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Authors(s)	Donahue, Thomas, Renou-Wilson, Florence, Pschenyckyj, Catharine, Kelly-Quinn, Mary
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A REVIEW OF THE IMPACT ON AQUATIC COMMUNITIES OF INPUTS FROM PEATLANDS DRAINED FOR PEAT EXTRACTION

Thomas Donahue, Florence Renou-Wilson, Cat Pschenyckyj and Mary Kelly-Quinn

ABSTRACT

Peatlands, which account for 21% of total land area in the Republic of Ireland, have been subjected to large-scale drainage and extraction of peat for fuel since the Industrial Revolution, continuing throughout the twentieth and early twenty-first century. Though the effects of this extraction on the water quality of streams draining peatlands have received extensive study, little research has investigated the direct effects of peat drainage and extraction on the aquatic biota (e.g. fish, macroinvertebrates, macrophytes, phytobenthos) of these streams and no studies have yet been published on these effects in Irish peatland streams. This paper reviewed the available data on the effects of physico-chemical stressors such as ammonia, heavy metals and peat sedimentation associated with drainage and extraction on the aquatic biota of peatland streams. Results suggest that peatland drainage and extraction provide several serious challenges to aquatic life, including increased mortality, reduced richness, behavioural changes, habitat alterations and changes in community structure. This review also details the benefits of peatland restoration via drain-blocking on aquatic biota and highlights the extensive knowledge gaps that exist in understanding the effects of peatland derived stressors on surface water communities. These include the impact of peat sedimentation on aquatic biota, the paucity of information on the effects on phytobenthos communities and a lack of knowledge of how the various peat-derived stressors interact to impact aquatic communities in addition to inadequate knowledge of the responses of stream biota to peatland restoration.

Thomas Donahue, Florence Renou-Wilson, Cat Pschenyckyj and Mary Kelly-Quinn (corresponding author; email: mary.kelly-quinn@ucd.ie. ORCID iD: https://orcid.org/0000-0002-0110-5360), School of Biology and Environmental Science University College Dublin.

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INTRODUCTION

Peatlands are a type of wetland, comprising transitional ecosystems that contain a minimum depth of peat, typically 45cm on undrained land and 30cm on drained land (Rydin and Jeglum 2006; Renou-Wilson *et al*. 2011). Peat refers to the partially decomposed remains of organic material that accumulates in anoxic, water saturated conditions. Though peat is largely composed of plant material, its composition may vary depending on local or historical conditions unique to the area in which its formation occurs (Yeloff and Mauquoy 2006). As such, peat includes the remains of trees, grasses and mosses that accumulate under highly saturated conditions exceeding the rate of decomposition (Renou-Wilson 2018).

Across Europe, peatland habitat has been estimated to account for 12.5% of total land area (Xu *et al*. 2018). A survey of published data for Europe as a whole, showed that the percentage of peatland compared to total national area was highest in Finland (26.7%) and lowest in Cyprus (< 0.1%) with peatlands in Ireland accounting for 21% of total land area (Renou-Wilson *et al*. 2011). Evidence of anthropogenic use and modification of peatlands dates to the Neolithic (6,500 BC to 2,000 BC) in Ireland and Britain when they were subject to drainage for farming and use as fuel (Swindles *et al*. 2019). However, large-scale exploitation of peat as a resource did not truly commence until the time of the Industrial Revolution during the nineteenth century (Feehan *et al*. 2008). This is not confined to Ireland, an assessment by the International Mire Conservation Group (IMCG) reported a total loss of 130,000km^2 for Europe (including European Russia) compared to that found at its estimated maximum extent (Rydin and Jeglum 2006). Irish peatlands have been extensively used for domestic turf cutting (Renou-Wilson 2018). In 1946, the Republic of Ireland passed the Turf Development Act and created the state-sponsored body Bord na Móna for the industrial extraction of peat with the purpose of generating economic benefits in peatland areas and to expand the use of turf as a fuel source. All activities from peat extraction to forestry

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and agriculture require some form of drainage which impacts on peatland ecosystem services (biodiversity, climate, water retention) (Renou-Wilson *et al.* 2011). *In-situ* emissions of gaseous and fluvial carbon from damaged peatlands together with off-site emissions of carbon from burning and use of peat in horticulture, have been estimated to release upwards of 3 million tonnes (Mt)s of carbon annually in the Republic of Ireland alone (Wilson *et al.* 2013).

While the degradation of peatlands has been shown to produce deleterious effects on terrestrial flora (Ramchunder *et al.* 2009) and carbon emissions, less is known of the effects on water quality and associated aquatic biota of streams and rivers draining degraded peatlands. The drainage and cutting of peatlands produce changes in the hydrology of the peatland itself, potentially releasing a range of known stressors from heavy metals to sediments and nutrients (Holden *et al.* 2008; Armstrong *et al.* 2010; Kløve *et al.* 2010; Ramchunder *et al.* 2011; Evans *et al.* 2016; Gandois *et al.* 2020). In a previous review about the effects of drainage on UK upland peatlands, Ramchunder *et al.* (2009) related how drainage lowers the surrounding water table, alters natural flow patterns in peatlands and leads to deterioration of the soil. Erosion related to peatland drainage was seen as a major source of sediment input in surrounding catchments (Ramchunder *et al.* 2009). A major concern related to the effects of drainage and extraction on water quality is the possibility of ammonification and further nitrification which can result in the death of aquatic biota and the promotion of anoxic conditions in aquatic habitats (Camargo and Alonso 2006). Furthermore, stressor inputs to small streams can produce knock-on effects on water quality and aquatic communities further downstream (Wipfli *et al.* 2007).

In Ireland, nearly 50% of surface waters are not in a satisfactory ecological condition, thus failing to meet the legal requirements of the European Union (EU) Water Framework Directive (WFD) (EPA 2020). In this report 7.5% of water bodies at risk of not achieving good status were due to peat extraction (EPA 2020). Many of these waterbodies are in the midlands (Figure 1) where elevated ammonia and sediment inputs are considered to be the key potential stressors impacting ecological conditions in these rivers. Variable ammonium thresholds for individual peat extraction sites and the reported exceedance of ammonium concentrations by existing Integrated Pollution Control (IPC) licenses for Bord na Móna have led the Environmental Protection Agency (EPA) to investigate whether peat extraction is a contributory factor in elevated ammonia concentrations in water bodies (Government of Ireland 2018).

Given these pressures and the suite of potential impacts on water quality in peatland streams, this paper reviews the available information both on potential impacts on aquatic biota and the processes involved. The focus is on temperate and boreal streams within peatlands that have been drained for peat extraction. As such, this paper is specifically concerned with streams draining bogs that have been subject to drainage and extraction rather than fens, which though a type of peatland, have predominantly been affected by agricultural reclamation (Bragg and Lindsay 2003).

METHODS

The literature search was performed via Google Scholar and OneSearch which access prominent databases such as Web of Science, ScienceDirect and Scopus. Keywords included 'Ireland', 'peatland', 'drainage', 'restoration', 'stream', 'river', 'macroinvertebrates', 'phytobenthos' and 'macrophytes'. Only published papers have been included.

In terms of limitations, no cut-off dates were utilised though recent research was given precedence and when older studies were used an attempt was made to find more literature corroborating results. While the review has a strong focus on peatland studies in Ireland, the geographical limitations were extended to include peatlands occurring in boreal and other temperate climates but did not include those found in tropical environments owing to the large differences in local climate and biodiversity. Irish sources were a priority but studies originating in the UK, Scandinavia, continental Europe and Canada were also utilised. Where studies specific to peatlands were lacking, papers detailing the effects of stressors from peatlands that have been described from other land-use changes were used to highlight potential impacts. A considerable body of literature is available on the aquatic communities of rivers draining afforested peatlands (Feeley *et al.* 2013; O'Driscoll *et al.* 2012; O'Driscoll *et al.* 2014; Kelly-Quinn *et al.* 2016) and where relevant reference is made to these studies.

RESULTS

Here we present what has been published on impacts of the aforementioned peatland-derived stressors (sediment, nutrients, heavy metals, dissolved organic carbon (DOC)/acidity and changes in flow) on aquatic biota.

PEATLAND DERIVED SEDIMENTATION IMPACTS ON AQUATIC BIOTA

Research on the effects of peat-derived sediment on aquatic biota has included field observations

