

# Strategies to Improve Water Quality from Managed Peatlands (SWAMP) Project

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5. Office of Communications and Corporate Services

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## What did this research aim to address?

The Strategies to Improve Water Quality from Managed Peatlands (SWAMP) project sought to tackle critical knowledge gaps about water quality degradation in Ireland's peatland regions, driven by decades of drainage for industrial and domestic peat extraction. It focused on (i) identifying hotspots of pollution; (ii) quantifying the impacts of drainage of peat soils on stream chemistry and aquatic biodiversity; (iii) assessing current in situ pollution measures as well as innovative methods to treat polluted effluents; and (iv) developing improved hydrological models to predict peatlands' responses to drainage and restoration. The study focused on the Irish midlands, a hotspot for peatland degradation, to provide actionable science for governmental organisations at the local and national levels as well as for policymakers and restoration practitioners.

## What did this research find?

The SWAMP project revealed how decades of peatland drainage for extraction and other land use have harmed Ireland's rivers and streams. Drained peatlands have leached harmful nutrients, acids and carbon-loaded water into streams. The research identified some protected peatlands that also had poor water chemistry, highlighting the widespread legacy effects of drainage. Aquatic biota in degraded bogs showed reduced biodiversity (e.g. fewer sensitive species and dominance of pollution-tolerant species), although downstream effects were linked to upstream water quality, not bog status alone, underscoring the need for site-specific ecological indices to assess peatland restoration projects.

While sedimentation ponds at the edge of a cutaway bog removed suspended solids, this approach failed to treat dissolved pollutants, with seasonal acidity and nutrient surges further violating ecological standards. Edge-of-field treatments with biochar showed inconclusive results, warranting further study.

Finally, the SWAMP project exposed challenges with hydrological modelling, revealing that no single hydrological model suited all peatland types/catchments due to their heterogeneous responses to peatland management (drainage/rewetting/restoration), demanding site-specific approaches.

## How can the research findings be used?

The SWAMP project's findings arrive at a critical time, as Ireland needs to address the specific challenges of achieving compliance with the EU Water Framework Directive and delivering on the Nature Restoration Law. Reversing centuries of unsustainable peatland management and the declining trend in water quality in streams in the midlands will require considerable effort from all stakeholders. It will require the coordination of multiple agencies to expand the monitoring of peatlands both in space and time and in particular to include water quality indicators at site and regional levels.

Tackling the pollution recorded in bog streams at specific sites will be largely addressed by (i) rewetting and rehabilitation projects within existing drained bogs and (ii) stricter legal requirements for the treatment of effluents associated with peat extraction activities. The SWAMP project's results provide baseline information for future long-term monitoring and set achievable targets. They also highlight the need for tailored, site-specific approaches rather than one-size-fits-all solutions. Success hinges on interdisciplinary collaboration to address compounding pressures, from legacy drainage impacts to climate change.

**EPA RESEARCH PROGRAMME 2014–2020**

# **Strategies to Improve Water Quality from Managed Peatlands (SWAMP) Project**

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## **EPA Research Report**

A copy of the end-of-project Technical Report is available on request from the EPA

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by

University College Dublin

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The EPA Research Programme addresses the need for research in Ireland to inform policymakers and other stakeholders on a range of questions in relation to environmental protection. These reports are intended as contributions to the necessary debate on the protection of the environment.

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# Executive Summary

Ireland's water quality continues to decline,<sup>1</sup> driven partially by peatland drainage for agriculture, forestry and industrial extraction. These disturbances alter peatland hydrology and biogeochemistry, exacerbating downstream pollution, flood risks and carbon/nutrient losses – pressures that are all intensified by climate change (e.g. extreme rainfall and droughts). The Strategies to Improve Water Quality from Managed Peatland (SWAMP) project investigated these impacts in the Irish midlands, focusing on water chemistry, aquatic biota and hydrological modelling in drained/extracted peatlands. Four key findings are described below.

## Water chemistry degradation

- Streams in industrial cutaway peatlands showed elevated levels of pollutants compared with natural bogs, as follows:
  - Nutrients: higher total dissolved nitrogen (TDN), sulfate and total ammonia/nitrate hotspots.
  - Organic matter: increased dissolved organic carbon (DOC) and biological oxygen demand.
  - Conductivity: higher electrical conductivity in drained sites.
- Even protected special areas of conservation bogs leach significant nitrogen and carbon, highlighting widespread legacy effects of drainage.

## Aquatic biota shifts

- Degraded bog streams had:
  - reduced biodiversity, i.e. lower abundance of sensitive taxa (e.g. Ephemeroptera/Plecoptera/Trichoptera);
  - altered communities, i.e. dominance of pollution-tolerant taxa (e.g. *Asellus* vs *Gammarus* found in natural sites).
- Biota differences were linked to local water quality rather than bog status, underscoring the need for site-specific ecological indices to assess responses to restoration/rewetting efforts.

## Ineffective mitigation measures from industrial cutaway peatlands

- Sedimentation ponds in extracted peatlands failed to treat dissolved pollutants (e.g. DOC, TDN), while successfully reducing total suspended solids.
- Seasonal dynamics prevail in the effluents of the cutaway peatland, with neutral pH but high nutrients/ions in warm/dry periods and acidic pH with diluted pollutants in cold/wet periods.
- Annual exports were substantial (DOC: 148 kg C/ha/y; TDN: 15.7 kg N/ha/y), primarily during low-flow periods.
- Edge-of-field novel treatment using biochar showed inconclusive results.

## Hydrological modelling challenges

- No universal model exists for peatland hydrology due to data/resource limitations and heterogeneous responses to drainage/rewetting/restoration.
- A multi-model ensemble approach, using the Modular Assessment of Rainfall-Runoff Models Toolbox framework, revealed that model performance varied by catchment, with failures in wet regimes (critical under climate change). Overall, adaptive, site-specific configurations are essential.

The SWAMP project final recommendations are as follows:

- Immediate actions: target nitrogen pollution hotspots with effective treatment solutions and stricter effluent regulations.
- Long-term strategies: prioritise large-scale rewetting of drained peat soils, coupled with long-term monitoring to track recovery.
- Research gaps: develop standardised peatland models, improve biogeochemical process representation and refine ecological assessment tools.

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<sup>1</sup> EPA (Environmental Protection Agency). 2024. *Update on Pressures Impacting on Water Quality – May 2024*. EPA, Johnstown Castle, Ireland.

Reversing water quality declines in the midlands will require decades of coordinated restoration and rewetting of degraded peatlands. The SWAMP project's integrated findings underscore the need for

adaptive, multidisciplinary management – supported by flexible funding – to address peatland-specific complexities amid climate change.

# 1 Introduction

## 1.1 Peatlands and Water

Peatlands are an essential component of the world's freshwater resources, accounting for ~10% of the global freshwater (Xu *et al.*, 2018). In Ireland, peatlands are the main subclass of wetlands and cover c.23% (or 1.66 million hectares; Gilet *et al.*, 2024), occurring as raised bogs, blanket bogs or fens (Renou-Wilson, 2018). The variety of peatlands is the result of a key environmental factor that underpins their existence: water supply (Heathwaite and Göttlich, 1990). Water is the blood coursing through healthy natural peatlands, as high water saturation conditions (i.e. waterlogged) are required to slow the decay of dead plant material and accumulate organic matter as peat. Natural Irish bogs are typically more than 90% water by weight (Renou-Wilson *et al.*, 2022) and play a critical role in water cycling, regulating water regimes and immobilising pollutants in vegetation and peat (Shuttleworth *et al.*, 2019; Goudarzi *et al.*, 2021). Peatlands also interact with the landscape via hydrological pathways, affecting the chemical properties and biota of downstream aquatic ecosystems (Ramchunder *et al.*, 2012). However, land use or management changes and disturbances can lead to significant alterations in water quality and flood regime (Flynn *et al.*, 2022; Marcotte *et al.*, 2024).

## 1.2 Drained Peatlands and Their Impacts

Irish peatlands are naturally treeless open landscapes, which have been extensively modified over centuries, with more than 50% converted to conifer plantations or grassland and another 30% subject to peat extraction (Aitova *et al.*, 2023). Industrial peat extraction has been carried out on over c.100,000 ha, mostly by Bord na Móna (80,000 ha) and mostly on raised bogs in the Irish midlands. Following the cessation of all its industrial peat extraction operations, Bord na Móna is now currently rehabilitating 33,000 ha of these industrial cutaway peatlands. Rehabilitation, in the context of Bord na Móna's industrial peatlands and its EPA licence, aims to ensure that there is no risk to the environment arising from the peat extraction activity and its legacy. There should be no outstanding

environmental liability arising from the peat extraction. Opportunities to reduce carbon dioxide emissions will be highly site specific but are not directly regulated by a peat extraction licence. Therefore, rehabilitation can involve a large variety of practices, which may or may not include rewetting, and is compatible with certain activities, such as wind farms (NPWS, 2015).

Peat extraction is, however, still occurring widely in Ireland, especially large-scale illegal extraction for horticulture, currently under investigation by the EPA. Drainage, typically associated with these land use changes, has lowered the water table in and around those peatlands and compromised their ability to provide a range of ecosystem services, including water filtration and water supply, carbon storage/sequestration, biodiversity and sociocultural benefits (Renou-Wilson, 2018; Pschenyckyj *et al.*, 2021). Lowering the water table in any peatland leads to significant changes in peat properties due to subsidence and compaction (Renou-Wilson *et al.*, 2022). Drainage enhances the aerobic decomposition of organic matter, resulting in significant emissions of gaseous and fluvial carbon to the atmosphere (Evans *et al.*, 2015), as well as leaching of nutrients and pollutants (Price *et al.*, 2023).

## 1.3 Impacts of Extracted Peatlands on Water Ecosystem Services

The negative relationship between drained peatlands and water quality has now been internationally recognised (Evans *et al.*, 2015; Nieminen *et al.*, 2021; Williamson *et al.*, 2022). Drainage networks in extracted peatlands, be they cutaway or cutover bogs (Figure 1.1), control water table fluctuations, moisture storage and flow velocities, altering runoff pathways and residence time (Lane and Milledge, 2012). Prolonged drainage may cross thresholds, causing subsidence and enhanced cracking, triggering irreversible changes to hydraulic properties (Gatis *et al.*, 2023). Drainage and removal of vegetated surface layers for peat extraction have immediate and direct impacts on the hydrology of the peatlands, notably the way in which water is stored and flows off the peat surface (Price *et al.*, 2003). These altered



**Figure 1.1. Left: cutaway peatland (historical industrial extraction by Bord na Móna at Coolcraff, Co. Longford, 2020). Right: cutover peatland (domestic extraction by private contractors using the “sausage” technique at Ballydaly, Co. Offaly, 2021). Source: Florence Renou-Wilson.**

runoff regimes are accompanied by changes in the water quality of downstream water bodies, relative to intact peatlands, especially increased losses of fluvial dissolved organic carbon (DOC) and particulate organic carbon, the latter being potentially large due to fluvial erosion on bare peat (Strack *et al.*, 2011; Evans *et al.*, 2015). Drainage for peat extraction showed an increase in DOC concentrations of 54–83% (Glatzel *et al.*, 2003; Moore and Clarkson, 2007) as well as an increase in particulate organic carbon concentrations (Heikkinen, 1990; Worrall *et al.*, 2011). Furthermore, leaching of metals (iron, manganese) and nutrients, especially nitrogen (N) and phosphorus, have also been reported, impacting nitrogen and phosphorus dynamics in downstream water bodies (Heikkinen *et al.*, 1995; Holden *et al.*, 2004; Armstrong *et al.*, 2010; Kløve *et al.*, 2010). The intensive drainage network within cutaway peatlands leads to further hydrological changes, such as large runoff peaks and increased erosion, leading to siltation and eutrophication of downstream water courses due to high concentrations of suspended solids (Marttila and Kløve, 2010). Peatland drainage and extraction pose several significant challenges to aquatic life in downstream water bodies, including increased mortality due to habitat alterations, behavioural changes, shifts in community structure and an overall reduction in species richness (Ramchunder *et al.*, 2009; Donahue *et al.*, 2022).

In Ireland, some studies have investigated water quality from restored raised bogs (Schouten, 2002; Swenson *et al.*, 2019), while others have focused on fluvial carbon from agricultural peat soils (Renou-Wilson *et al.*, 2014) and on nutrients

from forested blanket bogs (Kelly-Quinn *et al.*, 2016; Flynn *et al.*, 2022) or afforested cutaway peatlands (Renou-Wilson and Farrell, 2007). Furthermore, O’Driscoll *et al.* (2018) demonstrated that peat soil presence in a catchment was a significant driver of total trihalomethanes exceedances (a carcinogen disinfectant by-product). This is because of the need to clean brown (carbon-loaded) water draining from degraded peat catchment. There is a paucity of Irish data pertaining to water quality associated with extracted peatlands, despite being correlated to poorer water quality in affected catchments (the EPA reports that 7.5% of poor inland water quality can be attributed to peat extraction (EPA, 2020a)).

#### 1.4 SWAMP Project Objectives

The EPA report *Water Quality in Ireland 2016–2021* highlighted that Irish surface waters are under a number of pressures resulting from human activities, thus leading to an overall decline in water quality (Trodd *et al.*, 2021). The report *Draft River Basin Management Plan for Ireland 2022–2027* indicates that 106 surface waters are currently impacted by peat extraction and drainage (DHLGH, 2021). This situation prevents Ireland from fully complying with the EU Water Framework Directive.

It is critical to understand the pressures on waters arising from degraded peatlands (cutaway and cutover). Although peat extraction for fuel is now largely restricted to domestic use, horticultural peat extraction is expected to continue due to rising global demand (Joosten and Tanneberger, 2017). Appropriate regulatory measures and innovative technologies

must be developed to ensure that the status of water bodies is protected, maintained and improved in line with the requirements of national and international environmental standards (Water Framework Directive and Floods Directive).

The aims of the Strategies to Improve Water Quality from Managed Peatlands (SWAMP) project were to (i) appraise the extent of contaminants in drained/extracted versus near-natural/restored peatland catchments (Chapter 2); (ii) investigate the

significance of peatland drainage and extraction, especially in terms of aquatic biodiversity (Chapter 3); (iii) evaluate the effluent dynamics from extracted peatlands, develop treatment methods to prevent and reduce pollution at peat extraction sites, and develop best practice guidelines to protect the environment (Chapter 4); and (iv) propose predictive tools for effective watershed management involving drained and restored/rewetted peatlands through the review of key physical processes involved in peatland hydrology and process-based modelling (Chapter 5).

## 2 Examination of the Influence of Peatlands on Regional Stream Water Chemistry

### 2.1 Introduction and Methodology

One of the objectives of the SWAMP project was to identify the impacts on water quality arising from the drainage and mining of peatlands by assessing a suite of water parameters vis-à-vis national thresholds and Water Framework Directive (2000/60/EC) targets. This chapter provides a snapshot of the quality of streams in the Irish midlands. Once home to the largest area of oceanic raised bog habitat, recent national monitoring surveys have indicated that 260,000 ha (84%) of the original 310,000 ha in this region have been impacted by industrial and domestic peat extraction (NPWS, 2017). A range of streams was sampled from 12 *degraded* bogs and 11 *natural* bogs during autumn 2020 and spring and autumn 2021 (Figure 2.1 and Appendix 1). The *degraded* bogs in this study were either under extraction or had just ceased industrial

extraction (fuel or horticultural peat) with additional domestic turf cutting on the margins. As there are no longer any pristine raised bogs in Ireland, the least-degraded or near-natural bogs (hereafter *natural* bogs) were included as part of the Natura 2000 network of raised bog special areas of conservation (SACs) and have been the subject of a range of restorative management practices over various periods, starting 15 years ago but mostly within the last 4 years (Appendix 1). At each bog, we sampled a cluster of streams: within-bog small streams and receiving streams, both upstream and downstream of the bog (Figure 2.2). These streams belong to catchments identified as being “at risk of not achieving good status” due to “extractive industry pressure associated with peat” or “peat-related activities” under the Water Framework Directive Significant Pressures classification (EPA, 2020b). Further details regarding

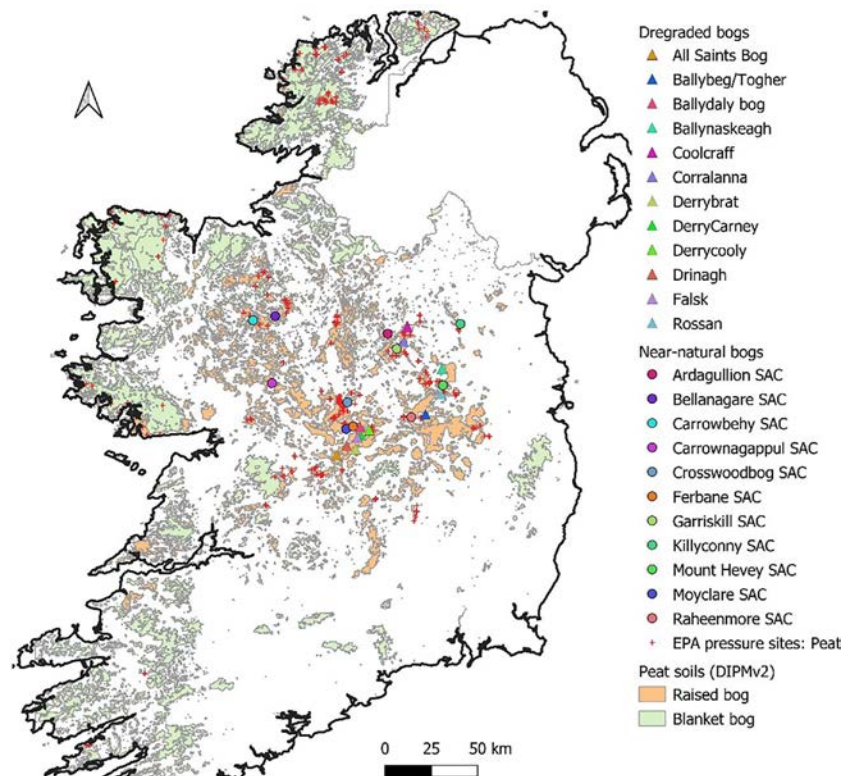
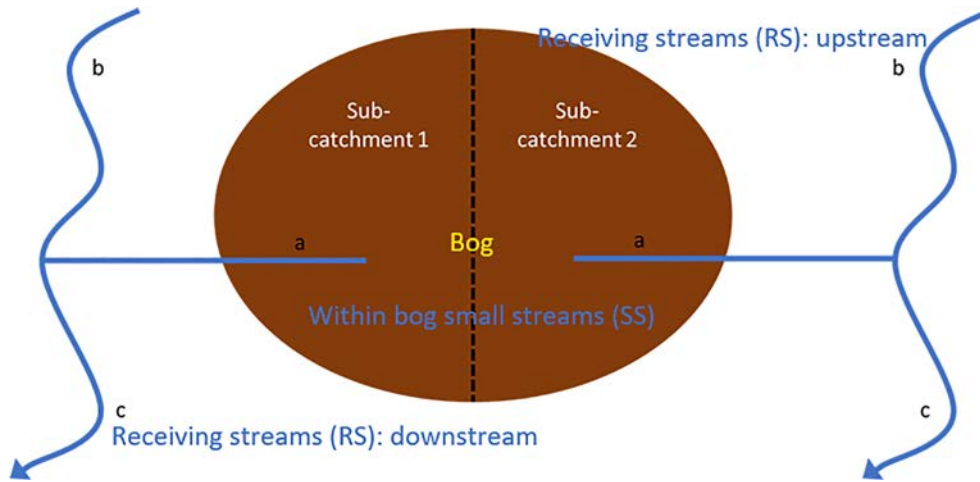


Figure 2.1. Map of Ireland showing peatlands types: EPA pressure sites (red dots; EPA, 2020b) associated with peat and locations of degraded (triangles) and near-natural (circles) bog sites sampled in this study. DIPMv2, Derived Irish Peat Map version 2. Source: adapted from Pschenyckyj *et al.*, 2023.



**Figure 2.2.** Conceptual diagram of the sampling strategy in a bog with two subcatchments. Samples were taken from (a) within-bog small streams (SSs), (b) upstream receiving streams (RSs) and (c) downstream RSs, where the within-bog SS was a tributary.

sampling and statistical analysis can be found in Pschenyckyj *et al.* (2023).

