Assessing the biodiversity outcomes of Wild Oxford, an Alkaline fen ecosystem restoration project

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Declaration of individual authorship

The author confirms that this research project contains no unacknowledged work or ideas from any publication or work by any other author.

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4 October 2021
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Abstract

The study investigated the effects of the Wild Oxford ecosystem restoration project on the plant communities, floral nectar resource and invertebrate pollinators of three small valleyhead Alkaline Fens in Oxford, Chilswell Valley, Lye Valley and Raleigh Park. It found that the Wild Oxford project has substantially increased biodiversity, with a shift to more biodiverse plant communities and the establishment of Oxfordshire Rare Plant Register species in the areas undergoing restoration within 4 – 7 years. These were not the target M13 *Schoenus nigricans-Juncus subnodulosus* mire characterising undisturbed Alkaline fens in the region, but novel ecosystems closest to M22a *Juncus subnodulosus-Cirsium palustre* fen-meadow. A substantial increase in floral units, nectar resources and visits by flying insect pollinators was also recorded in Chilswell Valley and Lye Valley, demonstrating by proxy the value of fens for invertebrates. This effect was not recorded at Raleigh Park as a consequence of high intensity grazing which drastically reduced the floral and nectar resource of the Alkaline fens under restoration.

Comparison with pre-restoration plant communities and control plots determined that these changes are largely explained by ecosystem restoration activities. The intensity of cut and collect biomass removal activities, application of hand-collected seed and grazing were found to have the strongest influence on biodiversity uplift. The role of rewetting could not be fully evaluated but the high mean annual water table recorded is an essential pre-requisite for the biodiverse wetland plant communities found.

Abiotic variables also influenced biodiversity with effects of water table depth and potassium detected for all sites as a whole and a small negative effect of nitrates observed at Chilswell and Lye Valley. Calcium concentrations are high in all plots and play an important role in aiding restoration activities by limiting phosphate and significantly neutralising the effect of elevated nitrates.

Compelling evidence was found of the effectiveness of fen ecosystem restoration techniques and similar sites with high water tables and calcium rich groundwaters offer good prospects for the ecosystem techniques to be successfully replicated in other local, degraded Alkaline fens at low cost primarily using volunteers.

(Total 15,900 words excluding title pages, headings, tables and references).

Table of Abbreviations

BBOWT	Berkshire, Buckinghamshire and Oxfordshire Wildlife Trust
FIT Count	Flying Insect Timed Count
FHT	Freshwater Habitats Trust
LWS	Local Wildlife Site
MAVIS	Modular Analysis of Vegetation Information System
NVC	National Vegetation Classification
OCC	Oxford City Council
ORPR	Oxfordshire Rare Plant Register
SAC	Special Area of Conservation
SSSI	Site of Special Scientific Interest
SD	Standard Deviation
WTD	Water Table Depth

1. Introduction

1.1 Alkaline Fens

1.1.1 Physical characteristics

Alkaline fens are peat forming mires sustained by continuous flows of minerotrophic groundwater, ensuring the water table remains at the ground surface most of the year (Rydin and Jeglum 2013). The mineral-rich groundwater is derived from calcareous bedrock and enters the mire via springs and seepages (Diack *et al.,* 2013), creating saturated, anoxic conditions which prevent the breakdown of organic matter and formation of peat (Price *et al.,* 2016). The base-rich groundwater is high in minerals in particular calcium and magnesium as well as potassium, iron and sodium (Rydin and Jeglum 2013). Unlike ombrotrophic, rainwater fed bogs with acidic waters, the waters of Alkaline fens have a high pH of 5.7-8.3 (Diack *et al.,* 2013).

Their topography in valley heads means springs are sometimes visible and the fens are strongly soligenous (Diack et al., 2013), which is defined by Rydin and Jeglum (2013) as having a spring/seep line with sloping laminar water flow and no distinct open water channels.

A key feature of Alkaline fens is the formation of tufa or travertine (Lamberth 2007), a white layer of hard Calcium carbonate deposited on plant matter and the fen surface which can form thick layers (Figure 1.1). This is formed by the release of carbon dioxide from the saturated calcium bicarbonate waters as they emerge from the springs and seepages into the air, precipitating insoluble calcium carbonate (Lamberth 2007, Webb 2018) with greatest volumes of tufa produced by flat, wide aquifers rather than isolated springs (Hájek *et al.,* 2002).

Probably the most important consequence of tufa formation is the co-precipitation of Phosphorus with calcite which limits its availability to plants (Boyer and Wheeler 1989). The limitation of Phosphorus is extreme in strongly tufa-forming fens (Rozbrojova, and Hajek 2008), becoming the critical constraint to plant growth in which only low productivity, stress tolerant plants can thrive (Webb 2015b). Alkaline fens consequently have a unique plant community dependent on low nutrients and their bryophyte and vascular plant flora earns the definition as extremely rich fens (Rydin and Jeglum 2013).

Figure 1.1: Tufa deposited on woody debris and organic matter in Raleigh Park

1.1.2 Plant community

Alkaline fens are classified as H7230 by the JNCC (2019a) into short sedges of three main National Vegetation Classification groups (Table 1.1). M13 is found mainly in lowland England and forms the mire plant community in the Oxfordshire fens. It is particularly biodiverse with the greatest concentration of overall species and rare mire specialist plants (Diack *et al.,* 2013). Other communities include the widespread M22a *Juncus subnodulosus - Cirsium palustre* fen meadow which can favour slightly more fertile conditions (Diack *et al.,* 2013) and lower summer water tables (Wheeler, 2002). The key influences on the plant community structure and composition are depth to water table, mineral concentration and nutrient status (Boyer and Wheeler 1989, Wheeler 2002, Rozbrojova, and Hajek 2008 and Rydin and Jeglum 2013).

Table 1.1: Alkaline Fen NVC Plant Communities

Plant Community	Description	Location
M9 *Carex rostrata – Calliergon cuspidatum / giganteum* mire	Diverse community of sedges on noticeable lawns of brown moss. Matts of vegetation always wet.	Mainly North England
M10 *Carex dioica – Pinguicula vulgaris* mire	Sward dominated by sedges and bryophytes which can cover more than 50% of the surface. Shorter, open and less species rich than M13. Mean species 32 per $4m^2$ quadrat, 264 species in total with 32 rare mire species	Uplands, especially North West England
M13 *Schoenus nigricans - Juncus subnodulosus* mire	Moderate sward height to 50cm, complex structurally with low growing and biodiverse patches characterised by bryophytes, pools and runnels. *Molinia caerulea*, sedges, smaller floriferous plants and orchids important. Average of 31 species per $4m^2$ quadrat, 367 species in total and 39 rare mire species.	Lowland England, concentrated in East Anglia with important localised sites in Oxfordshire, especially the Cothill Basin

Sources: Elkington *et al.,* (2001), European Commission (2013) Diack *et al.,* (2013) and JNCC (2019a)

1.1.3 Importance of Alkaline Fens

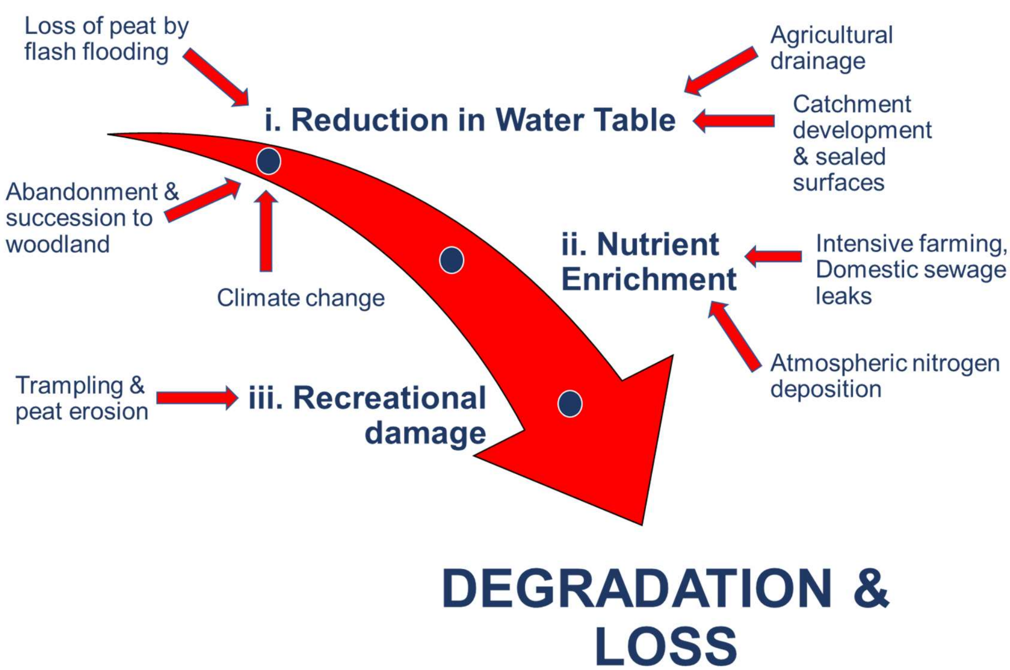
As a function of their high and unique plant diversity, Alkaline fens are considered important as biodiversity hotspots (Nielsson, 2015) especially the assemblages of brown mosses they support (European Commission, 2013). Ecologically their value extends beyond plants, sheltering and feeding many molluscs, invertebrates and birds (Rydin and Jeglum, 2013, Šefferová *et al.,* 2008) as evidenced by Wicken Fen hosting more than 8,000 species (Peh *et al.,* 2014). The ecological significance is such that Alkaline Fens are designated as an Annex I habitat under the Habitats Directive (JNCC 2019b).

Locally, many rare invertebrates are found; 120 rare species were recorded at Cothill Fen (Natural England, 2014) and 55 rare or scarce invertebrates in only 1ha at Lye Valley (Webb 2015). The fens of Raleigh Park are home to 40 species of spider, 34 beetles, 89 true flies, 32 true bugs and 17 bees, wasps and allies (Gregory, 2021) including the Nationally Rare and Vulnerable soldier fly *Oxycera analis* and 6 other Nationally Scarce or rare invertebrates. Oxfordshire's fens also have 44 and 13 species of snails and slugs respectively including 9 Nationally Rare, Notable or Local species (Gregory, 2002).

As peatlands, Alkaline fens are also carbon stores and offer a wide range of ecosystem services including water quality and flood risk prevention, local cooling and recreation (Joosten *et al.,* 2015 Lamers *at al.,* 2015). They play an important role in climate regulation, being carbon sinks in their natural state but shifting to net emitters of carbon if the peat dries out and oxidises (Joosten *et al.,* 2016) with UK peatlands estimated to emit 23,100 kt CO_2e annually (ONS, 2019)

1.1.4. Threats

Partly as a function of their dependence on reliable supplies of clean, low nutrient groundwater, lowland Alkaline fens are a fragile and sensitive habitat and subject to a range of threats (Figure 1.2). The main historical loss to date has come from drainage for agriculture with Šefferová *et al.* (2008) estimating that 95-98% of lowland Alkaline fens have been lost in the UK. They are now extremely rare and limited to small and fragmented sites (Diack *et al.,* 2013) covering only an estimated 28km^2 in the UK of which 57–75% was estimated to be in unfavourable condition (JNCC 2019b).



Sources: Diack et al., (2013), JNCC (2019a), Grand-Clement, E. et al., (2013), Lamers *et al.,* (2015), Morris (2002), Middleton (2013), Verhoeven *et al.,* (2011), Web (2014, 2015, 2018).

Figure 1.2: Threats to Alkaline Fens

Lowering the water table or loss of historical grazing or mowing for hay can accelerate transition to Willow Car and wet woodland (Morris, 2002, Middleton 2013). In Cothill Fen Snowdon (2017) recorded a shift to lower diversity plant communities dominated by *Phragmites australis* as a result of eutrophication from increased Nitrates arising from agriculture, with the diverse, light dependent, stress tolerant M13 community being outcompeted. Once degraded fens have lost their characteristic low nutrient plant specialists it is then difficult for recolonisation to occur due to chronic habitat fragmentation, loss of dispersal mechanisms and short-lived seedbanks for many wetland species (Rasran *et al.,* 2007, Hall *et al.,* 2010, Strouh *et al.,* 2012, Middleton 2013 and Lamers *et al*., 2015).

1.2 Ecological Restoration of Alkaline Fens

Given their ecological value and more recently an awareness of the carbon storage potential, restoration of degraded sites has come into increasing focus (Lamers *et al.,* 2015). Successful methods reported in the literature where grazing has ceased include mowing and scrub removal (Middleton *et al.,* 2006b, Klimkowska *et al.,* 2010, Menichino *et al.,* 2016, Nielsson 2015, Ross *et al.,* 2019 and Sundberg 2012). Primary goals are the removal of biomass to reduce nutrient levels, competition and shading from dominant successional species such as *Phragmites australis*. Middleton (2013) indicates that encroachment by woody shrubs and trees as a loss of traditional management methods needs to be countered with their removal.

Re-introduction of lost species appears to be important; Klimkowska *et al.,* (2007), Graf and Rochefort (2008) and Hedberg *et al.,* (2014) indicate application of green hay containing seeds of missing species from local fens is an effective tool for re-establishing low-nutrient plant communities. Hall *et al.,* (2010) evidenced that prospects for species returning from the seedbank were extremely limited after a decade of dominance by *Tyhpa* species with more than 17 species missing from the fen plant community of 30 years previous. Strouh *et al.,* (2012) concluded that the seedbank will be insufficient to restore degraded fens, instead it is likely to produce aggressive ruderal and competitive species needing control through mowing or grazing.

Furthermore, Decleer *et al.,* (2013) found *Pedicularis palustris* a hemiparasite of reeds and rushes now missing from many degraded fens, appears to play a key role as an ecosystem engineer without which restoration may proceed at a slower rate. They reported this species reduced the height of previously dominant *Carex acuta* from 1m to 0.4m and its biomass from 91% to 17% within just six years. The corresponding opening up of the sward and reduction in competition was correlated with a 40% uplift in plant species richness and an increase in non-carex biomass. Webb (2020) describes similar changes in plant community where it was applied during the Wild Oxford project.

In some situations hydrological changes may be irreversible and more drastic measures to create the wet year-round conditions are required, such as the deliberate removal of nutrient rich topsoil (Klimkowska *et al.,* 2007) and in former rich fens in Poland degraded by drainage (Hedberg et al., 2014). Morris (2002) found that by lowering the ground surface, former peat cuttings had delayed succession to wet woodland in Cothill Fen, keeping the water table high and partly explaining its enduring floristic diversity.

The resumption of Grazing has been used in fen restoration with the goal of simulating the ecological roles performed by extinct herbivores and then low intensity, traditional agricultural grazing (Middleton *et al.,* 2006b, Middleton 2013, Webb 2020a). Groome and Shaw (2015) found grazing at low stocking density to increase species richness in a lowland fen. However, the evidence seems mixed, with Stammel *et al.,* (2003) finding a reduction in plant species richness in grazed fen meadows, although still preferable to complete abandonment. Trampling of sensitive fen vegetation and peatland degradation by cattle were noted as problematic by Groome and Shaw (2015) and Middleton *et al.,* (2006b). It can also increase dominance of non-palatable species such as *Juncus inflexus* and reduction in flower production, as in Parsonage Moor, Cothill Fen as a consequence of keeping ponies on site too long has led to overgrazing (J Webb, personal communication 1 July 2021).

Fen Restoration is uncertain and not without problems. Lamers *et al.,* (2015) often found a lack of monitoring with the baseline plant community not recorded prior to restoration, making outcomes hard to determine. They also raised concerns that fen rewetting can inadvertently cause harm if the hydrological requirements of the target ecosystem are not understood, for example diverse *Junco-Molinion* fen meadow can shift to Sedge dominated fens if the water table is elevated year round. Another concern they raise is phosphate being released from rewetting desiccated peat, resulting in eutrophication.

1.3 The Wild Oxford Project

The Wild Oxford project is led by Berkshire, Buckinghamshire and Oxfordshire Wildlife Trust (BBOWT) working in partnership with Oxford City Council (OCC) and local community groups to secure resilient, thriving wildlife habitats in and around Oxford by engaging local people in nature and ecological restoration (BBOWT, 2019). One of the project's specific goals is to restore Alkaline fens on sites owned by OCC that had become degraded and invaded by *Phragmites australis* and Willow Carr.

1.3.1 The Study Sites

The research project studied three of the Alkaline fens undergoing restoration; Chilswell Valley and Raleigh Park to the West of Oxford on the Boars Hill escarpment and Lye Valley in East Oxford (Figure 1.3). Ecological monitoring of plant communities before or at the early stages of restoration was undertaken by Webb from 2017 onwards (2019a, 2019b, 2019c). Whilst they share common geology, each site has a different management history and plant community with only Lye Valley retaining an M13 plant community. The sites have been exposed to different restoration activities over different timeframes, but also retain some unrestored areas enabling the assessment of the effectiveness of restoration and the influence of abiotic factors both within and between sites. The ecological attributes, physical characteristics, plant community recorded by Webb (2019a, 2019b, 2019c) and threats are summarised for each site in Table 1.2.

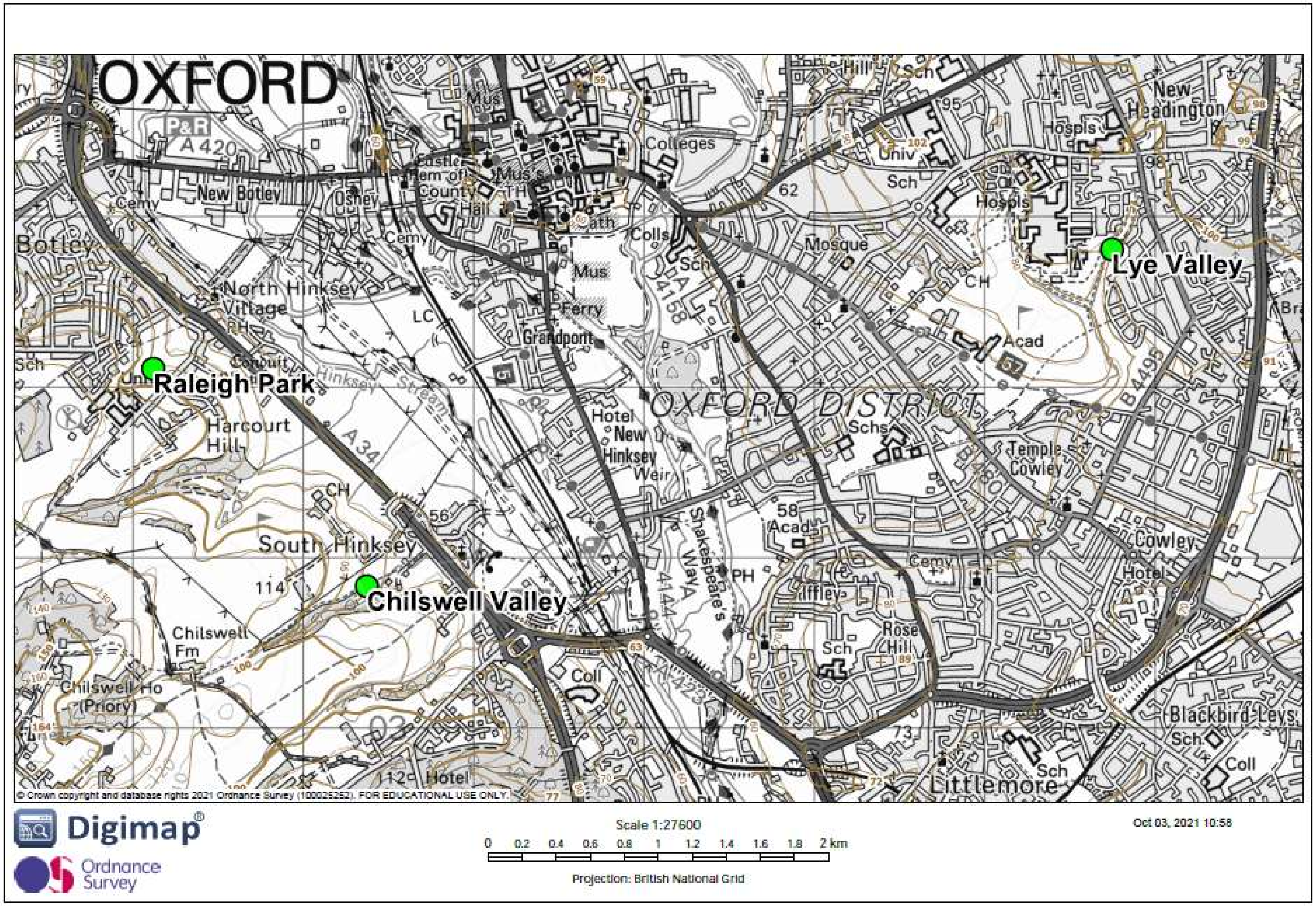


Figure 1.3: Location of Research Project Study Sites

Table 1.2: Ecological attributes, condition and threats of the sites studied

Attribute	Chilswell Valley	Lye Valley	Raleigh Park
Location	South Hinksey, West Oxford	Headington, East Oxford	Botley, West oxford
Grid reference	SP508035	SP548057	SP492052
Designation	LWS (Site 50b02)	SSSI (North), LWS (Site 50M02)	LWS (Site 40x03)
Size (ha)	6.3	4.5	9.6
Fen area	0.44 (0.35 unrestored) (Webb, 2019a)	1.45 (Webb, 2019b)	0.38 (0.5 unrestored) (Webb, 2019c)
Site Geology	Jurassic Corallian limestone and sandstone in the West, with Lower Corallian Clay and Oxford Clay to the East (Webb, 2015a)	Jurassic Corallian limestone overlying Jurassic clay (Webb 2015b)	Jurassic Corallian limestone and sandstone to the West, grading into Corallian Calcareous Grit and Oxford Clay to the East (Webb, 2018).
Catchment area*	1.3km^2	0.9km^2 (Lamberth, 2007)	0.4km^2
Fen hydrology	Fed by visible Alkaline springs and seeps on the Northern slope which have formed sheets of tufa deposits. These are stained orange / yellow with iron. High water table at or near the surface (Webb 2015a). A stream runs through the bottom of the fen.	Fed by tufa-forming, strong flowing calcareous spring lines and high water table, with a stream in the valley bottom which has become a storm drain for the local highways network carrying polluted surface water (Webb, 2019a)	Fed by calcium rich, seepage zones rather than individual springs. Abundant tufa deposits and petrified material. Water table close to surface. Some areas of red iron oxide due to the action of sub-surface chemotrophic bacteria (Webb, 2018).
Fen topography	Single unit of valley-head, sloping, soligenous fen.	Single unit of gently sloping, valley head, soligenous fen	Patchwork of valley head soligenous fens varying in slope gradient. Main fen is relatively flat.
Fen vegetation	Webb reported the areas of relic Alkaline fen undergoing restoration	East side of the stream is an intact M13 mire community. West of the stream undergoing restoration has	The main fen under restoration was reported by Webb (2019c) to be increasing in biodiversity with an

Attribute	Chilswell Valley	Lye Valley	Raleigh Park
	included the characteristic fen species (2019a): • *Anagallis.tenella* • *Carex distans* • *Galium uliginosum* • *Juncus subnodulosus,* • *Oenanthe lachenalii* • *Pedicularis palustris* • *Succisa.pratensis* The unrestored fen was dominated by *Phragmites australis*	characteristics of M22 *Juncus subnodulosus* - *Cirsium palustre* Fen Meadow dominated in areas by invasive, species poor *Phragmites australis* and *Carex acutiformis* (Webb, 2019b). In 2019 it included 20 vascular flowering plants from the Oxfordshire RPR and 14 species on the New Vascular Plant Red List for England	abundance of wetland species such as *Dactylorhiza fuchsia, Iris pseudacorus* and *Lotus pedunculatus* and the Oxfordshire RPR *Eleocharis uniglumis*
Fen condition	Fen plant diversity increased from 4 to 68 species between 2014 and 2019, around half are wetland plants (Webb, 2019a).	Webb (2019b) reported that the SSSI status is unfavourable but recovering. Species increased from 47 to 65 between 2017 and 2018.	Plant species increased from 45 to 52, with short turf and open fen returning. Drier areas dominated by ruderals (Webb, 2019c).
Threats to fen	• Groundwater nitrate pollution from adjacent agricultural land • Succession to wet woodland • Dominance of *Phragmites australis* shading and outcompeting target fen species • Erosion of peat from fen bottom due to flash flooding in stream • Vulnerability of groundwater recharge to irregular precipitation as a consequence of climate change	• Groundwater pollution from sewage leaks • Succession to wet woodland, dominance of *Phragmites australis* • Erosion from stream flash flooding • Contamination and eutrophication of M13 fen in the valley bottom from polluted stream • Reduction in spring flow due to development and area of impermeable surface in catchment • Fly tipping, arson • Vulnerability to climate change	• Invasion, dominance and out-competition by *Juncus inflexus, Typha latifolia*, *Salix cinerea* and *Salix fragilis* • Vulnerability of groundwater recharge due to climate change • catchment development • historic drainage and peat loss