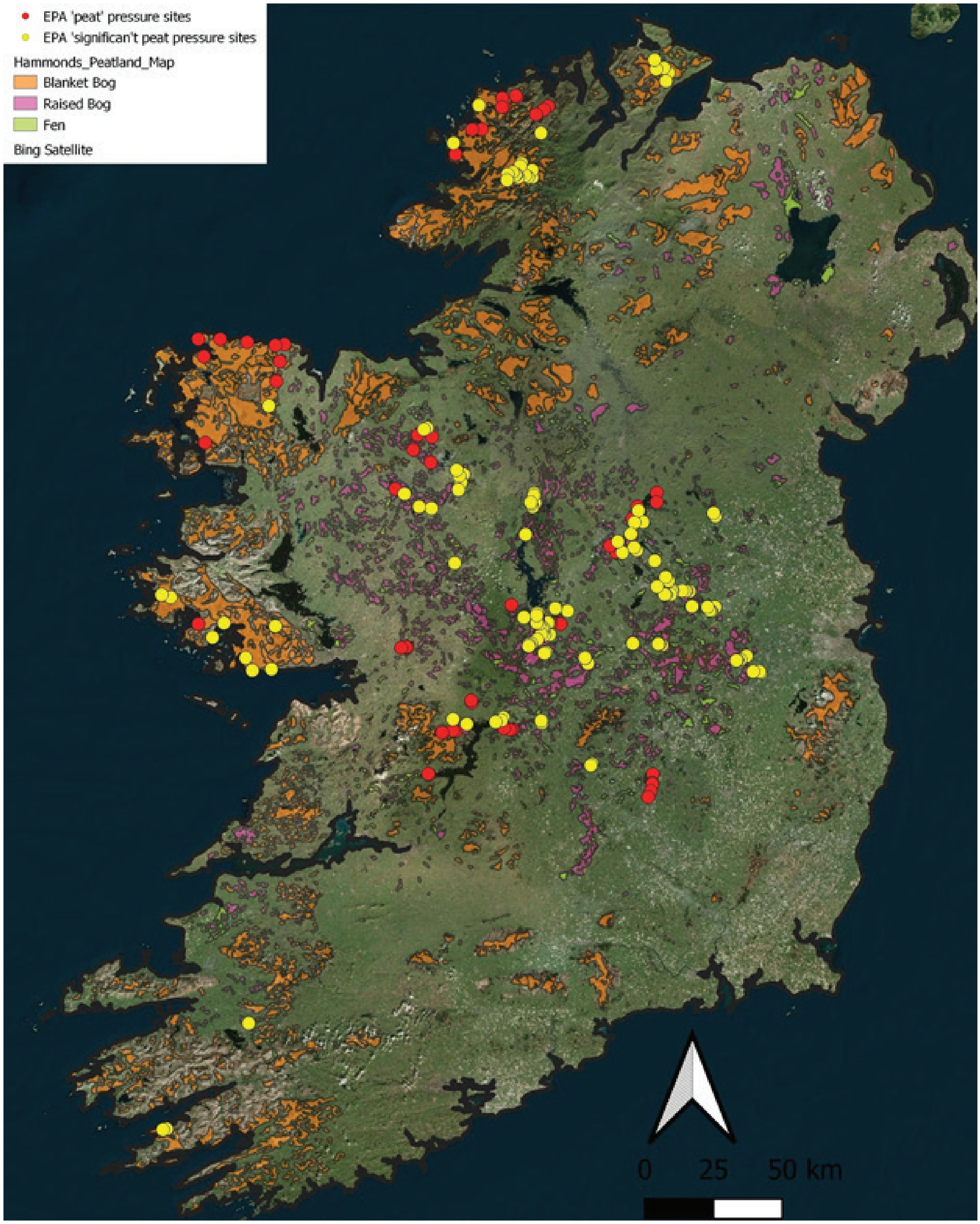

Fig. 1–Map of Ireland showing EPA pressure sites associated with peatland disturbances (peat map data from Hammond 1981.)

(Laine 2001; Ouellette *et al.* 2006) as well as experimental additions of peat sediment in the field (Aspray *et al.* 2017; Brown *et al.* 2018) and laboratory settings (Ouellette *et al.* 2003; Strychar 1997). Across all aquatic groups a range of impacts has been reported including changes in habitat preference (Ouellette *et al.* 2003; Ouellette *et al.* 2006), decreased growth at various stages of development (Laine 2001; Ouellette *et al.* 2006), increased mortality (Laine *et al.* 2001) and decreased gross primary production (GPP) (Aspray *et al.* 2017).

MACROINVERTEBRATES

Responses of macroinvertebrate taxa to peat sediment derived from field and experimental studies are generally similar and also broadly correspond with impacts of non-peat deposited sediment from other sources (Wood *et al.* 2005; Larsen and Ormerod 2010). These include decreased density and richness (Brown *et al.* 2018), increased drift (Aspray *et al.* 2017) and loss of abundance (Ramchunder *et al.* 2011).

Only three key papers (Ramchunder *et al.* 2011; Aspray *et al.* 2017; Brown *et al.* 2018) addressed community-level responses while a further seven papers (Olsson and Persson 1986; Strychar 1997; Laine 2001; Laine *et al.* 2001; Ouellette *et al.* 2003; Ouellette *et al.* 2006; Ramchunder *et al.* 2010) were focused on responses of specific species (Table 1). Brown *et al.* (2018) demonstrated loss of density and richness of Ephemeroptera and Plecoptera in particular, with the addition of peat to mesocosms in peatland rivers in Britain. Here, overall macroinvertebrate density decreased by 65% with increased bed sedimentation and there was a 50% decrease in richness, especially among Plecoptera. Field sampling also carried out by these authors showed declines in density of 95% across a density gradient of fine particulate organic matter (FPOM) with significant losses of stoneflies such as *Leuctra inermis* (Brown *et al.* 2018). Previous field experiments showed increased drift densities following the addition of peat, here again predominantly among plecopterans, including *L. inermis* (Aspray *et al.* 2017). The authors hypothesised that this may have been the result of smothering, abrasion, decreased habitat quality or a combination of such stressors.

Another experimental study showed significant macroinvertebrate community variation between streams draining drained peatlands, those with blocked drains and intact blanket bogs (Ramchunder *et al.* 2011). While intact sites and those with blocked drains were similar in species composition, drained sites had lower abundances of Ephemeroptera, Plecoptera and Trichoptera (EPT) and higher abundances of simuliids and chironomids. It was concluded that community structure, especially the greater abundance of EPT at drain-blocked sites correlated with reductions in the concentration of suspended sediment (SS) and fine particulate or FPOM.

Research has also reported variation in species composition in peatland streams according to stream order with a lack of habitat suitability in the fine substrate material of lower-order streams potentially influencing species distribution (Ramchunder *et al.* 2010). Species such as *L. inermis*, *Baetis rhodani*, *Rhithrogena semicolorata*, *Ecdyonurus torrentis*, *Hydroptila* spp. and *Esolus parallelepipedus* were shown to inhabit higher order peatland streams with coarser bed material, with some taxa (e.g. *B. rhodani*, *Ecdyonurus* spp., *Leuctra* spp.) having previously been shown to display negative reactions to increased sediment load (Wood and Armitage 1997; Larsen and Ormerod 2010; Conroy *et al.* 2016). This natural spatial variation in species composition among peatland streams needs to be considered when evaluating impact of sedimentation.

Several studies have focused on specific invertebrate species and the effects on physiology, behaviour and distribution. For example, peat extraction and associated sedimentation have been shown to affect the behaviour and physiology of invertebrates other than insects. Laboratory tests conducted on sand shrimp (*Crangon septemspinosa*) used modified aquaria containing sand and peat substrates to determine how peat influx from extraction interacted with substrate preference (Ouellette *et al.* 2003). Here 73% of shrimp displayed a preference for sand. Following the introduction of food to both substrates 75% of shrimp were observed on sand, only moving to the peat when food on sand had been depleted, with the majority (60%) returning to sand after all food on peat was consumed. Histological analysis of shrimp that had been feeding on the peat substrate showed ingested peat was extensive throughout the digestive tract with potential consequences on overall health. When starved, sand shrimp did not consume peat suggesting it is not viewed as a nutritional source but is taken up when eating provided food material. Field research by the same authors showed that peat fibres from commercial peat extraction contributed to a reduction in habitat availability for sand shrimp and a reduction in their condition (Ouellette *et al.* 2006). Fewer shrimp were observed over areas of high peat accumulation than elsewhere. A range of potential explanations were suggested for this avoidance behaviour including poor burrowing material, hydrogen sulphide (H_2S) production, or localised hypoxia but no specific conclusions were drawn (Ouellette *et al.* 2006). Research on the American oyster (*Crassostrea virginica*) has produced differing results (Strychar 1997; Mallet *et al.* 2005). One study reported decreased clearance and absorption efficiency with increasing peat sediment (Strychar 1997). However, the Mallet *et al.* (2005) experimental study did not detect deleterious effects on survival or reproduction. Nevertheless, as testing was performed at optimal temperature and food concentrations, it remains unclear how high levels of suspended peat could affect this species in a natural setting subject to the influence of multiple stressors. Sedimentation is expected to impact crustaceans such as *Gammarus* and *Austropotamobius pallipes*, the pearl mussel *Margaritifera margaritifera* and other aforementioned fine sediment-sensitive macroinvertebrate fauna (e.g. EPT) in Irish peatland rivers through multiple routes, including smothering, loss of habitat and food sources and other physiological effects. The sensitivity of pearl mussels to sedimentation is widely reported (e.g. Denic & Geist 2015; Lopes-Lima et al. 2017). Overall, these studies emphasise the need to protect stream fauna from elevated peat sedimentation.

PHYTOBENTHOS

There is a paucity of studies on effects of peat sediment on phytobenthos. The aforementioned study by Aspray *et al.* (2017) reported mean reductions in gross primary production (GPP) of 54% following peat addition compared to control sites. It was hypothesised that this reduction occurred due to the potential combination of substrate smothering, increased turbidity or abrasion of the biofilm, as found in other work showing declines in GPP in non-peatland catchments following storm events (Roberts *et al.* 2007). However, the authors did not conclusively distinguish between these or other potential stressors (e.g. nutrients, heavy metals) in their analysis.

FISH

Fish species, such as those within the family Salmonidae, have well-documented requirements of clean, silt-free gravel for the construction of redds and successful recruitment of alevins (Kelly-Quinn *et al.* 1996b; Suttle *et al.* 2004; Riebe *et al.* 2014). However, only three studies have investigated effects of peat sedimentation on salmonids (Olsson and Persson 1986; Laine 2001; Laine *et al.* 2001) (Table 1). Laine (2001) reported that recapture rates of stocked yearling salmon (*Salmo salar*) were significantly lower in riffles experiencing peat loading than in reference streams. The study also showed that recaptured salmon from drainage affected streams displayed lower weight and shorter lengths than those in reference conditions, with overall size inversely related to the estimated amount of sediment loading and a marked dietary shift from *Hydropsyche* and *Rhyacophila* larvae to *Ephemerella* species. Research by Laine *et al.* (2001) was focused on egg survival of brown trout (*Salmo trutta*) in constructed gravel beds. Survival at the eyed stage ranged from 53–84% but dropped to 0.2–9.1% before emergence as alevins, with the lowest rates associated with streams draining degraded peatlands. Survival was inversely related to an influx of sediment and iron (Fe) leaching from drained peatlands due to the precipitation of iron onto the stream bed which can result in smothering salmonid redds as well as physiological stress to developing embryos (Laine *et al.* 2001). This is in keeping with the results of earlier experimental research of Olsson and Persson (1986) using brown trout eggs and alevins in which redds constructed in the field were exposed to varying concentrations of peat. At concentrations exceeding 40% volume, egg survival decreased (≤65%), with high peat concentrations also linked to an increase in premature alevin emergence compared to control redds and extended rather than synchronous emergence. Furthermore, the prematurely emerging alevins were smaller and had larger yolk reserves, likely making them poorer swimmers and more susceptible to predation.

