Response of Tipulidae (Diptera) populations to grip-blocking at a low altitude blanket peatland: implications for ecosystem resilience in the context of climate change

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The written account that follows is original in form and content and may be made available for photocopying and for inter-library loan

Contents

Abstract

Grip-blocking is an extensively deployed yet understudied peatland restoration technique that involves blocking artificial drainage channels (grips) in order to raise water-table levels. Tipulids [Craneflies] represent a functionally important species of peatland ecosystems and constitute an essential prey item for breeding waders. As larvae, peatland tipulids require year-round wet peat, are many species are extremely vulnerable to desiccation in drought. Grip blocking has been identified as a potentially vital mitigation strategy for breeding wader conservation in the context of climate change by increasing tipulid abundance.

Using emergence traps, this research compared tipulid abundance within peatland soils adjacent to blocked and unblocked grips, together with soil moisture and vegetation data. Uniquely, the study also compared tipulid abundance, community composition and relative biomass at grips which have been blocked for varying lengths of time.

Significant covariations were identified between soil moisture, vegetation communities and tipulid community compositions. Larger tipulids species were positively associated with high soil moisture, which has important consequencs for prey biomass for predatory birds, in addition to the potential for tipulid size to act as a potential environmental indicator.

Among-site variations attributed to topographical differences appear to over-ride any positive influences of time since restoration, which highlights the importance of site-specific management and post-restoration monitoring.

Results indicate that a heterogeneous mosaic of hummocks and pools would allow for tipulid species diversity, which may reduce risks of Tipulidae population crashes during drought.

<u>Introduction</u>

Blanket peatland drainage and restoration

Blanket peatlands comprise upland ombrogenous ecosystems that develop on gentle slopes of upland plateaux at high latitudes. These unique environments are of exceptional conservation value, both owing to their globally unique biodiversity (Thompson *et al.,* 1995; Rydin & Jeglum 2013) and provision of essential ecosystem services. Blanket peatlands moderate water quality, floodwater runoff (Holden, 2005; Grayson *et al.*, 2010) and hold globally important stores of terrestrial carbon (Gorham, 1991; Parish *et al.*, 2007).

Peatlands are formed over millennia by accumulation of partly decomposed wetland plants, primarily *Sphagnum* moss (Parry *et al.*, 2014). Acidic, waterlogged conditions prevent full plant decomposition and thus high water-tables are crucial to ecosystem survival, rendering peatlands vulnerable to the effects of drought and climate change (Freeman, 1992; Worrall *et al.*, 2006). Within many parts of the world, this vulnerability has been exacerbated by extensive peatland drainage for agriculture and forestry (Hooijer *et al.*, 2010; Poulin *et al.*, 2004; Holden *et al.*, 2004). Within the UK alone, over 1.5 million hectares of blanket peatland has been subject to drainage, equating to over half of that present in the country (Stewart and Lance, 1983). This is of particular significance as the UK supports around 10–15% of the world's blanket peat resource (Tallis, 1998).

Peatland drainage in the UK, which reached its peak in the 1970s, largely involved the creation of regularly spaced drainage ditches, known as 'grips' in Northern England (Stewart and Lance, 1983). Grips are typically arranged in herringbone patterns across upland plateau moorland, with short lateral feeder ditches collecting into central ditches (Holden *et al.*, 2004).

Drainage was recorded to exert greatest influence on the hydrology and ecology of 'low altitude blanket peatland', defined by Coulson *et al.* (1990) as peatland annual rainfall levels of approximately 1200mm, typically less than 500m above sea level. This influence comprised extremely localised moisture reduction, often confined to within two metres of grip edges only (Coulson, 1990; Stewart & Lance, 1983; Stewart, 1991). For an ineffective drainage practice with low economic gain, environmental costs were high (Lance, 1978). The localised aeration of peat along grips triggered excessive erosion, carbon loss and peat subsidence (Boelter, 1972; Williamson *et al.*, 2016).

Owing to widespread degradation of essential peatland ecosystem services, peatland restoration has been increasingly integrated into high-level decision making on climate

change mitigation and land management, such as the Paris Agreement on Climate Change (UNFCCC, 2016) and EU Water Framework Directive (Council Directive 2000/60/EC, 2000).

Over recent decades, 'grip-blocking', the process of blocking grips with regular dams to restore high water tables, has been extensively deployed as a peatland restoration technique. Over £500 million has been spent on grip-blocking in the UK alone (Ramchunder *et al.*, 2012). The most extensively utilised material for grip blocking is peat from within the grip itself, which is piled as a dam immediately downstream from its point of excavation, creating both a grip block and an adjacent pool of standing water (Armstrong *et al.*, 2009).

Hydrological studies have documented rapid water table recovery as a response to grip-blocking (Price *et al.*, 2003; Jauhiaianen *et al.*, 2002; Worrall *et al.*, 2007), and long-term research has demonstrated subsequent gradual soil moisture increases for at least a decade post restoration, subject to *Sphagnum* growth and site topography (Haapalehto *et al.*, 2011; Wilson *et al.*, 2010; Bellamy *et al.*, 2011).

The majority of scientific research on the indirect impacts of grip-blocking has largely focused on aspects with legal/financial implications such as carbon sequestration, flood water retention and drinking water quality (e.g. Wallage *et al.*, 2006). It has been acknowledged that there is comparatively little scientific literature on the conservation benefits of grip blocking or long-term monitoring of site ecology post-restoration (Parry *et al.*, 2014; Grant *et al.*, 2012).

Breeding waders and tipulid prey
The blanket peatlands of the UK are of international conservation concern owing to the unique upland biodiversity they support (Thompson *et al.*, 1995). Globally significant populations of breeding wading birds (Charadrii) are of particular regard, many of which are at the southernmost edge of their distribution ranges and experiencing rapid population declines (Pearce-Higgins *et al.*, 2010; Fraizedas *et al.*, 2017).

Tipulids [craneflies] (Diptera: Tipulidae) constitute a vital dietary component for upland breeding birds including those of economic concern such as red grouse *Lagopus lagopus* (Butterfield, 1975), as well as conservation concern including curlew *Numenius arquata*, golden plover *Pluvialis fulva* and dunlin *Calidris alpina* (Pearce-Higgins, 2010). Peatland tipulids undergo mass synchronised emergence each May-June and often constitute over 75% of above-ground invertebrate biomass (Coulson & Butterfield, 1985). It is theorised that

the breeding season of many upland birds is centred around this mass emergence, which is crucial for wader chick survival (Pearce-Higgins, 2010).

Tipula subnodicornis is commonly recognised as the most important tipulid species for breeding waders (Coulson, 1962; Coulson & Butterfield, 1985; Carroll *et al.*, 2011), owing to its overwhelming abundance on high-altitude blanket peatland (peatland with annual rainfall levels of over 2000mm, typically over 500m above sea level (Coulson *et al.* 1990), and comprises one of the few species into which research has been undertaken on larvae autecology (Freeman, 1967; Coulson, 1962). Dominant tipulid species of lower altitude blanket peatland are less well studied, despite the comparative value of these peatlands for breeding wader assemblages (Stillman and Brown, 1994).

Prior to adult emergence, tipulids spend almost a full year in the upper layers of wet soil as eggs and larvae and are extremely vulnerable to desiccation in dry conditions. Dramatic population crashes have been recorded following early-Summer drought events, when tipulids are in the egg and first-larval instar phases of their lifecycles and at their most vulnerable to desiccation (Coulson, 1988). Wheelhouse (1995) recorded a 98% decrease in *T.subnodicornis* abundance following the severe drought of 1989 and Coulson (1962) observed a complete loss of *T.subnodicornis* larvae during a drought in 1955, at locations where the same study had recorded over 1300 larvae per m^2 in 1954. The only tipulids recorded by these studies to survive extreme drought conditions were low numbers within *Sphagnum* habitats, from which populations could recover (Wheelhouse, 1995; Coulson, 1962).

Drainage grips assist in the rapid removal of rainwater and thus can replicate some of the hydrological effects of drought at low-attitude blanket peatlands (Coulson *et al.*, 1990; Holden and Burt, 2002). Coulson (1988) reported marked effects in the distribution of virtually every invertebrate species at a low-altitude blanket peatland subject to drainage, converting from a typical blanket bog community composition to that of a northern heath and grassland.

Considering the vulnerability of tipulid larvae to desiccation, the largely degraded state of UK peatlands and predicted increases in drought intensity and frequency with climate change (Chun *et al.*, 2012), research into the influence of peatland restoration on tipulid populations is acknowledged as an urgent priority (Parry *et al.*, 2014; Buchanan *et al.*, 2006; Beadle *et al.*, 2015). The low-latitude peatlands of the UK are of particular concern, which are arguably

the most important for breeding wader populations whilst also being most at threat from temperature rise (Pearce-Higgins, 2010).

Carroll *et al.* (2011) provided some of the first experimental evidence that grip-blocking can indeed be expected to increase resilience of tipulid populations to drought. This two-year study at various blanket peatland sites (400-550m above sea level) in the UK recorded greater numbers of tipulids at blocked grips in comparison to open grips following dry weather, owing to a positive correlation between tipulid abundance and soil moisture. This important research represents the only published study to directly monitor the influence of drainage and grip blocking on tipulids. Study sites utilised by Carroll *et al.* (2011) had been subject to grip blocking two to three years prior to data collection, however as water-tables and vegetation can continue to alter for over a decade following peatland restoration (Haapalehto *et al.*, 2011; Belamy *et al.*, 2012), it is acknowledged that research into the long-term effects (beyond three years) of grip blocking on tipulids would be of value to further inform restoration management (Parry *et al.*, 2014). Parry et al. (2014) and Beadle et al. (2015) theorised that tipulid abundance will increase with time since restoration owing to high survival rates and population expansion between formally isolated pockets of wet peat and vegetation.