## 2.2 Key Results and Discussion

### 2.2.1 Water quality of within-bog small streams: degraded versus natural bogs

The chemical profile of bog streams was found to be very variable in space and time within both degraded and natural bogs (Table 2.1). Except for turbidity, all parameters had higher means in the degraded bog streams, but this difference was only statistically significant for electrical conductivity (EC), total dissolved nitrogen (TDN) and sulfate ( $\text{SO}_4$ ). The mean specific ultraviolet light absorbance at 254 nm ( $\text{SUVA}_{254}$ ) value was significantly lower in degraded bog streams. Overall, natural bog streams had a distinct cluster that was characterised by greater positive loading values for  $\text{SUVA}_{254}$  and negative loading values for all pollutants, EC and pH (Figure 2.3). In contrast, degraded bog streams were less tightly clustered and were most strongly influenced by total ammonia nitrogen (TAN), ammonia nitrogen ( $\text{NH}_3\text{-N}$ ) and sulfate loadings, as well as by greater EC, magnesium (Mg) and calcium (Ca) values. Of note, a subgroup of natural bog streams was directly associated with decreasing loading values for DOC and increasing TAN (Figure 2.3). These sites (Moyclare, Raheenmore, Carrowganappul and Mount Hevey bogs) have been partially restored, the

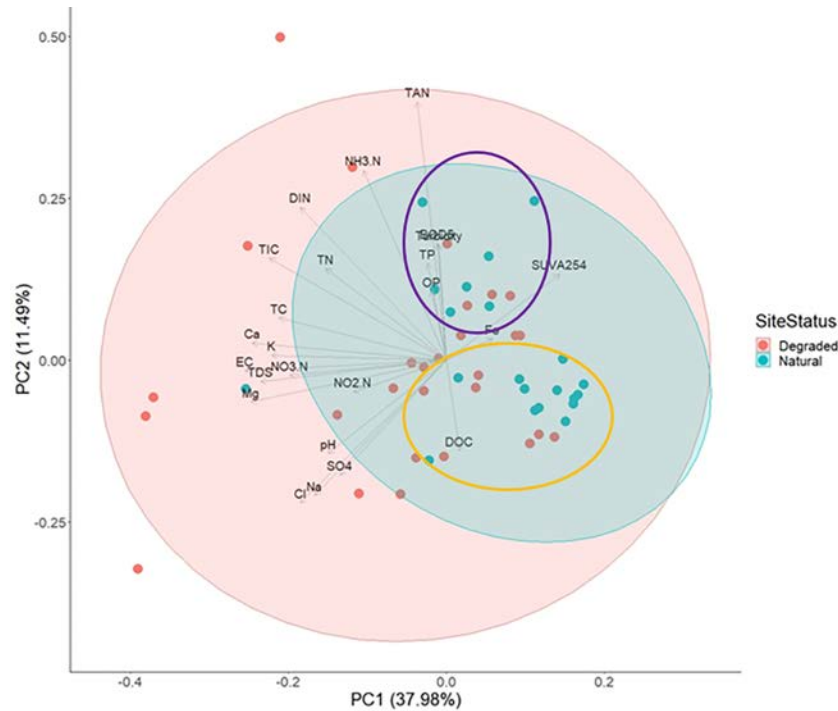
longest starting over 15 years ago (see Appendix 1), while more recently restored natural sites are more associated with increasing values of DOC, pointing to the need for time following restoration activities to have a positive impact on DOC fluxes.

**Table 2.1.** Key parameters means ( $\pm$  standard error) for within-bog streams in degraded and natural bogs

Parameter	Degraded	Natural
pH	7.04 $\pm$ 0.14	6.96 $\pm$ 0.11
EC ( $\mu\text{S/cm}$ )	<b>334.40<math>\pm</math>31.57</b>	<b>155.64<math>\pm</math>27.03</b>
TDS (ppm)	170.76 $\pm$ 16.98	64.44 $\pm$ 11.12
Turbidity (NTU)	4.11 $\pm$ 0.71	4.61 $\pm$ 1.22
DOC (mg/L)	34.74 $\pm$ 1.65	27.14 $\pm$ 2.20
$\text{SUVA}_{254}$ (L/mg/m)	<b>3.33<math>\pm</math>0.24</b>	<b>4.34<math>\pm</math>0.18</b>
TDN (mg N/L)	<b>1.89<math>\pm</math>0.15</b>	<b>1.04<math>\pm</math>0.09</b>
TAN (mg N/L)	0.48 $\pm$ 0.10	0.35 $\pm$ 0.12
$\text{SO}_4$ (mg/L)	<b>18.49<math>\pm</math>3.31</b>	<b>2.58<math>\pm</math>0.67</b>
Na (mg/L)	7.19 $\pm$ 0.57	6.25 $\pm$ 0.35
K (mg/L)	0.71 $\pm$ 0.10	0.55 $\pm$ 0.13
Mg (mg/L)	2.84 $\pm$ 0.35	1.83 $\pm$ 2.56
Ca (mg/L)	61.80 $\pm$ 7.90	34.39 $\pm$ 7.02
Fe ( $\mu\text{g/L}$ )	351.14 $\pm$ 68.78	543.48 $\pm$ 60.37

**Bold values signify significant differences ( $p \leq 0.05$ ). Variables in italics were excluded from the statistical analysis due to multicollinearity.**

**Fe, iron; K, potassium; Na, sodium; NTU, nephelometric turbidity unit; ppm, parts per million; TDS, total dissolved solids.**



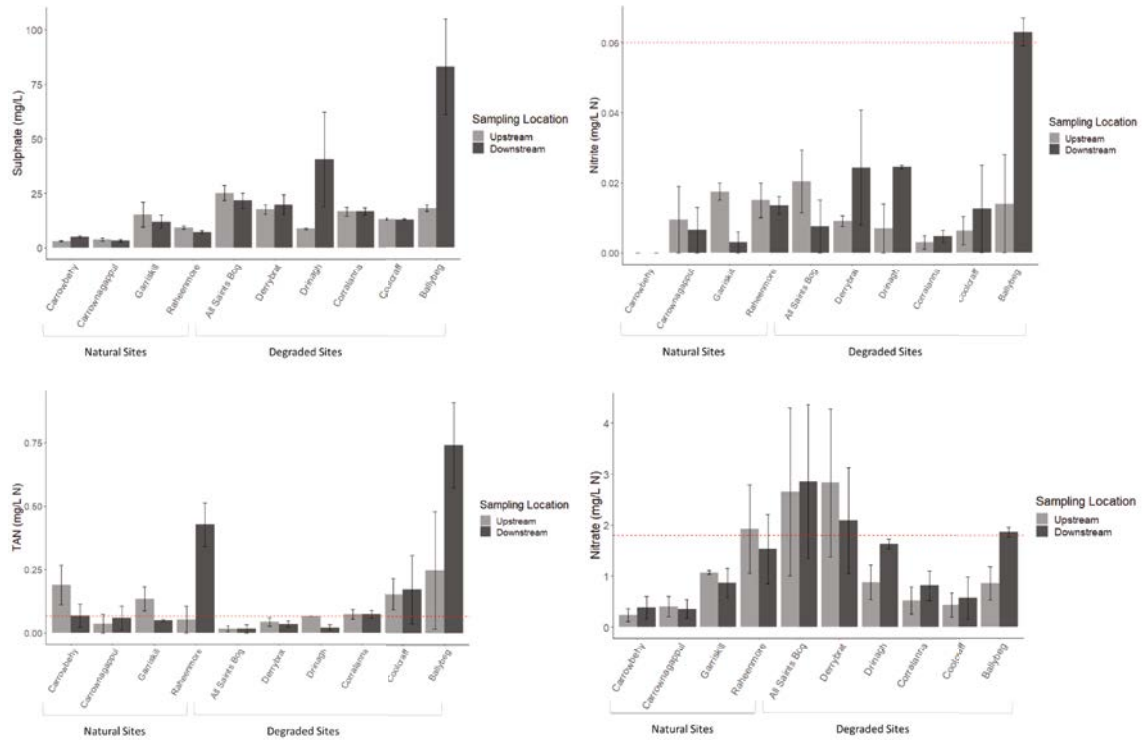
**Figure 2.3. Biplot of principal component analysis for within-bog small stream data comparing degraded (red) and near-natural (blue) sites. Sites within the purple ellipse are sites that have been restored over a long time frame (> 15 years), while sites within the orange ellipse are more recently restored sites (see Appendix 1). Source: adapted from Pschenyckj *et al.*, 2023.**

### 2.2.2 Water quality in receiving streams and issues at individual sites

With the exception of pH, magnesium and variables with negligible concentrations ( $\text{NH}_3\text{-N}$ , nitrite nitrogen ( $\text{NO}_2\text{-N}$ ) and organic phosphorus), mean concentrations of all parameters increased downstream of degraded sites. Some parameters (EC, total dissolved solids, total inorganic carbon, TDN, nitrate nitrogen ( $\text{NO}_3\text{-N}$ ), sodium, potassium, magnesium, iron) also increased downstream of natural sites.

Biological oxygen demand (BOD) concentrations were also higher downstream from degraded sites but lower in samples taken downstream from natural bogs over the course of this study. While showing a wide spectrum, the chemistry of the receiving streams (both upstream and downstream) was not significantly different between degraded and natural bogs. Taken together, the profile of streams draining bogs in the midlands is of concern. DOC concentrations across all receiving streams ranged from 7.2 to 63.9 mg/L with a mean value of 27.91 mg/L and a median value of 27.26 mg/L.

Finally, we compared the water quality of upstream and downstream streams of a certain cluster in the context of enforcement of existing regulations (Figure 2.4). Elevated BOD, sulfate, TAN, nitrate and nitrite concentrations were found in Ballybeg (degraded) bog, suggesting that the water quality of the Coolcar stream within the Yellow River catchment (north-east midlands) is impacted by this drained bog. BOD and TAN concentrations were also above “good status” thresholds at Corralanna and Coolcraff, both of which are located within the Inny River catchment. Of note, TAN concentrations downstream of Raheenmore were greater than the high-status guideline concentration by one order of magnitude, with a downstream mean value of  $0.43 \pm 0.09$  mg/L compared with an upstream mean value of  $0.05 \pm 0.05$  mg/L. Ammonia has been linked to microbiological shifts in bogs, from shifts in populations of typical bacteria and fungi to the proliferation of algae (Payne *et al.*, 2013). The Raheenmore site has seen an increase in algae slime growing on heather and, while nitrogen-sensitive species such as *Ramalina* spp. were still present, they were also being encroached on by algae (Kelleghan *et al.*, 2022). Meanwhile, the lowest and



**Figure 2.4. Bulk mean concentrations of sulfate, NO<sub>2</sub>-N, TAN and NO<sub>3</sub>-N upstream and downstream of individual bogs. The red dashed line represents the threshold for good ecological status (1.8 mg N/L) based on the EPA Environmental Quality Status for Ireland.**

highest values of DOC were recorded from the same downstream river of Garriskill, a natural bog that has been more recently restored (The Living Bog (Raised Bog Restoration Project)),<sup>2</sup> pointing to dynamic water chemistry soon after restoration activities.

### 2.3 Key Messages and Recommendations

This study presented a snapshot of the quality of streams in the Irish midlands, a region known for its historical expanse of raised bogs, most of which have been affected by peat extraction. Detailed observations and discussion on issues around EC, nitrogen, carbon and SO<sub>4</sub> at both the site and regional levels can be found in Pschenyckyj *et al.* (2023). In short, we presented evidence that bog streams within degraded bogs exhibit greater levels of pollutants, in particular TDN and sulfate, as well as higher EC and lower SUVA<sub>254</sub>. Since the relative contribution of dissolved organic nitrogen (DON) was greater than the mean dissolved inorganic nitrogen

in degraded bog streams, additional leaching of organic nitrogen is also likely. The higher EC also has serious implications for the rewetting and potential restoration of these degraded bogs. A neutral pH environment promotes peat mineralisation, especially in the presence of nutrients. Therefore, it is crucial to halt drainage-induced degradation immediately and to rewet and ideally restore the bogs to recover their biogeochemical functioning and associated water chemistry (Emsens *et al.*, 2020; Urbanová and Bárta, 2020). In addition, all bog streams (natural and degraded) contained a considerable amount of TAN with concentrations seven and five times the EPA Environmental Quality Status for Ireland for degraded and natural bogs, respectively. These levels are greater than those typically recorded in bog waters from northern and central Europe, but similar levels have been found in degraded bogs in Canada and the northern USA (Bourbonniere, 2009; Andersen *et al.*, 2010). Of note, the natural/restored bog Raheenmore had very high TAN concentrations that may be linked to the higher atmosphere NH<sub>3</sub> measured there

<sup>2</sup> <https://www.raisedbogs.ie/>.

(Kelleghan *et al.*, 2021), and warrants close monitoring of water and aerial nitrogen deposition.

In addition, at the regional level, the water quality of streams that receive bog waters did not significantly differ between natural and degraded sites. Only at specific sites were nitrogen pollutants attributed specifically to the bog stream and therefore should be acted on. This is evidence that the region investigated is experiencing long-term widespread loss of fluvial nitrogen and carbon into the streams and has a reduced influence from natural bog water, which is typically acidic and nutrient poor, confirming the widespread degradation of these ecosystems. Critically, DOC values were high (overall median value 27.2 mg/L), nearly five times higher than other Irish streams (Liu *et al.*, 2014), and also greater than any other regions with drained peatlands (Ryder *et al.*, 2014; Kelly-Quinn *et al.*, 2016). The high upstream DOC concentrations recorded in our study, regardless of the bog status of the tributary, explain the lack of impact of within-bog water in this landscape.

The implications of the degraded water quality in this region are manifold, but, critically, it is likely impacting other aquatic ecosystem services, including drinking water and biodiversity (see Chapter 3). To assist with meeting water quality standards in the region, our study points to required actions: (i) *at the site level, through water treatment at extracted bogs* (see Chapter 4); and (ii) *at the landscape level, via large-scale rewetting of drained bogs* (see Chapter 6). To this effect, and in view of these findings, we support *the long-term seasonal monitoring* of the water chemistry of bog water and receiving streams, as well as investigating groundwater within the wider landscape.

### ***2.3.1 Towards a seasonal monitoring tool: water quality indicators***

While our results show large spatial and temporal variations associated with the wide range of site characterisations, they also reveal the baseline water quality in a landscape, which is critical for future restoration work. While terrestrial biota of most protected bogs is carefully monitored in terms of habitat integrity (NPWS, 2017), there is often a lack of information regarding the aquatic environment associated with these ecosystems. Targets are often only concerned with terrestrial criteria (e.g. vegetation). Long-term monitoring of the chemical composition of bog water and associated water bodies in conjunction with bog hydrological data is critical for a full understanding of the impacts of large-scale efforts to rewet/restore peatlands and for the development of robust ecohydrological targets. In particular, EC, DOC and TDN have been identified as important indicators to monitor and develop models in order to evaluate current and future peatland management activities in the region, especially rewetting/restoration efforts. Since EC measurement is inexpensive and rapid, widespread and routine monitoring could be deployed around bog sites to ascertain the status of the site and also to monitor the effects of management actions. Finally, seasonal sampling may also impact water quality from these ecosystems and should therefore be carefully addressed in any monitoring programme. For example, our results indicate greater concentrations of TDN in the spring, as well as lower SUVA<sub>254</sub> values (i.e. less labile dissolved organic matter which is more difficult to treat). Therefore, seasonal monitoring may be important to develop end-of-pipe treatment solutions in this region.

# 3 The Impact of Peatland Drainage and Extraction on Aquatic Communities

## 3.1 Review of Impacts of Inputs from Drained Peatlands on Aquatic Communities

While peat extraction has been identified as a significant pressure on Irish surface waters, there are relatively few studies on the effect on aquatic biota of stressors from extracted peatlands, particularly for phytoplankton, macrophytes and fish. We carried out an extensive review of existing data on the impacts of physico-chemical stressors such as ammonia, heavy metals and peat sedimentation on the aquatic life of peatland streams and the results have been published in Donahue *et al.* (2022). The findings indicate that peatland drainage and extraction pose several significant challenges to aquatic life, including increased mortality, reduced species richness, behavioural changes, habitat modifications and alterations in community structure (Surette *et al.*, 2002; van Duinen *et al.*, 2003; Camargo and Alonso, 2006; van Duinen *et al.*, 2008; Ramchunder *et al.*, 2009; Verberk *et al.*, 2010; O'Driscoll *et al.*, 2014). The review also discusses the benefits of peatland restoration through drain-blocking for aquatic biota and highlights the substantial knowledge gaps in understanding the effects of peatland-derived stressors on surface water communities (van Duinen *et al.*, 2003; Ramchunder *et al.*, 2012), prompting the need for field collection of baseline aquatic biota data from streams draining Irish natural and drained peatlands.

## 3.2 Field Investigations: Aquatic Macroinvertebrate Communities

The ecological investigations targeted aquatic macroinvertebrates and phytoplankton, both being important indicators of water quality and representing key functional components of freshwater trophic structure. Macroinvertebrate samples were collected at sites on streams originating within the degraded (8 sites) and natural (10 sites) bogs in spring 2021 (April–May), hereafter referred to as within-bog sites (see Figure 2.2 and Appendix 1 for details of the

streams), and at a subset of these in autumn 2021 (October–November). A further 11 streams were sampled upstream and downstream of the receiving stream in spring 2021, hereafter referred to as pre- and post-bog sites. Phytoplankton were collected from artificial substrates planted in six natural and degraded sites in 2023. These were all within-bog streams, with the exception of Corralanna where a large drain that connects the site to a stream was used. Five unglazed tiles were attached to periphytometers and driven into the stream bank. Phytoplankton were allowed to colonise for 6–8 weeks. Further methods and statistical results can be found in the end-of-project report.

### 3.2.1 Macroinvertebrates

A total of 108 taxa were recorded across the 18 sites in spring, while 85 taxa were recorded in autumn. However, the number of taxa recorded at any one site was relatively small, ranging from 7 at a degraded site to 36 at a natural site. In spring, the natural sites were numerically dominated by *Gammarus duebeni* (27.4%) followed by Coleoptera (20.3%), Chironomidae (17.9%) and Trichoptera (16.4%) (Figure 3.1). In contrast, *Asellus aquaticus* (40.5%) accounted for the bulk of the abundances at the degraded sites in addition to high representation of Chironomidae (22.9%). Ephemeroptera (0.4%) and Plecoptera (0.05%) occurred in exceedingly small percentages at both site types (<1%). Coleoptera were poorly represented at the degraded sites (1.16%). Gammarids and Chironomidae made up a slightly lower and higher proportion of the community at the natural sites in autumn and spring, respectively. *Asellus* remained the dominant taxon at the degraded sites in autumn. Further investigations into the aquatic biodiversity showed no significant differences in any metrics from the autumn sampling. In the spring sampling, the differences in total taxon abundances and richness were not statistically significant, although measurements were higher at the natural sites (Figure 3.2). However, total abundances and richness of EPT (Ephemeroptera/Plecoptera/Trichoptera)

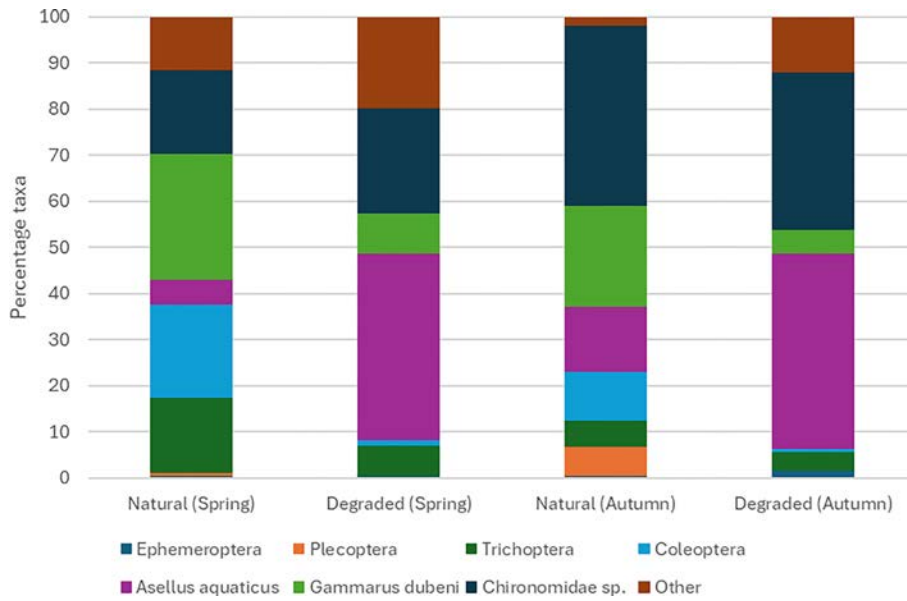


Figure 3.1. Distribution of macroinvertebrate taxa at the natural and degraded within-bog streams in spring and autumn.

