholes around Tory Hill are found with considerably higher concentrations of dissolved reactive phosphorus (DRP) than in the phreatic layers of the fen itself (Figure 9.18). The concentrations in the boreholes (median of 0.110 mg-P/l) were found to be statistically significantly lower than the phreatic tubes (median of 0.020 mg-P/l) with a reported p-value of 0.05. This however, was not the case when compared to concentrations in the piezometers (median of 0.090) where a p-value of 0.93 was returned. It therefore seems that the groundwater in the catchment is flowing up through the underlying substrate. This is possible that this nutrient rich water then is deposited into the phreatic water table since most of the piezometers had a screen intake at around 1 mBGL. However, this high DRP influx does not seem to be reflected in the phreatic tubes which implies the fen vegetation picks up this nutrient and is then internally cycled by the wetland itself as seen in the other sites.

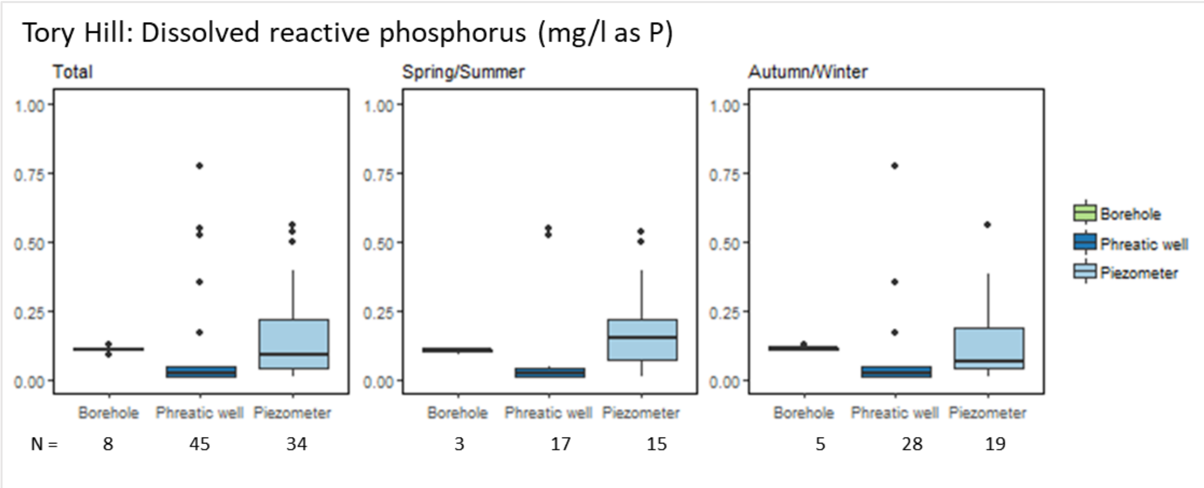


Figure 9.18. Dissolved reactive phosphorus in mg/l as P sampled from phreatic wells and piezometers inside and boreholes outside the fen in Tory Hill.

The total phosphorus is found with much higher concentrations in the phreatic layer (median of 0.459 mg-P/l) than outside the fen (median of 0.101 mg-P/l) which suggests that the levels are not due to direct supply from the outside aquifer, but rather due to active breakdown of organic matter in the upper layers of the peat (as seen in Figure 9.19) as well build up by internal wetland cycling. Indeed the concentrations in the phreatic layers were found to be significantly higher (p-value of 0.00). Furthermore, it seems that part of the TP is cycled into the substrate below due to downward gradients with the piezometers displaying a median concentration of 0.255 mg-P/l. Another possibility is that organic matter is also broken down in this deeper layer since oxygen rich water (from surface water fluxes) was also flowing into this layer speeding up the break down process of the larger proteins. The concentrations in piezometers were also proven to be statistically significantly greater than those in the boreholes (p-value = 0.00).

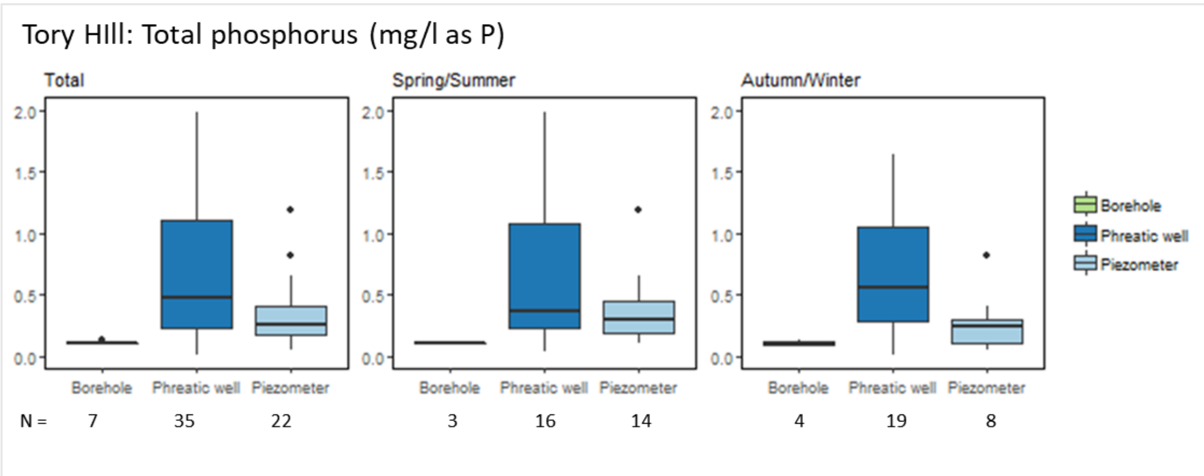


Figure 9.19. Total phosphorus in mg/l as P sampled from phreatic wells and piezometers inside and boreholes outside the fen in Tory Hill.

The DRP concentrations measured in the fen's catchment is somewhat reflective of concentrations found in the fen whereas TP is not found to be statistically significantly higher in the fen than the boreholes. Hence, this suggests that Tory Hill fen is reacting to accumulate the incoming P in its phreatic layer which is a contrast to Ballymore and Scragh Bog but not to Pollardstown site A and D which show similar trends.

The ratios calculated from the medians of the 'Total' boxplots showed ratios of DRP and TP is 1:1 in boreholes, somewhat higher in piezometers with 1:3 and 1:13 in the phreatic tubes.

Finally, the DRP concentrations in the boreholes and the piezometers substantially exceed the reported groundwater threshold values in Ireland (Government of Ireland, 2010) of 0.035 mg-P/l. mg-P/l. This is not true for water sampled in phreatic water table.

9.2.2. Nitrogen

Contrary to the other sites, total ammonia was found with higher values in the fen (Figure 9.20) with a median of 0.12 and 0.52 mg-N/l for phreatic tubes and piezometers respectively compared to the boreholes (median of 0.04 mg-N/l). The values in the fen were found to be statistically significantly higher (p-values of 0.00 and 0.01)

It therefore seems that relatively high ammonia is not being brought in from the groundwater aquifer, rather it is being generated in-situ by the breakdown of vegetation in the piezometric layers. The concentrations of ammonia are somewhat lower in the phreatic zone than in the piezometers, indicative of the more oxidised environment in conjunction with the subsequent nitrate uptake by the vegetation.

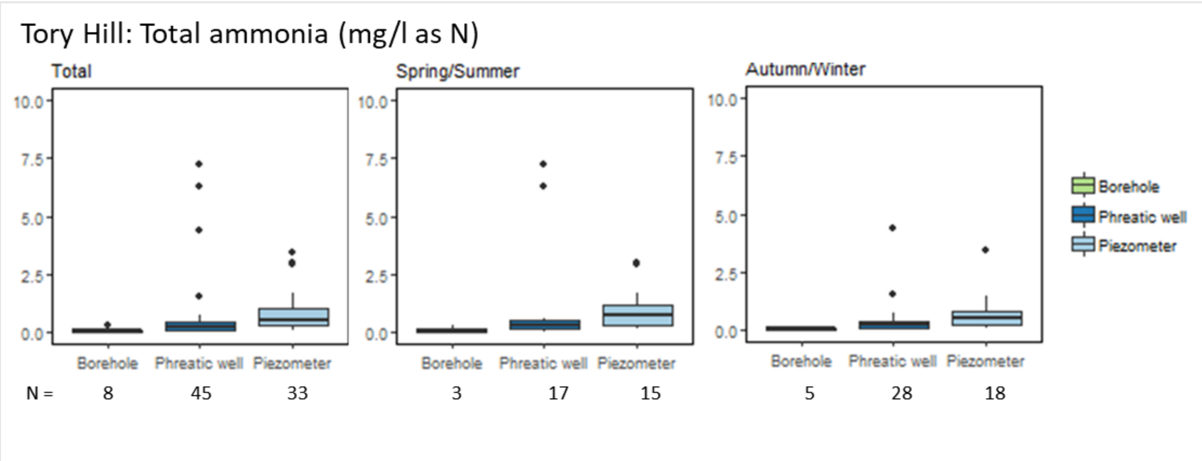


Figure 9.20. Total ammonia in mg/l as N sampled from phreatic wells and piezometers inside and boreholes outside the fen in Tory Hill.

Nitrite is barely found (Figure 9.21) in and around Tory Hill with almost all samples were analysed below the limit of detection which was 0.05 mg-N/l.

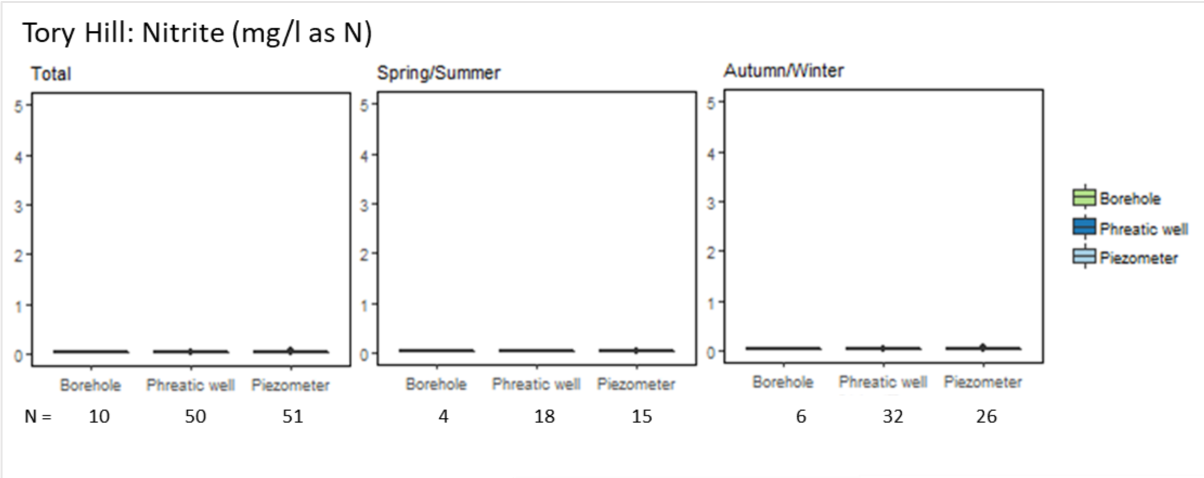


Figure 9.21. Nitrite in mg/l as N sampled from phreatic wells and piezometers inside and boreholes outside the fen in Tory Hill.

The total oxidised nitrogen (nitrate) concentrations measured in the catchment (median of 4.17 mg-N/l) were significantly higher than those in the phreatic tubes (median of 0.02 mg-N/l) and piezometers (median of 0.05 mg-N/l) with a p-values of 0.02 and 0.01 respectively (Figure 9.22). The low concentration in the fen is probably due to the vegetation picking up this nutrient. This process is more apparent when comparing the difference between the Spring/Summer and Autumn/Winter concentrations found in the phreatic tubes. The vegetation in the winter is not actively growing and therefore not taking up as much total oxidised nitrogen as in the summer. Furthermore, there may be some loss of nitrate due to denitrification when flowing from the aquifer through the organic rich substrate (which may show anoxic conditions in places) into the phreatic water table. However, Autumn/Winter concentrations were not proven to be statistically significantly higher with a p-value of 0.14.

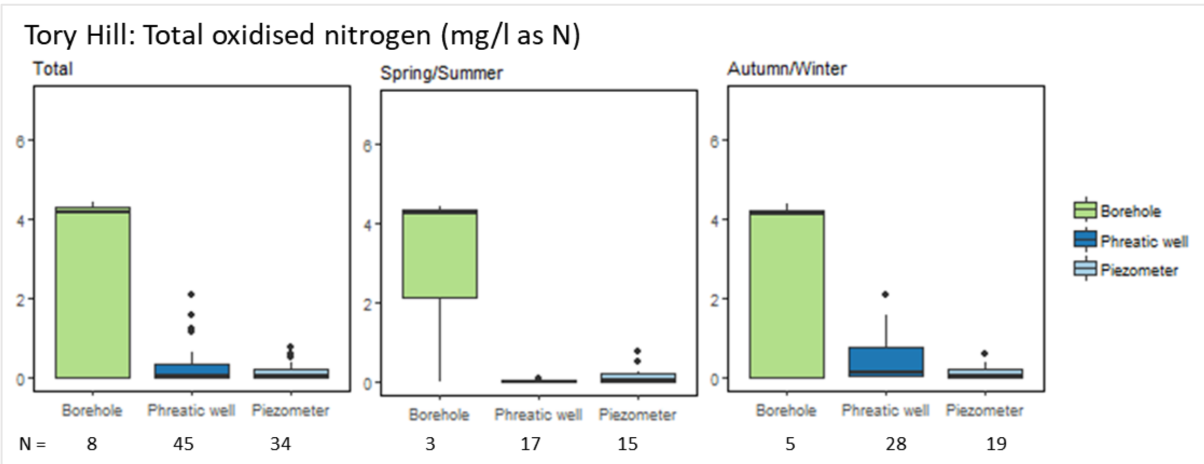


Figure 9.22. Total oxidised nitrogen in mg/l as N sampled from phreatic wells and piezometers inside and boreholes outside the fen in Tory Hill.

The TDN data in Figure 9.23 displays the same trend as earlier where total dissolved nitrogen is found to be at higher concentrations in the catchment groundwater (median of 5.28 mg-N/l) than in the phreatic tubes (median of 2.14 mg-N/l) and the piezometers (median of 2.04 mg-N/l). Both phreatic tubes and piezometers were found with statistically lower values with a p-value of 0.00 returned by both tests. Again, this could be reflective of the vegetation picking up the nitrate as well as the occurrence of denitrification processes in the fen's peat.

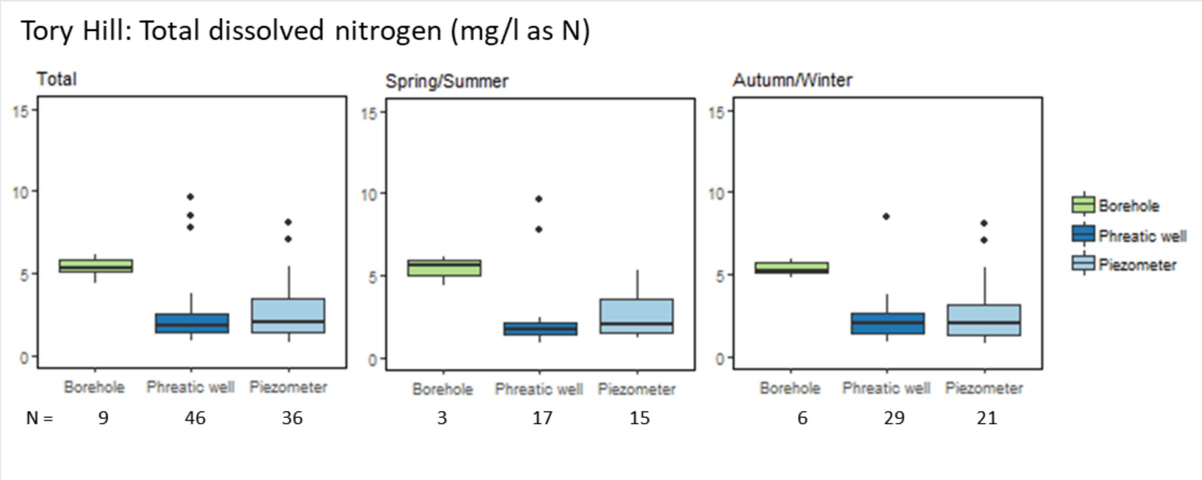


Figure 9.23. Total dissolved nitrogen in mg/l as N sampled from phreatic wells and piezometers inside and boreholes outside the fen in Tory Hill.

The ratios of total ammonia to total dissolved nitrogen were 1:135 in boreholes and smaller in piezometers (1:4) and phreatic tubes (1:18). The ratio of total oxidised nitrogen to total dissolved nitrogen was highest for the boreholes at 1:1, compared to much lower ratios 1:42 and 1:89 in the piezometers and phreatic tubes respectively.

From the reported medians, a nitrogen balance shows that 71% of TDN are in different forms of organic nitrogen in the piezometers. This percentage was even higher in phreatic tubes where 92% was deemed to be in a form of organic nitrogen.

The reported medians do not exceed groundwater threshold values for nitrite nor nitrate (Government of Ireland, 2010). None of the medians found in and around the fen were found to be higher than 0.114 mg-N/l for nitrite. There were also no outliers found that exceeded the threshold value.

The total oxidised nitrogen concentrations had an insignificant amount of nitrite and a therefore can be regarded as a reflection of nitrate. None of the measured concentrations exceeded the threshold value of 8.47 mg-N/l.

9.2.3. Other chemistry

The overall concentrations of alkalinity were somewhat higher in the fen compared to data gathered from the boreholes in the wider catchment, as seen in Figure 9.24. The median in the boreholes was 200.0 mg/l as $CaCO_3$ whereas the phreatic tubes and piezometers concentrations were higher with medians of 206.5 and 234.8 mg/l as $CaCO_3$ respectively. The concentrations in the boreholes were statistically significantly lower than the phreatic tubes (p-value = 0.00) as well as the piezometers (p-value = 0.01).

Interestingly, the alkalinity seems to fluctuate seasonally in the phreatic layer with higher concentrations during the Autumn/Winter (median of 234.7 mg/l as $CaCO_3$) than in the Spring/Summer (median of 184.7 mg/l as $CaCO_3$) but this does not seem to be caused by higher groundwater fluxes as the concentrations in the boreholes were lower. This seasonal increase, however, was not significantly higher (p-value of 0.16).

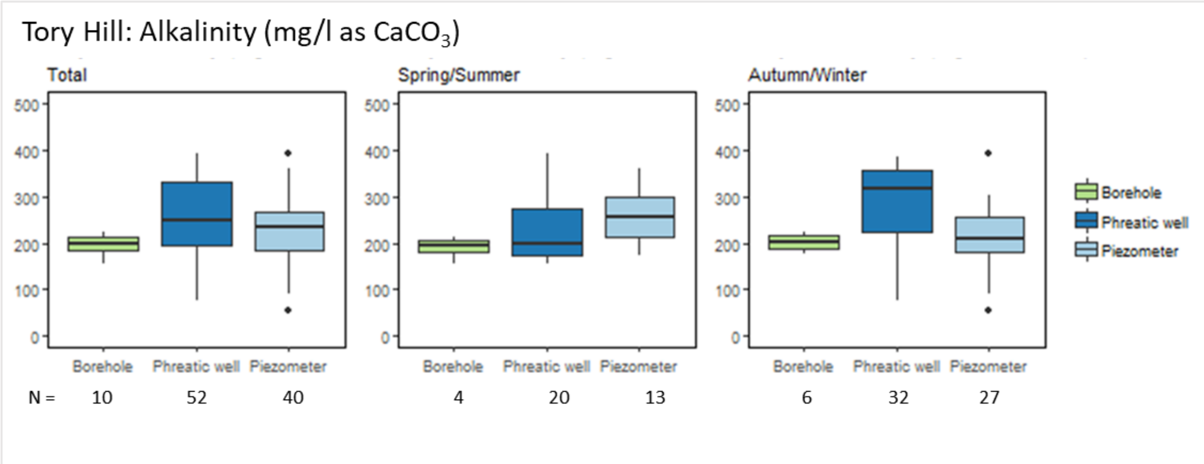


Figure 9.24. Alkalinity in mg/l as $CaCO_3$ sampled from phreatic wells and piezometers inside and boreholes outside the fen in Tory Hill.

The overall concentrations of chloride were much higher in the boreholes than the fen with a median of 39.4 mg/l (Figure 9.25). The median of the chloride concentration in the phreatic tubes was lower with 30.3 mg/l and statistically significant with a p-value of 0.00. The median of the piezometers was also much lower with 27.2 mg/l and but not significantly (p-value of 0.77). From the contrasts between the fen and the boreholes the implication can be made that both the phreatic and piezometric layer are receiving quite a large proportion of surface water. All values are lower in the winter period reflecting possibly shorter residence time in the aquifer (for the borehole samples) and higher effective rainfall / surface water inputs for the in-fen samples.

The reported medians are still far below the Irish groundwater threshold values for chloride (Government of Ireland, 2010) which is 187.5 mg/l.

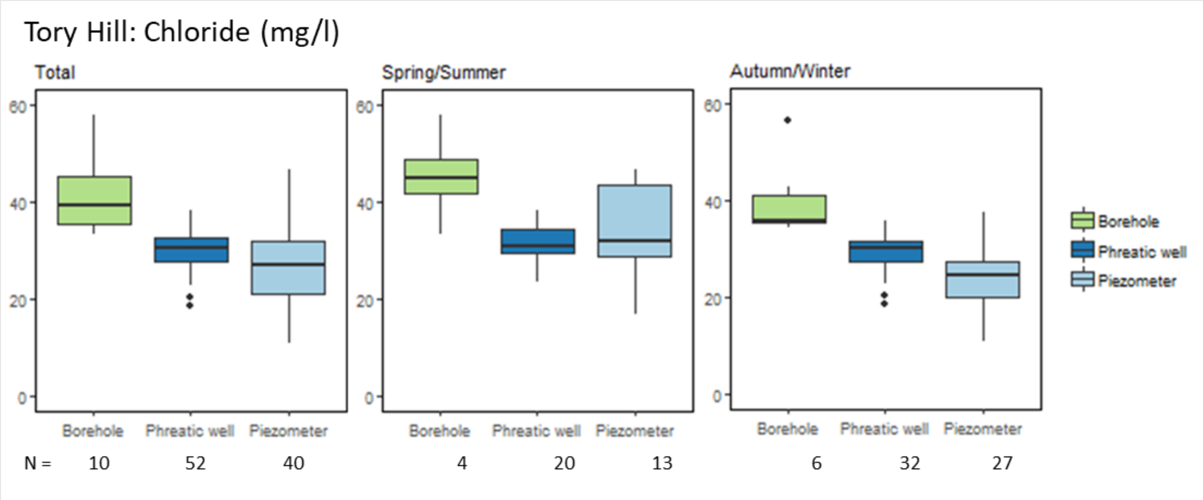


Figure 9.25. Chloride in mg/l sampled from phreatic wells and piezometers inside and boreholes outside the fen in Tory Hill.

The hypothesis that the fen receives a large proportion of surface water is further confirmed with the contrasting concentrations of silica in and around the fen. Figure 9.26 shows that the boreholes had the highest concentrations of silica with a median of 12.4 mg/l as SiO_2. The concentrations found in the fen are statistically significantly lower with a p-values of 0.00 for both phreatic tubes and piezometers. Median silica concentrations in the phreatic tubes were higher with a median concentration of 4.1 mg/l as SiO_2 compared to the concentrations in the piezometers with a median of 3.6 mg/l as SiO_2.

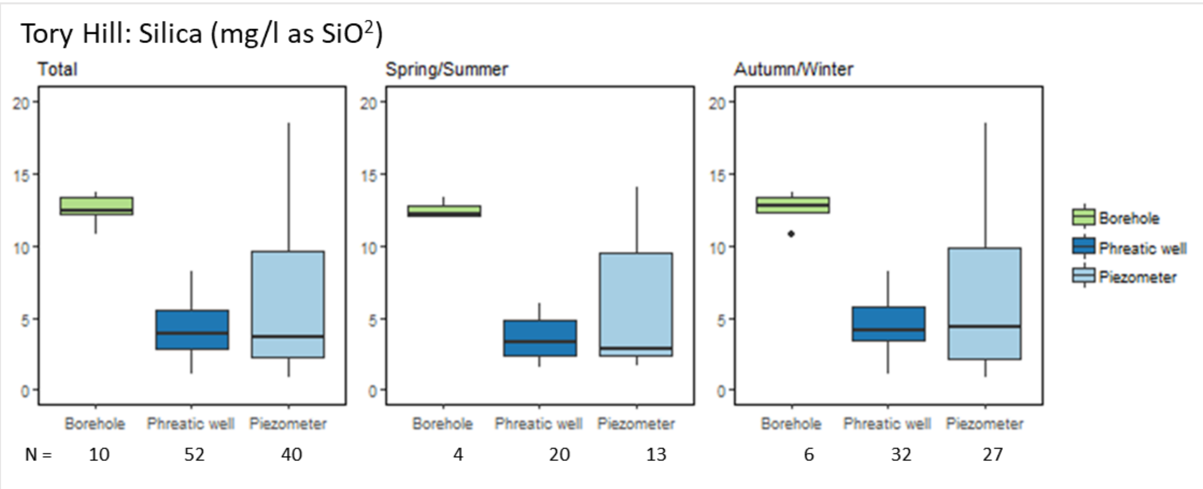


Figure 9.26. Silica in mg/l as SiO^2 sampled from phreatic wells and piezometers inside and boreholes outside the fen in Tory Hill.

Sulphate concentrations were found to be much higher in the piezometers (median of 83.4 mg/l as SO_4^{2-}) and the phreatic tubes (median of 42.0 mg/l as SO_4^{2-}) than in the surrounding catchment with a median 19.9 mg/l as SO_4^{2-} (Figure 9.27). These differences were found to be statistically significantly higher with p-values of 0.00 for both phreatic tubes and piezometers. Given that the nitrogen in the boreholes was totally in nitrate form, it shows that the aquifer seems to be in an aerobic condition, which means that the source of sulphate in the fen is unlikely to be from sulphide in groundwater (which subsequently gets oxidised to sulphate in the fen). This suggests that the high sulphate concentrations found in the fen must be originating from another source. The increase of sulphate could either be explained by a release of sulphate from decaying vegetation and/or higher concentrations of oxygen entering the fen via surface water causing sulphide to oxidise into sulphate (Wheeler and Proctor, 2000; Cusell et al., 2013; McBride et al., 2010).

The concentrations of sulphate in the catchment groundwater are far below the Irish groundwater threshold values for sulphate (Government of Ireland, 2010) of 187.5 mg/l, but this is not true for the concentrations found in the fen. In fact 15% of all samples collected in the fen were measured above this threshold value. It may be possible that these high values have negative effects on the growth of certain fen species. Indeed, Geurts, et al., (2009) found that fertilization of sulphate led to the dominance of fast-growing eutrophic species which outcompeted most key species for vegetation development in fens.

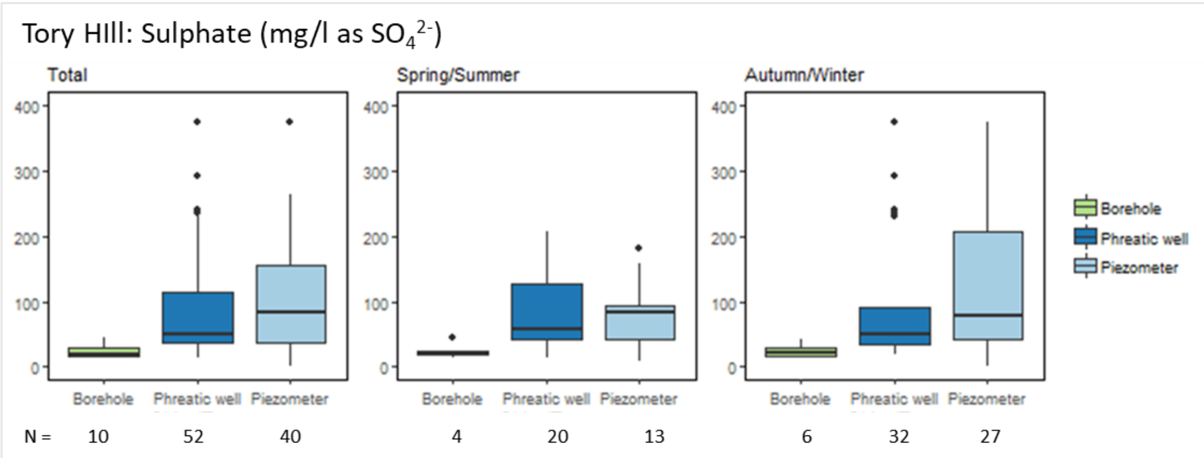


Figure 9.27. Sulphate in mg/l as SO_4^{2-} sampled from phreatic wells and piezometers inside and boreholes outside the fen in Tory Hill.

Dissolved organic carbon results showed higher concentrations in the fen than in the supporting aquifer which is expected from the breakdown of dead vegetation in peat (Figure 9.28). The boreholes in the fen catchment had a median of 3.5 mg/l compared to medians of 11.9 mg/l and 19.2 mg/l in the phreatic and piezometer sampling positions respectively. Welch T-tests proved that the boreholes had statistically significant lower values compared to both the phreatic tubes (p-value = 0.00) and the piezometers (p-value = 0.00).

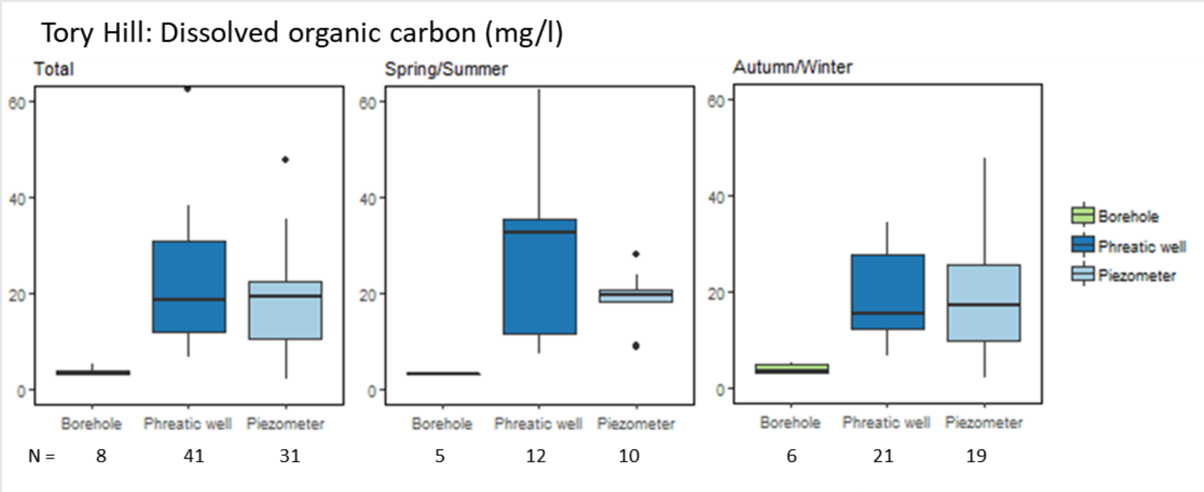


Figure 9.28. Dissolved organic carbon in mg/l sampled from phreatic wells and piezometers inside and boreholes outside the fen in Tory Hill.

No ferrous iron was detected in the boreholes around the fen with concentrations below the limit of detection (0.006 mg/l as Fe^{2+}), as seen in Figure 9.29. Concentrations were measured in the phreatic wells (median of 0.050 mg/l as Fe^{2+}) and the piezometers (median of 0.144 mg/l as Fe^{2+}) which was proven to be statistically significant with p-values of 0.00 returned by both tests. These high ferrous iron concentrations imply that the fen's phreatic layer as well as piezometric layer is under the influence of reducing conditions. These results corroborate the conclusion made previously regarding the fact that sulphur was very unlikely to be coming into the fen in the groundwater from the supporting aquifer in a reduced sulphide form, but is generated in more reducing conditions within the fen

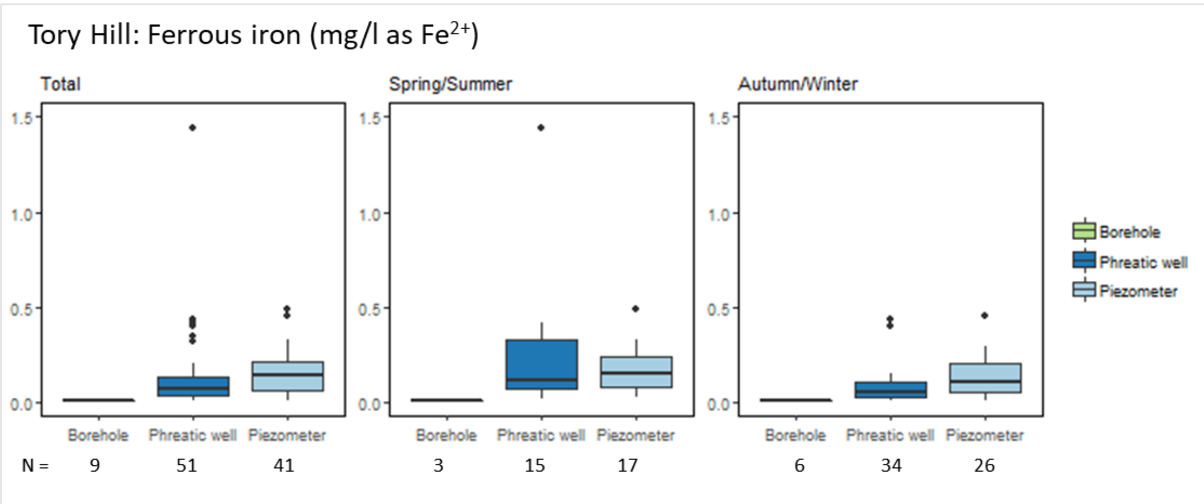


Figure 9.29. Ferrous iron in mg/l as Fe^{2+} sampled from phreatic wells and piezometers inside and boreholes outside the fen in Tory Hill.

9.2.4. Conceptual hydrochemistry model

- Higher DRP concentrations are flowing into the substrate of the fen which is taken up by vegetation in the phreatic layer and subsequently undergoes internal cycling.

- TP is high in the phreatic layer reflecting active breakdown of peat in an oxidized environment as well as build up by internal wetland cycling.
- There may exist some extremal airborne source of sulphur to the fen as high concentrations in the phreatic layers of the fen do not seem to originate from the surrounding catchment.
- Much lower concentrations of chloride and silica are proof that Tory Hill is fed by a large proportion of surface water (attributed to the overflow of the lake to the north).

9.3. Linkage to fen habitat

9.3.1. Hydrology and fen habitat

9.3.1.1. Boxplots water level

The boxplot in Figure 9.30 displays overall water levels for the different Fossitt habitats in the fen. Unlike the other fens Tory Hill has the lowest recorded phreatic water tables throughout the hydrological year for Rich fen and flush (PF1) habitat with a median of -0.18 mAGL. It was also established that this habitat was in a poor condition (see Section 10.1). This habitat further experiences a high rate of fluctuation in the 1st quartile (0.02 mAGL) and 3rd quartile (-0.52 mAGL). The Marsh (GM1) habitat experiences even lowers water levels with median of -0.37 mAGL and -0.51 mAGL for phreatic tubes and piezometers respectively. The piezometric water table especially seems to experience a high rate of fluctuation between the 1st quartile (0.01 mAGL) and 3rd quartile (-0.69 mAGL).

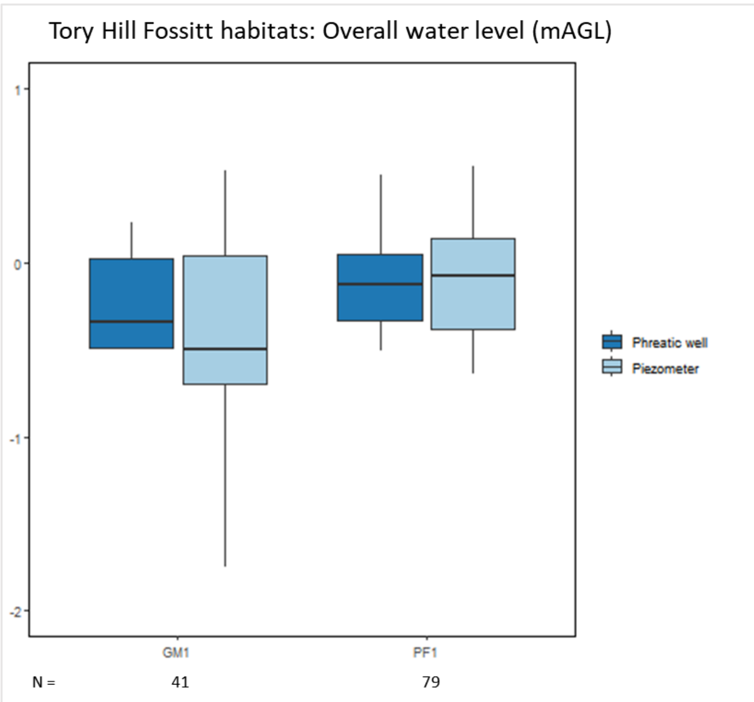


Figure 9.30. Overall water level in meters above ground level in the different habitats of Tory Hill measured in phreatic wells and piezometers.

Seasonally, the phreatic water levels in habitat PF1 remain fairly constant (Figure 9.31). This is, however, not true for the piezometric water table. Here the piezometers experienced a drop of 0.12 m in the Spring/Summer median. Waterlevels in GM1 also dropped during the summer with a median decrease of 0.23 m in the phreatic wells but no seasonal change was recorded in the piezometers. It has to be noted that the lowest recorded levels are limited as many phreatic tubes and piezometers stood dry, meaning that true levels should have been recorded even lower than reported here.

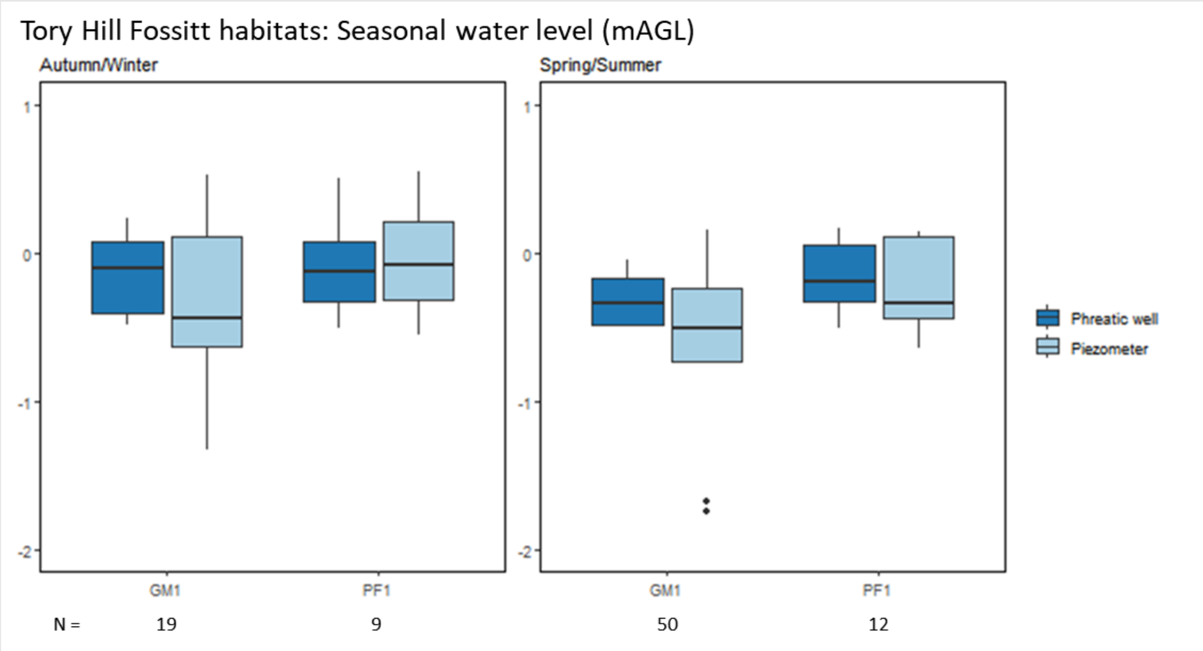


Figure 9.31. Seasonal water level in meters above ground level in the different habitats of Tory Hill measured in phreatic wells and piezometers.

9.3.1.2. Frequency duration curves

The GM1 habitat seems to be only supported by phreatic water levels above the surface elevation for <20% of the year (Figure 9.32). For about 50% of the time the levels were recorded at around 0.5 m below the surface which was actually the end of the screen of the phreatic tube. If the phreatic tube reached deeper it is expected that lower levels would have been recorded. Here the decreasing trend of the first 40% of the curve might be used to predict what the true phreatic level duration curve would look like. If this trend is continued water level could be expected to drop to a significant low level of up to 1 mBGL.

The same approach was used in the phreatic tube of a PF1 habitat. Here the end of the phreatic tube was reached at around 0.3 mBGL. Looking at the trend of the first 40% of the curve a lowest level of 0.8 mBGL could be expected in this area. Furthermore it seems that he PF1 habitat only seems to be supported by water levels above the surface for <37% of the year.

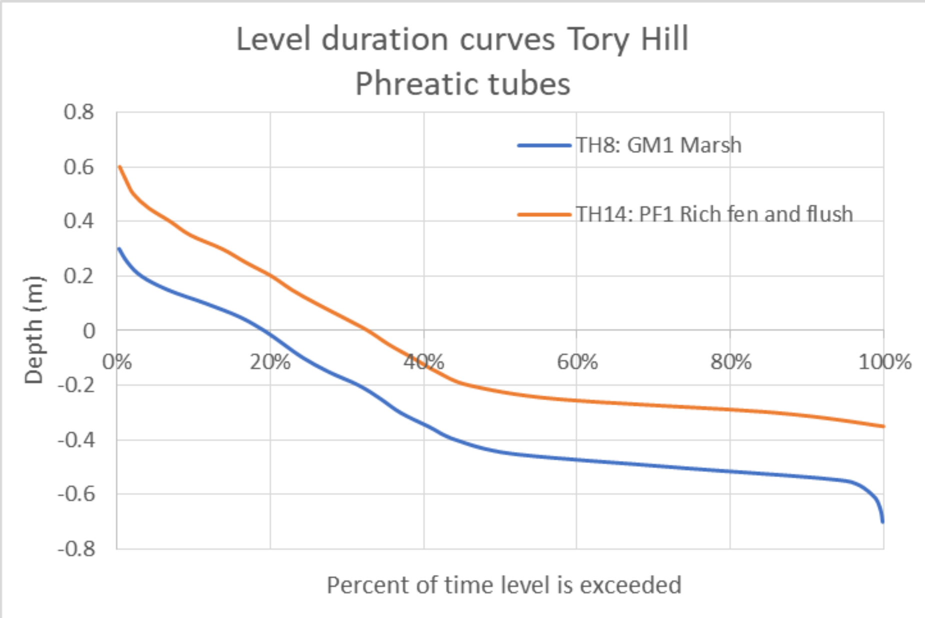


Figure 9.32. Phreatic level duration curves recorded in different habitats in Tory Hill. The negative numbers are water levels below groundlevel.

From the phreatic and piezometric water level data of TH8 and TH14 hydraulic gradients time-series were calculated, as shown in Figure 9.33. Both sites displayed strong downward gradients especially during the summer. These strong downward gradients were the most extreme out of all the fen sites indicating that the outlet level of the drain is clearly having a detrimental effect on the water levels within Tory Hill fen.

The downwards gradient is more extreme in TH14 than in TH8 which is located more upstream compared to the outlet. Here the hydraulic gradient is more maintained with support of flow from the lake even further upstream.

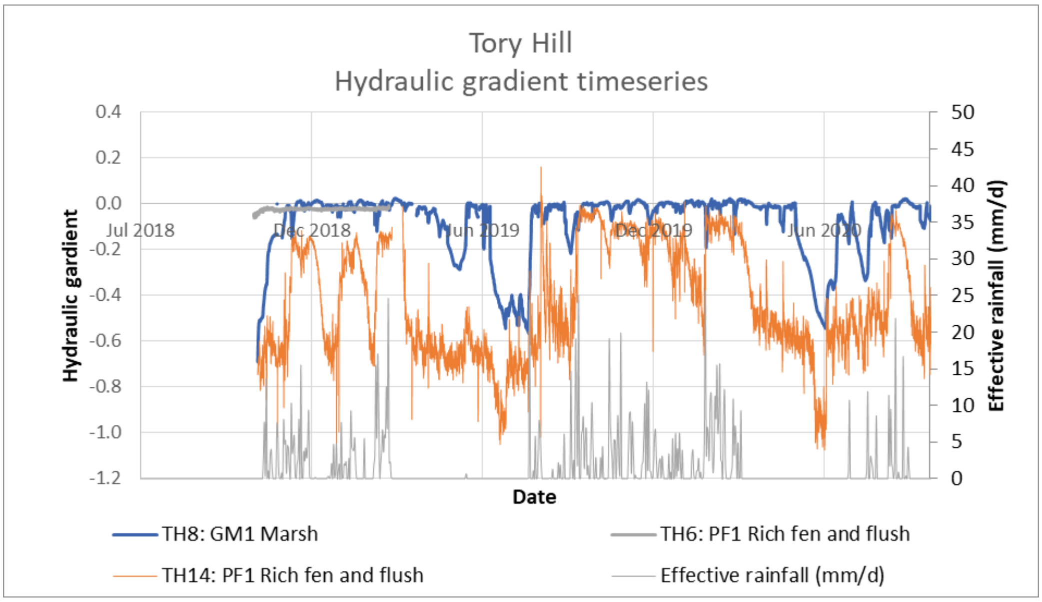


Figure 9.33. Hydraulic gradient timeseries calculated using the phreatic and piezometric water level timeseries in Tory Hill. Effective rainfall is displayed here as well.

Indeed, Figure 9.34 shows that the hydraulic gradient in TH8 is more or less maintained for around 70% of the time compared to TH14 which always seem to experience downward flows with extreme an extreme gradient of more than 0.5 for about 50% of the time.

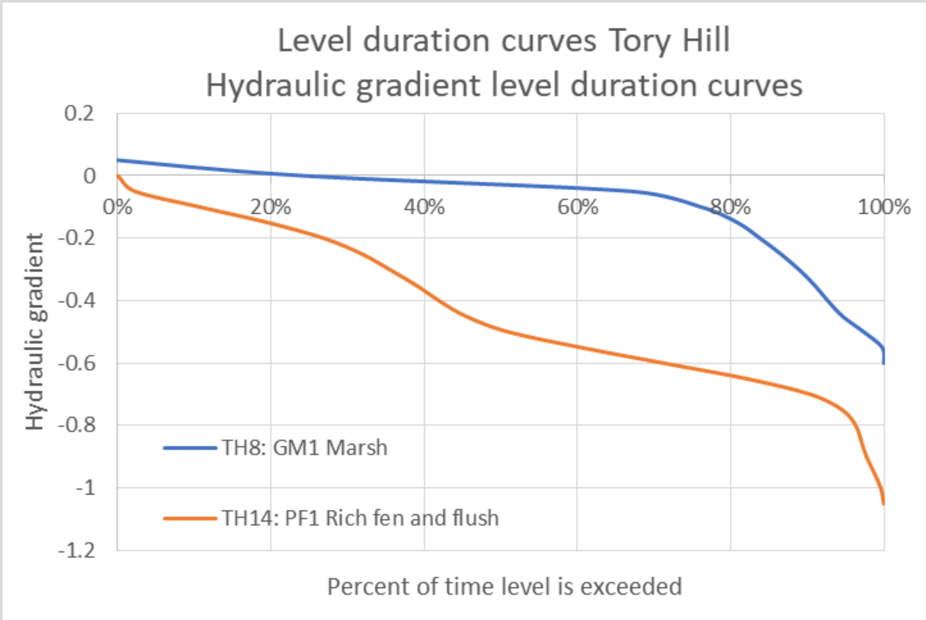


Figure 9.34. Level duration curves of the hydraulic gradients calculated from the water level time series in monitored phreatic tubes and piezometers.