To date, no studies have investigated long-term changes in tipulid abundance in response to peatland restoration. This research is especially important in the light of a climate modelling study by Pearce-Higgins (2011) which calculated that to mitigate the adverse effects of climate change on golden plover populations, peatland restoration would need to result in a two-fold increase in tipulid abundance for every 1°C rise in temperature. A monitoring study of earthworms at rewetted peat grassland in Germany recorded a two-fold increase in earthworms within six years of restoration work (Keplin & Broll, 2002; Plum, 2005). Earthworms are a species group that, like tipulids, can experience rapid population growth in suitable conditions.

The study by Carroll *et al.* (2011) solely focussed on total tipulid abundance, in line with other recent studies tipulids and breeding birds (Douglas & Pearce-Higgins, 2019; Samsom, 2016; Pearce-Higgins & Yalden, 2004). However, Tipulidae is large species group with much physiological variability. For example, the three tipulid indicator species of low-altitude blanket peatland identified by Coulson and Butterfield (1985), range in body length from ~3mm (*Molophilus ater*) to ~12mm (*T.subnodicornis*) and their distribution is likely to have important implications in terms of prey biomass for upland birds. As with most soil dwelling invertebrate groups, physiological differences among Tipulidae larvae cause much variation

in species tolerance of environmental stresses (Wheelhouse, 1995; Plum, 2005). As such, in addition to overall abundance, tipulid species community composition and biomass has the potential to alter with environmental conditions. To date, no published literature has investigated potential changes in tipulid community compositions, or associated biomass, in response to peatland restoration.

Research questions

To address some of the current knowledge gaps in relation to ecological impacts of grip-blocking as a peatland restoration technique, the current study aims to record, for the first time, tipulid abundance and species community composition at grips that have been blocked in phases over the past decade. Specifically, results and data analysis will aim to investigate the following research questions in relation to Spring-emergent tipulids:

1. Are there any species-specific changes in cranefly abundance with soil moisture and vegetation?
2. Does tipulid abundance and community composition alter with time since restoration?

The above research questions are of relevance to peatland-management for breeding waders in the context of climate change mitigation. The study also has high relevance to the effectiveness of grip blocking on maintaining high water tables during drought, as the data has been collected following the severe UK drought of Summer 2018.

Methods

Study site

West Pennine Moors Site of Special Scientific Interest (SSSI), Lancashire, England supports an extensive mosaic of moorland and moorland-fringe habitats including blanket bog and heath. Diverse breeding bird assemblages are listed qualifying features of the SSSI (Natural England, 2017), which include wader species such as dunlin, curlew, golden plover and redshank *Tringa totanus* (RSPB, 2017). The locality has an average annual rainfall of 1236mm (Climate-data.org, 2019), classing the SSSI as low-altitude blanket peatland (Coulson *et al.* 1990). Much of the moorland habitats present are historically drained and extensive grip blocking has taken place over the past 15 years using peat turves (K. Rogers, 2019. Pers. comm., 19 March).

Four study sites were allocated by Natural England that comprised:
- Open grips;
- Grips blocked 1 year prior to study;
- Grips blocked 7 years prior to study; and,

- Grips blocked 10 years prior to study.

Owing to the spatially phased nature of restoration work, the above grip classifications were necessarily located at four separate locations within the SSSI (Figure 1 & Table 1), which are referred to as '1yr-Blocked', '7yr-Blocked' and '10yr-Blocked' throughout the remainder of this report.

The four study sites possessed some unavoidable among-site topographical variations, summarised in Table 1. Topographical data was collected at each sampled grip following methods outlined by Evans *et al.* (2005) provided as Appendix 1. The 1yr-Blocked site possessed the flattest overall topography with the shallowest grips, whilst the 10yr-Blocked site contained the deepest and widest grips. Importantly, large peat pipes (natural subsurface conduits for water) were noted at the 10yr-Blocked site, which measured over 1metre in depth and 0.5metres in width where roof collapse allowed for visual inspection.
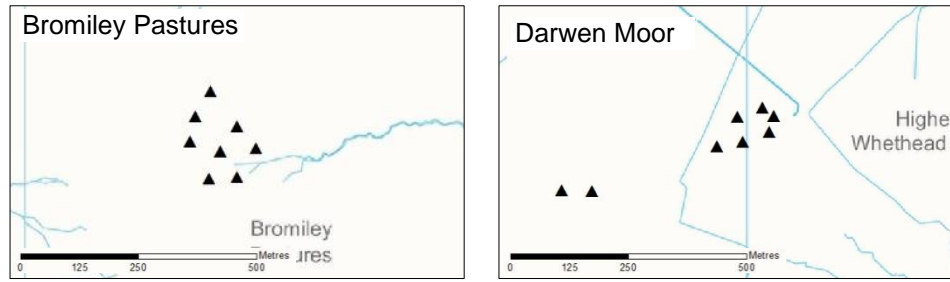
Figure 1: Study site locations in North-West England, Map of West Pennine Moors SSSI (blue stripes) showing locations of four study sites and sampling locations grips (triangles). Each sampling location held two traps. Full site details provided in Table 1. Map created using ArcMap 10.6.1 using Natural England Open Data for SSSI shapefile and EDINA Ordnance Survey data basemaps.

Field sampling

Fieldwork was collected May to July 2019, meaning all emergent adult tipulids collected will have been eggs and first-instar larvae throughout the May-July 2018 drought. 2018 experienced the driest May to July three month period for England since 1921 (Environment Agency, 2018).

Table 1: Study sites details including locations and key topographical attributes

Site / Attribute	Open			Blocked – 1yr			Blocked – 7yr			Blocked – 10yr		
Central GR	SD679196			SD654184			SD642203			SD753216		
OS Site Ref.	Darwen Moor			Bromiley Pasture			Withnell Moor			Musbury Heights		
	Min	Mn	Max	Min	Mn	Max	Min	Mn	Max	Min	Mn	Max
Altitude (m)	380	381	382	361	364	366	278	294	309	347	355	362
Grip depth (m)	0.09	0.19	0.35	0.02	0.08	0.21	0.08	0.14	0.22	0.20	0.46	0.70
Grip width (m)	0.50	1.14	2.52	0.30	0.50	0.65	0.57	1.80	3.54	2.87	3.62	5.03
Grip slope (%)	0	3.2	6.5	0.4	1.41	2.5	2.7	6.75	10.2	1.4	4.34	6.4
Grip bank angles (%)	19.8	38.3	74.5	12	13.4	15	12	13.4	15	25	27.6	37
Grip block spacing (m)	n/a	n/a	n/a	8	10	12	9	10.6	15	12	12.4	15
Grip spacing (m)	11	15.1	20	10	10.6	15	12	13.4	15	25	27.6	37

GR: Ordnance Survey Grid Reference; Min/Mn/Max: Minimum/Mean/Maximum measurements of all grips at locations subject to tipulid monitoring.

Practical field sampling methods for tipulids largely followed those employed by Carroll *et al.* (2011). Stratified random sampling identified eight sampling locations at each of the four study sites (Figure 1). Sampling locations were constrained to be separated by at least 50metres wherever possible and always located along different grips. Two emergence traps

were deployed at each sampling location, one located within 0.5metres of the drain edge (immediately upstream from grip blocks at blocked sites for consistency) and one located five metres downslope from the drain edge to allow for potential variations in soil moisture (Stewart, 1991) (Figure 2). This totalled 64 traps in total at 32 sample locations.

Carroll *et al*. (2011) placed another two traps mid-way between grip blocks at their sample locations, to allow for any further potential variations in soil moisture along grips. Their study recorded no differences in soil moisture at these mid-grip traps and as such, considering the finite number of traps available for the current study, these additional trap locations were not included.

Tipulid abundance and community composition
Emergence traps comprised plastic baskets (approximately L41cm x W28cm x H17cm), pegged into the ground open end down, with Yellow Dry-stick Traps (Oecos: Kimpton, UK) attached to every inner surface (Carrol *et al*., 2011). Upon collection, sticky traps were labelled and transferred to a cold room for tipulid preservation. All caught tipulids were identified to species level.
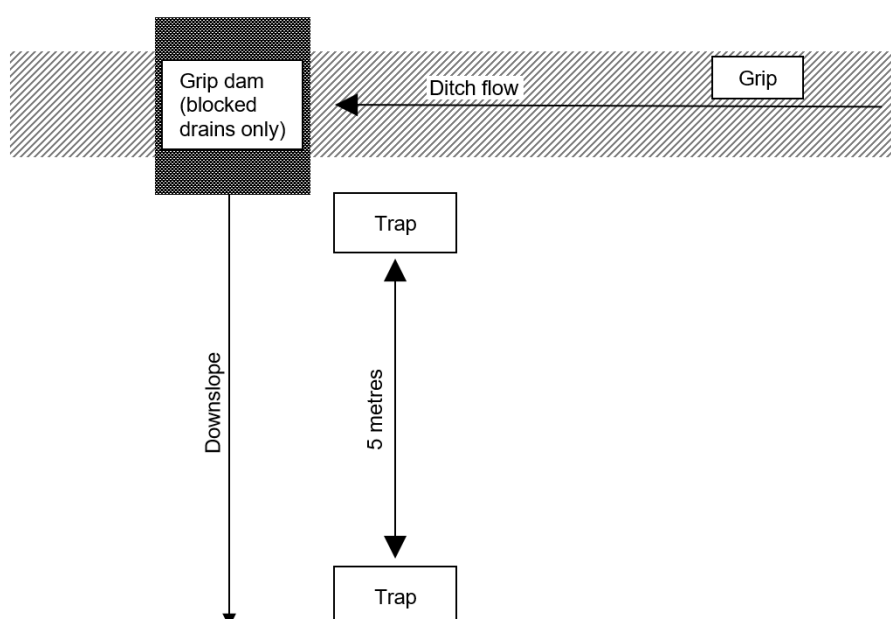

Figure 2: Diagram showing the positions of the two traps at sampling points. Diagram shows the layout at a blocked grip. The first trap is placed immediately upstream of grip dam, and the second positioned 5 metres downslope. Diagram and methodology adapted from Carroll *et al.* (2011).