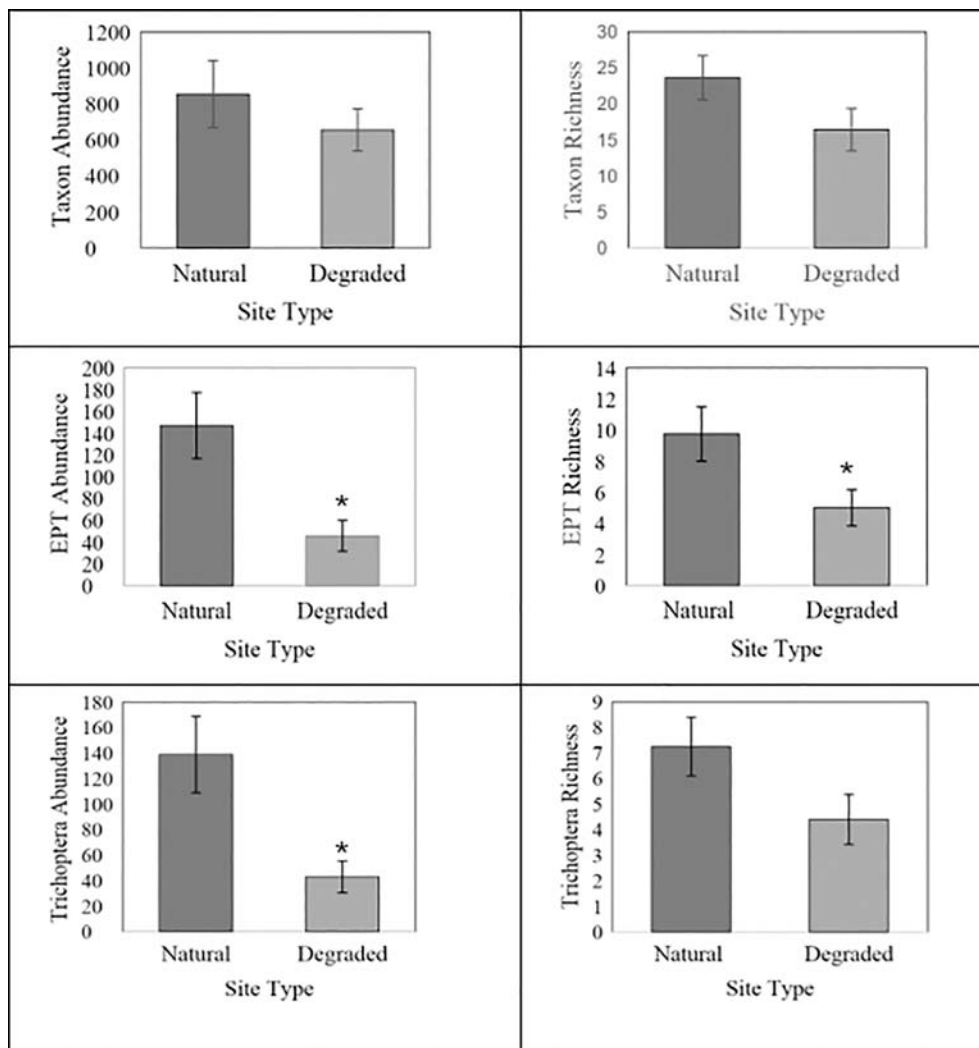


Figure 3.2. Macroinvertebrate metrics from the spring 2021 sampling period at natural and degraded within-bog streams. \*Significant difference ( $p < 0.05$ ).

taxa were significantly higher in the natural sites. Trichopteran abundances likewise showed significant differences, being three times more abundant in natural sites, but this difference did not hold for trichopteran richness. There was also a significant difference in terms of community structure. *Asellus aquaticus* was responsible for 21.17% of the difference in community structure, with a substantially greater average abundance found at degraded sites (Table 3.1). The opposite was true for *Gammarus duebeni*, with an average abundance of 232 found at natural sites compared with 57.1 at degraded sites, accounting for a 15.67% difference.

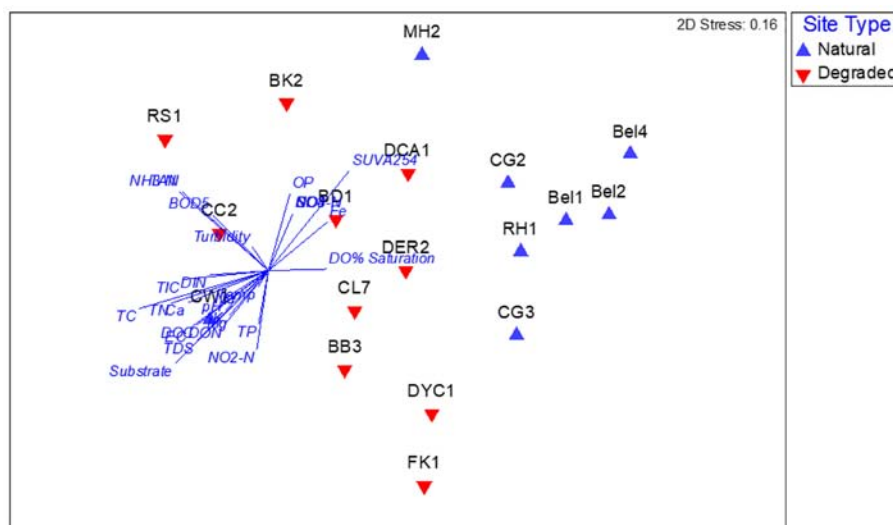
Community composition also showed significant differences between natural and degraded sites, with *Erpobdella octoculata* topping the list at 2.67% and found only in natural bog streams (see data in

the end-of-project report). While no taxon accounted for more than 3%, there was good separation of the sites on the non-metric multidimensional scaling plot analysis using the water quality variables as vectors (Figure 3.3), showing the impact of environmental conditions, in particular peaty substrates, slow flow and low oxygen concentrations, which limit the types of species that can inhabit bog streams.

No effect of the inflowing bog stream was detected in the comparison between pre- and post-bog sites. The only significant differences found were between the composition of natural and degraded pre-bog sites, which differed in their total species richness, total EPT richness and trichopteran richness. Pre-bog streams within the degraded bogs were already impacted, as shown in Figure 3.3, by severely reduced trichopteran richness.

**Table 3.1. Average taxon abundances (number from kick/sweep samples) and their percentage contribution to dissimilarity in the macroinvertebrate community structure of natural and degraded within-bog sites**

Taxon	Average abundance (individuals per sample)		Contribution (%)
	Natural	Degraded	
<i>Asellus aquaticus</i>	45.75	265.20	21.17
<i>Gammarus duebeni</i>	232.00	57.10	15.67
Chironomidae	151.75	149.90	11.35
<i>Elmis aenea</i>	100.00	0.50	9.34
<i>Pisidium</i> sp.	35.75	76.40	7.83
<i>Limnephilus marmoratus</i>	52.13	10.00	5.27



**Figure 3.3. Non-metric multidimensional scaling plots based on macroinvertebrate community structure with a subset of the environmental variables represented as vectors (see Appendix 1 for site abbreviation and description).**

### 3.2.2 *Phytobenthos: within-bog sites*

Average concentrations showed significant differences between natural and degraded sites for cyanobacteria and diatoms but not for green algae. Values were higher in the natural sites for both cyanobacteria and diatoms (Figure 3.4).

A total of 118 diatom taxa were recorded across all tiles from the within-bog sites. The number of taxa (100) recorded in the degraded bog sites was almost double that from the natural sites (57). However, differences in diatom taxa between the two site types were not statistically significant for richness, abundance, community structure or composition. Of note, some 55% of the taxa from the degraded sites had single-site occurrences compared with 39% for the natural sites (Table 3.2).

The taxa with the highest contribution to the abundance in the natural sites were *Achnanthydium minutissimum* (42.5%) and *Achnanthydium nanum* (20.2%). *Achnanthydium minutissimum* was also

the most abundant taxon across all sites, followed by *Achnanthydium nanum* (13.81%) and *Sellaphora saugerresii* (12.6%). Just over 30% (36 taxa) of diatom taxa were common to both degraded and natural sites with the percentage of taxa unique to degraded sites almost twice that recorded in the natural sites.

### 3.2.3 *Water quality metrics*

There was no difference in the site types using the Trophic Diatom Index, which scored sites BEL2 and CL7 as “good status” while all others were rated “high status”. Small stream impact score (SSIS) is a pollution risk assessment index based on macroinvertebrates and designed for assessment of small streams. Scores were calculated for within-bog sites sampled in spring (see the end-of-project report). Except for three of the natural sites, all scores were indicative of impacted conditions (<6.5).

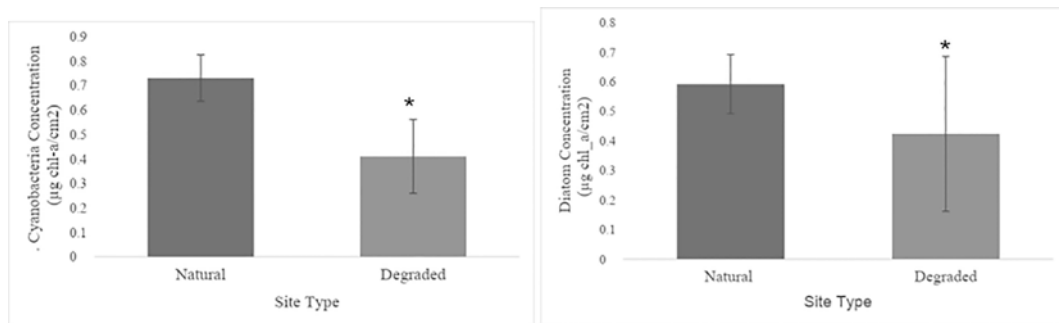


Figure 3.4. Mean concentration of cyanobacteria (µg chlorophyll a (Chl-a)/cm<sup>2</sup>) (left) and diatoms (µg Chl-a/cm<sup>2</sup>) (right) for natural and degraded within-bog sites. \*Significant difference ( $p < 0.05$ ).

Table 3.2. Diatom taxon richness counts at each site and average diatom taxon richness for natural and degraded bog sites

Natural site ID	Taxon richness	Degraded site ID	Taxon richness
CG2	15	BD1	49
CG3	22	BK2	18
MH2	21	DYC1	12
BEL1	14	FK1	18
BEL2	31	CL2	47
BEL4	20	CL7	32
<b>Average natural</b>	<b>20.5</b>	<b>Average degraded</b>	<b>29.3</b>

Site ID details can be found in Appendix 1.

### 3.3 Implications for Ecological Water Quality Assessment and Restoration

Our research gathered critical baseline aquatic biota data from bog streams draining natural and extracted peatlands and supports the existing evidence in the literature that drainage poses significant threats to aquatic biota in what is clearly a multi-stressor environment. Degraded bog streams had lower total abundance and richness of EPT and lower biomass of phytobenthos such as cyanobacteria but higher diatom richness due to the range of water chemistry. Macroinvertebrate community composition also differed, with *Asellus* as the dominant taxon in the degraded sites and *Gammarus duebeni* in natural sites. However, the distal downstream influence

was not observed, as differences were not due to the bog status but upstream water quality status. The environmental conditions, in particular peaty substrates, slow flow and low oxygen concentrations, limit the types of species that can inhabit bog streams. This has implications for the use of existing water quality metrics. We tested the applicability of SSIS for distinguishing the natural sites from the degraded sites. With the exception of three natural sites, the SSIS scores were indicative of impacted conditions. This spatial-temporal heterogeneity would require further research to develop more specific water quality indices to take account of the variation in natural bog streams (e.g. the presence of stony and peaty streams). This would help, in turn, with monitoring of restoration projects.

# 4 Investigations into the Treatment of Water Quality from Extracted Peatlands

## 4.1 Background and Experimental Site Description

Peat extraction has been shown to impact 7% of Irish surface waters (EPA, 2019). Chapter 2 revealed that nitrogen concentrations in particular are higher in streams receiving effluent from specific extracted peatlands than those from natural peatlands, pointing to the need for additional site-level treatment. Therefore, an in-depth investigation of the effluent water quality dynamics from an extracted peatland was carried out and innovative treatment methods tested.

The experimental site was an industrial cutaway raised bog named Corralanna (53°41'54.8"N, 7°24'53.6"W, 110 m above sea level) located in the Irish midlands. It is a relatively small site, comprising three mini-subcatchments connected to significant freshwater bodies, in particular the Inny River (order 4, "at risk" under the Water Framework Directive) (Figure 4.1). The site was drained in the 1980s (1 m deep drains every 15 m) and was cleared of vegetation and milled for horticultural peat from 2012 until 2020. Removal of peat stocks from the site ended in 2023. The average peat depth is over 5 m with a nutrient-poor

sphagnum peat at the surface displaying the following properties: pH 4.1, 48.5% carbon and 0.8% nitrogen. We instrumented the eastern subcatchment (16.4 ha), which drains water into a sedimentation pond, exiting into a ditch before entering the Ferskill stream (Mendes *et al.*, 2025). The mean annual rainfall was 1047 mm over the study period (2021–2023) and further climatic data can be found in Mendes *et al.* (2025).

## 4.2 Water Quality Monitoring of an Extracted Peatland Catchment

### 4.2.1 Monitoring station for effluent water quality and quantity

A monitoring station containing an automatic water sampler (Teledyne ISCO), a bubbler flow module, an area velocity flow meter, a multiparameter EXO2 sonde (YSI Xylem) and a rain gauge was installed at the outlet of the sedimentation pond (Figure 4.1). Fourteen storm event batches of samples were collected from March 2021 to March 2023 in the four seasons. Further methodologies are described

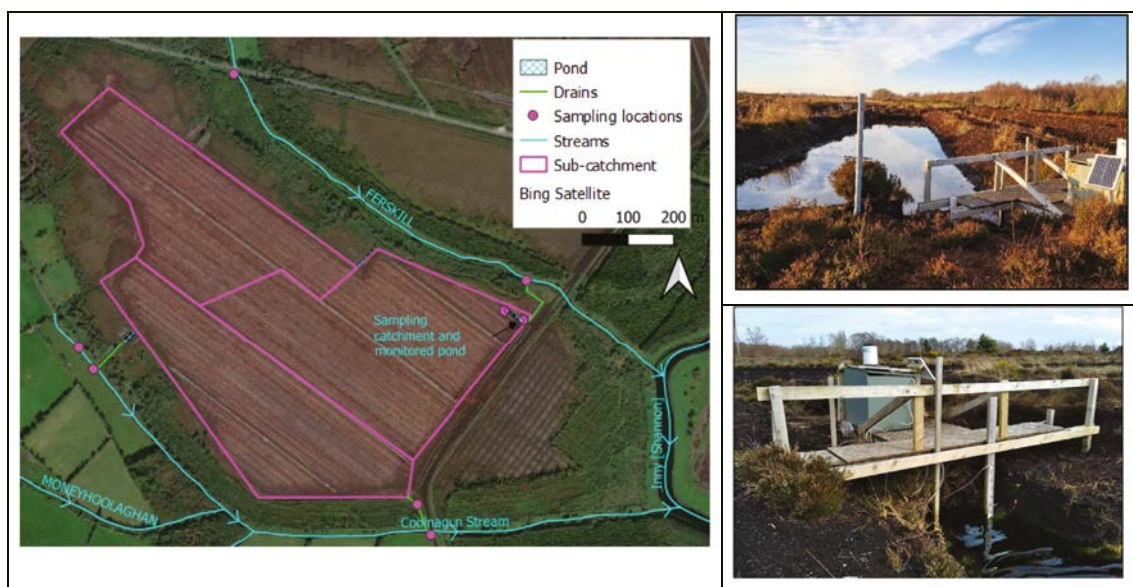


Figure 4.1. Aerial view of Corralanna, Co. Westmeath, a Bord na Móna industrial cutaway peatland showing subcatchments and drains reaching local stream network with photos of instrumented pond.

in Mendes *et al.* (2025). Mendes *et al.* (2025) comprehensively reported the nutrient dynamics of the extracted peatland catchment Corralanna and key results are presented here. The effluent water at the outlet of the pond had high concentrations of DOC, NO<sub>3</sub>-N and NH<sub>3</sub>-N, whereas concentrations of NO<sub>2</sub>-N, TDP, organic phosphorus and total suspended solids (TSS) were negligible. Variation in nutrient concentrations was great between storm events but generally small in single events (Figure 4.2). It was particularly low at high-flow events, suggesting a

dilution effect, which is supported by the decrease in concentration in events with a sharp increase in flow (Figure 4.3). The nutrient export was high only for short periods during extreme or peak flows, whereas low nutrient exports were common, occurring during continuous low to moderate flows (Table 4.1). While heavy rainfall greatly controlled nutrient exports, cumulative nutrient export during the long periods of low to moderate flow accounted for a greater fraction of total exports (Table 4.1).

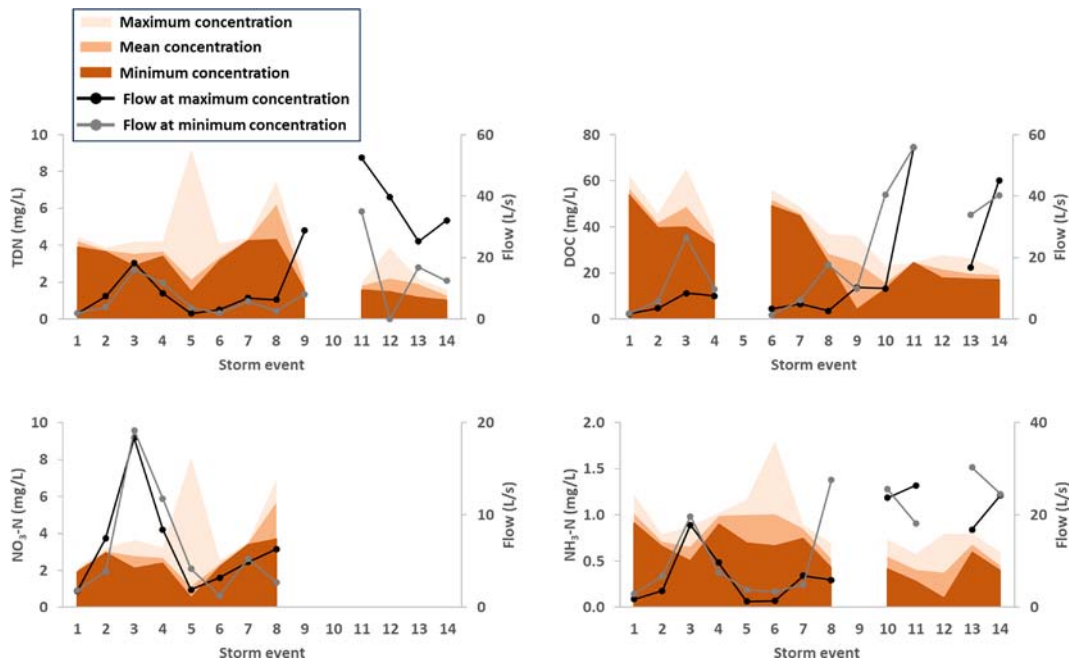


Figure 4.2. Nutrient concentrations and associated flows at the outlet of Corralanna during storm events, as reported in Mendes *et al.* (2025). Gaps indicate lack of data.

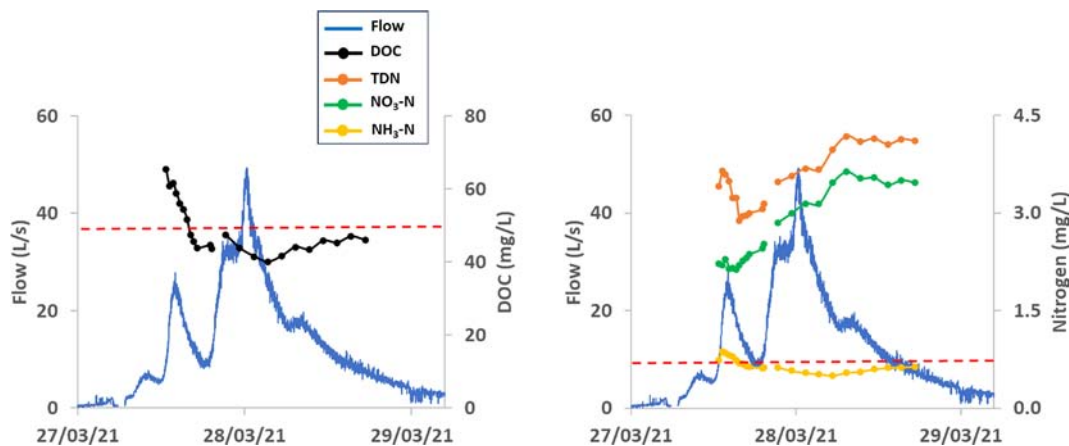


Figure 4.3. Nutrient concentrations in a storm event at Corralanna with a sharp increase in flow. The red dashed lines indicate DOC and nitrogen concentrations at 50 and 0.75 mg/L, respectively. Gaps indicate lack of data. Source: adapted from Mendes *et al.* (2025).

**Table 4.1. Nutrient export according to flow range and duration as reported in Mendes *et al.* (2025)**

Flow range (L/s)	Flow duration (% of time)	DOC (%) <sup>a</sup>	TDN (%) <sup>a</sup>	NO <sub>3</sub> -N (%) <sup>a</sup>	NH <sub>3</sub> -N (%) <sup>a</sup>
>25	1.7	74.8/19.9	86.0/30.1	85.9/34.3	74.1/16.4
5–25	13.3	21.6/39.6	11.8/32.7	12.6/37.7	20.9/31.1
<5	84.9	3.6/40.6	2.2/37.2	1.5/28.1	5.0/52.5

<sup>a</sup>Values on the left indicate export as measured during the storm events, and values on the right indicate export as estimated for the full monitoring period from March 2021 to March 2023.