*Estimated from measurements in Digimap (2021) using OS Terrain 5 and OS Terrain 50 datasets of contours and spot heights

1.3.2 Ecological Restoration Activities undertaken by Wild Oxford

Since 2014 a range of ecological restoration activities have been undertaken on the study sites, detailed in Table 1.3

Table 1.3: Description of Alkaline Fen restoration activities undertaken in the study sites through the Wild Oxford project

Restoration Activity	Description	Delivered by	Purpose
One-off activities on abandoned areas			
Removal of Carr Woodland / dense Willow scrub	Felling with chainsaws Extraction of stumps from fen by machinery	OCC and BBOWT specialist teams	• Raise water table, prevent peat loss • Remove shading and enable open, short fen sward to re-establish
Removal of Willow Scrub	Cutting with hand tools	Volunteers	• Prevent succession to Carr woodland / scrub and shading out the re-establishing target plant community • prevent peat drying out and water table dropping
Removal of Bramble Scrub	Brush cutter, hand tools	Volunteers, OCC specialist	
Rewetting	Blocking artificial drains by constructing log dams reinforced with earth and leaky log dams in stream channels	Contractors with volunteer support	• Rewet areas of former fen by reversing historical drainage and retain water within fens, raising the water table • Slow flows of surface water passing through fens in stream channels and minimise loss of peat through flash flood erosion.
Repeat activities – annually or more			
Mowing of herbaceous fen vegetation	Scything and hand raking undertaken autumn, early spring and early summer (high cuts to avoid damaging emerging target low-growing species). Arisings removed	Volunteers	• Biomass removal to lower fertility, reduce dominance of tall *Phragmites australis* and species that posing problems with vigorous growth such as *Juncus subnodulous and Juncus inflexus*
	Power scythe / tractor pulled mower (latter only if sufficiently dry enough to prevent damage to peat), all arisings removed	OCC and Contractors	

Restoration Activity	Description	Delivered by	Purpose
Maintaining blocked drains and dams	Drag-bagging cut arisings, placing them within blocked drainage channels / dams and treading the cuttings into the structure	Volunteers	• Reinforce and maintain rewetting structures, required annually to ensure water table remains close to the surface and peat is kept moist
Targeted hand removal of specific undesirable species	• Hand pulling of Alder, Birch and Willow seedlings, *Conyza canadensis, Cirsium arvense, Solanum dulcamara* and *Typha latifolia* • Hand digging of *Carex pendula, Rubus fruticosus* and *Juncus inflexus.*	Volunteers	• Prevent dominance of unwanted aggressive and vigorous species returning from the seedbank or via windblown seed, which if left unchecked outcompete/shade target plant community species and reduce net biodiversity
Grazing	Small herds of 7-10 Dexter cattle grazing and browsing freely during spring and summer in Raleigh Park only	OCC Grazier	• Reduce herb height to a low sward, prevent succession to Willow Carr, create opportunities for germination via light trampling, poaching and moving seeds around on their hooves and fur.
Re-introduction activities			
Application of Brown Hay[1]	Brown Hay raked from material cut from Lye Valley donor plots at end of October, then bagged and transported to recipient plot for prompt spreading by hand	Volunteers	• Re-introduction of missing species with short seedbank life by seed
Application of hand collected seed	Seed of *Filipendula ulmaria, Lotus pedunculatus, Lysimachia vulgaris, Lythrum salicaria, Oenanthe lachenalii, Pedicularis palustris*, *Succisa pratensis Valeriana officinalis* and *Vicia cracca* collected by hand from Lye Valley donor plots late summer when fresh and immediately sown by hand on recipient plots		

Adapted from Webb (2019a, 2019b, 2019c) and personal communication 17 August 2021

[1] Brown Hay is essentially a by-product of fen management but contains a residual amount of seed (J Webb, personal communication 02 October 2021).

1.4 Research Aims and Objectives

An increase in biodiversity within the study sites was reported by Webb (2019a, 2019b, 2019c) as a consequence of Wild Oxford. Building upon this work, the core aims of the research project were to quantify the scale of changes in biodiversity, identify the likely factors behind this change and the extent to which Wild Oxford successfully achieved Alkaline fen ecosystem restoration.

These aims were grouped around the 3 research objectives and null hypotheses described in Table 1.4

Table 1.4: Research objectives and Null hypotheses tested

Objective	Null hypotheses
Objective 1: Has Wild Oxford improved biodiversity outcomes?	"The Wild Oxford project has not increased biodiversity in the Oxford Alkaline fens"
Objective 2: To what extent do the duration and range of ecological restoration techniques influence biodiversity?	"There is no relationship between the application of different ecological restoration techniques and biodiversity."
Objective 3: How do abiotic factors influence Alkaline fen restoration?	"There is no relationship between abiotic factors, biodiversity and ecological restoration actions."

2. Methodology

2.1 Consent and risk assessments

Permission to collect data and installation water table monitoring equipment was obtained from the landowner in November 2019. All fieldwork was risk assessed for lone working, COVID19, slip and trips, inclement weather, tick and insect bites.

2.2 Data Sampling Strategy

The data sampling strategy was designed to:

1. Enable comparison with the historic plant community baseline data collected for 6 research plots by Webb in 2017 (2019a, 2019b, 2019c);
2. Assess how different restoration techniques affect biodiversity outcomes by including plots subjected to varying duration and intensity of ecosystem restoration activities, grazing, brown hay and seed application;
3. Include a control plot at each site with no history of restoration intervention to assess any differences with research plots undergoing restoration; and
4. Introduce a benchmark 'reference ecosystem' plot at Lye Valley, a representative Oxfordshire Alkaline fen M13 plant community, in keeping with the SERS principles established by Gann et al., (2019) for assessing ecosystem restoration.

Consequently 14 research plots were established across the study sites. Their names, purpose and restoration techniques experienced are described in Table 2.1 and their locations in Figures 2.1-2.3. A full description of restoration history is provided in Appendix 1.

Each research plot consisted of a 400m^2 20m x 20m square, defined by stout stakes in each corner marked with fluorescent tape to aid visibility during surveying. Space restrictions meant the LV_Bench and CV_Rest2 plots were provided as a rectangle and trapezoid of 400m^2 respectively.

Table 2.1: Research Plots Sampled

Research Plot Name	Aspect of Ecological Restoration[2]					Reason for sampling
	Duration (years)	Intensity (total times vegetation cut & collected)	Grazed (years)	Brown hay application (frequency)	Hand collected seed application (frequency)	
Chilswell Valley						
CV_Ctl1	0	0	0	0	0	Control plot, *Phragmites australis* reedbed, no historical record of cutting or restoration
CV_Ctl2(20)	0	0	0	0	0	Control plot, Willow Carr and *Phragmites australis* reedbed. Recorded in both 2020 prior to restoration commencing and again in 2021 to assess whether 1 year restoration had any notable differences
CV_Ctl2	1	3	0	0	0	
CV_Rest1	7	17.5	0	0	0	Restoration plot, no brown hay or seed added, Webb 2017 baseline plant community data available
CV_Rest2	7	17.5	0	1	4	Restoration plot, brown hay and seed added, Webb 2017 baseline plant community data available
Lye Valley						
LV_Ctl	0	0	0	0	0	Control plot, Willow Carr and *Phragmites australis* reedbed, no historical record of cutting or restoration
LV_Rest1	4.5	16	0	3	4	Restoration plot, frequent applications of brown hay and seed, Webb 2017 baseline plant community data available. Historical record of arson and groundwater eutrophication.
LV_Rest2	5	17	0	0	0	Restoration plot, no brown hay or seed added, Webb 2017 baseline plant community data available

[2] Adapted from Webb (personal communication, 17 August 2021), R Newton (personal communication 20 August 2021) and author's observations

Research Plot Name	Aspect of Ecological Restoration[2]					Reason for sampling
	Duration (years)	Intensity (total times vegetation cut & collected)	Grazed (years)	Brown hay application (frequency)	Hand collected seed application (frequency)	
LV_Bench	31	34	0	0	0	Research plot with the longest history of uninterrupted management and with plant community most closely resembling M13 Alkaline Fen vegetation. To provide a Benchmark ecological restoration reference
Raleigh Park						
RP_Ctl1	0	0	0	0	0	Control plot, Willow Carr, no historical record of restoration or grazing. Notable absence of *Phragmites australis,* unlike Chilswell and Lye Valley
RP_Ctl2	0	0	5	0	0	Grazed control plot, no historical record of restoration. Included to provide a contrast with the ungrazed control plot
RP_Rest1	4.75	6.5	5	0	0	Grazed restoration plot, low intensity restoration, Webb 2017 baseline plant community data available
RP_Rest2	4.75	6.5	5	0	0	Grazed restoration plot, low intensity restoration, Webb 2017 baseline plant community data available
RP_Rest3	3.75	6	5	0	0	Grazed restoration plot with artificial drain blocked and rewetted in Jan 2021,
RP_Rest4	2.75	4	5	0	2	Grazed restoration plot of young age and the only site in Raleigh Park receiving hand applied donor seed in low volumes

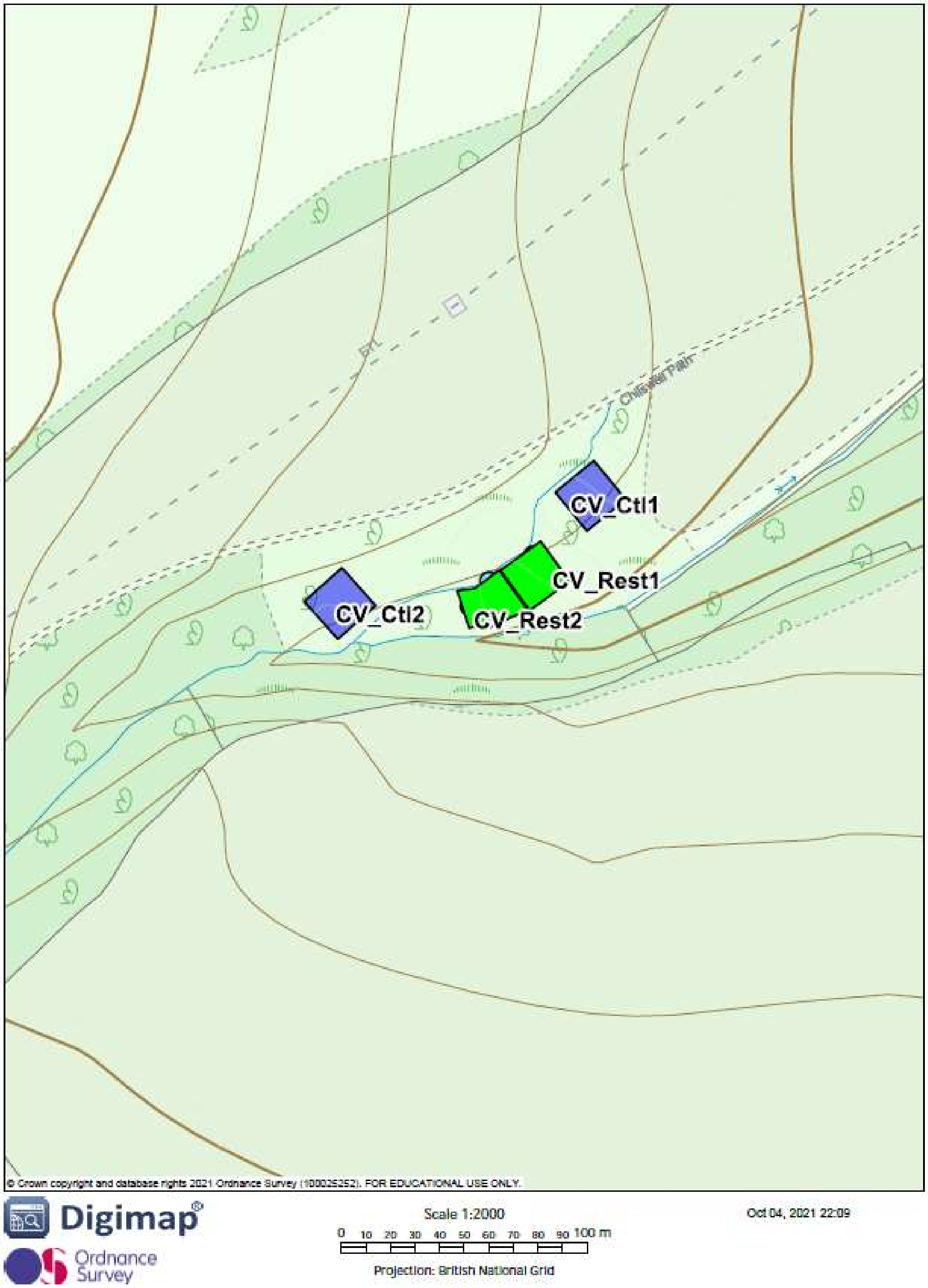


Figure 2.1: Chilswell Valley research plots

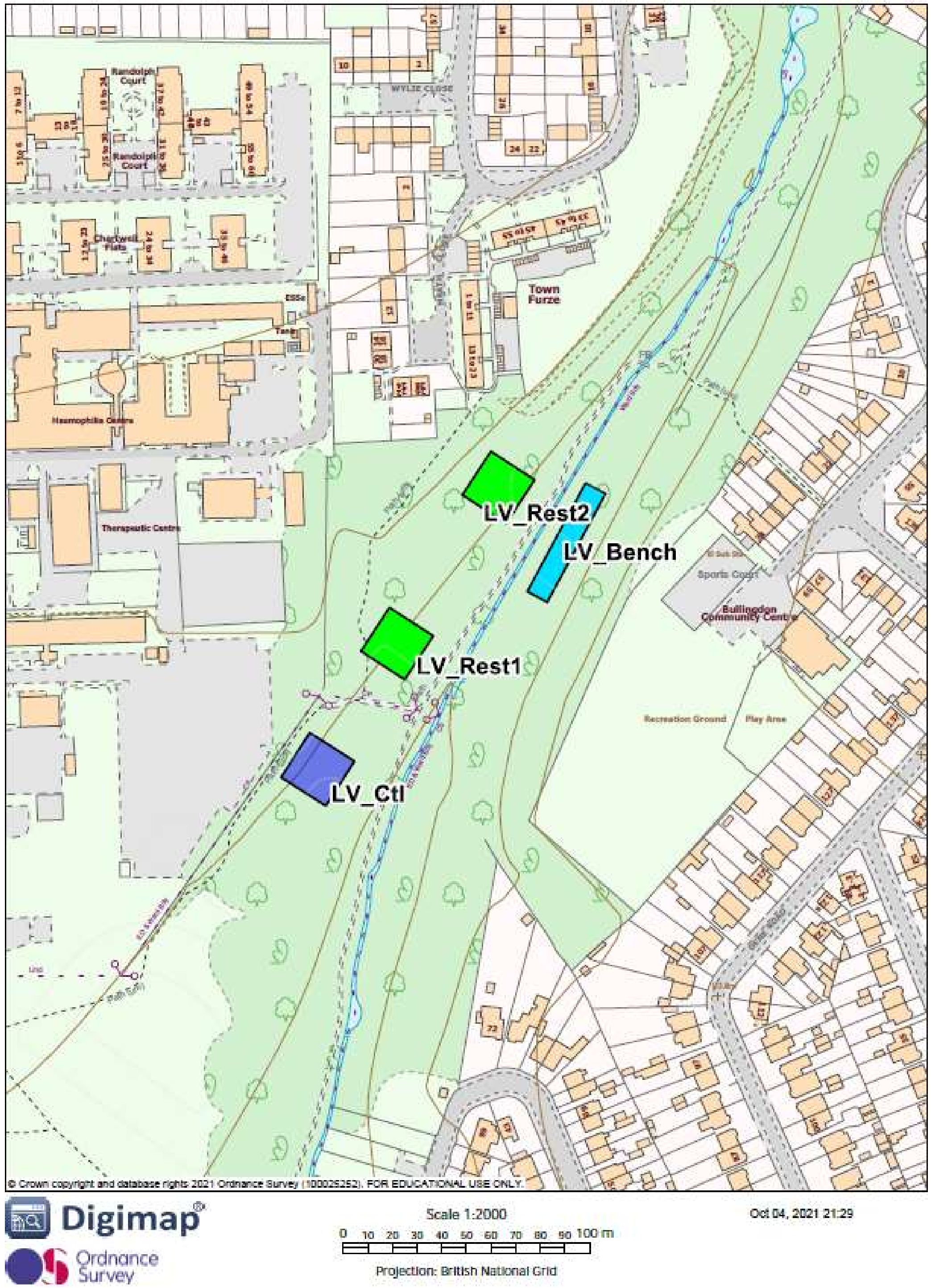


Figure 2.2: Lye Valley research plots

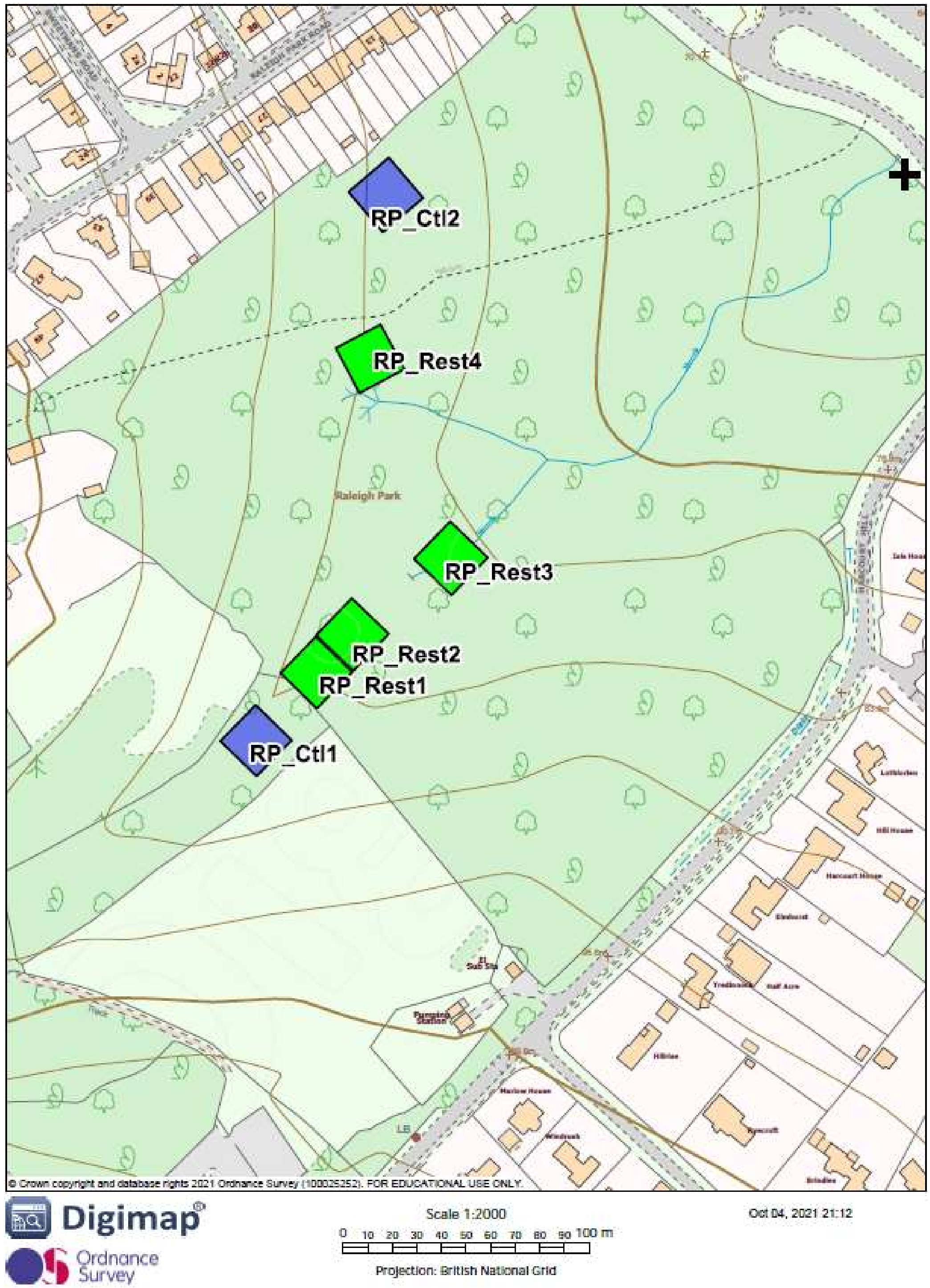


Figure 2.3: Raleigh Park research plots

2.3 Biotic Data Collection

2.3.1 Plant community and diversity

The diversity and abundance of the plant community in each research plot was sampled using the same methodology as Webb (2019a, 2019b, 2019c). This ensured the data collected could be directly compared with the baseline data recorded by Webb in 2017, providing a valuable opportunity to track any changes in plant community over time and test if this was a response to restoration or other variables. It was also consistent with methods used by Morris in 1975 and Snowdon (2017) to sample the Alkaline Fen plant community at Cothill Fen.

Forty samples of a 625cm^2 circular quadrat were taken per plot and all plant species present recorded. In total, 2.5 m^2 of each 400m^2 plot was sampled. Only individual species with at least 50% of their area within the quadrat were counted to minimise overestimation. To ensure consistency in sampling and to avoid potential bias due to the tall, dense vegetation in some research plots making it hard to see the edges of the plot, quadrat sampling was based around 5 transects taken from a known point at the bottom edge of the plot. Each transect within the plot was then walked with 8 quadrat samples taken by randomly throwing the circular quadrat ensuring no area was inadvertently sampled twice (Figure 2.4).

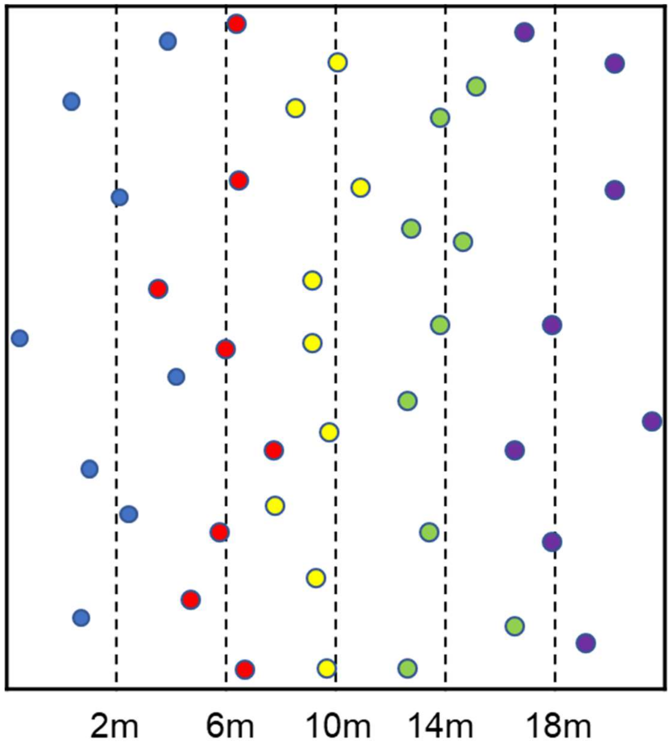


Figure 2.4 Quadrat sampling method using transects within each research plot

The use of a circular quadrat was expected to have minimised potential errors in counting, due to the reduced edge effect compared to square quadrats reported by Wheater *et al.,* 2011. Any species observed in the plot but not found within the quadrat sampling were also recorded.

The plant community in each plot was surveyed once, except the CV_Ctl2 plot sampled in 2020 and 2021 to determine any changes after 1 year of restoration. The LV_bench and all control plots were surveyed in August and September 2020 to spread workload and all restoration plots and final CV_Ctl2 plot sample taken during June and July 2021. The majority of species were identified in the field with a hand lens and identification keys (Rose 1989, Rose and O'Reilly 2006, Price 2016 and Poland and Clement 2020); in the few cases where this could not be achieved samples were collected in sealed bags and identified with the help of Judy Webb, an ecological expert in fenland vegetation.

Biotic data collected for each quadrat also included open floral units (see section 2.3.2), the height of the herb layer and where present, shading and estimated height of canopy cover, pools and tufa. Herb layer height was collected to provide an indicator of plant communities and restoration. In total, 600 quadrat samples of the plant community were recorded.