Emergence traps were in place by the 1st May 2019 and were active until 28th June 2019 to cover the mass emergence period of peatland tipulid species (Coulson, 1959). Sticky traps

were replaced every 20 days. Abundance counts from each trapping period confirmed the peak emergence season had been covered and allowed for comparisons in overall emergence trends between species.

Emergence traps are uniquely appropriate for the research objectives of this study as they are categorically plot specific, meaning they only capture invertebrates that emerge from peat directly covered by traps, with no other species caught from other more or less favourable habitats.

Alternative field data collection methods for tipulids include Malaise traps, window traps, sweep netting (Salmela, 2001), unenclosed sticky traps (Coulson, 1959), pitfall traps (Baines, 1988; Whelhouse, 1995) and more recently, visual transect counts (Pearce-Higgins & Yalden, 2004; Samsom, 2016; Douglas & Pearce-Higgins, 2019). None of these methods are plot specific and, as with many other flying insects (e.g. Pasek, 1988; Ludwig *et al.*, 2019), tipulids can be easily wind assisted to situations away from larval habitats, especially within open moorland habitats (A. Stubbs, 2019. Pers. comm., 9 Sept). In addition, the transect approach does not allow identification to species level, may overlook small or well-hidden tipulids, and relies on the accurate prediction of, and clement weather during, the peak tipulid emergence window, which for some species spans a matter of days (Coulson, 1959). Soil cores of final-instar larvae are an alternative plot-specific approach (Fisher & Walker, 2015; Baines, 1988; Coulson, 1962) however larvae of smaller species are often elusive and limited published technical detail is available (A. Stubbs, 2019. Pers. comm., 9 Sept).

Tipulid biomass
Weighing specimens to estimate tipulid biomass was not practical in this instance owing to the drying out of specimens in traps and practicalities of detaching them from sticky traps. Wing length has been identified as an alternative useful proxy for tipulid biomass (Butterfield, 1973). This study measured wing lengths from each tipulid species to estimate biomass. Only wings of male tipulids were measured, as female tipulids are mostly flightless with reduced wings.

Soil moisture & Vegetation data
Data to calculate the mean soil moisture per sample plot was collected from five points within the footprint of each trap at the end of each trapping period, using a Procheck 5TE moisture probe (Decagon Devices, Pullman, USA). Data was recorded in volumetric soil moisture (m^3 water m^{-3} soil), which ranges from 0 to 1, 1 implying the water table is at or

above ground level. Soil moisture at the time of the survey is unlikely to reflect year-round conditions, however readings provide an indication of relative differences in soil moisture among the sites.

Plant species data provides additional evidence of year-round soil moisture conditions and can also influence the distribution of tipulid species communities (Wheelhouse, 1995). Plant species cover within the footprint of each emergence trap was recorded using the DOMIN scale (Kent, 1992) including cover of bare peat.

Data analysis

All data analysis was completed in RStudio (RStudio Team, 2018). Field data collected provided the following information for each trap location:

- Average soil moisture
- Plant species cover
- Total adult tipulid abundance
- Community composition of adult tipulids
- Relative adult tipulid biomass

Shapiro-Wilk & Bartlett tests indicated that soil moisture data did not differ significantly from a Normal or Homogenous distribution. A two-way ANOVA and post-hoc Tukey test established differences in average soil moisture between the four sites (Open, 1yr-Blocked, 7yr-Blocked, 10yr-Blocked) and between 'near' and 'far' plots (traps located 0.5m and 5m from grip edges), and any interactions between these variables.

Species abundance for each trap was converted to tipulids per m^2 for analytical context. Shapiro-Wilk tests indicated that tipulid abundance data significantly differed from a Normal distribution and as such Kurksal Wallis and post-hoc testing was utilised to establish any significant differences in tipulid species abundance between the between the four sites (Open, 1yr-Blocked, 7yr-Blocked, 10yr-Blocked) and between 'near' and 'far' plots (traps located 0.5m and 5m from grip edges), and also between the three trapping periods.

To estimate relative tipulid biomass among sites, the accumulative wing length (species wing length*species abundance per m^2) of each species present was summed to gain a total accumulative wing length per m^2 for each sampling plot. Shapiro-Wilk tests indicated that tipulid biomass data significantly differed from a Normal distribution and Kurksal Wallis and post-hoc testing was utilised to establish any significant differences in biomass between

sites (Open, 1yr-Blocked, 7yr-Blocked, 10yr-Blocked) and between 'near' and 'far' plots (traps located 0.5m and 5m from grip edges).

To investigate similarities or differences of species community compositions among the sites, Non-metric Multi-Dimensional Scaling (NMDS) was utilised for both plant species and tipulid species.

Spearmans rank correlation co-efficient was employed to establish strength of relationships between tipulid abundance (total, individual species and biomass), plant species cover and soil moisture.

<u>Results</u>

643 tipulids were caught in total by the study consisting of nine species. *Ormosia pseudosimilis* (54%) and *Tricyphona immaculata* (31%) were the most abundant species. *Tipula subnodicornis* (8%) and *Idioptera pulchella* (6%) were regularly caught in lower numbers. The remaining species (*Phylidorea fulvonervosa, Erioconopa trivialis, Hoplolabis vicina, Pedicia rivosa* and *Molophilus ater*) accumulatively represented 1% of the total number caught.

Relationship between tipulid species abundance, soil moisture and vegetation

There was no significant correlation between total tipulid abundance per m^2 and mean soil moisture (rho=-0.01,*n*=64,p=n.s.). However upon modelling the four most abundant species separately, each was found to significantly correlate with soil moisture, with *O.pseudosimilis* being negatively correlated and the remainder positively correlated to varying degrees (Table 2).

Scatter plots of correlations imply that each of these four species has a potential soil moisture 'threshold', below or above which the species rarely occurs (Figure 3). *O.pseudosimilis* was rarely recorded at plots with a mean soil moisture exceeding $0.5m^3/m^{-3}$, and the remaining three species rarely occurred at plots with a mean soil moisture below $0.45m^3/m^{-3}$. Numerous zero scores occur for all species owing to the considerable variability in species presence/absence among the four study locations.

Comparison of correlation strengths implies that soil moisture exerts greater influence of tipulid distribution than plant species, although there is much covariation (Table 2). Tipulid species most strongly associated with high soil moisture (*T.subnodicornis* and *T.immaculata)* positively correlate with wetland plant species *E.angustifolium* and *E.tetralix* and negatively

correlate with dry heath plants *C.vulgaris* and *H.jutlandicum*. The reverse is true for *O.pseudosimilis*, which was negatively associated with soil moisture. *I.Pulchella* did not possess any significant correlations with individual plant species.

T.immaculata was the only species to possess a significant positive correlation with *M.culina* and no tipulid species negatively correlated with *M.culina* to counteract this, hence overall tipulid abundance possessed a positive relationship with *M.culina*. Similarly, the only two significant correlations with *C.vulgaris* were negative and as such overall tipulid abundance possessed a negative relationship with *C.vulgaris* (Table 2).

Relationship between tipulid biomass, soil moisture and vegetation

The largest species identified was *P.rivosa* with a wing length of 23mm, and the smallest was *M.ater* with a wing length of 2mm. A positive correlation was identified between mean soil moisture and relative tipulid biomass (rho=0.25,*n*=64,p=<0.05). Figure 4 demonstrates this is mainly owing to the influence of four specific 1yr-Blocked plots. These study samples returned high numbers of *T.immaculata* as well as low numbers of *T.subnodicornis*, both relatively large species (wing lengths of 9 and 12mm respectively) in comparison to *O.pseudosimilis* (wing length of 4mm). Positive relationships were also recorded between tipulid biomass and *M.caerulea* and *E.tetralix*, whilst a strong negative correlation was

Table 2: Spearmans rank correlation matrix of tipulid abundance, plant species and soil moisture. (N=64)

	Soil moisture	Total tipulid abundance	Total tipulid biomass	*O.pse* (69)	*T.imm* (47)	*T.sub* (36)	*I.pul* (16)
Soil moisture	-	-0.014	**0.247**[d]	**-0.574**[a]	**0.487**[a]	**0.441**[b]	**0.281**[d]
Cv (70)	**-0.426**[b]	**-0.309**[d]	**-0.424**[b]	0.171	**-0.420**[b]	**-0.312**[d]	0.170
Ea (63)	**0.680**[a]	-0.144	0.163	**-0.532**[a]	**0.486**[a]	**0.390**[c]	0.009
Mc (38)	0.027	**0.413**[c]	**0.397**[c]	0.068	**0.336**[c]	0.149	-0.200
Hj (23)	**-0.581**[a]	0.004	-0.175	**0.423**[b]	**-0.307**[d]	**-0.281**[d]	-0.128
Et (17)	**0.248**[d]	0.078	**0.263**[d]	**-0.405**[b]	**0.598**[a]	**0.266**[d]	-0.194
Sf (13)	*0.240*	-0.103	-0.090	-0.040	0.085	-0.029	-0.029

Tipulid species that occur at less than 5% of plot locations excluded from table. Plant species that occur at less than 10% of plot locations excluded from table and only species with significant correlations included. Bracketed numbers: % of plots at which species present. Significant values in bold. *Cv: Calluna vulgaris, Ea: Eriophorum angustifolium, Mc: Molinia caerulea, Hj: Hypnum jutlandicum, Et: Erica tetralix, Sf: Sphagnum fallax. O.pse: Ormosia pseudosimilis, T.imm:*

Tricyphona immaculata, T.sub: Tipula subnodicornis, I.pul: Idioptera pulchella.

[a]: correlation significant to the level of 0.0001

[b]: correlation significant to the level of 0.001

[c]: correlation significant to the level of 0.01

[d]: correlation significant to the level of 0.05

Italics: correlation marginally significant to the level of 0.055

identified between biomass and *C.vulgaris*. Each of these significant correlations was heavily influenced by the vegetation species compositions within the four 1yr-Blocked plots with highest *T.immaculata* abundance, in a similar fashion to Figure 4.