9.3.2. Hydrochemistry

9.3.2.1. Nonmetric Multidimensional Scaling ordination

The NMDS plots were produced with the IVC data set containing the recorded species percentage at each surveyed relevé (this set contained 6 relevés and 31 species) and the environmental set (ENV) which consisted out of vegetation type cover (%), Fossitt habitat codes and the hydrochemistry results (mg/l). Unfortunately, no associations between the hydrochemistry data and the vegetation was found so only the NMDS plot with vegetation type cover (%) and Fossitt habitat codes can be presented (see Figure 9.33).

From this plot is observed that a high cover of bryophytes as well as bare ground is correlated with habitat PF1. Furthermore, it seems that the specific height of the vegetation is not particularly correlated with either PF1 or GM1.

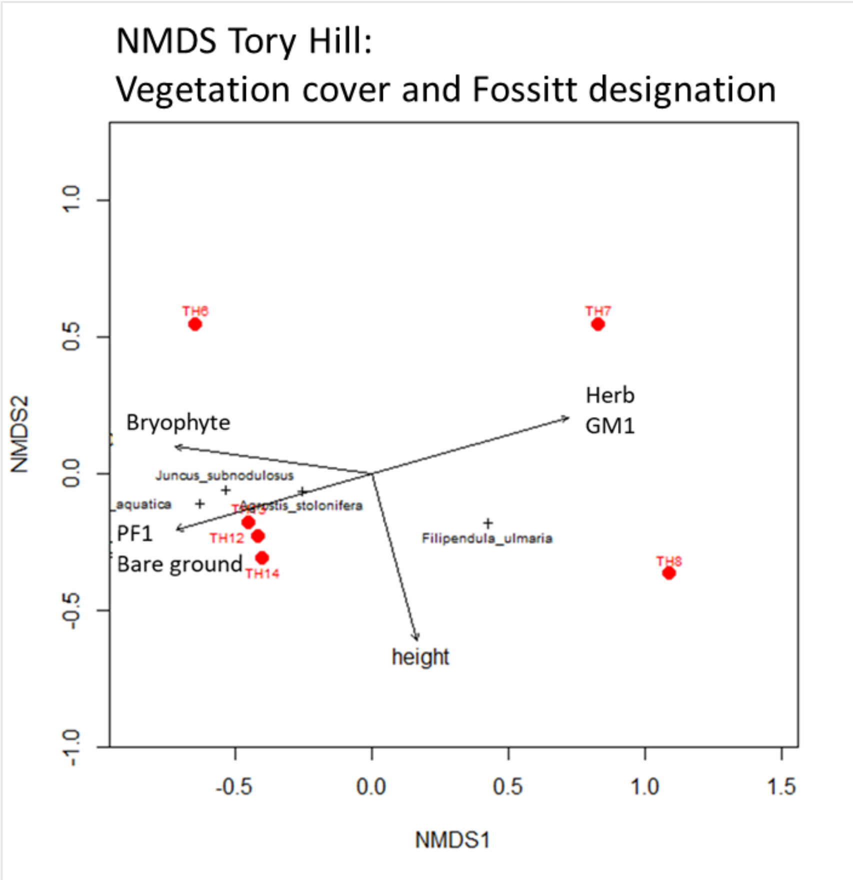


Figure 9.35. Multidimensional Scaling ordination of dimensions 1 and 2 with vegetation cover and Fossitt habitats plotted as vectors (max p-value = 0.2) in Tory Hill. The phreatic and piezometer nest locations are shown in red and the names of the species with the highest abundances (10%) are also plotted.

9.3.3. Mean seasonal hydraulic gradients and hydrochemistry

9.3.3.1. Dissolved reactive phosphorus

Figure 9.36 displays Tory Hill transect 1 with data collected in August 2019. The hydraulic gradients show greater water inputs at the base of the limestone hill (TH12) which is where the groundwater is expected to move into the fen via discrete springs. In location TH14 a steep drawdown towards the outlet is seen which implies that the water feeding the fen moves laterally in the upper levels of the peat and is drained rather quickly. This changes in February 2020 (Figure 9.37) where some small upward gradients were found, caused by high water levels in the outlet.

Furthermore, higher concentrations of DRP were found in August 2019. Interestingly, it seems that the hydraulic gradient in TH14 indicates that high concentrations may be moving from the underlying substrate to the phreatic water table. Concentrations decreased in February 2020 which can be attributed to high percentages of surface water diluting the water in the fen.

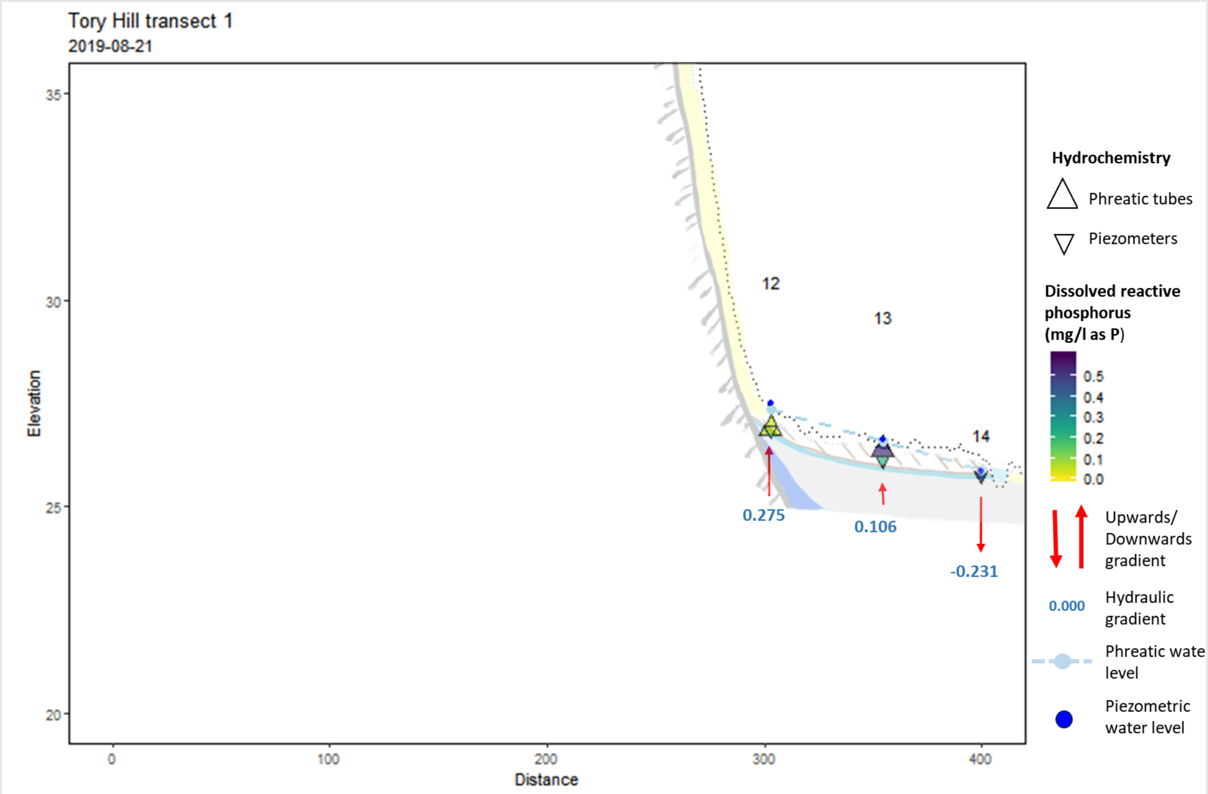


Figure 9.36. Hydrology and dissolved reactive phosphorus (mg/l as P) of Tory Hill transect 1 in August 2019. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

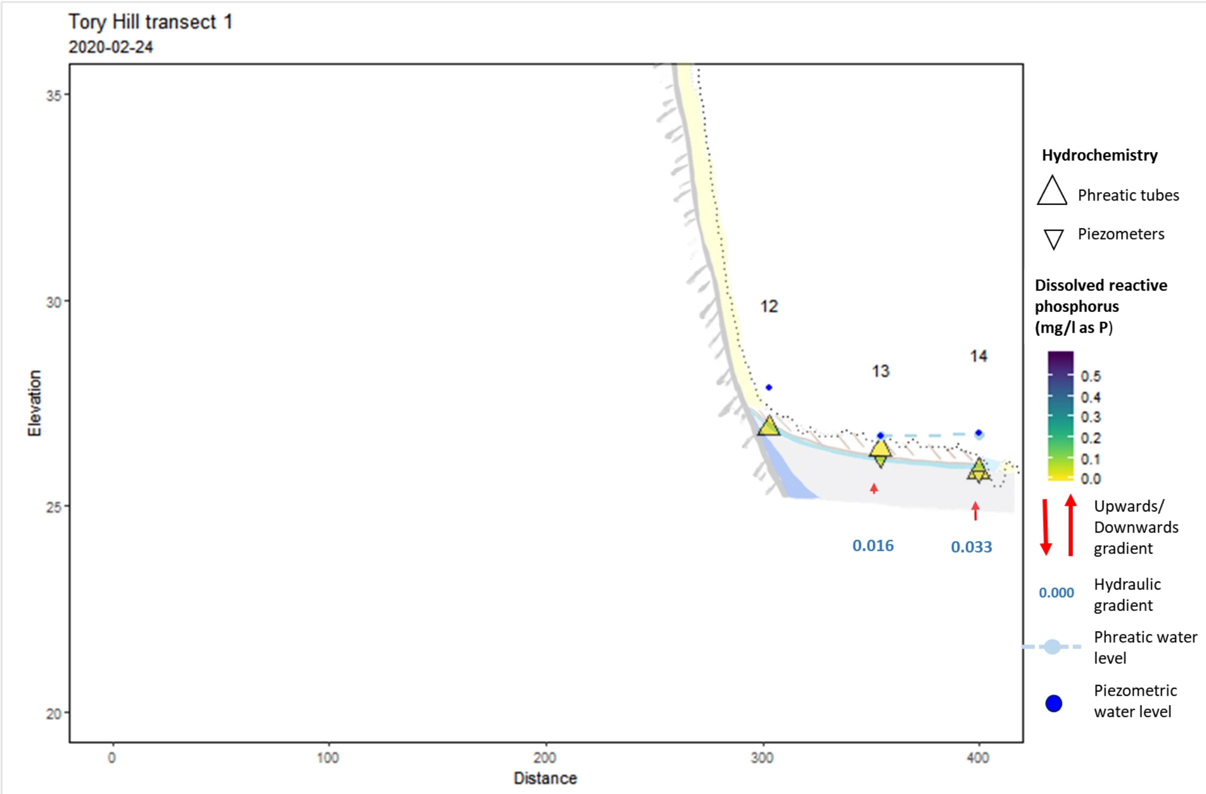


Figure 9.37. Hydrology and dissolved reactive phosphorus (mg/l as P) of Tory Hill transect 1 in February 2020. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

From Figures 9.38 and 9.39 it can be observed that high concentrations of DRP are present throughout the year in both the Pumphouse and TH8 at considerable depth. This implies that the underlying substrate of Tory Hill undergoes nutrient transformations and dispersion which is mainly visible in the underlying layers of the fen peat. Furthermore, concentrations are also somewhat elevated in the peat layer of TH7 which might be due to internal cycling as seen in the seasonally gradient shifts.

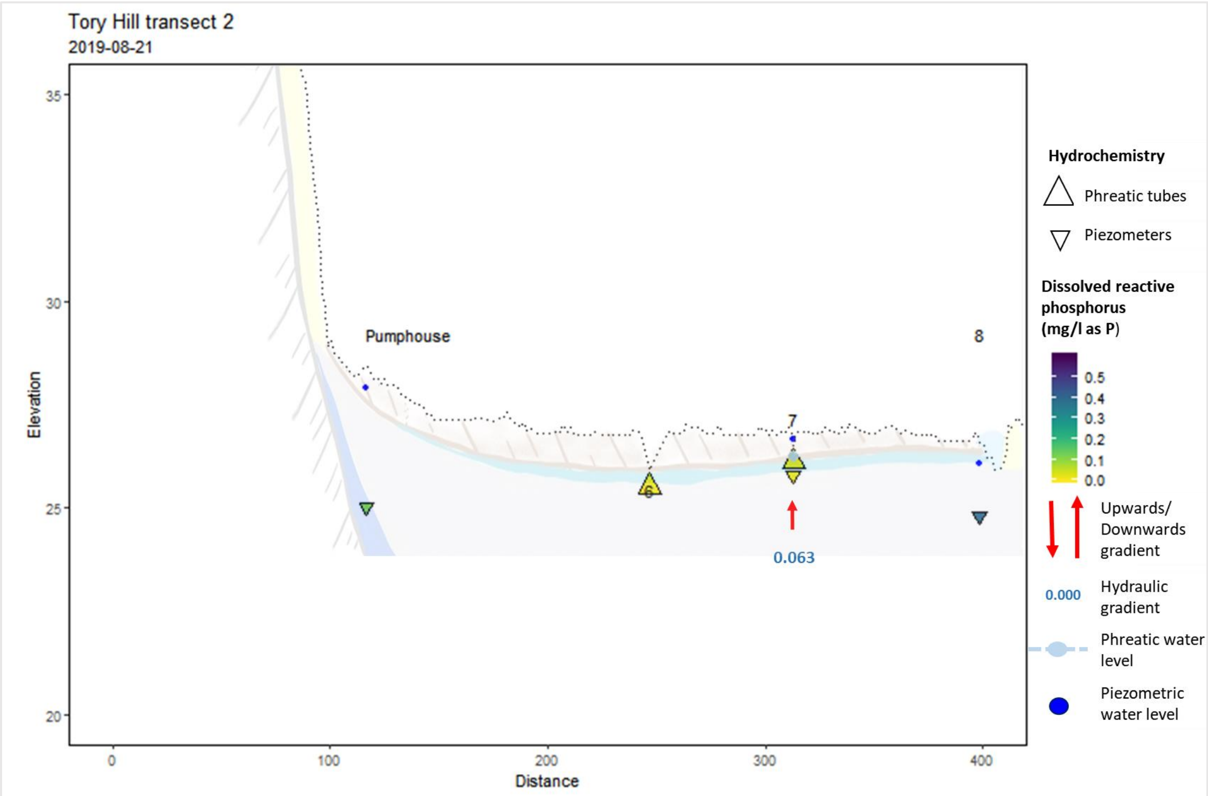


Figure 9.38. Hydrology and dissolved reactive phosphorus (mg/l as P) of Tory Hill transect 2 in August 2019. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

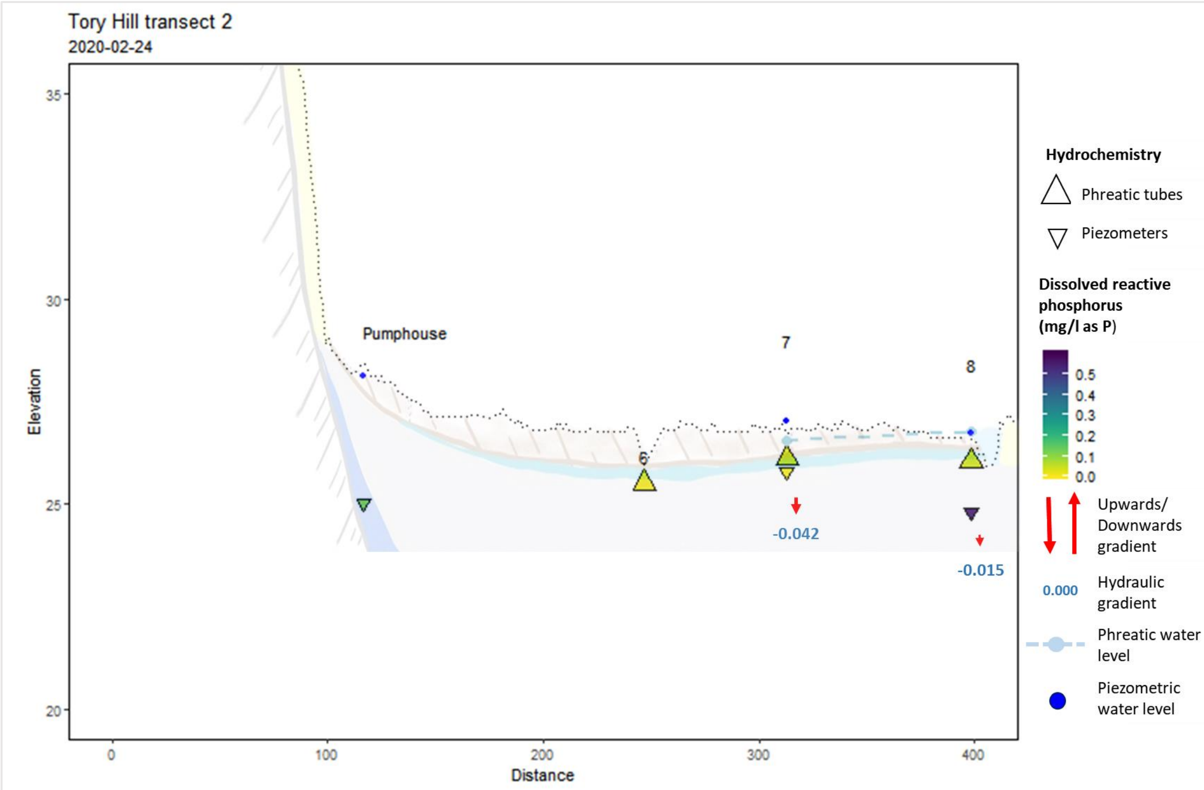


Figure 9.39. Hydrology and dissolved reactive phosphorus (mg/l as P) of Tory Hill transect 2 in February 2020. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

9.3.3.2. Total ammonia

Figures 9.40 and 9.41 show a similar trend for ammonia as for DRP (Section 9.3.3.1) where higher concentrations of total ammonia were found in August 2019. Again, it seems that the hydraulic gradient in TH14 might cause high concentrations to flow from the substrate to the phreatic water table. The concentrations then decrease in February 2020 due to dilution by surface water (increased effective rainfall).

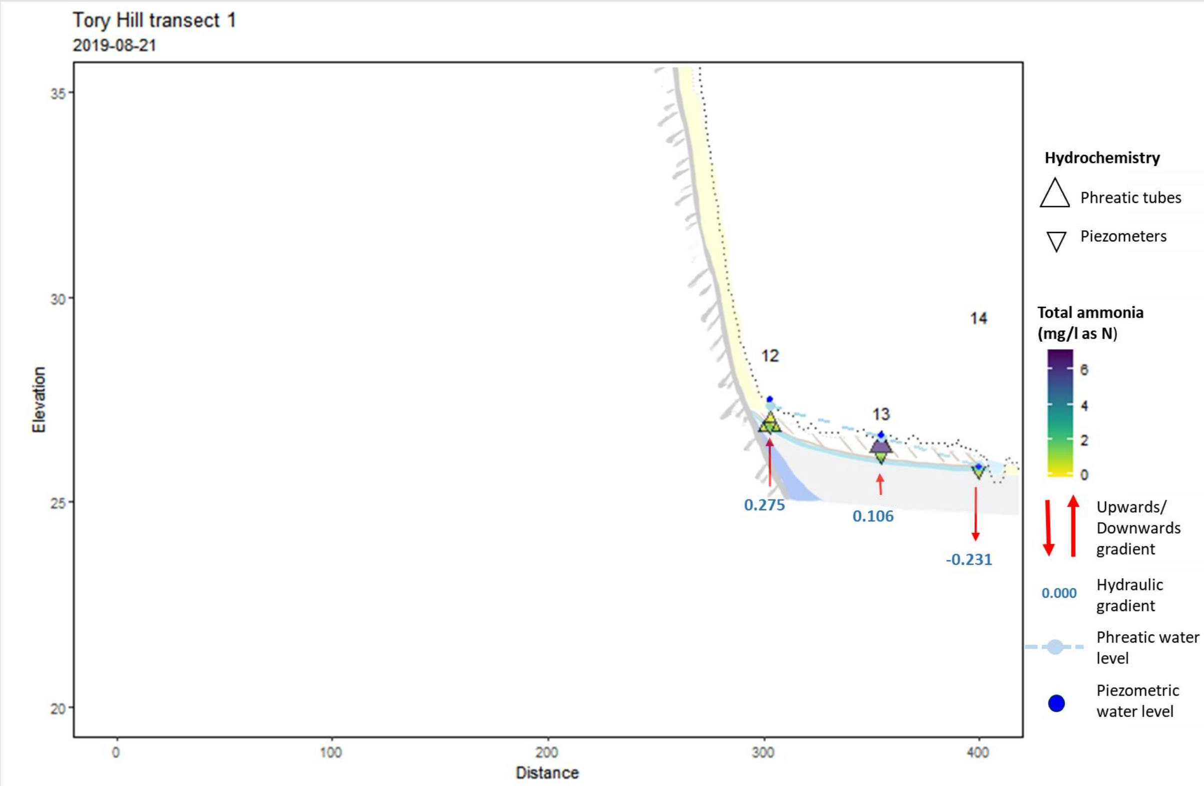


Figure 9.40. Hydrology and total ammonia of Tory Hill transect 1 in August 2019. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

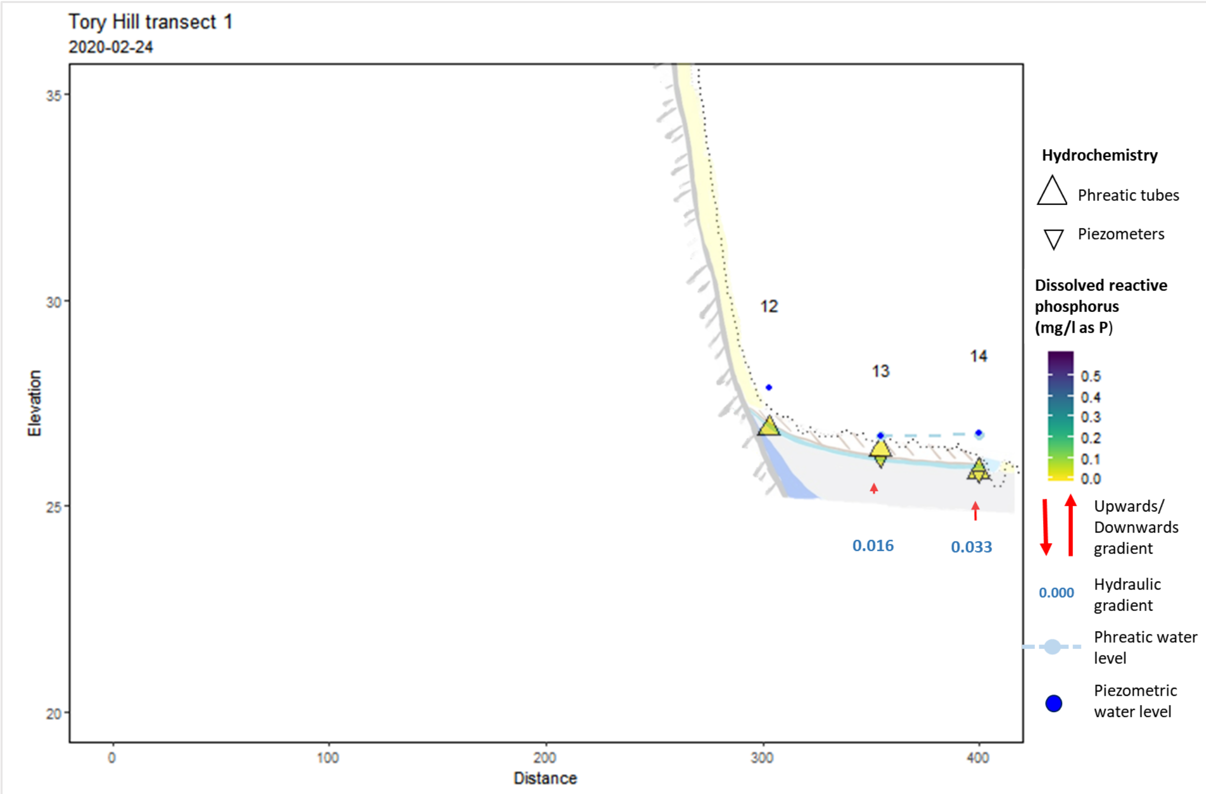


Figure 9.41. Hydrology and total ammonia of Tory Hill transect 1 in February 2020. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

Figures 9.42 and 9.43 show again a similar trend for ammonia as for DRP (Section 9.3.3.1) with higher concentrations present in TH7 and TH8. Interestingly, no elevated values are found in the Pumphouse implying oxic conditions there.

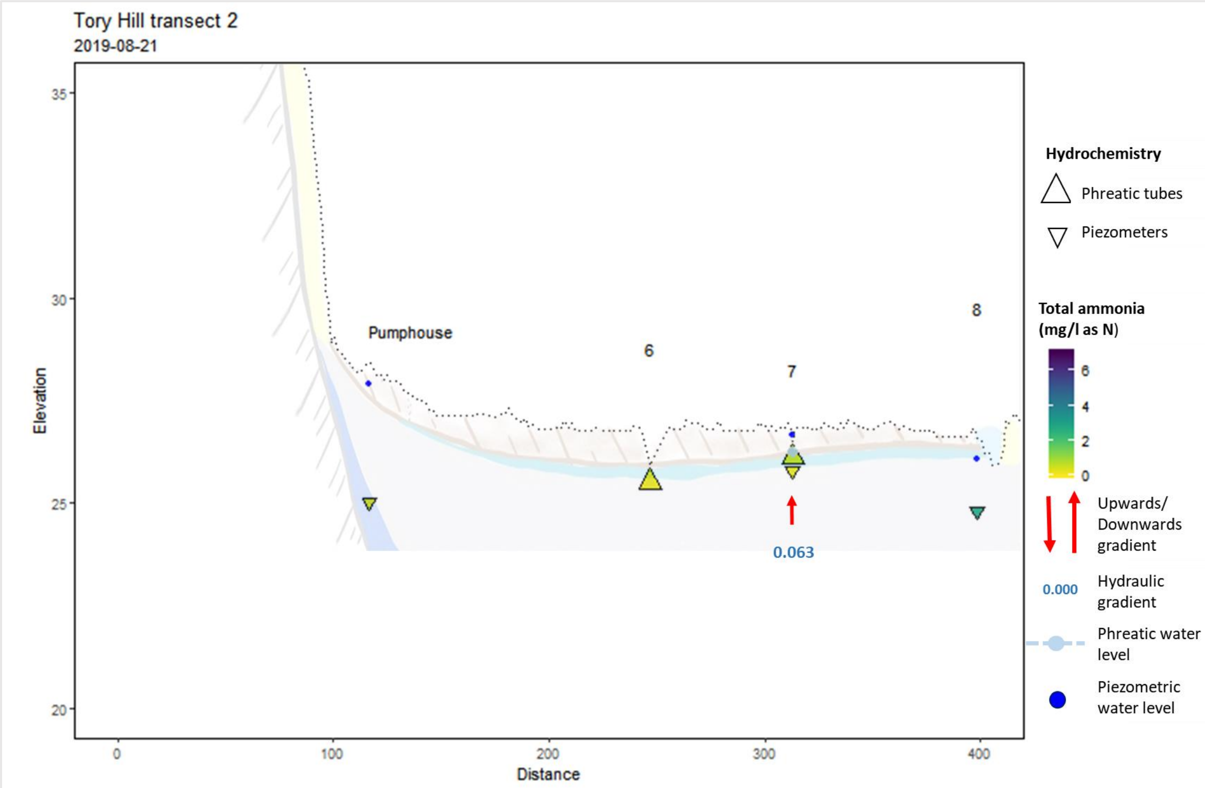


Figure 9.42. Hydrology and total ammonia of Tory Hill transect 1 in August 2019. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

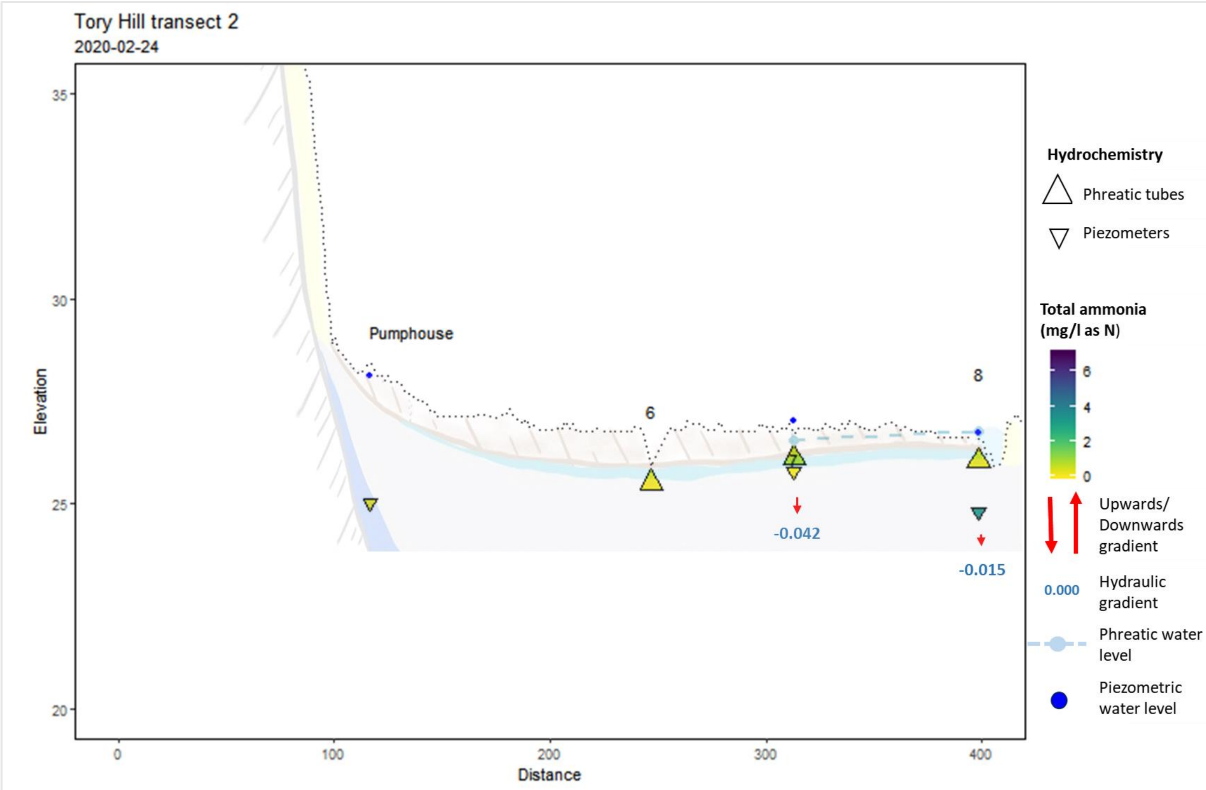


Figure 9.43. Hydrology and total ammonia of Tory Hill transect 2 in February 2020. The phreatic water levels are displayed by a light blue line connected in places where water levels were measured and the piezometric water levels are displayed by dark blue points. The hydrochemistry data is placed at the height were the sample was taken from. The hydraulic gradients flows are shown by red arrows with the number of the vector reported below.

9.4. Conceptual model

9.4.1. Site summary

Tory Hill has a fen area of 0.16 km^2 and is fed by a topographical catchment estimated to be 0.36 km^2

The outlet of the site has a strong draining effect which aims to keep the surrounding fields from flooding. The catchment feeding this outlet is estimated at 13.35 km^2. The site contains many different habitats such fen, marsh and swamp but also woodland and grassland. The fen supports 0.07 km^2 designated fen habitat (Rich fen and flush PF1) which encompasses 42% of the entire site. From the four assessed relevés conducted during the vegetation survey as specified in Section 4.1.3 all failed the fen assessment criteria in Appendix D which means that the fen vegetation of Tory Hill is considered to be in a bad condition.

The fen receives both groundwater and surface water in a hydrological year with a higher proportions of groundwater during the summer. This water enters the fen at the bottom of the steep limestone escapement in the west, presumably via discrete points. Discharge via an artificial outlet has a great effect on the water table of the fen as both the phreatic tube and the piezometer display water level fluctuations that follow the trend of the outlet stage.

Like the other fens studied in this research, Tory Hill also displays internal nutrient accumulation and cycling between peat and underlying substrate layers. High concentrations of DRP in the borehole and the piezometers might be indicative of similarity of the substrate in which they are located. Higher concentrations of TP in the phreatic layer is reflective of active breakdown of peat in an oxidized environment as well as build up by internal wetland cycling. Lower concentrations of chloride and silica are proof that Tory Hill receives a relative large proportion of surface water.

9.4.2. Conceptual model

A conceptual box model is displayed in Figure 9.44, showing the water balance for the catchment feeding the fen and the outlet, surface water level fluctuation and median nutrient concentrations in the fen and its catchment.

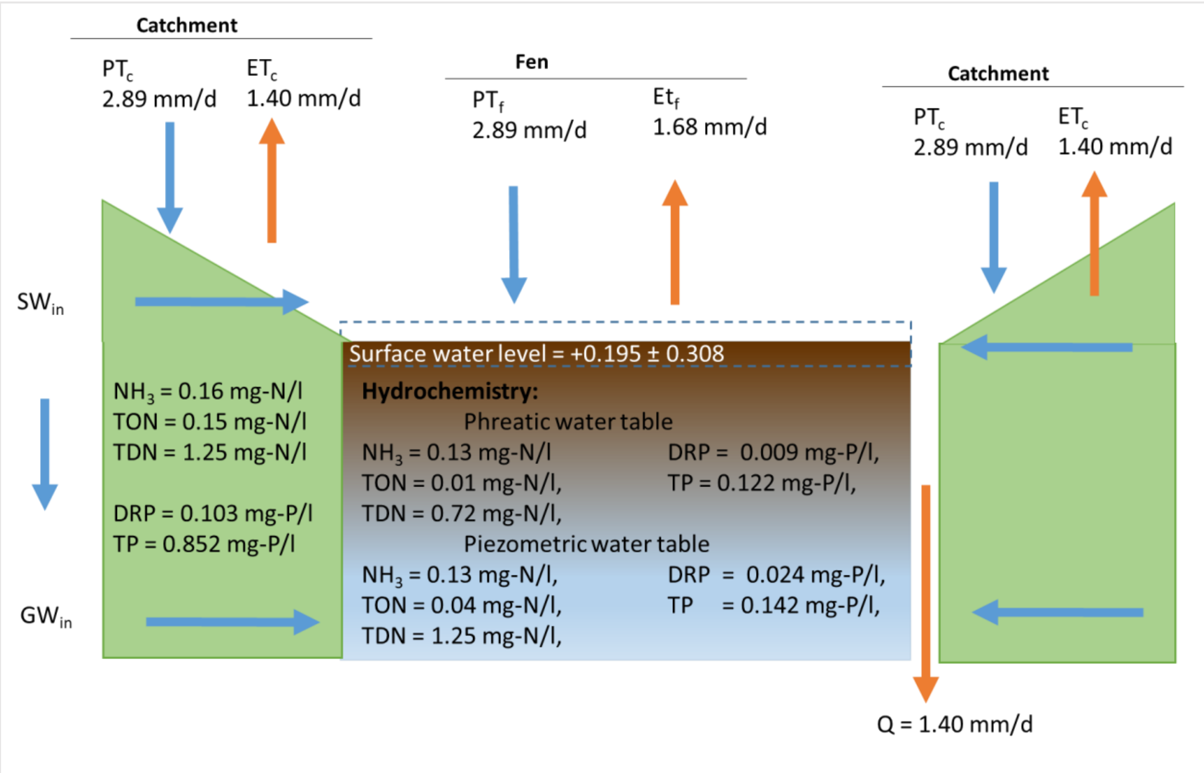


Figure 9.44. Conceptual box model of Tory Hill displaying the water balance, surface water level fluctuation and median nutrient concentrations in the fen and its catchment.

10. Environmental supporting conditions for Irish calcareous fens

Chapter 5 to 9 aimed to investigate and assign hydrological and hydrochemical controls that supports the present site specific habitat, which included a wide array of habitat such as fen, bog and swamp as well as scrub and woodland. This chapter aims to focus on the gathered knowledge of all research sites specifically collected in the fen habitats. These are (as specified under the EU Habitats Directive) 7230 Alkaline fens, 7210 Calcareous fens with Cladium mariscus and species of the Caricion davallianae and 7140 Transition mires and quaking bogs. Through the thesis and this chapter these habitats are identified by their Fossitt Habitat scheme equivalents as specified in Table 2.1. Utilising habitat quality assessments on these fen habitats, controls are investigated that support healthy (good) fen vegetation in the following sections. An assessment of the relative value of each of the study fen areas is reported in chapter 11.

10.1. Habitat quality assessment

The fen habitats Rich fen and flush (PF1), Poor fen and flush (PF2) and Transition mire and quaking bog (PF3) were assessed by the method specified in Section 4.1.3 and subsequently divided into habitats that either passed or failed the fen assessment criteria in Appendix D, based on the vegetation surveys. The habitats that passed the criteria are named 'good' fen habitat whereas the failed habitats are called 'poor' throughout this chapter. Sample locations BM12, BM42 and BM162 (in Ballymore) were excluded from this assessment since no vegetation survey was executed in these areas.

Under this categorization, updated NMDS plots were generated as seen in Figures 10.1, using all data from the research sites taken together. This integration resulted in an IVC (ecological) data set comprising 43 relevés and 157 species. Again, the environmental data sets (ENV) consisted of vegetation type cover (%) and Fossitt habitat codes.

The NMDS plot in Figure 10.1 was generated with the environmental variables of vegetation cover (%) and the presence of the Fossitt habitats displayed as vectors on top of the fen quality scatterplot. The reported stress after 100 randomised runs was 0.189. The stress value represents the difference between distance in the reduced dimension compared to the complete multidimensional space. The rule of thumb for reported stress is that values reported below 0.2 are deemed passable.

On this plot a cluster of good fen habitat (in green) is correlated against different parameters in varying strengths along the negative NMDS1 and NMDS2 axes. Both the PF3 and the mosaic habitat PF1/PF3 are associated with a higher cover of surface water and bryophytes. A select few sites, all surveyed in Ballymore, have the strongest connections with this vegetation composition.

These sites also seem strongly associated with high abundances of *Scorpidium scorpiodes* and *Menyantes trifoliate.* The species *Equisetum fluviatile* and *Carex diandra* have moderate association.

Another smaller cluster of good relevés can be seen positively correlated on the NMDS1 axis. This cluster is both moderately correlated with the habitats FS1 and PF2. The FS1 habitat seems to be associated with vegetation height which means that higher stands of fen vegetation can be expected here. Furthermore, higher abundances of *Juncus subnodulosus* are somewhat associated with this habitat since it is also positively correlated on the NMDS1 axis. However, it is noted that some poor relevés (in red) are associated with this habitat as well such as Pollardstown sites PA34 and PD27. Habitat PF2 is associated with high abundance of species *Schoenus Nigrigans* and *Depranoclidus cossonii recolvens.* This habitat is further associated with a high cover of sedges.

Overall, poor relevés have very weak correlations with any Fossit habitat or specific vegetation. However, the abundance of *Molinia cearulea* and *Juncus Subnodulosus* seemed to be higher in some of these relevés.

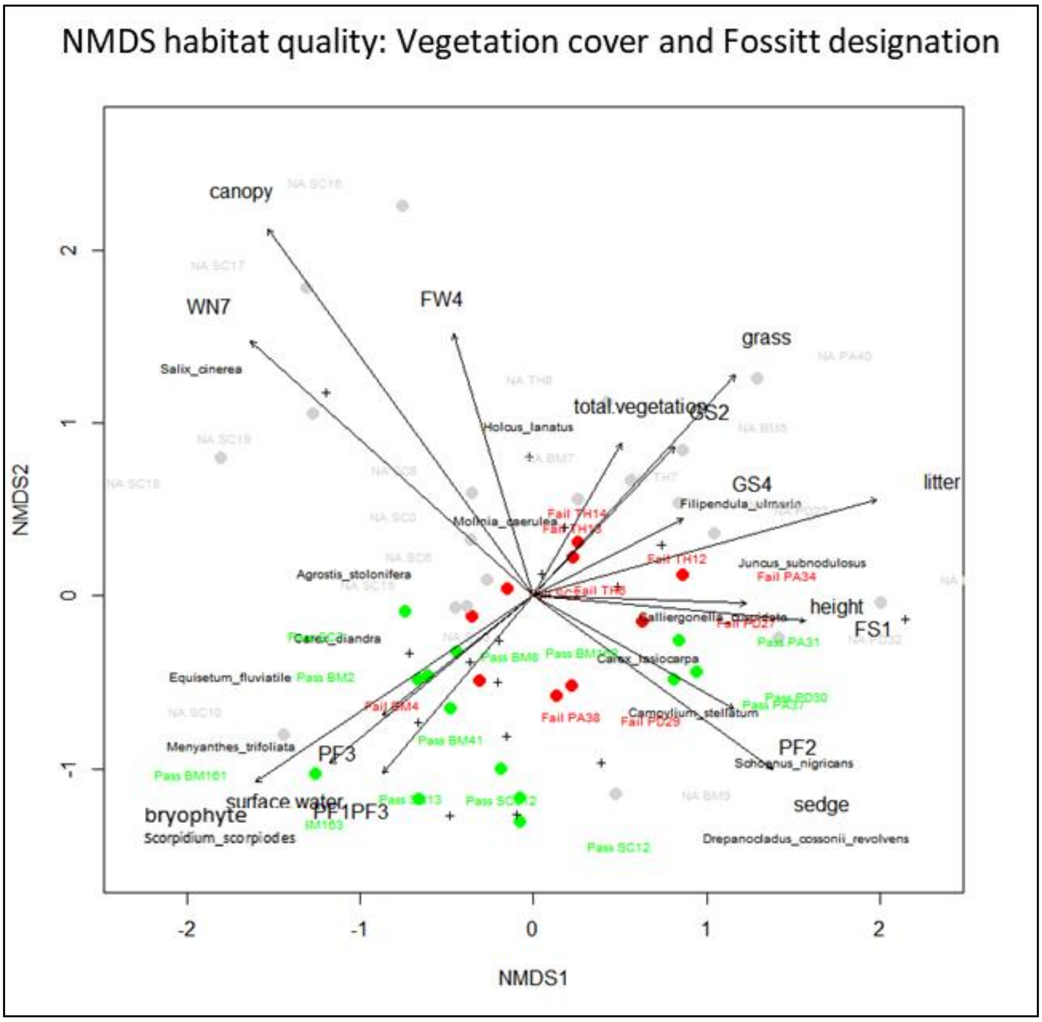


Figure 10.1. Multidimensional Scaling ordination of dimensions 1 and 2 with vegetation cover and Fossitt habitats plotted as vectors (max p-value = 0.2) in good, poor or non-fen (NA) habitats. The phreatic water sampling locations

with their specific quality are shown in green for good, red for poor and grey for non-fen (NA) habitats. The names of the species with the highest abundances (10%) are also plotted.

10.2. Hydrological supporting conditions fen vegetation

10.2.1. Phreatic and piezometric water levels

Using the new fen quality groupings (good or poor), boxplots were generated with phreatic as well as piezometric water levels in Figures 10.2 and 10.3. Again, these figures used the integrated dataset of all fens. The division of these habitats in the different research sites is presented in Tables 10.1 and 10.2. The quality groupings poor or good are shown for habitats PF1, PF2 and PF3. Any data that did not have the Fossitt habitat designation for fens were grouped together under NA.

Table 10.1. Phreatic water level data used per site in Figure 10.2

	BM	PA	PD	SC	TH
PF1poor					40
PF1good				29	
PF2poor		21	23		
PF2good		9	6		
PF3poor	9			10	
PF3good	55			8	
NA	69	22	29	114	20

A clear difference in the behaviour of the phreatic water level can be seen between the poor and good versions of PF1 (Figure 10.2). The poor habitat had the lowest median water level of all displayed habitats of 0.175 m below ground level. It has to be noted that all of the data for the poor sites were recorded in Tory Hill. PF1 good has a much higher phreatic water level (median of 0.231 m above ground level) and was solely measured in Scragh Bog. For the PF2 and PF3 habitats, both good versions have slightly higher median levels than their poor counterparts with 0.016 and 0.089 mAGL respectively. All PF2 habitats (good and poor) were recorded on Pollardstown fen, whilst the PF3 habitats were found on both Ballymore and Scragh Bog.

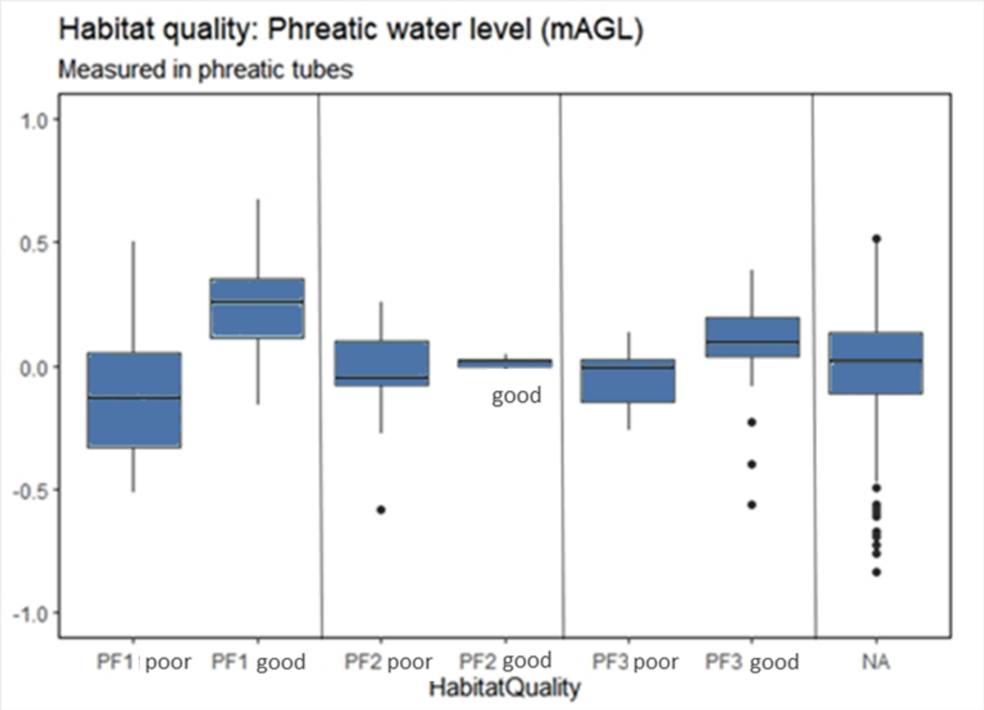


Figure 10.2. Phreatic water level in metres above ground level in good, poor or non-fen (NA) habitats.

Table 10.2. Piezometric water level data used per site in Figure 10.3

	BM	PA	PD	SC	TH
PF1poor					40
PF1good				27	
PF2poor		32	35		
PF2good		21	10		
PF3poor	10			10	
PF3good	55			10	
NA	77	21	49	103	21

The differences in the piezometric water levels, as measured approximately 1 m below ground level, are less clear between the different habitat qualities (Figure 10.3). This is to be expected as it is the phreatic layer that has a direct impact on the fen vegetation, whereas the link between the piezometric pressure and the vegetation in the phreatic layer is more indirect.

It is notable that the medians of all good fen habitats have lower piezometric levels and compared to the relative higher surface water elevation of the good quality fen habitat in Figure 10.2 this implies that downward gradients are more common in the areas of good fen vegetation. As discussed before, the hypothesis is that fens have a nutrient cycling mechanism incorporating this downwards gradient. It therefore seems that the areas with good habitats can move the excess nutrients to the substrate below at a more advantageous rate.