2.3.2 Floral nectar resources

As there was insufficient resource for a full invertebrate survey, the potential value of fens for insect pollinators was instead used as a proxy by estimating the total floral nectar sugar resource value of the research plots. Firstly, the number of open floral units within each quadrat were counted by plant species during the plant community survey. To aid sampling in the field, Floral units were defined as those a flying insect could reach all nectaries by walking across rather than flying between (Baude *et al.,* (2016). For example, a single flower of *Pedicularis palustris* and a capitulum of tightly packed florets of *Jacobaea erucifolia* were both recorded as a single floral unit (Figure 2.5)

Figure 2.5 Example of circular quadrat containing floral units of *Pedicularis palustris* (left) and *Jacobaea erucifolia* (right)

Secondly, a database of sugar nectar values per floral unit was compiled for the 63 species recorded flowering within quadrats (Appendix 2). The nectar resources per quadrat and research plot were then calculated by multiplying the number of floral units by the species nectar value. This was based on the Agriland database of empirically measured nectar sugar content for 260 UK wildflowers, shrubs and trees developed by Baude *et al.,* (2016). Sugar nectar values were also sourced from Gutowski (1990), Olivencia and Alcaraz (1993), Comba *et al*., (1999), Gallego Piñol *et al*., (2012), Kowalkowska *et al*., (2015), Claessens and Kleynen (2016), Hicks *et al*., (2016) and Timberlake *et al.,* (2019). Where sugar nectar values were unavailable, the value for the nearest genus or family was used. For 6 species a sugar nectar value could not be found and these were assigned a value of 0 to avoid over-estimating nectar resource.

To ensure fair comparison in nectar values, a supplementary survey of floral units in all research plots was conducted simultaneously during peak flowering season between 16 and 27 July 2021, with herb layer height also recorded. Combined with the surveys undertaken during August and September 2020 and June 2021, this ensured the phenology of flowering fen vegetation was recorded through the bulk of the flowering season. This generated a total of 1,000 quadrat samples.

2.3.3 Insect Pollinators

To supplement the nectar resource estimates, Flower-Insect Timed (FIT) counts were undertaken to provide a snapshot of the actual visitation and utilisation of floral resources within each research plot by groups of insect pollinators, following the methodology developed by the UK Pollinator Monitoring Scheme (UKPoMs 2021).

This involved selecting a single plant species in flower, placing a 50cm x 50cm quadrat over the flowers (Figure 2.6), recording the number of flowering units and counting all insects landing on or walking over the target species for 10 minutes. Target plant species were limited by those in flower, but species occurring most frequently and representative of that particular research plot were sampled where possible. Species with high nectar resources, in particular *Cirsium palustre*, *Eupatorium cannibinum, Lythrum salicaria* and *Vicia cracca* (Baude *et al.,* 2016, Hicks *et al*., 2016 and Timberlake *et al.,* 2019) were all sampled. Although the exact nectar value was unknown, *Pedicularis palustris* was targeted as Macior (1993) found the species was entirely reliant on bumble bees for pollination and Webb (2020b) indicated it was an important food resource.

Insects were classified into simple groups (Bumble Bees, Solitary Bees, Butterflies etc) and any that could not be identified counted as “Other insects”. Each individual insect was counted only once. Basic information about quadrat connectivity to other floral units, shading and weather was also collected. FIT Counts were undertaken weekly in each research plot from end-May 2021 to mid-August 2021 in dry weather when the temperature was above 15°C. This generated 11 counts for the research plots in Chilswell Valley and Lye Valley and 10 in Raleigh Park, yielding a total of 148 FIT counts.

Figure 2.6: FIT count of 17 floral units of *Pulicaria dysenterica* in Raleigh Park (RP_Rest3) showing utilisation by *Syrphidae* hoverflies (inset)

2.4 Abiotic data collection

2.4.1 Hydrology

A dipwell was installed in the centre of each research plot between mid-July and early August 2020 using an auger according to the methodology recommended by Rothero *et al.,* (2016) to enable measurement of Water Table Depth (WTD) from the surface (Figure 2.7). Where possible, at least one 2m dipwell was installed per plot in case drought meant the WTD dropped below 1m; this was not achieved in Lye Valley due to the geology. The stratigraphy of each dipwell core was recorded to supplement understanding of peat depth and the superficial deposits beneath the study sites (Appendix 3).

Figure 2.7: 2m dipwell, auger and core, Raleigh Park (RP_Rest3b test core)

Due to the presence of grazing cattle, all dipwells in Raleigh Park were sunk flush with ground level with stakes and a drain cover secured with pegs to stop cattle trampling and pushing the dipwell below ground level (Figure 2.8).

Figure 2.8: Dipwell protection from cattle at Raleigh Park (RP_Rest4)

WTD was measured weekly in every dipwell for 12 months from early August 2020 using an Eijkelkamp dipwell water level tape, or a ruler where water levels were 10cm or less from the top of the dipwell. Research plots CV_Ctl2, LV_Bench, RP_Ctl1 and RP_Rest3 also had an In-Situ RuggedTroll100 data logger suspended by cable from the cap sitting at the very bottom of the dipwell, programmed to automatically record hourly WTD readings by measuring the pressure of water above the sensor. Data was collected and downloaded every 6 months using win-situ software to calibrate the readings to local atmospheric pressure using an In-Situ BaroTroll100 located in the author's garden in Botley.

2.4.2 pH and Conductivity

Water pH within each dipwell was measured weekly from January 2021 until June 2021 using a Jenway 570 electronic pH meter loaned by Oxford Brookes University. The sensor was placed in the dipwell for 1 minute before the reading noted. In between readings the meter sensor bulb was rinsed in distilled water and stored in laboratory pH 4.0 solution when not in use. Calibration was undertaken monthly according to the device instructions. Samples were taken directly from the dipwell rather than open water or mats of saturated surface vegetation as Tahvanainen and Tuomaala (2003) found these provide readings most representative of actual pH values. Conductivity was recorded using a Jenway 470 Conductivity Meter at the same time as the pH readings using the same methodology. The sensor unit was stored dry and re-calibration was unnecessary.

2.4.3 Water chemistry

The original plan had been to collect samples each season and measure nutrient loads in the laboratory. The COVID19 pandemic and time pressures rendered this unworkable and instead the author paid for one sample to be professionally laboratory tested. This involved collecting a 400ml sample from each dipwell on 2 February 2021 using a 100ml syringe and plastic tubing, rinsed with distilled water between samples. The samples were tested by Chemtech Environmental within 24 hours for nitrates and phosphates using Ion Chromatography and for dissolved Calcium, Iron, Potassium and Sodium using Inductively Coupled Plasma Mass Spectrometry. pH was also measured using a pH meter as a cross-reference for the field methods.

To supplement the laboratory testing for key potential pollutants, spring and summer levels of nitrates and phosphates in each dipwell were measured in March and July 2021 using Kyoritsu Pack Tests supplied by the Freshwater Habitats Trust. Whilst low resolution, Biggs *et al.,* (2016) found these sufficient to differentiate polluted from clean water. Where the kits could not be deployed directly into the dipwell, a 100ml syringe and plastic tubing was used, rinsed with distilled water between samples.

2.4.4 Peat depth

Peat depth was measured manually using a 1m peat probe with part open 25mm diameter barrel showing the core sample (Figure 2.9). This enabled the depth of transition from peat to clay or sand to be visually identified. Parry et al (2014) and NatureScot (2020) consider this a reliable method for estimating peat deposits provided the soil at the base of the probe can be sampled and samples are repeated 2-3m within the vicinity of the probing point. Peat was probed every 2m along two perpendicular transects, generating 20 peat core samples per research plot. The depth of peat and other deposits, peat colour, wetness, presence of tufa, marl and transition zones were visually recorded. Where peat depth exceeded that of the corer for analysis purposes it was recorded as 1m to avoid overestimating peat deposits. To ensure accuracy, sampling was undertaken mid-March to early May 2021 before vegetation growth obscured the ground surface. A total of 280 peat core samples were recorded.

Figure 2.9 Peat probe with peat core sample, Chilswell Valley (CV_Rest1)

2.5 Data analysis

Plant community data, floral units, sugar nectar values, flying insect counts, dipwell, pH and conductivity measurements were inputted into excel and basic descriptive statistics (mean, minimum, maximum, median) and summary tables generated. Statistical analysis was undertaken with R statistical software using Tidyverse and the Vegan community ecology package.

2.5.1 Analysis of biotic data

Plant species abundance was determined per research plot by expressing as a percentage the total number of times a species was found in each quadrat (Bullock, 1996). This and the data collect by Webb in 2017 was inputted to an excel table look-up formula to convert the species names in the datasets to match those used by NVC. The species recorded in every quadrat were then loaded into the MAVIS v1.03 package following the MAVIS manual (Smart *et al*., 2016) as a .txt file per research plot. This was time consuming but considered more accurate than using constancy values. MAVIS was then run to calculate an NVC plant community match for each plot; the purpose was identify differences in the plant communities between plots and any changes between 2017 and 2021 in plots undergoing ecological restoration activities.

To identify any spatial and temporal differences in species richness and diversity between research plots, diversity indices (Shannon-Wiener Index, Gini-Simpson Index, the Fisher Alpha Index, Renyi Diversity and Pielou's Evenness) were calculated in R for the presence absence project data and Webb's baseline 2017.

2.5.2 Analysis of abiotic data

WTD data gathered from the 4 data loggers was used to provide greater detail for research plot WTD measurements. First the data logger dataset was checked against manual readings and the mean difference between data logger and manual readings was calculated and the data logger dataset adjusted accordingly. Data logger and manual readings were then plotted on scatter plots with smoothed conditional means to enable visual comparison. The Conductivity and pH dataset was scrutinised and adjusted due to problems with instrument failure; sections 3.6.1 and 3.6.2 provide full details.

2.5.3 Inferential statistical data analysis – T-Tests

T-Tests were undertaken on plant community species richness datasets for the 2017 and 2021 research plot datasets, to determine any statistical significances. Data was first checked for normally distribution using a Shapiro-Wilk normality test. Unpaired Two Sample T-Tests were used to compare different control and restoration plots as the data being compared was drawn from different populations. Where data was not normally distributed the unpaired two-samples Wilcoxon test (Mann-Whitney test) was used as a non-parametric alternative. This process was repeated for the sugar nectar and FIT count datasets to compare one control and one restoration plot (receiving the greatest restoration activities) per study site. Details of the plots compared, data and tests used are summarised in Table 2.2.

Table 2.2: T-Tests conducted on research plots datasets

Research plots compared	Statistical Method			Dataset			Comment	Purpose
	Welch Two Sample T-Test	Unpaired Two Sample T-test	Unpaired two-samples Wilcoxon test	Species richness	Nectar Sugar	Flying Insects		
CV_Rest1 2017 vs 2021	✓			✓			Used 40 values of mean species per quadrat per research plot. Note that 2017 values per quadrat were not available for the Rest_2 plots.	To identify any statistically significant difference after 4 years of restoration
LV_Rest1 2017 vs 2021	✓			✓				
RP_Rest1 2017 vs 2021	✓			✓				
CV_Ctl2 2020 vs 2021	✓			✓			Used 40 values of mean species per quadrat per research plot	To identify any statistically significant difference after 1 year of restoration
CV_Ctl1 vs CV_Rest2,	✓			✓			Used 40 values of mean species per quadrat per research plot. Note that the F-Test for variance indicated a Welch Two Sample T-Test was required.	To identify any statistically significant difference between a control plot and the restoration plot receiving the greatest restoration intervention per study site for species richness, sugar nectar values and flying insects
			✓		✓		Nectar sugar; 40 values of mean nectar per quadrat. Flying insects. Data was not normally distributed so Wilcoxon test used;	
		✓				✓	Used 11 values for total FIT count per research plot.	

Research plots compared	Statistical Method			Dataset			Comment	Purpose
	Welch Two Sample T-Test	Unpaired Two Sample T-test	Unpaired two-samples Wilcoxon test	Species richness	Nectar Sugar	Flying Insects		
LV_Ctl vs LV_Rest1		✓		✓			Used 40 values of mean species per quadrat per research plot.	
			✓		✓	✓	Nectar sugar; 40 values of mean nectar per quadrat. Flying insects; 11 values for total FIT count per research plot.	
RP_Ctl1 vs RP_Rest1		✓		✓			Used 40 values of mean species per quadrat per research plot.	
			✓		✓		Nectar sugar; 40 values of mean nectar per quadrat. Data was not normally distributed so Wilcoxon test used.	
		✓				✓	10 values for total FIT count per research plot.	

2.5.4 Inferential statistical data analysis – Dissimilarity

Dissimilarity between research plots was estimated using the R-vegdist function for the plant species abundance dataset using the "Bray Curtis" method, a common tool in ecology to quantify differences between sites based in terms of species composition and number (Statology, 2021).

2.5.5 Inferential statistical data analysis – Hierarchical Cluster Analysis

HCA was performed on the entire plant species abundance dataset after Zelený (2019), first using the 'dist' function to create a matrix measuring distance between research sites in terms of plant community composition, secondly using the 'hclust' algorithm to cluster the samples and finally plotting outputs as dendrograms enabling visual comparison.

As noted by Shreeve, 2019a and (Zelený, 2019), HCA classifies sample data into groups with similar features that set them apart from others, enabling them to be clustered into groups sharing similar characteristics. The purpose was to understand which, if any, research plots possessed sufficient similar characteristics to be grouped together and comprehend the scale of difference.

2.5.6 Inferential statistical data analysis – Principal Component Analysis

PCA was undertaken to identify relationships between data. It is a multivariate analysis tool which condenses information from multiple variables into fewer dimensions, plotting them in a theoretical multi-dimensional space, visually enabling correlations between variables them be identified (Wildi, 2013). The 'prcomp', 'fviz_pca_var' and 'fviz_eig' functions were used to plot a series of PCA comparing the full range of biotic data with abiotic and restoration variables against each other to determine any relations and correlations to be identified.

2.5.7 Inferential statistical data analysis – Correlation Matrix

Finally, correlation matrices using the same dataset variables for the PCA were generated and plotted visually using the 'rcorr' and 'corrplot' functions respectively. This is an effective method of summarising large volumes of data to visually output observable relationships between data variables.

2.5.8 Inferential statistical data analysis – ANOVA

Linear regression modelling and ANOVA (Analysis of Variance) were undertaken using the R-"lm" and R-"anova" functions to establish any statistically significant variation in how plant community species richness, floral units, nectar values and flying insect pollinators changed according to the independent categorical variables associated with different ecosystem restoration activities. This same process was then followed for abiotic variables to determine any possible relationships with plant community species richness only. The restoration and abiotic variables tested are summarised in table 2.3.

Data was not checked for normal distribution because ANOVA is deemed to be a robust statistical test where sample sizes are large (Statology, 2019). By combining this with the linear regression model it served to check the statistical significance of the model outputs.

Table 2.3 Variables examined by Linear regression modelling and ANOVA

Biotic variables	Definition
Plant community species richness	Mean species per quadrat (40 values per research plot)
Floral units	Mean floral units per quadrat (40 values per plot)
Nectar	Mean nectar sugar (mg) per quadrat (40 values per plot)
Flying insect pollinators	Total Insect count (10-11 values per plot)
Restoration treatment variables	**Definition**
Duration	Number of years research plot managed (0 for control plots)
Intensity	Total number of times vegetation cut and all arisings removed (0 for control plots)
Brown hay application	Total number of times brown hay applied (0 if none)
Hand collected seed application	Total number of times seed applied (0 if none)
Grazing	Number of years grazing (0 if none)
Abiotic variables	**Definition**
WTD	Mean of annual WTD by dipwell (cm)
Minimum WTD	Minimum WTD value by dipwell (cm)
Peat depth	Mean of peat depth by research plot (cm)
pH	Mean of pH by dipwell (14 values per research plot) and 1 laboratory measured value
Conductivity	Mean of conductivity (17 values per research plot), MicroSiemens (µS)
Nitrates	Mean of 2 test kits and 1 laboratory measured value (ppm / mg/l)
Phosphates	Mean of 2 test kits and 1 laboratory measured value (ppm / mg/l)
Calcium	single laboratory measured values (mg/l)
Iron	single laboratory measured values (mg/l)
Potassium	single laboratory measured values (mg/l)
Sodium	single laboratory measured values (mg/l)

3. Results

3.1 Data reliability

Whilst the COVID19 pandemic delayed some elements of the project set-up, the data sampling framework proved robust in the field. The vegetative sampling method had already been proven and presented no problems. The citizen science approach of the FIT count method proved an effective mechanism for gathering data on flying pollinators without necessitating extensive time to identify insects to species level.

Some problems were encountered during data collection notably timing of flowering and instrument malfunction whilst nectar values for 6 species were missing. The solutions to these and other issues are explained in Table 3.1; consequently, they are not considered to have had an adverse effect on data reliability.

A considerable amount of time and effort was spent designing a robust project framework and data collection methods. Great care and attention was paid throughout the fieldwork to consistently adhere to the specified methodology and deliver quality data. Overall, the data collected is considered accurate and reliable for the purposes of scientific scrutiny and analysis.

Table 3.1: Data collection problems and solutions deployed

Problem encountered	Solution deployed / comment	Rating of Impact on data with solution
The cold, late spring and wet May delayed flowering by a few weeks compared to previous years, meaning that fewer flowers were present than anticipated in the June surveys	• Although the majority of research plots had already been sampled by this point, a supplementary flower survey was undertaken between 16 and 27 July 2021, ensuring that all research plots had floral units counted as close to peak flowering time as possible. This also enabled fair comparison between plots to the same timeframe. It avoided the problem of comparing one plot in early June when many flowers had not emerged and another in late July, which would not have been a fair and equal comparison between plots of floral unit and nectar content.	**GREEN – no adverse effect**
Difficulties in obtaining nectar values for all species found flowering in the field; some species have a value of '0', others required the use of nearest relative proxy values	• Despite intensive searches, neither a direct sugar nectar value or closest relative proxy value could be found in the literature for 6 species; *Centaurium erythraea, Filipendula ulmaria, Humulus lupulus, Hypericum tetrapterum, Mentha aquatica and Parnassia palustris*. These were assigned a value of 0 to avoid potentially assumptions, not a problem for *Humulus lupulus* (wind pollinated) or *Filipendula ulmaria* and *Parnassia palustris* which have very little nectar (J Webb 2021, personal communication 2 October). Nevertheless this likely means the nectar resource has been under-estimated, in particular for *Hypericum tetrapterum* which was numerous and floriferous. • Uncertainty should be acknowledged around the use of proxy nectar values from closest available relatives, in particular for key wetland species like *Pedicularis palustris* which was very floriferous and the true value could have a dramatic effect on total plot nectar value estimates	**AMBER – limited adverse effect; possible underestimation of nectar values. However, this is considered a more robust approach and less problematic than overestimation**
Difficulties in identifying *bryophytes* and *Marchantiophyta* to species level.	This was dealt with by: • Obtaining identification advice and verification from local expert Judy Webb; and • restricting identification to family/genus level to avoid potentially incorrectly identification to the wrong species	**GREEN – no adverse effect**
The pH and Conductivity meters	• Both units were replaced with newer models, resolving the problem	**GREEN – no adverse effect**

Problem encountered	Solution deployed / comment	Rating of Impact on data with solution
loaned from Oxford Brookes University malfunctioned in the field, producing erroneous data	• Erroneous data was identified and removed post-survey, not forming part of the data analysis. Sufficient viable data remained for analysis; this is fully described in section 3.6.1 and 3.6.2.	
Cool, wet weather delayed some of the FIT Counts, meaning that the Raleigh Park research plots had one less count than the Chilswell Valley and Lye Valley research plots	• It was essential to ensure the FIT Counts were all conducted under reasonable comparable weather conditions (dry, minimum of 15C and not too windy) otherwise flying insect pollinators would not be present regardless of floral nectar resource availability and comparison between plots would be suspect. Delaying and rescheduling was therefore unavoidable and missing one count at Raleigh Park was not considered to have affected the reliability of the dataset given the overall sample number	**GREEN – no adverse effect**
Vegetation surveys were undertaken some time apart; 10 months in the case of the control plot surveys, potentially affecting herb layer height comparison	• To overcome this problem, herb layer height was recorded again at that same time as the supplementary flower survey between 16 and 27 July 2021. This value was used for all herb layer heigh comparison between research plots.	**GREEN – no adverse effect**
Some plot boundary markers were knocked over by cattle / vandalised / by restoration activities	• Where possible the stakes were replaced, but in practice they were unnecessary as the author had visited the plots every week for over a year and knew them very well.	**GREEN – no adverse effect**

3.2 Biotic – plant community

3.2.1 Species richness

Figure 3.1. compares the total number of plant species recorded by research plot, including the data collected from Webb in 2017. The full species list and plant abundance per plot can be found in the raw data tables (Appendix 4). Also included are plots with plants on the Oxfordshire Rare Plant Register (OFG, 2015).

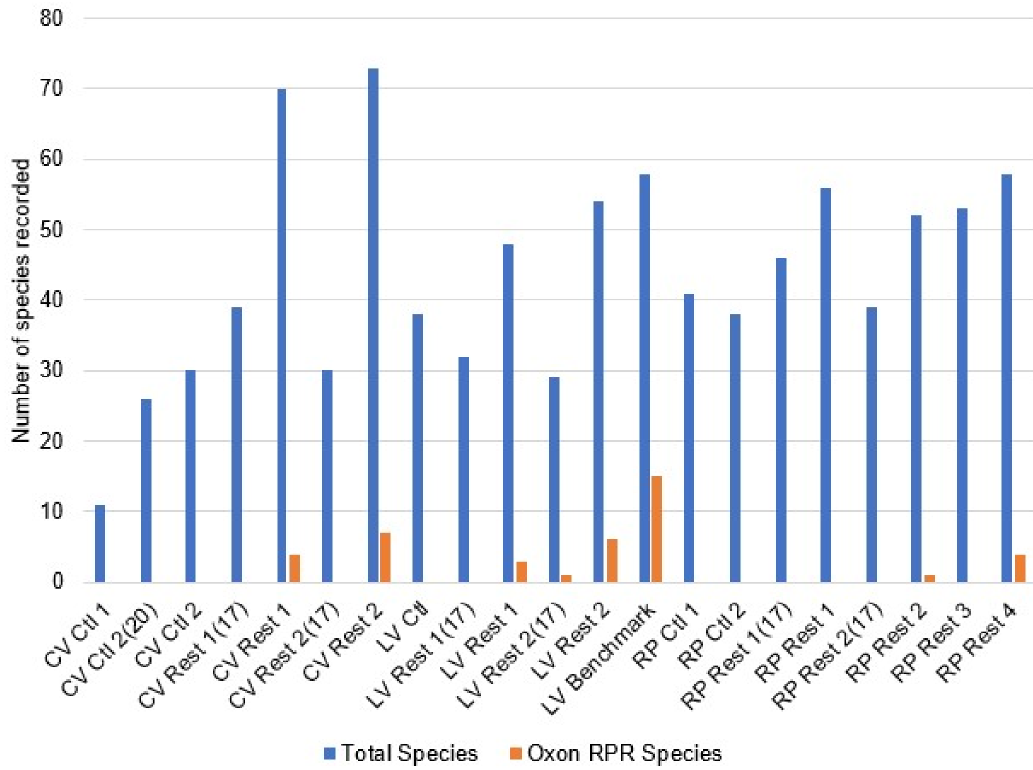


Figure 3.1: Total species and ORPR species recorded per research plot

Table 3.2 details the ORPR species present and their abundance in each plot. The restoration plots have more species than control plots, especially at Chilswell Valley. In 2021, ORPR species were only recorded in the restoration plots with *Pedicularis palustris* by far the most abundant ORPR species. Just 1 plot, LV_Rest2(17) contained a single species on the ORPR, *Pedicularis palustris,* prior to restoration.

Table 3.2: Abundance of all Oxfordshire RPR Species found in research plots

Oxfordshire RPR Species		Abundance in research plot (as % of quadrats sampled)							
		CV Rest 1	CV Rest 2	LV Rest 1	LV Rest 2(17)	LV Rest 2	LV Benchmark	RP Rest 2	RP Rest 4
Anagallis tenella	Bog Pimpernel		15%	3%		25%	48%		
Carex dioica	Dioecious Sedge						*		
Carex distans	Distant Sedge	3%							
Carex lepidocarpa	Long-stalked Yellow-sedge	*	8%			20%	33%		
Carex pulicaris	Flea Sedge						*		
Cirsium dissectum	Meadow Thistle						5%		
Eleocharis quinqueflora	Few-flowered Spike-rush						30%		
Eleocharis uniglumis	Slender Spike-rush							25%	
Eriophorum angustifolium	Common Cottongrass						25%		
Molinia caerulea	Purple Moor-grass						63%		
Oenanthe lachenalii	Parsley Water-dropwort		10%	63%		18%	8%		*
Ononis spinosa	Spiny Restharrow								*
Parnassia palustris	Grass of Parnassus		*			*	8%		*
Pedicularis palustris	Marsh Lousewort	28%	93%	25%	3%	60%	75%		10%
Pinguicula vulgaris	Common Butterwort						5%		
Schoenus nigricans	Black Bog-rush						5%		
Triglochin palustris	Marsh Arrowgrass		10%			15%	63%		
Valeriana dioica	Marsh Valerian	*	3%				38%		
Total RPR species in research plot		**4**	**7**	**3**	**1**	**6**	**15**	**1**	**4**

*Observed in research plot but not present within quadrats

Table 3.3 expresses the percentage increase in species richness for 6 restoration plots where both 2017 and 2021 data available. This shows substantial increases of 22% to 143% in species observed, with Chilswell Valley and Lye Valley seeing the greatest increase. Considered as mean species per quadrat the increases in species richness are higher, at least doubling across all plots, confirmed as a statistically significant relationship by T-Tests. After just 1 year of restoration, the CV_Ctl2 plot saw an uplift of 25% in mean species per quadrat.