Effect of time since restoration: soil moisture

Mean soil moisture differed significantly among the restoration sites ($F_{3,56}=24.401$, p=<0.0001). Post-hoc analysis identified that open grips and 10yr-Blocked grips possess lower average soil moisture to 1yr-Blocked and 7yr-Blocked grips (Figure 5).

Mean soil moisture also differed significantly with the distance of the sample plot from grips, with 'near' plots (0.5m from grip) possessing an overall higher moisture content than 'far' plots
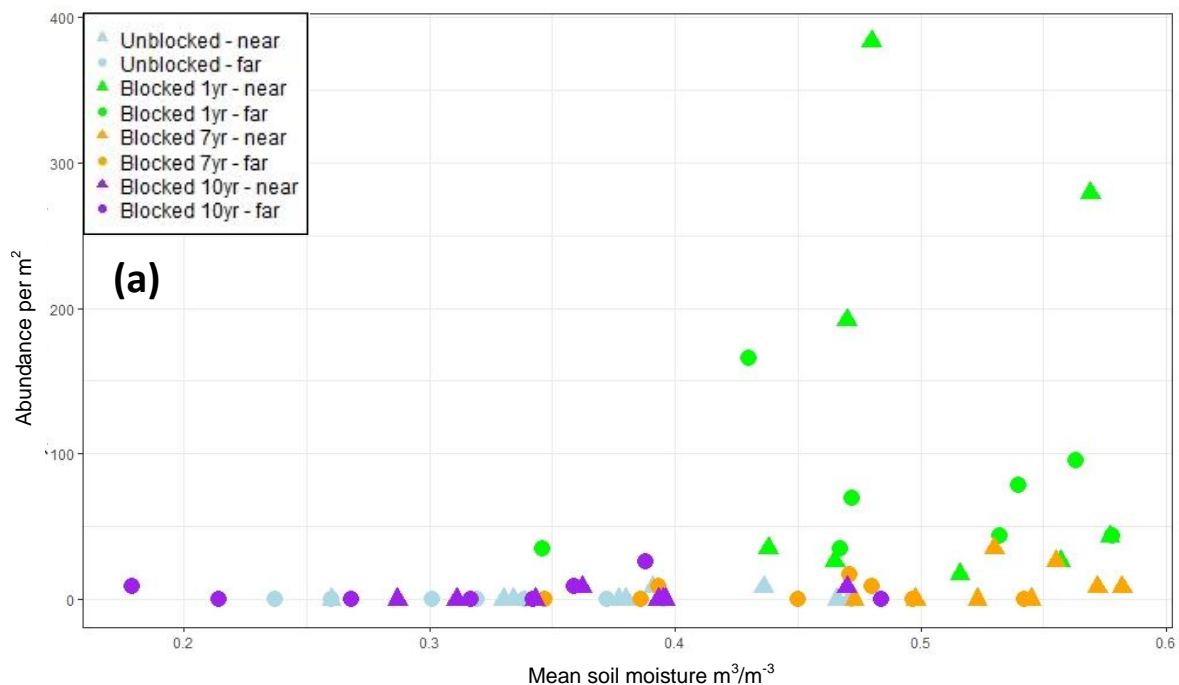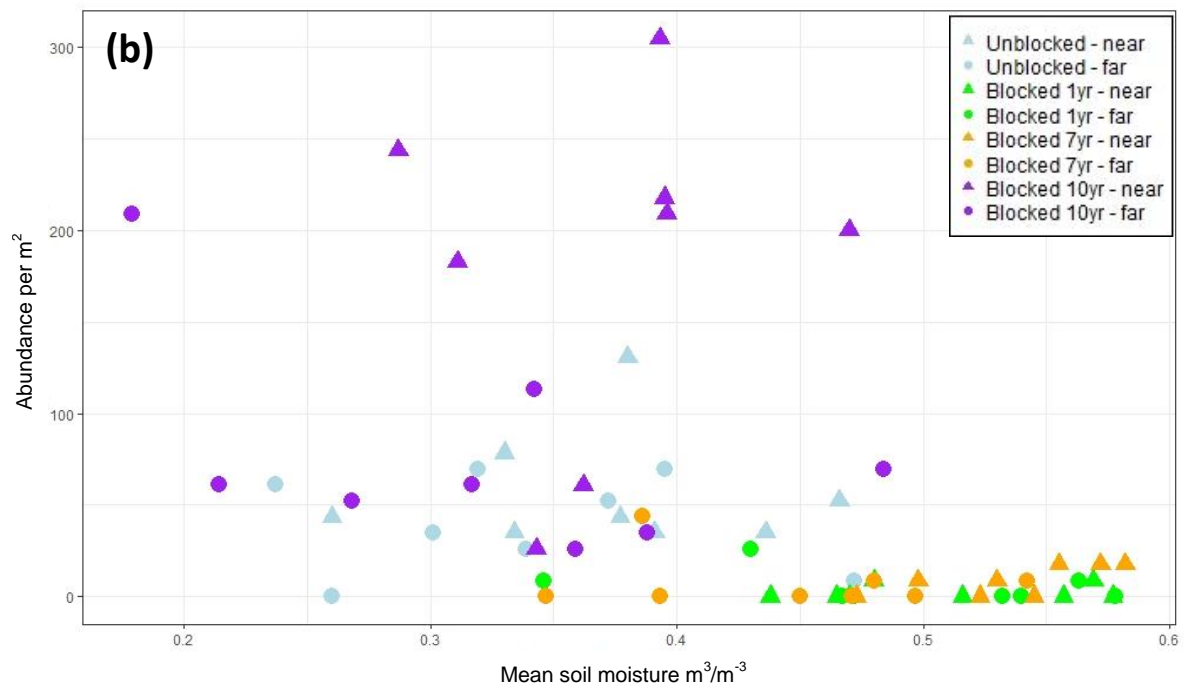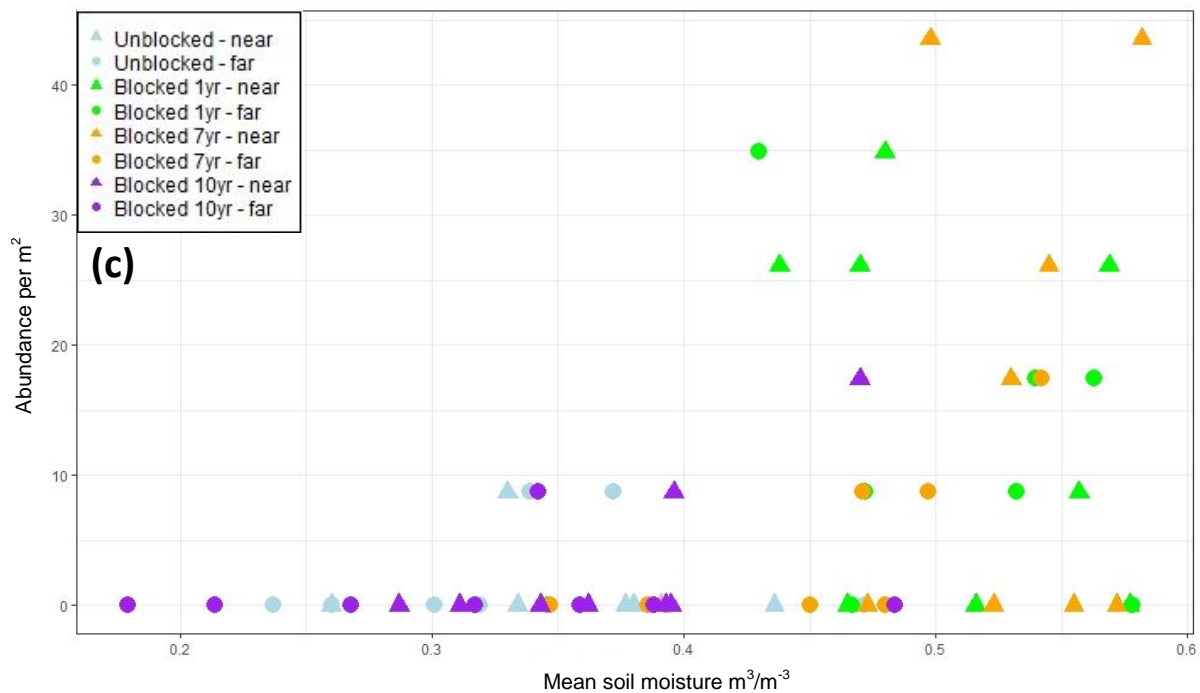
Figure 3: Relationships between mean volumetric soil moisture and tipulid abundance, for the four most abundant species caught: **(a)** *Tricyphona immaculata,* **(b)** *Ormosia pseudosimilis,* **(c)** *Tipula subnodicornis* and **(d)** *Idioptera pulchella* **(continues)**
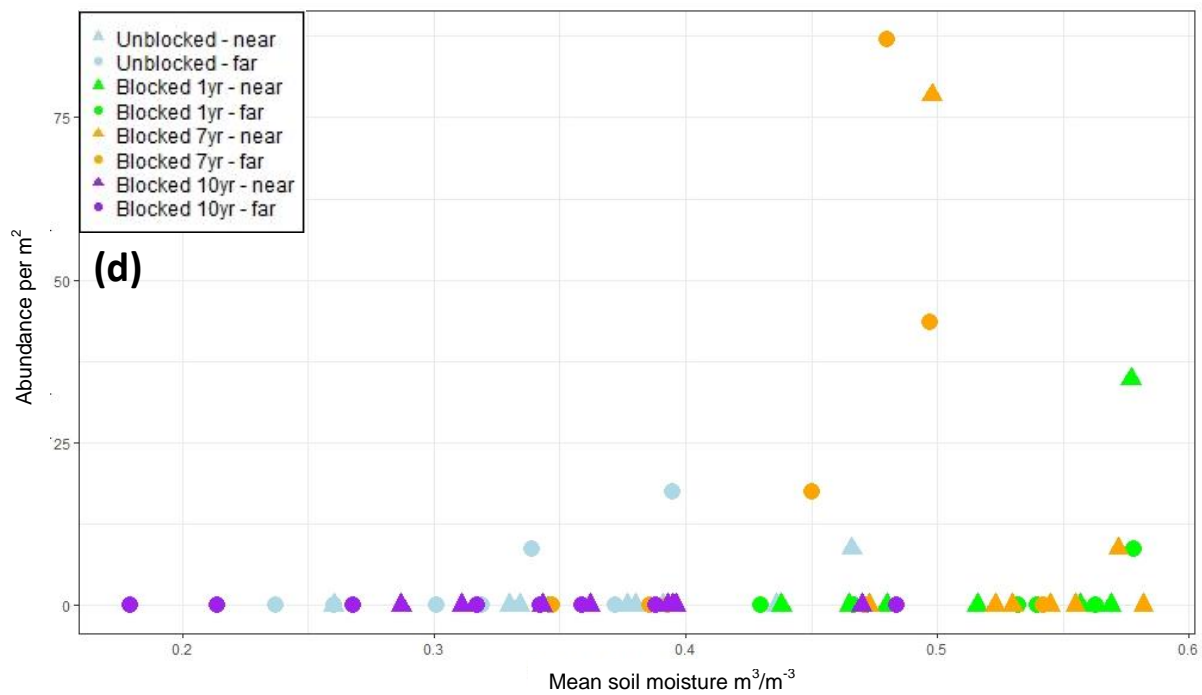
Figure 3: Relationships between mean volumetric soil moisture and tipulid abundance, for the four most abundant species caught: **(a)** *Tricyphona immaculata,* **(b)** *Ormosia pseudosimilis,* **(c)** *Tipula subnodicornis* and **(d)** *Idioptera pulchella*