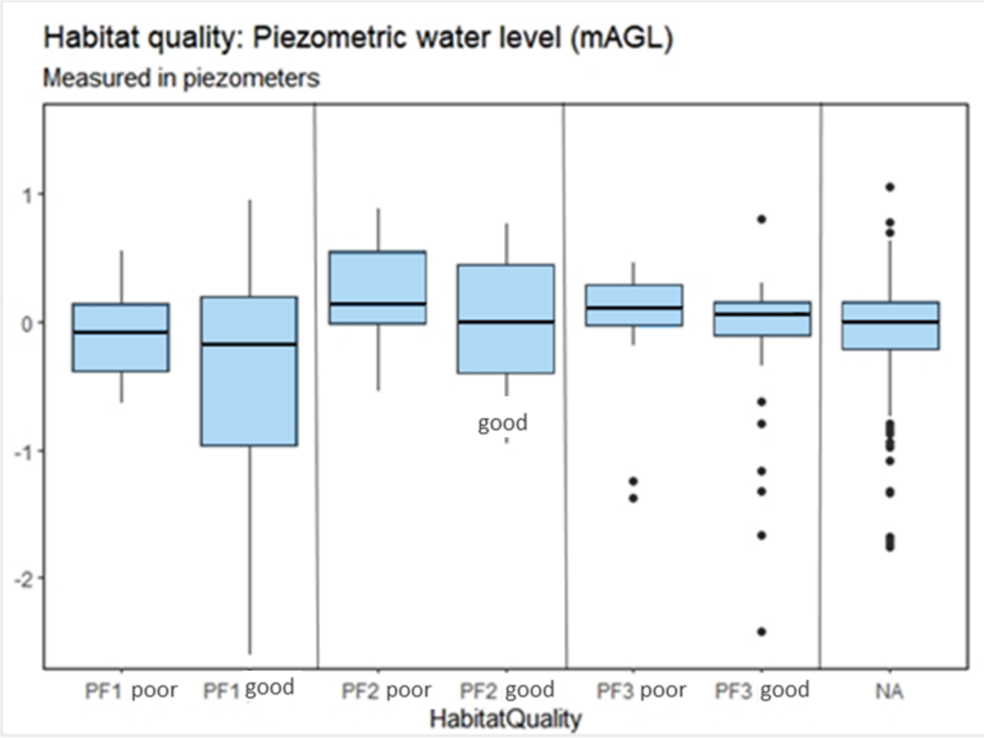


Figure 10.3. Piezometric water level in metres above ground level in good, poor or non-fen (Other) habitats.

To further investigate the influence of hydrology on the quality of the vegetation, frequency duration plots measured in the phreatic tubes situated in different fen habitat qualities are presented in Figure 10.4. These plots were generated from the logged water levels between October 2018 and November 2020. Most of the curves presented were measured in good quality fen habitat and show minimal change in levels. Most of levels were also recorded above ground level with some exceptions for <90% of the hydrological year were the level was found 0.1 m below ground level. Highest levels were recorded in habitat PF1 with some especially high elevations up to 0.73 m for <16% of the year. These 'flooding' levels were also seen in the PF1 poor quality, however here the levels change drastically and are found below ground elevation for most of the year (70%) with levels down to 0.43 mBGL. From this can be concluded that the overall hydrological controls for good quality fen vegetation seems to be reflected in phreatic water levels above ground level with minimal level changes for most of the hydrological year.

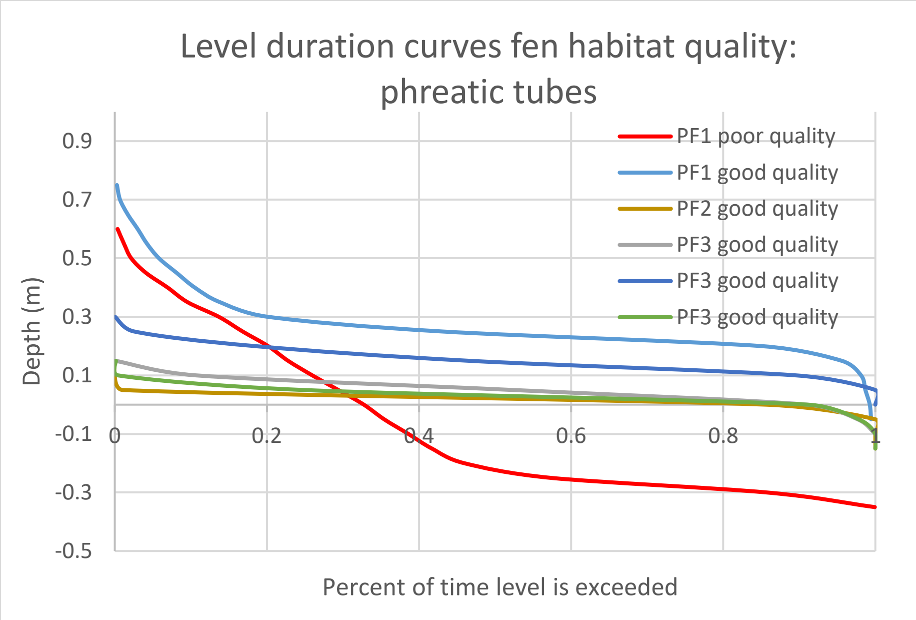


Figure 10.4. Phreatic level duration curves recorded in good and poor fen habitat. The negative numbers are water levels below groundlevel.

Additionally, the maximum hydraulic gradient change in the habitat groupings is presented in Table 10.3. Both the maximum seasonal change between August 2019 and February 2020 as well as the maximum change in the hydraulic gradient time series were observed. The numbers (*n*) of locations from which the data was observed is also displayed. It seems that relatively high gradient fluctuations occurred for the good quality PF3 habitats. PF2 habitats don't have big differences either in poor or good habitats. A significant difference is observed in PF1 habitats though, where the poor quality sites show much higher gradient fluctuations (0.528 seasonal change) than the good sites (0.286 seasonal change). It therefore seems that PF1 vegetation does not tolerate large differences in downward or upward gradient fluctuations.

Table 10.3. Maximum hydraulic gradient change in good, poor or non-fen (NA) habitats. The seasonal change of the hydraulic gradient was measured at two different points (August 2019 and February 2020). The time-series gradient was measured between October 2018 and February 2020.

Habitat	**Seasonal change (February - August)**		**Hydraulic gradient time-series**	
	n	*Maximum change*	*n*	*Maximum change*
NA	27	1.702	9	0.72
PF1 poor	3	0.528	1	1.25
PF1 good	3	0.286		
PF2 poor	6	0.128		
PF2 good	2	0.065	1	0.103
PF3 poor	1	0.060		
PF3 good	7	0.402	3	0.248

10.2.2. Electrical conductivity and pH

Boxplots in Figures 10.5 and 10.6 again used the full fen dataset to present habitat quality groupings. The divisions of the data points in these groupings are presented in Tables 10.4 and 10.5 to show in which fen they were measured.

Table 10.3. Phreatic water level data used per site in Figure 10.4 and 10.5

	BM	PA	PD	SC	TH
PF1poor					20
PF1good				18	
PF2poor		11	12		
PF2good		7	5		
PF3poor	9			11	
PF3good	55			8	
NA	41	13	17	68	12

Table 10.4. Piezometric water level data used per site in Figure 10.4 and 10.5

	BM	PA	PD	SC	TH
PF1poor					16
PF1good				17	
PF2poor		17	18		
PF2good		10	7		
PF3poor	7			9	
PF3good	35			6	
NA	46	12	29	61	12

The EC in surface water of non-fen habitats seems comparable to poor PF2 and PF3 habitats (Figure 10.5). The good quality habitats have higher EC which implies that these habitats are sustained by relative higher groundwater contributions. PF1 is an exception to this. EC recorded solely in the habitats of Scragh implies that the phreatic layer of this site is fed by a much higher surface water contribution than the other sites. The poor quality PF1, PF2, PF3 habitats seem to have lower EC in the piezometric measurement level which could mean that the substrate layer is dominated by a greater mixture of groundwater and surface water. However, it has to be noted that the EC is frequently variable in each surrounding catchment, depending on the routes of water feeding the fen, which in turn can influence the substrate layers of the fen.

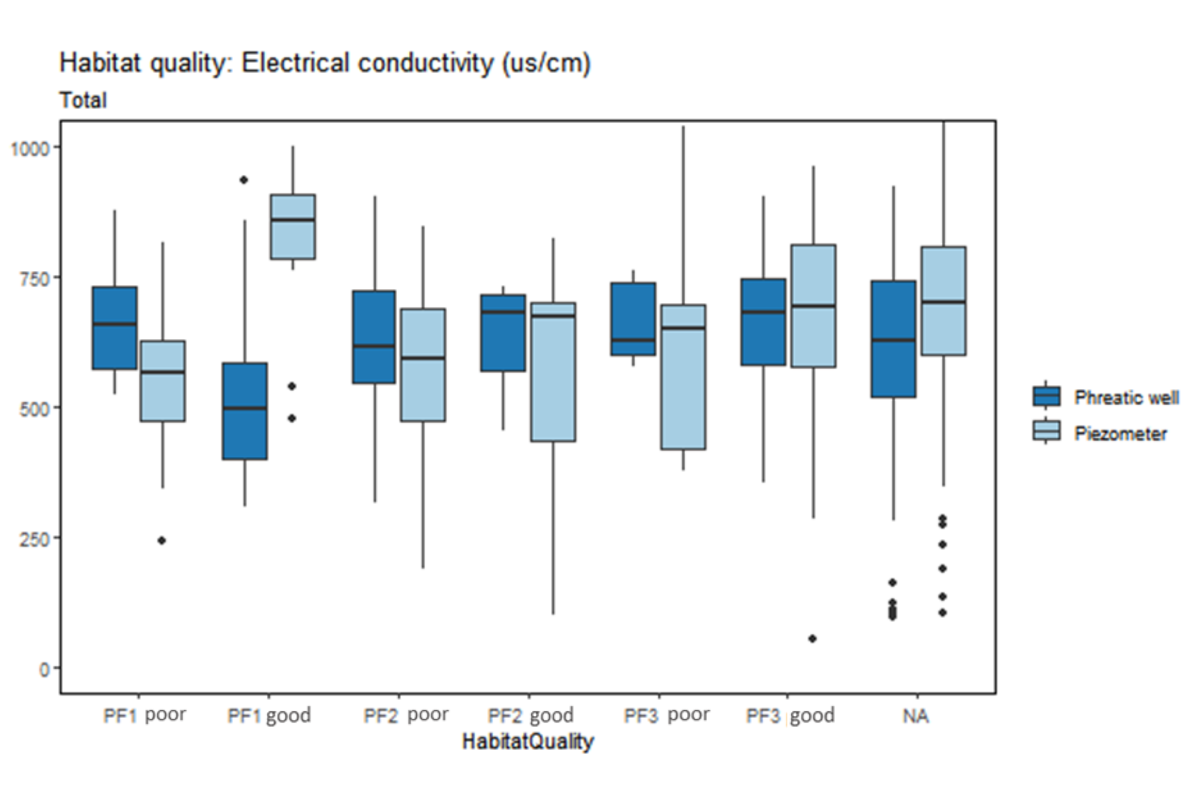


Figure 10.5. Electrical conductivity (µm/cm) in the phreatic tubes and piezometers of good, poor or non-fen (NA) habitats.

Figure 10.6 suggests that the pH in both phreatic and piezometric water levels is higher in the poorer quality PF1 and PF2 habitats. The median pH of the phreatic layer in the poor PF3 habitat is lower than in the good quality sites.

It is notable that in all the 'good' habitat sites, there is little change in pH between that at the lower piezometric level and the phreatic level which may indicate little hydrochemical influence of the intervening substrate. In the poor habitats, there appears to be a consistent reduction in alkalinity/ pH which may indicate a greater influence from direct rainfall or of the effect of intervening acidic peat layers.

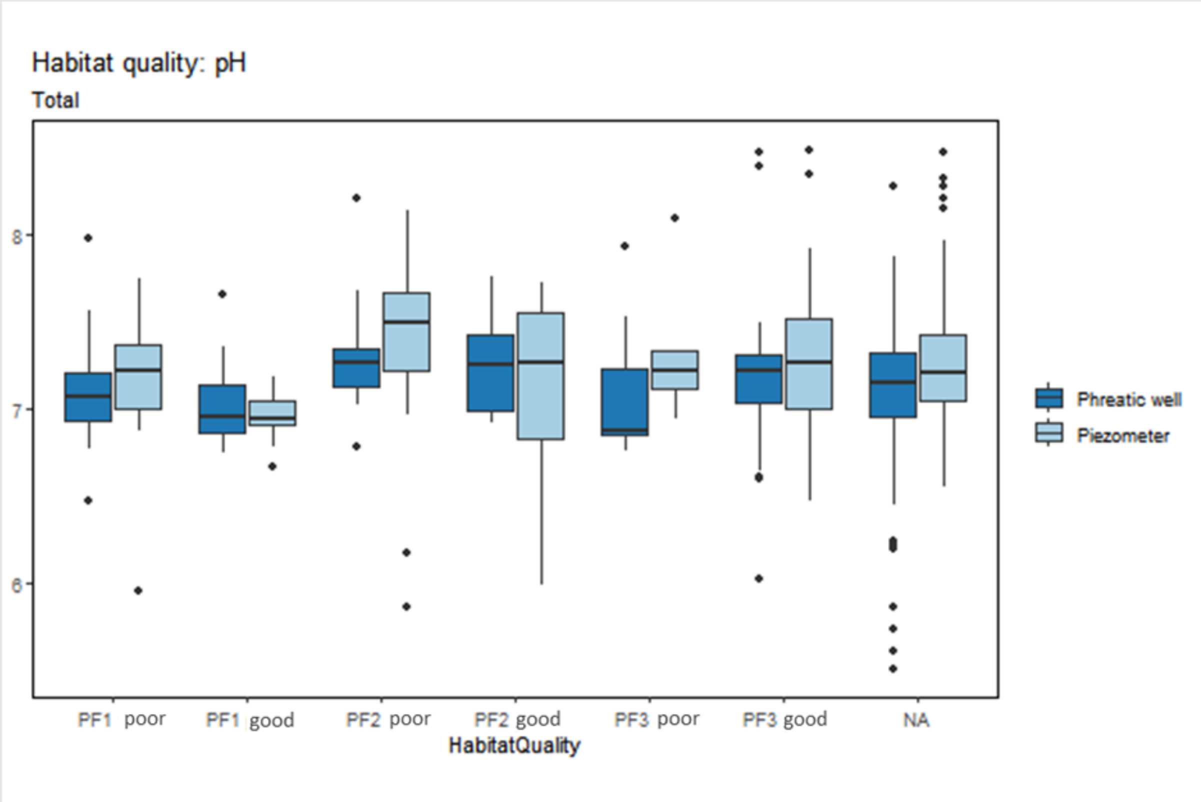


Figure 10.6. pH in the phreatic tubes and piezometers of good, poor or non-fen (NA) habitats.

10.3. Hydrochemical supporting conditions fen vegetation

10.3.1. Nonmetric Multidimensional Scaling ordination

Figure 10.7 displays the NMDS plots generated with the lumped IVC data set of 43 relevés and 157 species. The environmental sets (ENV) and the phreatic hydrochemistry results (mg/l) were measured in the growing season.

Associated hydrochemistry is found neither in habitats with vegetation in good condition nor in bad condition meaning that there is no particular association of nutrients (such as DRP, TP, total oxidised nitrogen and total dissolved nitrogen), sulphate and DOC here.

This means that higher concentrations of these components found in a fen site might be associated with non-fen habitat rather than with designated fen habitats (good and poor). Furthermore, from these plots then it also seems that good quality habitat has a much lower association with high concentrations of before mentioned components. The thus expected low concentrations here are evidence that habitat specific vegetation is much more effective in the uptake, recycling and storage of nutrients then vegetation in non-fen and poor habitats whereas the vegetation in non-fen and poor fen quality is less effective in the before mentioned processes.

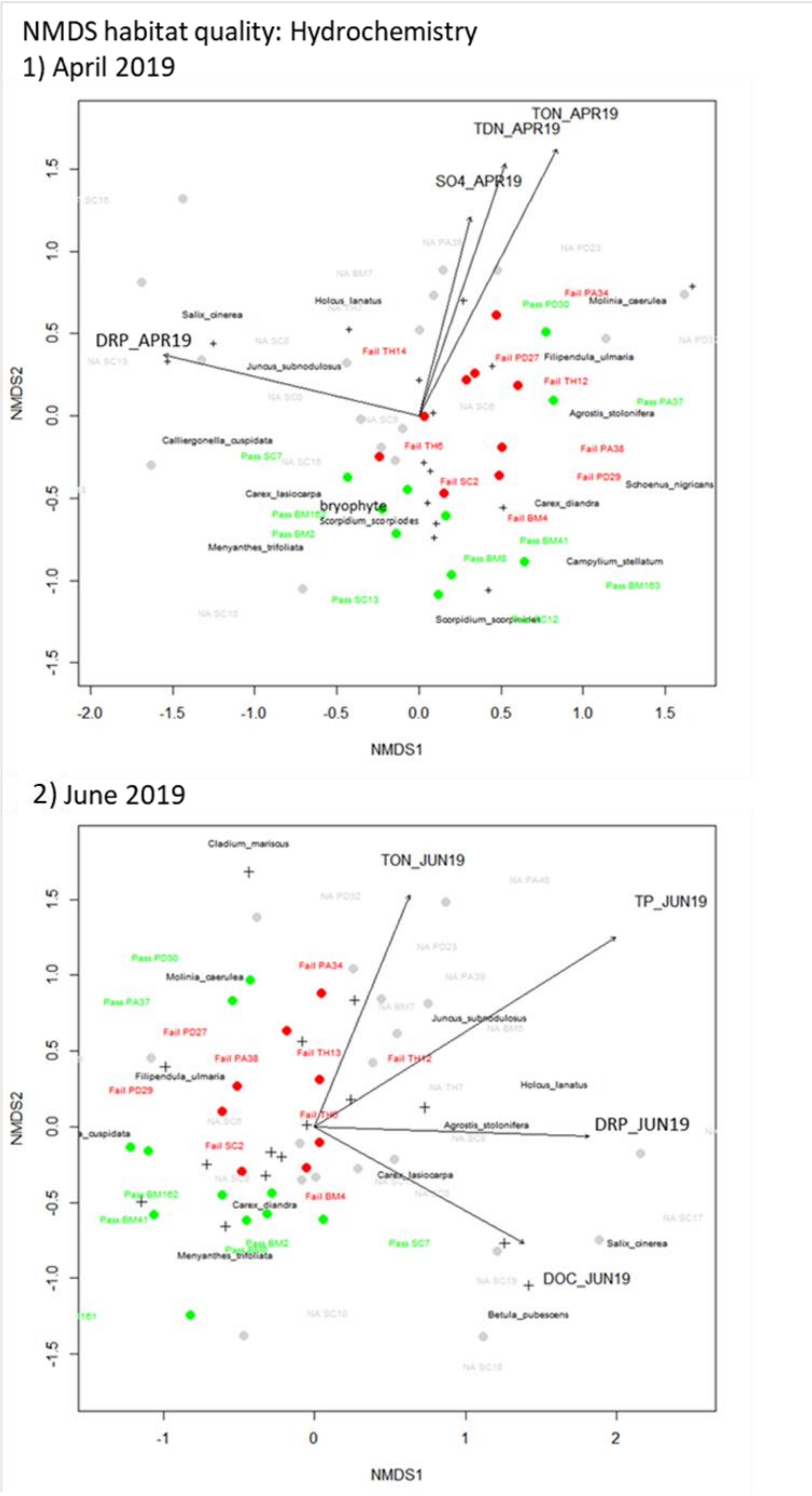


Figure 10.7. Multidimensional Scaling ordination of dimensions 1 and 2 with vegetation cover and Fossitt habitats plotted as vectors (max p-value = 0.2) in good, poor or non-fen (NA) habitats. The phreatic water sampling locations with their specific quality are shown in green for good, red for poor and grey for non-fen (NA) habitats. The names of the species with the highest abundances (10%) are also plotted.

10.3.2. Boxplots hydrochemistry

Boxplots were generated with hydrochemistry data collected in both phreatic tubes and piezometer. These are presented in Figures 10.8 – 10.19. The division of the data points in these grouping are presented in Tables 10.6 and 10.7 to indicate in which fen they were measured.

Table 10.5. Phreatic water level data used per site in Figure 10.7 to10.18

	BM	PA	PD	SC	TH
PF1poor					24
PF1good				24	
PF2poor		20	22		
PF2good		9	7		
PF3poor	10			14	
PF3good	55			9	
NA	59	19	25	98	17

Table 10.6. Piezometric water level data used per site in Figure 10.7 to 10.18

	BM	PA	PD	SC	TH
PF1poor					26
PF1good				25	
PF2poor		29	31		
PF2good		21	10		
PF3poor	11			15	
PF3good	48			11	
NA	68	18	43	90	18

10.3.2.1. Phosphorus

There is little difference in the DRP concentrations in the phreatic layers between the different habitat qualities (Figure 10.7). It therefore seems that free DRP in the water column near the vegetation does not have a direct link with fen vegetation quality, at the range of concentration levels found in the fens. However, consistent with the nutrient cycling hypothesis, it does seem higher concentrations of DRP were found in the piezometric layers of the good quality ('good') PF1 and PF3 habitats in comparison with the poor habitats. This implies that those 'good' habitats, which were also found to exhibit overall stronger downwards gradients, have a stronger nutrient flux from the phreatic layer to the underlying substrate. Another reason for this phenomenon could be that the vegetation more actively growing, creating more biomass which provides higher amounts of organic material to break down. Furthermore, the in the phreatic zone there is expected to be an higher concentration of oxygen which means biochemical reaction can act to break down the nutrients much more effectively here. This is especially expected in Tory Hill where the water table seems to be below the ground level for a large portion of the hydrological year and experiences a lot of fluctuations in water levels. Overall, concentrations found in the boreholes are found with much higher concentrations of DRP than in all the (non-fen, poor and good) phreatic tubes. This means that generally higher concentration in the surrounding catchment of the fens are not necessarily reflected in the surface water concentrations in the fens. Hence, this suggests that the substrates and vegetation in the fens are having a strong filtering action notwithstanding any fluctuating hydraulic gradients.

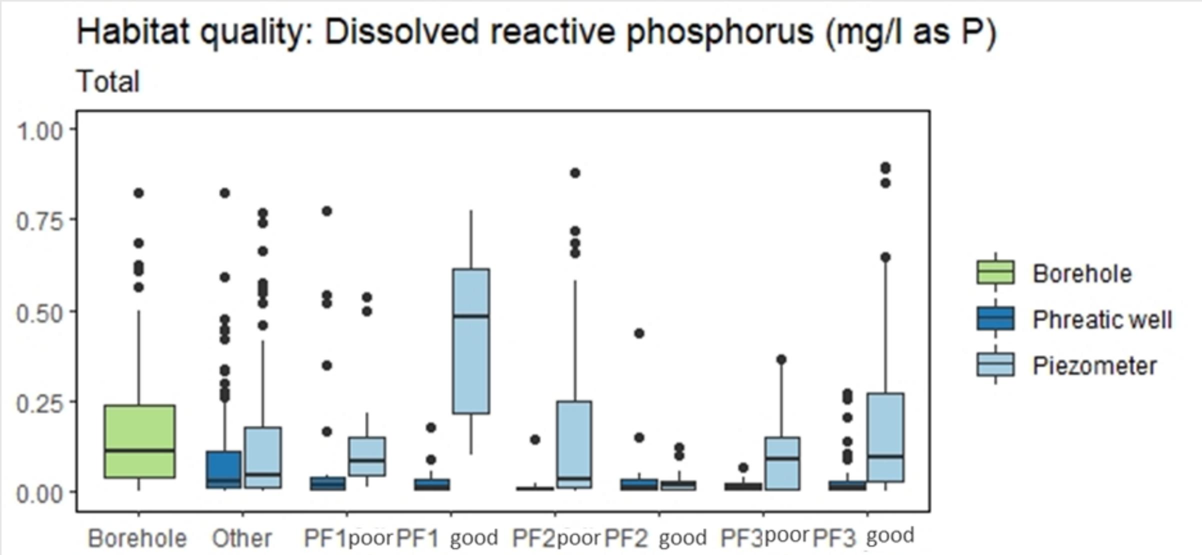


Figure 10.8. Dissolved reactive phosphorus in mg-P/l in the phreatic tubes and piezometers of good, poor or non-fen (NA) habitats.

The TP concentrations are more varied in the good and poor quality habitats as shown on Figure 10.9, making it more difficult to distinguish whether this parameter is having direct control on the fen vegetation. The poor quality (poor) PF1 habitat stands out as having the most significantly high TP concentrations in the phreatic tubes compared to the PF1 good sites. This may be due to there being a higher rate of peat soil humification found in the poor habitat sites which might also have affected the fen vegetation quality. In particular, brown mosses had a significant lower coverage in these relevés which also supported some negative indicator species such as *Holcus lanatus* and higher coverages of *Phragmites australis*. Interestingly, the higher TP concentrations found in the boreholes seem to match surface water concentrations measured in non-fen and PF1 poor (solely measured in Tory Hill) as well as PF2 good (measured in Pollardstown fen) habitats. This may reflect the different nature of the substrate at these locations clayey gravel and metamorphic till respectively.

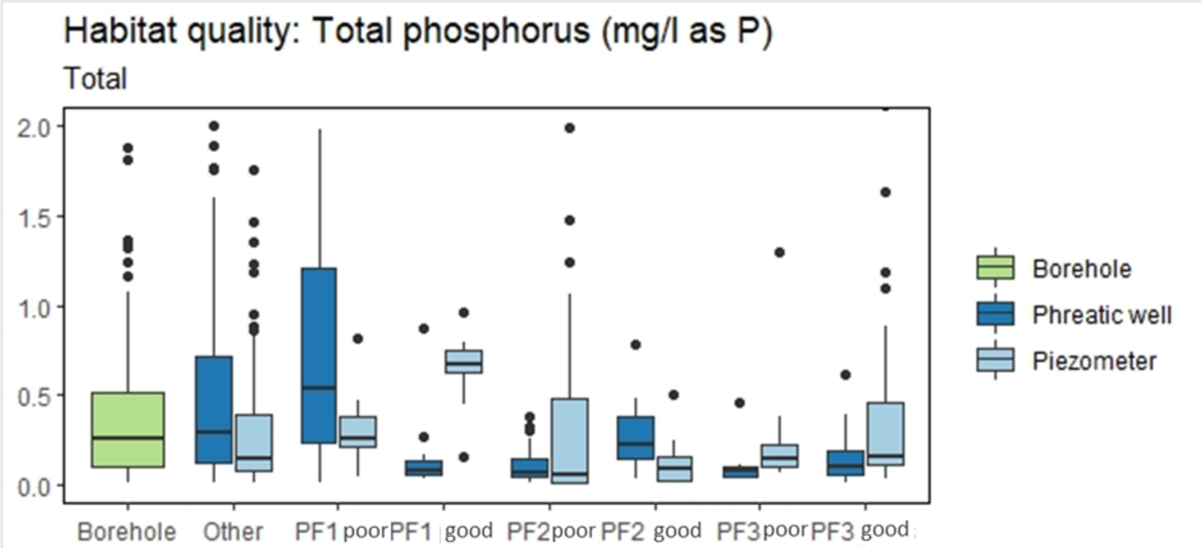


Figure 10.9. Total phosphorus in mg-P/l in the phreatic tubes and piezometers of good, poor or non-fen (NA) habitats.

10.3.2.2. Nitrogen

There is very little difference in the total ammonia concentrations in the phreatic layers between the different habitat qualities although it seems that the good habitat types have somewhat higher values than their poor counterparts. PF1 good was found with the highest concentrations which implies that the vegetation can tolerate them. These high concentrations also imply that this habitat was found in a reduced environment. In contrast, habitats with lower ammonia concentrations might be evidence for the conversion into another form of N at a faster rate, i.e. into nitrate which, as specified before, seemed to happen in a more oxidised environment in conjunction with N-uptake by the vegetation. Similar to elevated DRP concentrations at the piezometric level of the good fen habitats, higher total ammonia is found for this depth in the high quality (good) PF1, PF2 and PF3 habitats. This concentration difference is more obvious in PF1 and PF2. This again points to higher rates of nutrient breakdown and/or dispersion/adsorption in the soils/sediments below the phreatic water table.

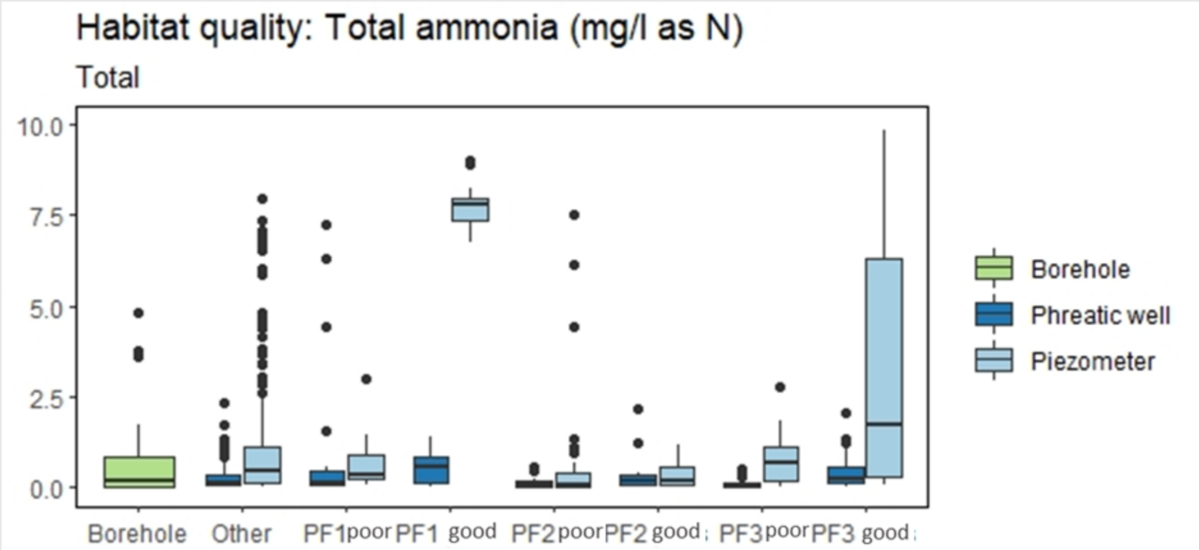


Figure 10.10. Total ammonia in in mg-N/l in the phreatic tubes and piezometers of good, poor or non-fen (NA) habitats.

Most nitrite concentrations were measured below the limit of detection, however some high outliers (up to 0.3 mg-N/l) can be found in the piezometers of PF3 good (Figure 10.11). This points to some temporary denitrification occurring in the piezometric zone. This organic matter decomposition could be important in PF3 habitat which is known for having lower maturity of humification in the peat habitats as larger parts of the peat column contain lower percentages of organic material making the habitat unstable underfoot (hence the name Transition mire and quaking bog). Higher rates of breakdown would signify a higher rate of peat accumulation in the layers below the phreatic zone which may be important for the integrity of the fen.

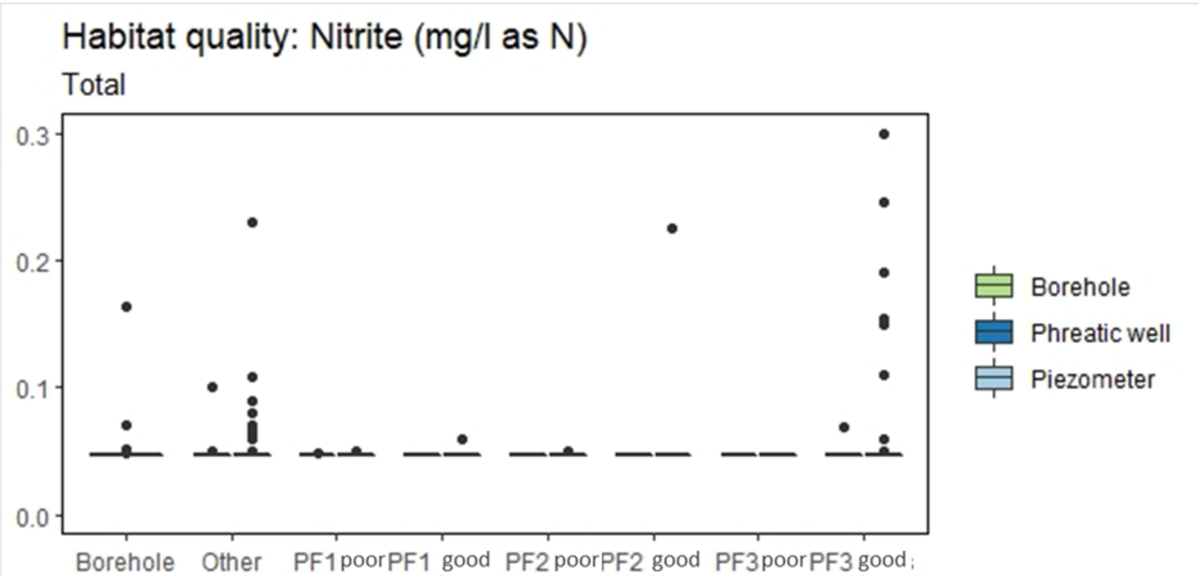


Figure 10.11. Nitrite in in mg-N/l in the phreatic tubes and piezometers of good, poor or non-fen (NA) habitats.

As seen in Figure 10.12 the total oxidised nitrogen is found with much higher concentrations in the boreholes surrounding the fen than in the phreatic tubes and most of the piezometers in the fen. It is likely that the higher concentrations found in the PF2 habitats are due to their screen location. The data was solely collected in Pollardstown Fen where the screens of the piezometers were installed in the gravel aquifer thus directly reflecting the higher concentrations found in the catchment.

As for the concentrations measured in the phreatic tubes, total oxidised nitrogen seems to be completely taken up by vegetation in the surface layer. The low concentrations found in the piezometers of habitats PF1 and PF3 (opposed to the higher ammonia concentrations seen in Figure 10.9) imply reducing conditions in the lower sediment/soil layers of the fen.

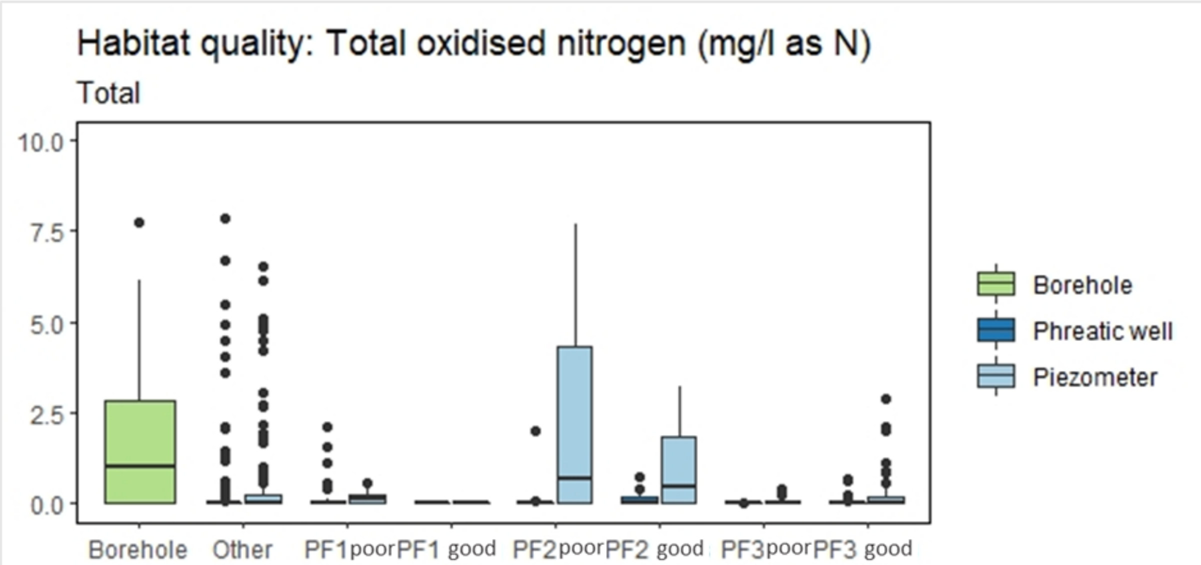


Figure 10.12. Total oxidised nitrogen in mg-N/l in the phreatic tubes and piezometers of good, poor or non-fen (NA) habitats.

The total dissolved nitrogen concentrations are found to be much higher in the piezometers than in the phreatic tubes. However there does not seem to be any correlation to either good or poor quality fen habitats.

The high piezometer concentrations are comparable to the values found in the boreholes although, again it is believed that this is due to internal fen nutrient cycling rather that inflow from the aquifer. PF1 good was found with high elevated values as was seen before in the total ammonia data (Figure 10.10). This data, gathered from Scragh bog, again, suggest that decaying processes are causing organic nitrogen and ammonia to build up over time in the piezometric layer, especially due to anoxic conditions.

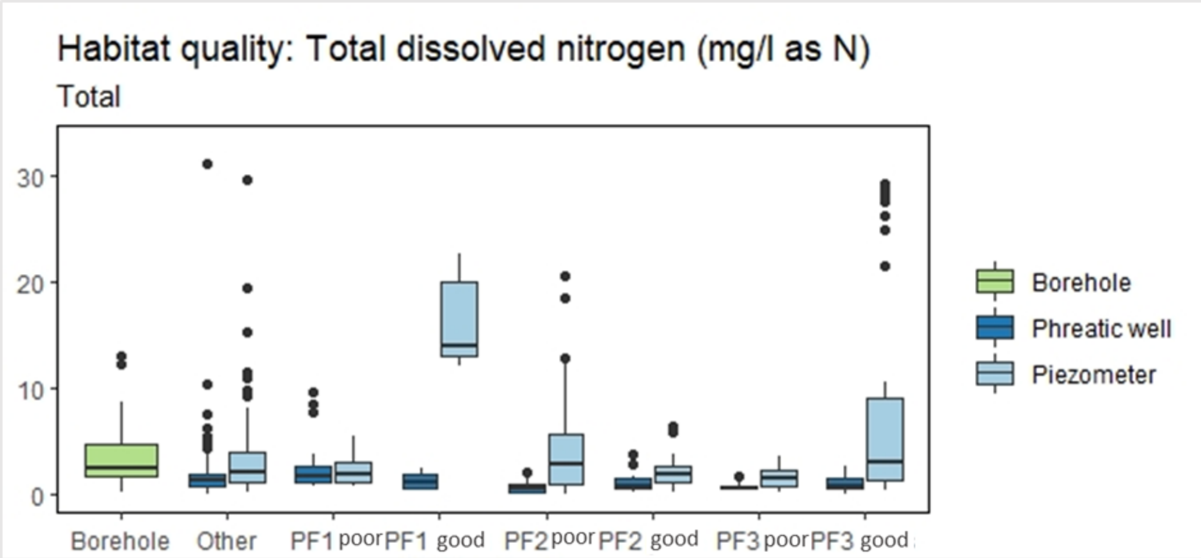


Figure 10.13. Total oxidised nitrogen in mg-N/l in the phreatic tubes and piezometers of good, poor or non-fen (NA) habitats.

10.3.2.3. Other chemistry

The alkalinity concentrations were consistent and do not differ much between the different habitat quality groupings (Figure 10.14) and reflect the earlier pH measurements. Overall low concentrations were found in the piezometric water levels of PF2 compared to PF1. This observation implies that the mineral content of the aquifer feeding the fen has an influence on supporting the different floristic composition of both Poor fen and flush (PF2) and Rich fen and flush (PF1).

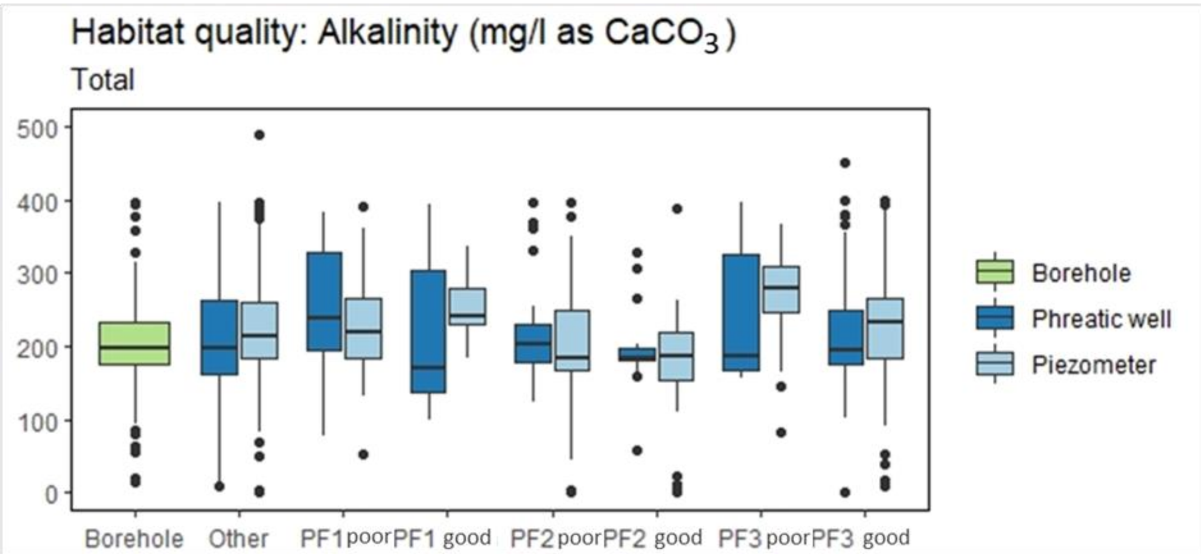


Figure 10.14. Alkalinity in mg/l as $CaCO_3$ in the phreatic tubes and piezometers of good, poor or non-fen (NA) habitats.

The chloride concentration was reported higher in the phreatic tubes of PF1 poor as well as PF3 good and these concentrations also match the values found in the boreholes (Figure 10.15). These differences are probably not showing that different fen habitats need different chloride concentrations in order to thrive but rather that this is site specific (since PF1 poor was solely collected in Tory Hill). However according to the Irish groundwater threshold values for chloride (Government of Ireland, 2010), which is 187.5 mg/l, the reported low values in the fens are probably not a significant issue. Within this threshold, chloride concentrations are not a useful metric for fen conservation.

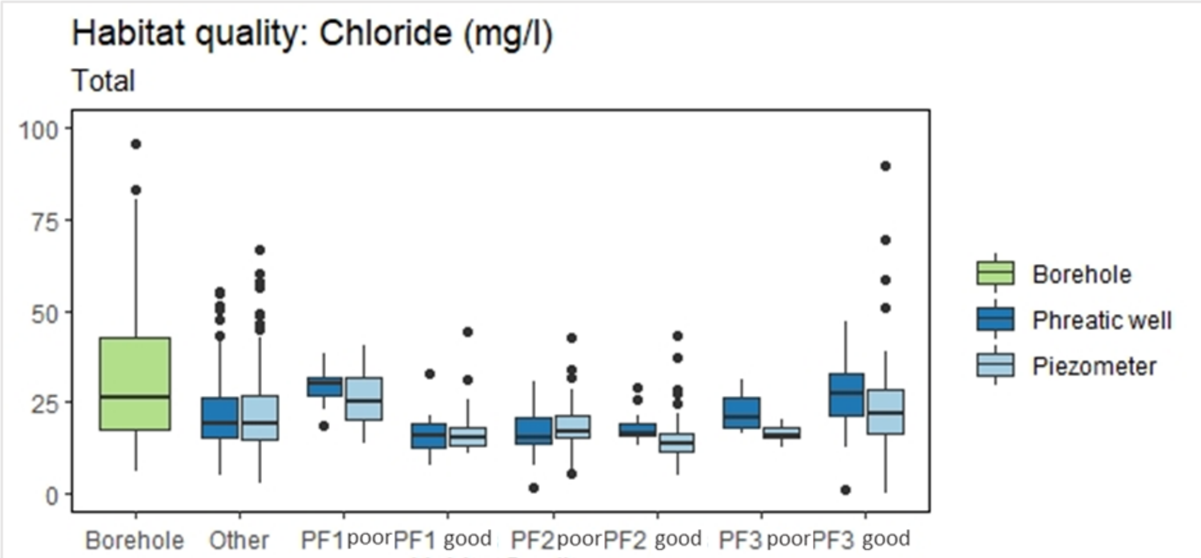


Figure 10.15. Chloride in mg/l in the phreatic tubes and piezometers of good, poor or non-fen (NA) habitats.

PF1 good has high concentrations of silica at the piezometric water level compared to PF1 poor (Figure 10.16). This could however again be specific to the character of the lower sediments/soils per site as PF1 poor sites were all located in Tory Hill whilst the PF1 good sites were all located in Scragh Bog. Interestingly, higher concentrations are found in the 'good' version of PF3 indicating higher groundwater contributions here. However, silica would not appear to be a significant limiting condition for fen sustainability.

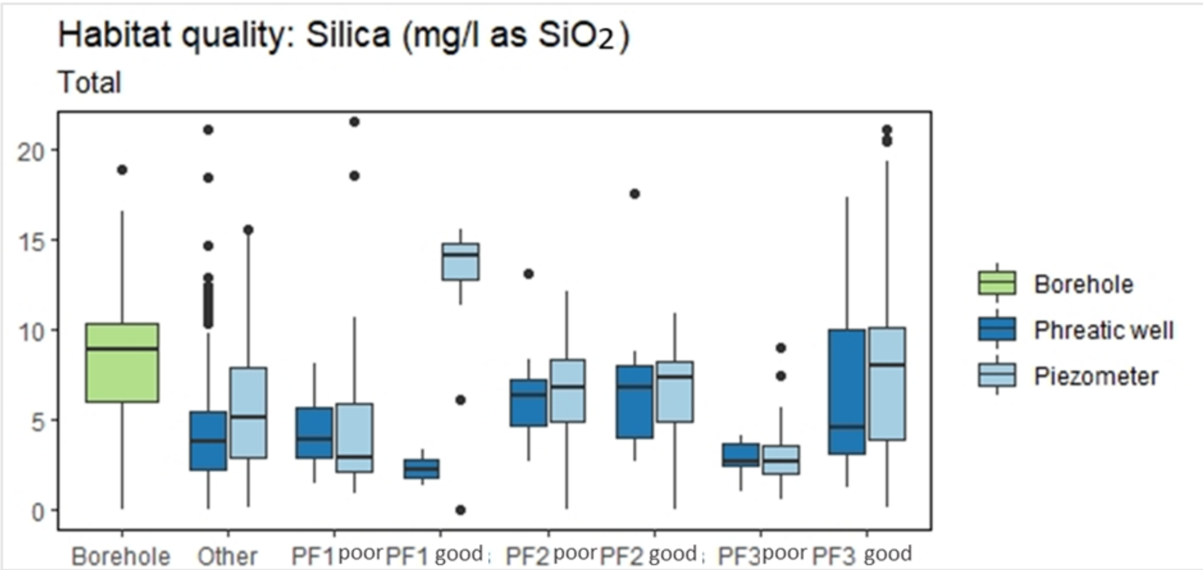


Figure 10.16. Silica in mg/l as SiO_2 in the phreatic tubes and piezometers of good, poor or non-fen (NA) habitats.

Sulphate concentrations were found to be much higher in the poor quality PF1 sites which might have negatively influenced the species composition leading to a 'poor' habitat (Figure 10.17). However, this may also be due to the wider site-specific nature of the data (as all PF1 poor samples were collected in Tory Hill) and therefore be more linked to other environmental controls around the fen. Higher concentrations were also found in the piezometers of PF2 poor compared to its good habitat version. These are also comparable to the concentrations found in the boreholes, measured in the catchment.