While this review has separately outlined the available knowledge on the impacts of sediment on three groups of aquatic organisms it should be noted that all three are interconnected in aquatic food webs and thus effects on any one group has consequences for other biota. Existing water quality metrics and in particular sediment specific metrics such as Proportion of Sediment sensitive Invertebrates (PSI) (Extence *et al.* 2013; Conroy *et al.* 2016) need to be tested in peatland systems in terms of their effectiveness as monitoring tools in peatland streams and rivers.

IMPLICATIONS OF AMMONIA, NITRITE AND NITRATE RUNOFF FROM DRAINED PEATLANDS FOR AQUATIC BIOTA

Drainage and cutting of peatlands produces changes in the hydrology of the peatland itself, resulting, as previously mentioned, in the release of nutrients such as nitrogen (N) and phosphorus (P) (Holden *et al.* 2004; Kløve *et al.* 2010). The effects of drainage-induced nutrient inputs to water have been documented (Kløve 2001; Vassiljev and Blinova 2012; Nieminen *et al.* 2017a,b) and other studies have described the effects of the availability of nutrients such as N from atmospheric deposition on aquatic communities within peatlands (van Duinen *et al.* 2003; van Duinen *et al.* 2006). However, the specific effects of nutrient inputs from drained peatlands on aquatic biota of surrounding catchments are largely unknown. Though ammonium (NH_4) and nitrate (NO_3) are found at low levels in natural peatlands due to the anoxic conditions, concentrations significantly increase when peat is exposed to oxygen (Schouten 2002). The production of ammonia is induced by the actions of aerobic bacteria that trigger the decomposition of organic matter present. This ammonia may then be carried into adjacent surface waters through runoff, where, at pH levels of 7.2 and above the equilibrium shifts towards the unionised form (NH_3) and can exceed toxic levels. Unionised ammonia is extremely toxic to fish, in particular salmonids, even at low concentrations (0.02mg/L N) (Camargo and Alonso 2006). Furthermore, nitrification depletes dissolved oxygen and can lead to anoxic conditions that are also detrimental to all aquatic life.

Like NH_3, nitrite (NO_2) is highly toxic to aquatic fauna at low concentrations. Alonso *et al.* (2005), cited in Camargo and Alonso (2006), proposed based on available toxicity data, that concentrations of nitrite below 0.08–0.36mg/L N may protect aquatic biota. Nitrate appears to be more important in terms of its contribution to the process of eutrophication (Camargo and Alonso 2006). The authors suggest that levels of total N lower than 1 mg/L N could prevent the development of eutrophication, although no ecologically relevant thresholds have been set with respect to the WFD.

AMMONIUM

Nitrogenous by-products such NH_4 and NO_3 are not typically found in high concentrations in intact peatlands (Rydin and Jeglum 2006) but increase significantly in extracted sites with research showing higher levels of ammonium and nitrate in the peat and water of extracted sites compared with controls (Wind-Mulder *et al.* 1996). No data have been published regarding nutrient inputs from extracted Irish peatlands.

Though no studies have yet addressed the impact of ammonification in peat water on aquatic biota, potential effects may be inferred from the existing literature. Ammonium is relatively neutral in aqueous settings but as mentioned, its unionised form (NH_3) and derivatives such as NO_2 can induce mortality in aquatic organisms even in low concentrations (Wang and Leung 2015). Alkaline conditions, such as those owing to limestone or marl substrate found underlying most Irish peatlands where pH can range from 7 to 9 (Renou-Wilson *et al.* 2008), induces the formation of NH_3 (Camargo and Alonso 2006) which, unlike NH_4, is capable of permeating biological membranes (Eddy 2005).

Freshwater fish species, such as salmonids, are particularly susceptible to the toxicity of unionised ammonia, with concentrations of 0.027mg/L N capable of inducing mortality in rainbow trout (*Oncorhynchus mykiss*) eggs (Solbé and Shurben 1989) and 0.02mg/L N for adults (Camargo and Alonso 2006). Furthermore, concentrations of 0.12 mg/L N unionised ammonia can reduce growth and cause death in the European eel (*Anguilla anguilla*), with statistically significant decreases occurring at 0.24mg/L N and a lethal threshold concentration of 1mg/L N (Sadler 1981). Furthermore, NH_3 can also induce disorientation, hyperventilation, organ damage, suppression of the immune system, and death in adult fish (Randall and Tsui 2002; Eddy 2005; Camargo and Alonso 2006) with larvae and juveniles considered to be most at risk. Exposure to increasing ammonia levels greater than 0.033mg/L N showed a significant decrease in critical swimming speed in Coho salmon (*Oncorhynchus kisutch*) due to metabolic and physiological stress (Wicks *et al.* 2002). Moreover, results from the same study showed that acute ammonia toxicity in *O. mykiss* was related to swim speed, with lethal concentration to kill half the population (LC50) values decreasing from 207±21.99mg/L N in resting fish to 32.38±10.81mg/L N in fish swimming at 60% critical swim speed, suggesting that ammonia toxicity may be especially dangerous during stages of salmonid lifestyle that require extended periods of exertion such as predator avoidance in juveniles or during spawning (Wicks *et al.* 2002).

Macroinvertebrate species display differing sensitivities to ammonia. *Gammarus pulex* has shown adverse reactions to NH_3 in experimental studies, with 0.15mg/L NH_3 and 0.91mg/L NO_2- resulting in a 98% decrease in abundance compared to controls, though in the same study, effects on the trichopteran *Limnephilus lunatus* were less apparent (Berenzen *et al.* 2001). The majority of research conducted on crustaceans and mollusc sensitivity to NH_3 has shown high levels of mortality with relatively short-term exposure (≤ 96 hours) (Hickey and Vickers 1994; Mummert *et al.* 2003; Wang *et al.* 2007). In contrast, mesocosm studies from New Zealand showed no significant declines in taxa richness, numbers of EPT taxa and no significant increases in drift with increasing (from mean of 0.141mg/L N (control) to 6.25mg/L N (high treatment)) levels of NH_3 (Hickey *et al.* 1999). Similarly, an earlier study showed that pollution-sensitive groups such as Plecoptera and Ephemeroptera were among the least sensitive to NH_3 toxicity of nine invertebrate species studied (Hickey and Vickers 1994) suggesting that it may not be a significant stressor among EPT taxa. Further study is clearly required in this area to enable identification of the impacting stressors on macroinvertebrates inhabiting degraded peatland streams, and within an Irish context.

NITRITE AND NITRATE

Like NH_3, nitrite (NO_2-), arising from oxidation of NH_{4+} or NH_3, is highly toxic to aquatic fauna such as fish and crustaceans at low concentrations. This is mainly due to hypoxia caused by the oxidation of Fe and copper (Cu) atoms in haemoglobin and haemocyanin, respectively, adversely affecting the binding and delivery of oxygen to tissues (Jensen 2003; Camargo and Alonso 2006). Nitrite also accumulates in the organs and body tissues, where it can induce deleterious physiological changes such as organ damage (Arillo 1984), hyperplasia of respiratory epithelium (Kroupova *et al.* 2008) and destruction of red blood cells (Jensen 2003). No known studies have considered whether this is a stressor in peatland streams and its potential impacts on aquatic biota.

The final compound in the nitrification of NH_4, nitrate (NO_3), is less toxic than NH_3 or NO_2- but is integral to the process of eutrophication when in elevated concentrations, contributing to algal blooms and associated depletion of oxygen (O_2) in aquatic habitats (Smith 2003; Camargo and Alonso 2006). Though no studies directly relating to the effects of NO_3- toxicity on the biota of peatlands streams have been sourced, research conducted on peatland lakes in the west of Ireland draining surrounding blanket bog, including intact bogs, mature conifer plantations and recently clearfelled sites observed changes in community structure attributable to eutrophication (Drinan *et al.* 2013). Though the nutrient source was linked with application of fertiliser during afforestation, the authors considered that nutrient enrichment of lakes and thus changes in macroinvertebrate community structure were exacerbated by leaching of these nutrients into surface waters from drained peatlands. The reported changes in macroinvertebrate assemblages may therefore serve as an indicator of the potential effects of nutrient leaching due to peat extraction and drainage, rather than to NO_3- toxicity.

IMPACTS OF HEAVY METAL CONTAMINATION ON AQUATIC BIOTA IN DRAINED PEATLAND STREAMS

The impacts of heavy metals on aquatic biota are well described in the literature especially from studies dealing with mine drainage (e.g. Harding 2005). Streams draining degraded peatlands have been shown to export significantly higher concentrations of heavy metals such as aluminium (Al), iron (Fe), lead (Pb), arsenic (As), nickel (Ni) and cadmium (Cd) compared to undrained peatland (Gandois *et al.* 2020; Ramchunder *et al.* 2009) with Fe, As, Cd and zinc (Zn) being strongly linked with the presence of dissolved organic matter (DOM) from degraded peatlands (Gandois *et al.* 2020). Atmospherically deposited Pb from an eroding peatland in the UK showed inputs of 30.0 ± 6.0g ha^{-1} a^{-1}, with 85% of mobilised Pb associated with particulate matter from the peatland (Rothwell *et al.* 2008), in keeping with subsequent research from the USA linking changes in water table depth to the mobilisation of Pb by DOM (Jeremiason *et al.* 2018). The export of mercury-containing peat particles has also been reported (Surette *et al.* 2002).

Al and Fe are two heavy metals commonly reported in studies investigating inputs from drained or degraded peatlands (Vuori *et al.* 1998; Krachler *et al.* 2010 and 2012) and their potential impact on aquatic communities (Vuori *et al.* 1998; Laine *et al.* 2001). In terms of macroinvertebrate studies, researchers were generally unable to attribute impacts directly to the heavy metals because of other confounding factors such as elevated sediment, acidity and vegetation coverage (Vuori *et al.* 1998). For example, Vouri *et al.* (1998) found that 94% of the variance in EPT richness, the most species rich macroinvertebrates present, was positively correlated with percent coverage of aquatic vegetation, intensity of drainage, maximum Al, and mean Fe concentration while maximum Al concentration and mean Fe concentration accounted for 37 and 15% of variance, respectively. Though Al and Fe were found to induce changes in community structure, and maximum Al concentrations (250–1100µg/L Al) and mean Fe concentrations (236–1682µg/L Fe) were considered high enough to negatively impact fish and macroinvertebrates, no conclusive statement was made with regard to the direct effect of these heavy metals (Vuori *et al.* 1998).