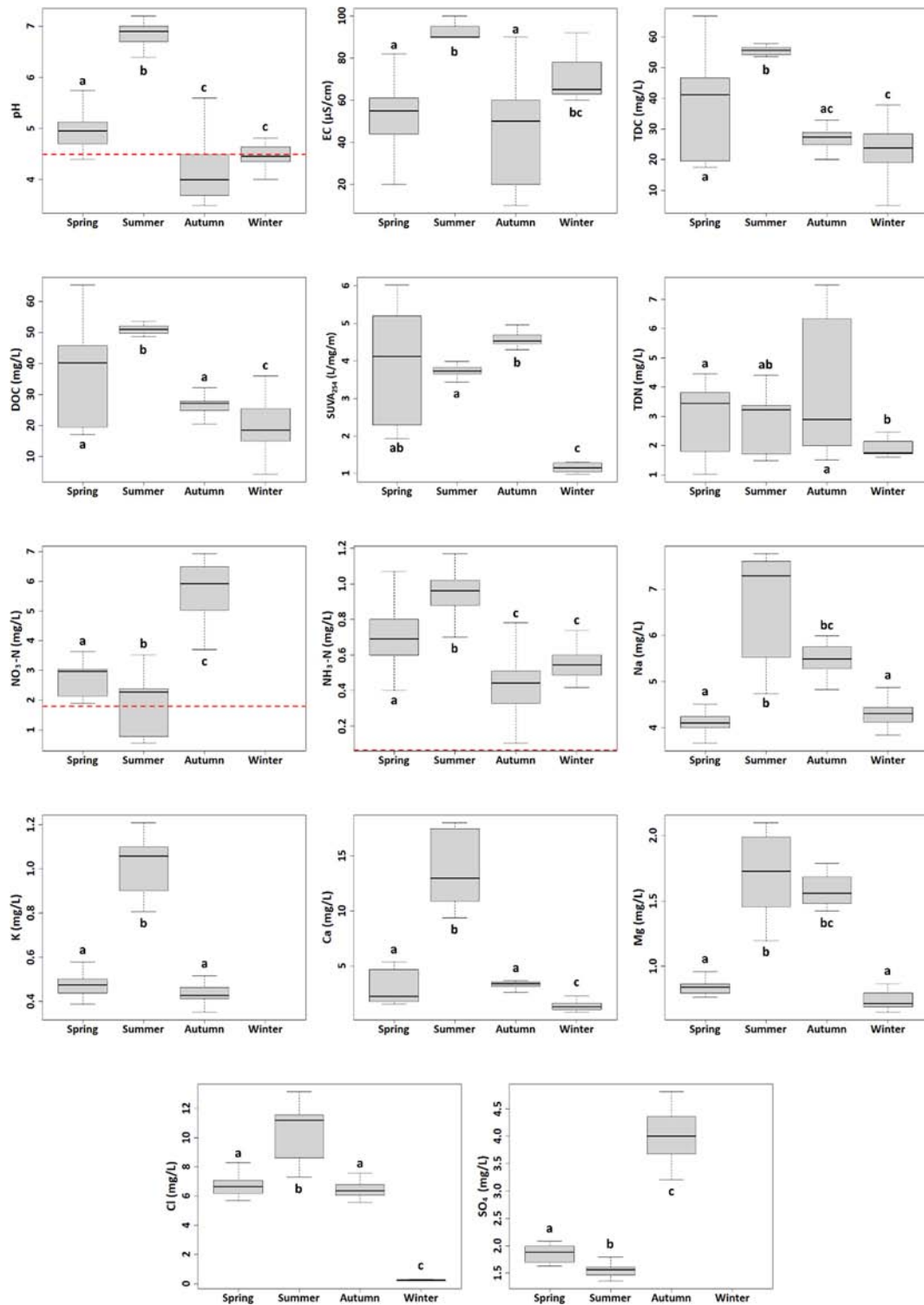
Annual mass export of nutrients was estimated using the concentration data from the 14 storm events captured by the sampler and the associated flows at the time of sampling, as described in Edokpa *et al.* (2015). The total fluxes for DOC were 148 kg C/ha/y and for TDN 15.7 kg N/ha/y. The annual mass exports were calculated as follows: 2432 kg DOC and 258 kg TDN (mostly nitrate but including 50 kg NH<sub>3</sub>-N). The fluvial exports are similar to other elevated nutrient leaching loads measured in Finnish peat mining wastewater (15 kg N/ha/y) (Kløve, 2001) and from bare peat blanket bogs in the UK (Edokpa *et al.* 2015, 2017; Pickard *et al.*, 2022). The DOC export figures are unexpectedly low (close to natural blanket bogs in the west of Ireland (Koehler *et al.*, 2009)), given the high level of disturbance (high drainage intensity) at this cutaway peatland site, which typically correlates with increases in waterborne DOC losses (Evans *et al.*, 2015). This is probably due to several causes: (i) low annual mean rainfall; (ii) presence of a poorly humified sphagnum peat (DOC fluxes are similar to agricultural catchment on drained nutrient-poor peat soils in the midlands (Barry *et al.*, 2016)); (iii) high conductivity/ionic strength and high sulfate concentrations (see Chapter 2), both of which reduce the solubility of DOC (Clark *et al.*, 2005); and (iv) the fact that some of the drains may intersect the underlying mineral soils can increase the retention of DOC on mineral surfaces (Åström *et al.*, 2001).

The water quality of the effluent at the outlet of the pond (i.e. at the drain entering the Ferskill stream) varied widely, commonly including significant differences between seasons (Figure 4.4). Water pH was acidic in spring, autumn and winter (i.e. <6.0), and neutral in summer. DOC, NH<sub>3</sub>-N and several ion concentrations were clearly higher in summer. SUVA<sub>254</sub> was clearly lower in winter, indicating an increase in the proportion of labile carbon compounds. Water pH, NO<sub>3</sub>-N and NH<sub>3</sub>-N often exceeded the environmental quality standards (pH range 4.5–9.0,

1.8 mg NO<sub>3</sub>-N/L and 0.065 mg NH<sub>3</sub>-N/L) for good ecological status of surface waters (Murphy, 2019). Climatic factors, including rainfall (proxy for flow) and water temperature, explained generally well the variability in water quality parameters, with pH also regulating carbon and ion concentrations. Mendes *et al.* (2025) observed that warm periods with little rainfall appear to neutralise the water pH but increase the concentration of nutrients, especially carbon, and ions, whereas cold and rainy periods appear to acidify the water pH but decrease the concentration of nutrients, especially carbon and ions, which may be due to a dilution effect.

#### 4.2.2 Continuous high-resolution monitoring

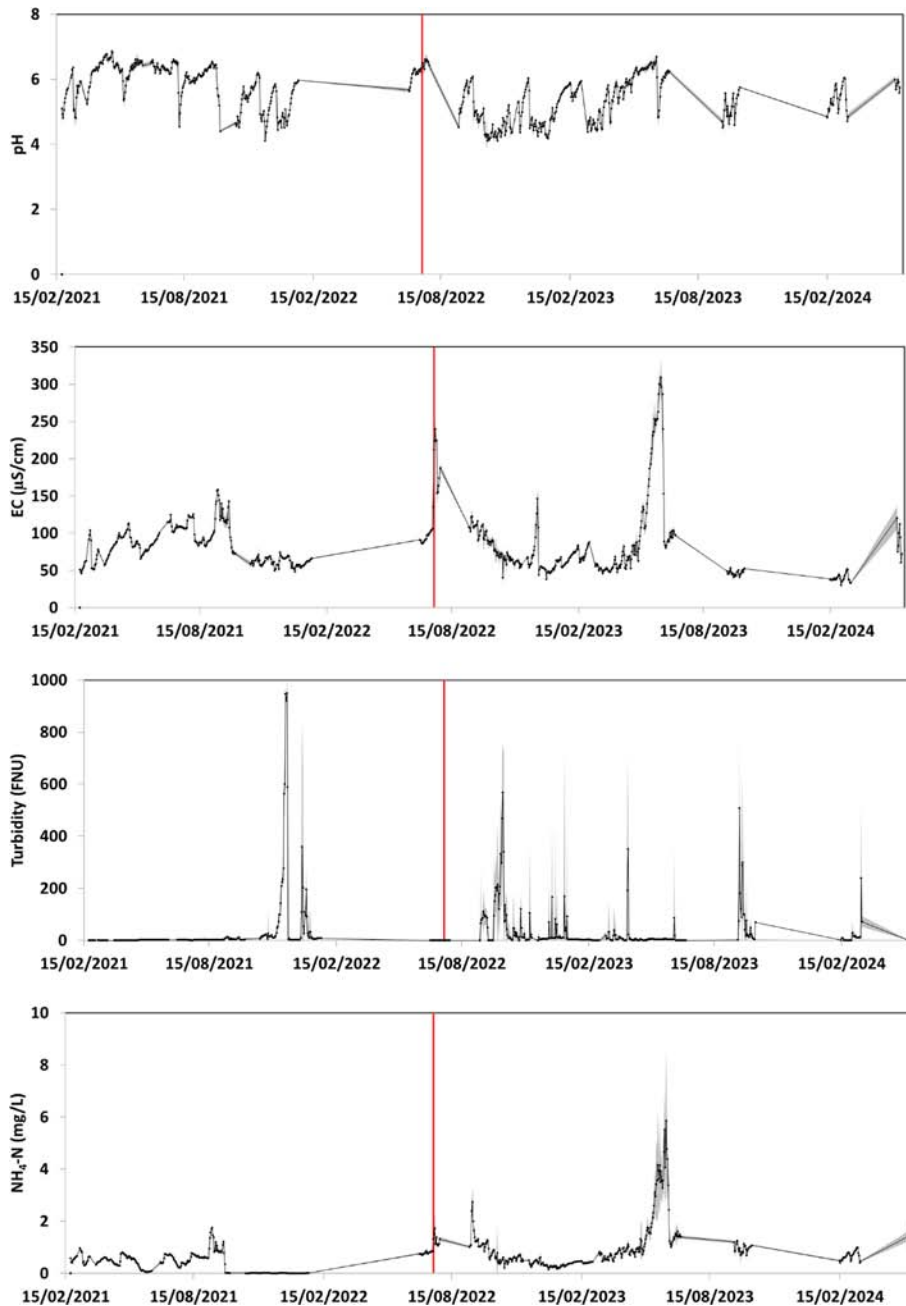
A multiparameter EXO2 sonde was used in the monitoring station at the pond outlet to carry out high-resolution measurements of pH, EC, turbidity, ammonium nitrogen (NH<sub>4</sub>-N) concentration and fluorescent dissolved organic matter every 30 min (see the end-of-project report for maintenance and operation details). The sonde operated from March 2021 to July 2022, when it was moved to a monitoring station at the pond inlet, where it operated until May 2024. Sonde measurements and measurements from grab samples were not significantly different ( $p > 0.05$ ) for all parameters, demonstrating that we obtained a robust dataset using the sonde. Fluorescent dissolved organic matter was also measured, but this dataset will be converted into DOC data once a model currently developed for this purpose has been published (Cox, 2025). The Sonde datasets revealed continuous variation in the effluent pH, EC, turbidity and NH<sub>4</sub>-N concentration (Figure 4.5). Water pH varied quickly within a range of 4–7. Similarly, EC varied quickly and was mostly in the range of 50–150  $\mu$ S/cm, with periodic slow increases followed by sharp decreases after high-flow events (e.g. in July 2022 and June 2023). At the inlet, this is explained by increased flow from low EC water that ran off over sphagnum peat. At the outlet,



**Figure 4.4. Water quality parameters at the outlet of Corralanna during storm events across seasons. The red dashed lines indicate the threshold environmental quality standards for good ecological status of surface waters as per the Water Framework Directive. Letters indicate where significant differences occur between groups ( $p < 0.05$ ). Source: adapted from Mendes *et al.* (2025).**

these cycles suggest a small additional subsurface inflow to the pond characterised by high EC (in contact with the sub-peat mineral soils), which supports the findings that the hydraulic load at the outlet was

higher than at the inlet (Mendes *et al.*, 2025). Water pH and EC had relatively small diel changes, unlike turbidity, which showed sharp diel variations. Turbidity was commonly close to or equal to zero formazin



**Figure 4.5. Variation in the effluent pH, EC, turbidity and NH<sub>4</sub>-N concentration at the outlet of Corralanna as measured by the sonde when it was at the pond outlet (left of red line) and inlet (right of red line).**

nephelometric units (FNU), except for sharp spikes up to around 950 FNU during autumn and winter. NH<sub>4</sub>-N varied mostly below 2 mg/L, except for a sharp rise that occurred from May to June 2023. Overall, the sonde data were in harmony with the data collected using the automatic water sampler but gave a more accurate understanding of the rapid changes in water parameters in these extracted peatlands. The effect of climate on the concentration and export of nutrients in effluents may have further implications. This is supported by a few studies simulating the response

of degraded peatland catchments to increased temperatures and rainfall, in which increased DOC and NH<sub>4</sub>-N fluvial exports were found (Salimi and Scholz, 2021; Rosset *et al.*, 2022).

### 4.3 Site-based Mitigation Measures

#### 4.3.1 Water treatment by sedimentation pond

Bord na Móna has an Integrated Pollution Control (IPC) licence from the EPA for all extracted bogs and

follows similar peat extraction operations on all its industrial cutaway peatlands, including the excavation of sedimentation ponds that collect effluents from a suitably large drainage network, allowing treatment before discharge into surface waters. The IPC licence, in this instance, contains specific ranges for pH, emission limit values for suspended solids, and trigger levels for ammonia (as nitrogen) and chemical oxygen demand (EPA, 1996). The ponds aim to retain excess particles through sedimentation due to an extension of the effluent hydraulic retention time (HRT). This study investigated the efficiency of the ponds and whether prolonging the HRT of the effluents on site is sufficient to achieve adequate treatment. The water treatment performance of the instrumented pond was assessed and reported in Mendes *et al.* (2025). In short, the pond has a rectangular shape, with a length, width and depth of 49.4, 8.2 and 2.4 m, respectively, and an area and volume of 405 m<sup>2</sup> and 959 m<sup>3</sup>, respectively, representing 0.25% of the catchment area (Figure 4.1). Grab sampling campaigns at the pond inlet and outlet, recording pH, EC, nutrients, ions, TSS and turbidity, as well as manual measurements of dissolved oxygen and temperature across the pond, and measurements of the amount of sediments deposited in the pond

between cleaning operations, were carried out over a 3-year period.

There were small and non-significant changes in the water quality of the effluents from the pond inlet to the outlet, except for TSS (Figure 4.6). This was also evident for SUVA<sub>254</sub>, biochemical oxygen demand and iron. Hence, the results indicated minimal treatment of soluble components. This occurred despite the pond having long HRTs (90 ± 551 days, median 15 days). Water pH at the pond outlet was still outside the range of limit values for discharges to water as set in the IPC licence (6–9; EPA, 1996), often being below 6. However, the pond retained particles such as TSS (Figure 4.6). This successful retention was demonstrated particularly during more intense rainfall events that mobilised more particles to enter the drainage system and the pond. Therefore, the study findings suggest that retention of TSS increases when a pond's amount of TSS increases. The high outlet TSS in the first sampling occurred shortly before the pond was cleaned, when it was considerably silted, indicating resuspension and discharge of particles. The pond retained an average of 23 and 15 L of sediment per day in two measurements.

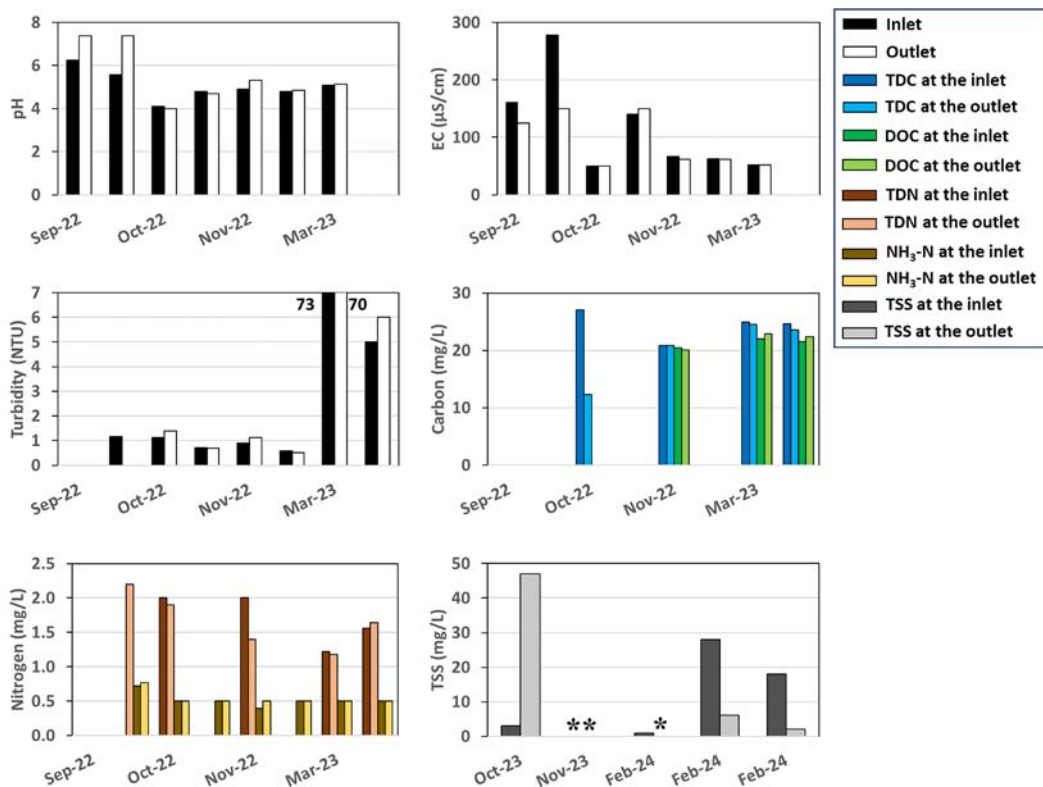


Figure 4.6. Water quality parameters at the pond inlet and outlet. Gaps indicate lack of data and asterisks denote 0 mg/L. NTU, nephelometric turbidity units. Source: adapted from Mendes *et al.* (2025).

The sedimentation pond is subject to subsurface inflows low in dissolved oxygen, which can worsen effluent water quality. Ultimately, the results highlight the need for alternative or additional mitigation measures on site that can safeguard downstream surface waters from harmful effluents.

#### 4.3.2 Field experiments using pipe-encased biochar

Recent studies have tested the potential of biochar in the treatment of effluents from degraded peatland sites (Lafdani *et al.*, 2020, 2021; Saarela *et al.*, 2020; Mosquera *et al.*, 2024). This is due to the adsorption properties of biochar, which can reduce the concentration of nutrients in organic and inorganic forms as water moves through the medium. Batch experiments demonstrated that nitrogen adsorption increased with concentration, with negligible desorption observed post adsorption (Saarela *et al.*, 2020; Lafdani *et al.*, 2021). Laboratory column experiments attested to the efficiency of biochar in reducing nitrogen concentrations when inlet concentrations were greater than 0.45 mg/L (Lafdani *et al.*, 2020). In field conditions, a pile of jute bags containing biochar and partially filling a drain reduced carbon and nitrogen concentrations (Mosquera *et al.*, 2024). Despite promising results, field experiments focusing on an appropriate design to provide optimised conditions are necessary to ensure the applicability

of biochar. We hypothesised that HRT is a critical factor for the performance of biochar and that marl may improve biochar adsorption conditions, thus enhancing treatment. Testing the first hypothesis involves evaluating the amount of biochar required for satisfactory treatment according to the flow dynamics of the catchment. Investigating the second hypothesis involved testing whether marl, a high pH (8.5) material commonly found beneath the peat substrate of raised bogs, combined with biochar improved the treatment performance by increasing the effluent pH as water flows through the medium. Since the pH of effluents from extracted peatlands is acidic, this second hypothesis is important and is supported by studies that show the influence of water pH on biochar adsorption properties (Yin *et al.*, 2017; Saarela *et al.*, 2020). Experiments were conducted at Corralanna (described earlier) and at All Saints (another extracted peatland site described in Chapter 5 and Table 5.1) bogs. The experimental design (Figure 4.7) is described in full in the end-of-project report and involved testing (i) different pipe volumes, (ii) different receiving flows to produce a variety of HRTs and (iii) different biochar particle sizes. Water was sampled at the inlet and outlet of the pipes once a week over 5 weeks in 2023 and 2024 to assess the treatment performance on pH, EC, TDC, DOC, TN, NO<sub>3</sub>-N, NH<sub>3</sub>-N and DON.

Overall, the experiment showed that pipe-encased biochar had an effect on water quality parameters, with



Figure 4.7. Experimental setups in (a) All Saints and (b) Corralanna, where effluent treatment in pipe-encased biochar was tested at the outlets of the catchments.

longer HRTs tending to decrease slightly DOC, TN,  $\text{NO}_3\text{-N}$ ,  $\text{NH}_3\text{-N}$  and DON (Figure 4.8). However, this was not significant, implying the need for even longer HRTs. Similarly, while smaller particle size tended to decrease DOC and  $\text{NH}_3\text{-N}$ , this was not significant (see the end-of-project report). The addition of marl did increase pH and TDC (as a result of increased dissolved inorganic carbon) but decreased DOC and DON, albeit not significantly (Figure 4.9). Increasing the amount of marl did not further impact water quality parameters.

#### 4.4 Implications for Peat Extraction and Best Practice Guidelines

Mendes *et al.* (2025) discussed in depth the implications of our findings, including (i) highly dynamic water quality in the effluents from extracted peatland catchments, (ii) year-round disturbance of the ecological status of downstream aquatic communities and (iii) the efficacy of removing TSS but the inadequacy of sedimentation ponds as a standalone mitigation measure in the treatment of these effluents.