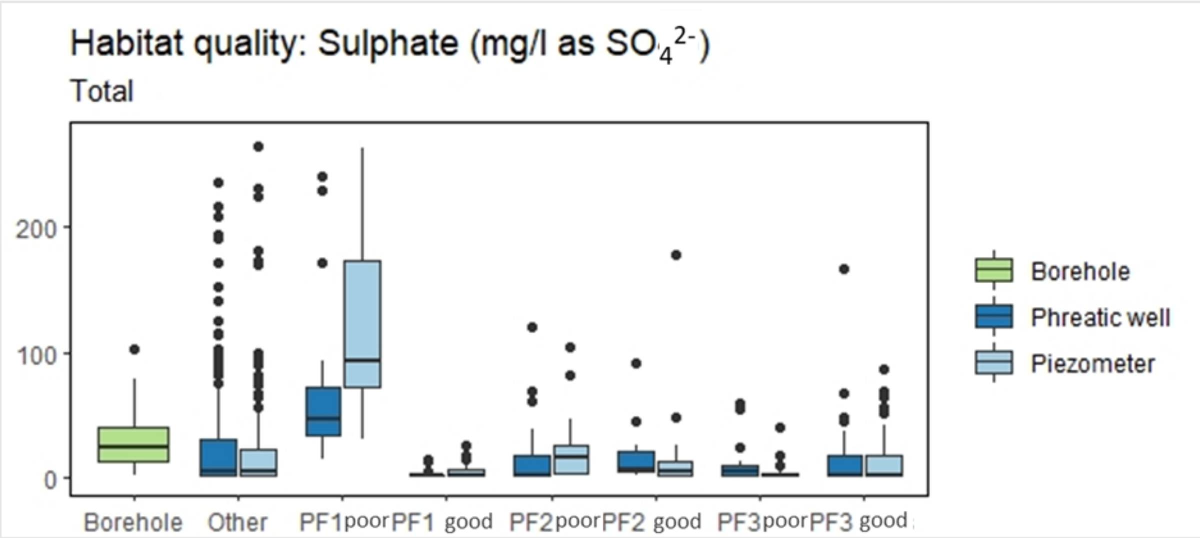


Figure 10.17. Sulphate in mg/l as SO_4^{2-} in the phreatic tubes and piezometers of good, poor or non-fen (NA) habitats.

Figure 10.18 suggests higher rates of organic decomposition in the PF1 habitat (poor and good) as seen from higher DOC concentrations.

Good PF2 and PF3 habitats do not seem to reveal higher rates of decomposition compared to poor quality habitats where elevated concentrations are also found. This implies that these designated habitats could be in poorer condition if decomposition rates were increased. The poor habitats could further be linked to lower water tables (as seen in Figure 10.2) which would support a higher rate of organic breakdown.

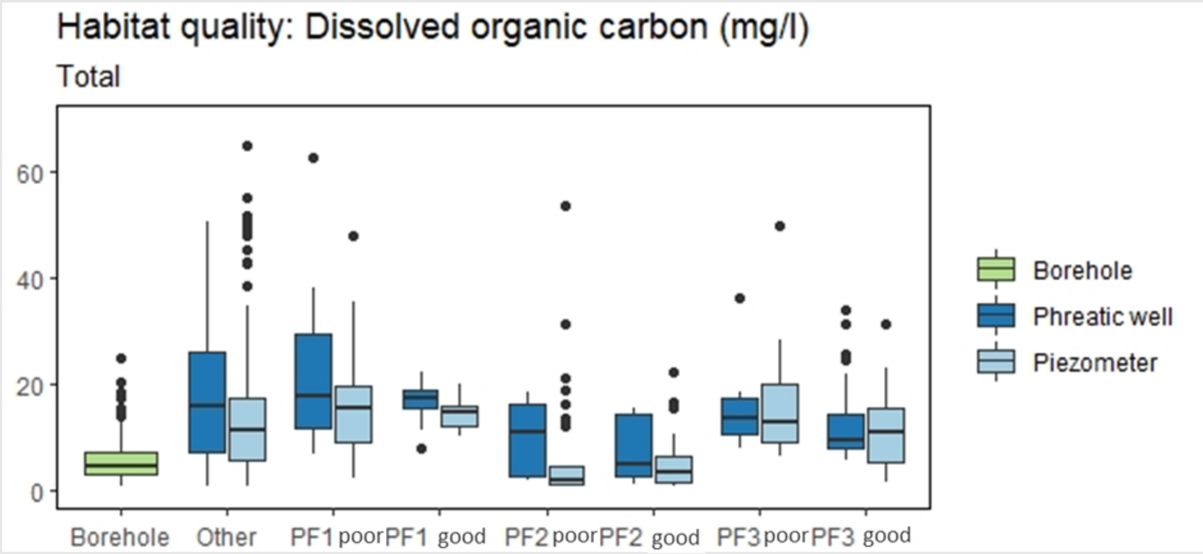


Figure 10.18. Dissolved organic carbon in mg/l in the phreatic tubes and piezometers of good, poor or non-fen (NA) habitats.

Finally, Figure 10.19 shows higher ferrous iron concentrations in the phreatic tubes of PF3poor than in PF3good, which is indicative of slightly more reducing conditions. This condition also implies a higher chance for ferrous iron to oxidise to ferric (3^+) iron due to higher seasonal surface water inputs and thereby releasing bonds to phosphorus allowing organic phosphorus to enter the free water column.

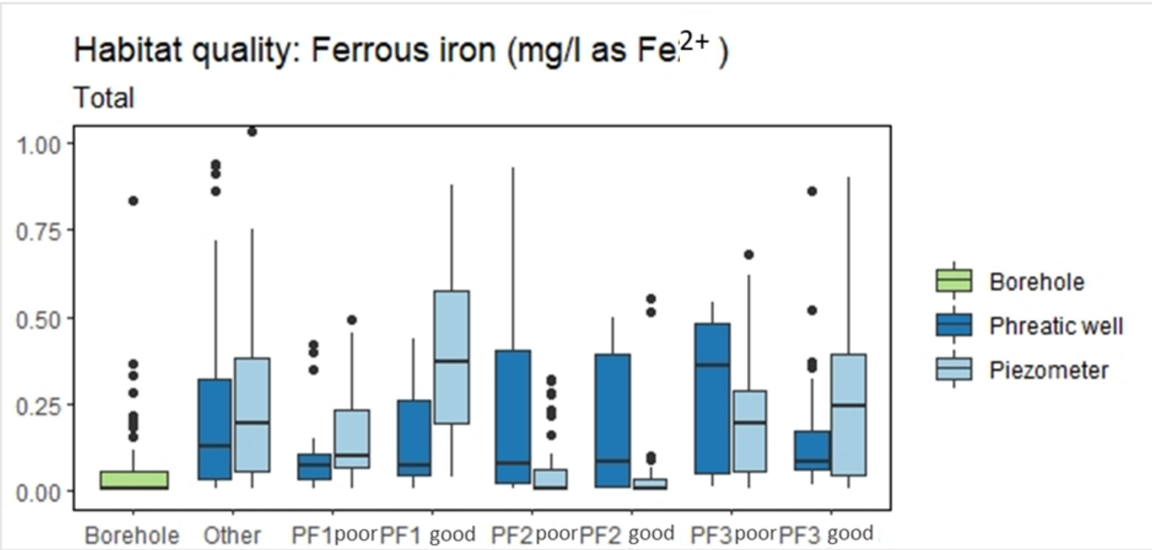


Figure 10.19. Ferrous iron in mg/l as Fe^{2+} in the phreatic tubes and piezometers of good, poor or non-fen (NA) habitats

10.4. Conceptual model

10.4.1. Description

Hydrologically, a fen, being a wetland, arises from an area of retained water on the landscape – a temporary storage. In time, characteristic soils develop within it along with a distinguishing vegetation and ecology. The dynamics of the water supply, its retention and its quality dictate the nature of the wetland. What distinguishes a fen wetland is that the principal source of water is groundwater and its chemistry depends on the surrounding geology through which it passes. However, as the wetland vegetation exists close to the surface of the fen, the soils of the fen itself can modify the chemistry of the water that arrives at the root zone. The hydrogeological characteristics of the soils or substrate of the fen are, therefore, a key control on the growing conditions for the fen vegetation at the surface.

Thus, the proportion of groundwater supply relative to direct rainfall falling on the fen is an important influence on the hydrochemistry of the water sustaining the vegetation/ecology. The corresponding fluctuations in the fen seepages, springs and water table are also a fundamental control on the nature of the vegetation and associated habitats.

A representative conceptual model of lowland calcareous fens in Ireland is shown in Figure 10.20 indicating the diversity of the groundwater pathways supplying the wetland and in particular, the controlling nature of the wetland soils and substrates on modifying that hydrology. The underlying bedrock is typically fractured or fissured Carboniferous limestone. Over the bedrock, glacially derived subsoil including tills and outwash gravels usually provide the topography which hosts the conditions for the fen to develop. Organic soils or peat then accumulate, often on a lacustrine clay, depending on the topography and drainage – the fen vegetation develops on the surface, and at the margins of these wetland soils, influenced by the pathways taken by the water supply. The substrate soils are rarely areally uniform, allowing discrete inflow pathways to persist as the piezometric level of the regional groundwater is above the substrate or the margins of the fen, creating the seepages and springs feeding the fen. As the fen develops, the piezometric level may become above or below the fen surface but will seasonally and climatically fluctuate. Thus, the model may conceive of separate but connected hydrological regimes – the one in the catchment surrounding the fen and feeding the other within the fen itself. The characteristics and dynamics of the connections between the two dictate the sustainability of the supported vegetation. Surplus water from groundwater inflow and direct rainfall on the fen, that is, runoff, is typically routed via natural or artificial drainage to an outlet, discharging to the natural surface water stream network. The morphology of this on-fen surface drainage can also influence the nature of the fen vegetation and its sustainability.

The dynamics between the piezometric surface and the phreatic water table in the fen also play a role in the nutrients cycling utilising the fen vegetation and fluctuating hydraulic gradients. As seen in Figure 10.19 upward flows may bring nutrients to the surface, where the fen vegetation can access the nutrients for growth, thereby acting to reduce (or "treat") the levels of nutrients in the phreatic water. When the vegetation dies down, especially in the winter amongst annual species, the natural degradation then releases organic form of nutrients back in to the water column. Seasonal hydraulic / hydrologic fluctuations may then cause small gradients to support a downward nutrient flux, driving higher concentrations in to the sediments underlying the peat. Diffusion as well as dispersion act to further spread out the concentrations throughout the different fen sediment layers.

Thus, the determined model for a lowland alkaline fen is one of a dynamic water balance at the dependent vegetation between groundwater, whose chemistry is modified by the pathways involved, and direct rainfall. The resilience of the vegetation depends on the buffering effect of the fen soils and substrate.

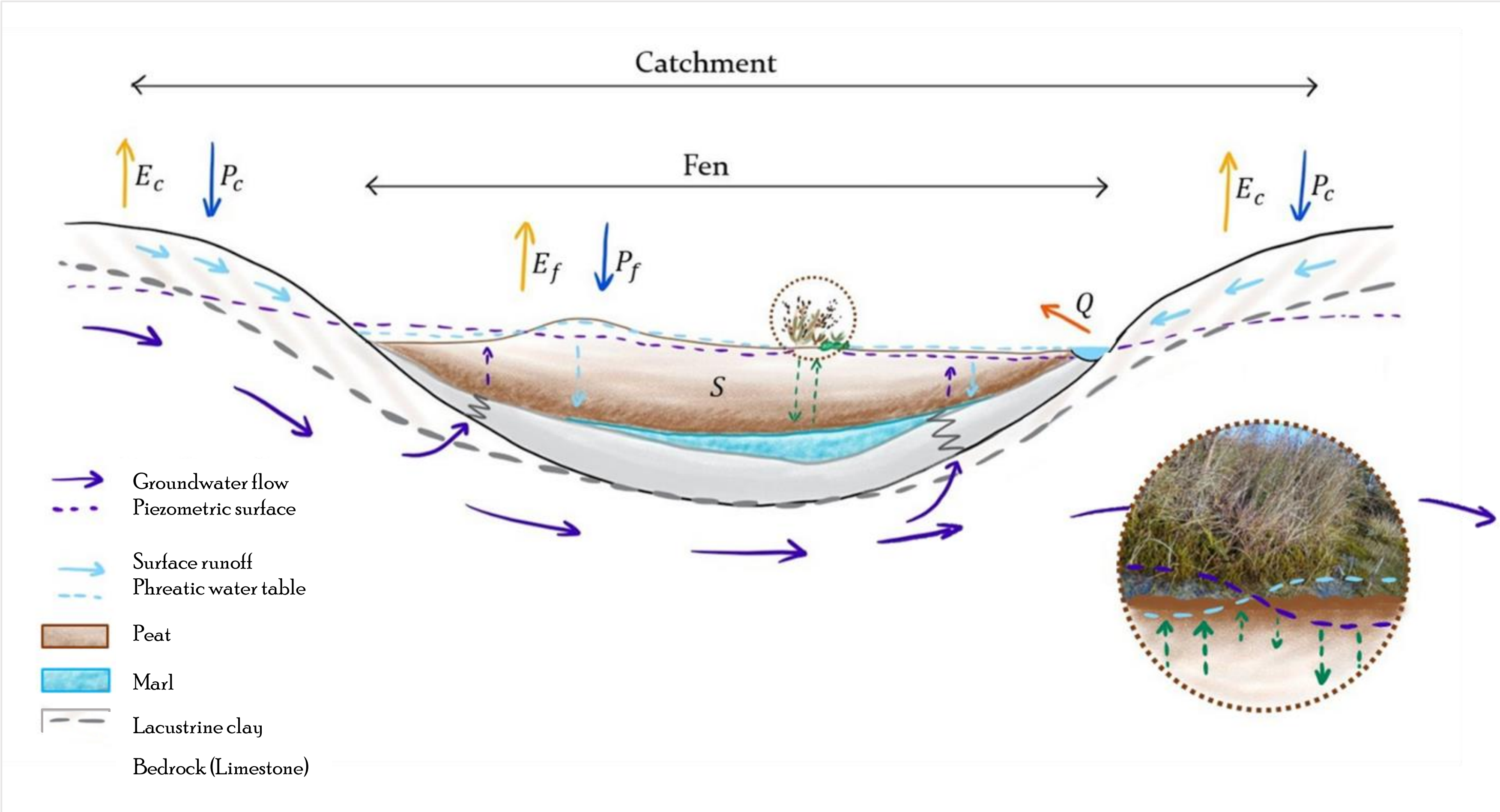


Figure 10.20. Conceptual model of Irish calcareous fens. The cut-out circle shows how the hydraulic gradient may change in specific fen vegetation from upward to downward and how this affects the nutrient circulation between peat and lower sediments.

10.4.2. Conceptual model specifics

To meet the specifics of this conceptual model hydrological and hydrochemical metrics are outlined in this section which can be used as a general baseline for Irish calcareous fens. Hydrological controls include surface and groundwater proportions feeding the fen. Generally, groundwater feeds the substrate below the peat, however groundwater proportions may vary in different sites. Therefore, the average EC in the substrate may be found between 550 and 790 µm/cm (based on 1st, 3rd quartile of all collected data). Surface water may also be a mix of groundwater and surface water with average values between 535 and 735 µm/cm (based on 1st, 3rd quartile of all data). To support good fen vegetation the overall surface water level needs to be within a range of 29 mm to 277 mm above ground level and these levels should be sustained for 60% of the year. These envelope values were calculated by taking the 1st, 3rd quartile of the good fen habitats which passed the assessment criteria specified in Section 4.1.3. The fens both support up and downward flows which may change between seasons, however, environmental controls should not change the hydraulic gradients by more than 0.4 during a hydrological year (as derived from Table 10.3).

Hydrochemistry controls should enable an internal cycling system where the fen recycles nutrients (which were released from the breakdown of vegetation) from the phreatic water levels into the sediments below. This is visible by a statistically significant difference in nutrient concentrations between the phreatic water levels (low concentrations) and the deeper substrate as well as surrounding catchment (high concentrations). Statistical analysis found the following median values of nutrients found in the near-surface phreatic zone of good quality fen habitat: 14 µg-P/l for dissolved reactive phosphorus, 110 µg-P/l total phosphorus, 0.26 mg-N/l for ammonia, 0.01 mg-N/l for total oxidised nitrogen and 0.95 mg-N/l for total dissolved nitrogen. However, this does not mean that levels found above these medians are not necessarily regarded as nutrients pollution, especially when the concentration of nutrients are reported below the threshold values as reported by Government of Ireland (2010). The nutrient concentrations in boreholes in the catchment surrounding the fen that supported these eventual values found in the fen had the following medians: 110 µg-P/l for dissolved reactive phosphorus, 260 µg-P/l total phosphorus, 0.17 mg-N/l for ammonia, 1.00 mg-N/l for total oxidised nitrogen and 2.59 mg-N/l for total dissolved nitrogen.

It is further possible that specific vegetation needs a certain array of nutrients in order to survive. Envelope recommendations for each Fossitt habitat are based on the 1st and 3rd quartile in boxplots of phreatic water table concentrations recorded in good fen habitats (Section 10.3.2). However, it is important to note that these values should be viewed as being representative of “remnant” nutrients not taken up by the vegetation or cycled into lower sediments rather than the nutrient feed needed to sustain the fen. These levels are also reflective of the dilution by rainwater evident from the mixed surface/groundwater at phreatic levels. The reported values in Table 10.8 are, however, a representative of typical conditions found in the phreatic zone of good quality fen habitat.

Table 10.7. Envelope recommendations of nutrients as a hydrochemical control on Irish calcareous fens.

	Dissolved reactive phosphorus		Total phosphorus		Total ammonia		Total oxidised nitrogen		Total dissolved nitrogen	
	as mg-P/l	St. dev	as mg-P/l	St. dev	as mg-N/l	St. dev	as mg-N/l	St. dev	as mg-N/l	St. dev
PF1	0.006 - 0.037	0.04	0.062 - 0.135	0.21	0.116 - 0.836	0.48	0.000 - 0.010	0.00	0.666 - 2.017	0.67
PF2	0.010 - 0.037	0.12	0.152 - 0.382	0.21	0.100 - 0.354	0.61	0.023 - 0.171	0.20	0.695 - 1.511	1.00
PF3	0.010 - 0.030	0.07	0.056 - 0.191	0.13	0.154 - 0.568	0.48	0.000 0.021	0.14	0.654- 1.502	0.64

Nutrients levels found in other fens such as sites studied in Scotland, Canada and Michigan show varied results when compared to the concentrations found the Irish fens (Table 10.8). It has to be noted that the international studies collected data in fen surface water but at varied depths. Furthermore, concentrations were sometimes recorded as a mean from three fens, such as the Canadian riparian fens (Duval, 2010), or is the median of a sample pool collected in 20 fens, as reported in Scotland (Schutten, 2019). This makes it harder to compare these values to the concentrations found in the Irish fens. It was also not reported if the habitat of these fens were in good condition, as is the case for the envelope recommendations reported in Table 10.7.

Nevertheless, some observations good be made comparing the nutrients concentrations of these fens to the Irish fens. It seemed that both the earlier study in Pollardstown fen and the Scottish fens found much higher concentrations of dissolved reactive phosphorus, while total ammonia is comparable to the concentrations found in Ireland. This could result in the fen becoming more nutrient polluted as phosphorus is more available to vegetation. This could then change the habitat, where the fen vegetation is replaced by other habitats that can thrive on these high nutrient concentrations. In contrast the Canadian and Michigan fen both report phosphorus levels way below the values found in Ireland and total ammonia and total oxidised nitrogen are also found at the lower end of the scale.

Table 10.8. Fen surface water nutrients concentrations reported in international literature.

	No. of sites	Dissolved reactive phosphorus	Total phosphorus	Total ammonia	Total oxidised nitrogen
		as mg-P/l	as mg-P/l	as mg-N/l	as mg-N/l
Ireland Pollards-town Fen (Kuczyńska 2008)	1	0.16 – 0.90		range 0.08 - 0.34	
Scottish fens (Schutten, 2019)	20	median 0.100		median 0.25	
Canadian riparian fens (Duval 2010)	3	mean ± SD 0.001 ± 0.0004		mean ± SD 0.52 ± 0.97	
Michigan fens, US (Schwintzer et al., 1982)	6	mean ±1 SD 0.008 ± 0.006	mean ± SD 0.032 ± 0.018	mean ± SD 0.18 ± 0.15	mean ± SD 0.006 ± 0.003

If the previously mentioned specific components of the conceptual model for a 'healthy' fen are met, the system has a higher natural resilience against natural environmental stressors such as drought or flooding. Furthermore, groundwater and surface water influx with high concentration of nutrients causes the fen to act as a self-cleaning system and cycles the nutrients internally.

Cusell et al. (2014) researched the process of filtering fens in phosphorus. . P-availability was lowest in relatively isolated floating and vegetation indicated P-limitation even though the fens had high phosphorus inflow. The author found that this pattern was primarily due to precipitation of Fe-phosphates. Verhoeven et al. (1985) found that in Dutch fens the P availability to the vegetation is reduced as a result of enhanced absorption and precipitation of phosphate due to the inflow of ground water. In addition, high total ammonia concentrations in the seepage water had no eutrophicating effect on the vegetation, as it is absorbed in deeper peat layers before it reaches the plant roots. They concluded that groundwater seepage is vital for the conservation of rich fens.

Finally, fen vegetation also shows patterns in nutrients cycling in a research where nitrogen-use efficiency (NUE) and litter decomposition was compared between different habitats (Aerts et al., 1999). There was a clear difference between predominant growth forms, with evergreens having the highest NUE.

Ultimately, the build-up of organic material (including nutrients) causes peat formation. In general, groundwater containing elevated nutrient concentrations are effectively supporting the fen vegetation. The maximum values in Table 10.8 should therefore not be used as a threshold value above which damage will be afflicted upon the fen as no direct link was found between high nutrient concentrations and 'poor' fen habitats. The vegetation was also found relatively robust against short to medium term changes in nutrient supply from groundwater or surface water through a process of self-recycling/cleaning. Therefore the threshold metrics should include a time dimension.

The natural resilience of Irish calcareous fen can be decreased by significant hydrological changes in the fen catchment (as was found in Tory Hill). Even though the research did not find damage by nutrient pollution, very high levels of nutrient pollution can cause the nutrient cycling system to become overloaded. This may result in levels toxic to fen vegetation coupled with them being outcompeted by species found in other habitats such as swamps, scrub or woodland.

11. Conclusions and Recommendations

11.1. Conclusions

The primary aim of this research was to investigate the hydrological and hydrochemical supporting conditions of Irish fen wetlands from which metrics for good quality fen vegetation could be developed. Appropriate controls were found for hydrological variables which are considered a primary metric. These are closely associated with a metric found for hydrochemistry related to typical nutrient variables that control fen vegetation growth. However, these values were not considered to be fixed criteria. They are considered as a 'threshold of deterioration' implying that if levels are found somewhat above the determined values, a fen should not automatically be considered to be in poor condition.

The fen metrics were established through an intensive hydrochemical monitoring programme. Four fens (5 research sites) were instrumented and monitored over a two-year period, investigating both their hydrology and hydrochemistry in relation to their different vegetation habitats. The fens were selected to represent a range of different conditions with at least one fen wetland considered to be under water quantity pressure (i.e. under pressure by drainage), one site considered to be under water quality pressure (i.e. under pressure from nutrient pollution), and at least one site that is considered to be still in a relatively pristine (intact) state. They were located along a climatic gradient in Ireland, from the drier east (Pollardstown Fen) through the midlands (Scragh Bog and Ballymore Bog) to the slightly wetter west (Tory Hill).

With respect to the expected conditions of the research sites, Tory Hill was indeed found to have serious drainage issues significantly affecting the phreatic water level. This left all surveyed fen habitats in an undesirable state, as seen in Table 11.1, even though they made up about 42% of the entire site. Pollardstown fen was not affected to the same extent, although some areas were found under pressure of drainage and a few areas were possibly under pressure from nutrient pollution. These combined impacts could be the reason some of the surveyed habitats were 'damaged' in the sense of not being classified as healthy fen. Pollardstown fen also supported smaller portions of fen habitat compared to the other sites with 31% in site A and a merely 20% in site D. Scragh Bog was chosen as a site under water quality pressure, possible from surrounding agriculture. However, the fen supported a high percentage of fen habitat (45%) and even though significantly higher nutrients were found at the phreatic level/water table, almost all fen habitat was in a pristine state. Ballymore also supported pristine fen vegetation and was neither affected by drainage nor nutrient pollution, as predicted. This site was supporting 53% of designated fen habitat at the time of surveying. More detailed information on the surveyed fen habitats shown for every research site is displayed in Table 11.1.

Table 11.1. Habitat quality of all sample sites surveyed in each research site.

	Ballymore	Pollardstown A	Pollardstown D	Scragh Bog	Tory Hill
Total sample sites	15	6	6	15	6
Good quality	9	2	2	5	0
Poor quality	0	2	3	0	4
Non fen habitat	6	2	1	10	2

The measured vertical hydraulic gradients in the fens also suggest that the incoming water from the aquifer enters the fens via discrete pathways straight into the phreatic zone, bypassing the lower permeability substrate. In terms of water level, the field investigations suggested that a threshold water level envelope of between 29 mm to 277 mm above ground level for at least 60% of the year seems to be required for healthy fen vegetation. However, for particular habitats, such as the designated Fossitt habitats, the controls were more stringent. The PF1 (rich fen and flush) habitat was found to require a tighter envelope of water levels always above the ground surface from approximately 225 mm to 370 mm depth of flooding all year round. These values are considered the primary hydrological control for healthy fen vegetation are hence are suitable as an appropriate metric. These controls were written up from data of good quality fen habitats collected from all research sites. However, to show the contrast between the research sites, detailed surface water values are reported in Table 11.2.

Table 11.2. Detailed surface water levels (in mAGL) specified for all research sites. Results include water levels from all recorded sites including habitats with good and poor conditions as well as non-fen habitats.

	Ballymore	Pollardstown A	Pollardstown D	Scragh Bog	Tory Hill
Median	0.039	-0.053	-0.026	0.127	-0.030
Average	0.006	-0.116	-0.041	0.119	-0.049
Minimum	-0.565	-0.668	-0.836	-0.760	-0.490
Maximum	0.388	0.255	0.513	0.675	0.500

Regular water quality monitoring across different transects on the fens comparing samples taken at the free water surface (phreatic) and the water from piezometers at depth, in the sediment between the glacial till and fen peat, generally revealed higher concentrations of nutrients in the sediments compared to those in the surface water. There also appeared to be little variation of the surface water nutrient concentrations across the seasons.

Statistical analysis of the surface water nutrient concentrations associated with the different fen habitat types suggests typical maximum values of 37 µg-P/l for dissolved reactive phosphorus, 382 µg-P/l total phosphorus, 0.57 mg-N/l for ammonia, 0.17 mg-N/l for total oxidised nitrogen and 2.01 mg-N/l for total dissolved nitrogen.

How these concentrations compare between research sites is shown

The median water quality values for good quality fen vegetation habitats were found to be 14 µg-P/l for dissolved reactive phosphorus, 110 µg-P/l total phosphorus, 0.26 mg-N/l for ammonia, 0.01 mg-N/l for total oxidised nitrogen and 0.95 mg-N/l for total dissolved nitrogen.

How nutrient concentrations compare between research sites is shown in Table 11.3. It has to be noted that these medians are taken from data that was collected in good as well as poor quality fen vegetation habitat.

Table 11.3. Median nutrient concentrations (in mg/L) specified for all research sites. Results include water levels from all recorded sites including habitats with good and poor conditions as well as non-fen habitats

Nutrients (mg/L)	Ballymore	Pollardstown A	Pollardstown D	Scragh Bog	Tory Hill
Dissolved reactive phosphorus	0.010	0.014	0.013	0.055	0.020
Total phosphorus	0.123	0.112	0.134	0.200	0.459
Total ammonia	0.131	0.141	0.081	0.213	0.122
Total oxidised nitrogen	0.724	1.472	1.190	1.353	2.144

The levels of nutrients in the aquifers that feed the fens were generally much higher than the concentrations in the phreatic zone which suggests that the sediments of the fen act as a filtration mechanism and, more importantly, that the vegetation is taking up the incoming nutrients which thereby end up in an organic form. This vegetation will decay on an annual cycle to form the fen peat and there is then evidence that some of the nutrients come back out into solution and disperse into the underlying till substrate (which showed anaerobic conditions in some cases). It was therefore concluded that the fens seem to act as nutrient sinks using he evidence of net accumulation in the lower sediments of the fen.

Hence, it follows that the fen specific vegetation requires a minimum threshold of nutrients in the groundwater feed order to survive. The indicative in-fen water quality levels for good quality Fossitt habitats were measured at levels shown in Table 11.4.

	Dissolved reactive phosphorus	**Total phosphorus**	**Total ammonia**	**Total oxidised nitrogen**	**Total dissolved nitrogen**
	as mg-P/l	as mg-P/l	as mg-N/l	as mg-N/l	as mg-N/l
PF1	0.006	0.062	0.116	<0.005	0.666
PF2	0.010	0.152	0.100	0.023	0.695
PF3	0.010	0.056	0.154	<0.005	0.654

Table 11.4. Indicative minimum threshold of nutrients needed measured in good quality fen habitats.

These values however, are considered to be remnant concentrations after vegetation uptake, surface water dilution and dispersion rather than the nutrients entering the fen from the catchment. In short, the fen vegetation exhibited remarkable resilience to fluctuating water quality.

In summary, there was little previous knowledge of calcareous fen in Ireland with respect to both hydrological and hydrochemical data. Kimberley (2013) gave median levels for nitrate threshold values ranging from 1.30 to 2.00 mg-N/l based on available data. This investigation determined a median for fens in good ecological condition was 0.924 mg-N/l. The total oxidised nitrogen (which can be regarded as nitrate) determined in this research was found much lower in comparison with a median of 0.01 mg-N/l. It has to be noted that these values were recorded in specifically designated good quality fen habitats whereas Kimberley's data was collected from available data at unspecified locations across the fen.

11.2. Recommendations

This research has provided metrics of typical hydrological and hydrochemical values measured over a timespan of two years. The fens studied in this project should continue to be monitored for another 3 years in order to strengthen the findings and evaluate their response over a longer climatological period. Furthermore, the database should be further expanded with data collected from an even wider array of fen sites in order to define the current metrics even better. These fens could include bigger differences in the geological setting as well as fens under higher pressure from nutrient pollution in order to define better nutrient thresholds. Even though it is probably not necessary to collect data to the same extent as in this research, it is still recommended to screen new fen sites in different seasons instead of taking a singular 'grab' sample as there is a high change of this not being a great representative of the current state of the fen. It is therefore essential to take the time dimension into account.

In addition, more targeted research is needed in terms of understanding the fate and transport of nutrients within the phreatic zone and down into the substrate. The variation of redox potential over time needs to be studied at targeted depth profile in different vegetation habitats. The main driver of these processes is oxygen. However, in the anoxic sediment layers of the fen other chemical species —nitrate, manganese, iron, sulphate, and carbon dioxide can accept electrons in its place. These components should therefore also be considered in a more extensive redox investigation. Also, it would be advantageous to undertake tracing of incoming groundwater nutrients, using labelled isotopes for example, to follow the nitrogen and phosphorus cycles within the fens.

Finally, when setting up a multi-disciplinary monitoring programme it is advised not to use the piezometer nests for hydrological monitoring and hydrochemical sampling at the same time. The phreatic tubes as well as piezometers required purging before each sample was taken. However, some piezometers with screens installed near the substrate of the fen had slow acting hydraulic gradients. It would therefore sometimes take weeks before the true piezometric head was reached again and accurate water levels could be recorded. A better approach would be to install phreatic tubes and piezometers for the hydrological data collection and in parallel to lysimeters at similar depths as their screens for the hydrochemical data collection. If piezometers and phreatic tubes are still to be used for hydrochemical data collection a flow cell should be used as some nutrients and elements are sensitive to the interference of oxygen that may occur when sampling is done without one. It is further advised to install boardwalks on or above the surface of the fen before commencing the instrumentation and data collection in the research areas in order to minimise the disturbance on the environment.

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Appendices

Appendix A. Sample object information

Sitename	Northing	Easting	Object type	Sample site	Object length	Object top elevation	Ground elevation	Screen elevation
Ballymore Fen	624254	749337	Phreatic well	BM12	1.4	91.40	90.59	89.98
Ballymore Fen	624255	749337	Piezometer	BM12	1.7	91.29	90.59	89.56
Ballymore Fen	624184	749371	Piezometer	BM14	1.3	91.12	90.63	89.80
Ballymore Fen	624184	749371	Phreatic well	BM14	1.4	91.55	90.63	90.15
Ballymore Fen	624133	749395	Piezometer	BM16	1.0	90.78	90.71	89.75
Ballymore Fen	624132	749395	Phreatic well	BM16	1.5	91.60	90.71	90.13
Ballymore Fen	624090	749410	Piezometer	BM161	2.2	90.81	90.18	88.61
Ballymore Fen	624090	749410	Phreatic well	BM161	1.5	91.03	90.18	89.58
Ballymore Fen	623999	749475	Phreatic well	BM162	1.5	90.84	90.25	89.39
Ballymore Fen	623999	749475	Piezometer	BM162	3.4	91.55	90.25	88.12
Ballymore Fen	623932	749547	Phreatic well	BM163	1.5	91.02	90.41	89.55
Ballymore Fen	623932	749547	Piezometer	BM163	3.4	90.67	90.41	87.24
Ballymore Fen	623887	749594	Piezometer	BM164	1.2	90.83	90.49	89.63
Ballymore Fen	623887	749595	Phreatic well	BM164	1.4	90.99	90.49	89.59
Ballymore Fen	623939	749354	Phreatic well	BM2	1.4	91.12	90.23	89.71
Ballymore Fen	623939	749355	Piezometer	BM2	1.9	91.13	90.23	89.22
Ballymore Fen	623875	749396	Phreatic well	BM4	1.5	90.98	90.38	89.51
Ballymore Fen	623875	749396	Piezometer	BM4	2.0	91.00	90.38	89.05
Ballymore Fen	623824	749436	Phreatic well	BM41	1.5	91.16	90.47	89.69
Ballymore Fen	623824	749436	Piezometer	BM41	2.3	91.95	90.47	89.70
Ballymore Fen	623773	749473	Phreatic well	BM42	1.5	91.18	90.58	89.73
Ballymore Fen	623773	749472	Piezometer	BM42	2.2	91.86	90.58	89.68
Ballymore Fen	623998	749315	Piezometer	BM5	1.2	91.16	90.32	89.93
Ballymore Fen	623998	749315	Phreatic well	BM5	1.5	91.46	90.32	90.00
Ballymore Fen	624006	749246	Piezometer	BM7	1.5	91.64	90.71	90.19
Ballymore Fen	624006	749247	Phreatic well	BM7	1.4	91.69	90.71	90.26

Appendix A. Sample object information

Ballymore Fen	624039	749210	Piezometer	BM8	1.4	90.89	90.43	89.53
Ballymore Fen	624039	749209	Phreatic well	BM8	1.5	91.35	90.43	89.88
Ballymore Fen	624113	749190	Piezometer	BM9	1.3	91.48	90.81	90.16
Ballymore Fen	624113	749191	Phreatic well	BM9	1.4	91.90	90.81	90.50
Ballymore Fen	624344	749362	Borehole	BHKearney	5.3	97.11	97.17	91.78
Ballymore Fen	624344	749362	Borehole	BHKearney	2.2	97.11	97.17	94.96
Ballymore Fen	623858	749601	Borehole	BMBrownesLower	1.0	91.84	91.89	90.84
Ballymore Fen	623858	749601	Borehole	BMBrownesLower	5.0	91.84	91.89	86.84
Ballymore Fen	623717	749694	Borehole	BMBrownesUpper	3.5	94.74	94.74	91.23
Ballymore Fen	623717	749694	Borehole	BMBrownesUpper	2.7	94.74	94.74	92.04
Ballymore Fen	623959	749231	Outlet	Outlet	0.0			
Pollardstown site A	676358	715922	Phreatic well	PA31	1.5	87.27	86.17	85.77
Pollardstown site A	676358	715922	Piezometer	PA31	2.6	86.81	86.17	84.24
Pollardstown site A	676369	715952	Phreatic well	PA34	1.5	85.83	85.18	84.33
Pollardstown site A	676368	715953	Piezometer	PA34	4.3	86.48	85.18	82.16
Pollardstown site A	676369	715953	Piezometer	PA34	7.3	87.00	85.18	79.73
Pollardstown site A	676376	715973	Phreatic well	PA37	1.5	85.12	84.24	83.62
Pollardstown site A	676376	715973	Piezometer	PA37	6.3	85.65	84.24	79.34
Pollardstown site A	676389	716031	Phreatic well	PA38	1.5	83.79	83.03	82.29
Pollardstown site A	676389	716031	Piezometer	PA38	2.5	84.28	83.03	81.83
Pollardstown site A	676406	716080	Phreatic well	PA39	1.5	83.68	83.15	82.18
Pollardstown site A	676406	716080	Piezometer	PA39	3.5	84.44	83.15	80.99
Pollardstown site A	676419	716124	Phreatic well	PA40	1.5	83.68	82.86	82.18
Pollardstown site A	676418	716125	Piezometer	PA40	3.5	83.99	82.86	80.54
Pollardstown site A	676308	715698	Borehole	MB38	20.0	95.71	95.73	75.75
Pollardstown site A	676401	715786	Borehole	MB39		93.85	93.85	
Pollardstown site A	676425	716131	Outlet	Outlet	0.0	82.47	81.50	
Pollardstown site D	677098	716835	Phreatic well	PD23	1.5	84.58	83.87	83.08
Pollardstown site D	677099	716834	Piezometer	PD231	10.5	84.64	83.87	74.14
Pollardstown site D	677099	716835	Piezometer	PD23	3.3	84.14	83.87	80.82
Pollardstown site D	677094	716801	Phreatic well	PD27	1.5	83.56	83.06	82.06
Pollardstown site D	677093	716801	Piezometer	PD272	11.7	83.85	83.06	72.20
Pollardstown site D	677094	716801	Piezometer	PD27	3.8	83.80	83.06	80.04

Appendix A. Sample object information

Pollardstown site D	677076	716793	Phreatic well	PD28	1.5	83.33	82.91	81.83
Pollardstown site D	677075	716793	Piezometer	PD281	10.9	83.96	82.91	73.07
Pollardstown site D	677076	716793	Piezometer	PD28	3.7	83.85	82.91	80.18
Pollardstown site D	677084	716769	Phreatic well	PD29	1.5	83.16	82.74	81.66
Pollardstown site D	677084	716769	Piezometer	PD29	2.5	83.82	82.74	81.37
Pollardstown site D	677052	716747	Phreatic well	PD30	1.5	83.30	82.78	81.80
Pollardstown site D	677051	716747	Piezometer	PD30	3.5	84.03	82.78	80.58
Pollardstown site D	677034	716729	Phreatic well	PD32	1.5	83.30	82.79	81.80
Pollardstown site D	677034	716729	Piezometer	PD32	4.5	84.45	82.79	79.95
Pollardstown site D	677303	717330	Borehole	MB37	18.2	88.85	88.98	70.65
Pollardstown site D	677237	717089	Borehole	MB43		90.48	90.48	
Pollardstown site D	677123	717027	Borehole	MB44		89.45	89.45	89.45
Pollardstown site D	677389	716815	Borehole	MB45	7.4	89.09	89.09	81.74
Scragh Bog	641922	759809	Borehole	GW08	7.0	107.65	107.65	100.65
Scragh Bog	642050	758668	Borehole	GW09	43.0	122.49	122.49	79.49
Scragh Bog	641640	759881	Borehole	GW15	25.0	115.95	115.95	90.95
Scragh Bog	642309	759452	Borehole	BHDuffyHigh	3.0	112.03	112.00	109.03
Scragh Bog	642309	759452	Borehole	BHDuffyHigh	1.5	112.03	112.00	110.58
Scragh Bog	642222	759400	Borehole	BHDuffyLow	1.0	104.79	105.19	103.79
Scragh Bog	642222	759400	Borehole	BHDuffyLow	5.0	104.79	105.19	99.79
Scragh Bog	641975	759301	Borehole	BHWallaceN	15.0	114.39	114.46	99.39
Scragh Bog	641975	759301	Borehole	BHWallaceN	3.0	114.39	114.46	111.39
Scragh Bog	642240	758683	Borehole	BHWallaceS	2.3	119.36	119.22	117.11
Scragh Bog	642240	758683	Borehole	BHWallaceS	5.5	119.36	119.22	113.86
Scragh Bog	641993	759657	Outlet	Outlet	0.0	104.66	104.66	104.66
Scragh Bog	642313	758869	Phreatic well	SC0	1.5	105.16	104.56	103.66
Scragh Bog	642313	758869	Piezometer	SC0	2.5	105.73	104.57	103.28
Scragh Bog	642104	759323	Phreatic well	SC10	1.5	105.01	104.33	103.51
Scragh Bog	642104	759323	Piezometer	SC10	2.0	105.15	104.24	103.15
Scragh Bog	642148	759350	Phreatic well	SC12	1.5	105.05	104.45	103.55
Scragh Bog	642148	759351	Piezometer	SC12	6.0	106.18	104.41	100.18
Scragh Bog	642182	759372	Phreatic well	SC13	1.5	105.04	104.41	103.54
Scragh Bog	642183	759373	Piezometer	SC13	6.0	105.76	104.48	99.76
Scragh Bog	642215	759393	Piezometer	SC15	1.0	105.15	104.55	104.15
Scragh Bog	642216	759393	Phreatic well	SC15	1.5	105.39	104.48	103.89

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Scragh Bog	642681	758523	Phreatic well	SC16	1.5	105.34	104.90	103.84
Scragh Bog	642681	758522	Piezometer	SC16	2.0	105.90	104.90	103.90
Scragh Bog	642612	758697	Phreatic well	SC17	1.5	105.12	104.43	103.62
Scragh Bog	642613	758697	Piezometer	SC17	3.0	105.43	104.56	102.43
Scragh Bog	642528	758795	Phreatic well	SC18	1.5	105.28	104.70	103.78
Scragh Bog	642528	758795	Piezometer	SC18	6.0	106.28	104.71	100.28
Scragh Bog	642456	758847	Phreatic well	SC19	1.5	105.11	104.42	103.61
Scragh Bog	642456	758847	Piezometer	SC19	5.0	105.67	104.64	100.67
Scragh Bog	642345	758888	Phreatic well	SC2	1.5	104.80	104.46	103.30
Scragh Bog	642345	758888	Piezometer	SC2	3.5	105.73	104.46	102.28
Scragh Bog	642421	758936	Phreatic well	SC6	1.5	105.09	104.54	103.59
Scragh Bog	642421	758937	Piezometer	SC6	8.5	105.44	104.46	96.99
Scragh Bog	642311	759100	Phreatic well	SC612	1.5	105.12	104.40	103.62
Scragh Bog	642311	759100	Piezometer	SC612	6.0	105.01	104.40	99.01
Scragh Bog	642466	758970	Phreatic well	SC7	1.5	105.05	104.47	103.55
Scragh Bog	642466	758970	Piezometer	SC7	8.0	105.36	104.50	97.36
Scragh Bog	642500	758988	Phreatic well	SC8	1.5	105.00	104.49	103.50
Scragh Bog	642500	758988	Piezometer	SC8	4.0	105.22	104.49	101.22
Scragh Bog	642083	759308	Piezometer	SC9	1.0	105.23	104.54	104.23
Scragh Bog	642083	759309	Phreatic well	SC9	1.5	105.47	104.49	103.97
Scragh Bog	643140	758245	Borehole	GW21	4.6	115.52	115.52	110.92
Scragh Bog	642811	758375	Turlough	Turlough	0.0	111.00	111.00	111.00
Tory Hill	553363	643006	Piezometer	TH12	1.2	28.06	27.39	26.86
Tory Hill	553363	643007	Phreatic well	TH12	1.0	27.88	27.39	26.88
Tory Hill	553391	642957	Piezometer	TH13	1.0	27.07	26.50	26.12
Tory Hill	553391	642957	Phreatic well	TH13	0.6	27.00	26.53	26.40
Tory Hill	553431	642936	Piezometer	TH14	1.1	26.82	26.21	25.77
Tory Hill	553431	642936	Phreatic well	TH14	0.9	26.75	26.23	25.85
Tory Hill	553489	643326	Piezometer	TH6	1.1	26.35	25.89	25.25
Tory Hill	553489	643326	Phreatic well	TH6	0.8	26.27	25.88	25.52

Appendix A. Sample object information

Tory Hill	553537	643281	Piezometer	TH7	1.3	27.10	26.50	25.80
Tory Hill	553537	643281	Phreatic well	TH7	0.8	26.88	26.47	26.13
Tory Hill	553592	643213	Phreatic well	TH8	1.0	27.01	26.55	26.06
Tory Hill	553592	643212	Piezometer	TH8	2.0	26.81	26.56	24.81
Tory Hill	553739	643489	Lake	Lake	0.0			
Tory Hill	553381	643368	Borehole	Pumphouse	4.0	29.04	28.45	25.04
Tory Hill	553344	642752	Outlet	Outlet	0.0			

Appendix A. Sample object information

Appendix B. Soil logs

Ballymore - Fen					
Sample site	Peat	Marl	Blue clay with sand and gravel	Sandy clay	Sandy gravelly clay
42	0 -0.5	0.5 -0.9			0.9>
41	0 -0.9				0.9>
4	0 -2	2 -2.2	2.2 -2.35		
2	0 -0.8		0.8>		
5	0 -0.4				1.5>
7	0 -0.3				
8	0 -0.3				
9	0 -0.4			0.4 -0.6	0.6>
164	1.2				1.2
163	1.8	3.3	3.3		
162	2				
161	1.7				0.7
16	0.7				0.5
14	0.5				0.5
12	0.2	0.5			0.5

Ballymore - Boreholes							
Sample site	Sand	Gravelly sand	Sandy gravel	Clay	Wet clay	Gravelly clay	Bedrock
Brownes Upper	2.6					3.6	3.6
Brownes Lower	1.8			2.5	5.4		5.4
John Kearney	1.6	1.9	3.5			6.3	6.3

Pollardstown site A								
Sample site	Peat	Topsoil	Marl	Grey clay	Gravel	Sandy clay	Sand	Boulder
40	0 - 0.5, 0.7-2.5		0.5-0.7	2.5>				
39	0 -0.4, 0.5-2.4		0.4-0.5	2.4>				
38	0-1.3				1.3>			
37	0-1.2			1-4.5	4.5-6		6>	
34	0-1			1-1.5, 4-5		3.2-4	1.5-3.2, 5>	
31	0-0.5			0.5-0.8, 3-5	0.8-3, 6			
MB38		0-0.8			0.8-19	19-20		20>

Pollardstown site D							
Sample location	Peat	Grey clay	Gravel	Brown clay	Sand	Marl	Boulder
32	0-3.3		3.3				
30	0-1.25		1.25				
29							
28	0-0.7	0.7-1.7, 3-3.5	1.7-3	3.5-11			
27	0-0.8	0.8-1.5, 2.5-4	1.5-2.5	4-11			
23	0-0.9	0.9-1.5	1.5-2	2-9			

Scragh Bog- Fen							
Sample location	Peat	Livermud	Shell marl	Livermud	Gyttja	Boulder clay	End core
0	0					1.3	1.3
2	0	1.8			3.4	4.8	4.8
3	0	1.3			5.5	6.4	6.4
6	0	2.5			6.9	8.3	8.3
7	0	1.8			6.4	7.6	7.6
8	0	1.3				2.6	2.6
9	0	0.1					1
10	0	0.4			1.4	1.7	1.7
12	0	1.1	2.2	3	4.1	5.6	5.6
13	0	1.2	1.6	3.1	3.8	4.9	4.9
15	0	0.1					0.62
612	0	2			5.7	6.3	6.3
19	0	1.9			2.7	5.5	5.5
18	0	1.3				1.5	1.5
17	0					3	3
16	0						1.1

Scragh Bog - Boreholes						
Sample location	Sand	Gravelly clay	Sandy gravel	Sandy clay	Wet clay	Bedrock
Wallace North	1.2	10		15.5	17.5	
Duffy Lower		6.9		1.8		6.9
Duffy Upper	1		3.2			3.2