The responses of fish to heavy metal contamination are well documented (Jezierska *et al.* 2009; Authman *et al.* 2015;) including Al (Poléo *et al.* 1997; Alwan *et al.* 2009) and Fe (Peuranen *et al.* 1994; Dalzell and Macfarlane, 1999). Few studies have focused specifically on impacts of heavy metals in peatland streams, although aluminium-induced impacts have been reported for streams draining forested peatland streams (e.g. Feeley *et al.* 2013; Kelly-Quinn *et al.* 2016). An experimental study by Laine *et al.* (2001) reported decreased hatching success of *S. trutta* with influxes of peat sediment, as previously mentioned, and associated Fe. Survival was inversely related to Fe concentration, ranging from 0.2–9.1% during pre-emergence. The authors therefore hypothesised that precipitation of Fe from peat-derived organic matter can produce strong negative effects on the survival of salmonid eggs in degraded peatland streams. Other experimental studies have shown that iron leached from degraded soils can damage the gill epithelium of brown trout leading to impaired oxygen uptake and ion regulation (Peuranen *et al.* 1994) while Al mobilised under acidic conditions has been experimentally shown to induce detrimental effects on macroinvertebrates (Vuori 1996) and fish (Buckler *et al.* 1995).

ACIDITY AND IMPACTS ON AQUATIC BIOTA

The acidic conditions associated with peatland streams have been well documented (Kelly-Quinn *et al.* 1996a; Tierney *et al.* 1998; Worrall *et al.* 2003; Åström and Spiro 2005; Haraguchi 2007; Feeley *et al.* 2013; Kelly-Quinn *et al.* 2016; Gandois *et al.* 2020) as well as the effects on macroinvertebrate distribution, and community structure, some of which is related to degraded peatlands. As expected, macroinvertebrate species tolerant of acidic or episodically acidic conditions predominate due to unfavourable conditions for pH-sensitive taxa (e.g. Ephemeroptera) (Tierney *et al.* 1998; Verberk *et al.* 2001; Feeley *et al.* 2011; O'Driscoll *et al.* 2012; O'Driscoll *et al.* 2014) or indirect effects on the food chain (e.g. change is the composition of primary producers) (O'Driscoll 2012; O'Driscoll *et al.* 2014). For example, a study conducted in a Dutch raised bog remnant showed that species composition of aquatic Coleoptera in bog pools was significantly correlated with pH and alkalinity (Verberk *et al.* 2001). *Hydroporus* species were typically associated with acidic water in which *Sphagnum* dominated, as they not only display a tolerance for acidic conditions but also may utilise the mosses for shelter and during spawning. In contrast, beetles of the family Haliplidae were strongly associated with more alkaline sections of the bog remnant. The authors posited that this may be due to higher nutrient availability in less acidic bog pools, leading to a greater abundance of filamentous algae that beetles utilise for food and floating vegetation on which they deposit eggs. It is unclear how taxa in similar conditions in peatland streams in the midlands of Ireland would respond as these streams are not typically as acidic as bog pools or streams draining mountain blanket bog.

The effects of acidification on macroinvertebrates are not limited to alterations in species distribution due to pH conditions outside of their preferred range but can also result in developmental abnormalities, as evidenced in experimental trials utilising translocated Hydropsychidae (Trichoptera) in Finland, during which larvae from uncontaminated sites were relocated to stretches of river affected by acid sulphate runoff (Vuori 1995). The sulphate originated from acid sulphate soils which, according to the author, is released when oxidised following draining and restructuring of the river. The translocated larvae exhibited high levels of mortality and displayed developmental abnormalities, however, tolerance was not uniform, with some species displaying greater plasticity when exposed to acidic runoff (Vuori 1995). While stabilisation in pH resulted in the growth of morphologically normal larvae, this stabilisation was also linked with reductions in heavy metal concentrations, nutrient runoff, and organic material and it is likely that these stressors interacted to produce the observed increase in mortality. Further research has shown that drainage, ditch-digging and the lowering of the water table in areas with acid sulphate soils results in considerable changes in the spatial structure of fish and phytobenthic communities due to the production and runoff of sulfuric acid (H_2SO_4) via oxidation of metal sulphides in the soil that is subsequently leached into surrounding rivers during rainfall events. Suggested mitigation measures include efforts to raise the water table to reduce oxidation related influx of H_2SO_4 (Vehanen *et al.* 2022).

Acidity has also been shown to be an important factor in determining the structure of diatom communities in low alkalinity streams (Kahlert and Andrén 2005; Kovács *et al.* 2006; Andrén and Jarlman 2008) and in a study that examined those in streams draining upland blanket bogs where forestry and sheep grazing were the predominant land-use (O'Driscoll *et al.* 2012). This analysis of diatom assemblages conducted in Co. Mayo in the north-west of Ireland showed that alkalinity and conductivity, rather than nutrient inputs, were the major factors influencing diatom biodiversity in the studied sites (O'Driscoll *et al.* 2012). Anthropogenic derived acidification from forestry and peat drainage, along with increased DOC were considered potentially responsible for the paucity of diatom species less tolerant of low-alkalinity waters.

Further research in the Burrishoole catchment in Co. Mayo, investigated the macroinvertebrate and diatom assemblages in the upper, mid-upper, middle and lower reaches of two rivers draining degraded blanket peat (O'Driscoll *et al.* 2014). The upper reaches drained afforested peatland, while mid-upper and middle directly drained degraded peatlands, and the lower reaches were downstream from drained areas that supported sheep grazing. The macroinvertebrate and diatom communities displayed the expected seasonal shifts associated with temperature and life cycle, but community composition for both macroinvertebrates and diatoms varied along the upstream/downstream gradient. Acid-tolerant diatom species such as *Eunotia rhomboidea* were found in greater abundance at lower pH with an alkalinity associated shift to circumneutral species *Achnanthes oblongella* and *Gomphonema parvulum* further downstream.

The impact of acidity on the distribution of fish in upland streams draining mountain blanket bog in Ireland has been documented where it is related to the severity of episodic acidity and associated increased in inorganic aluminium (Kelly-Quinn *et al.* 1996b and 2016) Acidity-related impacts are not expected to be an issue for salmonids and other aquatic biota in the streams draining drained bogs in the midlands of Ireland. However, this requires further study.

DRAINAGE AND FLOW

Drainage of peatlands induces changes in flow in the peatland itself. Drained peatlands have been shown to have a greater volume of macropores, soil pipes and pipeflow, which alters discharge regimes for baseflow and storm events (Holden *et al.* 2006). Soil degradation following drainage promotes the erosion of peat which is then carried into surrounding peatland streams (Ramchunder *et al.* 2009). The accumulation of this sediment in peatland streams may have the potential to alter the flow of said streams. Though no studies specifically assess the effects of altered flow regimes in rivers arising from peatland drainage on aquatic biota, research on alterations in flow from other land uses can inform how biological communities in stream draining degraded peatlands may be impacted. Profound effects on the structure of aquatic communities and river ecosystem processes have been reported due to direct (e.g. loss of preferred habitat) and indirect (e.g. food webs or changes in water quality) (McKinney *et al.* 1999; Palmer and Ruhi 2019) effects of altered flow and subsequent changes in habitat.

VARIABILITY IN FLOW IMPACTS

Macroinvertebrate communities within streams have been shown to be sensitive to changes in flow regime and in particular flow extremes with species such as *Nemurella pictetii* and *Leuctra nigra* preferentially utilising flow refugia (areas of limited hydraulic stress) during and after periods of high or fluctuating flow (Lancaster and Hildrew 1993). Certain macroinvertebrate taxa, such as Plecoptera and most Heptageniidae, exhibit a preference for fast-flowing, well oxygenated mesohabitats with coarse substrates whereas others with weak swimming ability (e.g. Coleoptera) benefit from reductions in flow (Harper *et al*. 1997; Garcia *et al*. 2011).

As well as influencing habitat heterogeneity, variability in flow can influence species richness. Changes to the natural variability in flow can result in an increase in the uniformity of flow or generation of extremes such as spate conditions. For example, the highest net biomass production and species richness of primary producers (periphyton) is associated with streams displaying highly variable flow regimes rather than streams exhibiting low disturbance flow regimes (Cardinale *et al*. 2005). The authors suggested that higher rates of production in high disturbance streams could be linked to the presence of taxa that generally achieve greater biomass, reflecting differences in species richness between high and low disturbance streams, as well as the possibility that greater species richness may contribute to increased biofilm accumulation.

Uniformity of flow, as can occur in some peatland streams, has been shown to negatively impact trichopteran species diversity (Cardinale *et al*. 2002). Laboratory mesocosm studies utilising three caddisfly species (*Hydropsyche depravata*, *Ceratopsyche bronta* and *Cheumatopsyche* sp.) showed that in substrates not subject to flow related disturbance, 97% of pore spaces in the substrate were occupied, leading to competitive displacement and dominance of the largest species, *H. depravata*. This resulted in low abundance of the two smaller species, while competitive displacement was not observed in streams subject to flow disturbance. The dominant *H. depravata* was also characterised by low rates of nutrient excretion leading to a loss of algal productivity, with potential negative effects on the greater trophic web in streams with uniform flow (Cardinale *et al*. 2002). In contrast, elevated flow has also been associated with the loss of less-well attached taxa of diatoms such as *Epithemia* and *Ctenophora* (O'Driscoll *et al*. 2012).

FLOW AND AQUATIC MACROPHYTES

Though no studies were found that specifically examined flow effects on macrophytes in peatland streams, flow has been found to produce a strong influence on aquatic vegetation in other stream ecosystems. Hydraulic conditions such as current velocity, turbulence and boundary layer shear stress determine factors like particle size distribution, movement of sediment, suspension of FPOM and therefore the growth of phytobenthos and macrophytes (Garcia *et al*. 2011). In fact, the hydraulic resistance of aquatic macrophytes can reduce mean flow velocity across an entire catchment (Sand-Jensen *et al*. 1989). Furthermore, research has shown that flow reduction by dense strands of aquatic macrophytes such as *Callitriche cophocarpa* and *Elodea canadensis* (Sand-Jensen and Mebus 1996; Sand-Jensen 1998) can promote the accumulation of inorganic and organic sediment among the root structures upstream from flow, while accelerating flow around and downstream from macrophyte patches. The increase in sediment deposition can have negative impacts on sediment-sensitive taxa as previously outlined. However, the presence of macrophytes can provide habitat for phytobenthos and macroinvertebrates, especially suspension feeders (Cotton *et al*. 2006). The retention of organic particles by macrophytes and subsequent consumption by attached suspension feeders such as simuliids may play a key part in the downstream distribution of organic matter (dissolved (DOM) and particulate (POM)) and nutrient distribution in these streams owing to the reduced flow patterns created (Cotton *et al*. 2006).