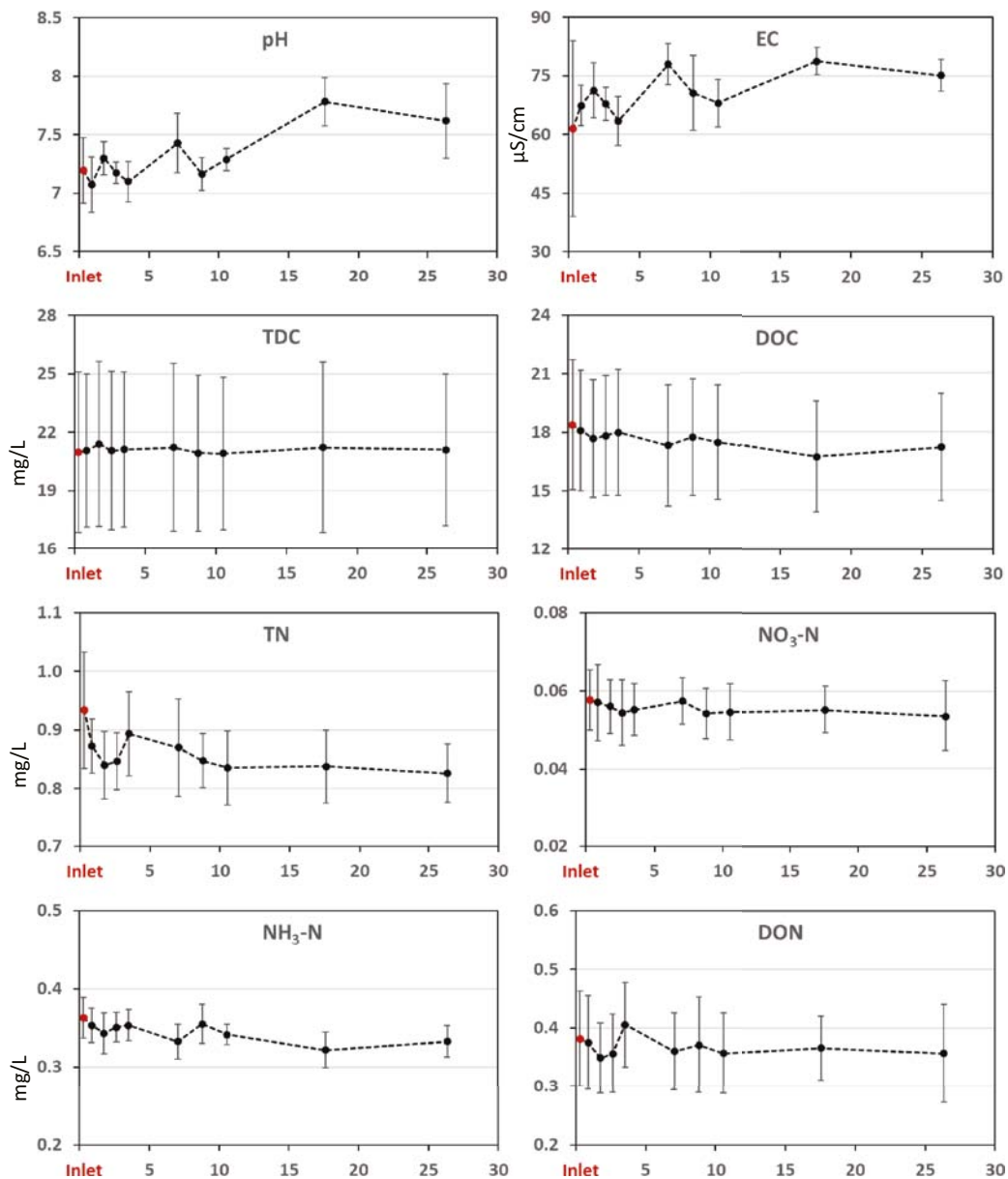
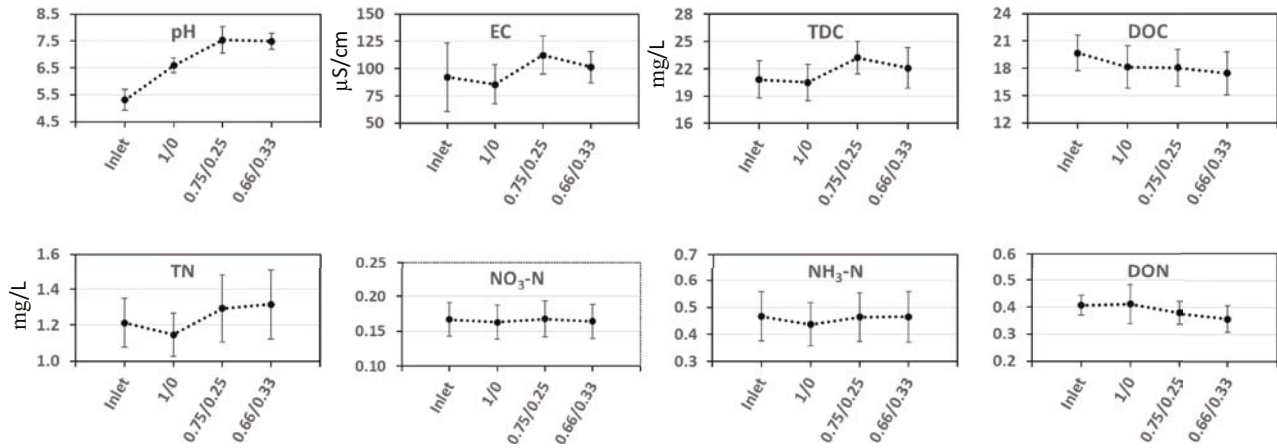


Figure 4.8. Means  $\pm$  standard deviations of water quality parameters at the inlet and outlet of pipes containing biochar according to the HRTs (minutes) shown on the x-axes.



**Figure 4.9. Means ± standard deviations of water quality parameters at the inlet and outlet of pipes containing biochar according to the volumetric ratio of biochar–marl shown on the x-axes.**

Effluents tend to be acidic in cold seasons, and nutrient and ion rich in warm seasons, thus leading to varied harmful effects on aquatic communities year round. In addition, nutrient exports are flow controlled. Hence, it is recommended that mitigation measures at the edge of extracted peatland catchments that treat soluble components before entering freshwater bodies are developed and implemented. The measures must

prioritise either a reduction in the concentration or the export of nutrients, as flow has opposite effects on concentration and export, and this may depend on the sensitivity of downstream aquatic communities.

To date, the application of biochar has had limited results. Further testing is recommended, particularly with longer HRTs and during the summer when nutrient concentrations and temperatures are higher.

# 5 Modelling Peatland Hydrology and Peatland Dynamics

As industrial peat extraction winds down under EU and national policies, cutaway bogs pose certain rewetting/restoration challenges (Gaffney *et al.*, 2020). This is because of complex non-linear feedbacks that control drainage legacy effects (e.g. shifting drainage configuration and water storage dynamics) and restoration trajectories over time (Gatis *et al.*, 2023). Numerical process-based and conceptual models provide explanatory or heuristic tools for discovering influential mechanisms driving complex behaviours in peatland systems (Jakeman *et al.*, 2006) and have been increasingly applied to predict exchanges of energy, water, carbon and nutrients in peatlands and land use change impacts (Mozafari *et al.*, 2023). Among the different types of models, climate or Earth system models represent global or regional precipitation and temperature but operate too coarsely to capture heterogeneity. Meanwhile, terrestrial ecosystem and land surface models focus on surface energy budgets, carbon pools, nutrient fluxes and coupled water-carbon cycles but simplify lateral water flows. Hydrological models concentrate specifically on water fluxes but lack ecological components (Mozafari *et al.*, 2023). Available numerical models often lack peatland-specific modules, and therefore they inadequately represent unique hydraulic properties of peat soils, such as pronounced non-linear hydraulic conductivity, preferential flows through cracks and pipes, and structural changes over time (Holden, 2005). In this chapter, we first synthesise evidence on drainage and restoration impacts on peatland flood regulation (Mozafari, 2024) before reviewing and appraising all process-based models previously applied to peatlands (Mozafari *et al.*, 2023). Further analysis has been carried out to test a multi-model rainfall-runoff ensemble technique using three raised bog datasets (Mozafari, 2024).

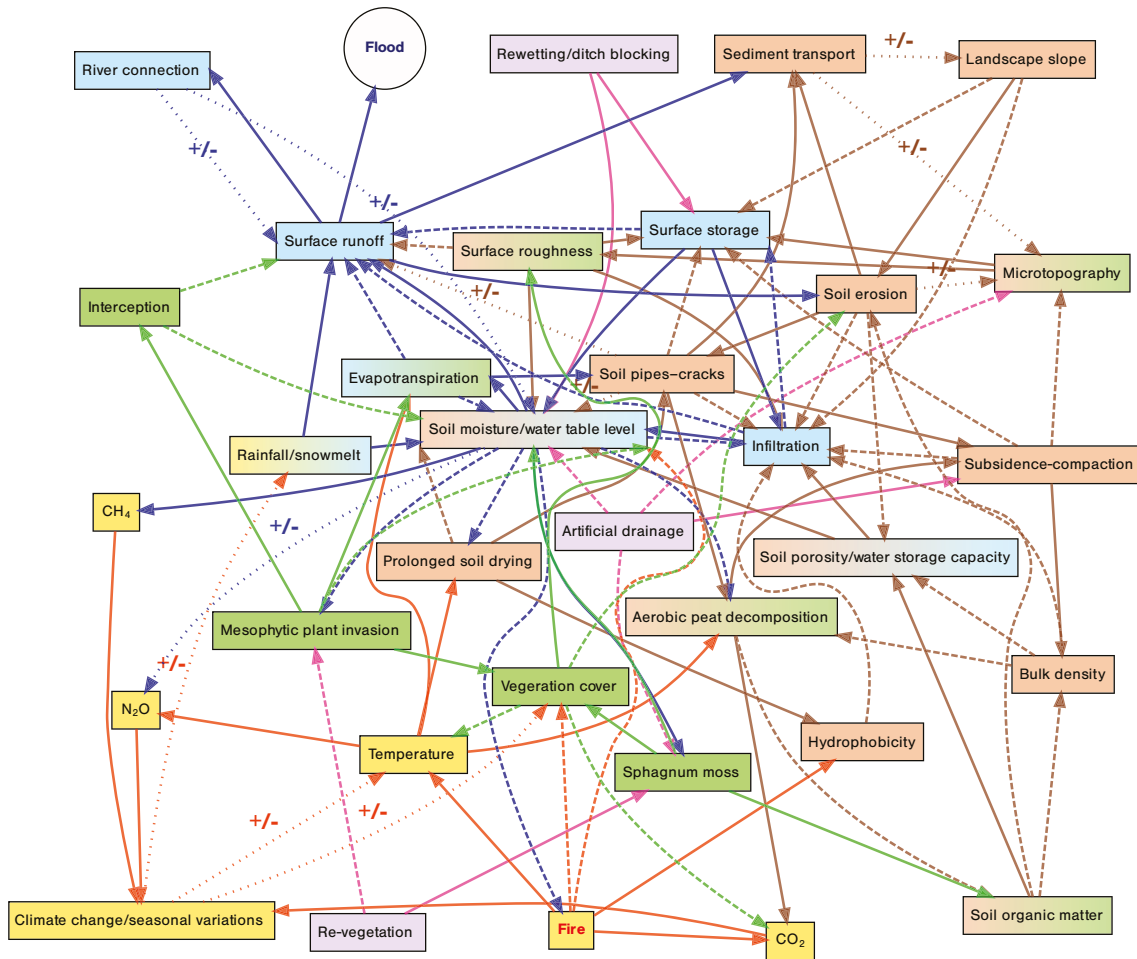
## 5.1 Conceptual Framework of Factors Influencing Peatland Runoff Dynamics and Flood Regulation

Peatlands provide essential hydrological regulation services through their high moisture storage

capacity and as transitional landscapes for aquatic environments. However, extensive artificial drainage has severely disrupted these functions across all peatland types. The complex interplay between climate, peatland topography, soils and vegetation make it difficult to predict hydrological alterations and flooding consequences from any intervention across peatlands worldwide. In particular, limited mechanistic understanding complicates process-based modelling regarding upscaling fine-scale changes to larger-scale flooding regimes (Holden, 2005). Furthermore, any gaps in conceptualising these factors can translate into difficulties in predicting other interrelated issues, such as water quality degradation, that may emerge from shifting drainage configurations.

A systematic review, following the PRISMA guidelines (Mozafari, 2024), identified key factors influencing runoff generation and flood regulation in peatlands globally. From the evidence compiled, a conceptual model (Figure 5.1) was developed to synthesise key factors and linkages.

The conceptual model (Figure 5.1) indicates that the master variables are soil moisture and water table depth. Climate factors, including precipitation (intensity, timing and phase) and temperature, influence antecedent moisture in peatlands, governing water inputs for storm runoff and flooding (Metcalfe and Buttle, 2001, Lane and Milledge, 2012; Menberu *et al.*, 2018; Laine *et al.*, 2019, Edokpa *et al.*, 2022; Karimi *et al.*, 2023). However, changes in vegetation composition can offset or reinforce the impacts over time through transpiration and interception feedbacks (Marttila and Kløve, 2009; Menberu *et al.*, 2017). Local topographic and landscape factors, including slope, surface complexity and landscape position, further modulate hydrological responses (Quinton *et al.*, 2003; Marttila *et al.*, 2010; Lane and Milledge, 2012; Edokpa *et al.*, 2022). Meanwhile, headwater bogs act as storage reservoirs, and floodplain fens have limited capacity for attenuation by routing flows downstream (Quinton *et al.*, 2003). Artificial drainage lowers water tables by enhancing subsurface flow losses (Kløve and Bengtsson, 1999; Moorhead, 2003; Harvey and McCormick, 2008) while also decreasing antecedent



**Figure 5.1. Diagram showing key processes and interactions governing peatland flood regulation, grouped into climate (yellow), soil (brown), vegetation (green), land management (purple) and hydrological response (blue) categories. CH<sub>4</sub>, methane; CO<sub>2</sub>, carbon dioxide; N<sub>2</sub>O, nitrous oxide.**

soil moisture, converting peatlands from surface saturation–excess runoff systems into ones dominated by infiltration–excess flows. This leads to increased aerobic peat decomposition, subsidence and altered runoff pathways, which reduce porosity, saturated hydraulic conductivity and water storage capacity (Billett *et al.*, 2011; Zhong *et al.*, 2020; Ikkala *et al.*, 2021). This contributes to increased surface runoff and peak flows during storms, reducing flood regulation capacity (Grayson *et al.*, 2010; Gatis *et al.*, 2023). Alterations to the physical structure of peat persist over time (Alderson *et al.*, 2019), creating hydrophobic, compacted layers (Shuttleworth *et al.*, 2019).

Drainage configuration strongly affects hydrological response, with closer ditch spacing causing greater water table drawdown (Hillman, 1992), which differs between events and seasons (Kløve and Bengtsson, 1999). Drier conditions increase bulk density and shrinkage cracks, enhancing preferential flow through

macropores and soil pipes, which in turn increase runoff peaks (Holden *et al.*, 2004; Holden, 2005; Regensburg *et al.*, 2021). Water table drawdown also increases fire risk, which leads to exposed bare peat, exacerbating peat loss, increasing runoff generation and causing further drying due to albedo effects and increased hydrophobicity (Kamrath *et al.*, 2020; Nelson *et al.*, 2021). A projected shift towards a warmer, drier climate with more sporadic intense rainfall events would directly reduce moisture retention in bogs while also promoting shrub expansion and loss of sphagnum cover through evaporative drying (Edokpa *et al.*, 2022; Karimi *et al.*, 2023). This would initiate a positive feedback mechanism, whereby lower water tables sustain further shrub encroachment, which in turn would continue to diminish infiltration and detention storage capacities. The combined climate and vegetation shifts could dramatically reduce natural flood mitigation capacities during subsequent

extreme rainfall events. Widespread surface erosion and loss of microtopography would directly accelerate overland flow velocities while also removing surface depressions for detention storage (Daniels *et al.*, 2008; Howson *et al.*, 2023). This highlights the importance of ensuring that peatland modelling research carefully considers drivers that climate change could intensify.

There is mixed evidence on whether or not restoration fully reverses drainage effects over the long term. Hysteretic subsidence and changes in soil structure can persist, creating an alternative stable regime (Ikkala *et al.*, 2021). Loss of mosses and herbaceous communities after drainage reduces surface roughness and exposes bare peat, which increases erosion risk and accelerates near-surface flows (Goudarzi *et al.*, 2021). Revegetation can help stabilise eroding surfaces and mitigate these impacts (Alderson *et al.*, 2019). Threshold behaviour in which initial drying triggers irreversible hydraulic changes may constrain restoration potential in some cases (Moorhead *et al.*, 2008; Johnson *et al.*, 2022).

## 5.2 Review of Process-based Models for Peatlands

To identify models that have the potential to be most useful to the modelling of Irish peatlands processes, a systematic review was conducted of the most widely used process-based models for simulating

ecohydrological interactions that are driven by the exchanges of energy and mass (e.g. water, carbon and nitrogen). The methodology is summarised in Figure 5.2, and further details and findings have been published in Mozafari *et al.* (2023). It was decided that any model considered must meet both FOSS (free and open-source software) principles, to ensure that all models had an open-source licence for the model's source code, and FAIR (findable, accessible, interoperable, reusable) principles (Hasselbring *et al.*, 2020; Hut *et al.*, 2022; Tucker *et al.*, 2022), which requires that data for all formal scholarly digital publishing, as well as the tools and algorithms generating the data (e.g. computer codes), be made available globally. A final list of 12 models that met the criteria was drawn up and used for further technical analysis (see Appendix 2, Table A2.1, for the spatial and temporal resolution of each model). The review highlighted the lack of systematic calibration, validation and intercomparisons across studies and serves as a useful guide for model selection in future peatland research that can benefit from shared databases and standardisation. Furthermore, each model granularity or modelling approach was appraised under each process category, namely energy balance, water or nitrogen cycles (see Appendix 2, Table A2.2), focusing on the commonalities and the challenges and opportunities for peatland dynamics research. We concluded that, given the compromises between the complexity and the limited availability of data or

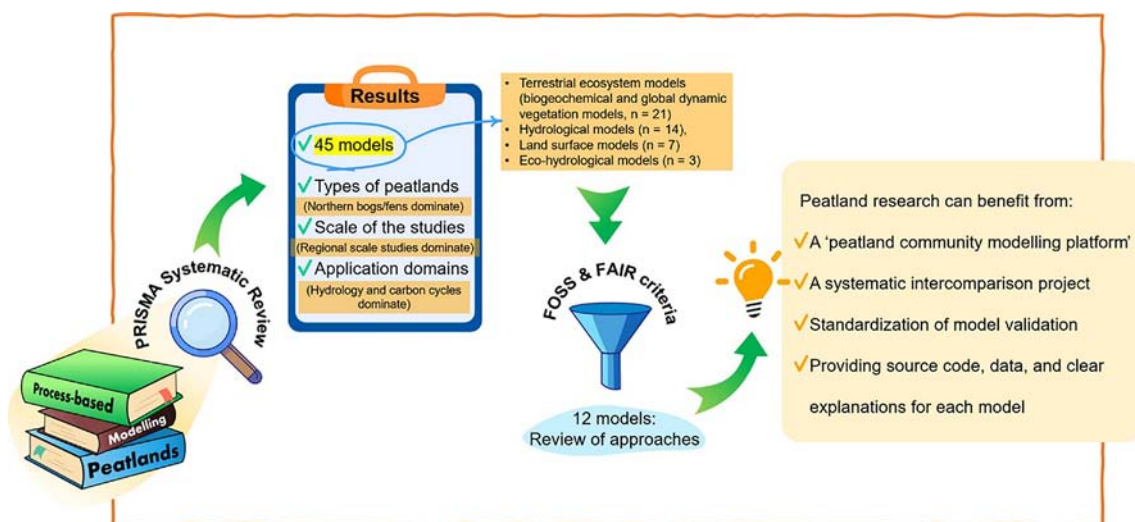


Figure 5.2. Graphical representation of methods and conclusions of the review of process-based models and approaches for simulating ecohydrological interactions in peatlands. Source: reproduced from Mozafari *et al.*, 2023; licensed under the Creative Commons Attribution 4.0 International licence (<https://creativecommons.org/licenses/by/4.0/>).

computing resources, a widely accepted one-size-fits-all model or approach is not available. This study streamlined the process of model selection and highlighted that standardisation and coordination are required for both data exchange and model calibration/validation to facilitate intercomparison.

### 5.3 Multi-model Ensemble Approach for Peatland Hydrological Modelling

Three peatlands located in the midlands (from east to west: Corralanna, All Saints and Carrownagappul) were used to explore the use of a parsimonious multi-model rainfall-runoff ensemble technique for hydrological modelling of Irish peatlands (see location in Figure 2.1 and key properties in Table 5.1). The study sites represent both drained (Corralanna, All Saints) and rewetted (Carrownagappul) systems, allowing for analysis across a gradient of hydrological conditions.

#### 5.3.1 Model selection

Conceptual lumped models were chosen over more complex process-based distributed models to explore

the use of ensembles for several reasons. First, detailed spatial data on soil properties, geology, topography and drainage networks were not available at sufficient resolutions. Second, the short record period of daily discharge would be challenging for parameterised process-based models (Mozafari *et al.*, 2023). Third, simple lumped models can achieve a performance similar to more complex models when predicting catchment outlet flows (Lobligeois *et al.*, 2014; Vansteenkiste *et al.*, 2014).

Ten lumped conceptual rainfall-runoff models with varying structures and complexity were selected from the Modular Assessment of Rainfall-Runoff Models Toolbox (MARRMoT) (Knoben *et al.*, 2019).