Tory Hill					
Sample site	peat	marl	lacustrine clay	blue clays	gravel
12	0.5	0.75	4.7	13.5	13.6
13	0.5	0.75	4.7	13.5	13.6
14	0.35	0.6	4.7	13.5	13.6
6	0.4	0.65	4.7	13.5	13.6
7	0.2	0.45	4.7	13.5	13.6
8	0.2	0.45	4.7	13.5	13.6

Appendix C. Vegetation surveys relevé data

Site	Ballymore (1)				
Relevé code	B10	B15	B17	B19	B23
Sample site code	BM162	BM41	BM4	BM2	BM8
Easting ITM	623992.0	623870.3	623876.6	623941.5	624039.8
Northing ITM	749490.0	749444.5	749400.5	749358.7	749210.7
Distance from sample point	16	46	4	5	0.5
Quadrat size (m2)	4	4	4	4	4
Annex	7140	7140	7140	7140	7140
Fossitt	PF3	PF3	PF3	PF3	PF3
Fen quality	Good	Good	Good	Good	Good
% surface water	0	10	1	0	3
% bare ground	0	0	0	0	0
% total vegetation	100	90	99	100	98
% grass	0	0	0	0	0
% sedge	0	0	0	0	0
% herb	80	60	50	70	55
% bryophyte	60	80	80	75	90
% shrub	0	1	15	5	2
% canopy	0	0	0	0	0
% rush	0	0	0	0	0
% litter					
Veg max ht Q1					
Veg max ht Q2					
Veg max ht Q3					
Veg max ht Q4					
Height (cm)	30 to 50	10 to 30	20 to 30	20 to 40	20 to 30
NOTES					
Species No.	17	16	22	24	23
Agrostis stolonifera	3	3	7		1.5
Angelica sylvestris		1.5	1.5	0.5	
Betula pubescens				1.5	
Bryum pseudotriquetrum				20	
Calliergonella cuspidata	30	20	30	42	7
Caltha palustris	1.5	1.5	3	3	
Campylium stellatum	20	0.5		30	85
Cardamine pratensis	0.5	3	1.5	1.5	0.5
Carex diandra		20	20	30	20
Carex disticha	3				
Carex rostrata				7	1.5
Carex viridula brachyrrhyncha	20	3			3
Cirsium palustre			0.5		
Comarum palustre	0.5			3	
Ctenidium molluscum					3

Epilobium palustre		3	1.5	0.5	
Epilobium parviflorum					
Epipactis palustris		3	7	3	
Equisetum fluviatile	7	0.5		3	1.5
Equisetum palustre		1.5	20	1.5	7
Eriophorum angustifolium			1.5		1.5
Filipendula ulmaria	7		0.5	0.5	1.5
Galium palustre	1.5			0.5	1.5
Holcus lanatus	1.5		3		
Juncus articulatus					3
Mentha aquatica	3			0.5	1.5
Menyanthes trifoliata	30	30	20	20	20
Parnassia palustris	0.5		20	7	
Pyrola rotundifolia			7	0.5	7
Ranunculus flammula					3
Sagina nodosa				1.5	
Salix cinerea		0.5	0.5	0.5	1.5
Scorpidium scorpioides		42			
Silene flos-cuculi			3	7	3
Succisa pratensis	20		3	3	0.5
Triglochin palustre					0.5

Site	Ballymore (1)				
Relevé code	B9	B12	B21	B22	B24
Sample site code	BM163	BM161	BM7	BM5	BM9
Easting ITM	623969.1	624089.4	623992.2	624011.6	624123.9
Northing ITM	749527.4	749431.0	749256.0	749288.9	749201.6
Distance from sample point	42	22	18	29	16
Quadrat size (m2)	4	4	4	4	4
Annex	7140	7140	none	none	none
Fossitt	PF3	PF3	GM1	GM1	GM1
Fen quality	Good	Good	Not fen	Not fen	Not fen
% surface water	10	75	0	0	0
% bare ground	0	0	0	0	0
% total vegetation	95	80	100	100	100
% grass	0	0	0	0	0
% sedge	0	0	0	0	0
% herb	90	80	95	95	80
% bryophyte	60	0	10	50	70
% shrub	0	0	0	0	0
% canopy	0	0	0	0	0
% rush	0	0	0	0	0
% litter					
Veg max ht Q1					
Veg max ht Q2					
Veg max ht Q3					
Veg max ht Q4					
Height (cm)	30 to 40	30 to 50	60 to 90	50 to 70	50 to 80
NOTES					
Species No.	9	8	16	14	19
Angelica sylvestris			3		
Anthoxanthum odoratum			20	7	7
Arrhenatherum elatius			30		20
Berula erecta		20			
Calliergonella cuspidata			7		
Calluna vulgaris					7
Caltha palustris		3			0.5
Campylium stellatum					7
Carex diandra	7		1.5	7	0.5
Carex flacca				0.5	1.5
Carex nigra					3
Carex panicea	7				3
Carex rostrata					20
Carex viridula brachyrrhyncha	3	7			
Centaurea nigra				0.5	85
Chara sp.					20
Cirsium palustre					3

Comarum palustre					0.5
Dactylorhiza fuchsii					3
Dactylorhiza sp.					0.5
Drosera rotundifolia					0.5
Epilobium palustre	0.5				
Epipactis palustris	0.5				3
Equisetum fluviatile	1.5	20			7
Equisetum palustre			3		
Eriophorum angustifolium	1.5				
Filipendula ulmaria			42	30	
Holcus lanatus			3	3	
Hydrocotyle vulgaris		20			
Hylocomium splendens				30	
Juncus inflexus				42	
Lathyrus pratensis			7		
Luzula camplestris				0.5	
Lythrum salicaria			20		
Mentha aquatica		7			
Menyanthes trifoliata	85	42			
Plagiomnium undulatum			3		
Potamogeton coloratus		7			
Potentilla erecta			7	7	
Ranunculus acris				0.5	
Scleropodium purum			7	20	
Scorpidium scorpioides	85				
Stellaria gamineae			1.5		
Succisa pratensis				3	
Valeriana officinalis			0.5		

Site	Pollardstown site A					
Relevé code	PF_01	PF_02	PF_03	PF_04	PF_05	PF_06
Sample site code	PA31	PA34	PA37	PA38	PA39	PA40
Easting ITM	676367.0	676379.0	676385.0	676396.0	676410.0	676422.0
Northing ITM	715933.0	715955.0	715975.0	716031.0	716079.0	716123.0
Distance from sample point	2	2	2	2	2	2
Quadrat size (m2)	4	4	4	4	4	4
Annex	7230	7230	7230	7230	None	None
Fossitt	PF2	PF2	PF2	PF2	GS4	GS2
Fen quality	Good	Good	Good	Good	Not fen	Not fen
% surface water	0	0	5	70	0	0
% bare ground	0	0	0	0	0	0
% total vegetation	95	95	95	75	100	100
% grass	33	43	11	9.5	97	87.5
% sedge	62.5	37	72	28	2	11
% herb	10	14.5	13	32	43	18.8
% bryophyte	40	18.6	47	22.6	0.9	0.8
% shrub	0.5	0.1	0.1	0	0	0
% canopy	0	0	0	0	0	0
% rush	3	30	21	40	0	1
% litter	20	30	0	7	60	30
Veg max ht Q1	110	75	95	145	105	70
Veg max ht Q2	150	80	120	110	110	100
Veg max ht Q3	85	105	100	140	75	110
Veg max ht Q4	105	85	100	135	90	90
Height (cm)	85 to 150	75 to 105	95 to 120	110 to 145	75 to 110	70 to 110
NOTES						
Species No.	29	25	29	29	20	21
Agrostis stolonifera	7	1	1	0.5		
Anagallis tenella			0.1			
Angelica sylvestris				1	3	
Anthoxanthum odoratum	0.3	0.3			10	1
Apium nodiflorum				1		
Arrhenatherum elatius					0.3	5
Brachythecium rutabulum					0.3	0.5
Briza media		0.3				
Bryum pseudotriquetrum	0.3					
Calliergon cordifolium				0.3		
Calliergon giganteum				3		
Calliergonella cuspidata	15	3	15	15		0.3
Caltha palustris				3		
Calypogeia fissa	0.5					
Calystegia sepium						7
Campyliadelphus elodes				3		
Campylium stellatum	10		25			
Cardamine pratensis	0.1			0.3		

Carex diandra				1		
Carex echinata	0.3					
Carex flacca	1	30	15	7	1	1
Carex hirta						10
Carex lasiocarpa				7		
Carex lepidocarpa	0.5		10	7		
Carex nigra					1	
Carex panicea	0.7	7	7	5		
Carex pulicaris		0.3				
Carex rostrata				1		
Centaurea nigra					3	3
Charophyte sp.			0.3			
Chrysosplenium oppositifolium						
Cirsium palustre	0.3		0.3			
Climacium dendroides		0.3				
Ctenidium molluscum	10		3			
Dactylorhiza sp.			0.1			
Dactylus glomerata					1	20
Deschampsia cespitosa			0.3			7
Epilobium palustre			0.1	0.5	0.1	
Equisetum palustre			0.1			0.3
Eriophorum angustifolium				0.3		
Festuca rubra		0.5	0.3	1	0.5	15
Filipendula ulmaria		2			25	
Fissidens adianthoides	1		0.5			
Fraxinus excelsior	0.5	0.1	0.1			
Galium palustre						
Galium uliginosum	0.3	0.5	0.7		0.3	
Holcus lanatus	1	0.5	0.3		0.3	1
Hylocomium splendens		5				
Hypericum pulchrum	0.1					
Juncus acutiflorus			1			
Juncus inflexus						1
Juncus subnodulosus	3	30	20	40		
Kindbergia praelonga						
Lathyrus pratensis	0.7	1			1	7
Lotus corniculatus						0.3
Mentha aquatica				3		
Molinia caerulea	25	40	10	3	85	3
Palustriella commutata				1		
Pedicularis palustris			7	3		
Pellia sp.	0.3					
Phleum pratense						35
Phragmites australis	1	0.5	3	5		
Picea sitchensis						
Pinguicula vulgaris			0.1			
Plagiomnium elatium				0.3		

Plagiomnium undulatum		0.1			0.3	
Poa trivialis						0.5
Potamogeton coloratus				5		
Potentilla anserina						0.5
Potentilla erecta	7	5	5	0.3	10	1
Pseudoscleropodium purum	5	10	1			
Ranunculus acris					0.3	
Ranunculus flammula				0.3		
Rhizomnium punctatum						
Rhytidiadelphus squarrosus		0.3			0.3	
Riccardia multifida	0.1					
Schoenus nigricans	60	0.3	40			
Scorpidium cossonii			3			
Stellaria graminea					0.5	
Succisa pratensis	0.3	5				
Utricularia minor				15		

Site	Pollardstown site D					
Relevé code	PF_07	PF_08	PF_09	PF_10	PF_11	PF_12
Sample site code	PD23	PD27	PD28	PD29	PD30	PD32
Easting ITM	677096.0	677094.0	677076.0	677085.0	677052.0	677035.0
Northing ITM	716837.0	716803.0	716796.0	716770.0	716750.0	716733.0
Distance from sample point	2	2	2	2	2	2
Quadrat size (m2)	4	4	4	4	4	4
Annex	6410	7230	7210	None	7230	7210
Fossitt	GS4	PF2	FS1	PF2	PF2	FS1
Habitat name						
Fen quality	Not fen	Good	Poor	Poor	Good	Poor
% surface water	3	5	0	10	0	0
% bare ground	0	0	0	0	0	0
% total vegetation	100	95	90	90	95	100
% grass	87	15	1	40	46.3	1
% sedge	7	20	90	40	43	95
% herb	18	25	1	15	8.3	7
% bryophyte	0.3	40	0	80	3	0
% shrub	0	0	0	1	0	0.5
% canopy	0	0	0	0	0	0
% rush	3.5	60	0	0.3	10	0
% litter	80	3	80	1	80	80
Veg max ht Q1	105	100	148	130	130	140
Veg max ht Q2	110	105	148	130	120	135
Veg max ht Q3	90	110	150	135	135	145
Veg max ht Q4	77	150	150	170	130	140
Height (cm)	77 to 110	100 to 150	148 to 148	130 to 170	120 to 135	135 to 145
NOTES	Cirsium dissectum occurs nearby					
Species No.	16	27	6	22	22	11
Agrostis stolonifera				0.3	5	
Angelica sylvestris		7		3		1
Anthoxanthum odoratum	0.5	0.5			1	
Calliergonella cuspidata		25		70		
Calluna vulgaris				0.3	5	
Caltha palustris					1	
Calypogeia fissa					0.3	
Campylium stellatum		3				
Cardamine pratensis		0.1		0.3		
Carex diandra		1				
Carex echinata		1	0.3			
Carex flacca	7	0.5			5	
Carex lasiocarpa		15		40		
Carex lepidocarpa		1				
Carex panicea		3		1	3	0.5
Centaurea nigra	3					
Cirsium dissectum					3	3

Cladium mariscus			90			95
Cratoneuron filicinum		3				
Deschampsia cespitosa	5					
Epilobium palustre		0.5		0.5		
Equisetum fluviatile				0.3	0.3	
Equisetum palustre	0.3	1				
Erica tetralix				0.1	3	0.5
Eriophorum angustifolium				0.3		
Eupatorium cannabinum	3	7	0.5	0.5		1
Festuca rubra	7			3		
Filipendula ulmaria		1	0.1			0.5
Fissidens adianthoides					0.3	
Holcus lanatus	5	10		1		
Hookeria lucens					0.1	
Hydrocotyle vulgaris				7		
Hylocomium splendens				0.1		
Hypericum pulchrum					0.3	
Juncus acutiflorus	0.5			0.3		
Juncus subnodulosus	3	60			10	
Lathyrus pratensis	0.3					
Lophocolea bidentata		0.1				
Mentha aquatica	1					
Menyanthes trifoliata				3		
Molinia caerulea	70	5			40	1
Myosotis scorpioides						
Oxyrrhynchium speciosum	0.3				0.3	
Phragmites australis		3	1	40	0.3	0.5
Plagiomnium elatium		0.3				
Potentilla erecta	10	10	0.5		3	3
Pseudoscleropodium purum		10		10	1	
Salix x multinervis				1		
Schoenus nigricans					35	1
Succisa pratensis	1	1			1	
Thuidium tamariscinum					0.5	
Trifolium repens		1				
Utricularia minor				0.5		
Valeriana officinalis		0.7				

Site	Scragh Bog (1)							
Relevé code	SB_01	SB_02	SB_03	SB_04	SB_05	SB_06	SB_07	SB_08
Sample site code	8	7	6	2	0	19	9	10
Easting ITM	642505.0	642470.0	642420.0	642346.0	642318.0	642459.0	642087.0	642103.0
Northing ITM	758992.0	758971.0	758944.0	758893.0	758875.0	758847.0	759312.0	759326.0
Distance from sample point	2	2	2	2	2	2	2	2
Quadrat size (m2)	4	4	4	4	4	4	4	4
Annex	None	7140	None	7140	None	91D0	None	None
Fossitt	FS2	PF3	WS1	PF3	GM1	WN7	FS2	FS2
Fen quality	Not fen	Good	Not fen	Good	Not fen	Not fen	Not fen	Not fen
% surface water	80	2	70	20	5	0	10	100
% bare ground	0	0	0	0	0	0	0	0
% total vegetation	95	100	100	95	100	100	100	90
% grass	2	0.5	10	10	40	5	15	0
% sedge	0.5	20	20	70	0	0.5	15	2
% herb	90	60	45	20	40	1	65	85
% bryophyte	3	55	5	60	15	100	90	10
% shrub	5	15	50	0	0	1	0	0
% canopy	0	0	0	0	0	75	0	0
% rush	0	7	0	5	0	0	0.7	10
% litter	1	1	3	0.5	3	5	0.5	0.3
Veg max ht Q1	100	120	190	60	60	65	65	131
Veg max ht Q2	50	90	75	70	60	300	64	127
Veg max ht Q3	90	110	70	85	65	55	76	116
Veg max ht Q4	120	95	170	65	70	70	56	127
Height (cm)	50 to 120	90 to 120	70 to 190	60 to 85	60 to70	55 to 300	56 to 76	116to131
Species No.	16	31	22	25	17	12	23	16
Agrostis stolonifera	1	0.3	10	0.5	5		3	
Angelica sylvestris		0.3	0.5	3			0.7	
Apium nodiflorum								3
Arrhenatherum elatius					0.5			
Aulacomnium palustre						0.1		
Berula erecta								
Betula pubescens	0.3	0.7				45		
Brachythecium rivulare	0.5	35	5	3	0.3	0.1	0.3	
Bryum pseudotriquetrum		0.1						
Calliergon cordifolium			0.5	0.3				
Calliergon giganteum		1		3	1		1	10
Calliergonella cuspidata	2	5	0.3	40	15		50	
Calluna vulgaris								
Caltha palustris		0.1	0.1	0.3			0.5	
Cardamine pratensis				0.1			0.3	0.3
Carex appropinquata			20					
Carex diandra		3		15			5	
Carex lasiocarpa		15		50			10	1

Carex nigra	0.5			1		0.5	0.5	
Carex panicea		0.5	0.3					
Carex rostrata			0.3	3				1
Comarum palustre		0.7	0.3	3	5	1	5	0.7
Drepanocladus aduncus				15				
Epilobium palustre	0.3	0.1	2	0.3				
Epilobium parviflorum				3	5		3	0.7
Equisetum fluviatile	35	0.7	10	0.5	1	0.3	1	35
Festuca rubra				7	35		1	
Filipendula ulmaria	35	1	25	15	15		20	
Galium palustre	0.3		0.3	0.1	3		0.3	
Galium uliginosum		0.3	0.3					
Hamatocaulis vernicosus		3						
Hedera hibernica								
Holcus lanatus	0.3	0.3		5	1	0.1	15	
Hydrocharis morsus-ranae								55
Hydrocotyle vulgaris								7
Juncus acutiflorus		7		5				10
Juncus articulatus							0.7	
Kindbergia praelonga		15				0.3		
Lemna minor	65		10					1
Mentha aquatica		0.3	0.7	0.3			0.1	0.3
Menyanthes trifoliata		15	1	3	7		30	3
Molinia caerulea						5		
Oxyrrhynchium speciosum	0.5							
Plagiomnium elatium			0.1					
Potamogeton coloratus								3
Pseudoscleropodium purum	0.3							
Pyrola rotundifolia		2						
Ranunculus flammula		0.1			0.3			
Rhytidiadelphus squarrosus	0.5	1			0.3		40	
Salix aurita			50					
Salix cinerea	5	10				30		
Salix repens		3						
Silene flos-cuculi		0.7						
Sphagnum flexuosum						25		
Sphagnum palustre						75		
Succisa pratensis		0.7	0.3		0.5		1	
Typha latifolia								1
Vaccinium oxycoccos		40						
Valeriana officinalis		0.3	0.5	1	2		3	
Vicia cracca	1							

Site	Scragh Bog (2)						
Releve code	SB_09	SB_10	SB_11	SB_12	SB_13	SB_14b	SB_15
Sample site code	12	13	15	612	16	17	18
Easting ITM	642148.0	642188.0	642216.0	642315.0	642682.0	642611.0	642534.0
Northing ITM	759355.0	759368.0	759396.0	759105.0	758512.0	758710.0	758798.0
Distance from sample point	2	2	2	2	2	2	2
Quadrat size (m2)	4	4	4	4	4	4	4
Annex	7230	7230	None	7230	None	None	91D0
Fossitt	PF1	PF1	FS2	PF1	FW4	WN7	WN7
Fen quality	Good	Good	Not fen	Good	Not fen	Not fen	Not fen
% surface water	15	30	35	20	5	0	0
% bare ground	0	0	0	0	0	0	0
% total vegetation	90	100	100	100	100	85	97
% grass	0.3	0	0.7	0.3	80	0.1	0.5
% sedge	60	35	5	50	0	0	45
% herb	20	5	30	10	30	1	10
% bryophyte	70	90	95	95	0	85	85
% shrub	1	0	0	0	5	5	12
% canopy	0	0	0	0	70	90	40
% rush	7	3	0.5	0	0	0	0
% litter	3	3	0.1	3	30	15	7
Veg max ht Q1	78	81	40	75	40	20	210
Veg max ht Q2	87	66	57	60	35	99	220
Veg max ht Q3	80	72	49	71	35	130	85
Veg max ht Q4	83	81	51	77	40	10	77
Height (cm)	78 to 87	66 to 81	40 to 57	60 to 77	35 to 40	10 to 130	77 to 220
Species No.	25	18	19	22	12	17	18
Agrostis stolonifera			0.5			0.1	
Angelica sylvestris	0.5	0.5		1			
Aulacomnium palustre							1
Betula pubescens					5	10	45
Brachythecium rutabulum						0.1	
Calliergon cordifolium			0.5				
Calliergon giganteum	0.1	0.1		5			
Calliergonella cuspidata		0.3	95	3			
Caltha palustris	0.1		0.1	0.5			
Campylium stellatum	30	3		10			
Cardamine pratensis	0.3	0.3	0.3	0.1			
Carex diandra			5				
Carex lasiocarpa				15			
Carex limosa	0.7						
Carex nigra			0.3				
Carex panicea							0.1
Carex rostrata							7
Carex viridula ssp. brachyrrhyncha	0.3	0.3		0.1			

Chrysosplenium oppositifolium					30		
Cirriphyllum piliferum						0.1	
Comarum palustre			15				
Ctenidium molluscum		1					
Dactylorhiza sp.	0.3	0.5		0.3			
Dicranum scoparium						0.1	
Dryopteris dilatata						0.1	
Epilobium palustre			1	0.1			
Equisetum fluviatile	0.3	0.1	0.3	0.3			0.3
Eriophorum vaginatum							40
Fagus sylvatica					3		
Filipendula ulmaria	0.1		10	0.1			
Fraxinus excelsior					20		
Galium palustre		0.1	1				
Galium uliginosum	0.5	0.1					
Geranium robertianum					1		
Hedera hibernica					1	1	
Holcus lanatus			0.3		80		
Hydrocotyle vulgaris	0.3						
Hylocomium splendens							5
Hypnum cupressiforme						0.3	0.5
Hypnum resupinatum							0.1
Ilex aquifolium						3	
Juncus acutiflorus	7	3	0.5				
Kindbergia praelonga						0.1	
Lepidozia reptans						0.1	
Lonicera periclymenum					0.3		
Mentha aquatica	0.3		0.3	0.1		0.1	
Menyanthes trifoliata	0.5	3	1	7			
Microlejeunea ulicina							0.1
Mnium hornum						0.1	
Molinia caerulea	0.3			0.3			0.5
Picea sitchensis					40		
Potamogeton coloratus				0.5			
Pyrola rotundifolia	0.3						
Quercus robur							0.3
Rubus fruticosus					5		
Salix repens	0.5						
Salix x multinervis	0.7						
Schoenus nigricans	60	35		35			
Scleropodium purum							
Scorpidium cossonii	0.3	65		75			
Scorpidium scorpioides	40	20		1			
Silene flos-cuculi			0.3				

Appendix C. Vegetation surveys relevé data

Sorbus aucuparia					1	3	
Sphagnum angustifolium							1
Sphagnum contortum	0.3						
Sphagnum fallax							80
Sphagnum palustre							1
Sphagnum squarrosus						15	
Salic cinerea					80	5	
Succisa pratensis	0.3	1	0.3	0.5			
Taraxacum agg.				0.1			
Thuidium tamariscinum						70	0.1
Vaccinium myrtillus							5
Vaccinium oxycoccos	15	1		0.7			10
Valeriana officinalis			0.1				

Site	Tory Hill			
Relevé code	T1	T35	T13	T14
Sample site code	TH6	TH12, TH13, TH 14	TH8	TH7
Easting ITM	553491.6	553411.2	553589.2	553533.0
Northing ITM	643324.9	642965.1	643271.7	643269.7
Distance from sample point	2	64	59	13
Quadrat size (m2)	4	4	4	4
Annex	7230	7230	none	none
Fossitt	PF1	PF1	GM1	GM1
Fen quality	Poor	Poor	Not fen	Not fen
% surface water	0	0	0	0
% bare ground	1	1	0	0
% total vegetation	99	99	100	100
% grass	0	0	0	0
% sedge	0	0	0	0
% herb	95	95	100	100
% bryophyte	30	20	0	0
% shrub	8	0	0	0
% canopy	0	0	0	0
% rush	0	0	0	0
% litter	0	0	0	0
Veg max ht Q1				
Veg max ht Q2				
Veg max ht Q3				
Veg max ht Q4				
Height (cm)	30 to 80	50 to 70	60 to 90	40 to 60
NOTES				
Species No.	16	18	13	14
Agrostis stolonifera	20	20	7	7
Arrhenatherum elatius			30	
Calliergonella cuspidata	30			
Carex diandra	7			
Carex flacca				3
Carex nigra	3			
Carex panicea		3		0.5
Carex viridula brachyrrhyncha		1.5		
Dactylis glomerata			7	
Epilobium palustre	0.5	0.5		
Equisetum fluviatile	3			3
Equisetum palustre	1.5	3		7
Eupatorium cannabinum		1.5		
Festuca pratensis			7	3
Festuca rubra		3	7	
Filipendula ulmaria	0.5	20	30	20
Galium palustre	3	1.5		
Holcus lanatus	3		3	
Juncus subnodulosus	42	42		7

Lathyrus pratensis			20	3
Lythrum salicaria	3	3	7	7
Mentha aquatica	20	20		
Menyanthes trifoliata	30			
Molinia caerulea				42
Persicaria amphibium	7			
Phragmites australis	7	3	3	
Plagiomnium undulatum		7	0.5	
Scutellaria galericulata		3		
Stachys palustris			3	
Succisa pratensis	7	7		7
Valeriana officinalis	0.5	0.5		0.5
Vicia cracca			3	3

Appendix D. Draft assessment criteria for Annex I fen habitats and fen quality assessments

(1) Draft Assessment criteria for Annex I fen habitats surveyed during Pilot Fen Survey 2019-2020

7140 Transition mires

	Assessment criteria	Scale of assessment
Vegetation composition		
1a†	FE2B: number of positive indicator species from Groups i or ii present ≥ 3	Relevé
1b†	FE2E: number of positive indicator species from Groups i or ii present ≥ 3	
1c†	FE2C/D: number of positive indicator species from Groups i or ii present ≥ 6 (see below)	
2	Number of species from Group i present** ≥ 1 (see below)	Relevé
3	Cover of the following species: small to medium sized *Carex* spp., *Equisetum fluviatile*, *Hydrocotyle vulgaris*, *Hypericum elodes*, *Mentha aquatica*, *Menyanthes trifoliata*, *Potentilla palustris*, *Sphagnum* spp. collectively ≥ 25%	Relevé
4	Cover of the following species: *Anthoxanthum odoratum*, *Epilobium hirsutum*, *Holcus lanatus* collectively < 1%	Relevé
5	Cover of non-native species < 1%	Relevé
Vegetation structure		
6	†FE2C/D/E: ≥ 50% of the tips of live leaves and/or flowering shoots of vascular plants should be more than 15 cm above the ground surface	Relevé
Physical structure		
7	Cover of disturbed bare ground < 10%	Relevé
8	Cover of disturbed bare ground < 10%	Local vicinity
9	Area showing signs of drainage resulting from heavy trampling or tracking or ditches < 10%	Local vicinity

†Assess only the criteria relevant to the community being assessed.
Typical species for 7140:

FE2B (infilling pool community)	**FE2C/D/E (*Carex rostrata* fen and flush)**	
Group i	**Group i**	
Carex canescens	*Caltha palustris*	*Equisetum fluviatile*
Carex lasiocarpa	*Carex diandra*	*Galium palustre*
Carex limosa	*Carex lasiocarpa*	*Mentha aquatica*
Menyanthes trifoliata	*Carex limosa*	*Menyanthes trifoliata*
Sphagnum spp. (count separately)	*Carex nigra*	*Phragmites australis*
	Carex rostrata	*Potentilla palustris*
Group ii	*Carex viridula*	*Sphagnum* spp. (count separately)
Drosera spp. (count separately)	*Epilobium palustre*	
Eriophorum angustifolium	**Group ii**	
Rhynchospora spp. (count separately)	*Angelica sylvestris*	*Lythrum salicaria*
	Cardamine pratensis	*Selaginella selaginoides*
	Eriophorum angustifolium	*Succisa pratensis*
	Lysimachia vulgaris	*Viola palustris*

7210 *Cladium* fens and swamps

	Assessment Criteria	7210 (fen variant)	7210 (swamp variant)	Scale of assessment
Vegetation composition				
		Value required to pass	*Value required to pass*	
1	*Cladium mariscus* present	Y	Y	Relevé
2	Number of brown mosses	≥ 1	Not assessed	Relevé
3	No. of positive vascular indicator species	≥ 3	≥ 2	Relevé
4	Cover of brown mosses and positive vascular indicator species	≥ 75%	≥ 75%	Relevé
5a	Total cover of *Anthoxanthum odoratum, Epilobium hirsutum, Holcus lanatus, Ranunculus repens*	<1%	Not assessed	Relevé
5b	Total cover of *Epilobium hirsutum* and *Typha latifolia*	Not assessed	<5%	Relevé
6	Cover of non-native species	<1%	<1%	Relevé
7	Cover of scattered native trees / shrubs	<10%	<10%	Local vicinity
8	Total cover of *Juncus effusus* and *Phragmites australis* < 10%	< 10%	Not assessed	Local vicinity
Vegetation structure				
9a	Live shoots > 5 cm high	≥50%	Not assessed	Relevé
9b	Live shoots > 1 m high	Not assessed	≥50%	
Physical structure				
10	Cover of disturbed bare ground	< 10%	< 10%	Relevé
11	Cover of disturbed bare ground	< 10%	< 10%	Local vicinity
12	Area showing signs of drainage resulting from heavy trampling or ditches	< 10%	< 10%	Local vicinity
13	Disturbed vegetation (if tufa present)	< 1%	< 1%	Relevé

Typical species for 7210:

FW3H/FE1 (fen variant)

Brown mosses	**Vascular plants**
Bryum pseudotriquetrum	*Anagallis tenella*
Calliergon sarmentosum	*Carex dioica*
Campylium stellatum	*Carex lasiocarpa*
Ctenidium molluscum	*Carex panicea*
Drepanocladus revolvens	*Carex viridula*
Drepanocladus cossonii	*Carex rostrata*
Fissidens adianthoides	*Cirsium dissectum*
Palustriella commutata	*Eleocharis quinqueflora*
Palustriella falcata	*Juncus bulbosus*
Scorpidium scorpioides	*Molinia caerulea*
	Pinguicula vulgaris
	Schoenus nigricans
	Selaginella selaginoides

FW3H (swamp variant)

Carex lasiocarpa
Phragmites australis
Equisetum fluviatile
Lemna trisulca
Potentilla palustris
Menyanthes trifoliata

7230 Alkaline fens

	Assessment Criteria	Scale of assessment
Vegetation composition		
1	At least one brown moss species present	Relevé
2a†	FE1C number of positive vascular indicator species present ≥ 2	Relevé
2b†	FE1A/B : number of positive vascular indicator species present ≥ 3	Relevé
3a†	FE1C : vegetation cover of brown mosses and vascular indicator species ≥ 20%	Relevé
3b†	FE1A/B: vegetation cover of brown mosses and vascular indicator species ≥ 75%	Relevé
4	Total cover of the following species: *Anthoxanthum odoratum, Epilobium hirsutum, Holcus lanatus, Ranunculus repens* < 1%	Relevé
5	Cover of non-native species < 1%	Relevé
6	Cover of scattered native trees and scrub < 10%	Local vicinity
7	Total cover of *Juncus effusus* and *Phragmites australis* < 10%	Local vicinity
Vegetation structure		
8	At least 50% of the live leaves/flowering shoots are more than 5 cm above ground surface	Relevé
Physical structure		
9	Cover of disturbed, bare ground < 10%	Relevé
10	Cover of disturbed, bare ground < 10%	Local vicinity
11	Area showing signs of drainage resulting from ditches or tracking < 10%	Local vicinity
12	Where tufa is present, disturbed proportion of vegetation cover < 1%	Local vicinity

†Assess only the criteria relevant to the provisional community (see below) being assessed.

Typical species:

Brown mosses

Bryum pseudotriquetrum
Calliergon sarmentosum
Campylium stellatum
Ctenidium molluscum
Drepanocladus revolvens
Fissidens adianthoides
Palustriella commutata
Palustriella falcata
Scorpidium cossonii
Scorpidium scorpioides

FE1A/B (*Schoenus* flushes and fens)

Anagallis tenella
Carex dioica
Carex lasiocarpa
Carex panicea
Carex viridula
Carex rostrata
Cirsium dissectum
Molinia caerulea
Pinguicula vulgaris
Schoenus nigricans
Selaginella selaginoides

FE1C (small-sedge flushes and fens)

Carex panicea
Carex viridula
Eleocharis quinqueflora
Juncus bulbosus
Pinguicula vulgaris

(2) Quality assessment: 7230 Alkaline Fen – PF1 Rich fen and flush/PF2 Poor fen and flush

	Site Name		Scragh Bog		Scragh Bog		Scragh Bog		Tory Hill		Tory Hill		Pollardstown	
	Relevé Number		SB_09		SB_10		SB_12		T1		T35		PF_01	
	Sample site		SC12		SC13		SC612		TH6		TH12, TH13, TH14		PA31	
	Fossitt community		PF1		PF1		PF1		PF1		PF1		PF2	
	Date		03/09/19		03/09/19		03/09/19		2015		2015		20/08/19	
	Surveyor		KMN/GS		KMN/GS		KMN/GS		JC		JC		RH/KMN	
	Assessment Criteria	**Scale**	**Result**	**Pass/Fail**	**Result**	**Pass/Fail**	**Result**	**Pass/Fail**	**Result**	**Pass/Fail**	**Result**	**Pass/Fail**	**Result**	**Pass/Fail**
	Vegetation composition													
1	At least one brown moss species present	R	3	P	3	P	3	P	0	F	0	F	3	P
2a*	FE1C number of positive vascular indicator species present ≥ 2	R												
2b*	FE1A/B : number of positive vascular indicator species present ≥ 3	R	3	P	3	P	3	P	0	F	2	F	3	P
3a*	FE1C : vegetation cover of brown mosses and vascular indicator species ≥ 20%	R												
3b*	FE1A/B: vegetation cover of brown mosses and vascular indicator species ≥ 75%	R	83	P	90	P	88	P	0	F	4.5	F	70	F
4	Total cover of the following species: Anthoxanthum odoratum, Epilobium hirsutum, Holcus lanatus, Ranunculus repens < 1%	R	0	P	0	P	0	P	3	F	0	P	1	P
5	Cover of non-native species < 1%	R	0	P	0	P	0	P	0	P	0	P	0	P
6	Cover of scattered native trees and scrub < 10%	LV	2	P	5	P	0.5	P	20	F	5	P	0	P
7	Total cover of Juncus effusus and Phragmites australis < 10%	LV	0	P	0	P	0	P	7	P	3	P	1	P
	Vegetation structure													
8	At least 50% of the live leaves/flowering shoots are more than 5 cm above ground surface	R	82	P	75	P	70	P	70	P	70	P	112	P
	Physical structure													
9	Cover of disturbed, bare ground < 10%	R	0	P	0	P	0	P	0	P	0	P	0	P
10	Cover of disturbed, bare ground < 10%	LV	0	P	0	P	0	P	0	P	0	P	0	P
11	Area showing signs of drainage resulting from ditches or heavy trampling or tracking < 10%	LV	0	P	0	P	0	P	0	P	0	P	0	P
12*	Where tufa is present, disturbed proportion of vegetation cover < 1%	LV	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
	Notes													
	* R = Relevé; LV = Local vicinity			Pass		Pass		Pass		Fail		Fail		Fail
	After expert judgement													Marginal pass

(2) Quality assessment: 7230 Alkaline Fen – PF1 Rich fen and flush/PF2 Poor fen and flush

	Site Name		Pollardstown		Pollardstown		Pollardstown		Pollardstown		Pollardstown		Pollardstown	
	Relevé Number		PF_02		PF_03		PF_04		PF_08		PF_11		PF_10	
	Sample site		PA34		PA37		PA38		PD27		PD30		PA29	
	Fossitt community		PF2		PF2		PF2		PF2		PF2		PF2	
	Date		20/08/19		20/08/19		20/08/19		20/08/19		20/08/19		20/08/19	
	Surveyor		RH/KMN		RH/KMN		RH/KMN		RH/KMN		RH/KMN		RH/KMN	
	Assessment Criteria	**Scale**	**Result**	**Pass/ Fail**	**Result**	**Pass/ Fail**	**Result**	**Pass/ Fail**	**Result**	**Pass /Fail**	**Result**	**Pass/ Fail**	**Result**	**Pass/ Fail**
	Vegetation composition													
1	At least one brown moss species present	R	0	F	3	P	1	P	2	P	1	P	0	F
2a*	FE1C number of positive vascular indicator species present ≥ 2	R												
2b*	FE1A/B : number of positive vascular indicator species present ≥ 3	R	3	P	3	P	4	P	3	P	3	P	3	P
3a*	FE1C : vegetation cover of brown mosses and vascular indicator species ≥ 20%	R												
3b*	FE1A/B: vegetation cover of brown mosses and vascular indicator species ≥ 75%	R	34	F	65	F	28	F	48	F	71	F	43	F
4	Total cover of the following species: Anthoxanthum odoratum, Epilobium hirsutum, Holcus lanatus, Ranunculus repens < 1%	R	0.5	P	0.3	P	0	P	10	F	0	P	0.5	P
5	Cover of non-native species < 1%	R	0	P	0	P	0	P	0	P	0	P	0	P
6	Cover of scattered native trees and scrub < 10%	LV	0	P	0	P	0	P	0	P	0	P	0	P
7	Total cover of Juncus effusus and Phragmites australis < 10%	LV	0.5	P	1	P	3	P	1	P	0.5	P	30	F
	Vegetation structure													
8	At least 50% of the live leaves/flowering shoots are more than 5 cm above ground surface	R	85	P	103	P	120	P	110	P	128	P	150	P
	Physical structure													
9	Cover of disturbed, bare ground < 10%	R	0	P	0	P	0	P	0	P	0	P	0	P
10	Cover of disturbed, bare ground < 10%	LV	0	P	0	P	0	P	0	P	0	P	0	P
11	Area showing signs of drainage resulting from ditches or heavy trampling or tracking < 10%	LV	0	P	0	P	0	P	0	P	0	P	0	P
12*	Where tufa is present, disturbed proportion of vegetation cover < 1%	LV	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
	Notes													
	* R = Relevé; LV = Local vicinity			Fail		Fail		Fail		Fail		Fail		Fail
	After expert judgement					Marginal pass						Marginal pass		

(2) Quality assessment: 7140 Transition mires – PF3 Transitions mires and quaking bogs

	Site		Ballymore		Ballymore		Ballymore		Ballymore		Ballymore	
	Relevé Number (BEC)		B10		B15		B17		B19		B23	
	Sample site		BM162		BM41		BM4		BM2		BM8	
	Fossitt community		PF3		PF3		PF3		PF3		PF3	
	Date		Jul-Sep 2017		Jul-Sep 2017		Jul-Sep 2017		Jul-Sep 2017		Jul-Sep 2017	
	Surveyor		JC		JC		JC		JC		JC	
	Assessment Criteria	**Scale**	**Result**	**Pass/Fail**	**Result**	**Pass/Fail**	**Result**	**Pass/Fail**	**Result**	**Pass/Fail**	**Result**	**Pass/Fail**
	Vegetation composition											
1a*	FE2B: number of positive indicator species from Groups i or ii present ≥ 3	R	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
1b*	FE2E: number of positive indicator species from Groups i or ii present ≥ 3	R	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
1c*	FE2C/D: number of positive indicator species from Groups i or ii present ≥ 6	R	7	P	6	P	7	P	8	P	7	P
2	Number of species from Group i present ≥ 1	R	5	P	4	P	4	P	5	P	5	P
3	Cover of the following species: small to medium sized Carex spp., Equisetum fluviatile, Hydrocotyle vulgaris, Hypericum elodes, Mentha aquatica, Menyanthes trifoliata, Potentilla palustris, Sphagnum spp. collectively ≥25%	R	42	P	42	P	30	P	33	P	28	P
4	Cover of the following species: Anthoxanthum odoratum, Epilobium hirsutum, Holcus lanatus collectively <1%	R	1	P	0	P	3	F	0	P	0	P
5	Cover of non-native species <1%	R	0	P	0	P	0	P	0	P	0	P
	Vegetation structure											
6	†FE2C/D/E: ≥ 50% of the tips of live leaves and/or flowering shoots of vascular plants should be more than 15 cm above the ground surface	R	40	P	20	P	25	P	20	P	25	P
	Physical structure											
7	Cover of disturbed bare ground <10%	R	0	P	0	P	0	P	0	P	0	P
8	Cover of disturbed bare ground <10%	LV	0	P	0	P	0	P	0	P	0	P
9	Area showing signs of drainage resulting from heavy trampling or tracking or ditches <10%	LV	0	P	0	P	0	P	0	P	0	P
	Notes											
	* R = Relevé; LV = Local vicinity After expert judgement			Pass		Pass		Fail		Pass		Pass

(2) Quality assessment: 7140 Transition mires – PF3 Transitions mires and quaking bogs

	Site		Ballymore		Ballymore		Scragh Bog		Scragh Bog	
	Relevé Number (BEC)		B9		B12		SB_04		SB_02	
	Sample site		BM163		BM161		SC2		SC7	
	Fossitt community		PF3		PF3		PF3		PF3	
	Date		Jul-Sep 2017		Jul-Sep 2017		02/09/19		02/09/19	
	Surveyor		JC		JC		RH/GS		RH/GS	
	Assessment Criteria	**Scale**	**Result**	**Pass/Fail**	**Result**	**Pass/Fail**	**Result**	**Pass/Fail**	**Result**	**Pass/Fail**
	Vegetation composition									
1a*	FE2B: number of positive indicator species from Groups i or ii present ≥ 3	R	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
1b*	FE2E: number of positive indicator species from Groups i or ii present ≥ 3	R	3	P	4	P	n/a	n/a	n/a	n/a
1c*	FE2C/D: number of positive indicator species from Groups i or ii present ≥ 6	R	n/a	n/a	n/a	n/a	10	P	8	P
2	Number of species from Group i present ≥ 1	R	3	P	4	P	10	P	7	P
3	Cover of the following species: small to medium sized Carex spp., Equisetum fluviatile, Hydrocotyle vulgaris, Hypericum elodes, Mentha aquatica, Menyanthes trifoliata, Potentilla palustris, Sphagnum spp. collectively ≥25%	R	54	P	57	P	48	P	22	F
4	Cover of the following species: Anthoxanthum odoratum, Epilobium hirsutum, Holcus lanatus collectively <1%	R	0	P	0	P	5	F	0.3	P
5	Cover of non-native species <1%	R	0	P	0	P	0	P	0	P
	Vegetation structure									
6	†FE2C/D/E: ≥ 50% of the tips of live leaves and/or flowering shoots of vascular plants should be more than 15 cm above the ground surface	R	35	P	40	P	75	P	105	P
	Physical structure									
7	Cover of disturbed bare ground <10%	R	0	P	0	P	0	P	0	P
8	Cover of disturbed bare ground <10%	LV	0	P	0	P	0	P	0	P
9	Area showing signs of drainage resulting from heavy trampling or tracking or ditches <10%	LV	0	P	0	P	0	P	0	P
	Notes									
	* R = Relevé; LV = Local vicinity			Pass		Pass		Fail		Fail
	After expert judgement									Marginal pass

Reading in and filtering database

```
# joining data
> database = database %>% left_join(Hydrodata)
> database = database %>% left_join(Chemdata)

>
> #calculations
> database$Elevation=database$ObjectToplevel - database$Waterlevel
> database$WaterlevelmAGL=database$Elevation - database$SurfaceElevation
>
> ###############################
>
> #Filtering database
>
> PM = c("Nitrite (mg/l as N)") #Hydrochemical parameter
> sitename = c("Tory Hill")     #Fen site
> Obj = c(                      #Sampled objects
+   "Phreatic well",
+   #"Outlet",
+   #"Lake",
+   #"Turlough",
+   #"Borehole",
+   "Piezometer"
+
+ )
>
> #Seasons
> SeasS = c("Spring/Summer")
> SeasW = c("Autumn/Winter")
>
> GW = c("GW")   #Piezometers
> SW = c("SW")   #Phreatic tube
> BH = c("BH")   #Subsoil piezometers/boreholes/wells
>
>
> #Fossitt habitats
> PF1 = c("PF1")
> PF2 = c("PF2")
> PF3 = c("PF3")
> PF1PF3 = c("PF1/PF3")
> FS1   = c("FS1")
> FS2 = c("FS2")
> FW4   = c("FW4")
> GM1   = c("GM1")
> GS2   = c("GS2")
> GS4   = c("GS4")
> WN7 = c("WN7")
> WS1 = c("WS1")
> WS1PB4 = c("WS1/PB4")
>
> Lim = c(0,1000)
```