The higher velocity flow areas around and downstream from macrophyte patches create turbulence that can result in erosion of sediment and promotion of coarser substrata (Sand-Jensen 1998), potentially contributing to beneficial habitat heterogeneity for species such as brown trout (Mortensen 1977; Dolinsek et al. 2007). However, research has shown that macrophytes in streams can also be reduced by high-velocity flows, with even small increases in flow velocity resulting in a loss of aquatic macrophyte abundance (Chambers et al. 1991; Madsen et al. 2001). This has been attributed to uprooting (Haslam 1978; Madsen et al. 2001) or to decreased growth due to physical stress (Madsen et al. 2001) as well as a reduction in the ability to photosynthesise (Madsen et al. 1993). In such cases, low-growing bryophytes capable of attachment to rocks may be favoured (French and Chambers 1996; Madsen et al. 2001; Weekes et al. 2014). It is clear that macrophytes, like other aquatic biota, are sensitive to a combination of environmental conditions, either natural or exacerbated by anthropogenic activities such as peatland drainage, a point highlighted by Weekes et al. (2021).

Given the range of flow dependencies and sensitivities displayed by various species, potential alterations to flow from drained peatlands and associated sediment influx are likely to impact the ecological health of streams draining degraded peatlands. As such, research is needed to delineate the response of the aquatic communities in streams in drained peatlands to changes in flow from other co-occurring stressors.

In summary flow is important not only in terms of flow preferences of aquatic biota but it can also influence the delivery of other stressors (e.g. low flows typical of midland bog streams can lead to higher sedimentation, with wider knock-on effects for food webs, local and regional biodiversity etc). This points to the need to consider impacts in what is clearly a multi-stressor environment.

RESPONSES OF AQUATIC BIOTA TO PEATLAND RESTORATION

As noted by Andersen *et al.* (2016) peatland restoration projects often lack baseline data against which to measure change. Though the effects of restoration on bog pool communities are well reported (van Duinen *et al.* 2002; van Duinen *et al.* 2003; Hannigan *et al.* 2011), to date only one study (Ramchunder *et al.* 2012) based on drain blocking has addressed the effects of restoration on peatland stream biota. The authors compared the effects of peatland restoration via drain-blocking (3–11 years post-blocking) on macroinvertebrate community structure with those found at intact and drained sites. Results showed that taxon richness, species composition and community structure in the restored sites resembled intact sites. This was linked to a decrease in FPOM and suspended sediments due to drain-blocking via the creation of a series of dams to reduce outflow. These results suggest that though drainage produces demonstrable alterations to benthic macroinvertebrate assemblages, restoration methods such as drain-blocking are largely capable of reversing these changes (Ramchunder *et al.* 2012).

As noted most other studies are based on sampling bog pools, ditches or areas close to ditches and only a few are mentioned here. For example, field experiments performed in the UK, based on sampling for adult cranefly (Tipulidae) emergence from peat adjacent to blocked and unblocked drains showed a positive relationship between increased soil moisture following drain blockage and cranefly (Tipulidae) abundance (Carroll *et al.* 2011). Though the strength of this correlation was found to vary on a year-to-year basis, drain blocking was found to increase soil moisture by $0.06m^3m^{-3}$ in 2009 and $0.23m^3m^{-3}$ in 2010 and corresponded with a 1.3-fold increase in cranefly abundance in 2009 and 4.5-fold in 2010, respectively. Researchers in Finland assessed the effect of peatland drainage and restoration on the richness and abundance of larval Odonata species (Elo *et al.* 2015). Samples were collected from ditches at the drained sites and from bog pools at the undrained sites. The three samples within each site were taken from a randomly chosen small area of a single ditch or bog pool. The results showed that while no larvae were found in sites a year after restoration, a total of nine species had successfully recolonised after three years. A similar study in Ireland comparing aquatic macroinvertebrate and microcrustacean communities in bog pools at restored and intact bogs showed no significant differences between richness, abundance or composition fifteen years following restoration (Hannigan *et al.* 2011). However, a study by Verberk *et al.* (2010) using a Before-After-Control-Impact (BACI) design involved large-scale rewetting, achieved by drain blocking and construction of dams to retain rainwater, found a reduction in species richness within rewetted compartments, which they attributed to abrupt changes and homogenisation in environmental conditions as well as a decreased groundwater influence. A similar result was reported by Van Duinan *et al.* (2003) and the role of landscape heterogeneity was highlighted by Verberk et al. (2006). Thus, responses to the restoration strategies will depend on the methods used and their scale as well the time post restoration, factors that are also likely to influence responses of stream fauna.

Clearly further research relating to impact on stream biota is required especially in countries where there is an extensive natural network of streams draining or in close contact with peatlands. Furthermore, in terms of ecosystem services provision Parry *et al.* (2014) noted the paucity of research on the responses to different types of restoration that can inform restoration strategies.

KNOWLEDGE GAPS AND RESEARCH NEEDS

Despite the fact that 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 and particularly for phytobenthos, macrophytes and fish. While only published research was consulted for this review it may be useful to source the grey literature and other unpublished information to not only supplement information on the impact of peatland-derived stressors but also to help establish reference conditions for peatland streams. A second option is to collect baseline aquatic biota data from streams draining natural or minimally impacted peatlands and to make comparisons with biotic communities from streams draining degraded peatlands, a task which is being undertaken by the EPA-funded SWAMP project (www.ucd.ie/swamp).

Although the literature regarding the effects of sedimentation for both organic and inorganic particles is well detailed in other land uses associated with peatlands (Wood and Armitage 1997; Julien and Bergeron 2006; Jones *et al.* 2011a, 2011b; Jones *et al.* 2012), it cannot be assumed that organic particles arising from peat drainage and extraction will produce the same or similar effects on aquatic biota.

Research performed on the effects of forestry and agriculture on peaty soils is likely to provide some insight into the effects of this stressor (Ramchunder *et al.* 2011; Kelly-Quinn *et al.* 2016) but more comprehensive research specific to the effects of peat-derived sediment and relevant thresholds is required. The paucity of studies on effects of peat sediment on phytobenthos also needs to be addressed. Phytobenthos are monitored under WFD as a biological element, so research on the impacts of peat sedimentation in midland and lowland rivers would be advantageous.

Of additional concern, is how organic sediment arising from peat extraction interacts with other known stressors such as flow alterations, pH changes and heavy metals associated with peat extraction, and the type of effect these interactions produce as well as whether the organic nature of the peat sediment has a specific role to play in the reported impacts. For example, Aspray *et al.* (2017) showed that increasing organic sedimentation also resulted in higher concentrations of DOC, total organic nitrogen (TON) and suspended solids (SS), however, these changes were not examined in relation to the impacts detected on the aquatic biota nor did they distinguish these from the physical effects of the peat sediment (e.g. smothering). Additionally, changes in biotic response to peat sediment across catchment size should be considered as communities in intact peatland streams have been shown to naturally vary according to stream order (Ramchunder *et al.* 2011). For example, it has been shown that community composition of diatoms varies with alkalinity, stream order and catchment area (O'Driscoll *et al.* 2012), which further challenges efforts to identify impacts and clearly highlights the need for additional work on reference streams to characterise community structure in the absence of significant anthropogenic-derived stressors. This includes consideration of the role of landscape heterogeneity for conservation of biodiversity in these areas (Verbert *et al.* 2006).

A further challenge is identification of metrics that can effectively detect biologically impacted peatland streams. It is not known whether available metrics such as Q-value, Small Stream Risk Score (McGarrigle 2014) and EPT calculations can effectively detect community differences between streams draining natural and degraded bogs. This knowledge gap is being considered by the SWAMP project.

The issue of peat mining drainage water has been known and investigated for many decades in Scandinavia and Canada where water pollution control methods have been developed. In Ireland, Bord na Móna (BnM) was the only company that was subject to EPA licencing. The EPA BATNEEC Guidance Note Class 1.4 Extraction of Peat (1996, updated in 2006) sets emissions limit values for a range of pollutants in drainage water. However there has never been any scientific research into how these limit values are affected by specific Irish conditions such as peatland types and climatic conditions (e.g. no snow cover in the winter). In addition, no action has been taken to address water pollution from smaller private peat companies and bogs affected by turf-cutting, which represent an even greater land surface area (Renou-Wilson *et al.* 2022). Addressing the pollution from drained bogs will be largely addressed by restoration and rehabilitation projects that affect not only BnM bogs but also privately owned bogs.

Overall, this review identified considerable knowledge gaps that need to be addressed and that a multi-stressor perspective needs to be taken when researching the potential impact on aquatic biota of stressor inputs from extracted and degraded peatlands. Furthermore, adoption of measures to mitigate such impacts and restore peatlands first requires an understanding of the processes delivering the stressors but also identification of the dominant stressors and the key mechanisms of impacts on aquatic biota. Finally, improved and more spatially and temporally extensive post-restoration assessments of affected streams need to be undertaken to ensure that the works undertaken are yielding the desired ecological outcomes and inform works into the future.

REFERENCES

Alonso A. 2005 Valoración de la degradación ambiental y efectos ecotoxicológicos sobre la comunidad de macroinvertebrados bentónicos en la cabecera del río Henares. Doctoral Dissertation, Universidad de Alcalá, Alcalá de Henares (Madrid).

Alwan, S.F., Hadi, A.A. and Shokr, A.E. 2009 Alterations in haematological parameters of fresh water fish, *Tilapia zillii*, exposed to aluminum. *Journal of Science and Its Applications* **3**, 12–9.

Andersen, R., Farrell, C., Graf, M., Muller, F., Calvar, E., Frankard, P., Caporn, S. and Anderson, P. 2016 An overview of the progress and challenges of peatland restoration in Western Europe. *Restoration Ecology* **25**(2), 271–82.