MARRMoT is an open-source toolbox that contains a collection of 47 interchangeable model components that represent typical hydrological processes and storage elements, including snowmelt, evapotranspiration, soil moisture accounting, runoff generation and routing. The 10 models for developing the multi-model ensemble were chosen based on (i) model performance, (ii) variation in model structures and complexities and (iii) model parsimony. The 10 selected models combine different

**Table 5.1. Summary of key properties and data for each site**

Property	Corralanna	All Saints	Carrownagappul
Location	Co. Westmeath	Co. Offaly	Co. Galway
Latitude	53.6988°N	53.1526°N	53.4992°N
Longitude	-7.4131°W	-7.9683°W	-8.4900°W
Area (ha)	16.4	55	89.4
Elevation (m.a.s.l.)	61	35	66
Climate	Temperate oceanic	Temperate oceanic	Temperate oceanic
Mean annual precipitation (mm)	941	846	1174
Mean annual temperature (°C)	9.3	9.8	9.3
Ecosystem Limitation Index	Energy limited	Energy limited	Energy limited
Water table depth (m)	1–1.2	N/A	N/A
Runoff ratio	0.68	0.45	0.92
Average runoff (mm/d)	1.8	1.1	3.7
Observed runoff data period	December 2020 to April 2023	January 2021 to January 2023	July 2019 to July 2020
Peat type	Moss peat over woody fen peat	N/A	N/A
Average peat depth (m)	>3	N/A	N/A
von Post Humification Index	H5 mossy/heather peat over H6 woody fen peat	N/A	N/A
Drainage status	Drained (active drainage network)	Drained	Rewetted
Historical use	Peat extraction for fuel – last harvest: 2021	Peat extraction for fuel	Peat extraction for fuel
Vegetation	Mostly bare peat, with some recolonised areas of heather, graminoids and mosses	N/A	N/A

m.a.s.l., metres above sea level; N/A, not available.

model components for soil moisture accounting, evapotranspiration, runoff production, percolation and routing processes. The main structural features and complexities of each model are summarised in Appendix 3.

### 5.3.2 Ensemble construction

Ensembles were constructed using 14 different integration methods with ensemble sizes of 3, 5, 7 and 10 models. The individual models were selected based on a multi-objective ranking approach across the

**Table 5.2. Ensemble models and their approaches tested in this study**

Ensemble method	Abbreviation	Description	Advantages	Limitations	Reference
Simple average	SA	Calculates unweighted arithmetic mean of model outputs	Simple, intuitive, benchmark for other methods	Weights all models equally, poor models degrade ensemble	Shamseldin <i>et al.</i> , 1997
Weighted average	WA	Weighted arithmetic mean based on model performance	Accounts for model skill, flexible weighting	Requires predetermined or optimised weights	Shamseldin <i>et al.</i> , 1997
Trimmed mean	TM	Removes highest and lowest predictions before averaging	Robust to outliers, tuned by trimming level	Loses information from trimmed models	Huisman <i>et al.</i> , 2009
Median	M	Takes median of model outputs	Robust, equal model weights, no training	Loses magnitude information	Huisman <i>et al.</i> , 2009
Bayesian model averaging	BMA	Probabilistic weighting based on model evidence and likelihood	Accounts for within- and between-model uncertainties	Computationally intensive	Neuman, 2003
Granger–Ramanathan averaging	GRA	Accounts for correlation between model errors	Data-driven model weighting based on correlations	Assumes linear error correlations	Granger and Ramanathan, 1984
Dynamic weighting	DW	Time-varying model weights based on current conditions	Adapts weights based on current conditions	Requires a calibration period	Li and Sankarasubramanian, 2012
Multiple linear regression	MLR	Linear combination of model outputs	Simple, interpretable, baseline for non-linear methods	Assumes global linear relationships	Huisman <i>et al.</i> , 2009
Long short-term memory	LSTM	Learns temporal relationships	Adaptive non-linear integration	Prone to overfitting	Kratzert <i>et al.</i> , 2019
Decision tree	DT	Partitions input space to predict ensemble output	Non-parametric, handles interactions	Prone to overfitting	Solomatine <i>et al.</i> , 2007
Random forest	RF	Ensemble of decision trees trained on bootstrapped samples	Overcomes single tree overfitting	Loses interpretability	Schoppa <i>et al.</i> , 2020
Gradient boosting	GB	Iteratively trains weak learners to reduce ensemble error	Powerful non-linear technique, regularised	Complex parameter tuning	Song <i>et al.</i> , 2022
K-nearest neighbours	KNN	Weighted local averaging based on historical analogues	Non-parametric, conceptually simple	Performance depends on distance metric	Kumar <i>et al.</i> , 2023
Support vector regression	SVR	Finds optimal hyper-plane with allowance for errors	Handles high-dimensional spaces, less overfitting	Computationally intensive	Adnan <i>et al.</i> , 2022

calibration period using principal component analysis to extract dominant modes of variability, analysis of variance to rank models based on their variance, and K-means clustering to group models based on hydrograph shapes and features. With 10 models in total, there were 492 possible unique ensembles of sizes 3, 5 and 7 that could be constructed (Kumar *et al.*, 2015). However, this multi-objective ranking approach avoided the need to construct and evaluate this number of combinations. By selecting the top models based on average rank, complementary ensembles of predetermined sizes were objectively compiled. A summary of the different ensemble models and their potential advantages and limitations is given in Table 5.2.

### 5.3.3 Runoff simulations: individual models

As is evident in Figure 5.3, Carrownagappul experienced flashier storm responses with rapid runoff peaks that were distinct in duration and magnitude between the first and second halves of the time period used for calibration. Accordingly, no model could accurately predict these large differences arising in the validation period. With only 6 months of daily discharge data available for calibration, model

parameters lacked predictive accuracy. However, the evolving rewetting conditions suggest that a fixed parameter set would be insufficient to capture the non-stationary response occurring. GR4J and Susannah Brook v2 achieved slightly higher performance during calibration and validation for All Saints and Carrownagappul. However, the increased complexity (parameters and stores) did not result in better discharge simulations. No single model emerged as an outright top performer across all three sites. This highlights the unique challenges posed by peatlands, where subtle differences in properties can lead to heterogeneous behaviours.

Examination of event hydrographs during the evaluation periods (Figure 5.4) shows that models struggle to capture peaks, particularly for All Saints and Carrownagappul. Low flows were generally overestimated across catchments, with All Saints exhibiting prolonged recession flows, likely due to extensive ditch drainage networks. Only GR4J reproduced these delayed flows. Meanwhile, flashy responses due to intense rainfall were better simulated by GR4J and the Wetland model. The overestimation of low flows implies that models assume that water is retained for too long following rainfall events.

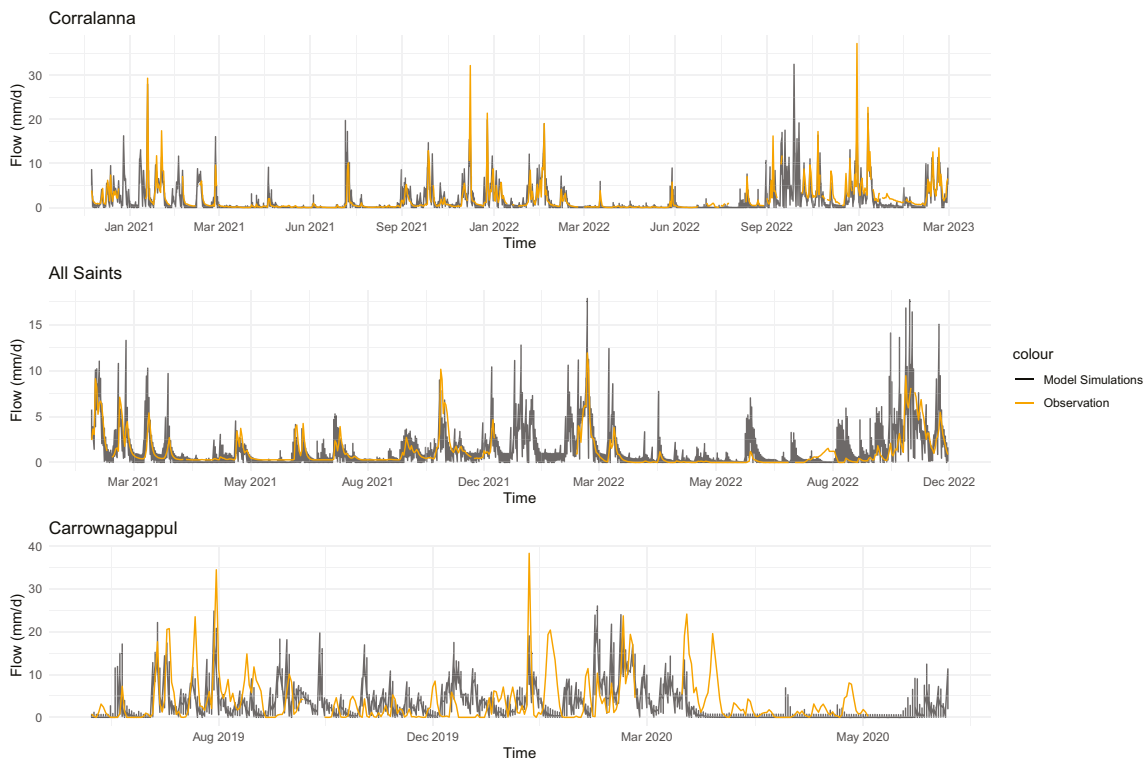


Figure 5.3. Individual model results for runoff simulations versus observations at each site.

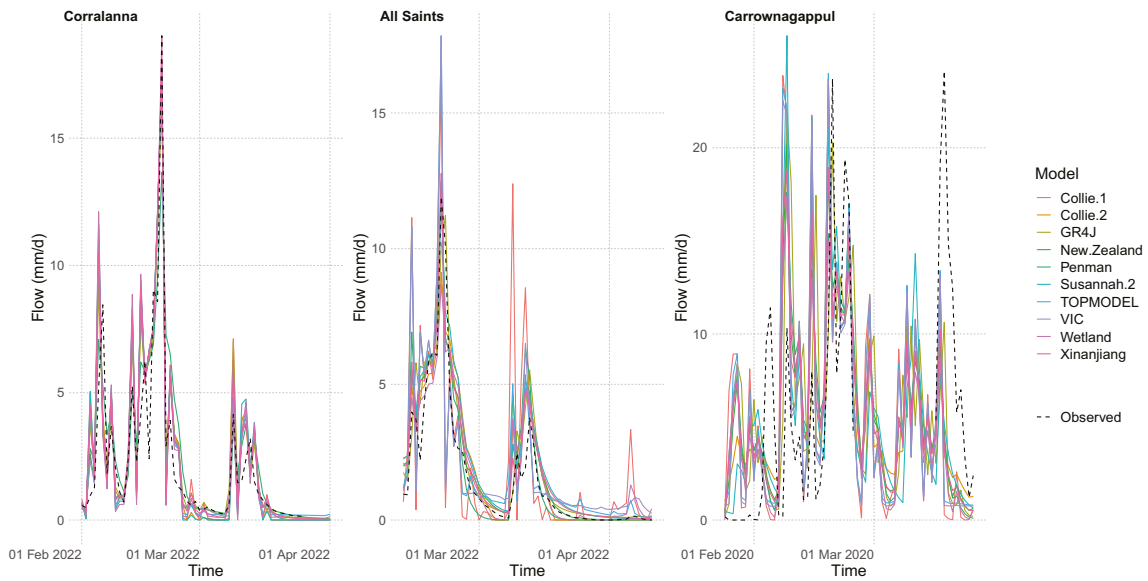


Figure 5.4. Examples of event-based hydrographs during the evaluation period at each site.

### 5.3.4 Runoff simulations: multi-model ensemble

The ensemble modelling results were condensed for visual interpretation in spider plots showing the distribution of Kling–Gupta and Nash–Sutcliffe efficiency scores (i.e. metrics used in hydrology to summarise model performance) for different integration techniques, catchments and ensemble

sizes (Figures 5.5 and 5.6). The results for Carrownagappul show that most techniques did not improve simulations compared with the top individual models. At All Saints, the size 3 GRA, MLR and BMA ensembles achieved the best scores. At Corralanna, several ensembles achieved significant improvements over the top individual model, GR4J. However, many ensemble configurations provided

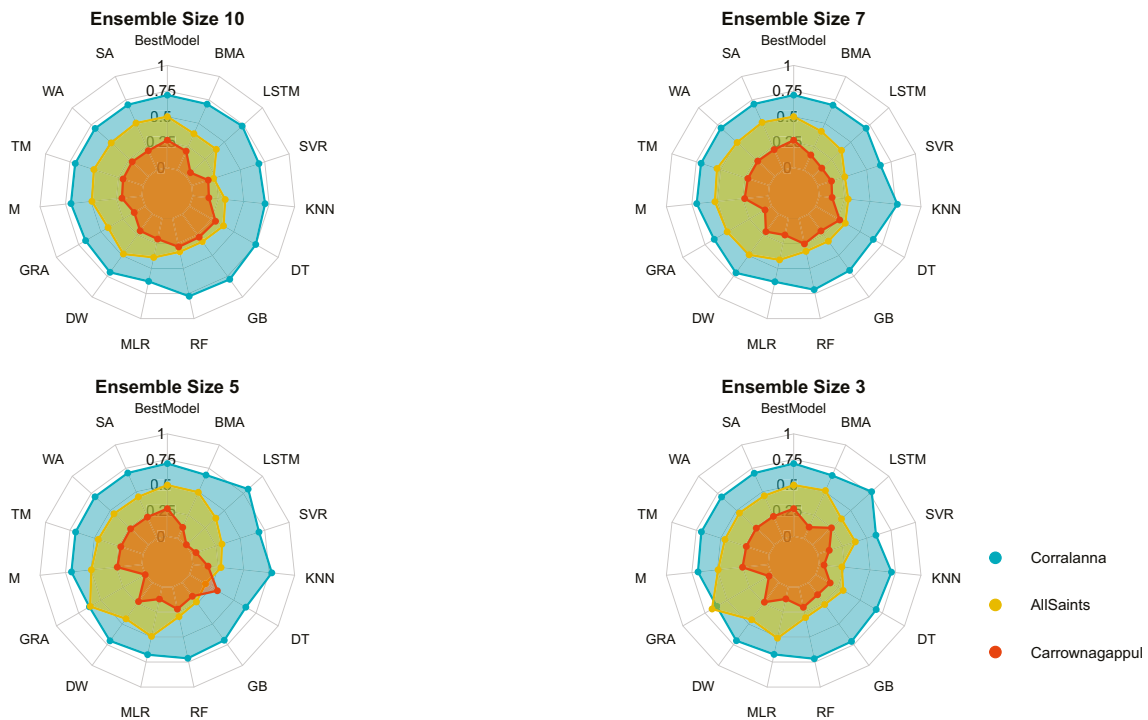
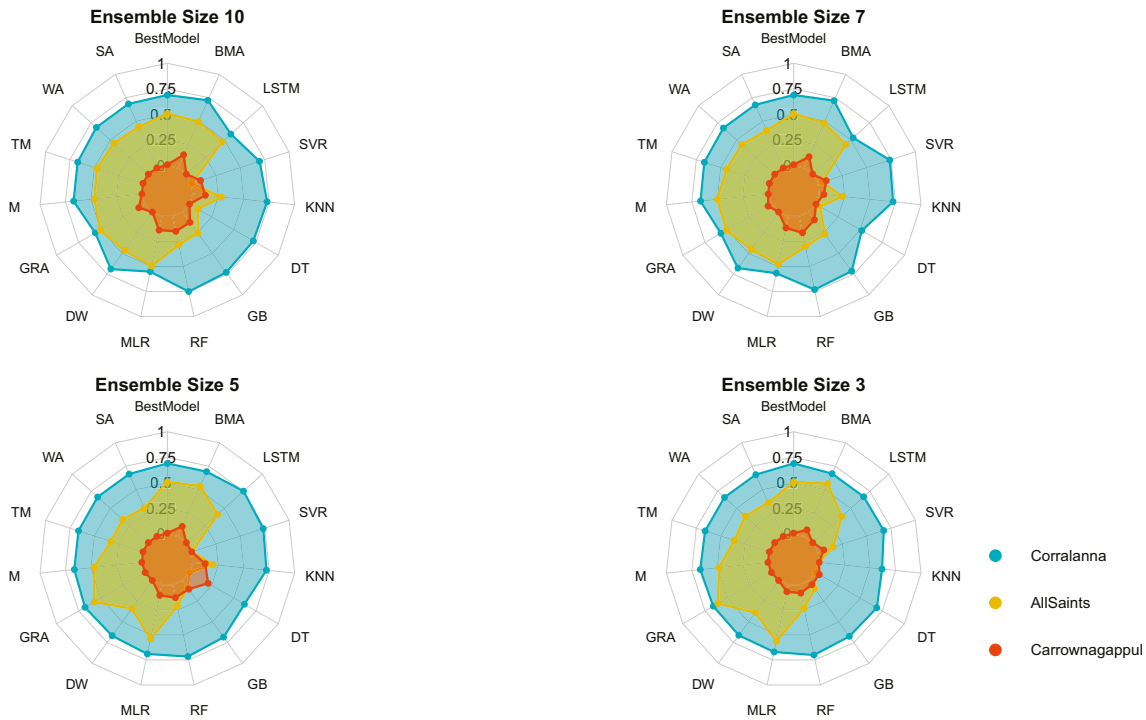


Figure 5.5. Kling–Gupta efficiency scores for different ensemble integration techniques on different ensemble sizes (1 being perfect simulation).



**Figure 5.6. Nash–Sutcliffe efficiency scores for different ensemble integration techniques on different ensemble sizes.**

negligible accuracy gains over GR4J, suggesting that there was not sufficient difference in model outputs to improve the results. In general, no single integration approach emerged as universally optimal across the diverse sites. The performances appeared sensitive to localised calibration data characteristics and runoff mechanisms. This highlights the need for adaptive configurations tailored to catchment behaviours rather than assuming generic relationships. We conclude that there is currently no widely accepted one-size-fits-all model or approach, given the compromises needed between the complexity of the model and the limited availability of data or computing resources.

## 5.4 Conclusions and Recommendations

### 5.4.1 Need for a community modelling platform

The review of current models for and approaches to modelling peatland dynamics highlights the need for a “community modelling platform” to reduce the proliferation of duplicative and redundant models. This approach offers modelling communities the opportunity to collaborate more efficiently and consolidate knowledge gleaned from various models.

The applicability of a community modelling platform for peatlands has not been well investigated, despite peatlands being widely simulated with “non-peatland-specific”, and in some cases community-based (e.g. CLM) and process-based, models.

### 5.4.2 Multi-model ensembles

No single parsimonious lumped model consistently achieved satisfactory performance across the diverse peatland catchments. Model structural inadequacies became more apparent under wetter regimes, with abundant peaks. The analysis of the multi-model ensemble of runoff simulation models reveals several key findings. First, combining models improved performance compared with individual models when simulation periods were sufficiently long. Second, the benefit of using the multi-model ensemble depended on model diversity (Huisman *et al.*, 2009; Velázquez *et al.*, 2011). The 10 diverse models used provided complementary behaviours to boost accuracy over most of the single models. Third, declining validation performance highlights transferability issues when clear signatures are unavailable to constrain model parameters (Kay *et al.*, 2006). While the ensemble could compensate for structural errors, it could

not overcome input data uncertainties from spatial transfers. No single technique performed best across all sites. Instead, adaptive optimisation was required based on catchment attributes and available data. Fourth, larger ensemble sizes did not reliably produce better performance; small ensembles, around five

members, tended to provide the optimal predictability across sites before the gains plateaued. However, this was not uniform, and some complex regimes like Corralanna benefited from a larger random forest ensemble to extract more complementary features.

# 6 General Conclusions and Recommendations for Policymakers

## 6.1 Key Observations

**Observation 1.** We presented evidence in Chapter 2 that bog streams within a degraded bog landscape exhibited greater levels of pollutants, in particular TDN and sulfate, as well as higher EC and lower  $SUVA_{254}$  values. This has an influence on downstream water quality and associated biodiversity, as well as future outcomes of bog restoration plans. The environmental impact of peat extraction is clearly not sufficiently mitigated by current control methods, and adequate site-level water treatment must be implemented to ensure that water quality standards are met.

**Observation 2.** While the water quality of streams that receive bog waters did not differ significantly between natural and degraded sites, high concentrations of nitrogen species and DOC within all the receiving streams (upstream and downstream from degraded and restored bogs) demonstrate the complexity of these bog landscapes that have undergone widespread disturbance over space and time. We conclude that this bog region is experiencing long-term widespread loss of fluvial nitrogen and carbon into the streams, evidence for the general poor health of all Irish bogs regardless of status. It complements existing evidence of the widespread drying of bogs (Swindles *et al.*, 2019) and additional pressures from, for example, nitrogen deposition (Kelleghan *et al.*, 2021).

**Observation 3.** Nitrogen pollutants in receiving streams were attributed specifically to bog streams only at specific sites and were significantly over the threshold limits for pollutants as currently regulated. These hotspot sites must be treated to remove this pollution source. While the *EPA BATNEEC Guidance Note Class 1.4 Extraction of Peat* (EPA, 1996), updated in 2006, currently sets emission limit values for a small number of pollutants in drainage water from extracted peatlands, the current level of pollution encountered at some extracted sites is not acceptable and must be immediately addressed by more robust

control methods at the site level and by monitoring of the impacts at the catchment level.

**Observation 4.** The negative impact of extracted peatlands was further evidenced in Chapter 3 through the investigation of aquatic biota. The degraded bog streams showed faunal profiles that indicate the presence of pollutants. The natural bog streams also exhibited distinct profiles that demonstrate the long-term impact of historical drainage in the region (e.g. stony- vs peat-bottomed streams). This is important for future bog restoration management plans, as reference conditions will vary between sites.