Boxplots

```
> ##############   OBJECTTYPE      ###################
> ####################################################
> #No Seasons site basis SW GW BH
>
> png(file = paste("plots/plot",sitename,"NO2 Boxplot.png", sep="_"), wi
dth = 300, height =300) #saving plots
>
>
> ggplot(data = database %>%
+          filter(.,
+                 Parameter == PM,
+                 Sitename == sitename,
+                 !is.na(Value),
+                 ObjectType %in% Obj)
+        #surface/cross section
+        #!is.na(Value)),
+ ) +
+   geom_boxplot(aes(y = Value, x=ObjectType, fill=ObjectType))+
+   labs(title = paste(sitename, PM), subtitle = ("Total"))+
+   theme(  panel.background = element_rect(fill = NA),
+           panel.border = element_rect( fill = NA),
+           panel.grid.minor=element_blank(),
+           axis.title.y = element_blank(),
+           #axis.ticks.x=element_blank()
+           panel.grid.minor.y=element_blank(),
+           legend.title = element_blank(),
+           legend.key = element_rect(colour = NA, fill = NA))+
+   theme(axis.text.x = element_text(angle =0, vjust=0.5))+
+   scale_fill_brewer(palette="Paired", direction = -1)+
+   coord_fixed(ratio = 4.5 , ylim = c(0, 0.6))
>
> dev.off()
>
> ##############Summer
>
> png(file = paste("plots/plot",sitename,"NO2 Summer Boxplot.png", sep="
_"), width = 300, height =300) #saving plots
>
> ggplot(data = database %>%
+          filter(.,
+                 Parameter == PM,
+                 Season == SeasS,
+                 Sitename == sitename,
+                 !is.na(Value),
+                 ObjectType %in% Obj)
+        #surface/cross section
+        #!is.na(Value)),
+ ) +
+   geom_boxplot(aes(y = Value, x=ObjectType, fill=ObjectType)) +
+   labs(title = paste(sitename, PM), subtitle = (SeasS))+
+   theme(  panel.background = element_rect(fill = NA),
+           panel.border = element_rect( fill = NA),
+           panel.grid.minor=element_blank(),
+           axis.title.y = element_blank(),
+           #axis.ticks.x=element_blank()
+           panel.grid.minor.y=element_blank(),
+           legend.title = element_blank(),
+           legend.key = element_rect(colour = NA, fill = NA))+
+   theme(axis.text.x = element_text(angle =0, vjust=0.5))+
+   scale_fill_brewer(palette="Paired", direction = -1)+
+   coord_fixed(ratio = 4.5 , ylim = c(0, 0.6))
>
> dev.off()
```

```
>
>
>
> ##########Winter
>
> png(file = paste("plots/plot",sitename, "NO2 Winter Boxplot.png", sep=
"_"), width = 300, height =300) #saving plots
>
> ggplot(data = database %>%
+          filter(.,
+                 Parameter == PM,
+                 Season == SeasW,
+                 Sitename == sitename,
+                 !is.na(Value),
+                 ObjectType %in% Obj)
+        #surface/cross section
+        #!is.na(Value)),
+ ) +
+   geom_boxplot(aes(y = Value, x=ObjectType, fill=ObjectType)) +
+   labs(title = paste(sitename, PM), subtitle = (SeasW))+
+   theme(  panel.background = element_rect(fill = NA),
+           panel.border = element_rect( fill = NA),
+           panel.grid.minor=element_blank(),
+           axis.title.y = element_blank(),
+           #axis.ticks.x=element_blank()
+           panel.grid.minor.y=element_blank(),
+           legend.title = element_blank(),
+           legend.key = element_rect(colour = NA, fill = NA))+
+   theme(axis.text.x = element_text(angle =0, vjust=0.5))+
+   scale_fill_brewer(palette="Paired", direction = -1)+
+   coord_fixed(ratio = 4.5 , ylim = c(0, 0.6))
>
> dev.off()
>
> ##############       FOSSITT     ##################
> ##################################################
> #No Seasons site basis Fossitt waterlevel
>
> png(file = paste("plots/plot",sitename,"WL mAGL Boxplot.png", sep="_")
, width = 500, height =500) #saving plots
>
>
> ggplot(data = database %>%
+          filter(.,
+                 Sitename == sitename,
+                 !is.na(WaterlevelmAGL),
+                 ObjectType %in% Obj)
+        #surface/cross section
+        #!is.na(Value)),
+ ) +
+   geom_boxplot(aes(y = WaterlevelmAGL, x=Fossitt, fill=ObjectType))+
+   labs(title = paste(sitename, "water level (mAGL)"), subtitle = ("Tot
al"))+
+   theme(  panel.background = element_rect(fill = NA),
+           panel.border = element_rect( fill = NA),
+           panel.grid.minor=element_blank(),
+           axis.title.y = element_blank(),
+           #axis.ticks.x=element_blank()
+           panel.grid.minor.y=element_blank(),
+           legend.title = element_blank(),
+           legend.key = element_rect(colour = NA, fill = NA))+
+   theme(axis.text.x = element_text(angle =0, vjust=0.5))+
+   scale_fill_brewer(palette="Paired", direction = -1)+
+   coord_fixed(ratio = 1.2 , ylim = c(-1, 1))
>
> dev.off()
>
```

```
> ###############Summer
>
> png(file = paste("plots/plot",sitename,"WLmAGL Summer Boxplot.png", se
p="_"), width = 400, height =400) #saving plots
>
> ggplot(data = database %>%
+          filter(.,
+                 Season == SeasS,
+                 Sitename == sitename,
+                 !is.na(WaterlevelmAGL),
+                 ObjectType %in% Obj)
+        #surface/cross section
+        #!is.na(Value)),
+ ) +
+   geom_boxplot(aes(y = WaterlevelmAGL, x=Fossitt, fill=ObjectType)) +
+   labs(title = paste(sitename, "water level (mAGL)"), subtitle = (Seas
S))+
+   theme(  panel.background = element_rect(fill = NA),
+           panel.border = element_rect( fill = NA),
+           panel.grid.minor=element_blank(),
+           axis.title.y = element_blank(),
+           #axis.ticks.x=element_blank()
+           panel.grid.minor.y=element_blank(),
+           legend.title = element_blank(),
+           legend.key = element_rect(colour = NA, fill = NA))+
+   theme(axis.text.x = element_text(angle =0, vjust=0.5))+
+   scale_fill_brewer(palette="Paired", direction = -1)+
+   coord_fixed(ratio = 1.2 , ylim = c(-1, 1))
>
> dev.off()
>
>
>
> ##########Winter
>
> png(file = paste("plots/plot",sitename, "WLmAGL Winter Boxplot.png", s
ep="_"), width = 400, height =400) #saving plots
>
> ggplot(data = database %>%
+          filter(.,
+                 Season == SeasW,
+                 Sitename == sitename,
+                 !is.na(WaterlevelmAGL),
+                 ObjectType %in% Obj)
+        #surface/cross section
+        #!is.na(Value)),
+ ) +
+   geom_boxplot(aes(y = WaterlevelmAGL, x=Fossitt, fill=ObjectType)) +
+   labs(title = paste(sitename, "water level (mAGL)"), subtitle = (Seas
W))+
+   theme(  panel.background = element_rect(fill = NA),
+           panel.border = element_rect( fill = NA),
+           panel.grid.minor=element_blank(),
+           axis.title.y = element_blank(),
+           #axis.ticks.x=element_blank()
+           panel.grid.minor.y=element_blank(),
+           legend.title = element_blank(),
+           legend.key = element_rect(colour = NA, fill = NA))+
+   theme(axis.text.x = element_text(angle =0, vjust=0.5))+
+   scale_fill_brewer(palette="Paired", direction = -1)+
+   coord_fixed(ratio = 1.2 , ylim = c(-1, 1))
>
>
> dev.off()
>
> ###################################################
> #No Seasons Fossitt basis SW GW BH
```

```
>
> png(file = paste("plots/plot",sitename,"NH3fos Boxplot.png", sep="_"),
 width = 300, height =300) #saving plots
>
>
> ggplot(data = database %>%
+          filter(.,
+                 Parameter == PM,
+                 Sitename == sitename,
+                 !is.na(Value),
+                 ObjectType %in% Obj)
+        #surface/cross section
+        #!is.na(Value)),
+ ) +
+   geom_boxplot(aes(y = Value, x=Fossitt, fill=ObjectType))+
+   labs(title = paste(sitename, PM), subtitle = ("Total"))+
+   theme(  panel.background = element_rect(fill = NA),
+           panel.border = element_rect( fill = NA),
+           panel.grid.minor=element_blank(),
+           axis.title.y = element_blank(),
+           #axis.ticks.x=element_blank()
+           panel.grid.minor.y=element_blank(),
+           legend.title = element_blank(),
+           legend.key = element_rect(colour = NA, fill = NA))+
+   theme(axis.text.x = element_text(angle =0, vjust=0.5))+
+   scale_fill_brewer(palette="Paired", direction = -1)+
+   coord_fixed(ratio = 0.6 , ylim = c(0,10))
>
> dev.off()
RStudioGD
        2
>
> ###############Summer
>
> png(file = paste("plots/plot",sitename,"NH3fos Summer Boxplot.png", se
p="_"), width = 300, height =300) #saving plots
>
> ggplot(data = database %>%
+          filter(.,
+                 Parameter == PM,
+                 Season == SeasS,
+                 Sitename == sitename,
+                 !is.na(Value),
+                 ObjectType %in% Obj)
+        #surface/cross section
+        #!is.na(Value)),
+ ) +
+   geom_boxplot(aes(y = Value, x=Fossitt, fill=ObjectType)) +
+   labs(title = paste(sitename, PM), subtitle = (SeasS))+
+   theme(  panel.background = element_rect(fill = NA),
+           panel.border = element_rect( fill = NA),
+           panel.grid.minor=element_blank(),
+           axis.title.y = element_blank(),
+           #axis.ticks.x=element_blank()
+           panel.grid.minor.y=element_blank(),
+           legend.title = element_blank(),
+           legend.key = element_rect(colour = NA, fill = NA))+
+   theme(axis.text.x = element_text(angle =0, vjust=0.5))+
+   scale_fill_brewer(palette="Paired", direction = -1)+
+   coord_fixed(ratio = 0.6 , ylim = c(0,10))
>
> dev.off()
RStudioGD
        2
>
>
>
```

```
> ##########Winter
>


> png(file = paste("plots/plot",sitename, "NH3fos Winter Boxplot.png", s
ep="_"), width = 300, height =300) #saving plots
>
> ggplot(data = database %>%
+          filter(.,
+                 Parameter == PM,
+                 Season == SeasW,
+                 Sitename == sitename,
+                 !is.na(Value),
+                 ObjectType %in% Obj)
+        #surface/cross section
+        #!is.na(Value)),
+ ) +
+   geom_boxplot(aes(y = Value, x=Fossitt, fill=ObjectType)) +
+   labs(title = paste(sitename, PM), subtitle = (SeasW))+
+   theme(  panel.background = element_rect(fill = NA),
+           panel.border = element_rect( fill = NA),
+           panel.grid.minor=element_blank(),
+           axis.title.y = element_blank(),
+           #axis.ticks.x=element_blank()
+           panel.grid.minor.y=element_blank(),
+           legend.title = element_blank(),
+           legend.key = element_rect(colour = NA, fill = NA))+
+   theme(axis.text.x = element_text(angle =0, vjust=0.5))+
+   scale_fill_brewer(palette="Paired", direction = -1)+
+   coord_fixed(ratio = 0.6 , ylim = c(0,10))
>
> dev.off()
```

Welch t-test

```
> ############## Total ##################
> ####Filtering data groups
> ###boreholes
>
> databh = database %>%
+   filter(.,
+          Parameter == PM,
+          Sitename == sitename,
+          !is.na(Value),
+          SWGW == BH,
+          Fossitt == PF1PF3)
>
> testbh = databh$Value
>
> ###preatic tubes
> datasw = database %>%
+   filter(.,
+          Parameter == PM,
+          Sitename == sitename,
+          !is.na(Value),
+          SWGW == SW)
>
> testsw = datasw$Value
>
> ###piezometers
>
> datagw = database %>%
```

```
+   filter(.,
+          Parameter == PM,
+          Sitename == sitename,
+          !is.na(Value),
+          SWGW == GW)
>
> testgw = datagw$Value
>
> #####Testing data groups
> ###Welch t-test BH - GW
> t.test(testbh, testgw, alternative = "greater", var.equal = FALSE)
>
> ###Welch t-test BH - SW
> t.test(testbh, testsw, alternative = "greater", var.equal = FALSE)
```

NMDS plotting

```
#vegetation correlations - default Bray distance

# Packages and data ----------------------------------------------------
---
if (!require(vegan)) install.packages('vegan') # checks for package, ins
talls if missing
if (!require(goeveg)) install.packages('goeveg') # checks for package, i
nstalls if missing
library(vegan) # package for vegetation analysis
library(goeveg) # package for selecting proportion of species for displa
y
veg = read.csv('data/Ecodata/IVCdata.csv', row.names = 1) # read in veg
data which should be in the working directory
env = read.csv('data/Ecodata/Envdata.csv', row.names = 1) # read in env
data which should be in the working directory
env$height = apply(env[,1:4],1, median) # calculate median of max height
s in columns 1 to 4

# Run ordination -------------------------------------------------------
---
set.seed = -2444 # sets the seed for random number generation so ordinat
ion will be same each time
ord = metaMDS(veg, k = 2, try = 100, trymax = 100) # runs 2-dimensional
ordination, ?metaMDS for details
ord # details of methodology and % stress on the solution for reporting
plot(ord, type = 'n') # plot blank graph
cols = c("red","blue", "green", "orange") # colours for each site
site = as.factor(substr(row.names(veg),1,1)) # extract site identities f
rom row names
points(ord, col = cols[site], cex = 1.4, pch = 16) # plot colour-coded q
uadrats
limited = ordiselect(veg, ord, ablim = 0.1) # select 10% of species with
 highest abundances
points(ord, display="species", select = limited, pch=3, col="black", cex
=0.6) # plot species overlay
ordipointlabel(ord, display="species", select = limited, col="black", ce
x=0.6, add = TRUE) # label species
ordipointlabel(ord, display="sites", col = cols[site], cex=0.6, add = TR
UE) # label quadrats
ef = envfit(ord, env[,6:28]) # calculate correlation with env variables
in columns 5 to 15
plot(ef, col = "black", p.max = 0.2) # create biplot for best correlatio
ns
```

Transect generation

```
> ########################################################
>
> #Transect with waterlevel and hydrochemistry
>
> #Variables:
> PM = c("DRP")
> DT = c("2020-02-11")
> TR = c("Ballymore transect 1")
> sitename = c("Ballymore Fen")
> Obj = c(
+   "Phreatic well",
+   "Outlet",
+   "Lake",
+   "Turlough",
+   "Piezometer",
+   "Borehole"
+ )
> GW = c("GW")
> SW = c("SW")
> GL = c ("REFERENCE")
>
> Lim = c(0,0.5)
>
> png(file = paste("plots/plots",TR,DT,"DRP Transect.png", sep="_"), wid
th = 750, height =525) #saving plots
>
> ggplot(data = database %>% #minmax#
+          filter(.,
+                 Abrev == PM,
+                 Date == DT,
+                 Sitename == sitename,
+                 Transect == TR,
+                 ObjectType %in% Obj),
+        #surface/cross section
+        #!is.na(Value)),
+        aes(y = Elevation, x = Distance)) +
+   labs(title = TR, subtitle = DT)+
+   theme(  panel.background = element_rect(fill = NA),
+           panel.border = element_rect( fill = NA),
+           panel.grid.minor=element_blank(),
+           #axis.title.x = element_blank(),
+           #axis.ticks.x=element_blank(),
+           panel.grid.minor.y=element_blank(),
+           legend.key = element_rect(colour = NA, fill = NA))+
+   coord_fixed(ratio = 45, xlim = c(150, 950), ylim = c(85, 100))+
+   geom_point(data = database %>% #nextpart#
+                filter(.,
+                       Abrev == PM,
+                       Date == DT,
+                       Sitename == sitename,
+                       Transect == TR,
+                       SWGW == SW,
+                       ObjectType %in% Obj,
+                       !is.na(Value)),
+              aes(y = SampleElevation, x = Distance, fill=Value),shape
= 24, size =5)+
+   geom_point(data = database %>% #nextpart#
+                filter(.,
+                       Abrev == PM,
+                       Date == DT,
+                       Sitename == sitename,
```

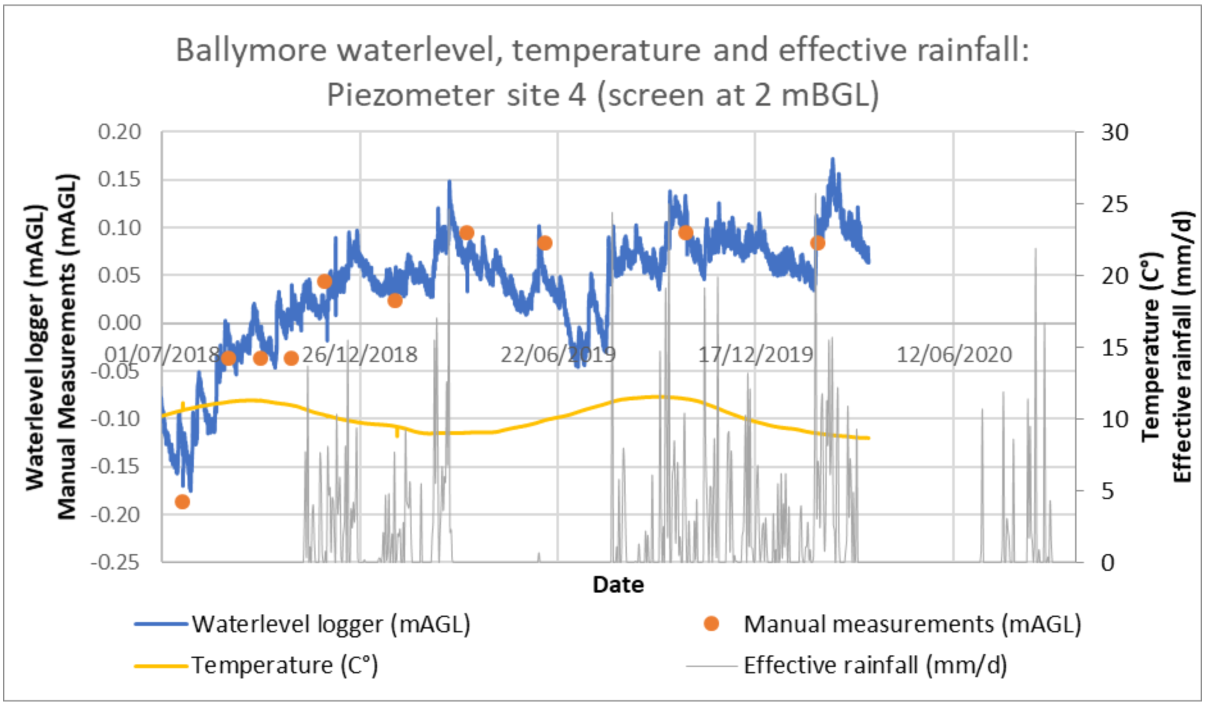
```
+                        Transect == TR,
+                        SWGW == GW,
+                        ObjectType %in% Obj,
+                        !is.na(Value)),
+               aes(y = SampleElevation, x = Distance, fill=Value),shape
= 25, size =3)+
+    geom_point(data = database %>% #nextpart#
+                 filter(.,
+                        Abrev == PM,
+                        Date == DT,
+                        Sitename == sitename,
+                        Transect == TR,
+                        SWGW == BH,
+                        ObjectType %in% Obj,
+                        !is.na(Value)),
+               aes(y = SampleElevation, x = Distance, fill=Value),shape
= 25, size =3)+
+    scale_fill_viridis(direction = -1, limits=c(Lim), name = paste(PM, s
ep =" ")) +
+    geom_text(data = database %>% #Boreholes
+                filter(.,
+                       Abrev == PM,
+                       Date == DT,
+                       Sitename == sitename,
+                       ObjectType %in% Obj,
+                       SWGW == BH,
+                       Transect == TR),size=4, aes(y = ObjectToplevel, x
 = Distance, label= LocationID),hjust=0, vjust=-2) +
+    geom_text(data = database %>% #piezometers
+                filter(.,
+                       Abrev == PM,
+                       Date == DT,
+                       Sitename == sitename,
+                       ObjectType %in% Obj,
+                       SWGW == SW,
+                       Transect == TR),size=4, aes(y = ObjectToplevel, x
 = Distance, label= LocationID),hjust=0.5, vjust=-2) +
+    geom_line(data = database %>% #Surfacelevels
+                filter(.,
+                       Sitename == sitename,
+                       Transect == TR),
+              aes(y = SurfaceElevation, x = Distance), size =0.25, linet
ype = "dotted")+
+    geom_line(data = database %>% #Surfacewaterlevels
+                filter(.,
+                       Abrev == PM,
+                       Date == DT,
+                       Sitename == sitename,
+                       ObjectType %in% Obj,
+                       #!is.na(WaterElevation),
+                       Transect == TR,
+                       SWGW == SW),
+              aes(y = Elevation , x = Distance), colour="lightblue", siz
e =0.75, linetype = "dashed")+
+    geom_point(data = database %>%
+                 filter(.,
+                        Abrev == PM,
+                        Date == DT,
+                        Sitename == sitename,
+                        ObjectType %in% Obj,
+                        Transect == TR,
+                        SWGW == SW),
+               aes(y = Elevation , x = Distance), colour="lightblue", si
ze =3)+
+    geom_point(data = database %>% #Boreholes
+                 filter(.,
+                        Abrev == PM,
```

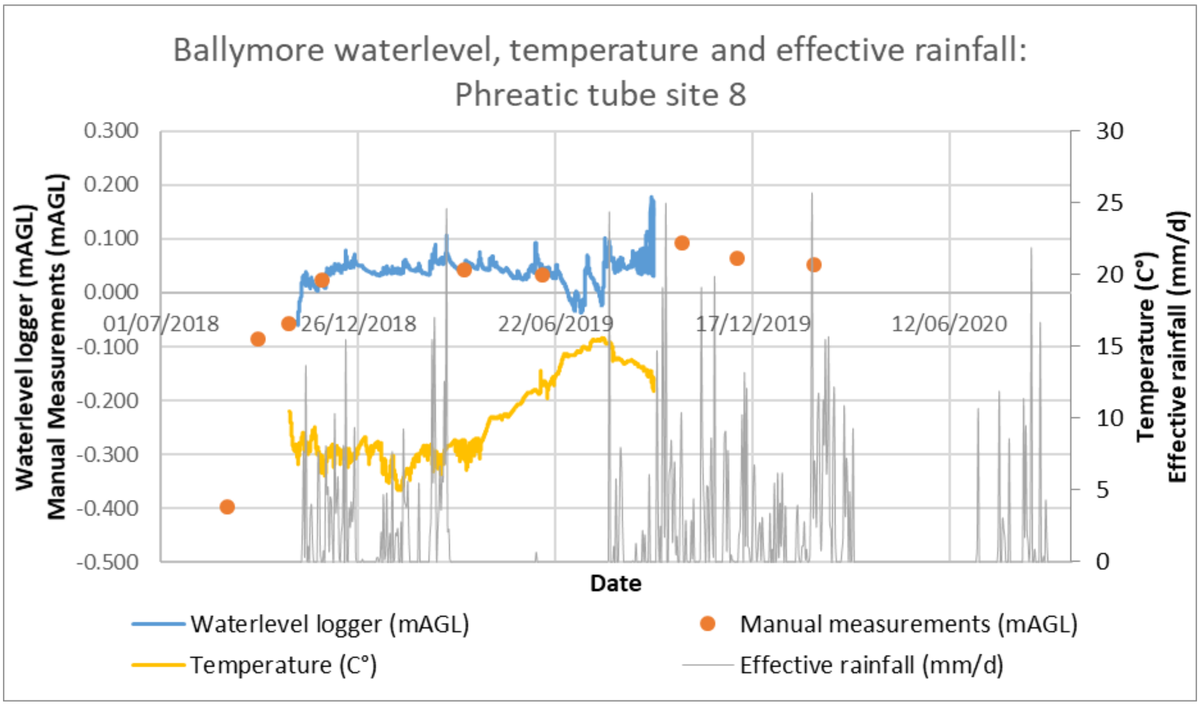
```
+                        Date == DT,
+                        Sitename == sitename,
+                        ObjectType %in% Obj,
+                        Transect == TR,
+                        SWGW == BH),
+               aes(y = Elevation , x = Distance), colour="blue", size =1
.5)+
+   geom_point(data = database %>% #Piezometers
+                filter(.,
+                        Abrev == PM,
+                        Date == DT,
+                        Sitename == sitename,
+                        ObjectType %in% Obj,
+                        Transect == TR,
+                        SWGW == GW),
+               aes(y = Elevation , x = Distance), colour="blue", size =1
.5)
```

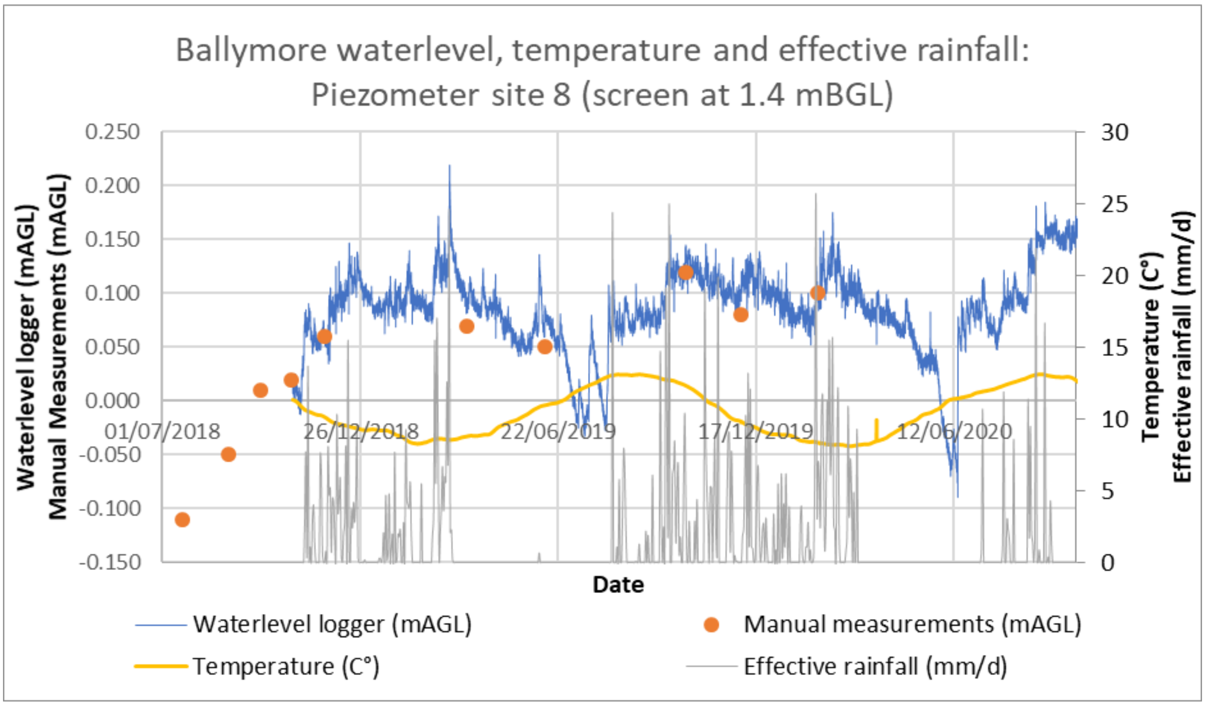
Appendix F. Water level logger time series

Appendix F. Water level logger time series

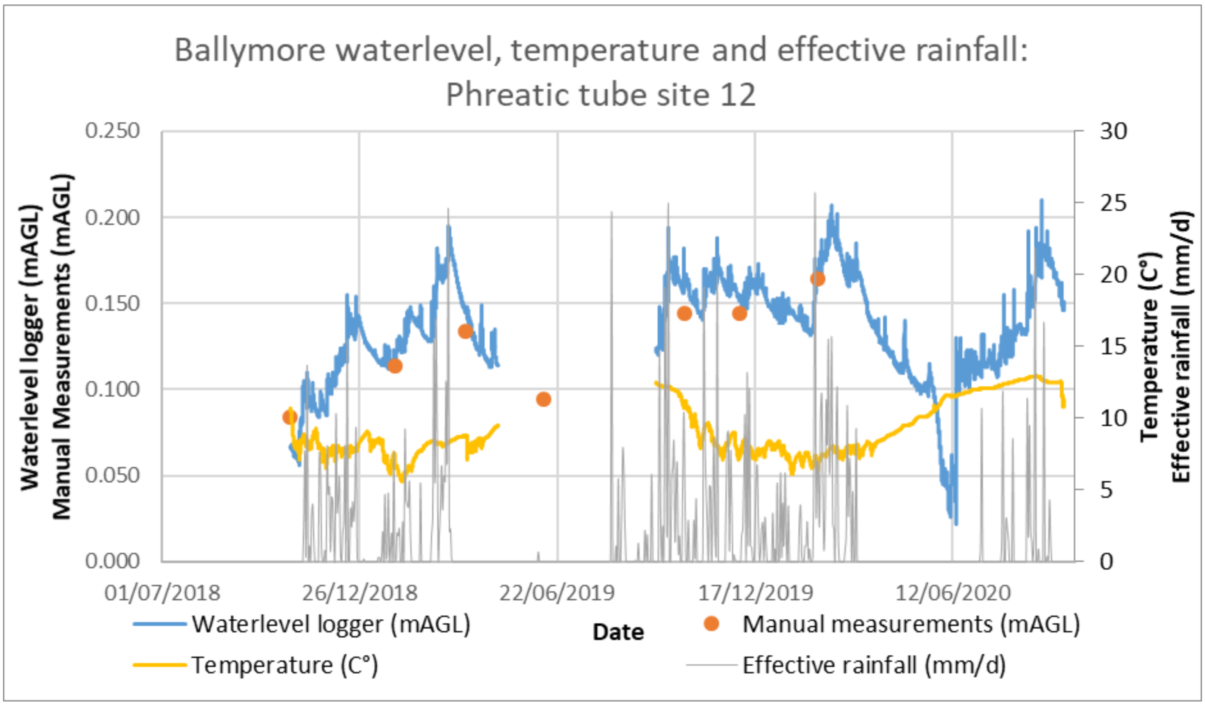
Ballymore water level time-series

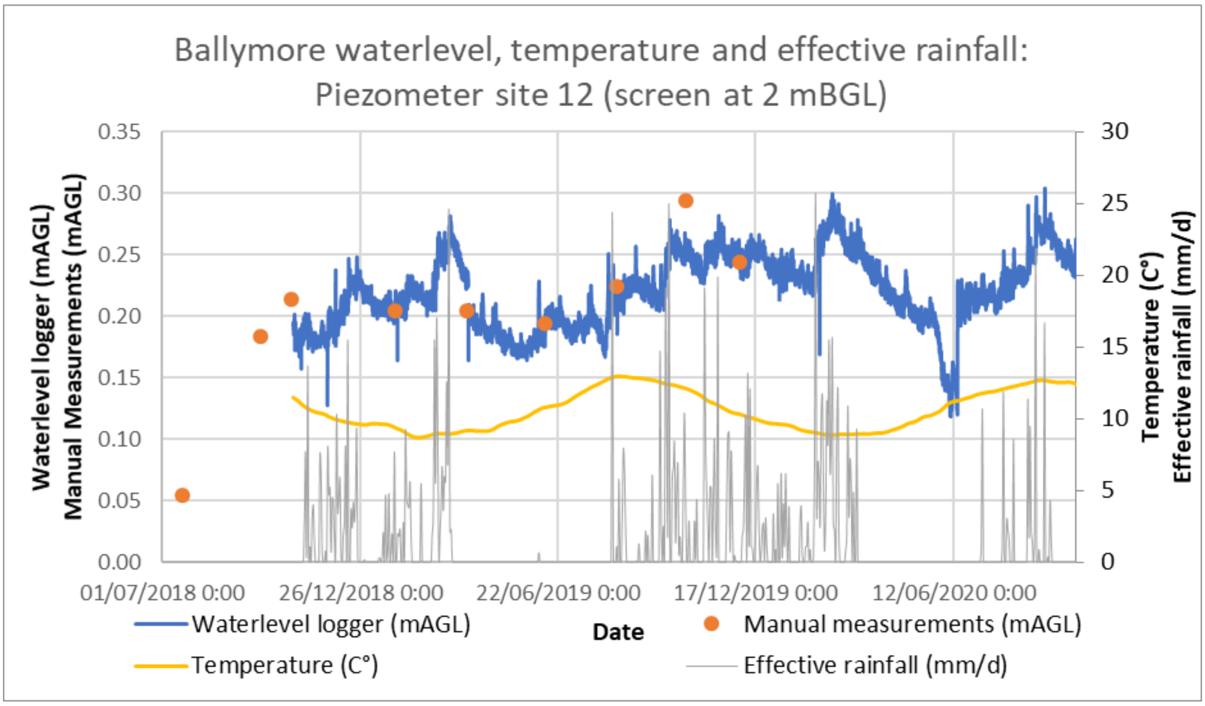




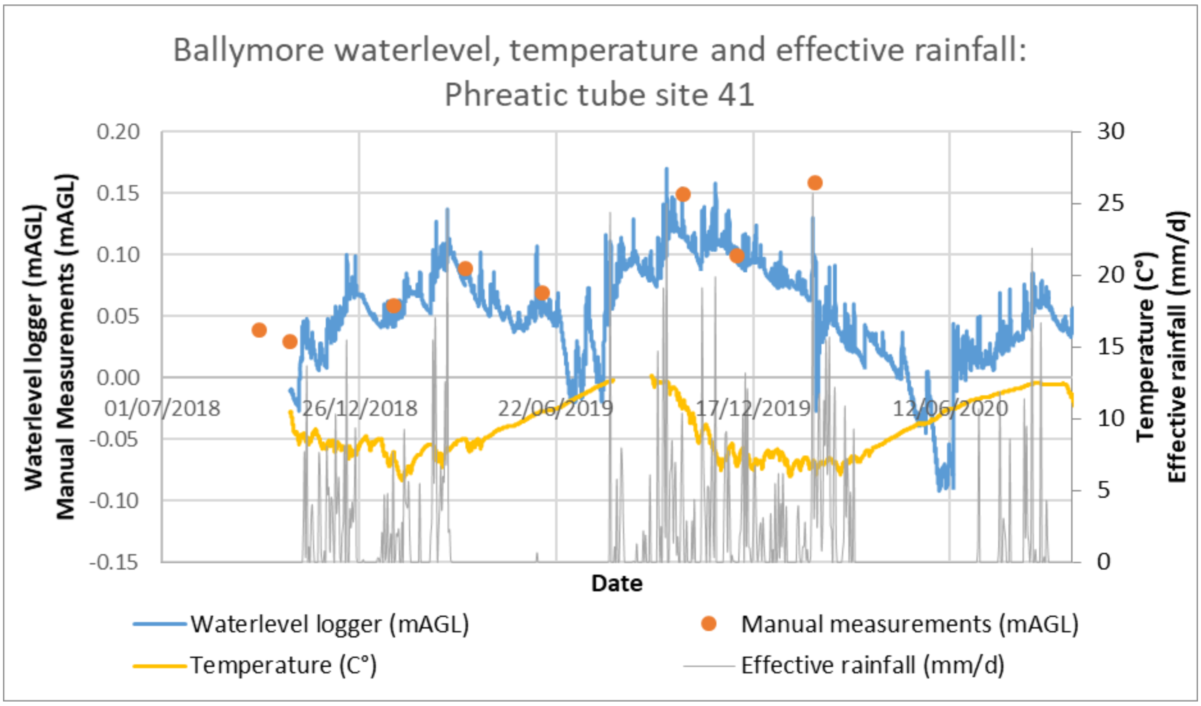


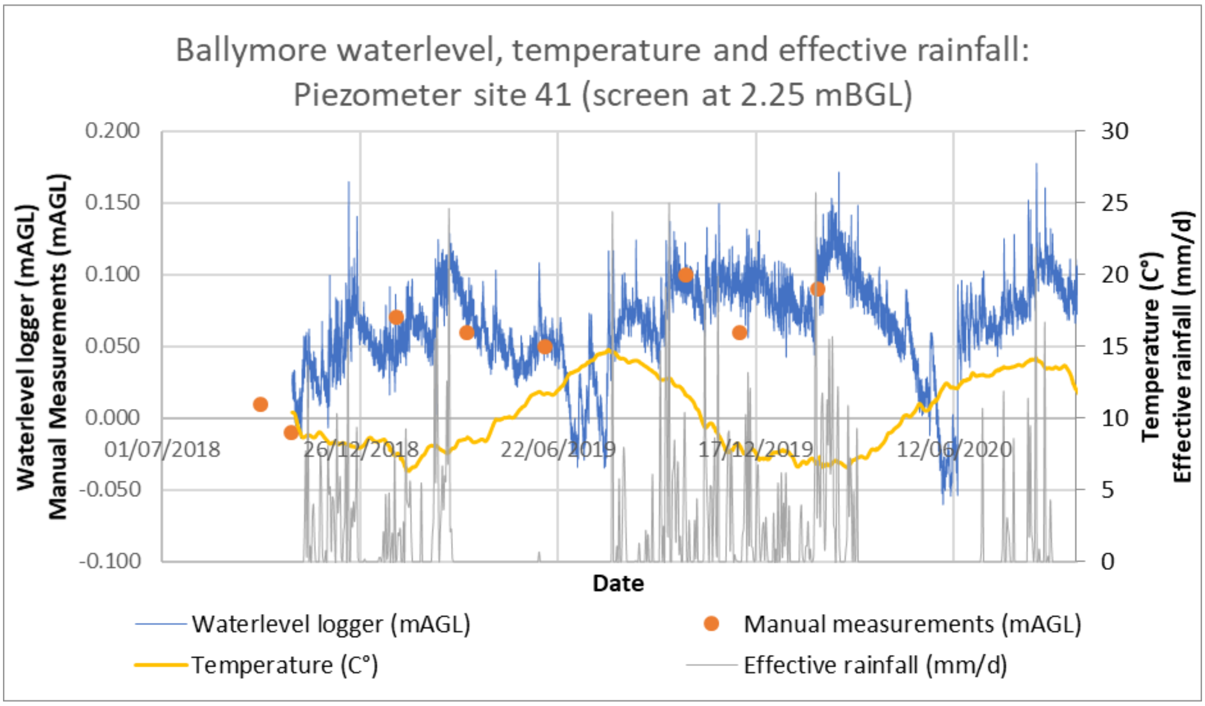
Appendix F. Water level logger time series



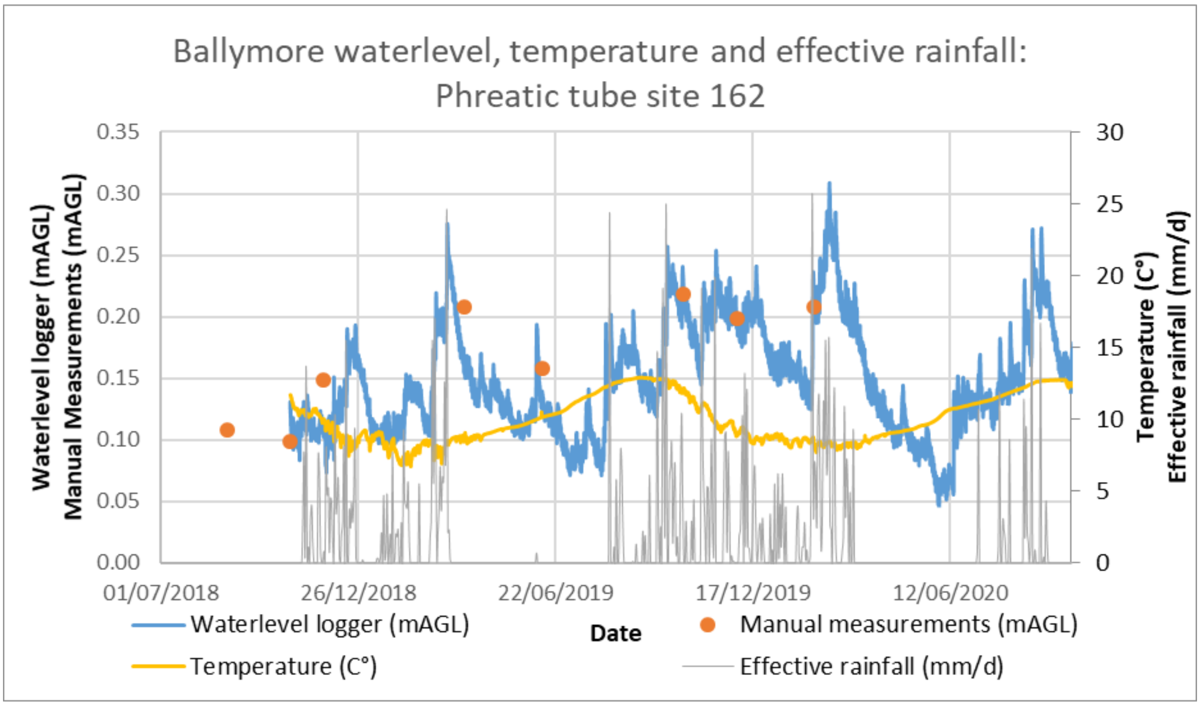


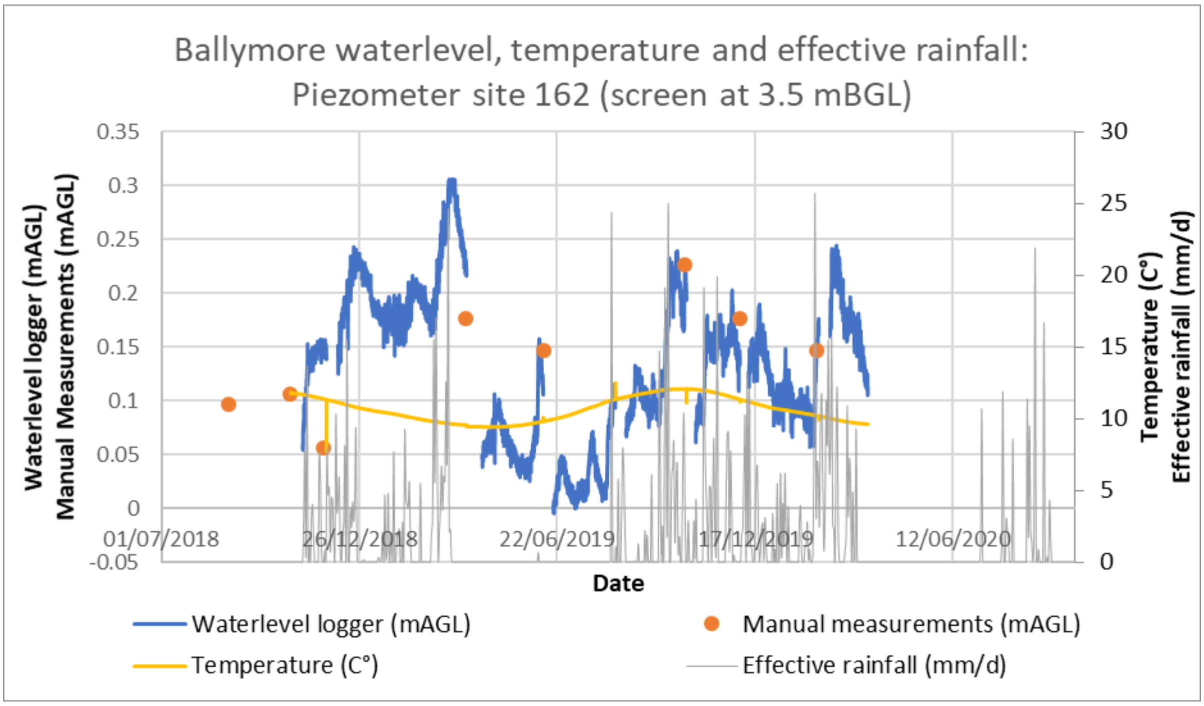
Appendix F. Water level logger time series



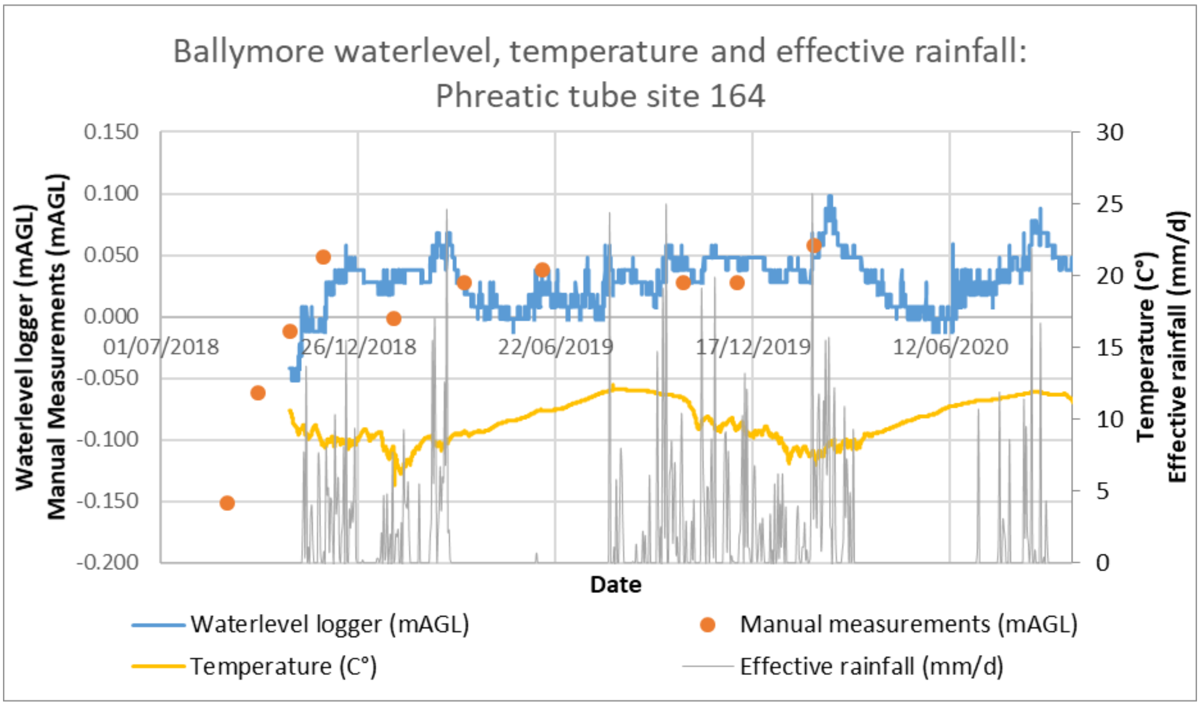


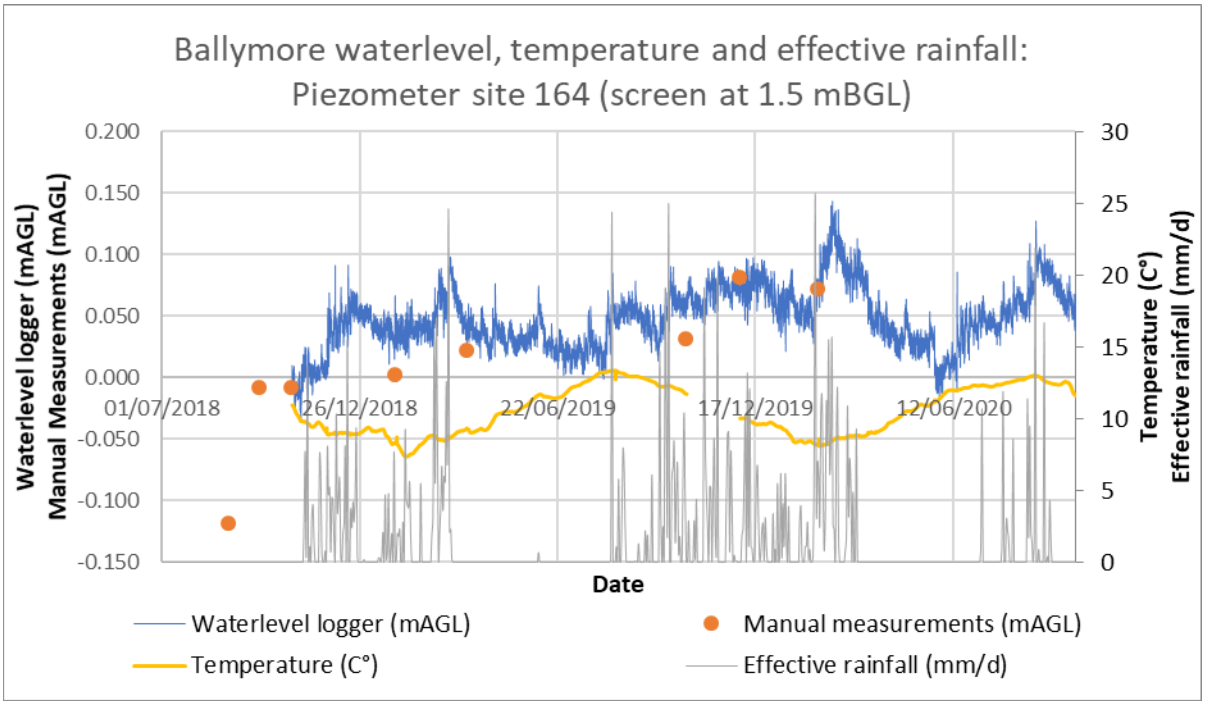
Appendix F. Water level logger time series