Andrén, C. and Jarlman, A. 2008 Benthic diatoms as indicators of acidity in streams. *Fundamental and Applied Limnology / Archiv für Hydrobiologie* **173**(3), 237–53.

Arillo, A., Gaino, E., Margiocco, C., Mensi, P. and Schenone, G. 1984 Biochemical and ultrastructural effects of nitrite in rainbow trout: Liver hypoxia as the root of the acute toxicity mechanism. *Environmental Research* **34**(1), 135–54.

Armstrong, A., Holden, J., Kay, P., Francis, B., Foulger, M., Gledhill, S., McDonald, A. and Walker, A. 2010 The impact of peatland drain-blocking on dissolved organic carbon loss and discolouration of water; results from a national survey. *Journal of Hydrology* **381**, 112–20.

Aspray, K., Holden, J., Ledger, M., Mainstone, C. and Brown, L. 2017 Organic sediment pulses impact rivers across multiple levels of ecological organization. *Ecohydrology* **10**(6), e 1855.

Åström, M. and Spiro, B. 2005 Sources of acidity and metals in a stream draining acid sulphate soil, till and peat, western Finland, revealed by a hydrochemical and sulphur isotope study. *Agricultural and Food Science* **14**(1), 34–43.

Authman, M.M.M., Zaki, M.S., Khallaf, E.A. and Abbas, H.H. 2015 Use of fish as bio-indicator of the effects of heavy metals pollution. *Journal of Aquaculture Research and Development* **06**(04), 1–13.

Berenzen, N., Schulz, R. and Liess, M. 2001 Effects of chronic ammonium and nitrite contamination on the macroinvertebrate community in running water microcosms. *Water Research* **35**(14) 3478–82.

Bragg, O. and Lindsay, R. 2003 *Strategy and action plan for mire and peatlands conservation in Europe*. Wageningen, The Netherlands. Wetlands International.

Brown, L., Aspray, K., Ledger, M., Mainstone, C., Palmer, S., Wilkes, M. and Holden, J. 2018 Sediment deposition from eroding peatlands alters headwater invertebrate biodiversity. *Global Change Biology* **25**(2) 602–19.

Buckler, D., Cleveland, L., Little, E. and Brumbaugh, W. 1995 Survival, sublethal responses, and tissue residues of Atlantic salmon exposed to acidic pH and aluminium. *Aquatic Toxicology* **31**(3), 203–16.

Camargo, J. and Alonso, Â. 2006 Ecological and toxicological effects of inorganic nitrogen pollution in aquatic ecosystems: A global assessment. *Environment International* **32**(6), 831–49.

Cardinale, B.J., and Palmer, M.A. 2002 Disturbance moderates biodiversity–ecosystem function relationships: experimental evidence from caddisflies in stream mesocosms. *Ecology* **83**, 1915–27.

Cardinale, B., Palmer, M., Ives, A. and Brooks, S. 2005 Diversity-productivity relationships in streams vary as a function of the natural disturbance regime. *Ecology* **86**(3), 716–26.

Carroll, M.J., Dennis, P., Pearce-Higgins, J.W. and Thomas, C.D. 2011 Maintaining northern peatland ecosystems in a changing climate: effects of soil moisture, drainage and drain blocking on craneflies. *Global Change Biology* **17**, 2991–3001.

Chambers, P., Prepas, E., Hamilton, H. and Bothwell, M. 1991 Current velocity and its effect on aquatic macrophytes in flowing waters. *Ecological Applications* **1**(3), 249–57.

Conroy, E., Turner, J.N., Rymszewicz, A., Bruen, M., O'Sullivan, J.J., Lawler, D.M., Lally, H. and Kelly-Quinn, M. 2016 Evaluating the relationship between biotic and sediment metrics using mesocosms and field studies. *Science of the Total Environment* **568,** 1092–101.

Cotton, J., Wharton, G., Bass, J., Heppell, C. and Wotton, R. 2006 The effects of seasonal changes to in-stream vegetation cover on patterns of flow and accumulation of sediment. *Geomorphology* **77**, 320–34.

Dalzell, D. and Macfarlane, N. 1999 The toxicity of iron to brown trout and effects on the gills: a comparison of two grades of iron sulphate. *Journal of Fish Biology* **55**(2), 301–15.

Denic, G. and Geist, J. 2015 Linking stream sediment deposition and aquatic habitat quality in pearl mussel streams: implications for conservation. *River Research and Applications* **31**(8). 943–52.

Dolinsek, I.J., Grant, J.W.J. and Biron, P.M. 2007 The effect of habitat heterogeneity on the population density of juvenile Atlantic salmon *Salmo salar* (L). *Journal of Fish Biology* **70**, 206–14.

Drinan, T., Foster, G., Nelson, B., O'Halloran, J. and Harrison, S. 2013 Macroinvertebrate assemblages of peatland lakes: Assessment of conservation value with respect to anthropogenic land-cover change. *Biological Conservation* **158**, 175–87.

Eddy, F. 2005 Ammonia in estuaries and effects on fish. *Journal of Fish Biology* **67**(6), 1495–513.

Elo, M., Penttinen, J. and Kotiaho, J. 2015 The effect of peatland drainage and restoration on Odonata species richness and abundance. *BMC Ecology* **15**(1), 11–18.

EPA 2020 *Ireland's Environment - An Integrated Assessment 2020.* Johnstown's Castle, Ireland. Environmental Protection Agency.

Evans, C., Renou-Wilson, F. and Strack, M. 2016 The role of waterborne carbon in the greenhouse gas balance of drained and re-wetted peatlands. *Aquatic Sciences* **78**, 573–90.

Extence, C.A., Chadd, R.P., England, J., Dunbar, M.J., Wood, P.J. and Taylor, E.D. 2013 The assessment of fine sediment accumulation in rivers using macroinvertebrate community response. *River Research and Applications* **29**(1), 17–55.

Feehan, J., O'Donovan, G., Renou-Wilson, F. and Wilson, D. 2008 *The bogs of Ireland - An introduction to the natural, cultural and industrial heritage of Irish peatlands.* Second Edition, Digital Format. Dublin. University College Dublin.

Feeley, H., Kerrigan, C., Fanning, P., Hannigan, E. and Kelly-Quinn, M. 2011 Longitudinal extent of acidification effects of plantation forest on benthic macroinvertebrate communities in soft water streams: evidence for localised impact and temporal ecological recovery. *Hydrobiologia* **671**(1), 217–26.

Feeley, H., Bruen, M., Blacklocke, S. and Kelly-Quinn, M. 2013 A regional examination of episodic acidification response to reduced acidic deposition and the influence of plantation forests in Irish headwater streams. *Science of the Total Environment* **443**, 173–83.

French, T. and Chambers, P. 1996 Habitat partitioning in riverine macrophyte communities. *Freshwater Biology* **36**(3), 509–20.

Gandois, L., Hoyt, A., Mounier, S., Le Roux, G., Harvey, C., Claustres, A., Nuriman, M. and Anshari, G. 2020 From canals to the coast: dissolved organic matter and trace metal composition in rivers draining degraded tropical peatlands in Indonesia. *Biogeosciences* **17**(7), 1897–909.

Garcia, X.F., Schnauder, I. and Pusch, M. 2011 Complex hydromorphology of meanders can support benthic invertebrate diversity in rivers. *Hydrobiologia* **685**(1), 49–68.

Government of Ireland 2018 *River basin management plan for Ireland 2018–2021 - SEA*: Dublin.

Hammond, R.F. 1981 *The peatlands of Ireland. Soil Survey Bulletin No. 35.* Dublin, Ireland. An Foras Taluntais.

Hannigan, E., Mangan, R. and Kelly-Quinn, M. 2011 Evaluation of the success of mountain blanket bog pool restoration in terms of aquatic macroinvertebrates. *Biology and Environment: Proceedings of the Royal Irish Academy* **111**(2), 1–11.

Haraguchi, A. 2007 Effect of sulfuric acid discharge on river water chemistry in peat swamp forests in central Kalimantan, Indonesia. *Limnology* **8**(2), 175–82.

Harding, J.S. 2005 Impacts of metals and mining on stream communities. In T.A. Moore, A. Black, J.A. Centeno, J.S. Harding, D.A. Trumm (eds) *Metal contaminants in New Zealand*, 343–57. Christchurch, NZ. Resolutionz press.

Harper, D., Mekotova, J., Hulme, S., White, J. and Hall, J. 1997 Habitat heterogeneity and aquatic invertebrate diversity in floodplain forests. *Global Ecology and Biogeography Letters* **6**, 275–85.

Haslam, S.M. 1978 *River plants: The macrophytic vegetation of watercourses*. Cambridge. Cambridge University Press.

Hickey, C. and Vickers, M. 1994 Toxicity of ammonia to nine native New Zealand freshwater invertebrate species. *Archives of Environmental Contamination and Toxicology* **26**(3), 292–8.

Hickey, C., Golding, L., Martin, M. and Croker, G. 1999 Chronic toxicity of ammonia to New Zealand freshwater invertebrates: A mesocosm study. *Archives of Environmental Contamination and Toxicology* **37**(3), 338–51.

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

Holden, J., Evans, M. G., Burt, T. P., & Horton, M. 2006 Impact of land drainage on peatland hydrology. *Journal of Environmental Quality*, **35**(5), 1764–1778

Holden, J., Kirkby, M., Lane, S., Milledge, D., Brookes, C., Holden, V. and McDonald, A. 2008 Overland flow velocity and roughness properties in peatlands. *Water Resources Research* **44(**6), W06415.

Jensen, F. 2003 Nitrite disrupts multiple physiological functions in aquatic animals. *Comparative Biochemistry and Physiology Part A: Molecular and Integrative Physiology* **135**(1), 9–24.

Jeremiason, J., Baumann, E., Sebestyen, S., Agather, A., Seelen, E., Carlson-Stehlin, B., Funke, M. and Cotner, J. 2018 Contemporary mobilization of legacy Pb stores by DOM in a boreal peatland. *Environmental Science and Technology* **52**(6), 3375–83.

Jezierska, B., Ługowska, K. and Witeska, M. 2009 The effects of heavy metals on embryonic development of fish (a review). *Fish Physiology and Biochemistry* **35**(4), 625–40.

Jones, J., Collins, A., Naden, P. and Sear, D. 2011a The relationship between fine sediment and macrophytes in rivers. *River Research and Applications* **28**(7), 1006–18.