(5m from grip) ($F_{1,56}$=7.944, p=<0.01) at Open, Blocked-7yr and Blocked-10yr sites. No significant interactions were identified.
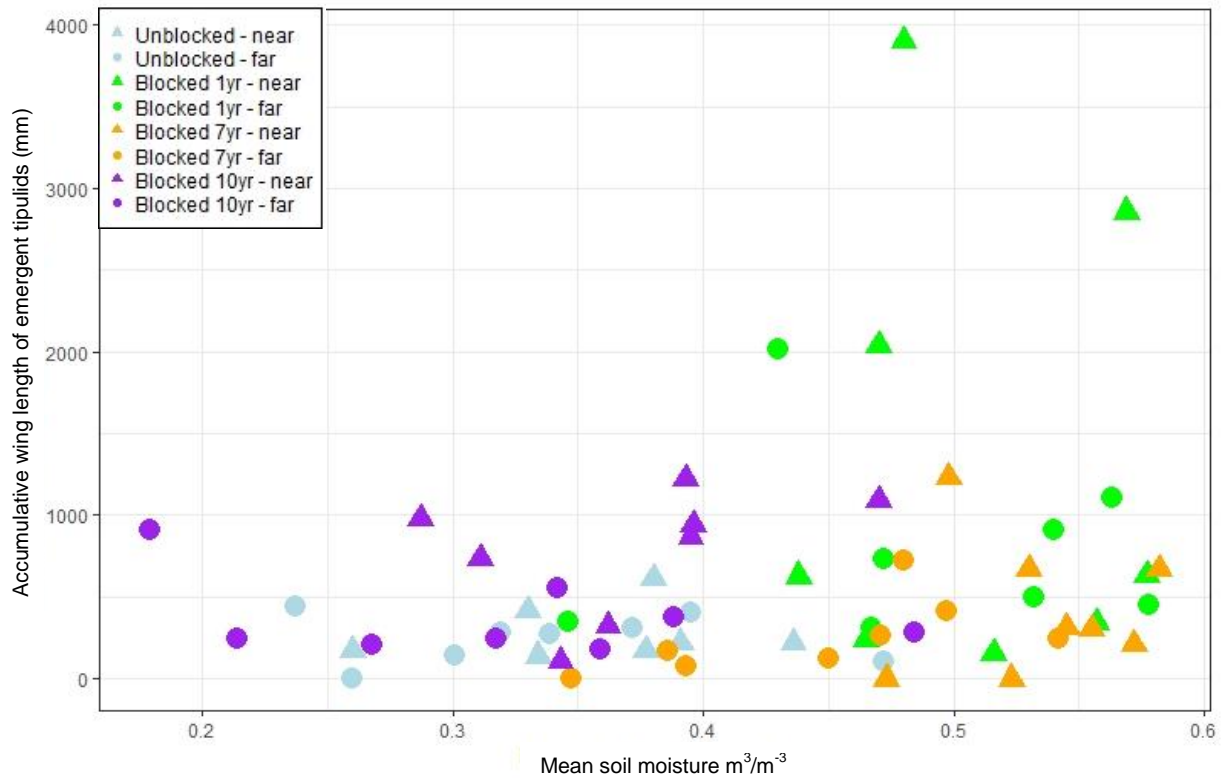
Figure 4: Relationship between mean volumetric soil moisture and relative biomass (accumulative wing length of all tipulids per m^2)
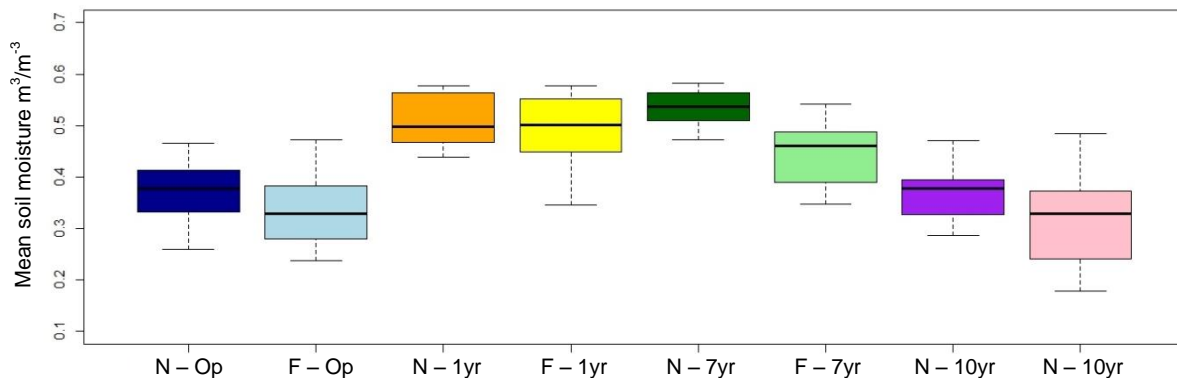

Figure 5: Comparison of Mean volumetric soil moisture, between open grips (Op) and grips that have been blocked for one (1yr), seven (7yr) and ten years (10yr), with additional comparisons of plots at the grip edge (N) and 5m from the grip edge (F). Box midline indicates median, box edges indicate interquartile range. Whiskers indicate range of data.

Effect of time since restoration: vegetation

Nineteen plant and bryophyte species were recorded by the study, typical of blanket bog, wet heath, dry heath and acid grassland habitats. An NMDS plot of vegetation communities (Figure 6) implies an abundance of wetland species at the 1yr-Blocked site such as *Eriophorum angustifolium* and *Erica tetralix*, which possessed strong positive correlations

with soil moisture (Table 2). Species associated with dry heath such as *Calluna vulgaris* and *Hypnum jutlandicum* were most prevalent at open grips, which possessed overall significant negative correlations with soil moisture (Table 2). The 7yr-Blocked and 10yr-Blocked sites possessed greatest variability in plant species communities. The NMDS model possessed moderate stress (<0.2) suggesting the clustering of communities is not likely to be random.
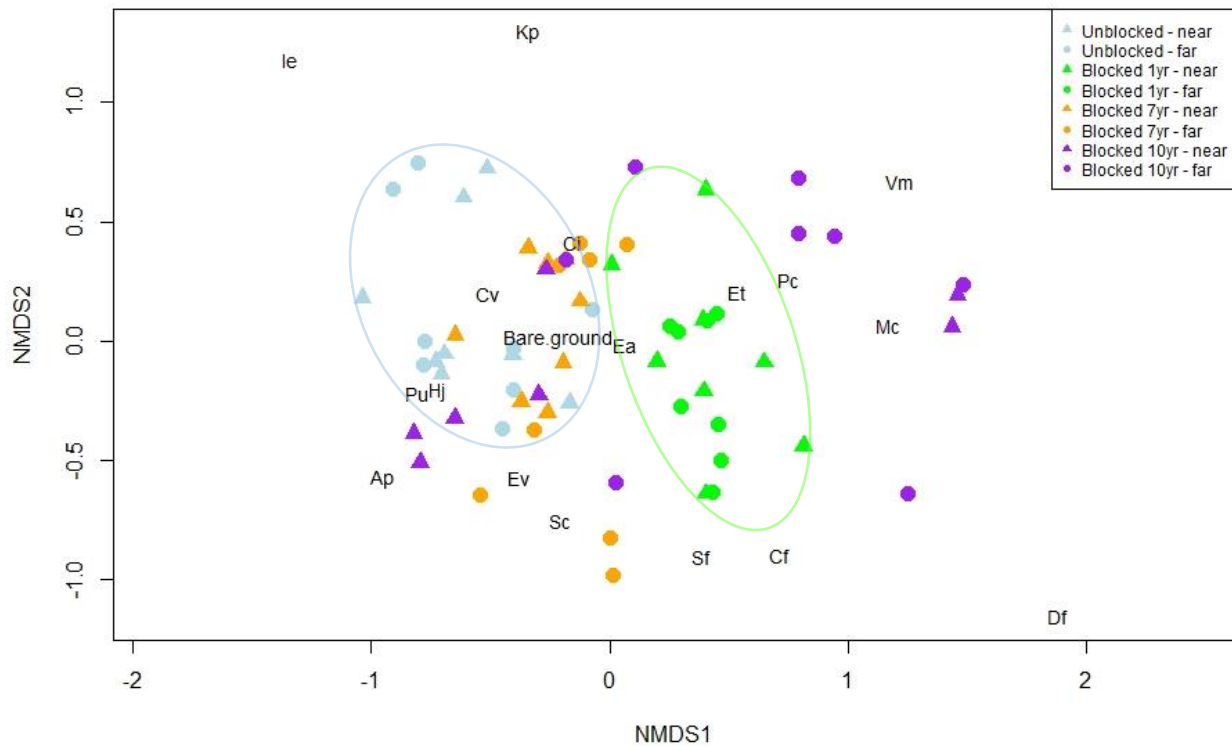