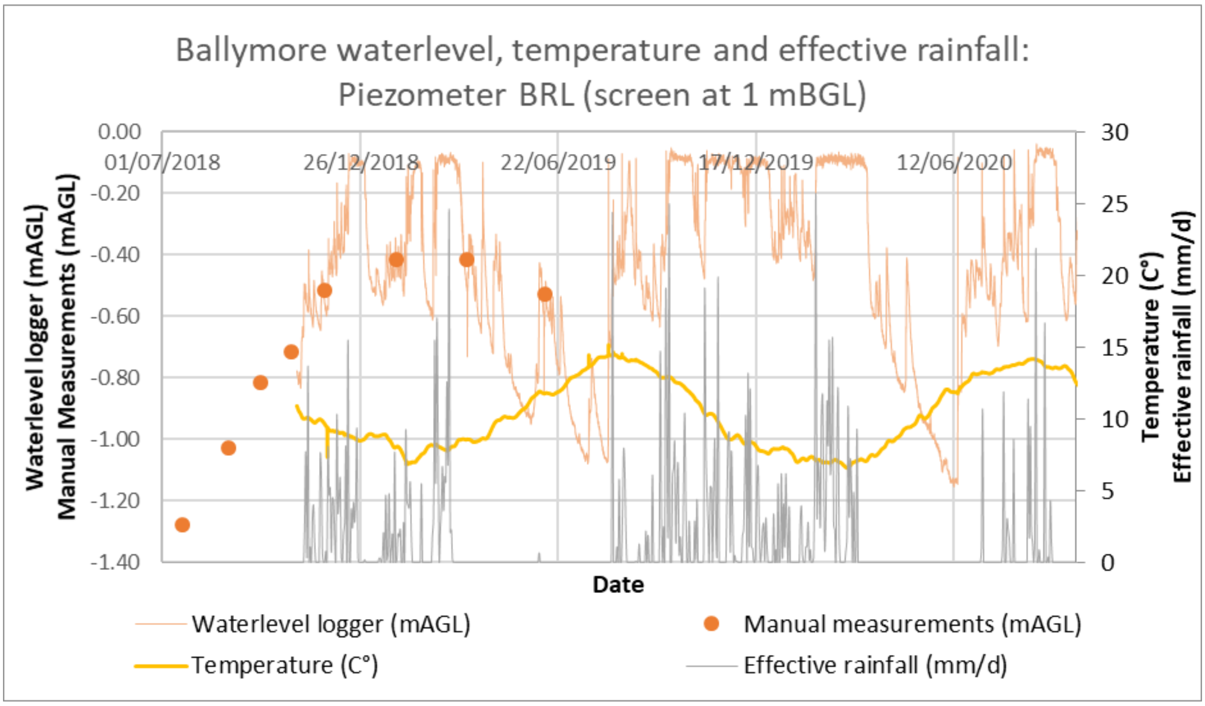


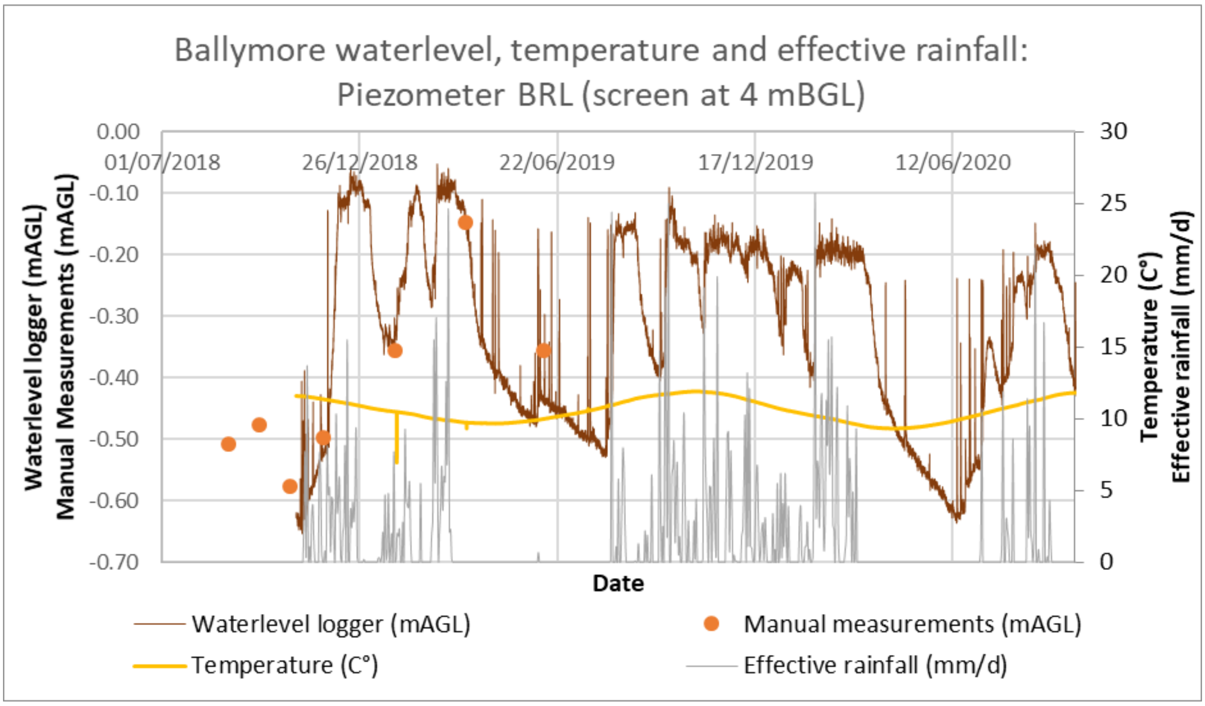
Appendix F. Water level logger time series





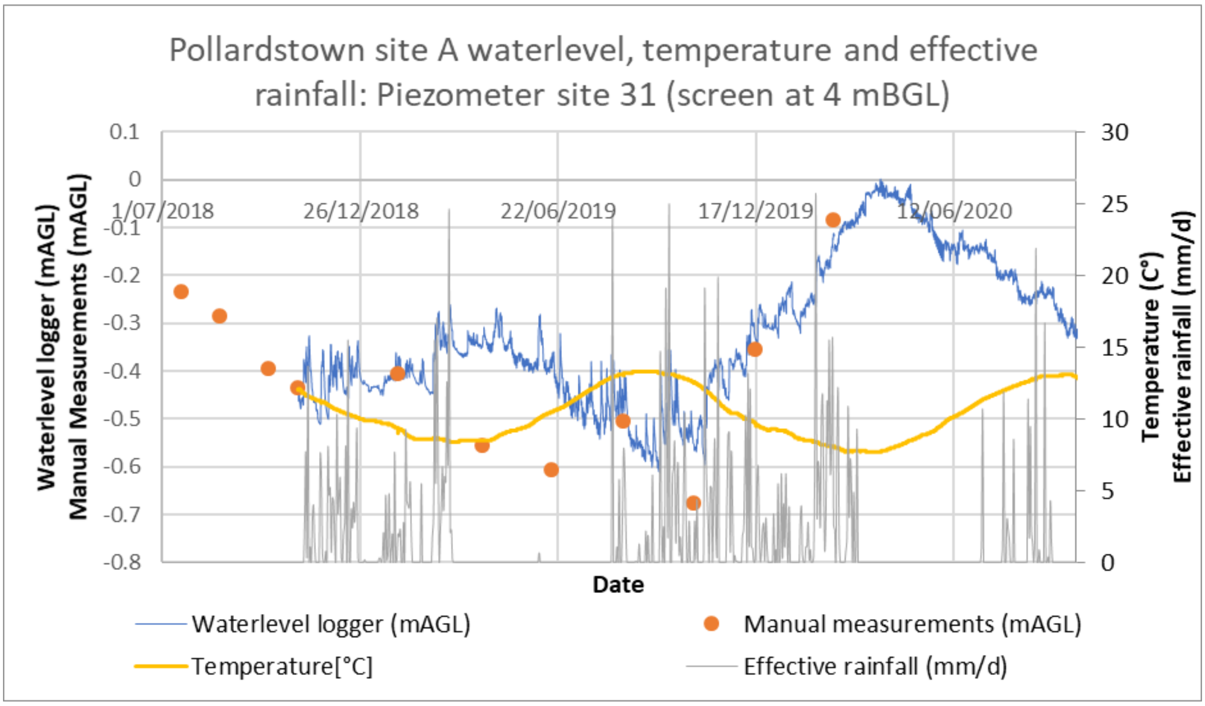
Appendix F. Water level logger time series



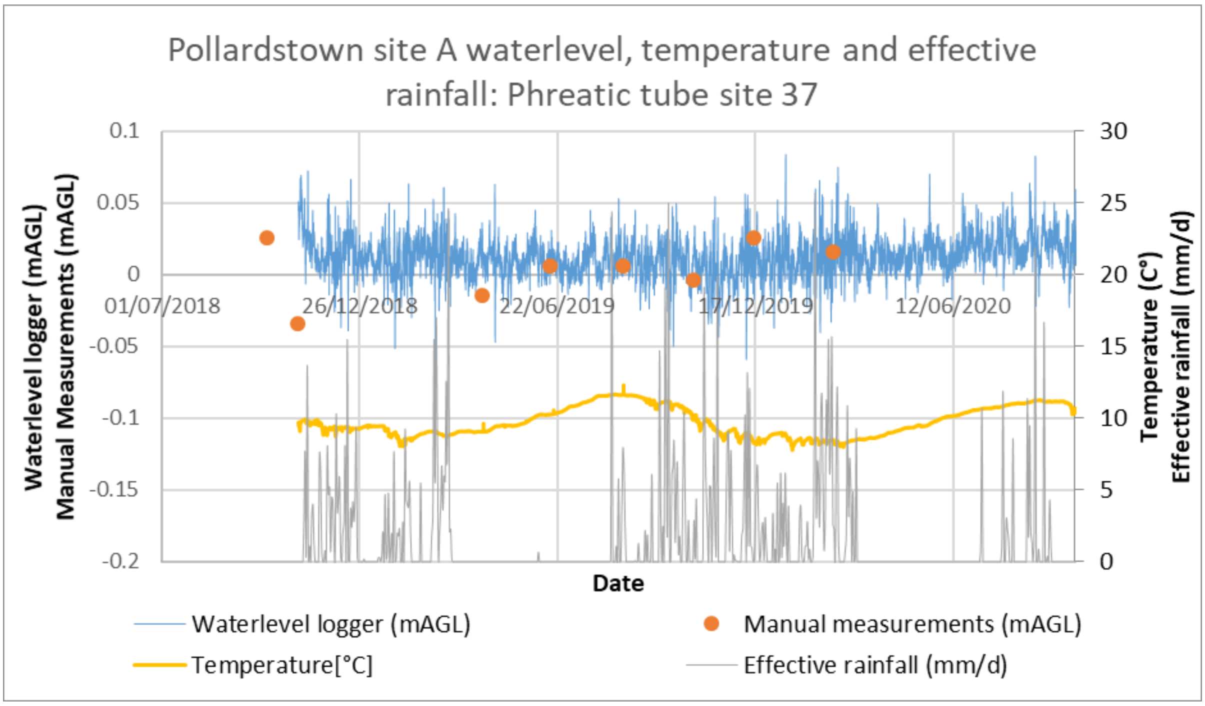


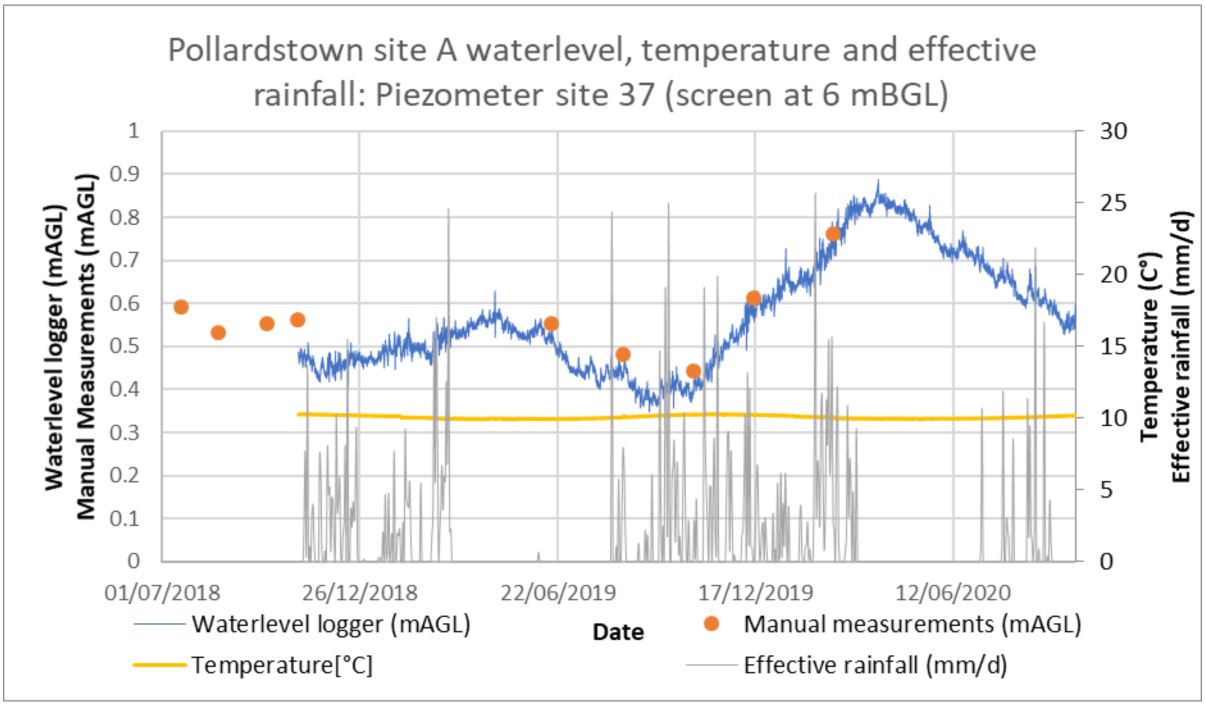
Appendix F. Water level logger time series

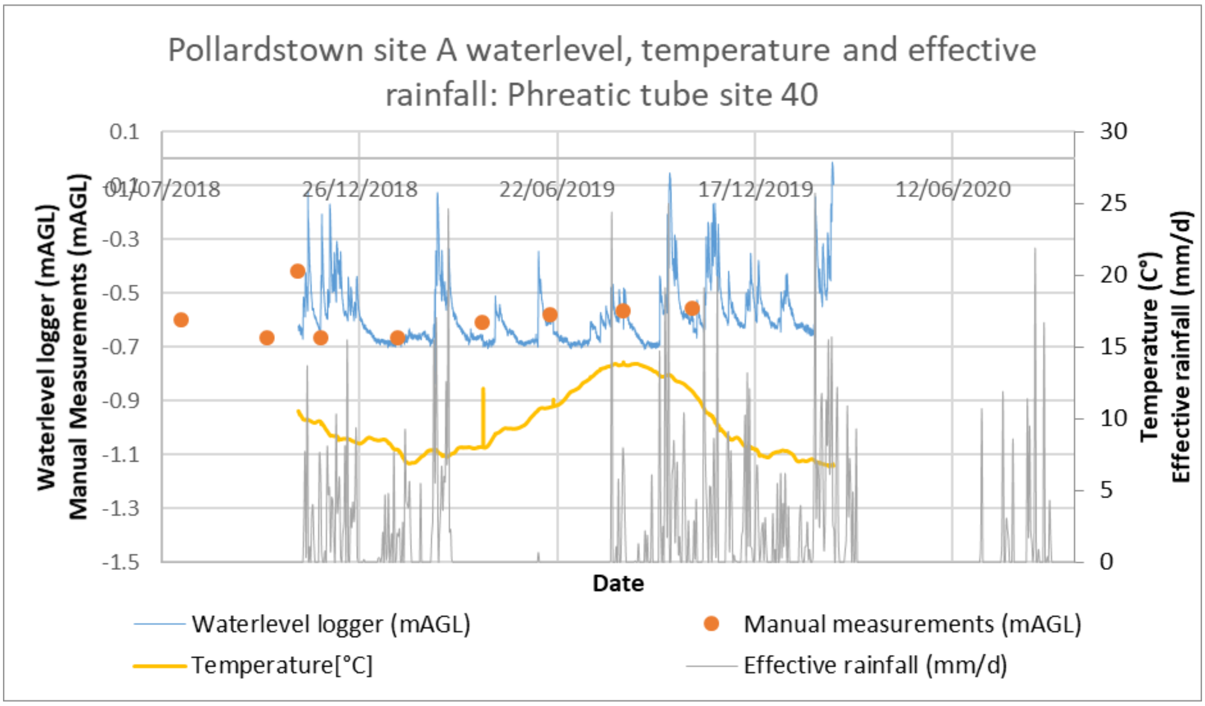
Pollardstown Fen site A water level time-series

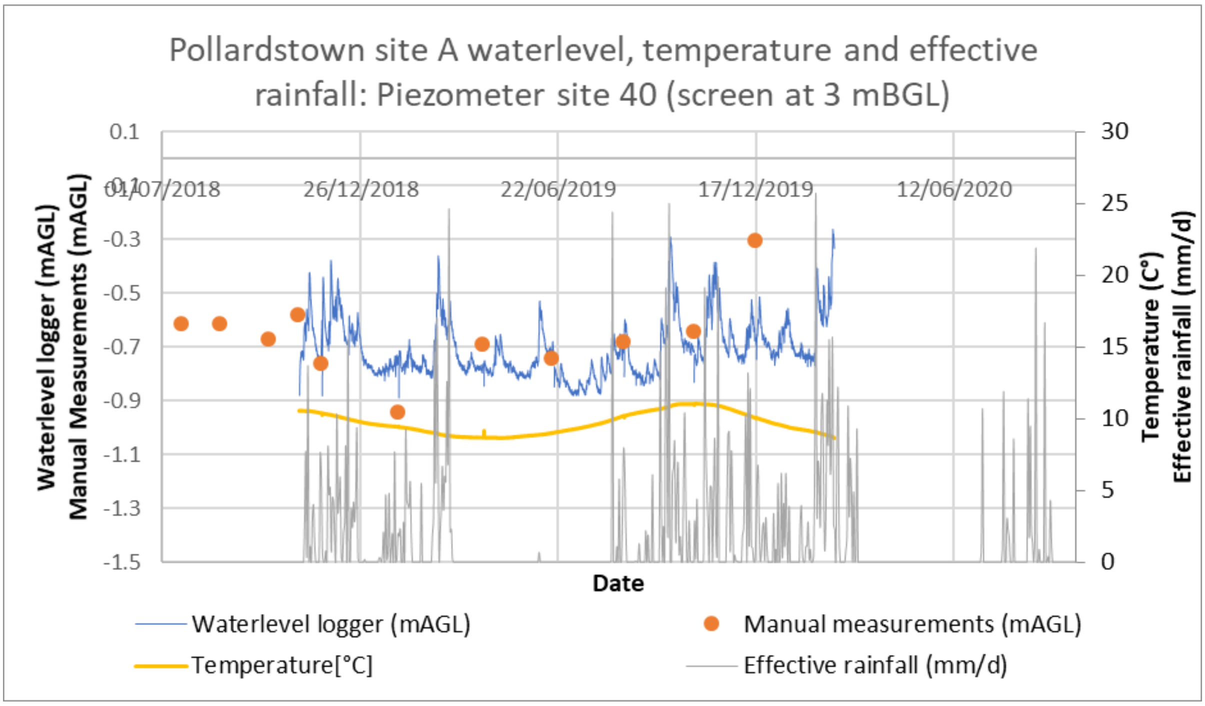


Appendix F. Water level logger time series

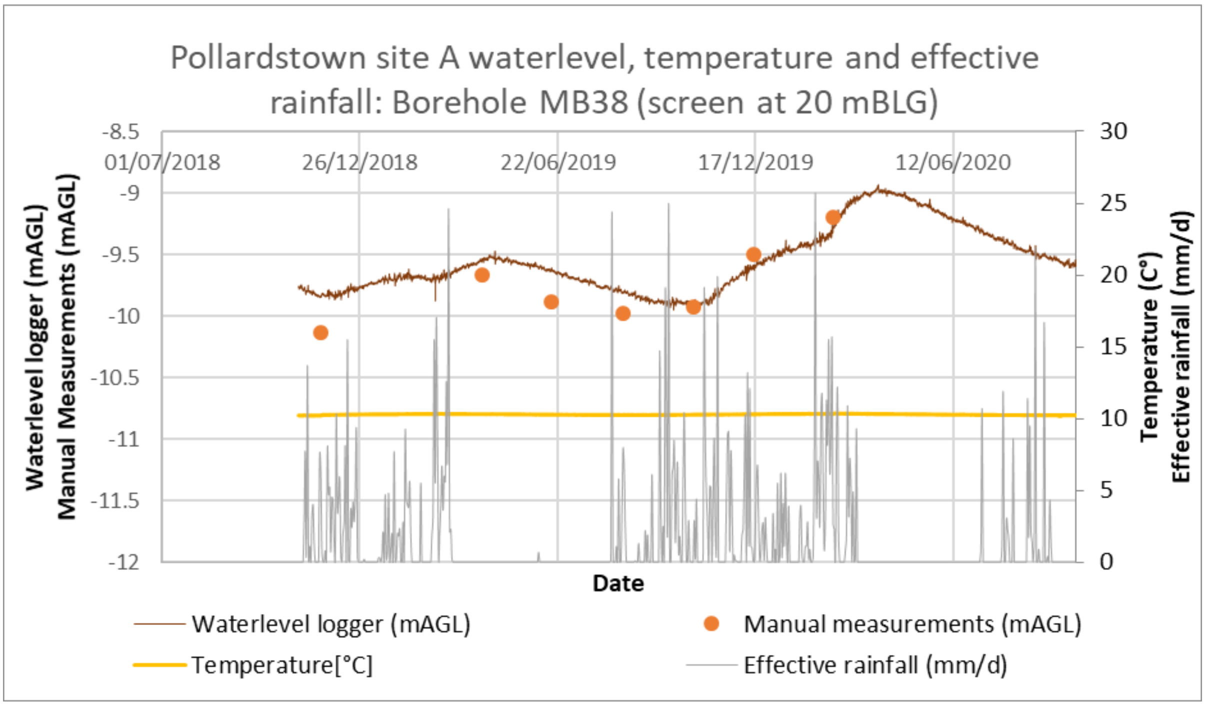

Pollardstown site A waterlevel, temperature and effective rainfall: Phreatic tube site 37
Waterlevel logger (mAGL)
Manual Measurements (mAGL)
Temperature (C°)
Effective rainfall (mm/d)
Date
0.1
0.05
0
-0.05
-0.1
-0.15
-0.2
30
25
20
15
10
5
0
01/07/2018
26/12/2018
22/06/2019
17/12/2019
12/06/2020
Waterlevel logger (mAGL)
Manual measurements (mAGL)
Temperature[°C]
Effective rainfall (mm/d)


Pollardstown site A waterlevel, temperature and effective rainfall: Piezometer site 37 (screen at 6 mBGL)
Waterlevel logger (mAGL)
Manual Measurements (mAGL)
Temperature (C°)
Effective rainfall (mm/d)
1
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
30
25
20
15
10
5
0
01/07/2018
26/12/2018
22/06/2019
17/12/2019
12/06/2020
Date
Waterlevel logger (mAGL)
Manual measurements (mAGL)
Temperature[°C]
Effective rainfall (mm/d)



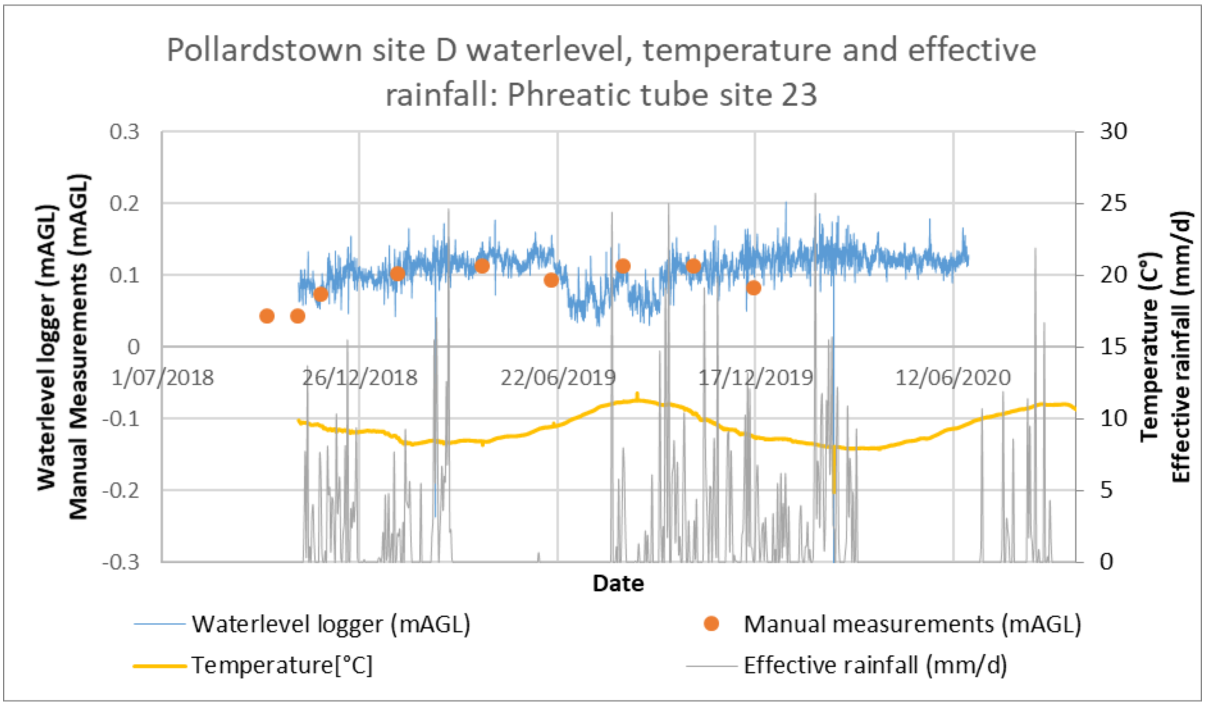


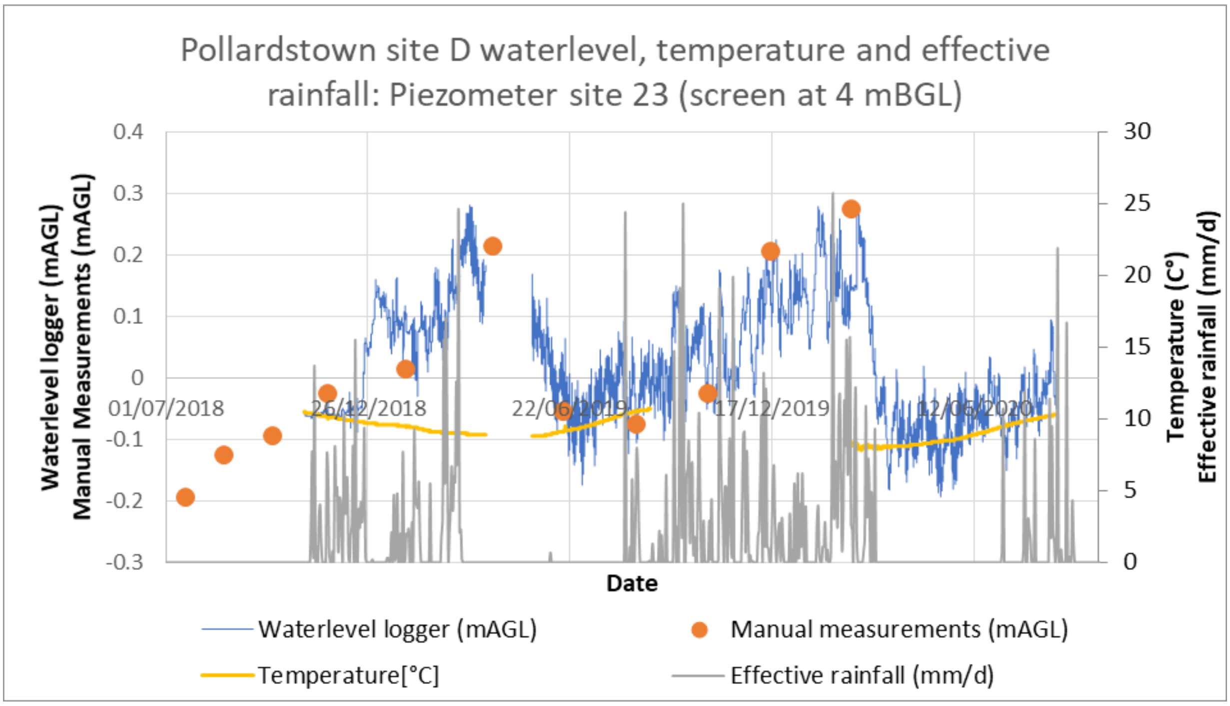
Appendix F. Water level logger time series

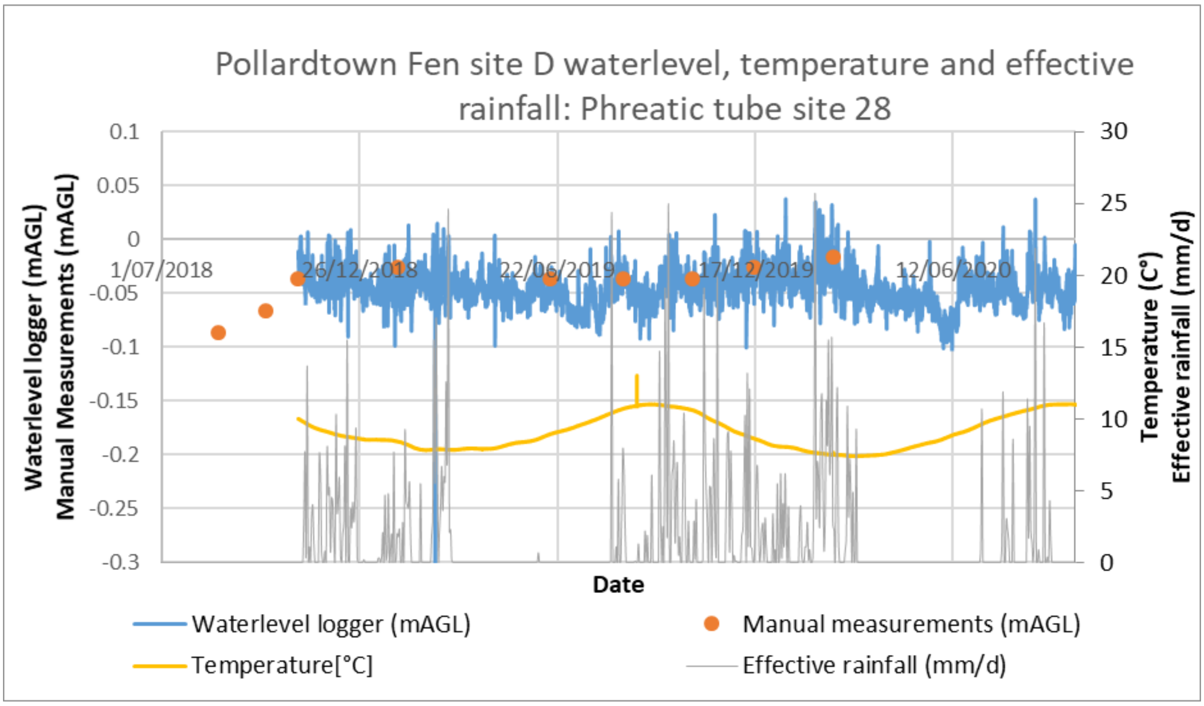


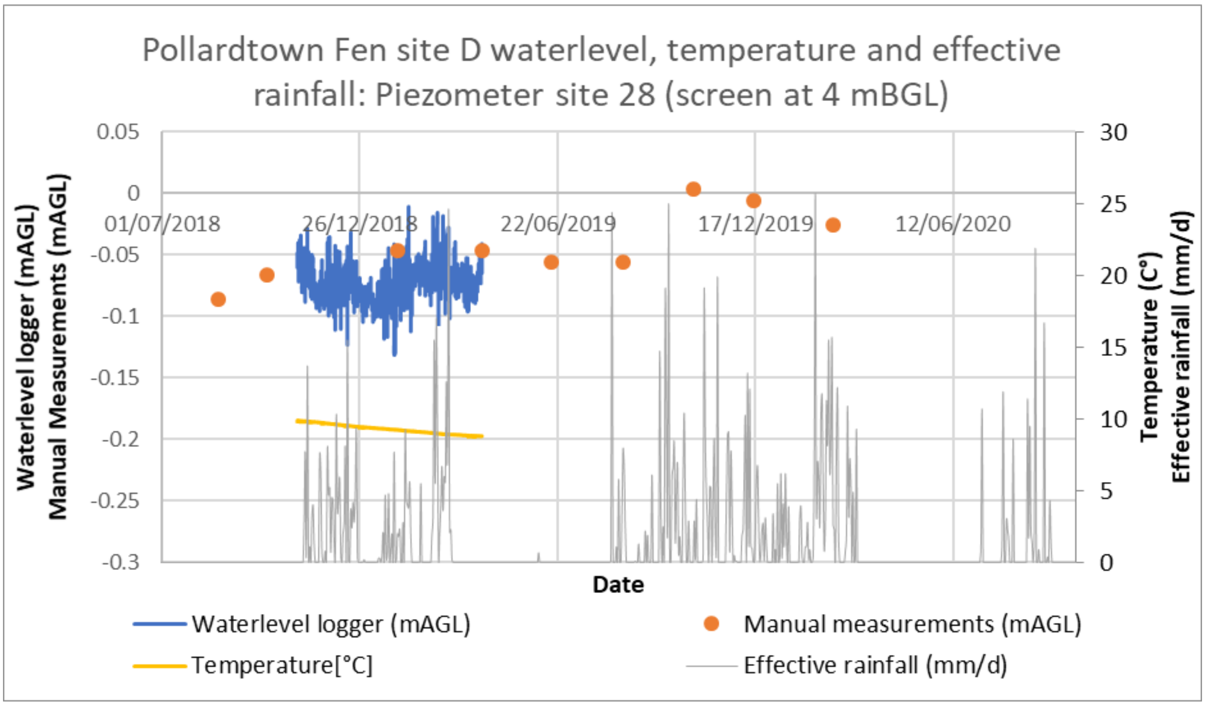
Appendix F. Water level logger time series

Pollardstown site D water level time-series

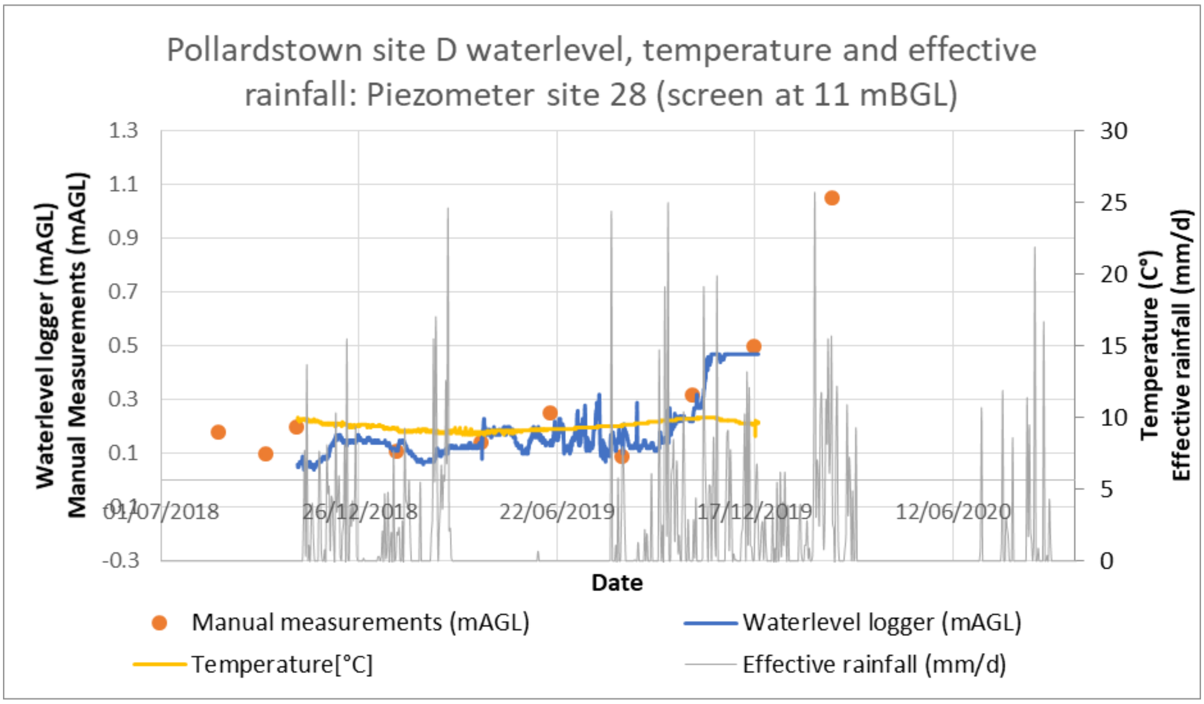


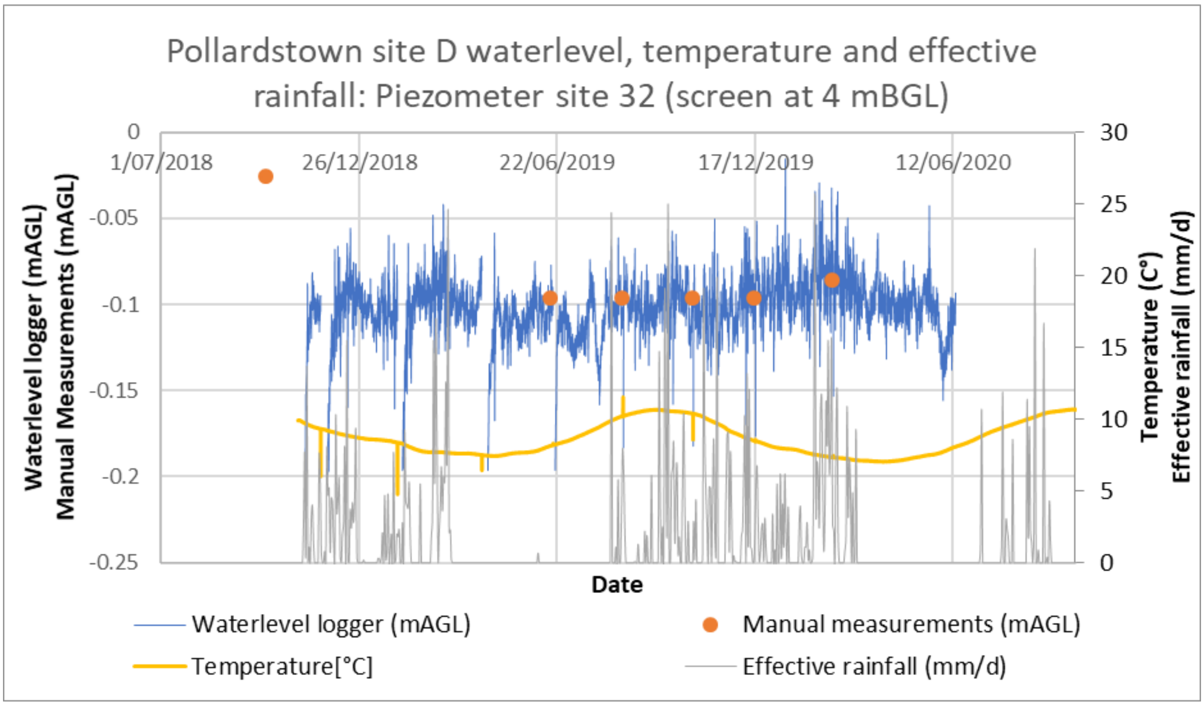




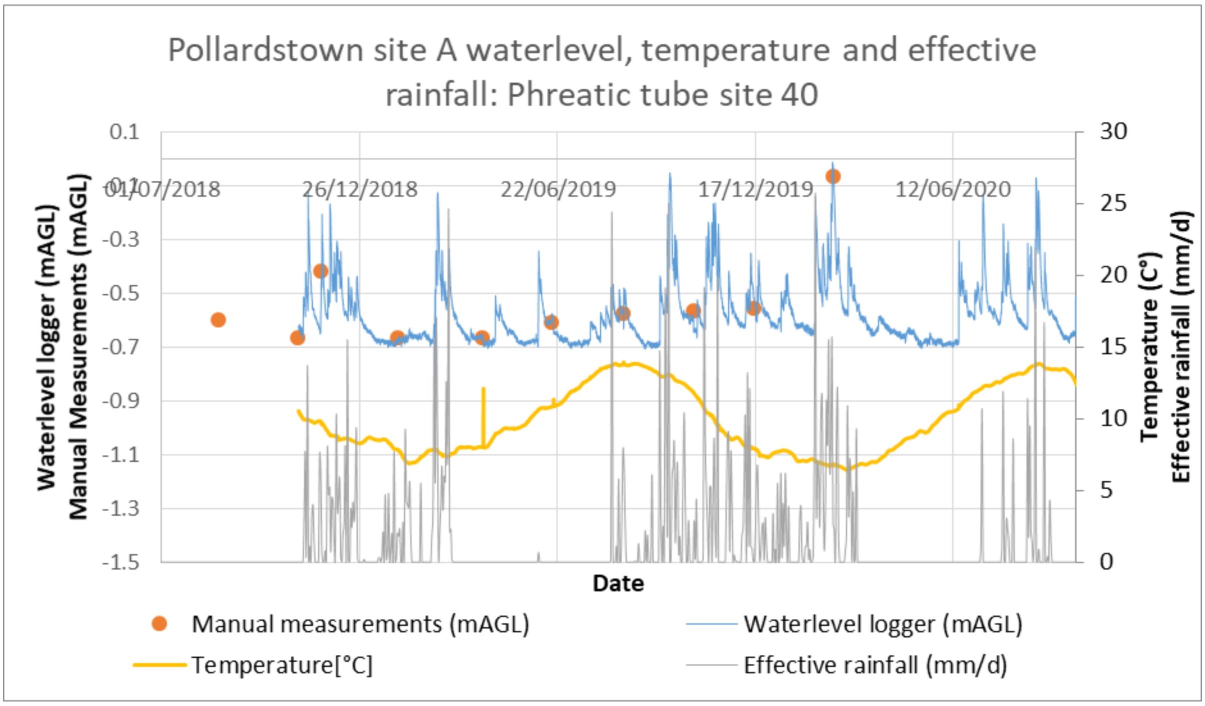


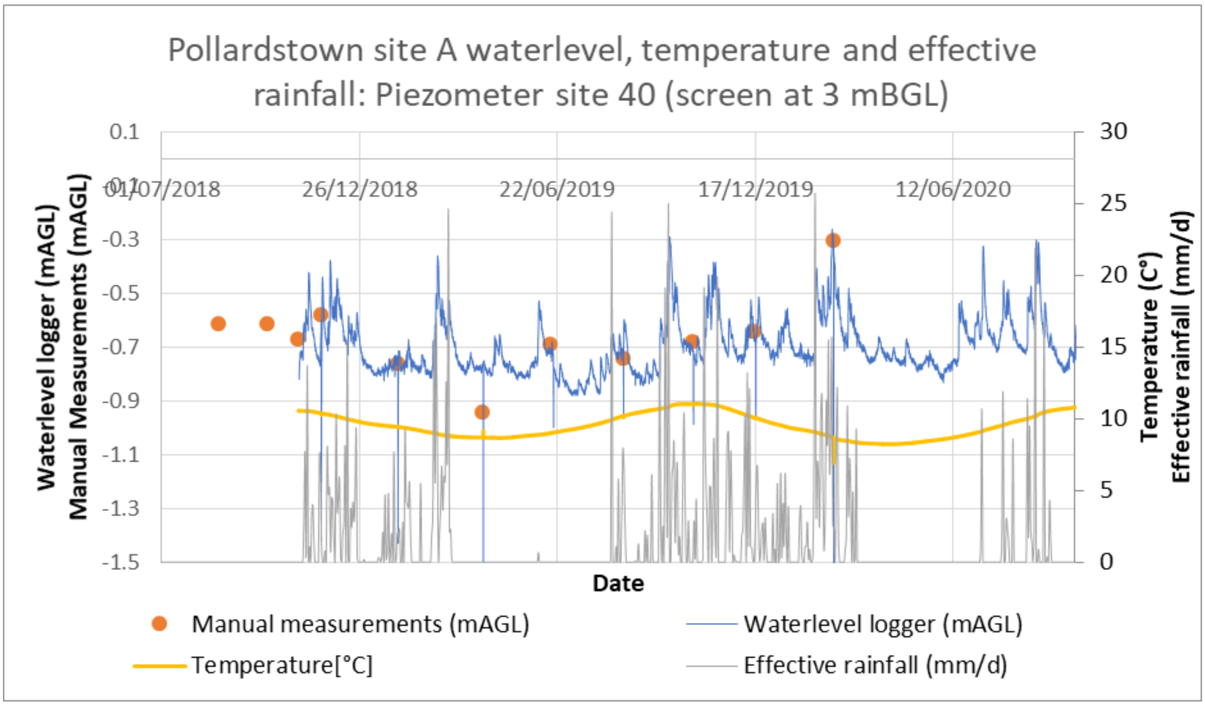
Appendix F. Water level logger time series





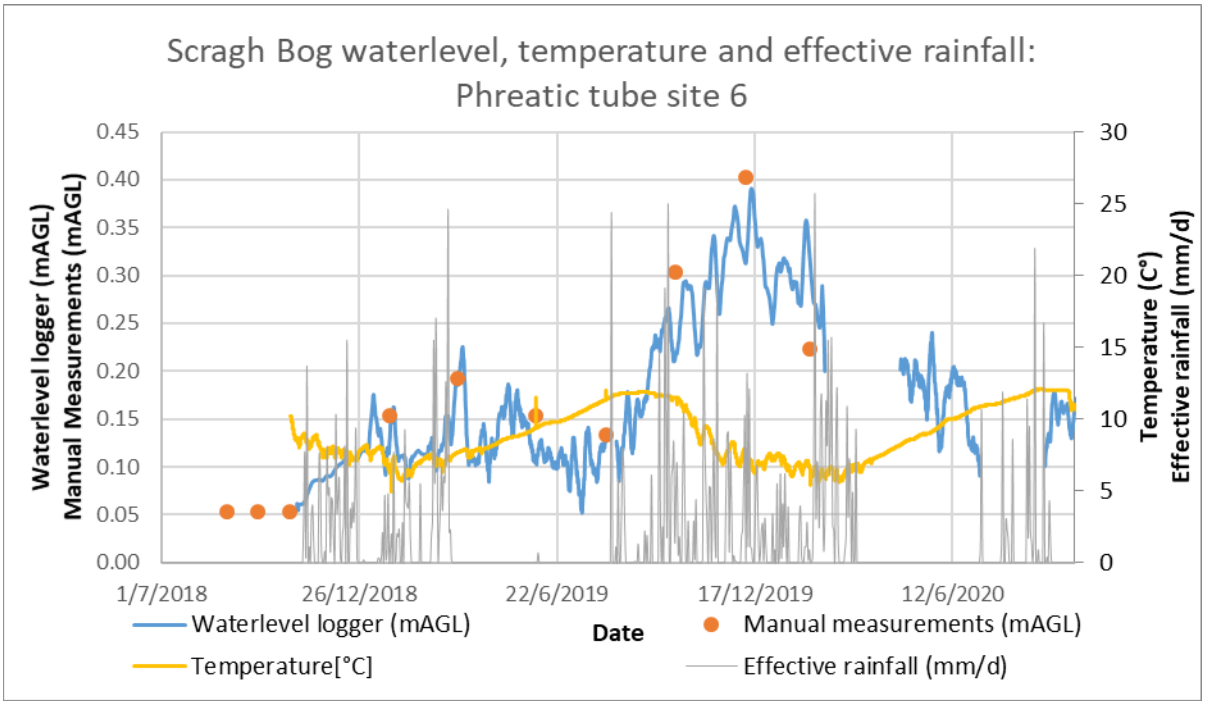
Appendix F. Water level logger time series