Jones, J., Murphy, J., Collins, A., Sear, D., Naden, P. and Armitage, P. 2011b The impact of fine sediment on macroinvertebrates. *River Research and Applications* **28**(8), 1055–71.

Jones, J., Duerdoth, C., Collins, A., Naden, P. and Sear, D. 2012 Interactions between diatoms and fine sediment. *Hydrological Processes* **28**(3), 1226–37.

Julien, H. and Bergeron, N. 2006 Effect of fine sediment infiltration during the incubation period on Atlantic salmon (*Salmo salar*) embryo survival. *Hydrobiologia* **563**(1), 61–71.

Kahlert, M. and Andrén, C. 2005 Benthic diatoms as valuable indicators of acidity. *SIL Proceedings 1922–2010* **29**(2), 635–39.

Kelly-Quinn, M., Tierney, D., Coyle, C. and J.J. Bracken, J.J. 1996a Factors affecting the susceptibility of Irish soft-water streams to forest-mediated acidification. *Fisheries Management and Ecology* **3**, 287–301.

Kelly-Quinn, M., Tierney, D., Roche, W. and Bracken, J.J. 1996b Distribution and abundance of trout populations in moorland and afforested upland nursery streams in County Wicklow. *Biology and Environment: Proceedings of the Royal Irish Academy* **96**B, 127–39.

Kelly-Quinn, M., Bruen, M., Harrrison, S., Healy, M., Clarke, J., Drinan, T., Feeley, H., Finnegan, J., Graham, C., Regan, J. and Blacklocke, S. 2016 *Assessment of the Impacts of Forest Operations on the Ecological Quality of Water (HYDROFOR).* Report No.16. Johnstown Castle, Co. Wexford. Environmental Protection Agency.

Kløve, B. 2001 Characteristics of nitrogen and phosphorus loads in peat mining wastewater. *Water Research* **35**(10), 2353–62.

Kløve, B., Sveistrup, T.E. and Hauge, A. 2010 Leaching of nutrients and emission of greenhouse gases from peatland cultivation at Bodin, Northern Norway. *Geoderma* **154**, 219–32.

Kovács, C., Kahlert, M. and Padisák, J. 2006 Benthic diatom communities along pH and TP Gradients in Hungarian and Swedish Streams. *Journal of Applied Phycology* **18**(2), 105–17.

Krachler, R., Krachler, R.F., von der Kammer, F., Süphandag, A., Jirsa, F., Ayromlou, S., Hofmann, T. and Keppler, B. 2010 Relevance of peat-draining rivers for the riverine input of dissolved iron into the ocean. *Science of the Total Environment* **408**(11), 2402–8.

Krachler, R., von der Kammer, F., Jirsa, F., Süphandag, A., Krachler, R.F., Plessl, C., Vogt, M., Keppler, B.K., Hofmann, T. 2012 Nanoscale lignin particles as sources of dissolved iron to the ocean. *Global Biogeochemical Cycles* **26**(3), GB3024.

Kroupova, H., Machova, J., Piackova, V., Blahova, J., Dobsikova, R., Novotny, L. and Svobodova, Z. 2008 Effects of subchronic nitrite exposure on rainbow trout (*Oncorhynchus mykiss*). *Ecotoxicology and Environmental Safety* **71**(3), 813–20.

Laine, A. 2001 Effects of peatland drainage on the size and diet of yearling salmon in a humic northern river. *Fundamental and Applied Limnology* **151**(1), 83–99.

Laine, A., Heikkinen, K. and Sutela, T. 2001 Incubation success of brown trout (*Salmo trutta*) eggs in boreal humic rivers affected by peatland drainage. *Fundamental and Applied Limnology* **150**(2), 289–305.

Lancaster, J. and Hildrew, A. 1993 Flow refugia and the microdistribution of lotic macroinvertebrates. *Journal of the North American Benthological Society* 12(4), 385–93.

Larsen, S. and Ormerod, S. 2010 Low-level effects of inert sediments on temperate stream invertebrates. *Freshwater Biology* **55**(2), 476–86.

Lopes-Lima, M., Sousa, R., Geist, J., Aldridge, D.C., Araujo, R., Bergengren, J., Bespalaya, Y., Bódis, E., Burlakova, L., Van Damme, D., Douda, K., Froufe, E., Georgiev, D., Gumpinger, C., Karatayev, A., Kebapçi, Ü., Killeen, I., Lajtner, J., Larsen, B.M., Lauceri, R., Legakis, A., Lois, S., Lundberg, S., Moorkens, E., Motte, G., Nagel, K.-O., Ondina, P., Outeiro, A., Paunovic, M., Prié, V., von Proschwitz, T., Riccardi, N., Rudzīte, M., Rudzītis, M., Scheder, C., Seddon, M., Şereflişan, H., Simić, V., Sokolova, S., Stoeckl, K., Taskinen, J., Teixeira, A., Thielen, F., Trichkova, T., Varandas, S., Vicentini, H., Zajac, K., Zajac T. and Zogaris, S. 2017 Conservation status of freshwater mussels in Europe: state of the art and future challenges. *Biological Reviews* **92,** 572–607.

Madsen, T., Sand-Jensen, K. and Beer, S. 1993 Comparison of photosynthetic performance and carboxylation capacity in a range of aquatic macrophytes of different growth forms. *Aquatic Botany* **44**(4), 373–84.

Madsen, J.D., Chambers, P.A., James, W.F., Koch, E.W. and Westlake, D.F. 2001 The interaction between water movement, sediment dynamics and submersed macrophytes. *Hydrobiologia* **444**, 71–84.

Mallet, A., Carver, C. and Daigle, J. 2005 The effect of peat moss particles on the physiology and survival of the American oyster, *Crassostrea virginica*. *Journal of Shellfish Research* **24**(1), 113–19.

McGarrigle, M. 2014 Assessment of small water bodies in Ireland. *Biology and Environment: Proceedings of the Royal Irish Academy* **14**B(3), 119–28.

McKinney, T., Rogers, R. and Persons, W. 1999 Effects of flow reductions on aquatic biota of the Colorado River below Glen Canyon Dam, Arizona. *North American Journal of Fisheries Management* **19**(4), 984–91.

Mortensen, E. 1977 Density-dependent mortality of trout fry (*Salmo trutta* L.) and its relationship to the management of small streams. *Journal of Fish Biology* **11**(6), 613–7.

Mummert, A., Neves, R., Newcomb, T. and Cherry, D. 2003 Sensitivity of juvenile freshwater mussels *(Lampsilis fasciola, Villosa iris*) to total and un-ionized ammonia. *Environmental Toxicology and Chemistry* **22**(11), 2545–53.

Nieminen, M., Palviainen, M., Sarkkola, S., Laurén, A., Marttila, H. and Finér, L. 2017a A synthesis of the impacts of ditch network maintenance on the quantity and quality of runoff from drained boreal peatland forests. *Ambio* **47**(5), 523–34.

Nieminen, M., Sallantaus, T., Ukonmaanaho, L., Nieminen, T. and Sarkkola, S. 2017b Nitrogen and phosphorus concentrations in discharge from drained peatland forests are increasing. *Science of The Total Environment* **609**, 974–81.

O'Driscoll, C., de Eyto, E., Rodgers, M., O'Connor, M., Asam, Z., Kelly, M. and Xiao, L. 2014 Spatial and seasonal variation of peatland-fed riverine macroinvertebrate and benthic diatom assemblages and implications for assessment: a case study from Ireland. *Hydrobiologia* **728**(1), 67–87.

O'Driscoll, C., de Eyto, E., Rodgers, M., O'Connor, M., Asam, Z. and Xiao, L. 2012 Diatom assemblages and their associated environmental factors in upland peat forest rivers. *Ecological Indicators* **18**, 443–51.

Olsson, T. and Persson, B. 1986 Effects of gravel size and peat material concentrations on embryo survival and alevin emergence of brown trout, *Salmo trutta* L. *Hydrobiologia* **135**, 9–14.

Ouellette, C., Boghen, A., Courtenay, S. and St-Hilaire, A. 2003 Influence of peat substrate on the distribution and behaviour patterns of sand shrimp, *Crangon septemspinosa*, under experimental conditions. *Journal of Applied Ichthyology* **19**(6), 359–65.

Ouellette, C., Courtenay, S., St-Hilaire, A. and Boghen, A. 2006 Impact of peat moss released by a commercial harvesting operation into an estuarine environment on the sand shrimp *Crangon septemspinosa*. *Journal of Applied Ichthyology* 22(1), 15–24.

Palmer, M. and Ruhi, A. 2019 Linkages between flow regime, biota, and ecosystem processes: Implications for river restoration. *Science* **365**(6459), eaaw2087.

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

Peuranen, S., Vuorinen, P.J., Vuorinen, M. and Hollender, A. 1994 The effects of iron, humic acids and low pH on the gills and physiology of Brown Trout (*Salmo trutta*). *Annales Zoologica Fennici* **31**, 389–96.

Poléo, A., Østbye, K., Øxnevad, S., Andersen, R., Heibo, E. and Vøllestad, L. 1997 Toxicity of acid aluminium-rich water to seven freshwater fish species: A comparative laboratory study. *Environmental Pollution* **96**(2), 129–39.

Ramchunder, S., Brown, L. and Holden, J. 2009 Environmental effects of drainage, drain-blocking and prescribed vegetation burning in UK upland peatlands. *Progress in Physical Geography* **33**(1), 49–79.

Ramchunder, S., Brown, L., Holden, J. and Langton, R. 2011 Spatial and seasonal variability of peatland stream ecosystems. *Ecohydrology* **4**(4), 577–88.

Ramchunder, S., Brown, L. and Holden, J. 2012 Catchment-scale peatland restoration benefits stream ecosystem biodiversity. *Journal of Applied Ecology* **49**(1), 182–91.

Randall, D. and Tsui, T. 2002 Ammonia toxicity in fish. *Marine Pollution Bulletin*, **45**, 17–23.

Renou-Wilson, F., Keane, M., McNally, G., O'Sullivan, J., Farrell, E.P. 2008 Developing a forest resource on industrial cutaway peatland. The BOGFOR programme. Dublin. National Council for Forest Research and Development (COFORD).