Figure 3.2 provides a box plot displaying the number of species found per quadrat in each research plot. This illustrates:

- that median species number and their abundance is notably higher in restoration plots and the LV_Benchmark than control plots across all sites; and
- variation in species found per quadrat within control plots tends to be less than that of restoration plots.

Together, these findings suggest that plots undergoing restoration have a greater plant biodiversity than the control plots, which by contrast had a more uniform composition and lower species diversity. This is reinforced by comparison of pairs of control and restoration plots (receiving the greatest intervention) from each site (Table 3.4). This demonstrates that mean species per quadrat values are higher in the restoration plots than the equivalent control plot for that site. T-Tests confirm this relationship to be statistically significant (p-value <0.05 indicates the difference between datasets is not random).

Table 3.3: Change in species richness (observed and mean per quadrat) for plots in 2017, 2020 and 2021 and statistical significance

Research Plot	Total species observed					Mean species per quadrat (n=40)					T-Test		
	2017	2020	2021	Change	% uplift	2017	2020	2021	Change	% uplift	t-value	df	p-value
CV_Ctl2		26	30	+4	15%		5.2	6.5	+1.3	25%	3.585	77.7	0.00059
CV_Rest1	39		70	+31	79%	6		12	+6	100%	15.799	61.2	< 2.2e-16
CV_Rest2	30		73	+43	143%	2017 data unavailable							
LV_Rest1	32		48	+16	50%	4.2		8.4	+4.2	100%	9.617	70.6	1.82E-14
LV_Rest2	29		54	+25	86%	2017 data unavailable							
RP_Rest1	46		56	+10	22%	4.4		11	+6.6	150%	11.748	64.5	< 2.2e-16
RP_Rest2	39		52	+13	33%	2017 data unavailable							

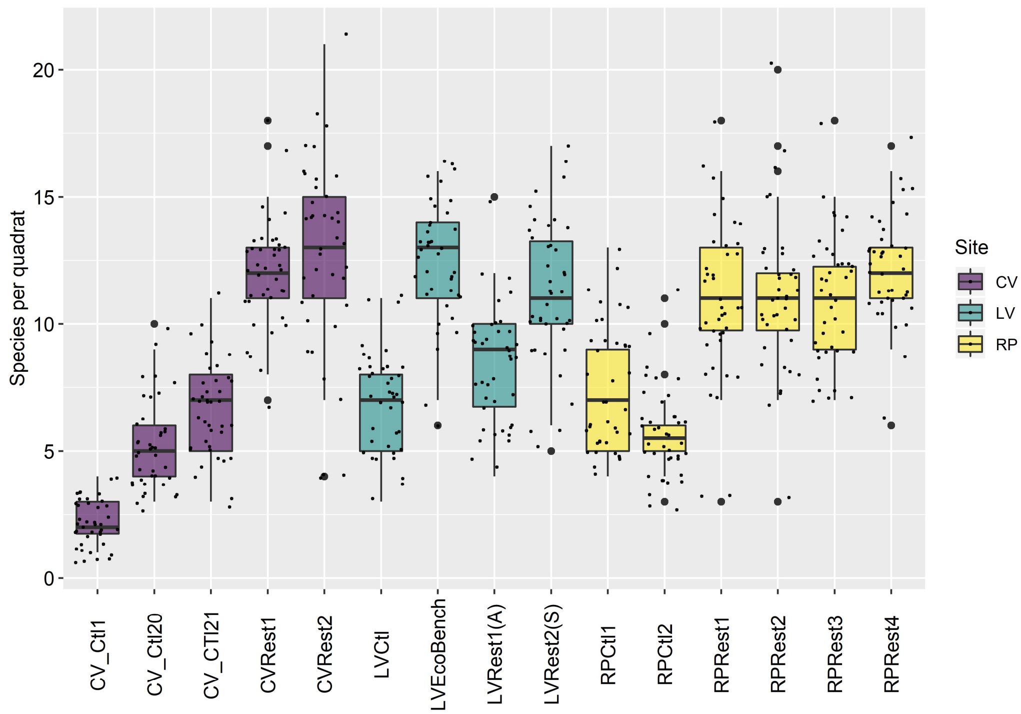


Figure 3.2: Box plot of species per quadrat by research plot

Table 3.4: Uplift in mean species per quadrat and T-Tests between pairs of control and restoration plots

Research plot pair	Difference between restoration plot vs control (mean species per quadrat)	Unpaired 2 samples T-Test conducted on species per quadrat (n=40 per plot)		
		t-value	df	p-value
CV_Ctl1 vs CV_Rest2	+10.6 (491% increase)	-17.047	42.983	< 2.2e-16
LV_Ctl vs LV_Rest1	+1.4 (19% increase)	-2.8686	78	0.005302
RP_Ctl1 vs RP_Rest1	+3 (47% increase)	-5.5509	78	3.775e-07

3.2.2 NVC plant community

Table 3.5 displays the highest match with the NVC plant community categories for each research plot (Appendix 5 details the top 3 NVC matches per plot). Matching was not high, with only 29% of plots achieving a match of 50% or more (highlighted green). Key patterns to note:

- The CV_Rest2 and LV_Rest2 restoration plots have shifted from *Phragmites australis* dominated S26 and S4 communities to M22a *Juncus subnodulosus-Cirsium palustre* fen-meadow between 2017 and 2021.
- RP_Rest1 and RP_Rest2 restoration plots have changed from OV26 *Epilobium hirsutum* community to M22a *Juncus subnodulosus-Cirsium palustre* fen-meadow;
- M22a is a more biodiverse plant community (Huxley-Lambrick, 2002), so the shift from S26, S4 and OV26 demonstrates an increase in biodiversity;
- The NVC community for CV_Rest1 and LV_Rest1 restoration plots remain little changed since 2017 (although the matches are low);
- The control plots are either *Phragmites australis* or *Epilobioum hirsutum* communities characterised by low species diversity (Huxley-Lambrick 2002, Rodwell 1997) and;
- Only the LV_Bench plot matches the NVC M13 plant community typical for herb-rich Alkaline Fens

Table 3.5: NVC plant communities matched using MAVIS

Research Plot	NVC	Description	% Match
CV_Ctl1	S4a	S4a *Phragmites australis* swamp and reed-beds, *Phragmites australis* sub-community	53.0
CV_Ctl2(20)	S26	S26 *Phragmites australis-Urtica dioica* tall-herb fen	48.2
CV_Ctl2	S26	S26 *Phragmites australis-Urtica dioica* tall-herb fen	48.5
CV_Rest1(17)	OV26	OV26 *Epilobium hirsutum* community	38.2
CV_Rest1	OV26	OV26 *Epilobium hirsutum* community	41.6
CV_Rest2(17)	S26d	S26d *Phragmites australis-Urtica dioica* tall-herb fen, *Epilobium hirsutum* sub-community	37.4
CV_Rest2	M22a	M22a *Juncus subnodulosus-Cirsium palustre* fen-meadow, typical sub-community	50.7
LV_Ctl	OV26	OV26 *Epilobium hirsutum* community	47.4
LV_Bench	M13	M13 *Schoenus nigricans-Juncus subnodulosus* mire	52.6
LV_Rest1(17)	S26	S26 *Phragmites australis-Urtica dioica* tall-herb fen	43.9
LV_Rest1	S4	S4 *Phragmites australis* swamp and reed-beds	45.9
LV_Rest2(17)	S4	S4 *Phragmites australis* swamp and reed-beds	50.0
LV_Rest2	M22a	M22a *Juncus subnodulosus-Cirsium palustre* fen-meadow, typical sub-community	47.0
RP_Ctl1	OV26	OV26 *Epilobium hirsutum* community	39.6
RP_Ctl2	OV26	OV26 *Epilobium hirsutum* community	51.2
RP_Rest1(17)	OV26	OV26 *Epilobium hirsutum* community	42.7
RP_Rest1	M22a	M22a *Juncus subnodulosus-Cirsium palustre* fen-meadow, typical sub-community	41.4
RP_Rest2(17)	OV26	OV26 *Epilobium hirsutum* community	46.3
RP_Rest2	M22a	M22a *Juncus subnodulosus-Cirsium palustre* fen-meadow, typical sub-community	44.3
RP_Rest3	OV26	OV26 *Epilobium hirsutum* community	46.3
RP_Rest4	M22a	M22a *Juncus subnodulosus-Cirsium palustre* fen-meadow, typical sub-community	51.7

3.2.3 Species diversity

Analysis of the plant species richness and abundance using a range of diversity indices is shown in Figures 3.3a and 3.3b. Each index places different weight on species richness, abundance and evenness, but broadly consistent trends are apparent:

- The CV_Ctl1 plot had markedly lower diversity than all other plots;
- The Chilswell and Lye Valley restoration plots show higher diversity restoration (comparing the 2017 data with 2021) and when compared with their control plots. The same is true for CV_Ctl2 after one year of restoration;
- The pattern is less clear at Raleigh Park; RP Rest 1 and RP Rest 2 restoration plots show little change in diversity indices between 2017 and 2021 and under the Fisher Alpha and Pielou's Evenness measures appear to have fallen slightly; and
- The trend for control plots to have lower diversity is not clearly mirrored at Raleigh Park, although the grazed RP_Ctl2 plot had the lowest diversity.

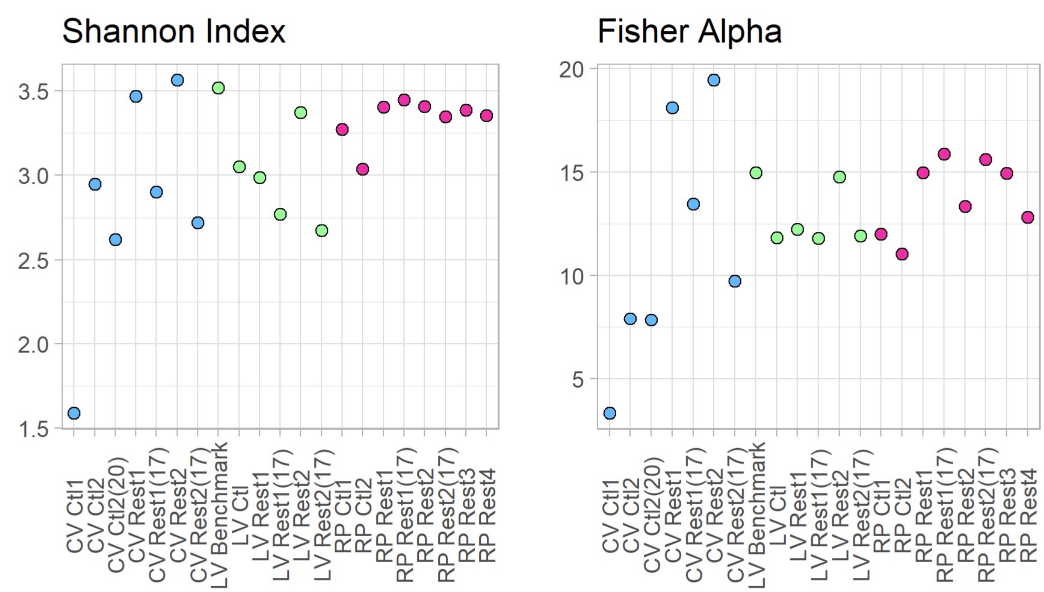


Figure 3.3a: Comparison of Shannon and Fisher Alpha diversity indices for research plot plant communities

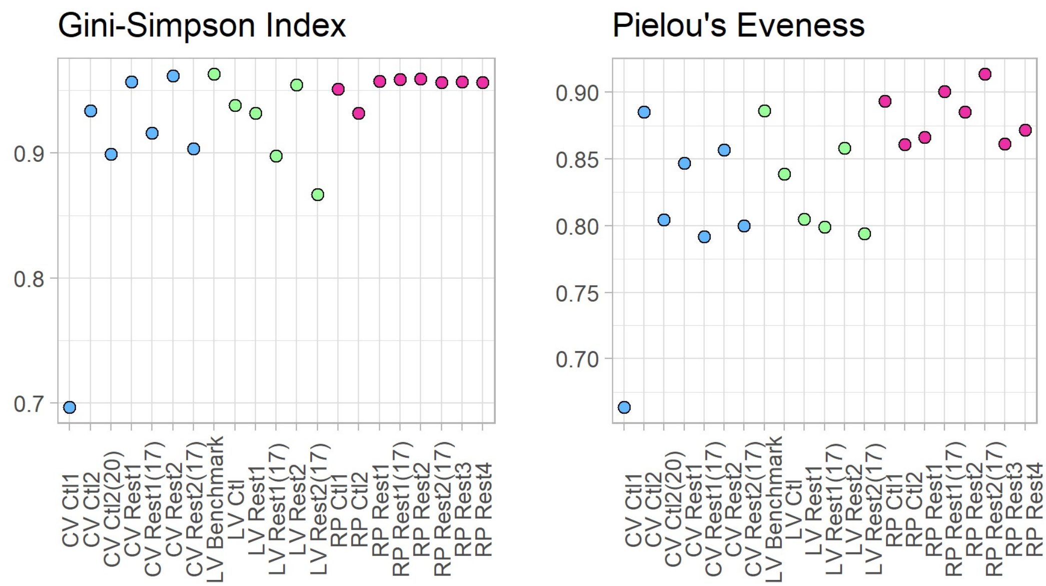


Figure 3.3b: Comparison of Gini-Simpson and Pielou's Evenness diversity indices for research plot plant communities

3.2.4 Dissimilarity and clustering

An assessment of research plot plant community 'Bray-Curtis' dissimilarity is plotted as a heatmap in Figure 3.4. This reveals a number of interesting groupings indicating:

- The 2021 restoration plot plant communities at all 3 study sites have low dissimilarity and share some characteristics, although the Chilswell and Lye Valley communities (group A) are markedly more similar than at Raleigh Park (group B);
- The plant community in the 2021 restoration plots at Chilswell and Lye Valley (group A) have changed and are clearly dissimilar from both their 2017 pre-restoration state (group C) and their control plots (group D)
- By contrast the Raleigh Park restoration plots don't show such clear dissimilarity to their condition in 2017 (group E)
- The LV_Benchmark plot shows a high degree of dissimilarity to the other research plots (Group F), apart from the Chilswell and Lye Valley restoration plots which show weaker dissimilarity.
- The control plots are dissimilar to their respective restoration plots in their study sites

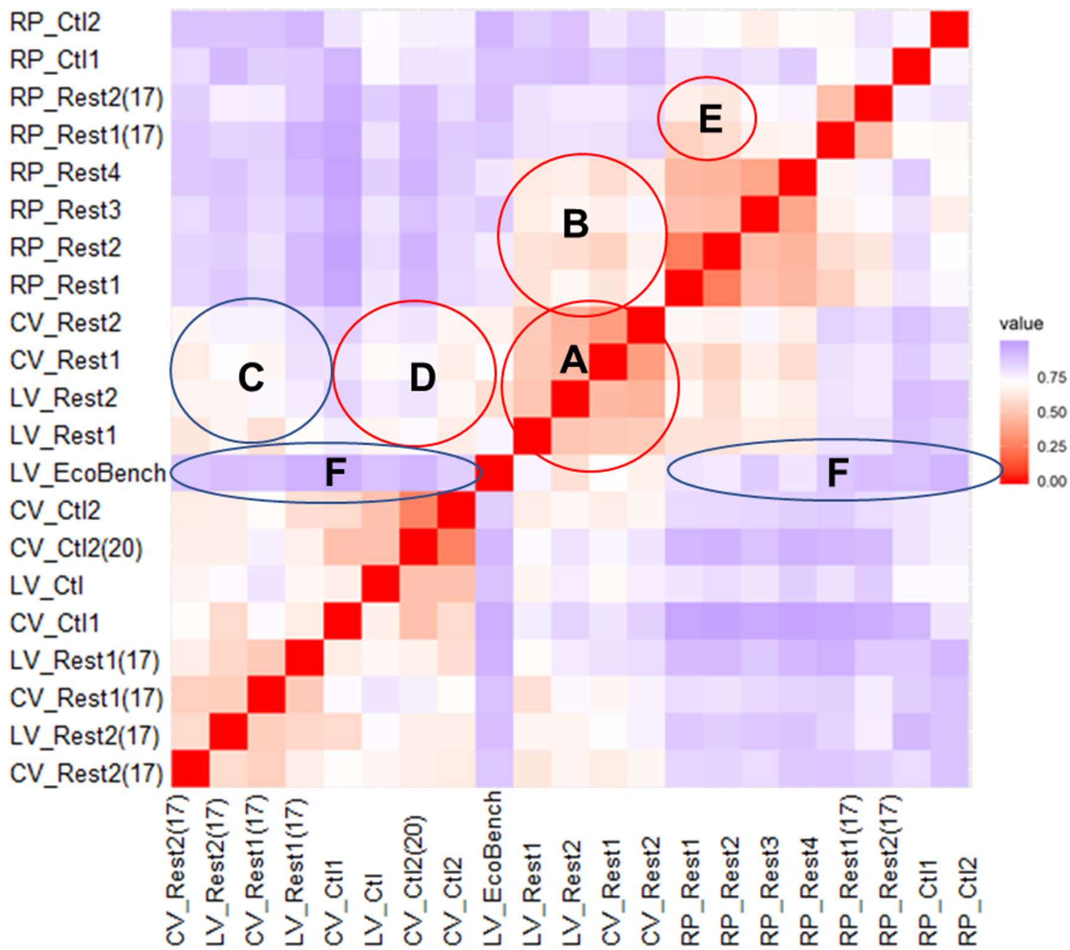


Figure 3.4: Notable groupings in research plots revealed by Bray-Curtis Dissimilarity Heatmap of plant community

HCA indicates the shortest distances between research plots are for the CV_Ctl2 plot, suggesting little change between the 2020 and 2021 surveys and restoration plots 1 and 2 at Raleigh Park and Chilswell Valley respectively (highlighted in red, Figure 3.5). HCA further reinforces the findings of the dissimilarity analysis by:

- clustering the Chilswell and Lye Valley restoration plots (dotted green grouping) together and most notably, separated entirely from their 2017 plant communities;
- placing the Chilswell and Lye Valley control plots into a separate cluster; and
- clustering together all the Raleigh Park plots, albeit with the controls and 2017 data as separate clades

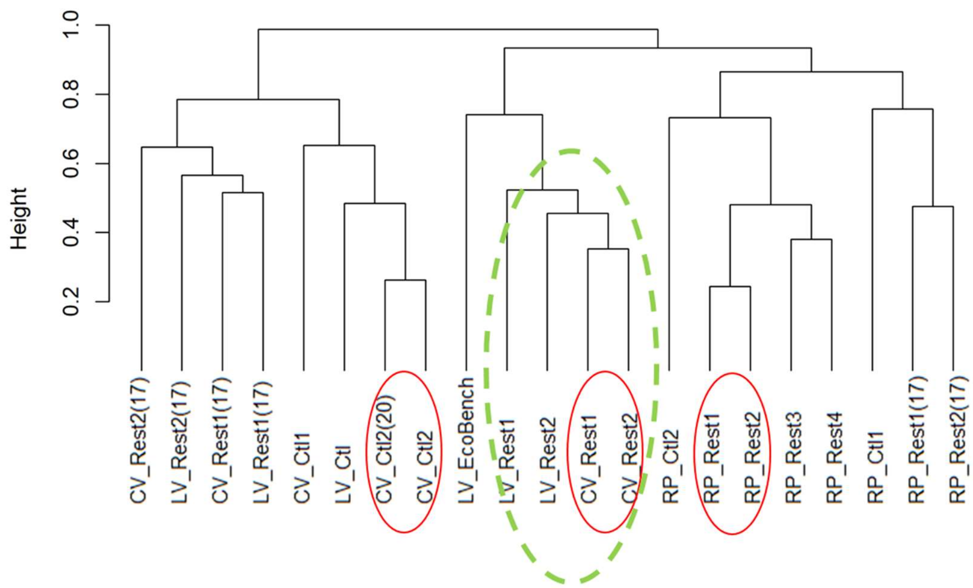


Figure 3.5: Clusters identified by HCA for the plant community dataset

3.2.5 Herb layer height

The research plot herb layer height dataset is displayed as box plots in Figure 3.6. Trends include:

- With the exception of the RP_Ctl2 (grazed) and CV_Ctl2 plots (partly cut in 2021), the control plots are dominated by tall herbs with median height of at least 170cm. However, there is considerable variation in height with data points spread across a far wider distribution than the restoration plots. This indicates that the RP_Ctl2 and CV_Ctl2 plots possessed patches of vegetation taller than that of the restoration plots;
- All restoration plots are characterised by herb communities of median height under 1m with less variation and lower, more uniform herb height than the controls;
- The LV_Benchmark and all grazed Raleigh Park restoration plots possess a low sward markedly shorter than all other plots and with little variation in height.

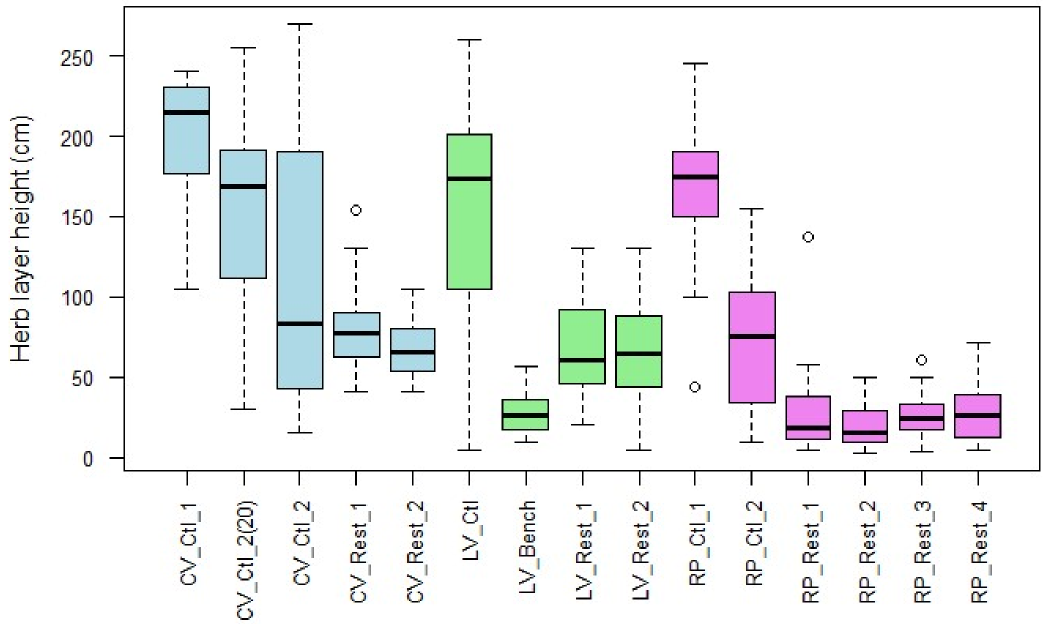


Figure 3.6: Box plot of herb layer height by research plot

3.3 Biotic – Floral nectar resources

3.3.1 Variation in floral resources

The raw data from the floral unit surveys is found in Appendix 4. Table 3.6 summarises the number of species in flower, floral unit count and nectar value for all plots surveyed between late summer 2020 and mid-summer 2021. Key patterns identified include:

- Seasonal variation in nectar resources; all control plots had 2 to 4 times higher nectar resources in August 2020 than in July 2021, notably RP_Ctl1 with the highest of any plot. June nectar resource in restoration plots was lower than in July 2021;
- In July 2021 the Chilswell and Lye Valley restoration plots had 2 to 3 times as many species in flower and far more floral units than their control plots. For example, CV_Rest2 and LV_Rest1 had 72 and 33 more floral units per quadrat than their respective control plots, a difference which was statistically significant (Table 3.7).
- All grazed Raleigh Park plots offered very low floral and nectar resource;
- The abundance of species in flower, volume of floral units and their contribution to total nectar values are not consistent. A small number of plant species appear to be providing the bulk of nectar with *Cirsium Palustre* delivering the majority of nectar in all but one restoration plots in spite of relatively small numbers of floral units.