Figure 6: Nonmetric multidimensional scaling (NMDS) ordination of the 64 plots representing plant community compositions of sites with open grips, and grips that have been blocked for one, seven and ten years. Cv: *Calluna vulgaris,* Ea: *Eriophorum angustifolium*, Mc: *Molinia caerulea*, Hj: *Hypnum jutlandicum*, Et: *Erica tetralix*, Sf: *Sphagnum fallax*, Ci: *Campylopus introflexus*, Ev: *Eriophorum vaginatum*, Bare.ground: bare peat, Pu: *Plagiothecium undulatum*, Ie: *Isopterygium elegans*, Kp: *Kindberga praelonga*, Pc: *Polytrichum commune*, Vm: *Vaccinium myrtillus*, Cf: *Calypogeia fissa*, Sc: *Sphagnum capillifolium*, Ap: *Aulacomnium palustre*, Df: *Deschampsia flexuosa*.

Effect of time since restoration: tipulid abundance

Overall tipulid abundance differed significantly between the sites (H=13.9,n=64,p=<0.01). Post-hoc testing identified higher numbers of tipulids at 1yr-Blocked and 10yr-Blocked grips in comparison to those that had been blocked for 7yrs (Figure 7). No significant difference was identified between the 'near' and 'far' plots (H=1.30,*n*=64,p=n.s.).
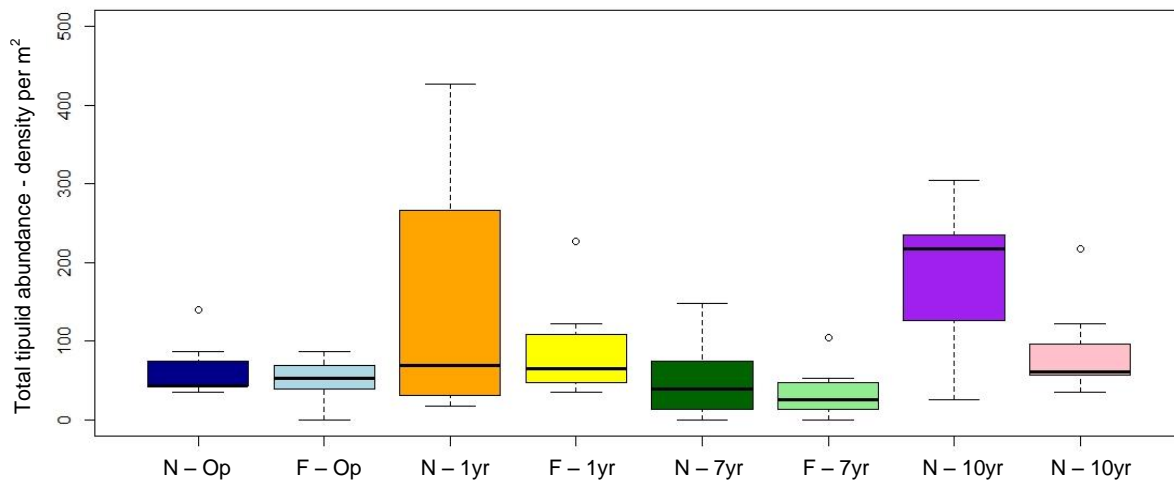
Figure 7: Comparison of total adult tipulid abundance (number per m²), between open grips and grips that have been blocked for 1, 7 and 10 years, with additional comparisons of plots at the grip edge (Near) and 5m from the grip edge (Far). Box midline indicates median, box edges indicate interquartile range. Whiskers indicate range of data; points indicate data outside 1.5 x the interquartile range.

Effect of time since restoration: tipulid community composition

An NMDS model of tipulid community composition suggests that community compositions present at Open and 10yr-Blocked grips centre around the abundance of *O.pseudosimilis,* significantly differing to the 1yr-Blocked species community compositions, which centre around the abundance of *T.immaculata* (Figure 8). The 7yr-Blocked site possesses greatest variation of tipulid species communities. The NMDS model has good stress (<0.1) suggesting the clustering of species communities is highly unlikely to be random.

Kurksall Wallis and Post-hoc testing confirmed significantly higher numbers of *O.pseudosimilis* at open and 10yr-Blocked grips in comparison to 1yr-Blocked and 7yr-Blocked grips (H=44.31,N=64,p=<0.0001) and greater numbers of *T.immaculata* at 1yr-Blocked grips to all other sites (H=42.00,N=64,p=<0.0001) (Figure 9). *T.subnodicornis* abundance was greater at 1yr-Blocked grips in comparison to open grips and 10yr-Blocked grips (H=11.75,N=64,p=<0.01) (Figure 9). *I.pulchella* abundance did not significantly differ between sites (H=6.53,N=64,p=n.s) and no significant difference between the 'near' and 'far' plots was recorded for any of the species (H=0.65/0.07/0.62/0.48, n=64, p=n.s.).
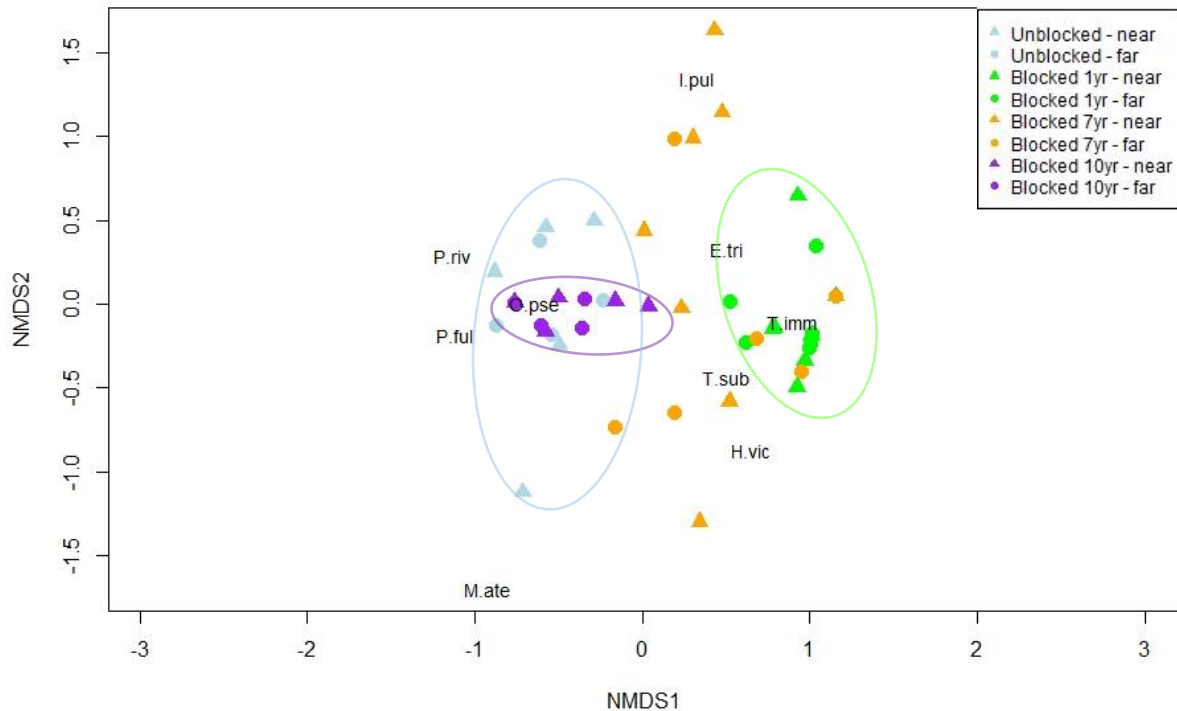
Figure 8: Nonmetric multidimensional scaling (NMDS) ordination of the 64 plots representing tipulid community compositions of open grips and grip that have been blocked for one, seven and ten years. O.pse: *Ormosia pseudosimilis*, T.imm: *Tricyphona immaculata*, T.sub: *Tipula subnodicornis*, I.pul: *Idioptera pulchella*, P.ful: *Phylidorea fulvonervosa*, E.tri: *Erioconopa trivialis*, H.vic: *Hoplolabis vicina*, P.riv: *Pedicia rivosa*, M.ate: *Molophilus ater*.

In terms of species phenology, *T. subnodicornis*, *T. immaculata* and *I. pulchella* all experienced peak numbers between 20th May and 8th June 2019, whereas highest numbers of *O.pseudosimilis* were recorded in the final collection period between 8th and 28th June 2019 (Figure 10).