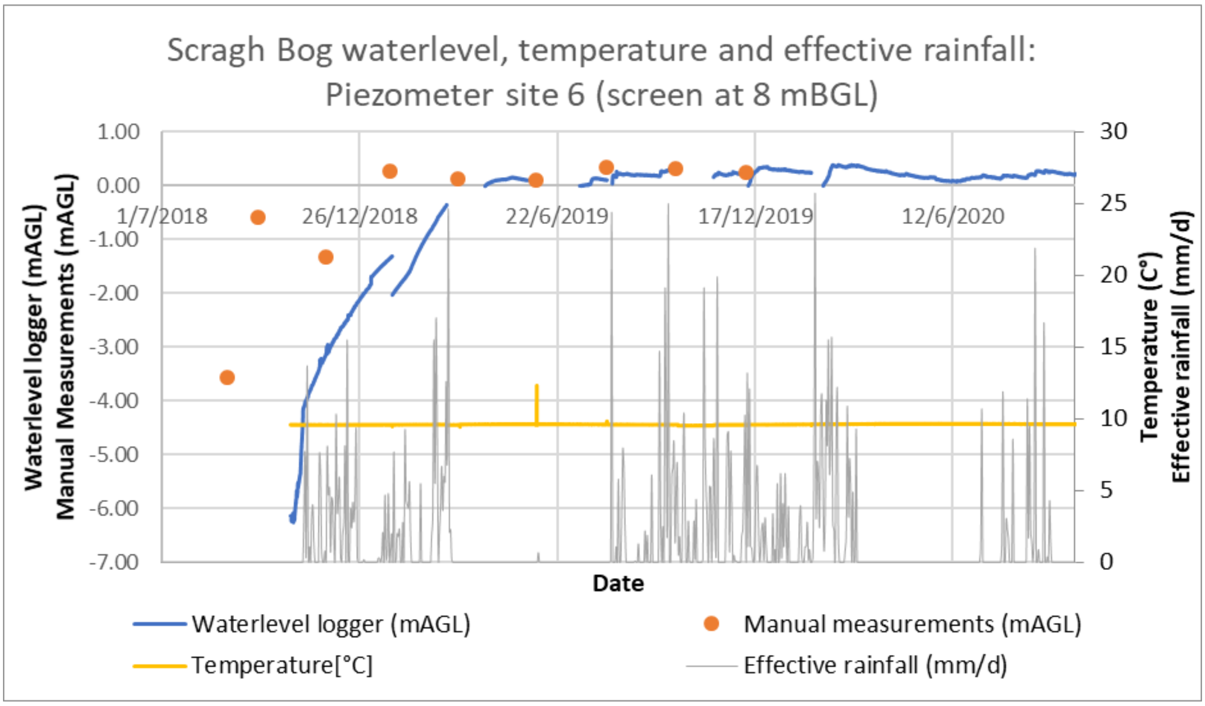


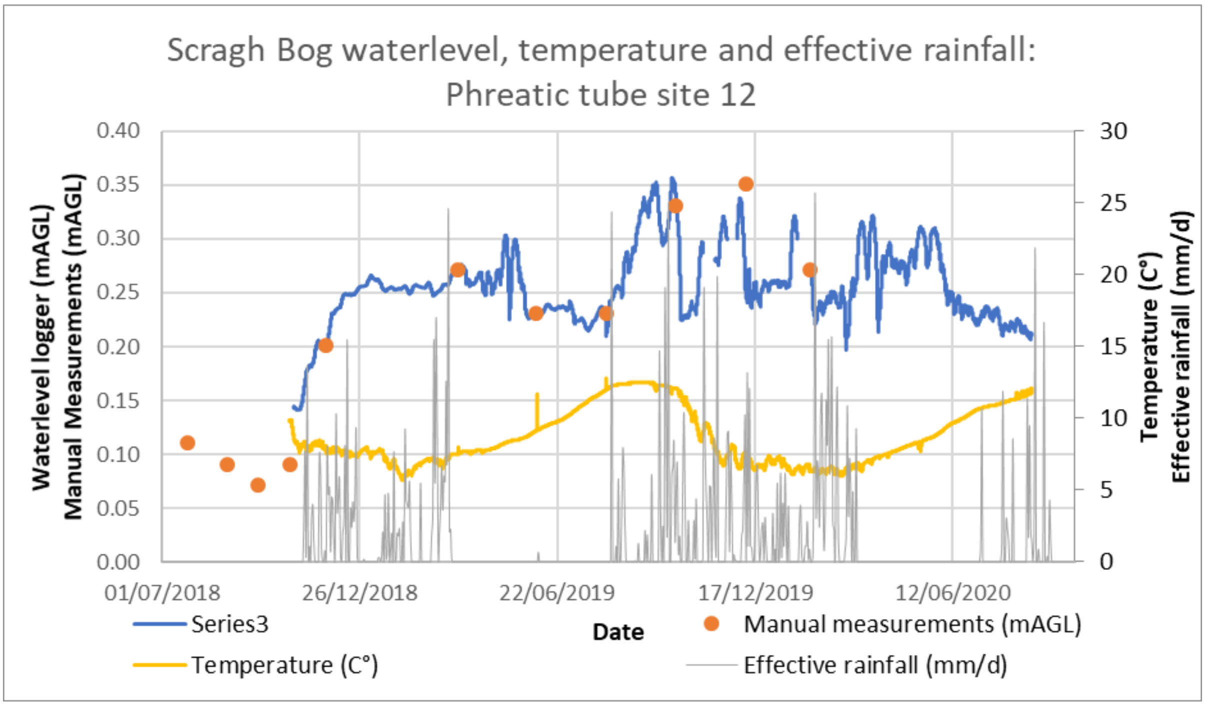


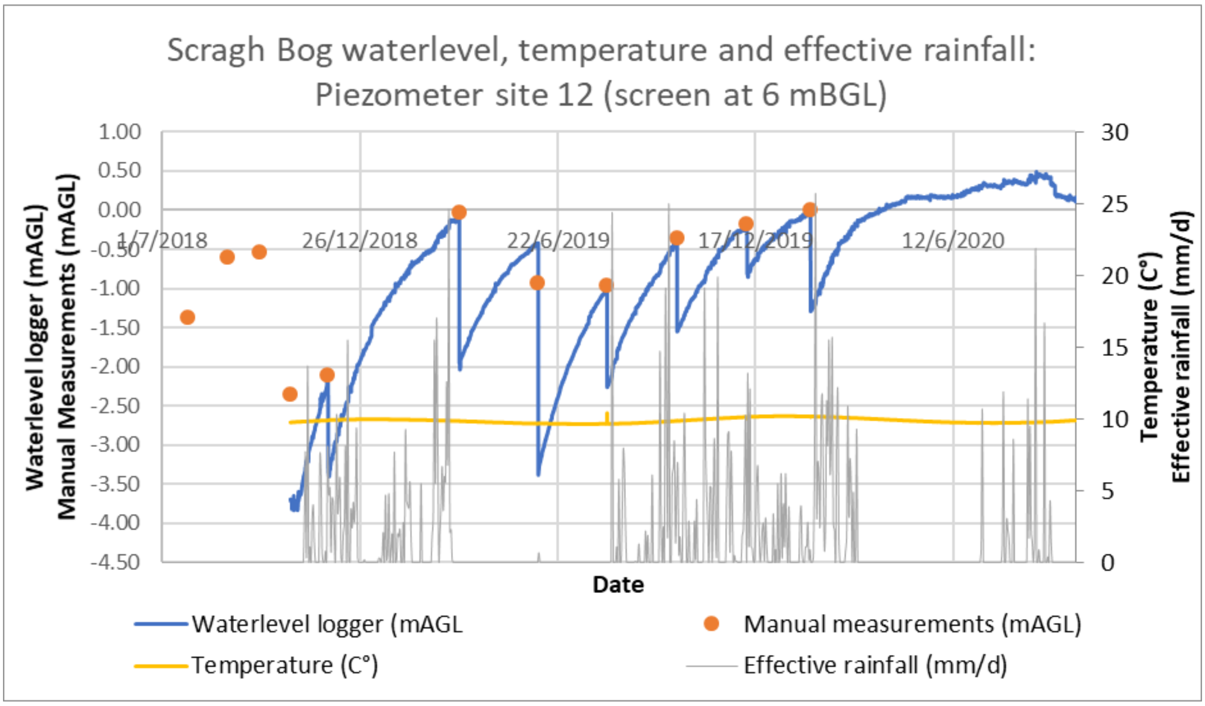
Appendix F. Water level logger time series

Scragh Bog water level time-series

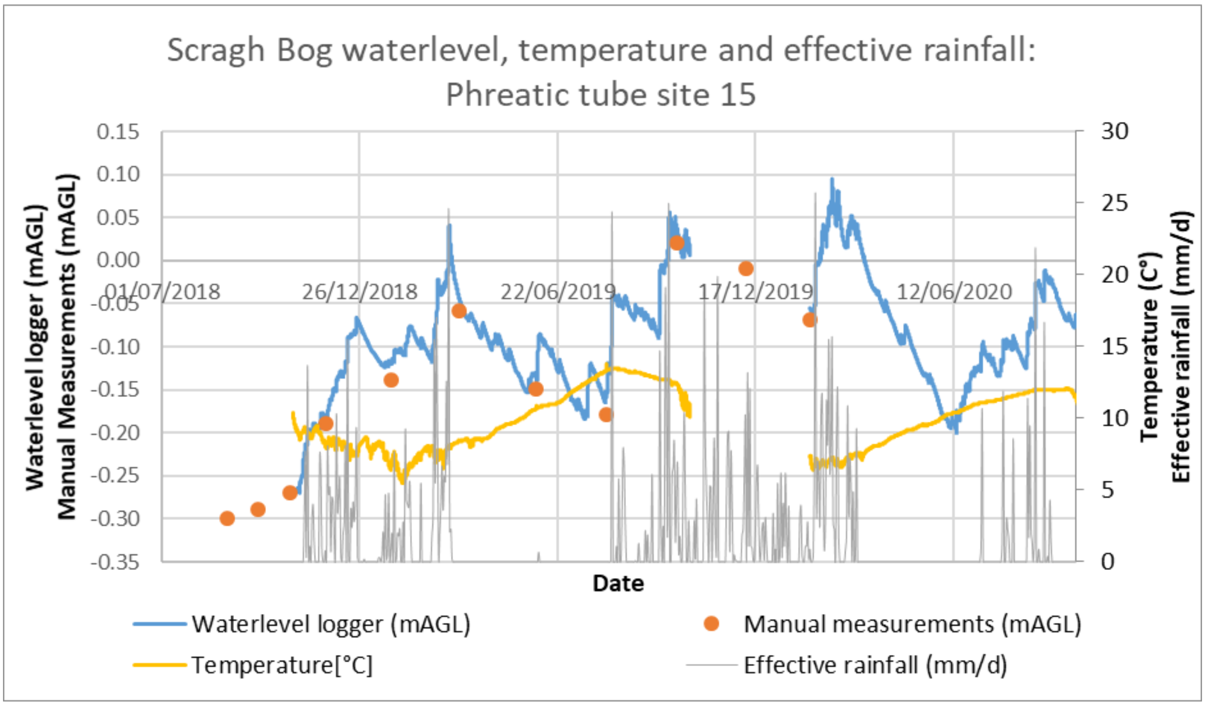


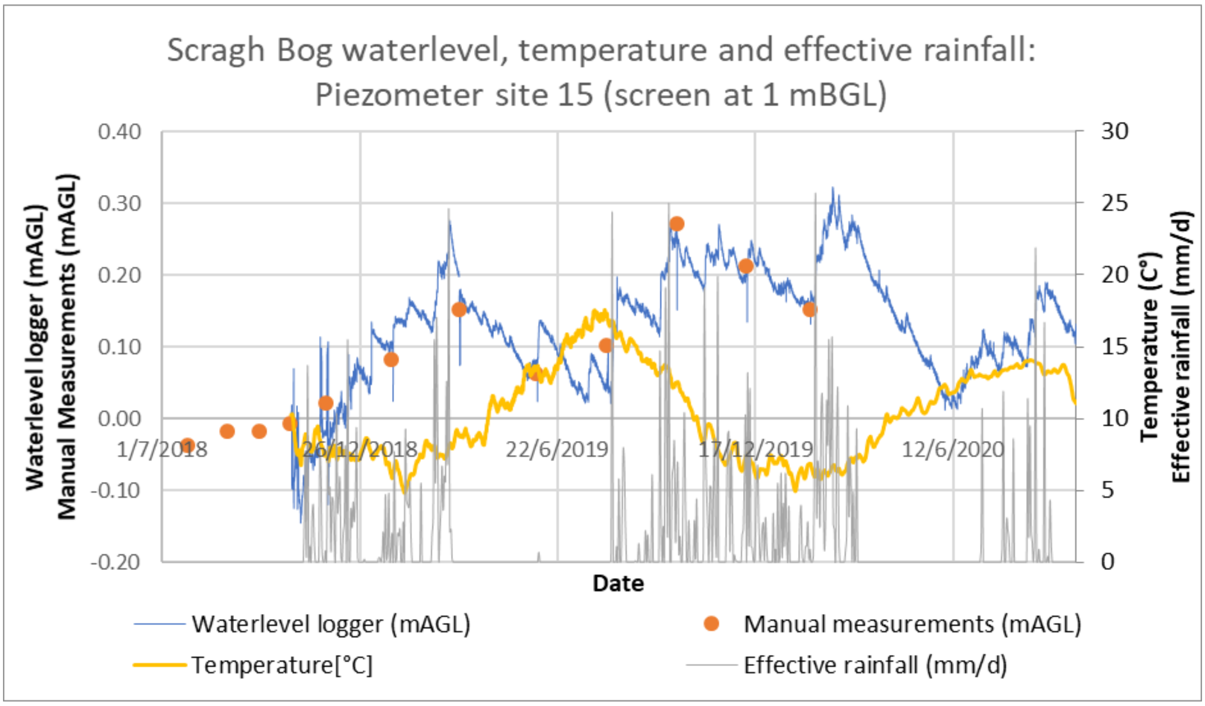


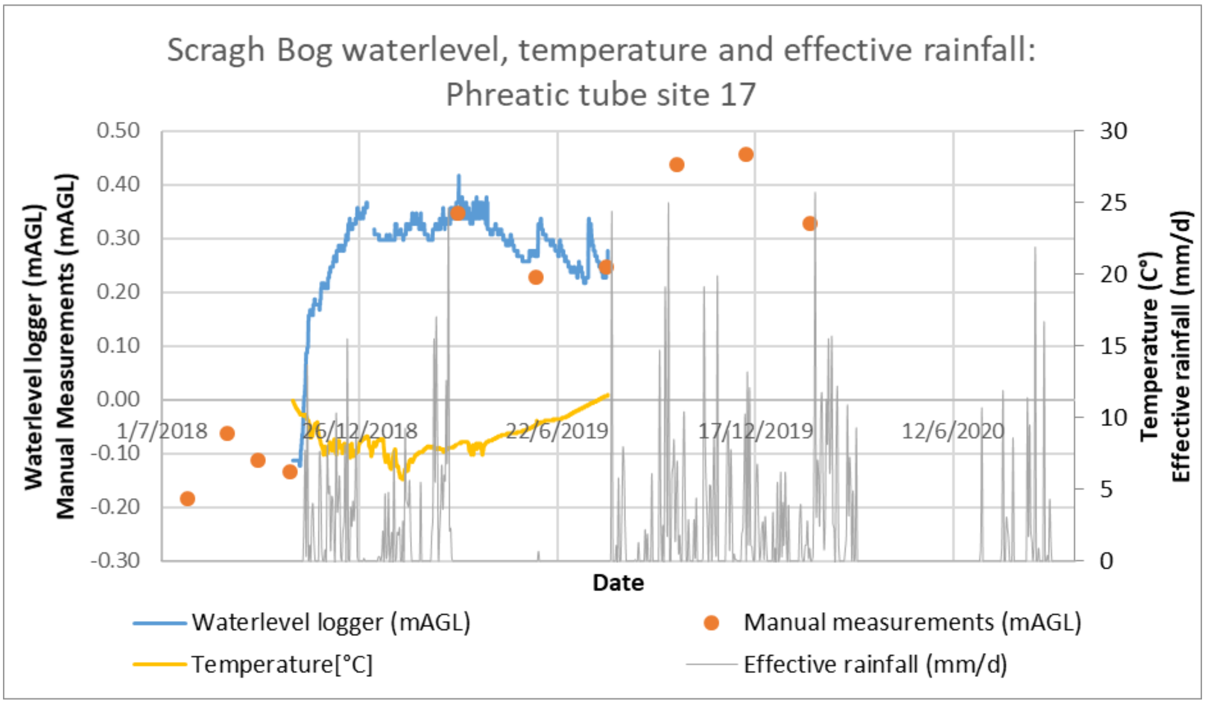


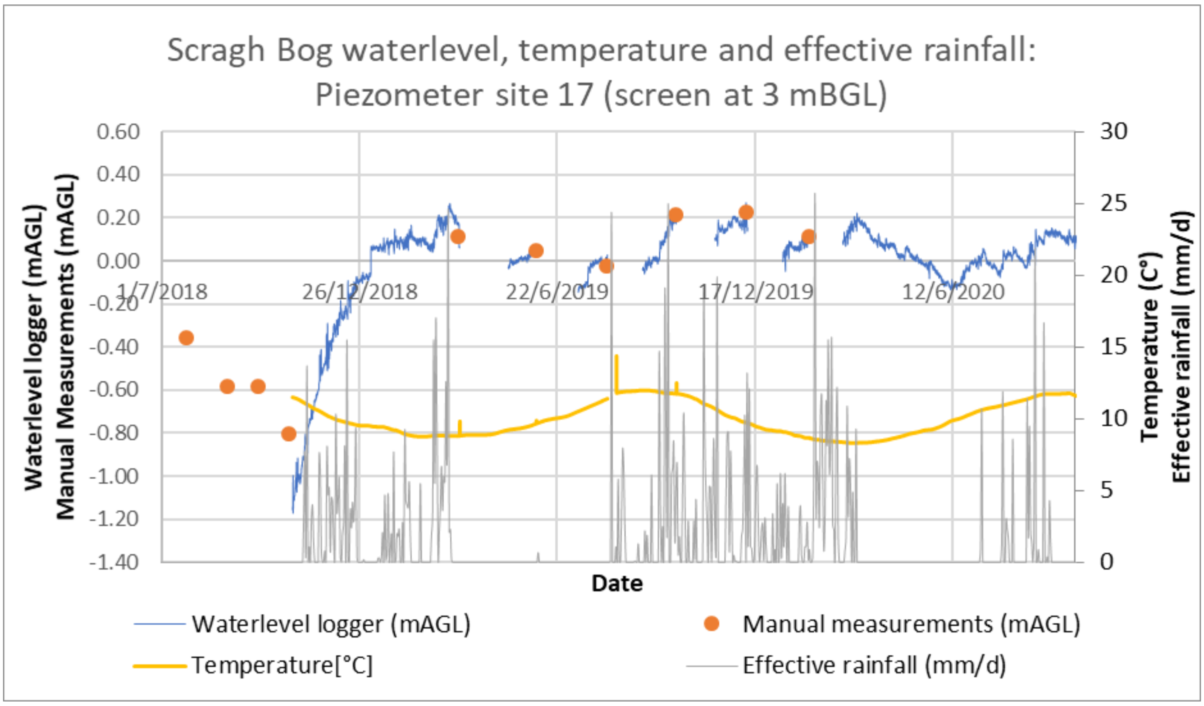


Appendix F. Water level logger time series

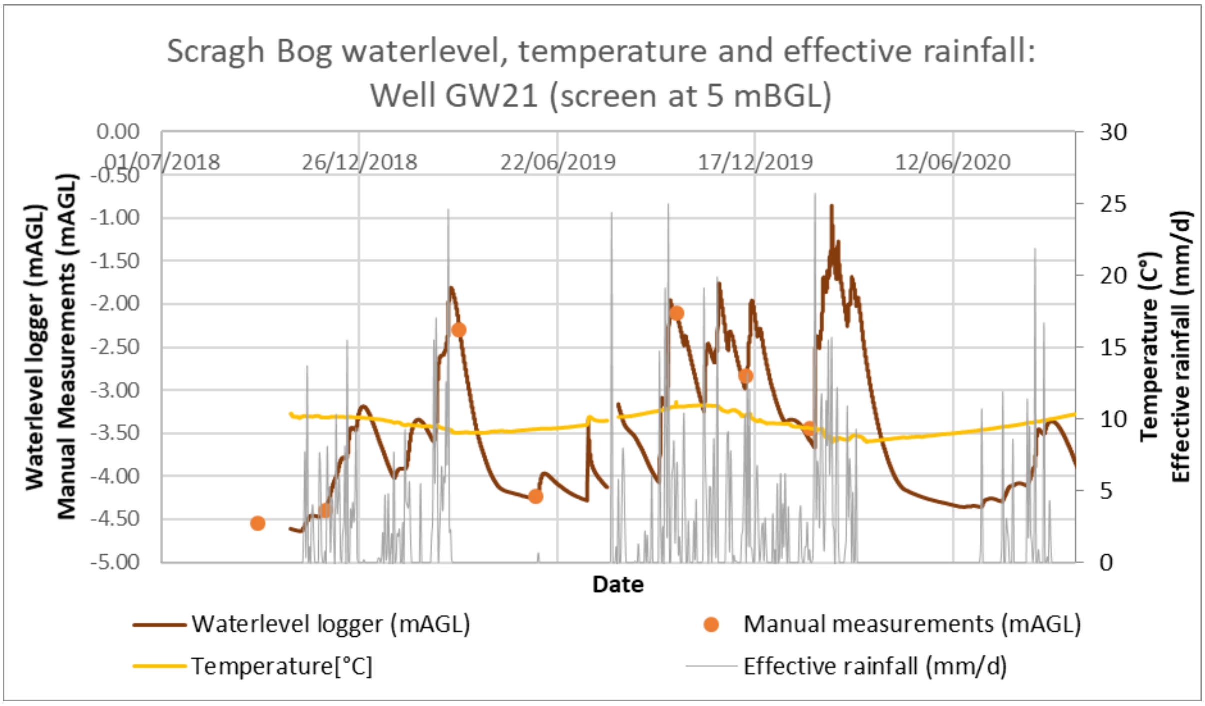

Scragh Bog waterlevel, temperature and effective rainfall:
Phreatic tube site 15
Waterlevel logger (mAGL)
Manual Measurements (mAGL)
Temperature (C°)
Effective rainfall (mm/d)
0.15
0.10
0.05
0.00
-0.05
-0.10
-0.15
-0.20
-0.25
-0.30
-0.35
30
25
20
15
10
5
0
01/07/2018
26/12/2018
22/06/2019
17/12/2019
12/06/2020
Date
Waterlevel logger (mAGL)
Manual measurements (mAGL)
Temperature[°C]
Effective rainfall (mm/d)

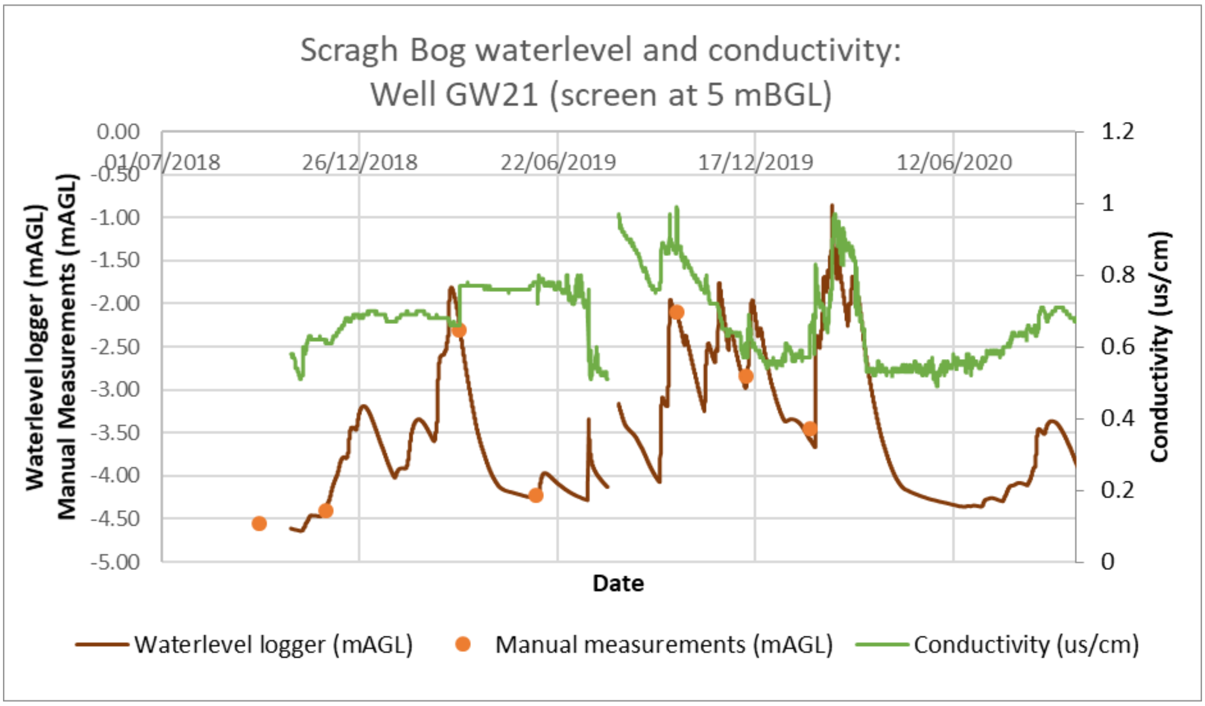

Scragh Bog waterlevel, temperature and effective rainfall:
Piezometer site 15 (screen at 1 mBGL)
Waterlevel logger (mAGL)
Manual Measurements (mAGL)
Temperature (C°)
Effective rainfall (mm/d)
0.40
0.30
0.20
0.10
0.00
-0.10
-0.20
30
25
20
15
10
5
0
1/7/2018
26/12/2018
22/6/2019
17/12/2019
12/6/2020
Date
Waterlevel logger (mAGL)
Manual measurements (mAGL)
Temperature[°C]
Effective rainfall (mm/d)



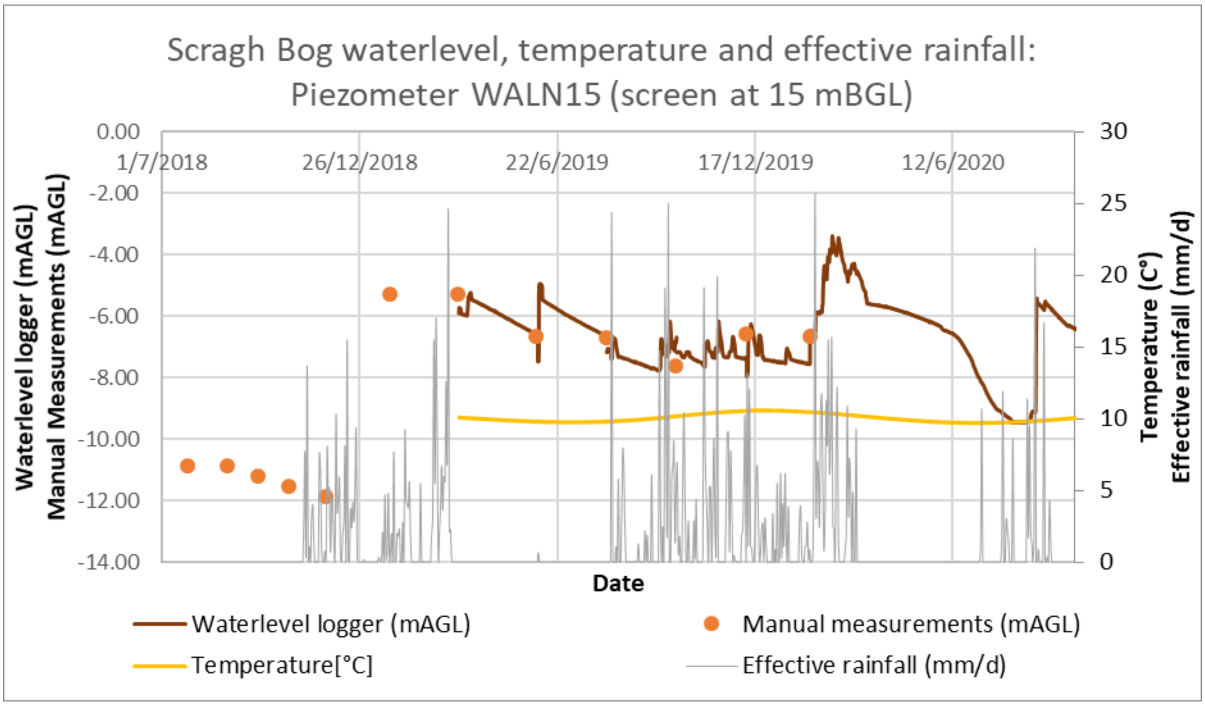


Appendix F. Water level logger time series



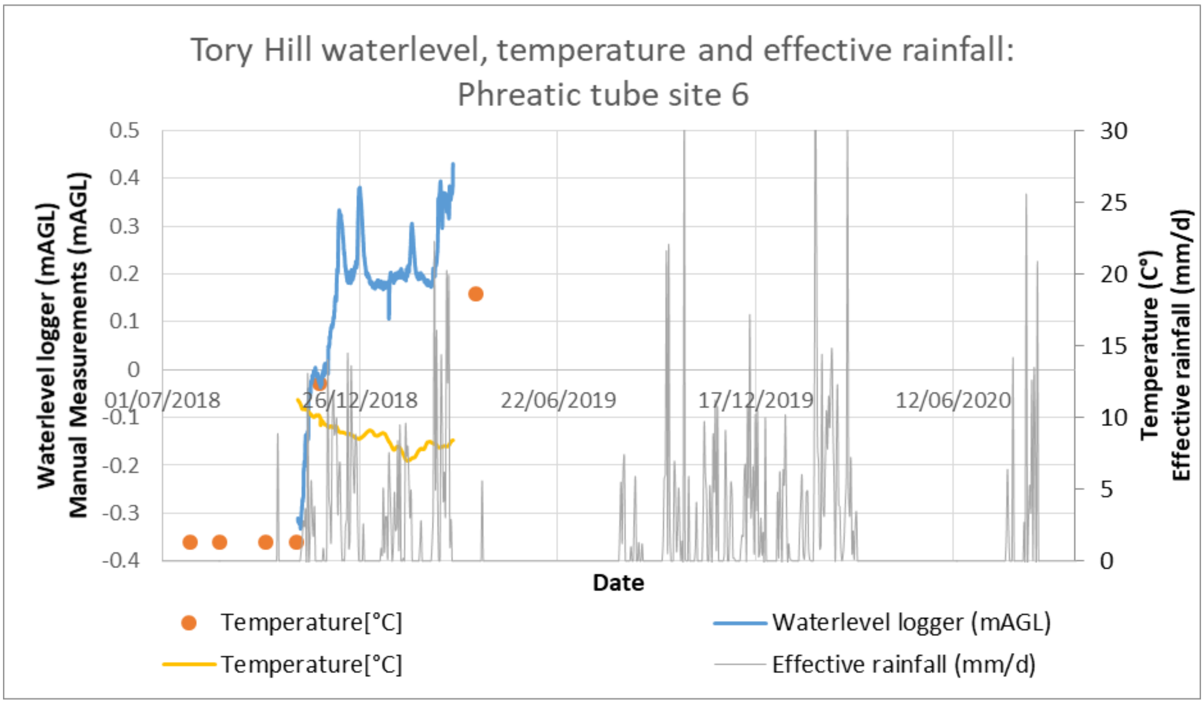


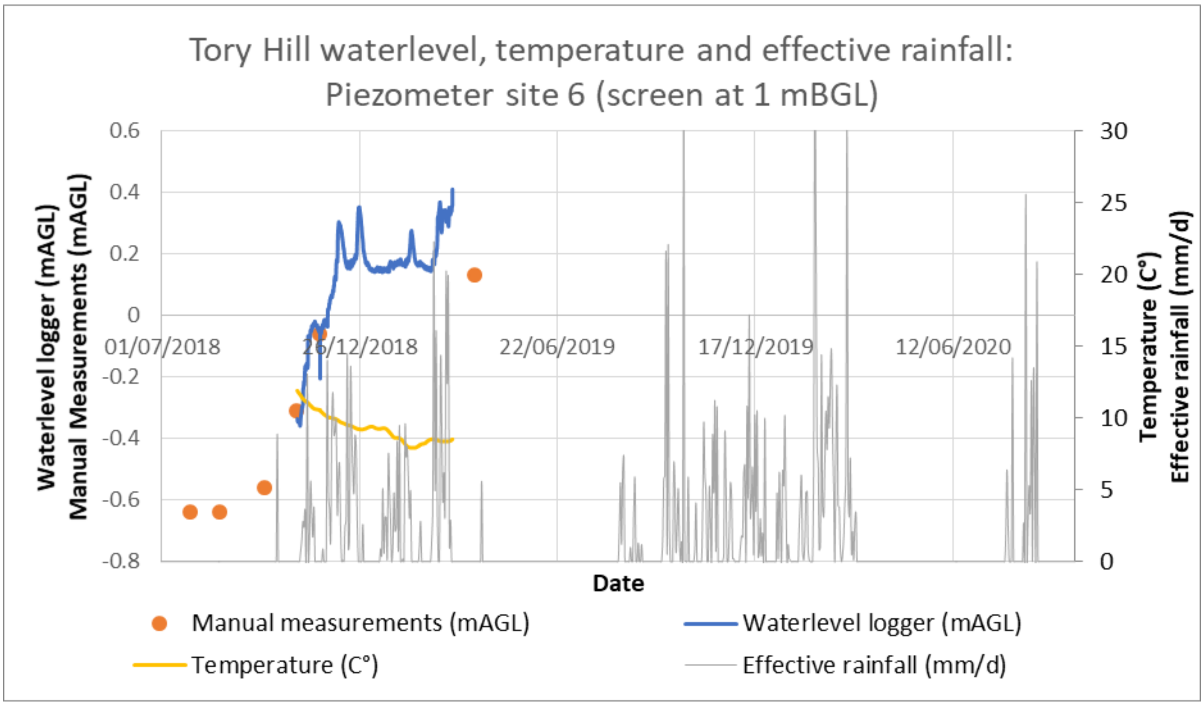
Appendix F. Water level logger time series

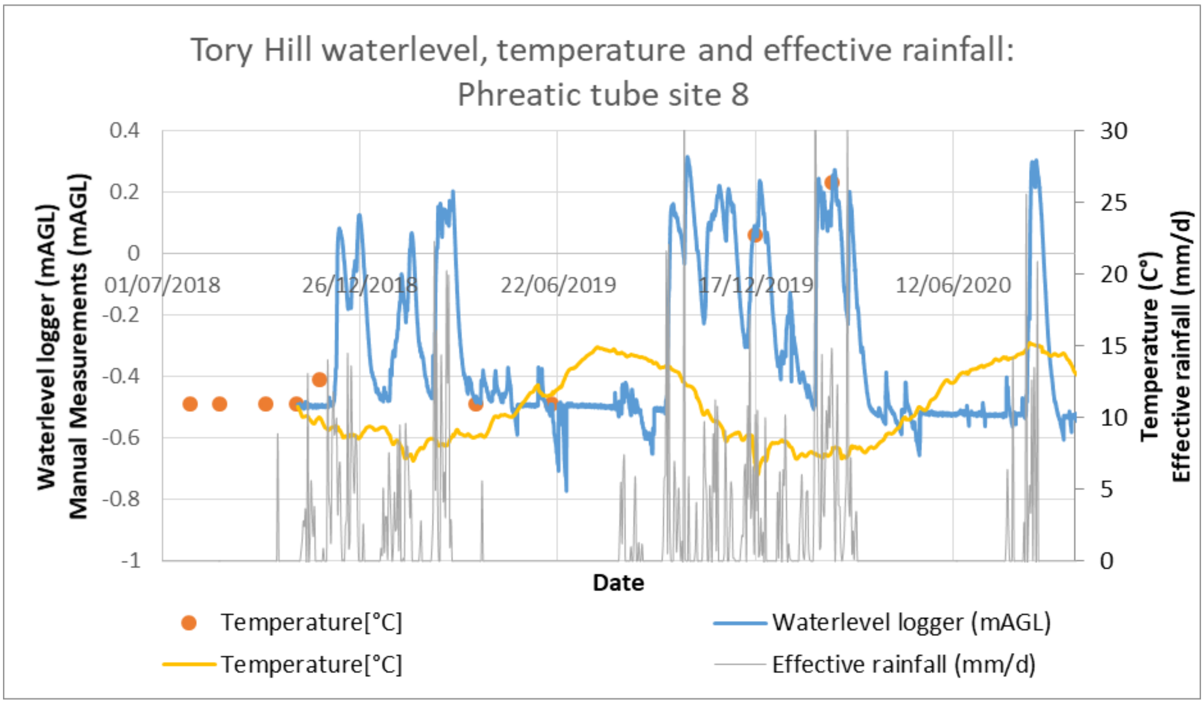


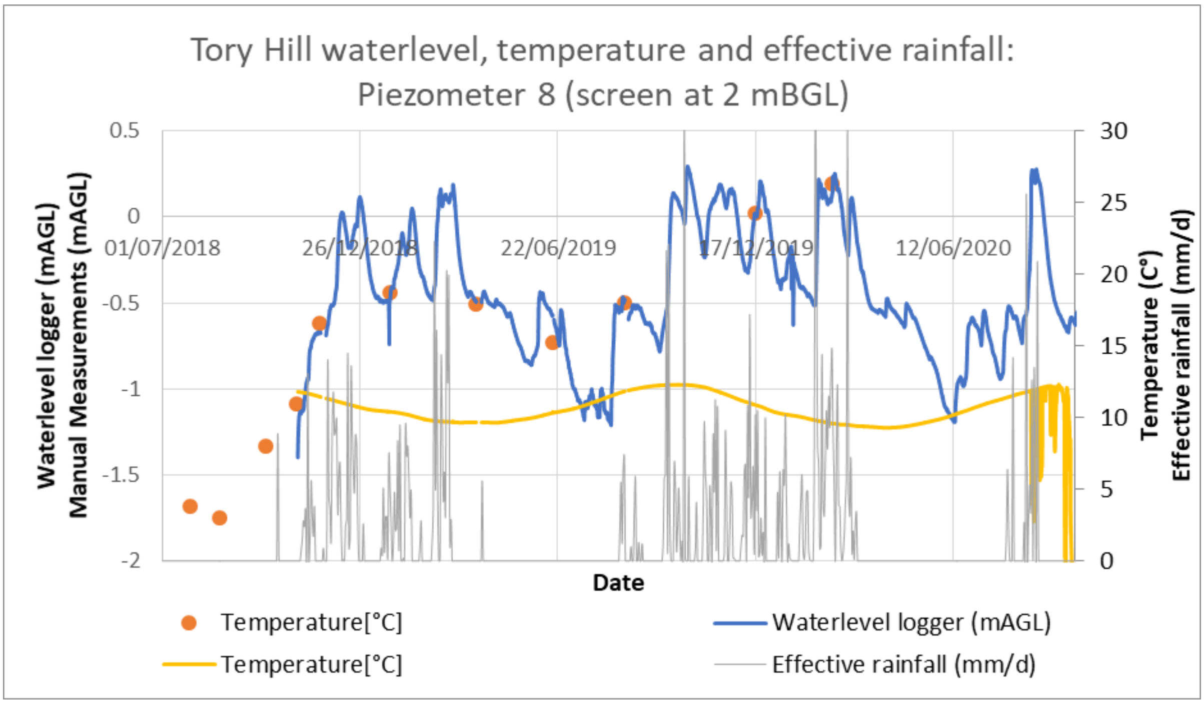
Appendix F. Water level logger time series

Tory Hill water level time-series

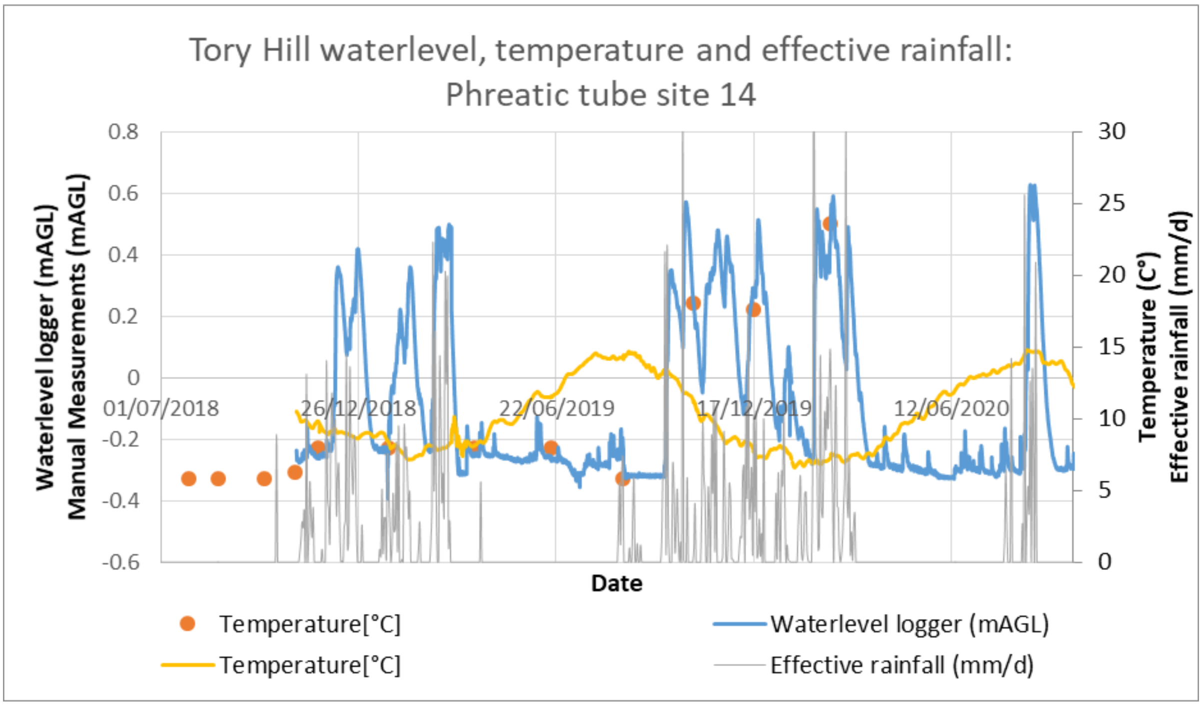




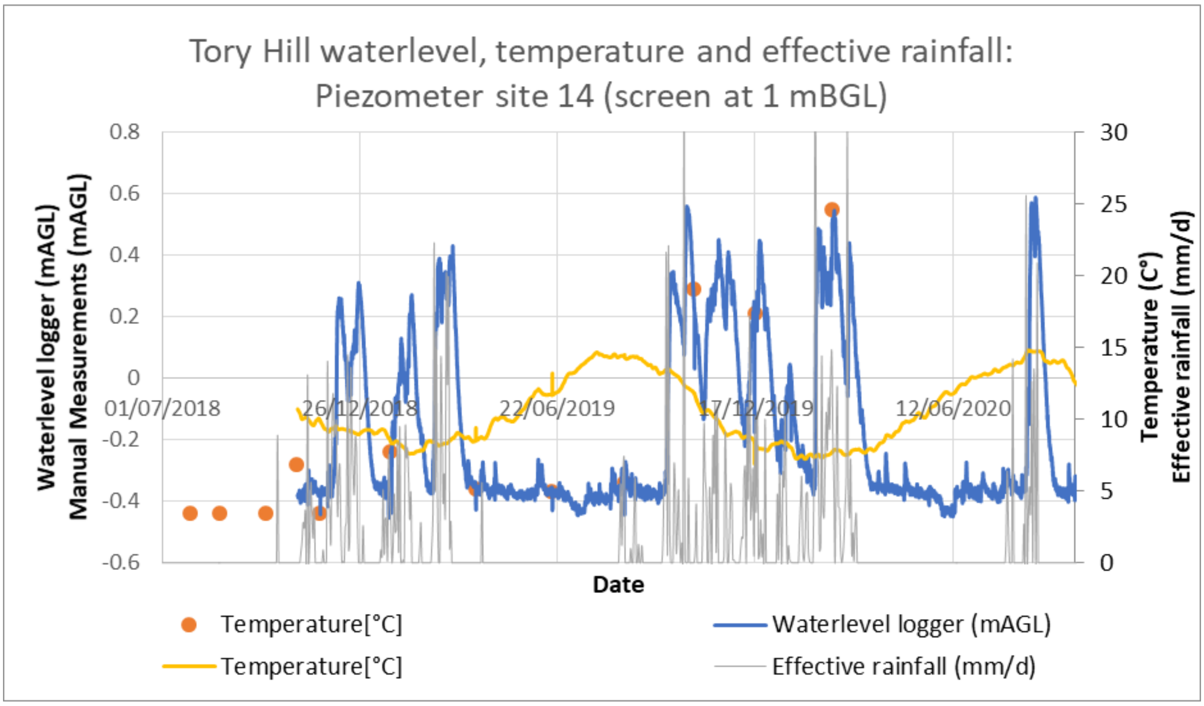




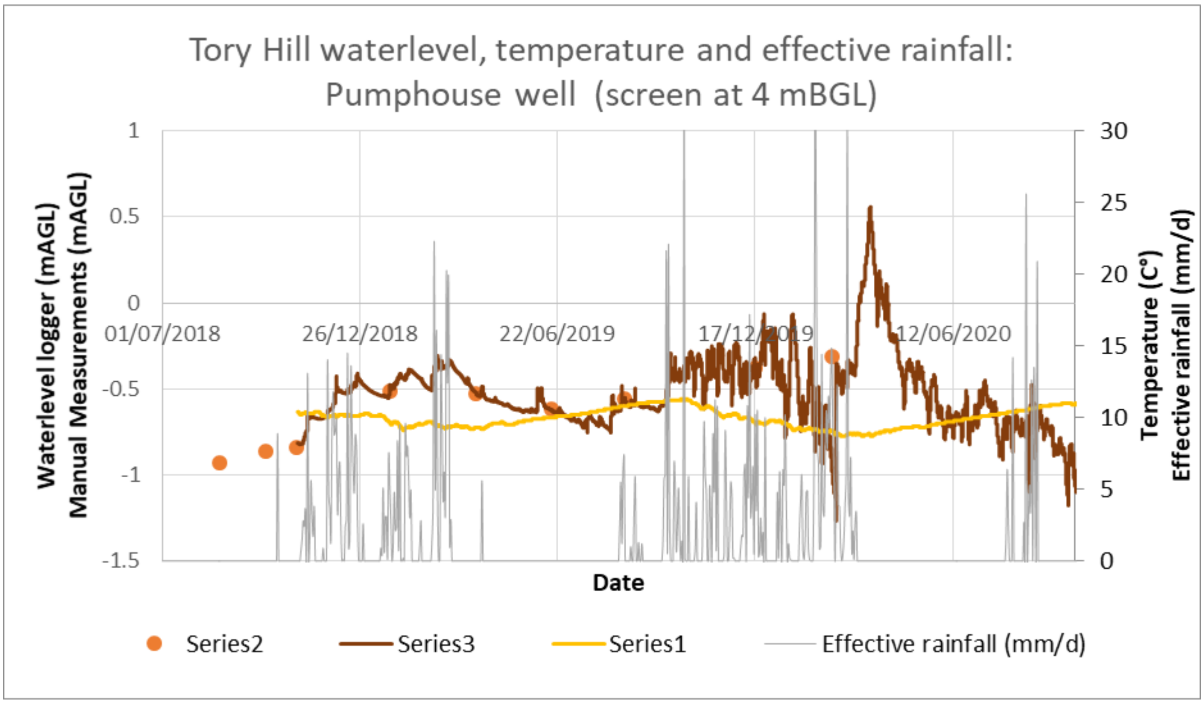
Appendix F. Water level logger time series



Tory Hill waterlevel, temperature and effective rainfall:
Phreatic tube site 14
Waterlevel logger (mAGL)
Manual Measurements (mAGL)
Temperature (C°)
Effective rainfall (mm/d)
0.8
0.6
0.4
0.2
0
-0.2
-0.4
-0.6
30
25
20
15
10
5
0
01/07/2018
26/12/2018
22/06/2019
17/12/2019
12/06/2020
Date
Temperature[°C]
Temperature[°C]
Waterlevel logger (mAGL)
Effective rainfall (mm/d)



Tory Hill waterlevel, temperature and effective rainfall:
Piezometer site 14 (screen at 1 mBGL)
Waterlevel logger (mAGL)
Manual Measurements (mAGL)
Temperature (C°)
Effective rainfall (mm/d)
0.8
0.6
0.4
0.2
0
-0.2
-0.4
-0.6
30
25
20
15
10
5
0
01/07/2018
26/12/2018
22/06/2019
17/12/2019
12/06/2020
Date
Temperature[°C]
Temperature[°C]
Waterlevel logger (mAGL)
Effective rainfall (mm/d)



Appendix F. Water level logger time series