Table 3.6: Summary of Floral nectar resource (highlighted rows were samples recorded in 2020)

Plot	Date sampled	Species in flower	Floral units	Nectar (mg/day)	Most abundant (% of quadrats flowering)	Most Floral Units (% of total units per plot)	Most Nectar (% of total nectar per plot)
CV_Ctl1	15/08/2020	3	119	65.6	*Calystegia sepium (28%)*	*Humulus lupulus (64%)*	*C. sepium (96%)*
CV_Ctl1	23/07/2021	1	13	23.4	*C. sepium (20%)*	*C. sepium (100%)*	*C. sepium (100%)*
CV_Ctl2	16/08/2020	8	470	1316.5	*C. sepium (38%)*	*H. lupulus (30%)*	*E. cannabinum (78%)*
CV_Ctl2	18/07/2021	5	71	791.9	*C. sepium (15%)*	*C. lutetiana (52%)*	*E. cannabinum (86%)*
CV_Rest1	13/06/2021	6	43	303.8	*Cirsium palustre (15%)*	*C. palustre (51%)*	*C. palustre (93%)*
CV_Rest1	23/07/2021	10	456	797.0	*C. palustre (28%)*	*P. palustris (63%)*	*C. palustre (57%)*
CV_Rest2	02/07/2021	15	1543	1159.8	*Galium uliginosum (33%)*	*G. uliginosum (55%)*	*C. palustre (49%)*
CV_Rest2	23/07/2021	13	2871	1183.2	*Pedicularis palustris (83%)*	*P. palustris (76%)*	*Jacobaea erucifolia (45%)*
LV_Ctl	20/08/2020	7	61	860.1	*Eupatorium cannabinum (20%)*	*Solanum dulcamara (59%)*	*E. cannabinum (93%)*
LV_Ctl	23/07/2021	7	120	194.8	*Circaea lutetiana (10%)*	*C. lutetiana (55%)*	*C. palustre (73%)*
LV_Bench	03/09/2020	6	79	245.2	*Succisa pratensis (18%)*	*P. palustris (44%)*	*C. palustre (53%)*
LV_Bench	23/07/2021	6	318	92.8	*Epipactis palustris (20%)*	*G. uliginosum (75%)*	*Vicia cracca (94%)*
LV_Rest1	09/07/2021	7	429	1063.3	*C. palustre (38%)*	*O. lachenalii (51%)*	*C. palustre (56%)*
LV_Rest1	23/07/2021	11	1461	1047.9	*Oenanthe lachenalii (53%)*	*H. tetrapterum (35%)*	*C. palustre (59%)*
LV_Rest2	19/06/2021	6	37	156.0	*C. palustre (20%)*	*Silene flos-cuculi (43%)*	*C. palustre (100%)*
LV_Rest2	23/07/2021	10	339	593.0	*C. palustre (20%)*	*P. palustris (33%)*	*C. palustre (63%)*
RP_Ctl1	11/08/2020	10	613	3618.0	*E. cannabinum (43%)*	*I. glandulifera (21%)*	*E. cannabinum (52%)*
RP_Ctl1	27/07/2021	8	397	582.5	*Impatiens glandulifera (28%)*	*Helosciadium nodiflorum(83%)*	*I. glandulifera (46%)*
RP_Ctl2	10/08/2020	7	111	472.5	*Epilobium parviflorum (10%)*	*E. parviflorum (41%)*	*E. hirsutum (2%)*
RP_Ctl2	27/07/2021	6	71	213.6	*C. sepium (28%)*	*C. sepium (28%)*	*C. sepium (17%)*
RP_Rest1	26/06/2021	9	92	105.9	*Veronica beccabunga (18%)*	*V. beccabunga (53%)*	*C. palustre (97%)*
RP_Rest1	23/07/2021	5	68	207.5	*Lysimachia nummularia (15%)*	*L. nummularia (28%)*	*C. palustre (99%)*
RP_Rest2	17/07/2021	7	100	100.0	*L. nummularia (13%)*	*L. nummularia (60%)*	*C. palustre (52%)*
RP_Rest3	16/07/2021	6	15	35.7	*Hypericum tetrapterum (8%)*	*H. tetrapterum (33%)*	*C. palustre (72%)*
RP_Rest4	11/07/2021	10	97	248.1	*Lotus pedunculatus (15%)*	*L. pedunculatus (48%)*	*C. palustre (93%)*
RP_Rest4	27/07/2021	8	91	190.2	*L. pedunculatus (23%)*	*H. tetrapterum (23%)*	*C. palustre (95%)*

Table 3.7: Difference in mean floral units per quadrat and Wilcoxon Tests between pairs of control and restoration plots

Research plot pair	Difference between restoration plot vs control (mean floral units per quadrat)	Unpaired two-samples Wilcoxon test	
		W-value	p-value
CV_Ctl1 vs CV_Rest2	+71.5	24.5	1.038e-14
LV_Ctl vs LV_Rest1	+33.5	248	4.909e-08
RP_Ctl1 vs RP_Rest1	-0.1	935	0.1612

3.3.2 Nectar resource

Figure 3.7 plots nectar values and number of floral units for all plots in July 2021. Relationships apparent in the data include:

- The Chilswell and Lye Valley restoration plots offered high floral resources comfortably exceeding nectar offered by their control plots;
- After only 1 year of restoration, CV_Ctl2 offered equivalent nectar to CV_Rest1;
- The LV_Benchmark plot and all the grazed Raleigh Park restoration plots had very low floral units and very low nectar resources. In both cases the LV_Ctl and RP_Ctl1 control plots offered higher nectar resource; and
- High numbers of floral units do not automatically translate into high nectar values. For example, CV_Rest2 produced 40 times as many floral units as CV_Ctl2 but only 50% more nectar resource. A Pearson's correlation determined only a weak positive correlation between floral units and nectar resource (Table 3.8).

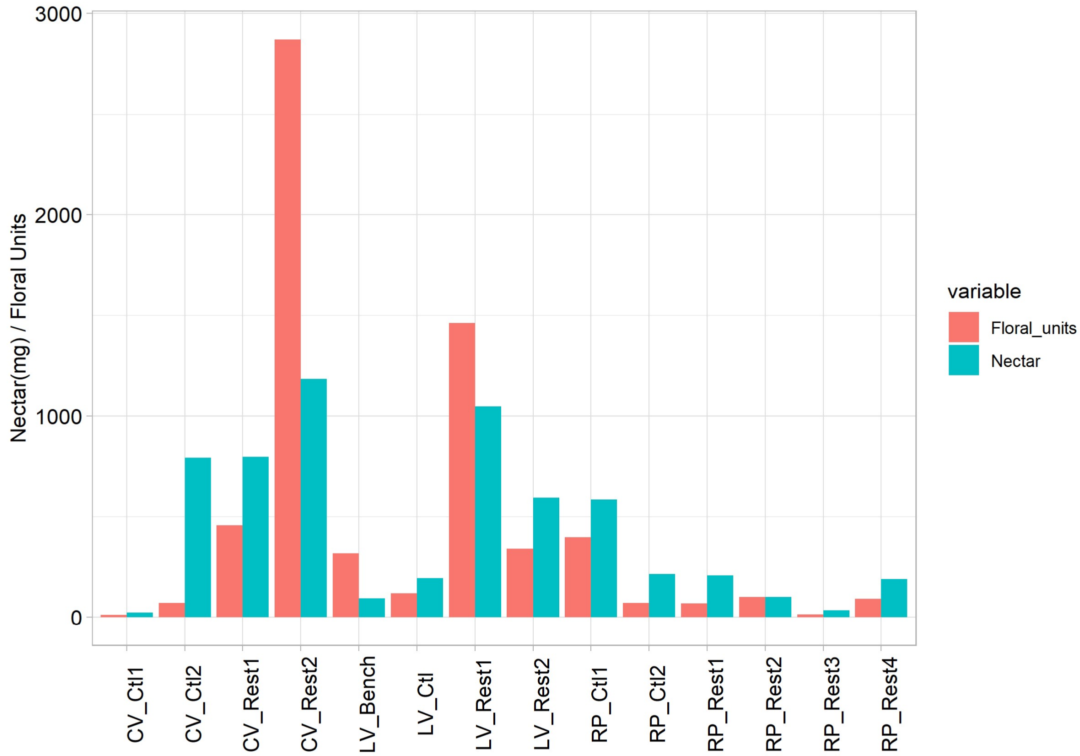


Figure 3.7: Nectar sugar resource by plot, late July 2021

Table 3.8: Pearson's product moment-correlation - floral units and nectar

	Cor value	t	df	p-value
Total Floral Units per plot correlated with total nectar value per plot	0.4272127	2.3148	24	0.0295

Wilcoxon tests conducted on mean quadrat nectar values (non-parametric data) for a pair of control and restoration plots per study site determined that differences in nectar values were statistically significant (Table 3.9). However, the direction of difference was not uniform, with some plots seeing a reduction in nectar value following restoration or in comparison with their control plot (CV_Ctl2 and RP_Rest1), whilst others demonstrated substantially more nectar resource than their controls (CV_Rest2 and LV_Rest1).

Table 3.9: Numerical and statistical differences in nectar values between control and restoration plots

Research Plot	Difference in total nectar value in restoration plot vs control (mg)	Unpaired two-samples Wilcoxon test	
		W-value	p-value
CV_Ctl2 (2020 vs 2021)	-525	1121	0.001375
CV_Ctl1 vs CV_Rest2	+1,160	47	5.924e-14
LV_Ctl vs LV_Rest1	+853	255	4.378e-08
RP_Ctl1 vs RP_Rest1	-375	1046	0.007745

3.3.3 Diversity of floral unit nectar resource

The diversity of floral nectar resource provided by each research plot is compared in Figure 3.8. The main patterns evident are:

- The control plots offer nectar from a small range of flowers, being dominated by a handful of high nectar producing species such as *Eupatorium cannabinum, Cirsium palustre, Impatiens glandulifera* and *Calystegia sepium;*
- In contrast, the ungrazed restoration plots provided a high diversity of nectar sources from 10-13 species in flower with *Lythrum salicaria, Jacobaea erucifolia, Pedicularis palustris, Vicia cracca, Scrophularia auriculata* and *Oenanthe lachenalii* being important as well as the ubiquitous *Cirsium palustre;* and
- The grazed restoration plots at Raleigh Park provided low nectar volume and diversity from just 5-8 species.

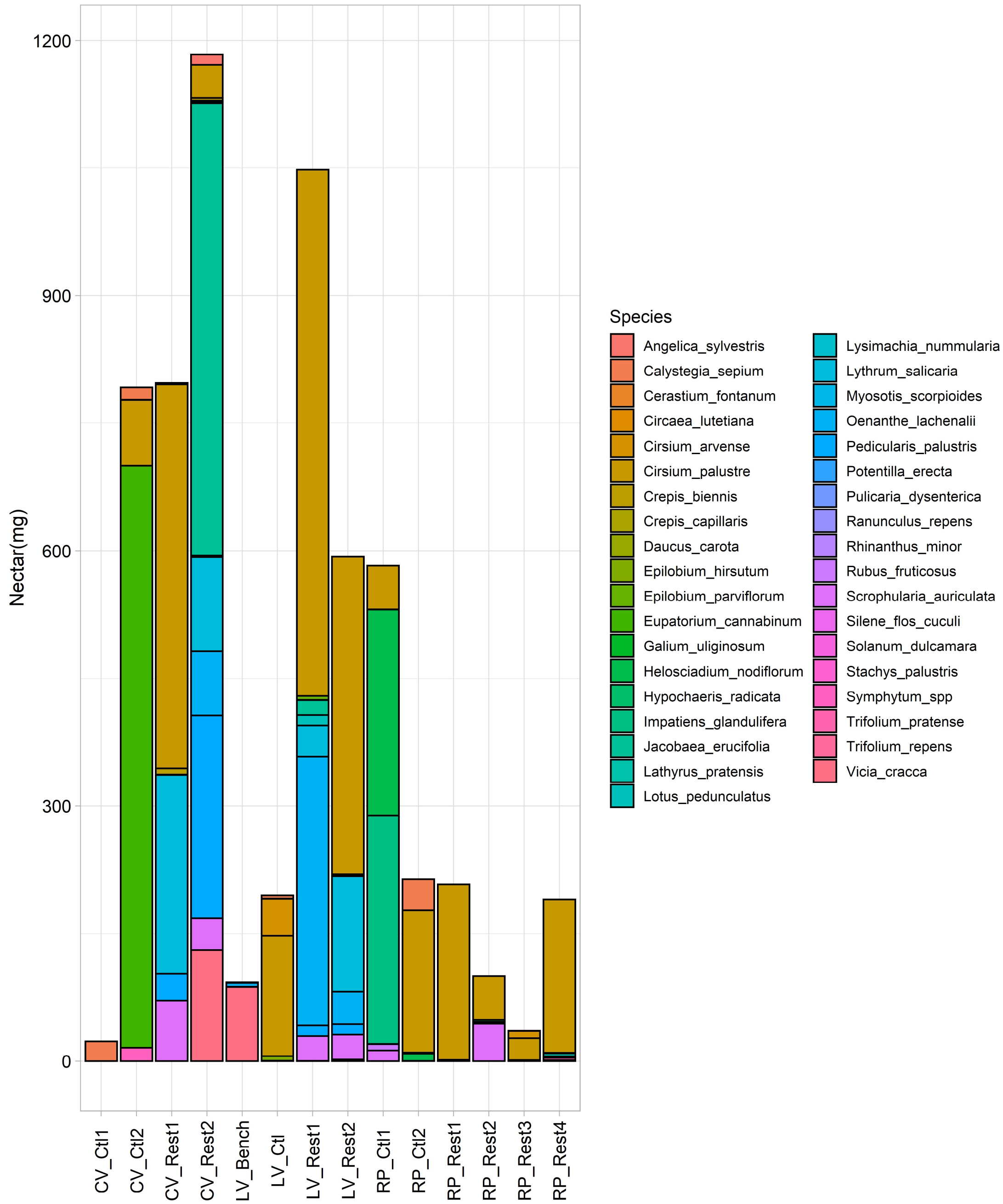


Figure 3.8: Nectar provision by species, late July 2021

3.4 Biotic – Flying Insect Pollinators

3.4.1 Research plot utilisation by flying insects

The FIT count raw data is found in Appendix 6. Figure 3.9 shows flying insects recorded utilising floral units within each research plot during the summer, with apparent trends including:

- Restoration plots had a higher median value of flying insects than the control plots. A notable exception was RP_Rest2 which supported very low flying insect pollinators. T-Tests and a Wilcoxon Test proved this difference to be statistically significant for CV_Ctl1 vs CV_Rest2 and LV_Ctl vs LV_Rest1 respectively, but not for RP_Ctl1 vs RP_Rest1 (Table 3.10);
- The high variation in numbers of flying insects visiting the control plots within the samples was notable;
- The restoration plots tended to have higher outliers with 1 or 2 samples recording very high numbers of flying insects diverging considerably from the median, in particular CV_Rest1 and Lye Valley restoration plots. This was also observed at the RP_Ctl2 control plot.

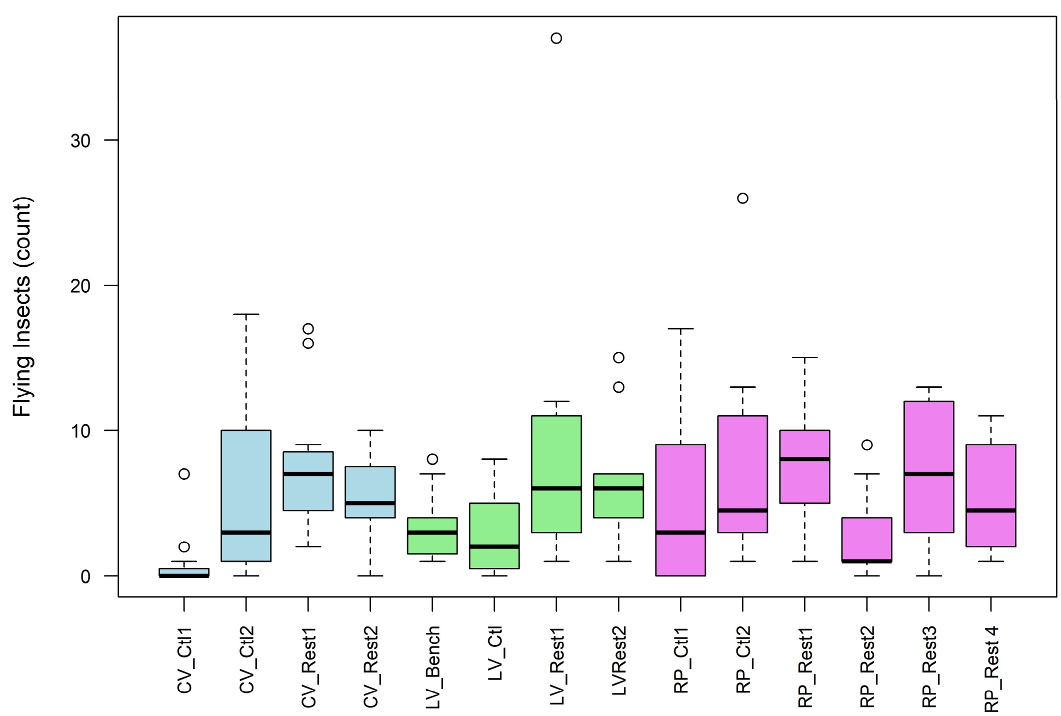


Figure 3.9: Boxplot showing median flying insects recorded 31 May – 18 August 2021

Table 3.10: Numerical and statistical differences in total flying insect numbers between control and restoration plots

Research Plot	Difference in total flying insect visits visiting restoration plot vs control plot	Unpaired two-samples T-test		
		t-value	df-value	p-value
CV_Ctl1 vs CV_Rest2	+48	-3.7736	17.367	0.001467
RP_Ctl1 vs RP_Rest1	+24	-1.0313	16.358	0.3174
		Unpaired two-samples Wilcoxon test		
		W-value	p-value	
LV_Ctl vs LV_Rest1	+66	27	0.02931	

3.4.2 Timing of flying insect visits

Figure 3.10 plots numbers of flying insects visiting plots by date. The Chilswell and Lye Valley restoration plots experienced more flying insect visits over a longer duration than their respective control plots, peaking mid to late July and tailing off into August. This relationship is not apparent in Raleigh Park, with the restoration plots attracting more flying insects throughout June and early July, whilst the control plots peaked later in July.

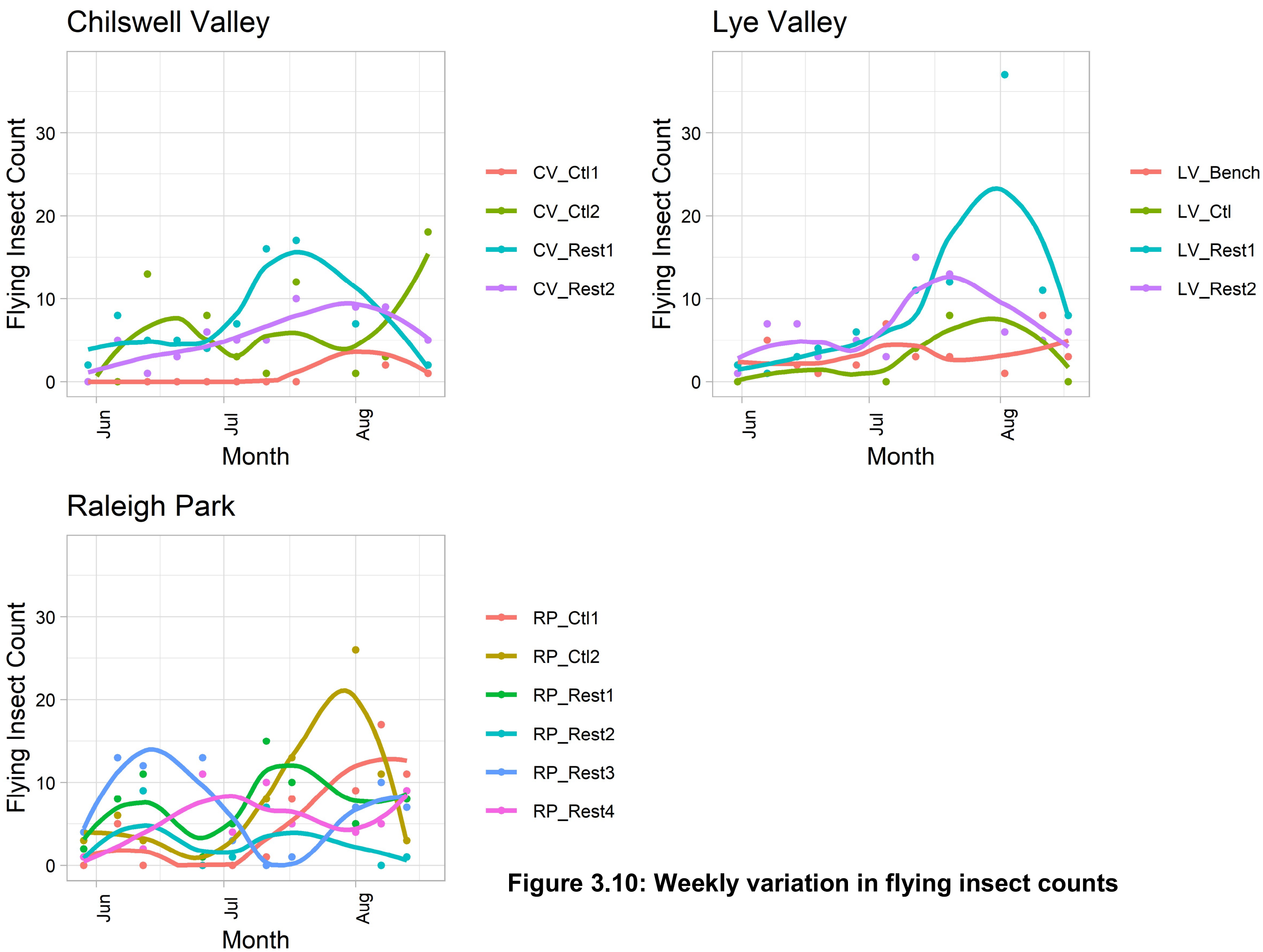


Figure 3.10: Weekly variation in flying insect counts

3.4.3 Utilisation of research plots by insect groups

Bumble bees and hoverflies were the insect groups most frequently using the research plot floral resources (33% and 23% of all insects recorded respectively). Figure 3.11 provides a comparison of insect groups recorded in total for each research plot. Although there is significant variation, some patterns are evident:

- A greater diversity of insect groups was recorded in the Chilswell and Lye Valley restoration plots than their respective controls. This was not repeated in the grazed Raleigh Park restoration plots;
- Honey bees and hoverflies were recorded significantly more frequently in the restoration plots than the control plots. There was no discernible preference for bumble bees, for example control plots CV_Ctl2 and RP_Ctl1 appeared equally attractive due to the presence of *Symphytum* and *Impatiens gladulifera*; and
- Very few butterflies, moths and wasps were recorded.

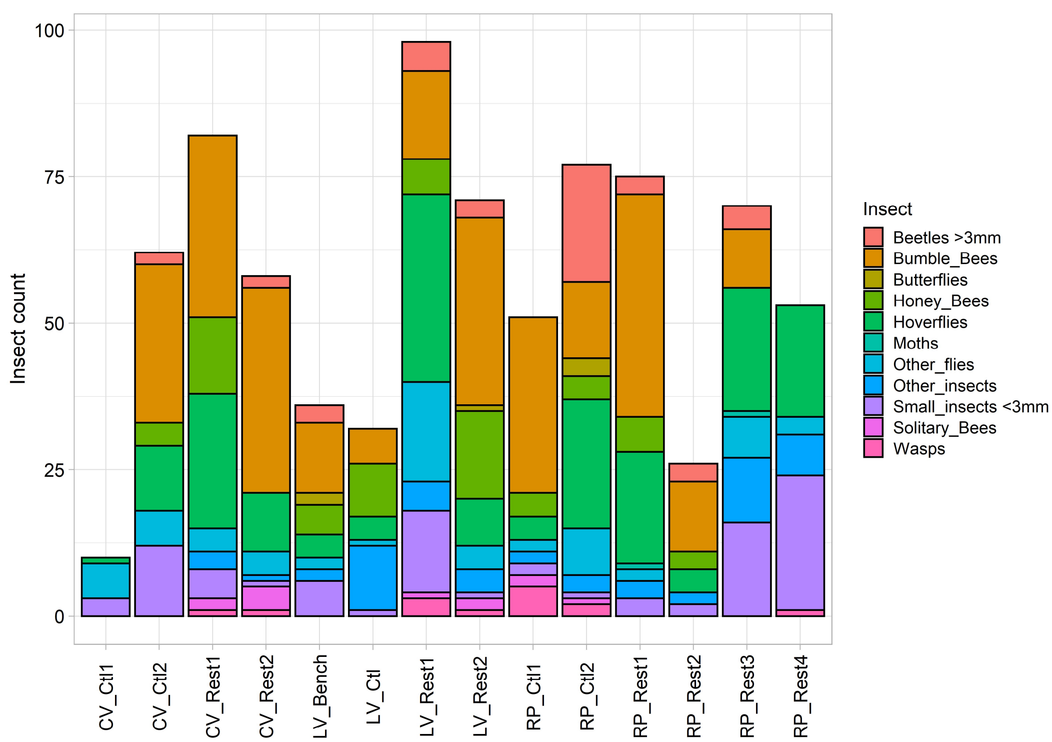


Figure 3.11: Insect groups recorded by research plot

3.4.4 Utilisation of flower resource by insect group

To account for uneven sampling in target flowers across research plots, mean FIT counts were calculated by flowering species for bumble bees, honey bees, hoverflies, other flies and small insects, these accounting for 83% of all visiting insect groups. Summary results are presented per study site in Table 3.11 (top 2 flowers by insect group highlighted) and Figure 3.12 plots these aggregated across all sites.