Effect of time since restoration: tipulid biomass

Based on wing length alone, overall biomass of emergent adult tipulids varied significantly between sites (H=15.57,N=64,p=<0.01). Post-hoc testing identified higher tipulid biomass at the 1yr-Blocked study site (Figure 11), despite high overall tipulid abundance at both 1yr-Blocked and 10yr-Blocked sites (Figure 7). This is owing to the small wing length of *O. pseudosimilis* (4mm) in comparison to *T.immaculata* (9mm) and *T.Subnodicornis* (12mm).
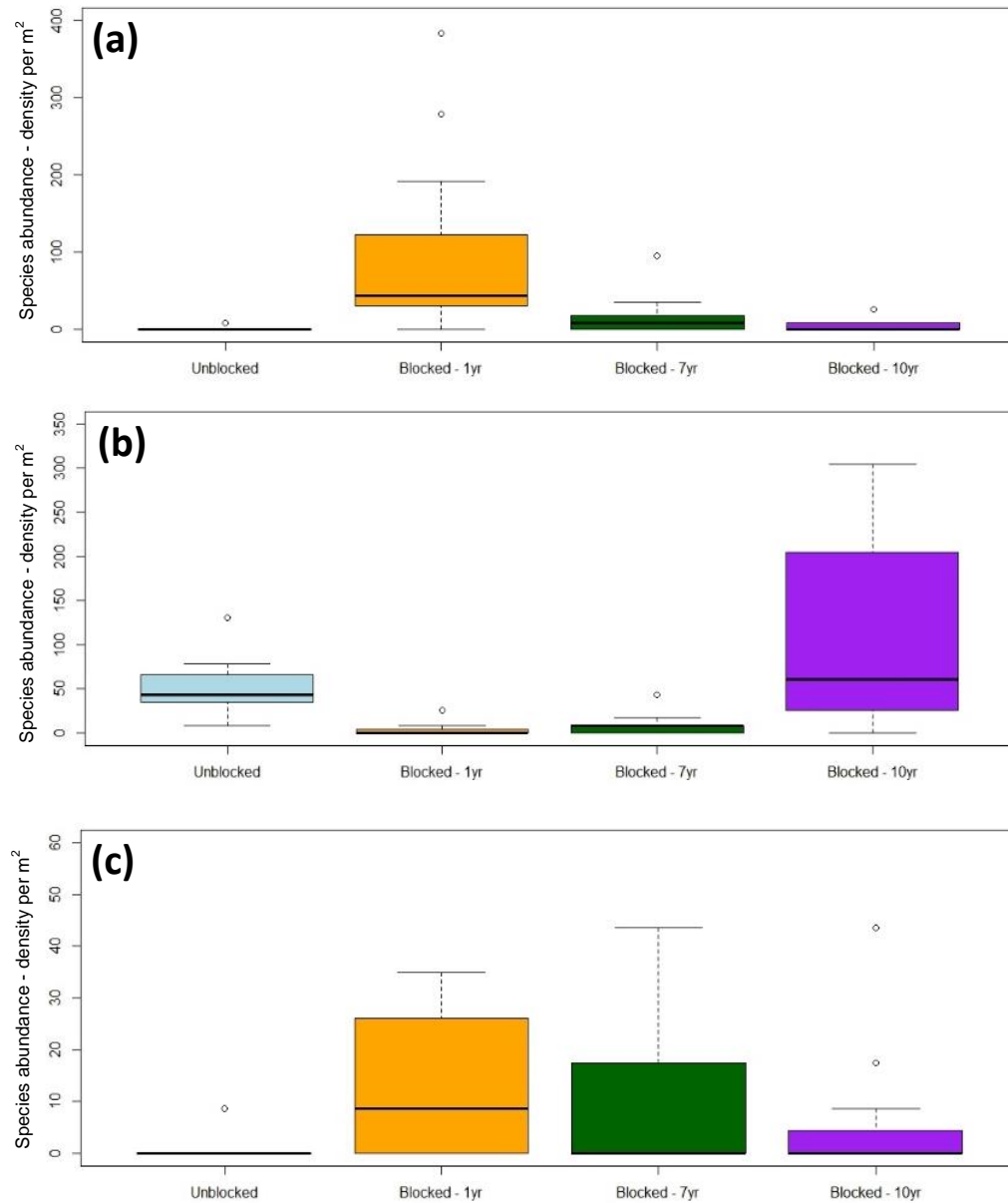
Figure 9: Comparison of tipulid species abundance (number per m^2) for **(a)** *Tricyphona immaculata,* **(b)** *Ormosia pseudosimilis* and **(c)** *Tipula subnodicornis* between open grips and grips that have been blocked for 1, 7 and 10 years. *Idioptera pulchella* is not represented owing to lack of significant differences. Box midline indicates median, box edges indicate interquartile range. Whiskers indicate range of data; points indicate data outside 1.5 x the interquartile range.
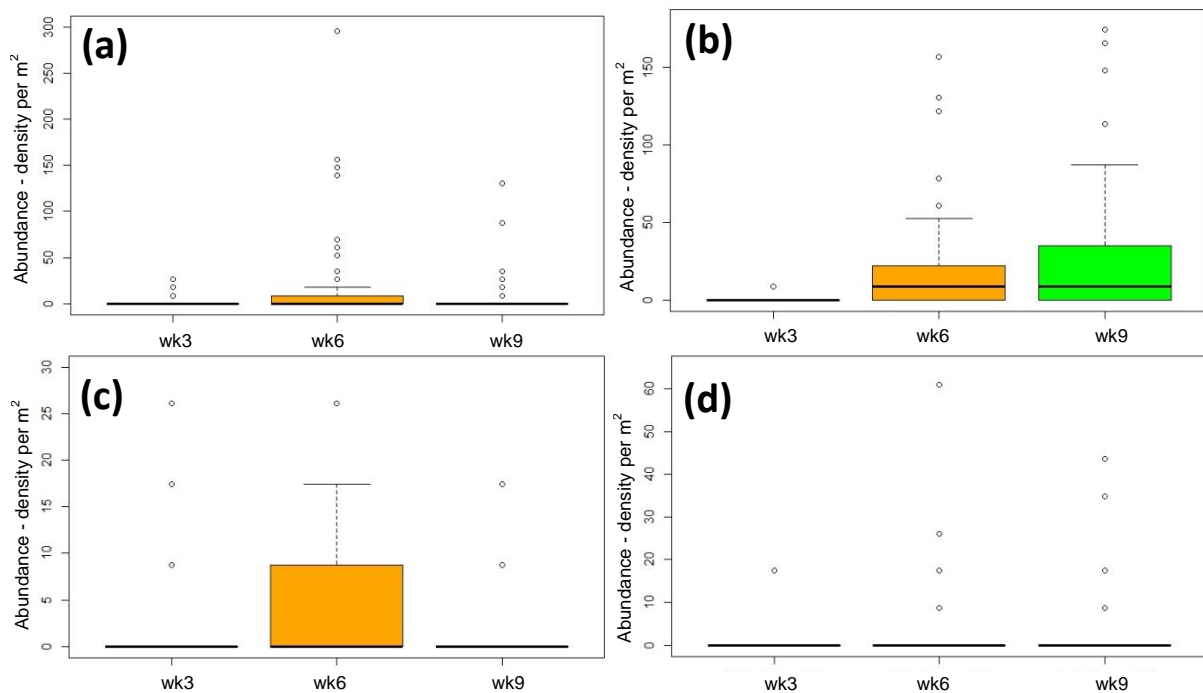
Figure 10: Comparison of tipulid abundance (number per m^2) between collection weeks 3 (wk3), 6 (wk6) and 9 (wk9), for the four species most frequently recorded **(a)** *Tricyphona immaculata,* **(b)** *Ormosia pseudosimilis,* **(c)** *Tipula subnodicornis* and **(d)** *Idioptera pulchella*. Box midline indicates median, box edges indicate interquartile range. Whiskers indicate range of data; points indicate data outside 1.5 x the interquartile range.

Figure 9: Comparison of mean relative biomass (accumulative wing length of all tipulids per m^2) between open grips and grips that have been blocked for 1, 7 and 10 years. Box midline indicates median, box edges indicate interquartile range. Whiskers indicate range of data; points indicate data outside 1.5 x the interquartile range.
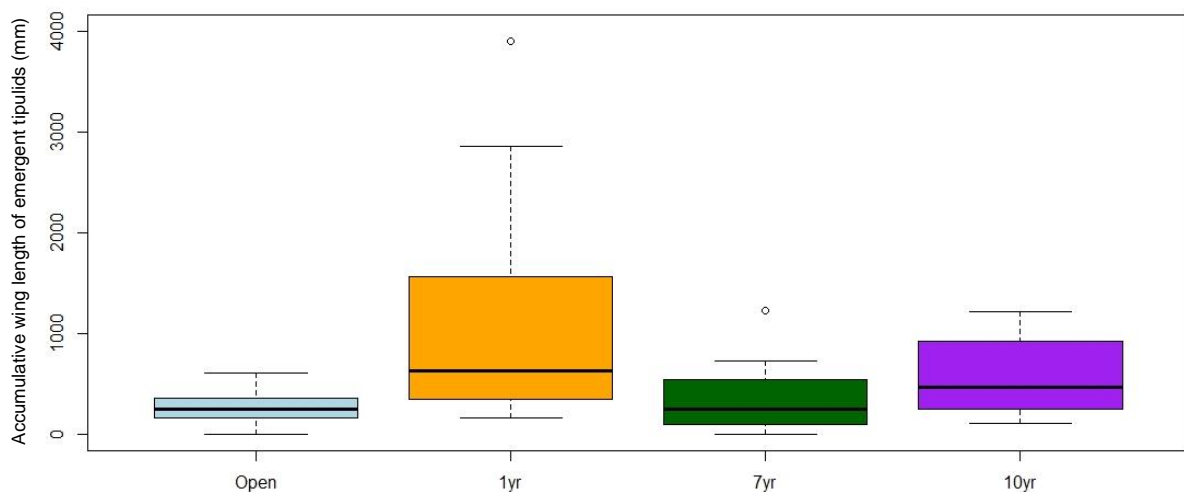
Discussion

Relationship between tipulid abundance, biomass, soil moisture and vegetation

The positive correlation between soil moisture and total tipulid biomass, in contrast to the lack of correlation between soil moisture and total tipulid abundance, highlights the importance of community composition analysis. Larger tipulid species *T.immaculata* and *T.subnodicornis* appear to positively correlate with soil moisture, whilst the small yet abundant species of *O.pseudosimilis* thrives only in relatively drained peat. Results suggest the potential existence of a critical moisture threshold, below or above which the above key moorland species occur, however this would require further year-round monitoring.