Renou-Wilson, F., Bolger, T., Bullock, C., Convery, F., Curry, J.P., S. Ward, Wilson, D. and Müller, C. 2011 BOGLAND: A protocol for the sustainable management of Irish peatlands. STRIVE Report No **76** prepared for the Environmental Protection Agency (EPA), Johnstown Castle, Co. Wexford.

Renou-Wilson, F. 2018 Peatlands. In R. Creamer and L. O'Sullivan (eds), *The Soils of Ireland*, 141–52. Cham. Springer International Publishing.

Renou-Wilson, F., Byrne, K.A., Flynn, R. Premrov, A., Riondato, E., Saunders, M., Walz, K., and Wilson, D. 2022 Peatland properties influencing greenhouse gas emissions and removal. Research Report No. 401 Environmental Protection Agency, Wexford.

Riebe, C., Sklar, L., Overstreet, B. and Wooster, J. 2014 Optimal reproduction in salmon spawning substrates linked to grain size and fish length. *Water Resources Research* **50**(2), 898–918.

Roberts, B.J., Mulholland, P.J. and Hill, H.R. 2007 Multiple scales of temporal variability in ecosystem metabolism rates: Results from 2 years of continuous monitoring in a forested headwater stream. *Ecosystems* **10**, 588–606.

Rothwell, J., Evans, M., Daniels, S. and Allott, T. 2008 Peat soils as a source of lead contamination to upland fluvial systems. *Environmental Pollution* **153**(3), 582–9.

Rydin, H. and Jeglum, J. 2006 *The biology of peatlands*. Second edition. Oxford. Oxford University.

Sadler, K. 1981 The toxicity of ammonia to the European eel (*Anguilla anguilla* L.). *Aquaculture* **26**, 173–81.

Sand-Jensen, K., Jeppesen, E., Nielsen, K., Bijl, L., Hjermind, L., Nielsen, L. and Iversen, T. 1989 Growth of macrophytes and ecosystem consequences in a lowland Danish stream. *Freshwater Biology* **22**(1), 15–32.

Sand-Jensen, K. and Mebus, J. 1996 Fine-Scale Patterns of Water Velocity within Macrophyte Patches in Streams. *Oikos* **76**(1), 169–80.

Sand-Jensen, K. 1998 Influence of submerged macrophytes on sediment composition and near-bed flow in lowland streams. *Freshwater Biology* **39**(4), 663–79.

Schouten, M.G.C. 2002 *Conservation and restoration of raised bogs: Geological, hydrological and ecological studies.* The Netherlands. Department of the Environmental and Local Government. Staatsboshbeheer.

Smith, V. 2003 Eutrophication of freshwater and coastal marine ecosystems a global problem. *Environmental Science and Pollution Research* **10**(2), 126–39.

Solbé, J.F. and Shurben. D.G. 1989 Toxicity of ammonia to early life stages of rainbow trout (*Salmo gairdneri*). *Water Research* **23**, 127–9.

Strychar, K.B. 1997 Peat particles, their associated microbial assemblages, and potential influence on feeding and absorption rates of the eastern oyster *Crassostrea virginica* (Gemlin, 1791). Unpublished MSc thesis, University of New Brunswick, Saint-John, NB, Canada.

Surette, C., Brun, G. and Mallet, V. 2002 Impact of a commercial peat moss operation on water quality and biota in a small tributary of the Richibucto River, Kent County, New Brunswick, Canada. *Archives of Environmental Contamination and Toxicology* **42**(4), 423–30.

Suttle, K., Power, M., Levine, J. and McNeely, C. 2004 How fine sediment in riverbeds impairs growth and survival of juvenile salmonids. *Ecological Applications* **14**(4), 969–74.

Swindles, G.T., Morris, P.J., Mullan, D.J., Payne, R.J., *et al.* 2019 The peatland map of Europe. *Mires and Peat* **19**, 1–17.

Tierney, D., Kelly-Quinn, M. and Bracken, J.J. 1998 The faunal communities of upland streams in the eastern region of Ireland with reference to afforestation effects. *Hydrobiologia* **389,** 115–30.

van Duinen, G.A., Brock, A.M.T., Kuper, J.T., Peeters, T.M.J., Smits, M.J.A., Verberk, W.C.E.P. and Esselink, H. 2002 Important keys to successful restoration of characteristic aquatic macroinvertebrate fauna of raised bogs. In G. Schmilewski and L. Rochefort (eds.), *Proceedings of the international peat symposium—Peat in horticulture—Quality and environmental changes,* 292–302.

van Duinen, G., Brock, A., Kuper, J., Leuven, R., Peeters, T., Roelofs, J., van der Velde, G., Verberk, W. and Esselink, H. 2003 Do restoration measures rehabilitate fauna diversity in raised bogs? A comparative study on aquatic macroinvertebrates. *Wetlands Ecology and Management* **11**(6), 447–59.

van Duinen, G., Zhuge, Y., Verberk, W., Brock, A., van Kleef, H., Leuven, R., van der Velde, G. and Esselink, H. 2006 Effects of rewetting measures in Dutch raised bog remnants on assemblages of aquatic rotifera and microcrustaceans. *Hydrobiologia* **565**(1), 187–200.

Vassiljev, A. and Blinova, I. 2012 The influence of drained peat soils on diffuse nitrogen pollution of surface water. *Hydrology Research* **43**(4), 352–8.

Vehanen, T., Sutela, T., Aroviita, J., Karjalainen, S.M., Riihimäki, J., Larsson, A., Vuori, K.M. 2022 Land use in acid sulphate soils degrades river water quality – Do the biological quality metrics respond? *Ecological Indicators* **141**,109085

Verberk, W.C.E.P., van Duinen, G.A., Peeters, T.M.J. and Esselink, H. 2001 Importance of variation in water types for water beetle fauna (Coleoptera) in Korenburgerveen, a bog remnant in The Netherlands. *Proceedings of the Experimental and Applied Entomology Section of the Netherlands Entomological Society* **12**, 121–8.

Verberk, W., van Duinen, G., Brock, A., Leuven, R., Siepel, H., Verdonschot, P., van der Velde, G. and Esselink, H. 2006 Importance of landscape heterogeneity for the conservation of aquatic macroinvertebrate diversity in bog landscapes. *Journal for Nature Conservation* **14**(2), 78–90.

Verberk, W.C.E.P., Leuven, R.S.E.W., Van Duinen, G.A. and Esselink, H. 2010 Loss of environmental heterogeneity and aquatic macroinvertebrate diversity following large-scale restoration management. *Basic and Applied Ecology* **11**(5), 440–9.

Vuori, K. 1995 Species- and population-specific responses of translocated hydropsychid larvae (Trichoptera, Hydropsychidae) to runoff from acid sulphate soils in tin River Kyronjoki, western Finland. *Freshwater Biology* **33**(2), 305–18.

Vuori, K. 1996 Acid-induced acute toxicity of aluminium to three species of filter feeding caddis larvae (Trichoptera, Arctopsychidae and Hydropsychidae). *Freshwater Biology* **35**(1), 179–88.

Vuori, K., Joensuu, I., Latvala, J., Jutila, E. and Ahvonen, A. 1998 Forest drainage: a threat to benthic biodiversity of boreal headwater streams? *Aquatic Conservation: Marine and Freshwater Ecosystems* **8**(6), 745–59.

Wang, N., Ingersoll, C., Greer, I., Hardesty, D., Ivey, C., Kunz, J., Brumbaugh, W., Dwyer, F., Roberts, A., Augspurger, T., Kane, C., Neves, R. and Barnhart, M. 2007 Chronic toxicity of copper and ammonia to juvenile freshwater mussels (Unionidae). *Environmental Toxicology and Chemistry* **26**(10), 2048–56.

Wang, Z. and Leung, K. 2015 Effects of unionised ammonia on tropical freshwater organisms: Implications on temperate-to-tropic extrapolation and water quality guidelines. *Environmental Pollution* **205**, 240–24.

Weekes, L., FitzPatrick, Ú. and Kelly-Quinn, M. 2021 Assessment of the efficiency of river macrophytes to detect water-column nutrient levels and other environmental conditions in Irish rivers. Hydrobiologia **848**(11), 2797–814.

Weekes, L., FitzPatrick, Ú. and Kelly-Quinn, M. 2014 . Composition and characteristics of macrophytes of macrophytes in small streams in Ireland. Biology & Environmnet, Proceedings of the Royal Irish Academy 114B, 3, 163-180.Wicks, B., Joensen, R., Tang, Q. and Randall, D. 2002 Swimming and ammonia toxicity in salmonids: the effect of sub lethal ammonia exposure on the swimming performance of coho salmon and the acute toxicity of ammonia in swimming and resting rainbow trout. *Aquatic Toxicology* **59**(1–2), 55–69.

Wilson, D., Müller, C. and Renou-Wilson, F. 2013 Carbon emissions and removals from Irish peatlands: present

trends and future mitigation measures. *Irish Geography* **46**(1–2), 1–23.
Wind-Mulder, H., Rochefort, L. and Vitt, D. 1996 Water and peat chemistry comparisons of natural and post-harvested peatlands across Canada and their relevance to peatland restoration. *Ecological Engineering* **7**(3), 161–81.
Wipfli, M., Richardson, J. and Naiman, R. 2007 Ecological linkages between headwaters and downstream ecosystems: transport of organic matter, invertebrates, and wood down headwater channels. *Journal of the American Water Resources Association* **43**(1), 72–85.
Wood, P. and Armitage, P. 1997 Biological effects of fine sediment in the lotic environment. *Environmental Management* **21**(2), 203–17.
Wood, P., Toone, J., Greenwood, M. and Armitage, P. 2005 The response of four lotic macroinvertebrate taxa to burial by sediments. *Archiv für Hydrobiologie* **163**(2), 145–62.
Worrall, F., Burt, T. and Adamson, J. 2003 Controls on the chemistry of runoff from an upland peat catchment. *Hydrological Processes* **17**(10), 2063–83.
Yeloff, D., and Mauquoy, D. 2006 The influence of vegetation composition on peat humification: implications for palaeoclimatic studies. *Boreas* **35**(4), 662–73.
Xu, J., Morris, P., Liu, J. and Holden, J. 2018 PEATMAP: Refining estimates of global peatland distribution based on a meta-analysis. *CATENA* **160**, 134–40.