There is considerable variation by study site, but this illustrates the value that different wetland flower species have for different pollinator groups, with some flowers only being visited by specific insect groups. *Oenanthe lachenalii, Filipendula ulmaria, Pulicaria dysenterica* and *Eupatorium cannibinum* are clearly important for hoverflies, flies and small insects. *Pedicularis palustris*, *Iris pseudacorus, Cirsium palustre, Silene flos-cuculi* and *Vicia cracca* for bumble bees and *Lythrum salicaria* and *Epilobium hirsutum* for honey bees

Two of the top 3 most visited flowers were plants not typical associated with healthy UK wetland ecosystems and potentially undesirable from a wider ecological perspective; *Cirsium arvense,* an invasive perennial ruderal, and *Impatiens glandulifera*, a non-native highly invasive annual. This pattern was only observed at Raleigh Park in the control plots.

Symphytum officinale agg. is also more typical damp grassland than fens but was the third most visited species by bumble bees despite being only found at Chilswell Valley.

Table 3.11: mean FIT count per flowering species by top 5 insect groups

Species in flower	Number of samples	Mean FIT count	Mean FIT count by top 5 insect groups				
			Bumble Bees	Honey Bees	Hoverflies	Other flies	Insects <3mm
Chilswell Valley							
Calystegium sepium	3	3.3			0.3	**2.0**	1.0
Cirsium palustre	5	5.0	3.2	0.6	0.6	0.2	0.2
Eupatorium cannibinum	3	**7.3**		**0.7**	**3.3**	**2.0**	**1.3**
Lythrum salicaria	3	**13.3**	0.7	**3.3**	**6.7**	1.3	0.3
Pedicularis palustris	8	6.3	**4.8**		0.8	0.4	
Symphytum officinale agg	9	6.1	**3.9**	0.2	0.2		**1.2**
Vicia cracca	2	5.0	1.0		1.5		0.5
Lye Valley							
Ajuga reptans	1	1.0					
Cardamine flexuosa	2	1.0					
Cirsium palustre	11	5.6	2.4	1.0	**1.6**	0.2	
Epilobium hirsutum	2	7.0		**3.0**			0.5
Eupatorium cannibinum	2	4.0		2.5	1.0		
Lythrum salicaria	3	**8.3**	1.0	**4.0**	0.3	**1.0**	0.3
Oenanthe lachenalii	4	**17.0**	0.3	0.3	**6.3**	**4.0**	**3.5**
Pedicularis palustris	3	3.0	2.7		0.3		
Ranunculus acris	3	2.7			0.3	0.7	**1.3**
Silene flos-cuculi	4	4.5	**3.3**			0.3	
Solanum dulcamara	4	1.5	1.3				
Valeriana dioica	1	1.0					1.0
Vicia cracca	3	5.0	**3.0**				0.3
Raleigh Park							
Ajuga reptans	1	1.0			1.0		
Cardamine pratensis	1	2.0			1.0		
Cirsium arvense	2	**19.5**	1.5	0.5	**6.0**	**2.5**	
Cirsium palustre	20	4.2	2.3	**0.6**	0.9	0.1	0.1
Eupatorium cannibinum	2	7.0			3.0	**1.5**	0.5
Filipendula ulmaria	4	7.5			2.3		**4.0**
Impatiens glandulifera	4	**11.3**	**7.5**	**1.0**	0.8		
Iris pseudacorus	7	10.0	**2.9**		1.4	0.7	**2.7**
Lotus penduncculatus	2	0.5			0.5		
Pulicaria dysenterica	6	7.0			**3.7**	0.8	1.2
Ranunculus repens	2	1.5			1.0		0.5
Silene flos-cuculi	4	3.5	1.3	0.3	0.5		
Veronica beccabunga	3	2.3			0.7	0.7	0.7

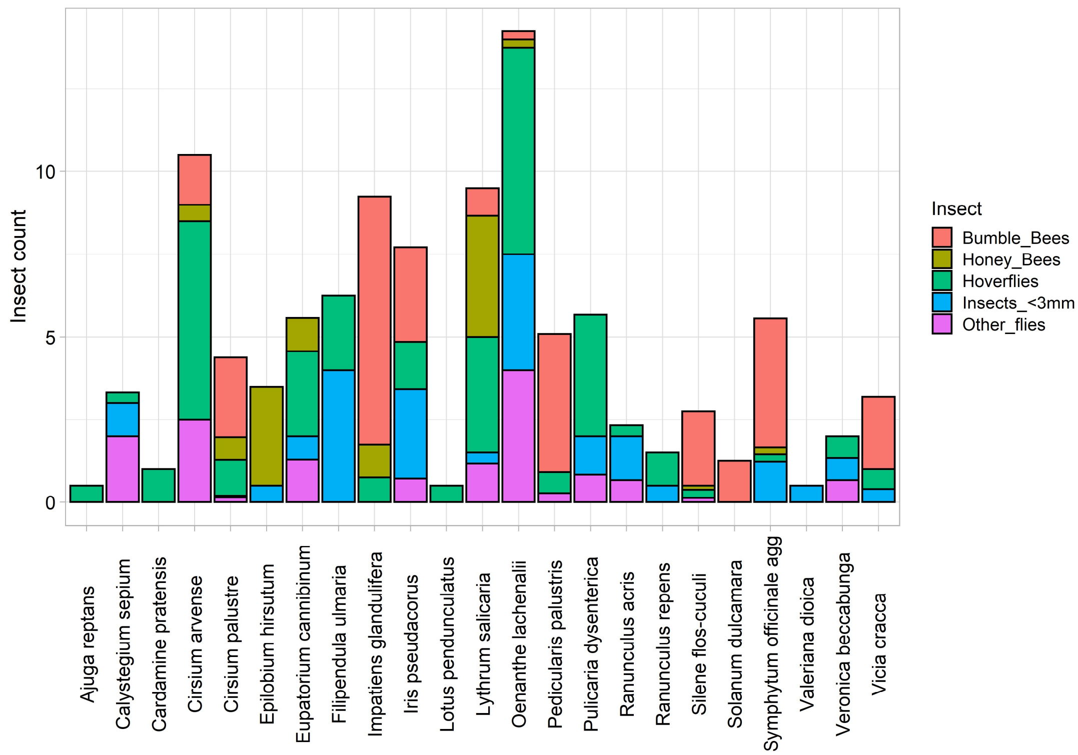


Figure 3.12: Mean FIT count of top 5 insect groups by flower resource

Pearson's product-moment correlations indicate there appears to be no relationship between the nectar value estimated in each FIT survey quadrat and the number of flying insect visits (Table 3.12). An exception are honey bees, whose numbers appear to have a weakly positive correlation with nectar values.

Table 3.12: Pearson correlation between FIT quadrat nectar and insect groups

Test for correlation with Nectar estimate per FIT quadrat and…	Corr value	t	df	p-value
All insects	0.185232	2.2776	146	0.0242
Bumble bees	0.06740078	0.81626	146	0.4157
Honey bees	0.5281102	7.5146	146	5.28e-12
Hoverflies	0.04306574	0.52085	146	0.6033
Other flies	-0.01744921	-0.21087	146	0.8333
Insects <3mm	-0.06611894	-0.80067	146	0.4246

3.5 Abiotic – Water Table Depth

Mean annual WTD is summarised in Table 3.13 and the full dataset of weekly manual dipwell measurements plotted by site in Figure 3.13. Key trends are evident:

- The Chilswell Valley restoration plots display substantial summer WTD drawdown, especially CV_Rest1. The control plot CV_Ctl2 showed less variation and retained water at the surface for much of the year;
- Lye Valley plots have little variation and a consistently high year round water table, remaining within 10cm of the ground surface even in the summer. Visual observations concur that the LV_Ctl and LV_Rest1 remain saturated under foot most of the year; and
- The main fen in Raleigh Park (plots RP_Ctl1, RP_Rest1 and RP_Rest2) and RP_Ctl2 have consistently high WTD, remaining saturated at the surface all year. RP_Rest3 shows extreme seasonal variation with WTD plunging during summer and autumn then returning to the surface in winter after the artificial drain was blocked.

Table 3.13: Mean annual and seasonal research plot WTD

Research plot	NVC	Mean WTD (cm)				
		Annual	Winter (Dec-Feb)	Spring (Mar-May)	Summer (Jun-Aug)	Autumn (Sep-Nov)
CV_ctl1	S4a	-7	-4	-6	-12	-6
CV_ctl2	S26	-5	-2	0	-8	-7
CV_rest1	OV26	-17	-9	-16	-24	-19
CV_rest2	M22a	-10	-4	-8	-15	-9
LV_ctl	OV26	-1	-2	-1	-2	-1
LV_rest1	S4	-3	-5	-4	-1	-2
LV_rest2	M22a	-5	-4	-5	-8	-5
LV_bench	M13	-5	-4	-5	-8	-5
RP_ctl1	OV26	-2	-1	-2	-4	-2
RP_ctl2	OV26	-2	-1	0	-5	-1
RP_rest1	M22a	0	-1	0	0	0
RP_rest2	M22a	-3	0	-2	-7	-2
RP_rest3	OV26	-18	-6	0	-29	-33
RP_rest4	M22a	-7	-4	-4	-12	-7

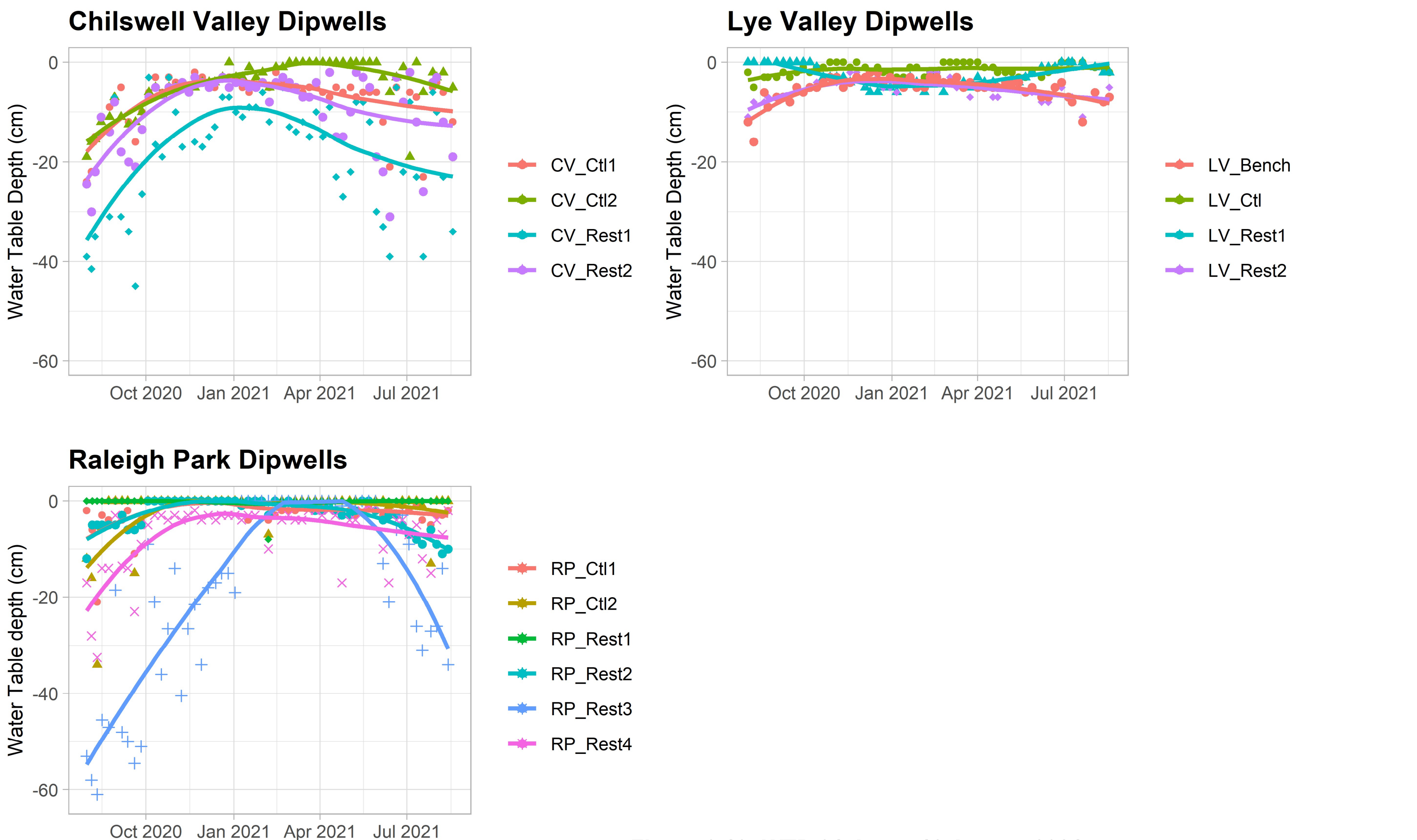


Figure 3.13: WTD 31 July – 18 August 2021

Figure 3.14 overlays the hourly WTD measurements recorded by the data loggers in 4 research plots with their equivalent manual readings. The close correlation between the two confirms the reliability of the manual dipwell readings and provides reassurance of data accuracy. Appendix 7 contains the raw data for the manual dipwell readings.

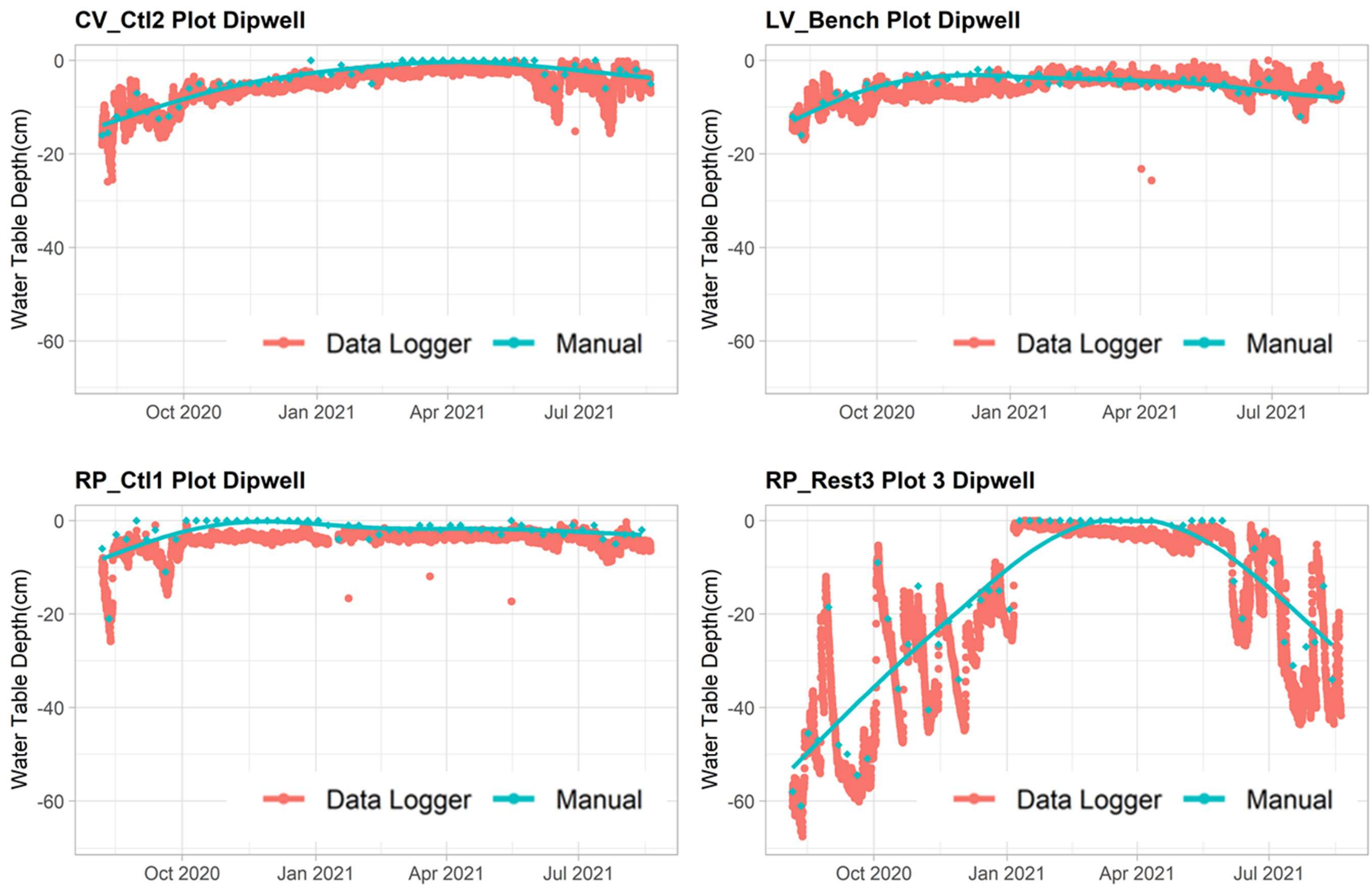


Figure 3.14: Overlay of WTD recorded by Data logger and manually

The data loggers also recorded dipwell water temperature. Figure 3.15 reveals a large seasonal difference between maximum and minimum water temperature of 6-8°C in Raleigh Park, more than twice that observed at Chilswell Valley. The Lye Valley data logger recorded variation midway between the Raleigh Park and Chilswell Valley plots, despite being installed 1m closer to the surface.

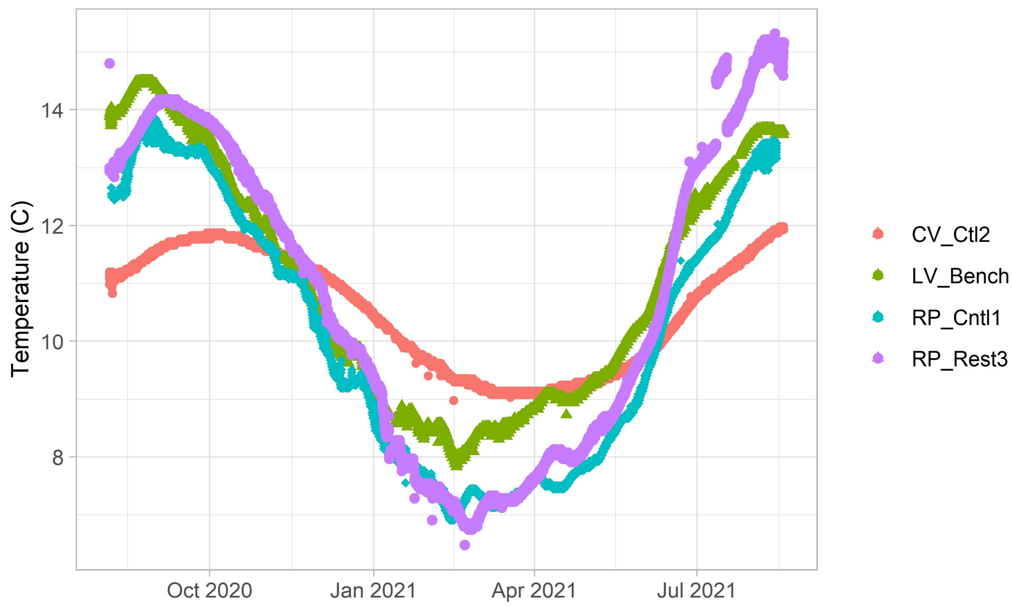


Figure 3.15: Dipwell water temperature for plots equipped with data loggers

3.6 Abiotic – Conductivity, pH and Water Chemistry

3.6.1 Conductivity

The conductivity meter malfunctioned and produced unusually high values in the field between 11 April and 16 May, so was replaced with a new model 18 May 2021. The scatter plot in Figure 3.16 indicates values in all plots during this period were consistently 40-50% higher. These erroneous readings from the malfunctioning unit were excluded from further analysis, with the low SD of the mean for the remaining measurements providing reassurance of their accuracy (Table 3.14).

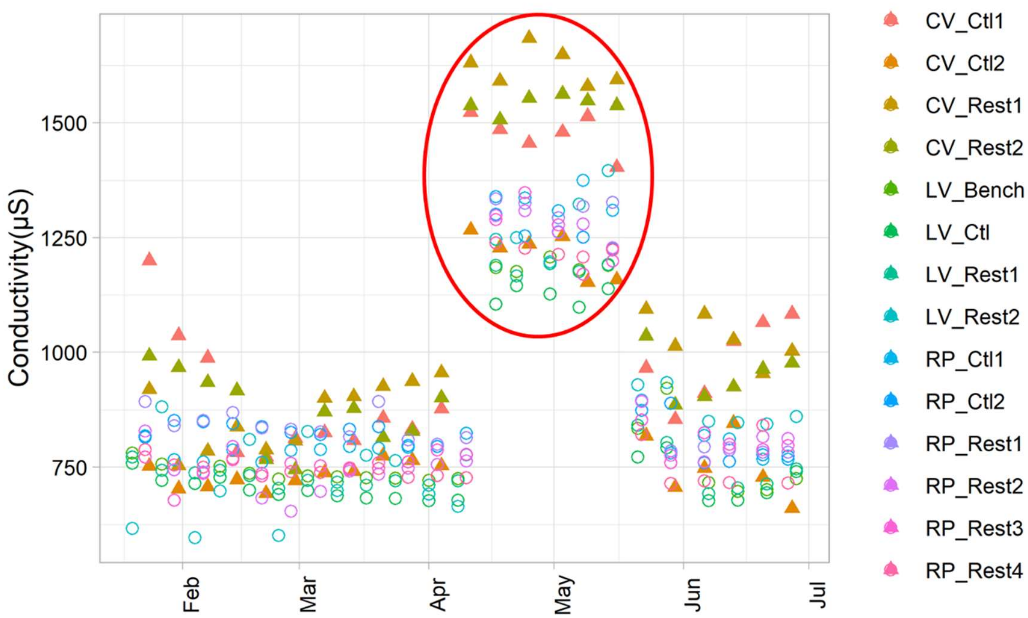


Figure 3.16: Conductivity (µS) readings; erroneous data highlighted and discarded due to meter malfunction

Conductivity does not appear to experience significant seasonal variation. Values within study sites remained broadly similar, with conductivity the lowest in Lye Valley plots and the greatest in Chilswell Valley plots (apart from CV_Ctl2).

Raw data is provided in Appendix 8.