Limited literature exists on *O.pseudosimilis* and this is potentially the first study to provide raw data in relation to the autecology of *O.pseudosimilis* larvae (J.Kramer, 2019. Pers. comm., 29 Sept). *M.ater* is an ecologically and physiologically similar species to *O.pseudosimilis* (Wheelhouse, 1995; Hadley, 1971) which has been recorded by previous studies to experience both no population change and significant population increase following drought (Hadley, 1966), suggesting these smaller species may be physiologically adapted to lower moisture conditions, or perhaps may experience competitive or predatory 'population release' in the absence of larger tipulid larvae that are more prone to desiccation. In line with research by Coulson (1958) and Wheelhouse (1995), the current study recorded continual increases in *O.pseudosimilis* throughout the study period, as this species has a longer emergence period than most, extending into August. It may be that the reduced reliance of optimum weather conditions within a short Spring emergence window ensures survival of this species at sample plots with low water tables. Larvae of small tipulid species are elusive and extremely understudied (A. Stubbs, 2019. Pers. comm., 9 Sept) and further autecological research would be needed to test these hypotheses.

T.immaculata is physiologically adapted to withstand high external water potentials (Freeman, 1995) and is particularly susceptible to desiccation during first-larval instars (Wheelhouse, 1995). *T.immaculata* and is a carnivorous tipulid larvae which primarily feeds upon enchytraeid worms [Enchytraeidae] (Horobin, 1971) and as such *T.immaculata* distribution will also depend on factors influencing its prey. Enchytraeid worm abundance increases with both soil moisture and temperature (Standen and Latter, 1977). As such it is possible that the hot summer of 2018 caused an increase in enchytraeid worms in areas where the water table was retained at a high enough level to support these species. This brings into question the potentially unique effects that temperature rise could have on the distribution of *T.immaculata*.

In line with the majority of research into peatland tipulid biology (e.g. Freeman, 1967; Wheelhouse, 1995), soil moisture was identified to exert greatest influence on species abundance in comparison to plant species cover, although there was much natural covariation and these intrinsically linked factors are difficult to categorically separate without further fine-scale survey work.

It is possible that plant species can independently influence tipulid abundance, effecting both nutrient content and ground temperature (Wheelhouse, 1995). Plant species such as *Sphagnum* and *C.vulgaris* can also directly influence soil moisture.

C.vulgaris was the only plant species to negatively correlate with overall tipulid abundance. This is theorised to be due to the woody roots of heather, which break up and aerate peat making it dry out but also making the ground more susceptible to flooding in heavy rain, so potentially resulting in suboptimal conditions for any tipulids with narrow soil moisture tolerance thresholds.

Within the current study, the presence of wetland tipulid species (*T.immaculata*, *T.subnodicornis* and *I.pulchella*) did not solely depend on Sphagnum, as recorded by Wheelhouse (1995) and Coulson (1962) post extreme drought. This suggests that grip blocking has enabled areas of peat to retain high enough water tables throughout the 2018 drought for tipulid populations to persist in the absence of Sphagnum microhabitats.

Tipulid abundance/biomass and community composition with time since restoration
Unblocked grips possessed significantly lower soil moisture and tipulid abundance in comparison to grips that had been blocked for just one year, supporting studies that highlight the potential rapid effectiveness of grip blocking (Price et al., 2003; Jauhiaianen et al., 2002; Worrall et al., 2007).

Previous literature has theorised that the positive effects of grip blocking on tipulid populations recorded by Carroll *et al.* (2011) will strengthen over time (Parry et al., 2014; Beadle et al., 2015), however linear increases in cranefly abundance and soil moisture was not observed with time since restoration by the current study, as 7yr-Blocked grips possessed low numbers of tipulids despite high soil moisture and the 10yr-Blocked grips possessed high cranefly abundance despite low soil moisture. The reason for the unexpected differences in soil moisture is considered likely to be due to unavoidable among-site topographical differences, which appear to override any potential influences of time since restoration. The 1yr-Blocked grips were notably shallow on flat ground and thus very

wet, whilst large peat pipes (natural subsurface drainage channels) were noted at the 10yr-Blocked grips. Peat pipes are responsible for significant water-table drawdown and subsurface erosion in peatlands (Holden, 2006) and are likely to have compromised the success of water table restoration at the 10yr-Blocked site. Artificially drained blanket peat catchments often contain greater soil pipe density than intact catchments (Holden, 2006). It is acknowledged that a larger scale study and/or long-term monitoring of individual sites would be of value to minimise variables outside of time since restoration.

The high tipulid abundance at 10yr-Blocked grips was attributed to the abundance of *O.psedosimilis* thriving in drained conditions, however, unblocked grips supported significantly fewer tipulids then 10yr-Blocked grips despite similar soil moisture levels. This is considered likely to be owing to extreme water table fluctuations that occur at open grips (Holden et al., 2011), however as with many assumptions relating to tipulid larvae, mechanisms have not yet been directly investigated, and reasons may relate to other/additional unobserved variables such as management history, pre-restoration soil moisture or peat structure.

Unobserved variables such as those listed above, in addition to factors effecting the distribution of enchytraeid worms, were also considered likely to be the reason for significantly lower tipulid abundance at 7yr-Blocked grips in comparison to 1yr-Blocked grips despite similar high average soil moisture levels.

Grip blocking reverses localised moisture reductions caused by drainage adjacent to grips (Wilson *et al*., 2010) and in line with this, overall soil moisture was higher at study plots 0.5 metres from grips as opposed to 5 metres. However, this trend was unexpectedly also the case at Unblocked grips. This could potentially reflect peat subsidence of grip banks over time down to the adjusted grip influenced water table (Williamson *et al*., 2016). The flat topography of 1yr-Blocked site is likely to explain the lack of variation that was found between 'near' and 'far' plots at this location. Further hydrological monitoring would be needed to verify these conclusions and potential explanations.

Further research is needed to fully investigate the overall influence of time since restoration on tipulid populations, either via long-term monitoring of individual sites or utilising a greater number of survey sites to decrease the influence of among-site topography.

Implications for breeding wader conservation & recommendations for further research

This study implies the relationship between tipulid abundance and soil moisture is potentially more complex than implied by recent research (Carroll et al., 2011; Pearce-Higgins & Yalden, 2004; Samsom, 2016; Pearce-Higgins, 2011), highlights the importance of monitoring rather than assuming the positive effects of restoration, and suggests caution in assuming grip blocking will result in linear increases in both soil moisture and tipulid abundance across all drained moorlands of all altitudes and conditions.

Further research is needed to investigate the overall influence of time since restoration on tipulid populations, and this research should be a conservation priority, especially in light of predictive modelling research which suggest that for every $1^{o}C$ rise in temperature, tipulid abundance will need to double in order mitigate for the adverse effects of climate change on breeding wader populations (Pearce-Higgins, 2011).

The tipulid species recorded possessed exceptionally patchy distributions and comprised species which showed affiliations to either blanket bog and dry heath habitat, obscuring any overall trend with soil moisture. It is recommended that any future tipulid data collection is documented to species level wherever possible in addition to abundance and biomass.

Much needed research into tipulid larvae autecology will enhance understanding of habitat requirements of key blanket peatland tipulids, and thus effects of management on breeding waders.

The observation that a positive relationship between tipulid biomass and soil moisture may exist, has potential use in relation to tipulid size as a useful indicator of year-round soil moisture conditions, however further research is needed to support this hypothesis. This potential trend would have important implications in terms of habitat value for foraging waders, as larger adult tipulids are generally selected by wader chicks (Schekkerman & Boele, 2009; Beintema *et al.*, 1991; Pearce-Higgins & Yalden, 2004), and adult waders will selectively feed upon largest final instar larvae as a rich source of protein prior to egg laying (Pearce-Higgins & Yalden, 2004).

It is suggested that future research and predictive modelling on tipulid populations and avian predators centres around tipulid biomass wherever possible, as suggested by Buchanan (2006). However, it is acknowledged that smaller species can be valued as prey for certain upland birds such as *M.ater* for red grouse chicks (Butterfield & Coulson, 1975; Savory, 1977). In addition, climate change has the potential to cause future phenological

mismatching of chick hatching and tipulid mass emergence dates (Pearce-Higgins *et al*., 2005). It is possible that species with comparatively lengthy emergence periods such as *O.pseudosimilis* may also be of value in this respect.

Tipulid species assemblages varied between blanket bog and dry heath vegetative species, suggesting that a greater interspersion of these will increase overall species diversity, as identified by Coulson & Butterfield (1985). If numerous species can co-exist across a heterogeneous habitat structure of hummocks and depressions, this is likely to lessen the risk of severe drought resulting in the overall loss of one dominant tipulid species, as has been recorded previously (Wheelhouse, 1995).

Considering the vast extent of moorland grip blocking that has occurred within the UK, there is a dearth of 'long-term' (>5yrs) published literature monitoring site ecology post-restoration (Parry et al., 2014; Beadle et al., 2015. Shepherd et al., 2013). Although the current study was subject to site-based constraints, it forms one of the very few papers to directly investigate post-grip blocking site ecology and highlights the crucial value of post-restoration monitoring. Recording and analysing long-term ecosystem responses to peatland restoration is key to adaptive evidence-based management, that is urgently required in the context of climate change adaption (Pullin *et al*., 2004; Sutherland *et al*., 2004) and it is recommended that this trajectory of research is continued.

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APPENDIX 1

Summary of topographical data types collected at artificially blocked grips	
Gully morphology and parameters	
Parameter	**Survey technique**
Gully width	Taped measurement between breaks of slope at upper limit of gully walls
Gully slope	Levelled height along a 10 metre stretch of ditch, avoiding the influence of dams or pools
Gully depth	Measured at gully mid-point, half way between successive blocks, perpendicular to a tape stretched between gully sides
Block Spacing	Taped in field
Grip spacing	Taped in field

Table and methods adapted from Evans