**Observation 5.** Degraded water quality in the overall region is likely to impact other aquatic ecosystem services, including drinking water and biodiversity. This was confirmed by the range of aquatic biota found in upstream water bodies (regardless of bog status) and points to widespread multi-source pollution. Therefore, management of peat soils in this region must be considered at the landscape level to ensure that water quality challenges are addressed.

**Observation 6.** In Chapter 4, we presented more detailed evidence from a peatland extracted for horticultural peat. The study found that the water quality of the effluents was very dynamic but tended to be acidic in cold seasons and nutrient and ion rich in warm seasons, thus leading to varied harmful effects on aquatic communities year round.

**Observation 7.** Data have shown that sediments are successfully retained in the sedimentation ponds (even during high rainfall events) but that these ponds fail as a standalone mitigation measure for soluble pollutants and low pH levels in the effluents.

**Observation 8.** Variation in nutrient concentrations was great between storm events, and strong seasonal patterns were observed. DOC and other nutrient concentrations were dependent on changes in stream flow (low during high-flow events). Warm periods with little rainfall appear to neutralise the water pH but increase the concentration of nutrients, especially carbon and ions, whereas cold and rainy

periods appear to acidify the water pH but decrease the concentration of nutrients, especially carbon and ions. Ultimately, water pH, NO<sub>3</sub>-N and NH<sub>3</sub>-N often exceeded the environmental quality standards (pH range 4.5–9.0, 1.8 mg NO<sub>3</sub>-N/L and 0.065 mg NH<sub>3</sub>-N/L) for good ecological status of surface waters.

**Observation 9.** Total annual exports were estimated based on periodic sampling during storm events, for DOC 148 kg C/ha/y and for TDN 15.7 kg N/ha/y. These fluvial exports are similar to those of other bare peat bogs, with relatively low DOC exports caused by the presence of nutrient-poor peat, high EC and/or high sulfate concentrations, all of which reduce the solubility of DOC.

**Observation 10.** Experiments conducted to date with pipe-encased biochar indicate that the impact of biochar was limited. Research is ongoing to investigate potential factors, such as longer HRTs and warmer temperatures.

**Observation 11.** While process-based models have been widely applied to study peatland hydrological dynamics, the plethora of models in the literature makes it difficult to identify the models that are “fit for purpose”. Our analysis concluded that, currently, given the compromises needed between the model complexity and the limited availability of data or computing resources, a widely accepted one-size-fits-all model or approach does not exist. The model descriptions in this report (see section 5.2) may serve as a useful guide for model selection in future peatland studies.

**Observation 12.** No single hydrological model emerged as the best option across the three peatland catchments we studied. Model structural inadequacies became more apparent under wetter regimes, with abundant peaks observed. Parsimonious models provided adequate accuracy in representing water balance but limited accuracy in representing intense event dynamics.

## 6.2 Key Recommendations

To assist in meeting water quality standards in the region and improve our understanding of peatland hydrology, we offer the following recommendations.

**Recommendation 1.** At the site level, additional site-specific water treatment at extracted bogs is

required to decrease soluble pollutants and manage pH levels. We recommend the development and implementation of mitigation measures at the edges of extracted peatlands to treat soluble components. These measures should prioritise the reduction of either nutrient concentrations or nutrient exports, as flow has been shown to have different effects on concentration and export. This approach may need to be tailored based on the sensitivity of downstream aquatic communities to threshold levels.

**Recommendation 2.** Stricter water quality controls should be included and regulated as part of the licensing of peat extraction activities. The water protection methods must be clearly validated at each specific environmental location in combination with specific regional objectives. The monitoring of data must also take cognisance of seasonal effects and additional site-specific factors, which may influence “key” nutrient species. In addition, action must be taken to address water pollution from the activities of unlicensed small private peat companies, as well as from bogs affected by domestic turf-cutting, which impact an even greater land surface area.

**Recommendation 3.** At the landscape level, large-scale rewetting of extracted bogs must be accompanied by long-term monitoring of the chemistry of the bog water and receiving streams. Reducing pollution from drained bogs will be largely addressed by restoration and rehabilitation projects that affect not only Bord na Móna bogs but also privately owned bogs within a catchment.

**Recommendation 4.** The full impacts of continued and new peat extraction must be acknowledged and properly assessed and more robust mitigation measures implemented. Revised mitigation measures to treat effluent waters from not only extracted bogs but also all sites that include drainage of peat soils (land use change, wind farms) must be implemented. The long-term water quality impacts of all projects that affect peatlands must be measured more regularly and for longer periods, while also following strict laboratory protocols (due to the difficulties associated with analysis of bog water).

**Recommendation 5.** While our results show large spatial and temporal variations associated with the wide range of site characterisations, they also provide baseline water quality values in the landscape, which is critical for future restoration work. Water polluted

with nutrients released from long-term degradation or other pressures (agricultural) may combine with the neutral, even alkaline, pH levels present in certain drainage waters (due to alkaline geology and the widespread presence of marl-bottom peat in this region), which provide optimal conditions for further peat mineralisation, leading to challenges for bog rewetting/restoration projects. In short, the quality of the water will affect the outcomes of peatland rewetting projects from the start and therefore require careful year-round monitoring. Investigation into groundwater within the wider landscape is also recommended along with its explicit inclusion in hydrological modelling.

**Recommendation 6.** While terrestrial biota of most protected bogs is carefully monitored in terms of habitat integrity (NPWS, 2017), there is often a lack of information with regard to the aquatic environment associated with these ecosystems. Targets are often only concerned with terrestrial criteria (e.g. vegetation). Long-term monitoring of the chemical composition of bog water and associated fauna is critical for a full understanding of the impacts of large-scale efforts to rewet/restore peatlands and for the development of robust ecohydrological targets for both public and private projects.

**Recommendation 7.** EC, DOC and TDN were identified as important indicators for evaluating current and future peatland management activities in the region, especially rewetting/restoration efforts. EC measurement is inexpensive and rapid, and thus widespread and routine monitoring could be deployed around bog sites to ascertain the status of the site and also monitor the effects of management actions (e.g. rewetting private bogs). This study recommends more specific investigations into the various drivers that affect water composition, and therefore more efforts should be deployed to continuously monitor DOC and TDN, in particular, to help develop more robust models for predicting their concentrations.

**Recommendation 8.** Seasonal sampling may also impact water quality from these ecosystems and should be carefully addressed in any monitoring programme. For example, our results indicate greater concentrations of various nitrogen species in the spring, as well as lower  $SUVA_{254}$  values (i.e. less labile dissolved organic matter, which is more difficult to treat). This is also correlated with high iron concentrations in the spring, which is not bound

to the higher aromatic hydrophilic natural organic matter. Therefore, seasonal monitoring and spring measurements, in particular, may be important in developing end-of-pipe treatment solutions in this region.

**Recommendation 9.** The poorer faunal profiles of degraded bog streams reflects the significant pressures linked to drained peatlands. Natural bog streams also exhibited distinct profiles, highlighting the long-term impact of historical management. Acknowledging this variability is crucial for future management plans, as reference conditions will differ. Consequently, new metrics must be developed to account for this heterogeneity in streams within the bog landscape.

**Recommendation 10.** Our results suggest that more efficient and varied treatment processes are needed to help improve and future-proof water quality from peaty catchments. Rewetting of all peat extracted sites is necessary to reduce the high nitrogen and carbon losses in freshwaters.

**Recommendation 11.** The failure of the mitigation measures we trialled to improve water quality at the site level highlights the need for continued research on a wider range of options.

**Recommendation 12.** The review of current hydrological models and approaches to modelling peatland surface runoff dynamics highlighted the need for a “community modelling platform” to reduce the proliferation of duplicative and redundant models, thereby offering the opportunity for efficient multidisciplinary collaboration in future peatland research.

**Recommendation 13.** The analysis of the multi-model ensemble for modelling peatland runoff dynamics revealed the need for adaptive configurations tailored to catchment behaviours rather than assuming generic relationships in modelling peatland hydrological responses. In this study we evidenced the unique challenges posed by peatlands, where subtle differences in properties can lead to heterogeneous behaviours, and we therefore recommend an “each site is unique” approach for future studies. Furthermore, as the lumped models considered in this study were unsuited to the complex hydrology of drained and rewetted peatlands, further research

is necessary, especially under changing climatic conditions.

## 6.3 Implications for Policy Decisions, Regulation and Future Research

### 6.3.1 Sustainable management of peatlands

There is a global recognition of the value of managing peatlands sustainably, specifically seen in the implementation of many international agendas that include sustainable development (e.g. United Nations Framework Convention on Climate Change, Convention on Biological Diversity, Ramsar Convention, Sendai Framework for Disaster Risk Reduction 2015–2030) and more targeted resolutions (e.g. EU climate regulations (EU, 2018)). There is increasing evidence that peatland rewetting/restoration can be a tool for meeting various regulatory targets, especially for climate and biodiversity (Renou-Wilson *et al.*, 2019; Wilson *et al.*, 2022), but also in relation to the Water Framework Directive and its target to achieve “at least good status” in European water bodies (Martin-Ortega *et al.*, 2014), as well as reducing the need for drinking water treatments (O’Driscoll *et al.*, 2018; Flynn *et al.*, 2021). Globally, Ireland has experienced the greatest level of wetland loss (mostly peatlands) since the 1700s (Fluet-Chouinard *et al.*, 2023), and the reversal of centuries of unsustainable peatland management will require considerable efforts from all stakeholders. At the national level, the utilisation of the peat soils resource must be integrated into cross-governmental policies to harmonise the sustainable management of this large but diminishing resource. A holistic view of the management of peatlands is required, reflecting the importance of peatland functioning within a regional ecosystem. Therefore, the focus should shift from meeting single policy targets, e.g. habitat protection, and move towards integrating all ecosystem services, including water quality and supply and climate change mitigation and adaptation. This would then require better cross-departmental communication to identify a wider range of targets associated with peatland management.

### 6.3.2 Peat extraction regulations

The full impacts of continued peat extraction must be acknowledged and properly assessed ahead of proposed land use, energy and horticultural projects

that all require drainage. If licensing activities are sought for continued peat extraction in Ireland, stricter regulations and associated monitoring must be implemented, updating licence thresholds to reflect regional needs. This would require new regulation to update the EPA BATNEEC guidance for peat extraction (EPA, 1996). The water protection methods must be clearly validated in each specific environmental location in combination with specific regional objectives and additional methods tested to ensure improvement, along with seasonal, long-term monitoring. This regulation must be connected with the new guidance for the preparation of bog rehabilitation plans with licensees (EPA, 2020c). While currently non-binding, it calls for (i) full characterisation of the bog, (ii) extensive consultations with all stakeholders, (iii) a test programme with list of criteria and (iv) an aftercare maintenance programme, before final surrender of the licence.

Our study provides evidence for the need for long-term monitoring of water quality at all sites.

The application of compulsory water protection guidelines by peat extraction companies varies widely across the globe, and their efficiency has been widely questioned given the exceedances of levels of pollutants and the lower fish abundance in affected streams (Pavey *et al.*, 2007; Clément *et al.*, 2009). Several Fenno-Scandinavian studies point to greater success of mitigation measures for reducing nutrient and sediment loads, such as overland flow areas (that purify the water by means of natural biophysical and chemical processes), when used in conjunction with other water pollution control methods, such as improved peat lifting practices, field ditch retainers and sedimentation basins (Heikkinen *et al.*, 1995; Huttunen *et al.*, 1996; Marttila and Kløve, 2009).

### 6.3.3 Monitoring of peatland rewetting/restoration

Beyond the need for prompt water quality remediation, this study stresses the importance of rewetting extracted peatlands within sensitive catchments, especially to reduce aerobic conditions. Restoring wetness can suppress decomposition and facilitate vegetation re-establishment to restart carbon accumulation and restore water ecosystem services (Bonn *et al.*, 2016). While water quality improvements have been reported in other rewetted/restored

peatland studies (Evans *et al.*, 2015; Lundin *et al.*, 2016), the current monitoring associated with rewetted/restored peatlands in Ireland has mainly focused on water table levels and greenhouse gas fluxes. Such monitoring must expand to include the monitoring of water quality at the site and regional levels in order to help set optimal targets for rewetting/restoration and subsequent maintenance programmes. To this effect, peatland studies must be financially supported to monitor the effect of management plans over a longer period. Restoration methods have only recently (<10 years) been intensively deployed on the network of Irish SAC raised bog sites. It is expected that the benefits of restoration for water quality will become evident only in the medium term (Menberu *et al.*, 2017). While it was expected that restoration/rewetting would reduce DOC concentrations in catchments (Evans *et al.*, 2015), high concentrations within all the receiving streams (upstream and downstream from degraded and restored bogs) demonstrated the complexity of this landscape, which has suffered widespread disturbance over space and time. It may be several decades before the effects of bog restoration have any beneficial impact on the region's water quality.

#### **6.3.4 Future research**

Overall, the integrated review and analyses carried out in this project provide both broad syntheses and targeted technical details on pressing knowledge gaps that currently limit sustainable peatland management. They outline clear monitoring, experimentation and modelling roadmaps to address complex ecohydrological interactions. The observed non-linear behaviours demand interdisciplinary perspectives that encompass disturbance regimes when predicting restoration outcomes and outcomes for associated ecosystem services. Water quality data from peatland catchments that span multiple hydrological years are essential to fully understand the impacts of peatland degradation on water quality and subsequent restoration projects, as well as to clearly inform policy. In addition, the aromatic and coloured nature of bog water complicates laboratory analysis, rendering some commonly used methods (e.g. colorimetric analysis) impractical. These issues and limitations should be considered in the implementation of future research and monitoring programmes, highlighting too the need to scrutinise management options under future climate change scenarios. Lastly, it is important to highlight that the successful implementation and long-term sustainability of these solutions depend significantly on support from funding schemes (both national and international) that prioritise flexible approaches and encourage cross-disciplinary collaboration.

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# Appendix 1

**Table A1.1. Summary of the bogs study sites (see Figure 2.1 for a map) and their status (SAC), together with stream type sampled, their code names, order, Water Framework Directive (WFD) subcatchment area and subcatchment/catchment name**

Bog site	Site status (year of restoration)	River name, order and WFD subcatchment area in km <sup>2</sup>	Biota study	River bottom type	Subcatchment and catchment name	Stream type sampled		
						Within-bog small stream	Upstream receiving stream	Downstream receiving stream
Ardagullion (AG), Co. Longford	Natural/restored SAC (2018)	Camlin 26, IE_SH_26C010050, order 1, 28		Soft peaty	Camlin_SC_010, 26C Upper Shannon			✓
Bellanagare (BEL), Co. Roscommon	Natural/restored SAC (2020)	Francis, IE_SH_26F050050, order 1, 24 IE_SH_26T030300, order 2, 24	✓	Cobble/gravel	Suck_SC_010, 26D Upper Shannon	✓		
Carrowbehy (CB), Co. Roscommon	Natural/restored SAC (2019)	Trk_of_D, IE_SH_26C020100, order 1, 19 Cloonfower stream, IE_SH_26C210200, order 2, 19 Gortaganwy stream, IE_SH_26C210200, order 1, 19	✓	Cobble/gravel	Breedoge_SC_010, 26B Upper Shannon	✓		
Carrowbegapull (CG), Co. Galway	Natural/restored SAC (1990 onwards and dams in 2019)	Gunnode stream, IE_SH_26S030100, order 2, 34 Cloonconore stream, IE_SH_26S030100, order 1, 34	✓	Soft peaty	Suck_SC_010, 26D Upper Shannon	✓		✓
Crosswood (CW), Co. Westmeath	Natural/restored SAC (2005)	IE_SH_26S021800, order 1, 109	✓	Soft peaty	Shannon [Lower]_SC_010, 26G Lower Shannon	✓		
Ferbane (FER), Co. Offaly	Natural/restored SAC (2019)	Holy Well Clongawny, IE_SH_25B270110, order 1, 32		Cobble/gravel	Shannon [Lower]_SC_030, 25B Lower Shannon			✓
Garriskill (GK), Co. Westmeath	Natural/restored SAC (2018)	IE_SH_26R030100, order 1, 41 Riffey, IE_SH_26R030100, order 3, 41	✓	Cobble/gravel	Inny [Shannon]_SC_051, 26F Upper Shannon		✓	
			✓	Soft peaty	Inny [Shannon]_SC_051, 26F Upper Shannon			✓

**Table A1.1. Continued**

Bog site	Site status (year of restoration)	River name, order and WFD subcatchment area in km <sup>2</sup>	Biota study	River bottom type	Subcatchment and catchment name	Stream type sampled		
						Within-bog small stream	Upstream receiving stream	Downstream receiving stream
Killyconny (KC), Co. Cavan	Natural/restored SAC (2006–2009)	Feegat stream, IE_EA_07M030700, order 1, 21		Soft peaty	Moynalty_SC_010, 07 Boyne			✓
Mount Hevey (MH), Co. Westmeath	Natural/restored SAC (2005)	Ballyhaw, IE_EA_07B340940, order 1, 18 Aghamore IE_EA_07K010200, order 1, 17	✓	Soft peaty Cobble/gravel	Boyne_SC_040, 07 Boyne Boyne_SC_030, 07 Boyne	✓		
Moyclare (MOY), Co. Offaly	Natural/restored SAC (1995–2005 and again in 2019)	Holy Well Clonbonniff, IE_SH_25B270200, order 1, 57.5 Lisdaly stream, IE_SH_25B270200, order 1, 57.5 Moyclare stream, IE_SH_25B270200, order 2, 57.5	✓ ✓ ✓	Soft peaty Cobble/gravel Cobble/gravel	Shannon [Lower]_SC_030, 25B Lower Shannon Shannon [Lower]_SC_030, 25B Lower Shannon Shannon [Lower]_SC_030, 25B Lower Shannon	✓ ✓ ✓		✓
Raheenmore (RH), Co. Offaly	Natural/restored SAC (1990 and again in 2020)	Bunsallagh, IE_EA_07Y020070, order 1, 29 Puttaghan 25 IE_SH_25S030100, order 1, 23	✓	Soft peaty Cobble gravel	Yellow [Castlejordan]_SC_010, 07 Boyne Silver[Tullamore]_SC_010	✓ ✓		✓
All Saints bog (ASB), Co. Offaly <sup>a</sup>	Degraded/peat extraction	Rapemills, IE_SH_25R010500, order 2, 36	✓	Cobble gravel upstream Soft peaty downstream	Shannon [lower] 25B, 25B Lower Shannon	✓		✓
Ballybeg/Togher (BB), Co. Westmeath	Degraded/peat extraction	Coolcor stream, IE_EA_07C080190, order 1 upstream and 2 downstream Derryron, IE_EA_07Y020100, order 1, 13	✓ ✓	Cobble gravel upstream Soft peaty downstream Soft peaty	Yellow [Castlejordan]_SC_010, 07 Boyne Yellow [Castlejordan]_SC_010, 07 Boyne	✓ ✓		✓
Ballydaly (BD), Co. Offaly	Degraded/peat extraction	Kilcolgan beg, IE_SH_25B090800, order 1, 15	✓	Soft peaty	BROSNA_SC_060, 25A Lower Shannon		✓	

**Table A1.1. Continued**

Bog site	Site status (year of restoration)	River name, order and WFD subcatchment area in km <sup>2</sup>	Biota study	River bottom type	Subcatchment and catchment name	Stream type sampled		
						Within-bog small stream	Upstream receiving stream	Downstream receiving stream
Ballynaskeagh (BK), Co. Westmeath	Degraded/peat extraction	Graffashtown, IE_EA_07D010200, order 1, 25	✓	Soft peaty	Deel [Rahamey]_SC_010, 07 Boyne	✓		
Coolcraff (CC), Co. Westmeath	Degraded/peat extraction	Inny River, IE_SH_26I010700, order 4, 70	✓	Cobble/gravel upstream Soft peaty downstream	Inny [Shannon] 26I01, 26E Upper Shannon	✓	✓	✓
Corralanna (CL), Co. Westmeath	Degraded/peat extraction	Ferskill, IE_SH_26F370890, order 3, 26	✓	Cobble/gravel upstream Soft peaty downstream	Inny [Shannon]_SC_019, 26F Upper Shannon	✓	✓	✓
		Coolnagun stream, IE_SH_26C080860, order 1 upstream, order 2 downstream, 15		Soft peaty upstream and downstream	Inny [Shannon]_SC_020, 26F Upper Shannon	✓	✓	✓
Derrybrat (DER), Co. Offaly	Degraded/peat extraction	Silver, IE_SH_25S020701, order 4, 16	✓	Soft peaty	Brosna_SC_070, 25A Lower Shannon	✓	✓	✓
Derrycarney (DCA), Co. Offaly	Degraded/peat extraction	Derrycarney, IE_SH_25B090950, order 1, 39	✓	Soft peaty	BROSNA_SC_070, 25A Lower Shannon	✓		
Derrycooly (DYC), Co. Offaly	Degraded/peat extraction	Derrycooly stream, IE_SH_25P050300, order 1, 35	✓	Soft peaty	Brosna_SC_050, 25A Lower Shannon	✓		
Drinagh (DRI), Co. Offaly	Degraded/peat extraction	Derrymullin and Loughderry River, IE_SH_25L010090, order 2, 11 Little [Cloghan], IE_SH_25L01009, order 3, 20		Soft peaty with some mineral sediments Soft peaty	Brosna_SC_070, 25 A Lower Shannon Brosna_SC_070, 25 A Lower Shannon		✓	✓
Falsk (FK), Co. Offaly	Degraded/peat extraction	Falsk 25, IE_SH_25B090950, order 1, 39	✓	Soft peaty	Brosna_SC_070, 25A Lower Shannon	✓		
Rossan (RS), Co. Meath	Degraded/peat extraction	Rossan 07, IE_EA_07K010100, order 1, 18	✓	Soft peaty	Boyne_SC_030, 07 Boyne	✓		

\*Shading denotes degraded bog sites.