Table 3.14: Mean conductivity, pH and water chemistry values

Research Plot	Measured manually in-situ						Laboratory tests by Chemtech Environmental						
	pH		Conductivity (µS)		Mean (n = 2)		Single sample						
	Mean (n = 14)	Sd	Mean (n = 17)	Sd	Nitrates (mg/l)	Phosphates (mg/l)	pH	Nitrates (mg/l)	Phosphates (mg/l)	Calcium (mg/l)	Potassium (mg/l)	Iron (µg/l)	Sodium (mg/l)
CV_Ctl1	6.9	0.14	923	125	0.2	0.02	6.9	0.2	0.3	242	0.4	4.1	18
CV_Ctl2	7.5	0.17	740	45	0.2	0.02	7.7	1.6	0.3	123	1.2	10	11
CV_Rest1	7.0	0.15	923	103	0.2	0.02	7.2	0.2	0.3	170	0.4	6.5	16
CV_Rest2	7.0	0.28	900	79	0.2	0.04	6.9	0.2	0.3	206	0.8	164	16
LV_Bench	7.0	0.16	747	54	0.2	0.02	7.2	0.2	0.3	137	1.5	8.1	26
LV_Ctl	7.0	0.17	709	35	3.0	0.02	7.1	3.4	0.3	154	0.1	20	21
LV_Rest1	6.8	0.21	735	38	0.2	0.02	7.1	0.2	0.3	160	0.1	7.9	39
LV_Rest2	6.9	0.23	772	110	0.2	0.02	6.9	0.2	0.3	157	1.3	7.5	16
RP_Ctl1	7.3	0.21	795	32	0.2	0.02	7.5	0.5	0.3	156	11	4.9	16
RP_Ctl2	7.0	0.27	818	38	0.5	0.04	7.4	0.5	0.3	189	0.9	3.9	17
RP_Rest1	7.1	0.19	828	39	0.2	0.02	7.1	0.2	0.3	186	6.6	5.6	16
RP_Rest2	7.1	0.24	763	56	0.2	0.02	7.0	0.2	0.3	192	6.3	6	15
RP_Rest3	7.4	0.23	770	38	0.5	0.26	7.8	4.5	0.3	158	7.6	3.9	14
RP_Rest4	7.0	0.24	749	36	0.2	0.02	7.4	0.3	0.3	157	4.1	3.2	12

3.6.2 pH

From 21 March the pH meter malfunctioned, with the temperature continually resetting to 100C. In the field the values recorded did not seem hugely different, so it was not replaced with a new meter until 18 May. However, when plotted graphically post data collection, the malfunction coincides with a period of notably reduced pH values, so these values have been deemed unreliable and removed (figure 3.17). The adjusted mean pH has low SD and compares well with the pH laboratory measurements, although the latter recorded 5-6% higher pH values in RP_Ctl2, RP_Rest3 and RP_Rest4 (Table 3.11). All dipwell plots had alkaline waters with mean values between 6.8 and 7.5, with Raleigh Park characterised by the highest pH measurements, fitting the definition of an Alkaline fen by Diack et al., (2013). Raw data is provided in Appendix 9.

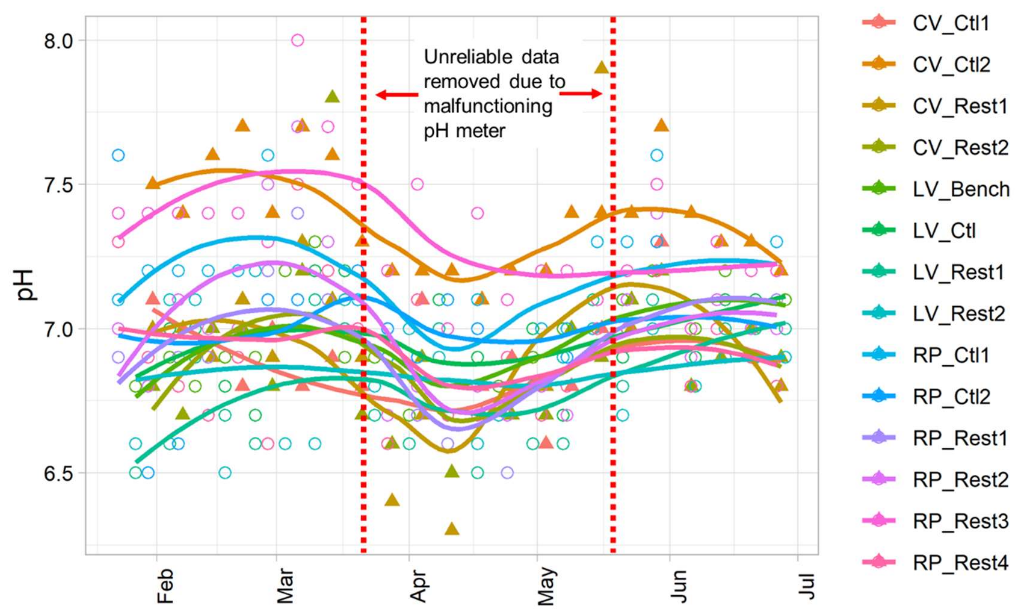


Figure 3.17: pH readings; unreliable data between dotted red lines discarded due to meter malfunction

3.6.3 Water chemistry

The laboratory tests for dipwell samples collected late winter 2021 recorded elevated levels of nitrate in CV_Ctl2, LV_Ctl and RP_Rest3; these were matched to some extent by both FHT test samples with more pronounced nitrate in the summer in LV_Ctl (Figure 3.18).

The other plots in Chilswell and Lye Valley were clean, whereas Raleigh Park had slightly elevated levels in a further 3 plots. The Laboratory and FHT tests for Phosphates revealed no detectable levels in winter or spring. RP_Rest3 was the only plot with a slightly elevated level in the summer.

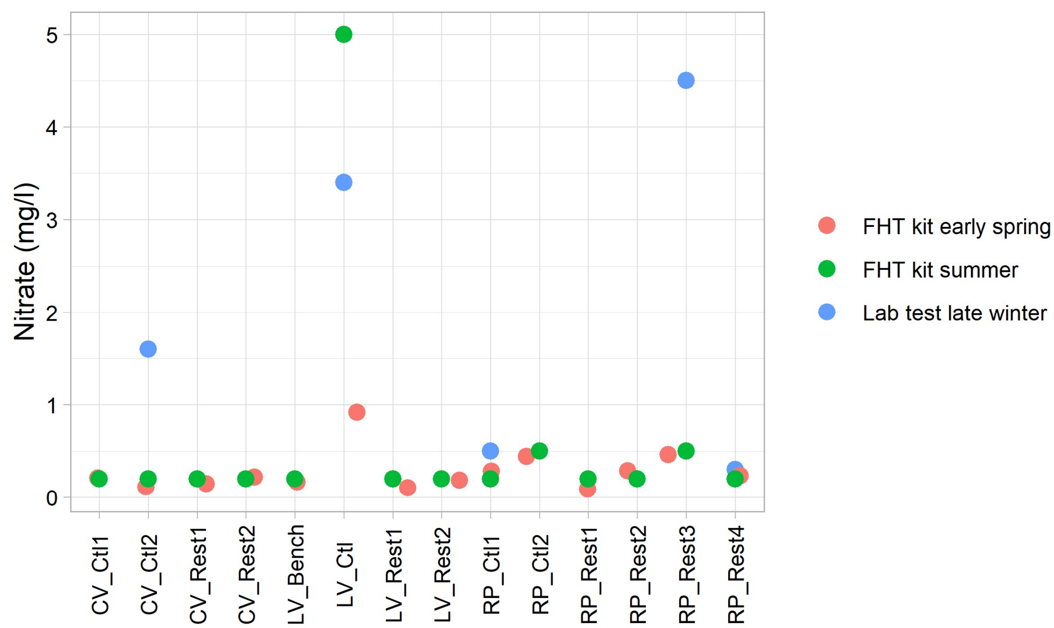


Figure 3.18: Dipwell nitrate levels winter, spring and summer 2021

Table 3.14 also lists the laboratory tests for other mineral ions; all plots have high levels of calcium, Raleigh Park (except for RP_Ctl2) has significantly higher levels of Potassium, Lye Valley have slightly higher levels of sodium and a notable iron rich spring was revealed in CV_Rest2.

3.7 Abiotic - Peat

Appendix 10a summarises the peat core depth by plot and Appendices 10b–10d contains the raw data by site. In summary, the Chilswell and Lye Valley research plots have thick peat deposits, frequently exceeding 1m in CV_Ctl2 and CV_Rest1 (Figure 3.19). Raleigh Park research plots tended to have shallow peat of around half the depth with less variation in thickness. RP_Rest3 had very little peat. CV_Rest2 had a 25cm layer of tufa overlying the peat; Marl deposits were also observed in all Chilswell plots.

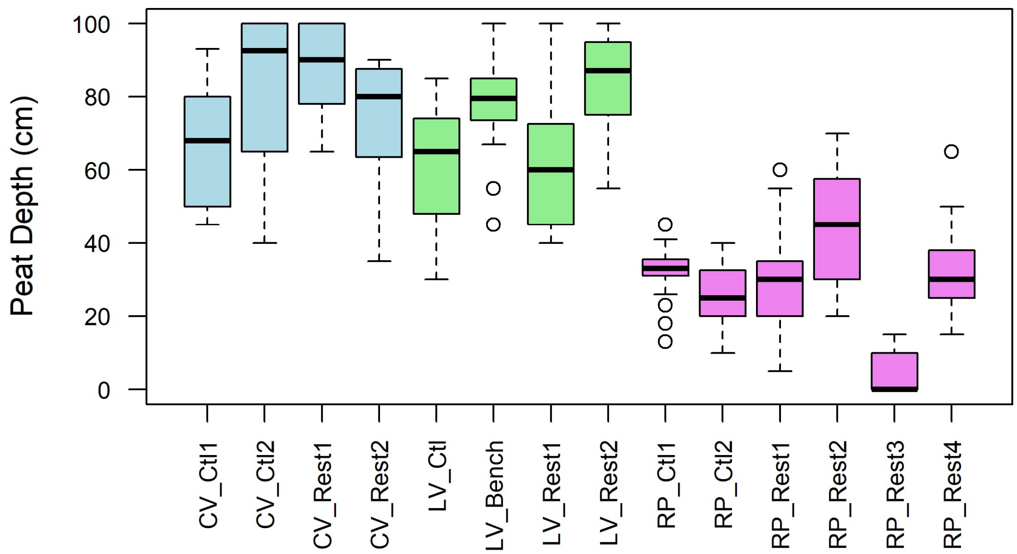


Figure 3.19: Box plot of plot peat depth by research plot

3.8 Relationships between Variables

3.8.1 Biotic

Biotic variables are generally positively correlated (Figure 3.20), strongly so for species richness, diversity indices and floral units, species flowering and nectar. Notable exceptions include

- as herb layer height increases, species richness and insect counts decline; the correlation is weaker with FIT counts but there is no relationship with floral units or nectar values; and
- A similar, but weaker, negative correlation for shading with species richness and hoverflies.

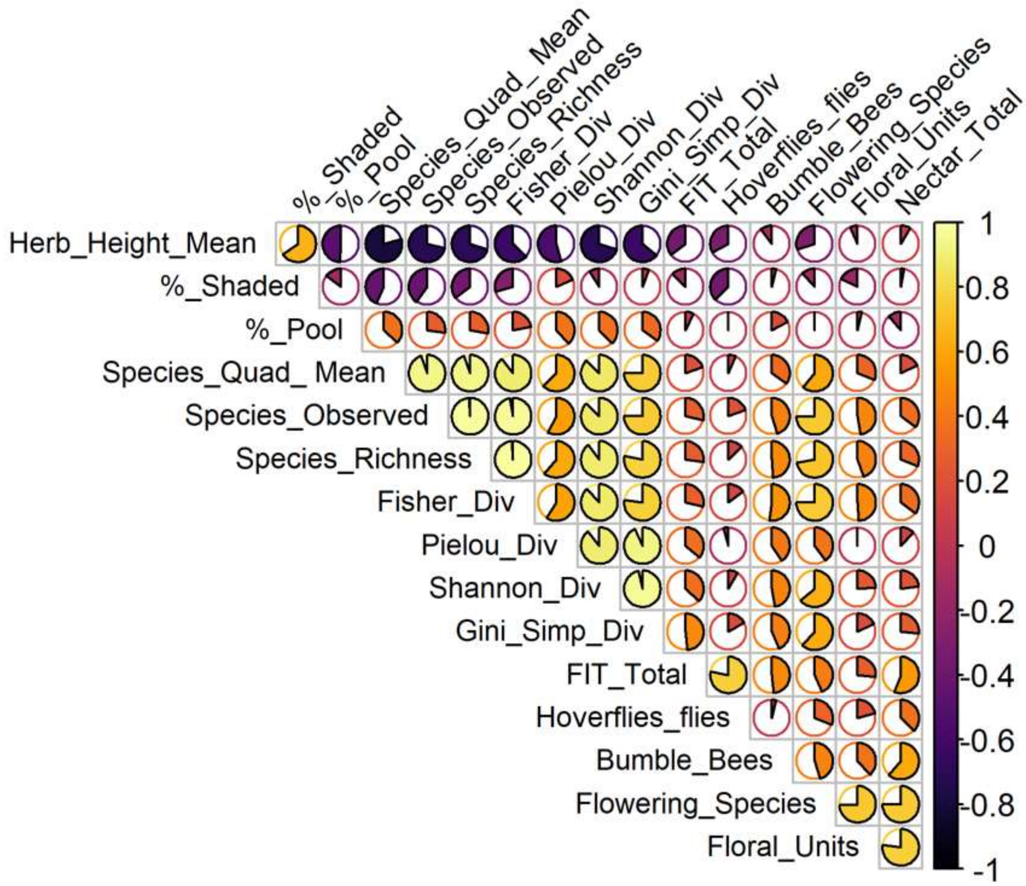


Figure 3.20: Correlation plot of biotic variables

3.8.2 Abiotic

Figure 3.21 suggests correlations of potential interest between the abiotic variables include:

- Nitrates, phosphates, pH and potassium are positively correlated with each other and high WTD, whilst being negatively correlated with low WTD, the other minerals and conductivity;
- Iron and in particular calcium are positively correlated with conductivity; and
- All peat depth variables show moderate negative correlations with high water table, nitrates, phosphates, pH and potassium; these are likely to be spurious as they apply for both the minimum and maximum peat depth which have substantial variation.

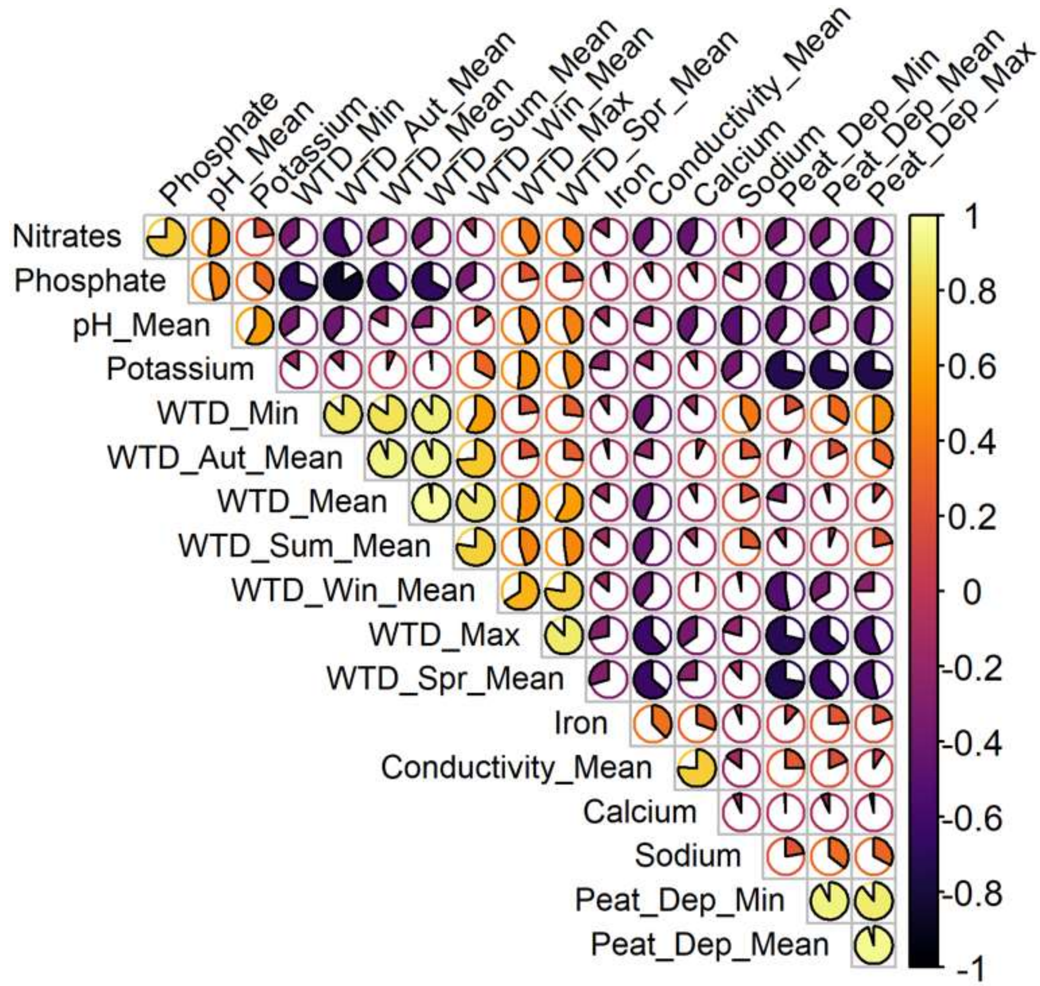


Figure 3.21: Correlation plot of abiotic variables

3.8.3 Plant Community relationships with restoration activities

Correlating the plant community including the 2017 data with ecological restoration activities (Figure 3.22), we can observe the following relationships:

- Intensity and then duration of ecological restoration have a moderate positive correlation with species richness and diversity, although there is no relationship with plant community evenness;
- The effect is weaker for applications of hand collected seed, whilst brown hay had no meaningful relationship with species richness or diversity; and
- Grazing has a weak positive relationship with some diversity measures.

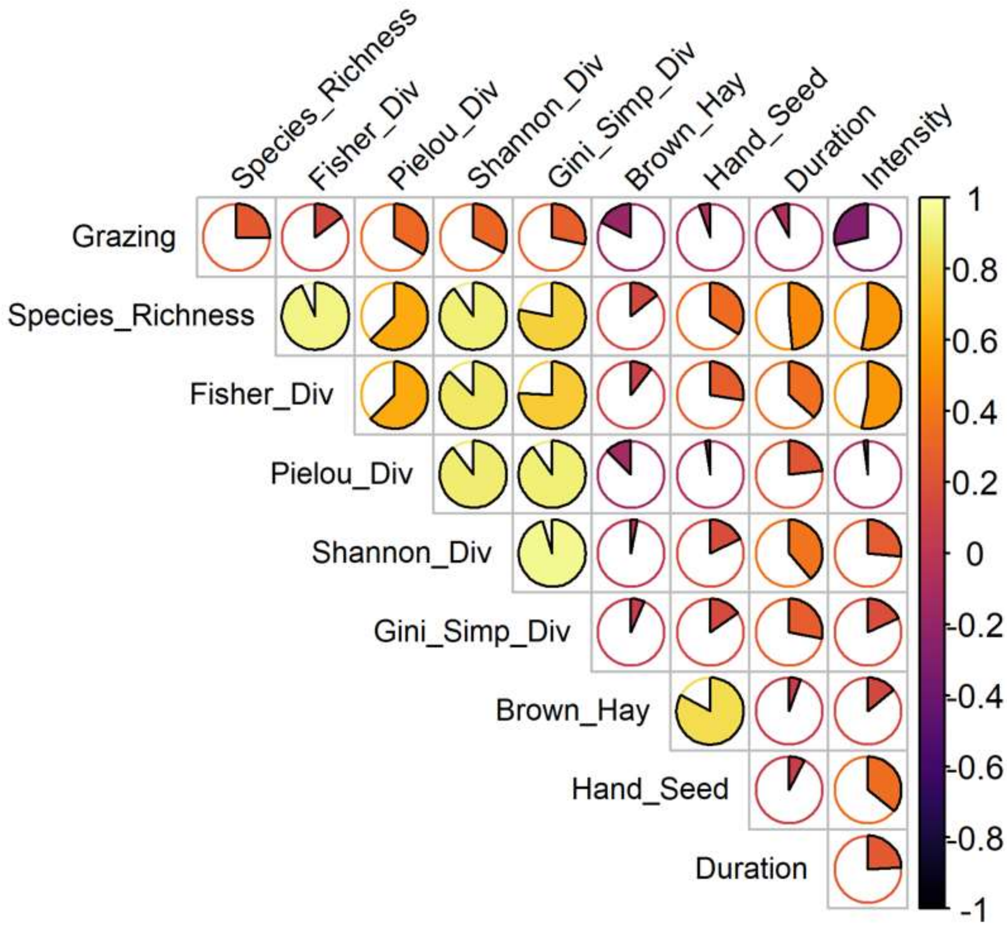


Figure 3.22: Correlation Plot of Plant Community species richness and diversity with ecological restoration activities

3.8.4 Plant community, Nectar and pollinator relationships with restoration activities

A PCA was undertaken on the plant community, nectar and FIT count data and the ecosystem restoration activities. The resulting biplot (Figure 3.23) illustrates the top 10 variables contributing to the two dimensions explaining 60% of the variance within the data, from which it can be concluded:

- Intensity and duration of restoration contribute strongly to the total plant species observed and the mean number of species per quadrat, around which the Raleigh Park restoration plots were clustered
- Nectar, floral units and flowering species are influenced by hand collected seed application and brown hay. This was most pronounced in the Chilswell and Lye Valley restoration plots; and

- All the control plots were associated with mean herb height, which was negatively correlated with all the other variables apart from shading

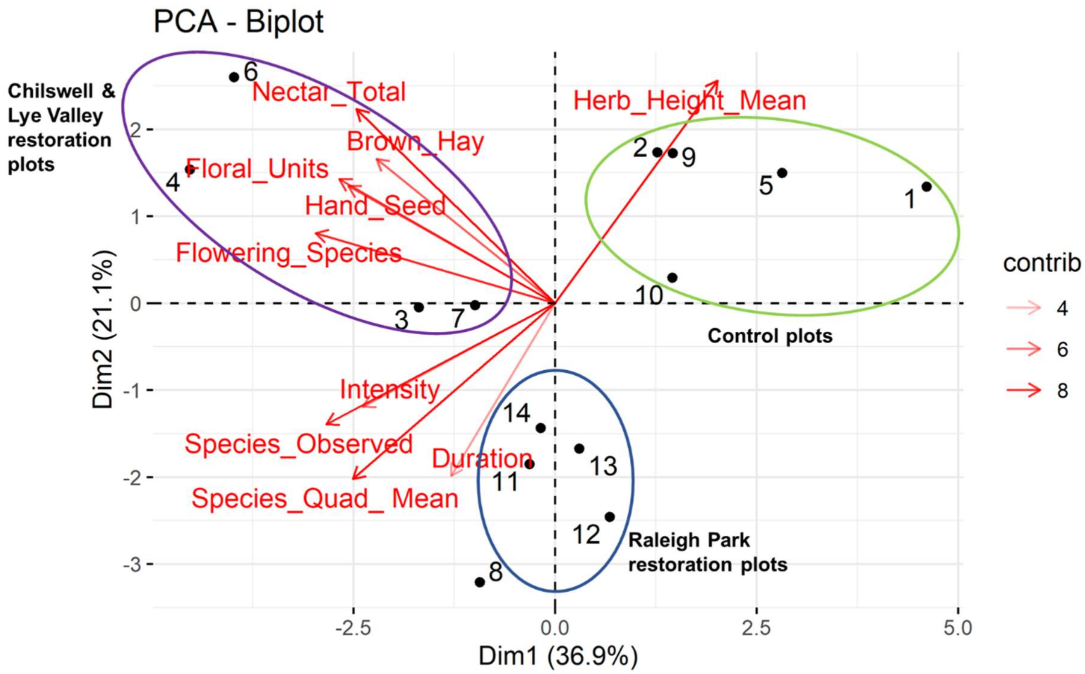


Figure 3.23: PCA Biplot of 10 top variables explaining relationships within the 2021 plant community, Nectar and FIT datasets and restoration techniques

Looking at correlations within the dataset, Figure 3.24 highlights further relationships between the restoration activities and response by biotic group:

- Grazing has a moderate negative relationship with nectar and floral units but no clear relationship with FIT counts;
- Brown Hay and Hand seed are moderately positively correlated with floral nectar resources but only weakly positively with FIT numbers, although brown hay correlates moderately with plot visits by hoverflies and flies; and
- No apparent relationship between duration of restoration and floral nectar resources and FIT counts, whereas intensity does have a weak positive relationship with these variables and numbers of bumble bees, hoverflies and flies.