## Appendix 2

**Table A2.1. Basic aspects of shortlisted FOSS/FAIR models: typical spatiotemporal resolution (peatland applications), grid structure, input/output data format and modularity**

Model	Spatial dimension	Typical spatial resolution	Typical temporal resolution	Grid structure	Input format	Output format	Modularity	Reference
MODFLOW 6	3D	0.4 – 200m	Daily – M	✓ Runs with regular/irregular grid/mesh structure	ASCII, Binary	ASCII, Binary	✓	Hughes et al., 2017; Langevin et al., 2017
TOPMODEL (in R)	2D	20 m – 1° (in LSMs)	6 min – M (in LSMs)	✓ Runs with regular grid structure (grid size ≤50m is recommended)	ASCII, Binary	ASCII, Binary	✗	Beven and Kirkby, 1979; Buytaert, 2018
SWAT+	2D	HRU	Daily	○ Sub-unit structure, but can be run on grid	ASCII, GIS vector	ASCII	✓	Bieger et al., 2017
GEOtop	1D, 3D	8 – 25m	Hr – M	✓ Runs with regular grid structure	ASCII	ASCII	✓	Rigon et al., 2006; Endrizzi et al., 2014
SUTRA	2D, 3D	1 cm – 1 m	15 min – Hr	✓ Runs with regular/irregular grid (and mesh) structure	ASCII, Binary	ASCII	✓	Provost and Voss, 2019
<b>DigiBog PDM</b>	1D, 2D (and 3D)	1 m – 1°	Daily – Annual	✓ Runs with square grid structure	ASCII	ASCII	✗	Baird et al., 2012
<b>E3SM (ELM) (ELM-SPRUCE)</b>	1D + spatial sub-grid variability	Single point – 1 km	30 min – M	✓ Runs with regular/irregular grid (sub-grid: tile) structure	netCDF-4, HDF5, DAP	netCDF, ASCII, Binary	✓	Golaz et al., 2019; Ricciuto et al., 2021
<b>CLASSIC (CLASS-CTEM)</b>	1D + spatial sub-grid variability	Single point – 0.22°, some processes not tuned for < 10 km	Daily (CTEM), < 0.5 Hr (CLASS)	✓ Runs with regular grid (sub-grid: tile/mosaic + composite) structure	netCDF-4	netCDF, csv	✓	Melton et al., 2020; Seiler et al., 2021
Community Land Model (CLM) 5 ( <b>CLM-SPRUCE</b> )	1D + spatial sub-grid variability	0.25 – 2.5°	6 Hr – Daily	✓ Runs with regular grid (sub-grid: tile/mosaic) structure	netCDF, csv	netCDF	✓	Shi et al., 2015; Lawrence et al., 2019
<b>ORCHIDEE (ORCHIDEE-PEAT)</b>	1D + spatial sub-grid variability	0.5° – 2°	30 min – M	✓ Runs with regular grid (sub-grid: tile) structure	netCDF	netCDF	✓	Krinner et al., 2005; Qiu et al., 2018
LPJml ( <b>LPJ-WHyMe</b> )	1D + spatial sub-grid variability	0.5 – 3.75°	Daily – Annual	✓ Runs with regular grid (sub-grid: mosaic) structure	netCDF, Binary	netCDF, Binary	✓	Wania et al., 2010; von Bloh et al., 2018
<b>CLSM (PEAT-CLSM)</b>	1D + spatial sub-grid variability	1 and 9 km – 0.25°	1 and 3 Hr – Daily	✓ Runs with regular grid (sub-grid: tile/mosaic) structure	netCDF	netCDF	✓	Koster et al., 2000; Bechtold et al., 2019

Models in bold are “peatland-specific” modules.

✓, fulfilled; ○, fulfilled to some extent; ✗, not fulfilled; D, dimensional; Hr, hourly; HRU, hydrological response unit; LSM, land surface model; M, monthly.

Source: adapted from Mozafari et al. (2023); licensed under the Creative Commons Attribution 4.0 International licence (<https://creativecommons.org/licenses/by/4.0/>).

Table A2.2. Process representation in the FOSS/FAIR models

Model	Water cycle			Carbon cycle			Nitrogen cycle														
	Subsurface			Surface runoff			GPP			CO <sub>2</sub> (R <sub>A</sub> and R <sub>H</sub> )			CH <sub>4</sub> (methanogenesis)			Nitrification			Denitrification		
	Energy balance	2D groundwater flow	Infiltration	Surface runoff	GPP	CO <sub>2</sub> (R <sub>A</sub> and R <sub>H</sub> )	CH <sub>4</sub> (methanogenesis)	Nitrification	Denitrification												
MODFLOW 6	x	✓ #∞ Darcy	✓ Richards, K(θ): Brooks-Corey	✓ f(GW/stream depth)	x	x	x	x	x												
TOPMODEL (in R)	x	x	✓ Green-Ampt	✓ f(topographic index)	x	x	x	x	x												
SWAT+	x	✓ #2 LRM	✓ Green-Ampt	✓ CN	x	✓ SWAT-MKT: f(MKT)	✓ SWAT-MKT: f(MKT)	✓ f(biomass volume)/ SWAT-MKT: f(MKT)	✓ Exp. rate Coef./SWAT-MKT: f(MKT)												
GEOtop	✓ Surface/ subsurface	✓ #∞ Richards	✓ Richards, K(θ): van Genuchten-Mualem	✓ Sat. excess/Infiltr. excess/f(GW)	x	x	x	x	x												
SUTRA	✓ Subsurface	✓ #∞ Darcy	✓ Richards, K(θ): van Genuchten	x	x	x	x	x	x												
DigiBog PDM	x	✓ #∞ Darcy (Dupuit-Boussinesq)	x	x	x	x	x	x	x												
E3SM (ELM) (ELM-SPRUCE)	✓ Surface/ subsurface	✓ #1 Darcy	✓ Richards, K(θ): van Genuchten-Mualem	✓ Sat. excess/ f(topographic index) /VIC/ <sup>a</sup> ELM-SPRUCE	✓ Farquhar/ ( <sup>a</sup> ELM-SPRUCE)	✓ R <sub>A</sub> (Rubisco), R <sub>H</sub> : f(T <sub>soil</sub> , θ, O <sub>2</sub> )	✓ (ELM-SPRUCE)	✓ Century N model	✓ Century N model												
CLASSIC (CLASS-CTEM)	✓ Surface/ subsurface	x	✓ Green-Ampt	✓ Infiltr. excess/f(soil)	✓ Farquhar	✓ R <sub>A</sub> (Rubisco), R <sub>H</sub> : f(WT, T, C <sub>soil</sub> )	✓ f(T, θ, sand content)	✓ f(T <sub>soil</sub> , NH <sub>4</sub> <sup>+</sup> )	✓ f(T <sub>soil</sub> , θ, NO <sub>3</sub> <sup>-</sup> )												

Table A2.2. Continued

Model	Water cycle			Carbon cycle			Nitrogen cycle		
	Subsurface			GPP			CH <sub>4</sub> (methanogenesis)		
	Energy balance	2D groundwater flow	Infiltration	Surface runoff	CO <sub>2</sub> (R <sub>a</sub> and R <sub>h</sub> )	CH <sub>4</sub> (methanogenesis)	Nitrification	Denitrification	
Community Land Model (CLM) 5 ( <b>CLM-SPRUCE</b> )	✓ Surface/subsurface	✓ #1 Darcy	✓ Richards, K(θ): f(ice impedance)	✓ Sat. excess/Infiltr. excess/f(topographic index/GW)/VIC/	✓ f(PFT, Farquhar)	✓ f(decomposition rate)	✓ Century N model	✓ Century N model	
ORCHIDEE ( <b>ORCHIDEE-PEAT</b> )	✓ Surface/subsurface	✗	✓ Richards/Fokker-Planck, K(θ): van Genuchten-Mualem	<b>CLM-SPRUCE</b> ✓ Infiltr. excess / f(soil/topographic index)	✓ f(R, T)	✓ <b>ORCHIDEE-PEAT</b>	✓ O-CN/Century model	✓ O-CN/Century model	
LPJml ( <b>LPJ-WHyMe</b> )	✗	✗	✓ #1 f(θ) or Green-Ampt	✓ Sat. excess/f(soil)	✓ Farquhar	✓ LPJ-WHyMe	✓ f(T, θ, pH)	✓ f(T, θ)	
CLSM ( <b>PEAT-CLSM</b> )	✓ Surface/subsurface	✗	✓ Richards/TOPMODEL combination/	✓ Sat./infiltr. excess f(topographic index)/	✗	✗	✗	✗	

Models in bold are “peatland-specific” modules.

<sup>a</sup>Peatland-specific module adopts a different approach when simulating the process. See Chapter 5 for more information.

✓, included; ✗, not included; #n, number of layers (∞: no limit); BC, boundary condition; CN, curve number approach; CO<sub>2</sub>, carbon dioxide; Coef., coefficient; C<sub>SOM</sub>, soil C mass; 2D, two-dimensional; Exp., exponential; GPP, gross primary production; GW, groundwater; K(θ), unsaturated hydraulic conductivity; LRM, linear reservoir model; MKT, microbial kinetics and thermodynamics; PFT, plant functional type, R, respiration; R<sub>a</sub>, autotrophic respiration; R<sub>h</sub>, heterotrophic respiration; T, temperature; VIC, variable infiltration capacity.

## Appendix 3

**Table A3.1. An overview of the structural features and complexities of MARRMoT models shortlisted for ensemble generation (sorted based on complexity)**

Model	No. of parameters	No. of storages	Structural feature(s)	Reference
Collie River Basin 1	1	1	Single soil moisture bucket; saturation excess runoff	Jothityangkoon <i>et al.</i> , 2001
Wetland model	4	1	Interception store; saturation excess runoff from distribution of soil depths; linear groundwater reservoir	Savenije, 2010
Collie River Basin 2	4	1	Separate bare soil and vegetation evaporation; saturation excess and subsurface runoff	Jothityangkoon <i>et al.</i> , 2001
New Zealand model v1	6	1	Separate vegetation and bare soil evaporation; saturation excess surface and subsurface runoff; linear baseflow reservoir	Atkinson <i>et al.</i> , 2002
GR4J	4	2	Soil moisture accounting; routing through two unit hydrographs; groundwater exchange	Perrin <i>et al.</i> , 2003a; Santos <i>et al.</i> , 2018
Susannah Brook model v2	6	2	Unsaturated and saturated zone stores; saturation excess runoff; non-linear subsurface and groundwater flow	Son and Sivapalan, 2007
TOPMODEL	7	2	Variable source area for saturation excess runoff; infiltration excess runoff; non-linear baseflow reservoir	Beven <i>et al.</i> , 1995
Penman model	4	3	Root zone storage; saturation excess runoff; deficit-based groundwater reservoir; linear routing reservoir	Wagener <i>et al.</i> , 2001
Variable Infiltration Capacity model	10	3	Dynamic interception storage; variable infiltration capacity; saturation excess overland flow; interflow; linear baseflow	Liang <i>et al.</i> , 1994
Xinanjiang	12	4	Tension water storage distribution; variable contributing area for saturation excess runoff; routing of surface, subsurface and groundwater flow	Ren-Jun, 1992; Jayawardena and Zhou, 2000

# Abbreviations

<b>BOD</b>	Biological oxygen demand
<b>DOC</b>	Dissolved organic carbon
<b>DON</b>	Dissolved organic nitrogen
<b>EC</b>	Electrical conductivity
<b>EPT</b>	Ephemeroptera/Plecoptera/Trichoptera
<b>FAIR</b>	Findable, accessible, interoperable, reusable
<b>FOSS</b>	Free and open-source software
<b>HRT</b>	Hydraulic retention time
<b>IPC</b>	Integrated Pollution Control
<b>MARRMoT</b>	Modular Assessment of Rainfall–Runoff Models Toolbox
<b>SAC</b>	Special area of conservation
<b>SSIS</b>	Small stream impact score
<b>SUVA<sub>254</sub></b>	Specific ultraviolet light absorbance at 254 nm
<b>SWAMP</b>	Strategies to Improve Water Quality from Managed Peatlands
<b>TAN</b>	Total ammonia nitrogen
<b>TDN</b>	Total dissolved nitrogen
<b>TSS</b>	Total suspended solids

# An Gníomhaireacht Um Chaomhnú Comhshaoil

Tá an GCC freagrach as an gcomhshaoil a chosaint agus a fheabhsú, mar shócmhainn luachmhar do mhuintir na hÉireann. Táimid tiomanta do dhaoine agus don chomhshaoil a chosaint ar thionchar díobhálach na radaíochta agus an truaillithe.

## Is féidir obair na Gníomhaireachta a roinnt ina trí phríomhréimse:

**Rialáil:** Rialáil agus córais chomhlíonta comhshaoil éifeachtacha a chur i bhfeidhm, chun dea-thorthaí comhshaoil a bhaint amach agus díriú orthu siúd nach mbíonn ag cloí leo.

**Eolas:** Sonraí, eolas agus measúnú ardchaighdeán, spriocdhírthe agus tráthúil a chur ar fáil i leith an chomhshaoil chun bonn eolais a chur faoin gcinnteoireacht.

**Abhcóideacht:** Ag obair le daoine eile ar son timpeallachta glaine, táirgiúla agus dea-chosanta agus ar son cleachtas inbhuanaithe i dtaobh an chomhshaoil.

## I measc ár gcuid freagrachtaí tá:

### Ceadúnú

- > Gníomhaíochtaí tionscail, dramhaíola agus stórála peitрил ar scála mór;
- > Sceitheadh fuíolluisce uirbhig;
- > Úsáid shrianta agus scaoileadh rialaithe Orgánach Géinmhodhnaithe;
- > Foinsí radaíochta ianúcháin;
- > Astaíochtaí gás ceaptha teasa ó thionscal agus ón eitlíocht trí Scéim an AE um Thrádáil Astaíochtaí.

### Forfheidhmiú Náisiúnta i leith Cúrsaí Comhshaoil

- > Iniúchadh agus cigireacht ar shaoráidí a bhfuil ceadúnas acu ón GCC;
- > Cur i bhfeidhm an dea-chleachtais a stiúradh i ngníomhaíochtaí agus i saoráidí rialáilte;
- > Maoirseacht a dhéanamh ar fhreagrachtaí an údaráis áitiúil as cosaint an chomhshaoil;
- > Caighdeán an uisce óil phoiblí a rialáil agus údaruithe um sceitheadh fuíolluisce uirbhig a fhorfheidhmiú
- > Caighdeán an uisce óil phoiblí agus phríobháidigh a mheasúnú agus tuairisciú air;
- > Comhordú a dhéanamh ar líonra d'eagraíochtaí seirbhíse poiblí chun tacú le gníomhú i gcoinne coireachta comhshaoil;
- > An dlí a chur orthu siúd a bhriseann dlí an chomhshaoil agus a dhéanann dochar don chomhshaoil.

### Bainistíocht Dramhaíola agus Ceimiceáin sa Chomhshaoil

- > Rialacháin dramhaíola a chur i bhfeidhm agus a fhorfheidhmiú lena n-áirítear saincheisteanna forfheidhmithe náisiúnta;
- > Staitisticí dramhaíola náisiúnta a ullmhú agus a fhoilsiú chomh maith leis an bPlean Náisiúnta um Bainistíocht Dramhaíola Guaisí;
- > An Clár Náisiúnta um Chosc Dramhaíola a fhorbairt agus a chur i bhfeidhm;
- > Reachtaíocht ar rialú ceimiceáin sa timpeallacht a chur i bhfeidhm agus tuairisciú ar an reachtaíocht sin.

### Bainistíocht Uisce

- > Plé le struchtúir náisiúnta agus réigiúnacha rialachais agus oibriúcháin chun an Chreat-treoir Uisce a chur i bhfeidhm;
- > Monatóireacht, measúnú agus tuairisciú a dhéanamh ar chaighdeán aibhneacha, lochanna, uiscí idirchreasa agus cósta, uiscí snámha agus screamhuisce chomh maith le tomhas ar leibhéil uisce agus sreabhadh abhann.

### Eolaíocht Aeráide & Athrú Aeráide

- > Fardail agus réamh-mheastacháin a fhoilsiú um astaíochtaí gás ceaptha teasa na hÉireann;
- > Rúnaíocht a chur ar fáil don Chomhairle Chomhairleach ar Athrú Aeráide agus tacaíocht a thabhairt don Idirphlé Náisiúnta ar Gníomhú ar son na hAeráide;

- > Tacú le gníomhaíochtaí forbartha Náisiúnta, AE agus NA um Eolaíocht agus Beartas Aeráide.

### Monatóireacht & Measúnú ar an gComhshaoil

- > Córais náisiúnta um monatóireacht an chomhshaoil a cheapadh agus a chur i bhfeidhm: teicneolaíocht, bainistíocht sonraí, anailís agus réamhaisnéisiú;
- > Tuairiscí ar Staid Thimpeallacht na hÉireann agus ar Tháscairí a chur ar fáil;
- > Monatóireacht a dhéanamh ar chaighdeán an aeir agus Treoir an AE i leith Aeir Ghlain don Eoraip a chur i bhfeidhm chomh maith leis an gCoinbhinsiún ar Aerthruailliú Fadraoin Trasteorann, agus an Treoir i leith na Teorann Náisiúnta Astaíochtaí;
- > Maoirseacht a dhéanamh ar chur i bhfeidhm na Treorach i leith Torainn Timpeallachta;
- > Measúnú a dhéanamh ar thionchar pleananna agus clár beartaithe ar chomhshaoil na hÉireann.

### Taighde agus Forbairt Comhshaoil

- > Comhordú a dhéanamh ar ghníomhaíochtaí taighde comhshaoil agus iad a mhaoiniú chun brú a aithint, bonn eolais a chur faoin mbeartas agus réitigh a chur ar fáil;
- > Comhoibriú le gníomhaíocht náisiúnta agus AE um thaighde comhshaoil.

### Cosaint Raideolaíoch

- > Monatóireacht a dhéanamh ar leibhéil radaíochta agus nochtadh an phobail do radaíocht ianúcháin agus do réimsí leictreamaighnéadacha a mheas;
- > Cabhrú le pleananna náisiúnta a fhorbairt le haghaidh éigeandálaí ag eascairt as tasmí núicléacha;
- > Monatóireacht a dhéanamh ar fhorbairtí thar lear a bhaineann le saoráidí núicléacha agus leis an tsábháilteacht raideolaíochta;
- > Sainseirbhísí um chosaint ar an radaíocht a sholáthar, nó maoirsiú a dhéanamh ar sholáthar na seirbhísí sin.

### Treoir, Ardú Feasachta agus Faisnéis Inrochtana

- > Tuairisciú, comhairle agus treoir neamhspleách, fianaise-bhunaithe a chur ar fáil don Rialtas, don tionscal agus don phobal ar ábhair maidir le cosaint comhshaoil agus raideolaíoch;
- > An nasc idir sláinte agus folláine, an geilleagar agus timpeallacht ghlan a chur chun cinn;
- > Feasacht comhshaoil a chur chun cinn lena n-áirítear tacú le hiompraíocht um éifeachtúlacht acmhainní agus aistriú aeráide;
- > Tástáil radóin a chur chun cinn i dtithe agus in ionaid oibre agus feabhsúchán a mholadh áit is gá.

### Comhpháirtíocht agus Líonrú

- > Oibriú le gníomhaireachtaí idirnáisiúnta agus náisiúnta, údaráis réigiúnacha agus áitiúla, eagraíochtaí neamhrialtais, comhlachtaí ionadaíochta agus ranna rialtais chun cosaint comhshaoil agus raideolaíoch a chur ar fáil, chomh maith le taighde, comhordú agus cinnteoireacht bunaithe ar an eolaíocht.

## Bainistíocht agus struchtúr na Gníomhaireachta um Chaomhnú Comhshaoil

Tá an GCC á bainistiú ag Bord lánaimseartha, ar a bhfuil Ard-Stiúrthóir agus cúigear Stiúrthóir. Déantar an obair ar fud cúig cinn d'Oifigí:

1. An Oifig um Inbhuanaitheacht i leith Cúrsaí Comhshaoil
2. An Oifig Forfheidhmithe i leith Cúrsaí Comhshaoil
3. An Oifig um Fhianaise agus Measúnú
4. An Oifig um Chosaint ar Radaíocht agus Monatóireacht Comhshaoil
5. An Oifig Cumarsáide agus Seirbhísí Corparáideacha

Tugann coistí comhairleacha cabhair don Gníomhaireacht agus tagann siad le chéile go rialta le plé a dhéanamh ar ábhair inní agus le comhairle a chur ar an mBord.

## EPA Research

**Webpages:** [www.epa.ie/our-services/research/](http://www.epa.ie/our-services/research/)  
**LinkedIn:** [www.linkedin.com/showcase/eparesearch/](http://www.linkedin.com/showcase/eparesearch/)  
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