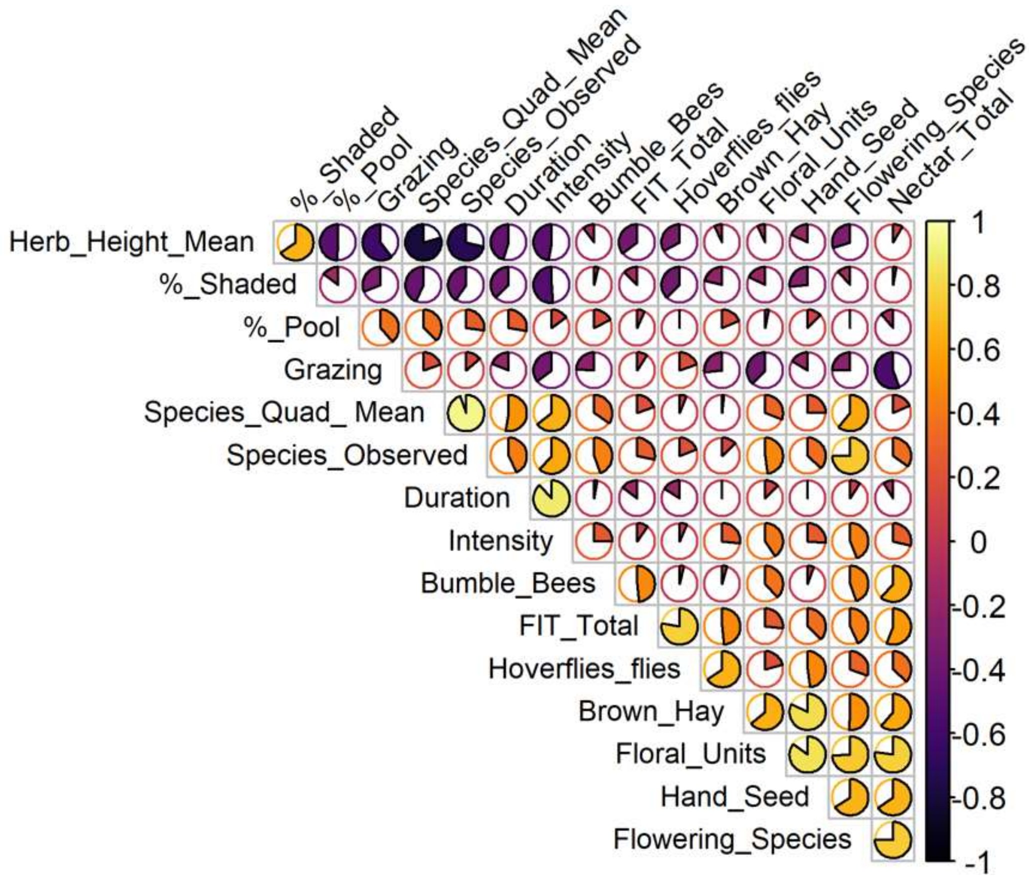


Figure 3.24: Correlation plot of plant community, Nectar and FIT variables with restoration activities

3.8.5 Analysis of Variance - Plant community and Restoration activities

The analysis in sections 3.8.3 and 3.8.4 revealed statistically significant relationships between plant community species diversity and ecological restoration activities. Linear regression modelling was used to understand how much the independent variables (restoration activities) contributed to the dependent variables (species richness). The linear regression model provides an estimate coefficient for the independent variable, predicting what numerical effect it would have on the dependent variable when all other variables are kept constant and it changes by 1 (Statology, 2018). The ANOVA then estimates how much variance is from each independent variable and its statistical significance.

Duration and Intensity and Brown Hay and Seed were correlated so could not be put into the same model. The model best fitting the dataset for all sites predicts that each unit of Intensity (1 cut & collect action), hand collected seed (1 application) and grazing (1 year) increase species per quadrat by 1.17 in total with grazing contributing the most (Table 3.15). The ANOVA finds the response to this combination of independent variables to be statistically significant; restoration techniques appear to be changing the plant community and increasing plant community diversity. The R-code, full results and the models tested are detailed in Appendix 11.

Table 3.15: Linear regression model and ANOVA explaining extent to which restoration technique affects response in plant community species richness

Linear regression model – dependent variable = species per quadrat				
Restoration (Independent Variable)	**Coefficient estimate (species added per variable unit)**	**Standard Error**	**t-value**	**Pr(>|t|)**
Intensity	0.25	0.014	18.35	< 2e-16
Seed	0.27	0.091	3.01	0.00274
Grazed	0.65	0.054	211.91	< 2e-16
Total	**1.17**			

Analysis of Variance					
Restoration	**Df**	**Sum Sq**	**Mean Sq**	**F-value**	**Pr(>F)**
Intensity	1	2613.9	2613.92	292.54	< 2.2e-16
Seed	1	48.6	48.60	5.44	0.02002
Grazed	1	1268.3	1268.32	141.95	< 2.2e-16
Residuals	594	5325.4	8.94		

Looking at sites individually there is considerable variation of response to restoration, summarised in Table 3.16 (full results in Appendix 11). Duration and intensity of management have had a considerably greater impact on species richness in Chilswell and Raleigh Park than at Lye Valley, although removing the LV_Benchmark plot (an outlier in terms of its very long management history and status as a benchmark ecosystem) reduces this somewhat.

Of particular interest is the large effect that Brown Hay has on increasing species richness at Chilswell Valley compared to Lye Valley. Removing the benchmark plot reduces this variance but was not statistically significant (highlighted in red). Hand applied seed increased species a similar amount at Chilswell Valley and Raleigh Park, but appeared negative at Lye Valley.

Table 3.16: Difference between sites in linear regression outputs of species richness for restoration variables

Research plots grouped by site		Restoration variable (modelled individually)			
		Duration	Intensity	Brown Hay	Seed
Chilswell Valley (n=5)	Coefficient estimate	1.18	0.48	6.22	1.55
	Pr(>|t|)p-value & ANOVA Pr(>F)	< 2.2e-16	<2e-16	2.42E-16	2.42E-16
Lye Valley (n=4)	Coefficient estimate	0.14	0.16	-0.63	-0.47
	Pr(>|t|)p-value & ANOVA Pr(>F)	1.56E-13	<2e-16	0.00121	0.00121
Lye Valley, no benchmark plot (n=3)	Coefficient estimate	0.63	0.18	-0.2542	-0.1906
	Pr(>|t|)p-value & ANOVA Pr(>F)	2.47E-08	7.06E-08	0.183	0.183
Raleigh Park (n=6)	Coefficient estimate	0.97	0.7	N/a	1.5
	Pr(>|t|)p-value & ANOVA Pr(>F)	<2e-16	<2e-16	N/a	1.87E-07

3.8.6 Analysis of Variance – Floral units and Restoration activities

Floral units were overwhelmingly influenced by hand applied seed, with the best fit model being each unit of intensity, seed and grazing producing 8.94 floral units per quadrat, with grazing having a negative effect (Table 3.17, full results for all models in Appendix 12). When considered as a single variable, brown hay also had a very strong association with increasing floral units, whilst grazing reduced floral units by -3.01 (Appendix 12). All these relationships are statistically significant.

Modelling the restoration activities individually by research site, the effect of brown hay and seed on floral units varies markedly, predicting a very large uplift effect on floral units at Chilswell Valley in comparison to Lye Valley (Table 3.18). No statistically significant relationship was observed at Raleigh Park, which only experienced seed application.

Table 3.17: Linear regression model and ANOVA explaining extent to which restoration technique affects numbers of floral units for all sites

Linear regression model – dependent variable = floral units per quadrat				
Restoration (Independent Variable)	**Coefficient estimate (floral units added per variable unit)**	**Standard Error**	**t-value**	**Pr(>|t|)**
Intensity	0.26	0.1095	2.331	0.020106
Seed	10.32	0.6939	14.873	< 2e-16
Grazed	-1.64	0.4288	-3.834	0.000141
Total	**8.94**			

Analysis of Variance					
Restoration	**Df**	**Sum Sq**	**Mean Sq**	**F-value**	**Pr(>F)**
Intensity	1	33411	33411	64.96	4.703e-15
Seed	1	119138	119138	231.63	< 2.2e-16
Grazed	1	7560	7560	14.70	0.0001406
Residuals	594	285975	514		

Table 3.18: Difference between sites in linear regression Coefficient estimates of floral units for restoration variables

Research plots grouped by site		**Restoration variable (modelled individually)**			
		Duration	**Intensity**	**Brown Hay**	**Seed**
Chilswell Valley (n=5)	**Coefficient estimate**	6.2	2.5	67.3	16.8
	Pr(>|t|)p-value & ANOVA Pr(>F)	1.55E-12	1.77E-12	<2e-16	<2e-16
Lye Valley (n=4)	**Coefficient estimate**	-0.2	0.11	10	7.5
	Pr(>|t|)p-value & ANOVA Pr(>F)	3.38E-01	0.59761	1.77E-08	1.77E-08
Lye Valley, no benchmark plot (n=3)	**Coefficient estimate**	3.6	1.1	10.3	7.7
	Pr(>|t|)p-value & ANOVA Pr(>F)	5.20E-03	3.07E-03	1.98E-07	1.98E-07
Raleigh Park (n=6)	**Coefficient estimate**	0.9	-0.7	N/A	-0.5
	Pr(>|t|)p-value & ANOVA Pr(>F)	0.00476	0.00306	N/A	0.574

3.8.7 Analysis of Variance – Nectar and restoration activities

Fewer restoration variables statistically influenced nectar values, with the best fit model being seed and grazing; although grazing was modelled to reduce nectar by 2.31mg, the impact of seed was to increase nectar by 0.91mg overall (Table 3.19, full models in Appendix 13).

Table 3.19: Linear regression model and ANOVA explaining extent to which restoration technique affects nectar values

Linear regression model – dependent variable = nectar (mg) per quadrat				
Restoration (Independent Variable)	**Coefficient estimate (nectar mg added per variable unit)**	**Standard Error**	**t-value**	**Pr(>|t|)**
Seed	3.22	0.8128	3.96	8.38e-05
Grazed	-2.31	0.4856	-5.104	2.17e-06
Total	**0.91**			

Analysis of Variance					
Restoration	**Df**	**Sum Sq**	**Mean Sq**	**F-value**	**Pr(>F)**
Seed	1	16441	16441	21.604	4.12e-06
Grazed	1	17418	17418	22.887	2.167e-06
Residuals	597	454344			

However, this varied by site, with no statistically significant association identified between nectar values and any restoration technique at Chilswell Valley (Table 3.20). Removing the floriferous CV_Ctl2(20) and CV_Ctl2 made no difference to this relationship (Appendix 13). Brown hay and seed did uplift nectar production in Lye Valley in a statistically significant manner both with and without the LV_Benchmark plot.

Table 3.20: Difference between sites in linear regression Coefficient estimates of nectar for restoration variables

Research plots grouped by site		Restoration variable (modelled individually)			
		Duration	Intensity	Brown Hay	Seed
Chilswell Valley (n=5)	Coefficient estimate	1.07	0.43	11.28	2.82
	Pr(>|t|)p-value & ANOVA Pr(>F)	0.20	0.198	0.101	0.101
Lye Valley (n=4)	Coefficient estimate	-0.38	-0.10	6.29	4.72
	Pr(>|t|)p-value & ANOVA Pr(>F)	0.0171	0.538	0.0000149	0.0000149
Lye Valley, no benchmark plot (n=3)	Coefficient estimate	3.08	0.91	5.45	4.09
	Pr(>|t|)p-value & ANOVA Pr(>F)	0.00443	0.00339	0.0014600	0.0014600
Raleigh Park (n=6)	Coefficient estimate	-1.4821	-1.0979	N/A	-0.4702
	Pr(>|t|)p-value & ANOVA Pr(>F)	0.00615	0.00401	N/A	0.748

3.3.8 Analysis of Variance – Insect pollinators and Restoration activities

The only restoration activity explaining some of the variation in FIT Counts was Brown Hay, with each application modelled to add 1.17 insect per quadrat (Table 3.21). The rest had no statistically significant relationships (Appendix 14). It was interesting that whilst grazing has reduced floral units it did not seem to have the same effect on flying insect visits.

Table 3.21: Linear regression model and ANOVA explaining extent to which restoration technique affected visits by flying pollinators

Linear regression model – dependent variable = flying insects per quadrat				
Restoration (Independent Variable)	Coefficient estimate (flying insects added per variable unit)	Standard Error	t-value	Pr(>|t|)
Brown Hay	1.17	0.5412	2.171	0.03155
Total	**1.17**			

Analysis of Variance					
Restoration	Df	Sum Sq	Mean Sq	F-value	Pr(>F)
Brown Hay	1	133.8	133.8	4.713	0.03155
Residuals	146	4144.1			

Looking at variations between sites, only in Lye Valley was Brown Hay found to be a statistically significant explanation for FIT counts and for Chilswell Valley only duration and intensity (Appendix 14).

3.8.9 Relationships between biotic and abiotic variables

A comparison of core abiotic and biotic variables undertaken in PCA suggests that iron and peat depth influences the biotic variables (except mean herb height) and are most pronounced for the Chilswell and Lye Valley restoration sites (figure 3.25). The other abiotic variables contribute little, suggesting they have no significant relationships in explaining the biotic variables.

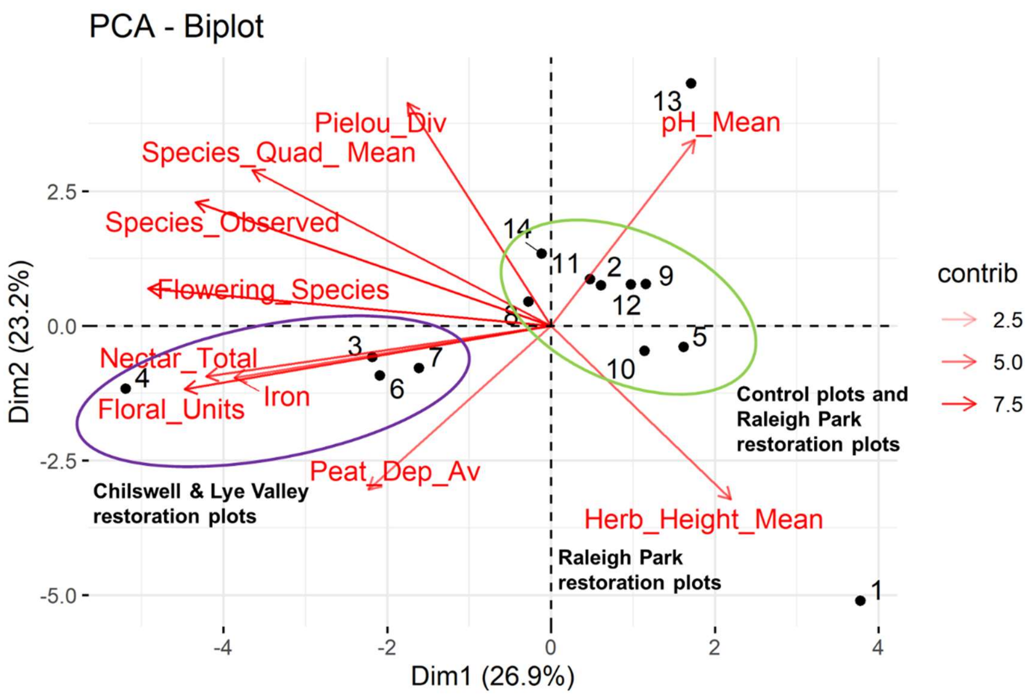


Figure 3.25: PCA Biplot of 10 top variables explaining contribution to dimensions and relationships within the biotic and abiotic datasets

The correlation plot in Figure 3.26 also suggests that abiotic variables explain little of the differences in the biotic dataset, with iron the only abiotic variable having a consistent positive correlation with a wide range of biotic variables including species richness, flowering species, nectar and a moderate to strong correlation with floral units. Peat depth is next, with a weak to moderate positive correlation with floral units and nectar.

Weak negative correlations included:

- nitrates, phosphates and potassium with floral & nectar resource;
- Mean water depth with plant species richness (i.e. species richness increases as mean WTD decreases),
- pH with flowering species and floral units; and
- calcium weak negative with FIT.

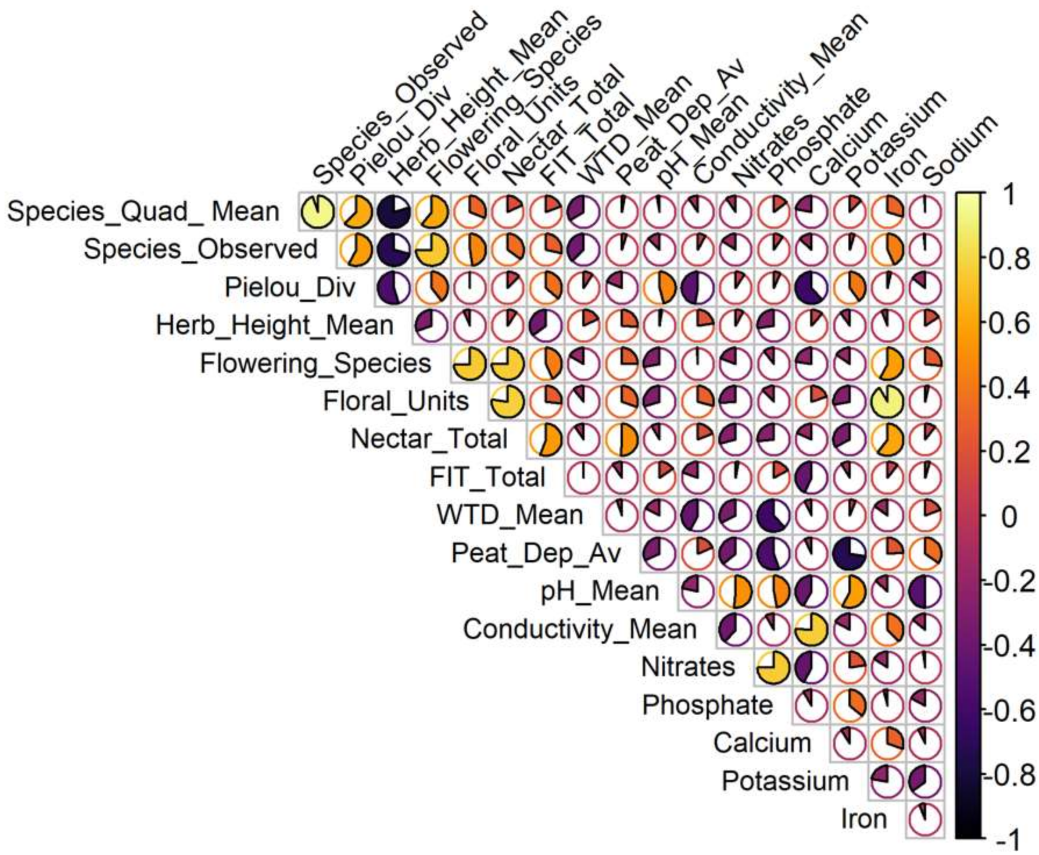


Figure 3.26: Correlation plot of biotic and abiotic variables

3.8.10 Analysis of Variance – Biotic and Abiotic

Linear regression modelling found that DWT, Phosphates, Calcium, Iron and Potassium have statistically significant relationships with species richness (Table 3.22). Nitrates, pH and Conductivity are modelled to have no statistically significant effect for sites overall. Modelling combinations of abiotic variables with the best fit biotic model (after first ensuring only one variable in pairs of highly correlated variables were included) is detailed in Appendix 15.

Table 3.22: Abiotic variables individually modelled against species richness

Linear regression model – dependent variable = species per quadrat				
Abiotic (Independent Variable)	**Coefficient estimate (Species added per variable unit)**	**Standard Error**	**t-value**	**Pr(>|t|)**
DWT (mean)	-0.19353	-0.02966	-6.525	1.53e-10
Phosphates	7.3135	2.6645	2.745	0.00625
Calcium	-0.023276	0.005564	-4.183	3.34e-05
Iron	0.022594	0.003942	5.731	1.63e-08
Potassium	0.11705	0.04878	2.40	0.0167

The models with the best fit for lowest residuals and sum of squares all included Phosphates, but this is deemed unreliable because of conflicting model outputs when the order of variables is changed and that only one plot (RP_Rest3) produce 1 different reading to the other plots. Consequently, phosphates were not considered further.

Excluding Phosphates, and combining the restoration variables the better fit model suggests that DWT and Potassium probably exert the most abiotic influence over species richness (Table 3.23). The model predicts that species richness increases as DWT decreases, whilst elevated Potassium increases species richness.

Looking at the abiotic variables individually by sites (Appendix 15) some interesting statistically significant relationships are apparent:

- The modelled effect of DWT on species richness is stronger at Lye Valley and Chilswell -1.22 and -0.55 respectively;
- Nitrates are predicted to reduce species richness in Lye Valley by -1.66 and in Chilswell Valley by -1.15

Due to the small numbers of samples for minerals and nutrients caution should be exercised in making conclusions about their role in affecting the biotic variables.

Table 3.23: Linear regression model and ANOVA explaining extent that biotic and abiotic variables affected species richness

Linear regression model – dependent variable = species richness				
Abiotic (Independent Variable)	**Coefficient estimate (quadrat species per variable unit)**	**Standard Error**	**t-value**	**Pr(>|t|)**
Intensity	0.245146	0.014553	16.845	< 2e-16
Grazed	0.528741	0.058834	8.987	< 2e-16
Seed	0.440881	0.088465	4.984	8.36e-07
DWT_Mean	-0.108956	0.023222	-4.692	3.42e-06
Calcium	-0.023852	0.004267	-5.590	3.57e-08
Potassium	0.248429	0.042817	5.802	1.10e-08
Total	**1.330389**			

Analysis of Variance					
Restoration	**Df**	**Sum Sq**	**Mean Sq**	**F-value**	**Pr(>F)**
Intensity	1	2151.1	2151.12	268.962	<2.2e-16
Grazed	1	1065.5	1065.55	133.229	<2.2e-16
Seed	1	198.6	198.6	24.837	8.357e-07
DWT Mean	1	126.6	126.57	15.826	7.869e-05
Calcium	1	316.0	315.98	39.508	6.630e-10
Potassium	1	206.1	206.13	25.773	5.255e-07
Residuals	553	4422.8	8.00		

4. Discussion

4.1 Wild Oxford biodiversity outcomes

Plant community biodiversity in all the research plots undergoing ecological restoration have experienced a statistically significant uplift over the lifetime of the Wild Oxford project. This applied both in comparison to prior or early in the restoration process and also to their respective control plots. Floral nectar resource, a proxy measure for invertebrate biodiversity, as well as utilisation by flying insect pollinators also showed statistically significant gains following ecological restoration at Chilswell and Lye Valley, but not at Raleigh Park.

Key findings from the results that support this finding are summarised in Table 4.1. The project has proved the null hypothesis that "The Wild Oxford project has not increased biodiversity in the Oxford Alkaline fens" to be false for plant community biodiversity at all three study sites and for floral nectar resource and utilisation by flying insect pollinators at Chilswell and Lye Valley. The null hypothesis could not be disproved for the proxy invertebrate measures at Raleigh Park.

Comparison with the control plots and for the plant community only, early or pre-restoration status, indicates the changes observed in the restoration plots could only have been caused by the restoration activities, as the control plots received no restoration intervention and abiotic factors were equal or very similar within and across the sites.

4.1.1 Plant community response

Plant communities in 4 of the 6 restoration plots have become more biodiverse. In 2017 the plots hosted low diversity S4 Swamp, S26d tall-herb fen or OV26 tall weed plant communities which are characterised by low diversity, lack rare plant species and generally have just 9-11 species on average per $4m^2$ sample (Huxley-Lambrick, 2002). After a minimum of 4 years restoration activities they had shifted to the more species rich M22a *Juncus subnodulosus-Cirsium palustre* fen-meadow. Wheeler (2002) found it to have a mean of 18.6 species per $4m^2$, almost double that of the S4, S26d and OV26 communities. Whilst sampling methods do not allow direction comparison, mean species per quadrat doubled between 2017 and 2021, matching the observations of Wheeler (2002) and providing confidence in the broad trends for increasing biodiversity identified in the NVC MAVIS analysis.

Table 4.1: Summary of Wild Oxford biodiversity outcomes

Biodiversity measure	Comparison	Finding	Statistically significant
Plant community			
Species richness	2017 v 2021	22% -143% uplift	✓
	Restoration vs control plots	19% - 491% increase	✓
RPR species	2017 v 2021	Increased from 1 to 10 species	N/a
NVC	2017 v 2021	4 plots shifted from S4/S26d/OV26 tall plant communities to more biodiverse M22a *Juncus subnodulosus-Cirsium palustre* fen-meadow. 2 plots showed little or no change	(MAVIS)
	Restoration vs control plots	Control plots remain dominated by less diverse, tall communities (S4a, S26 and OV26)	
Diversity indices	2017 v 2021	Diversity in 4 plots increased on all indices, little change for 2 Raleigh Park plots	(Dissimilarity & HCA)
	Restoration vs control plots	Controls have lower (or in the case of Raleigh Park, similar) diversity in all sites for all 4 indices than restoration plots	
Floral, Nectar resource & insect pollinators			
Species in flower	Restoration vs control plots	Over twice as many species flowering in Chilswell Valley and Lye Valley restoration plots (mean multiplier across sites is x2.7). Not observed at Raleigh Park.	N/a
Floral Units	Restoration vs control plots	Far greater floral units. 34 to 72 more floral units per quadrat for Chilswell and Lye Valley restoration plots receiving the most restoration activities. Not observed at Raleigh Park	✓
Nectar	Restoration vs control plots	Higher nectar values and diversity. 1,160mg and 853mg more nectar in the Chilswell and Lye Valley restoration plots receiving the most restoration activities. Not observed at Raleigh park where 1 control offered more nectar	✓
Insect Pollinators	Restoration vs control plots	More insect pollinators visiting restoration plots, +24 to +66 more insects for comparison pairs of control and research plots across the sites. This relationship was weaker at Raleigh Park with 1 research plot receiving less visiting insects than the control plot	✓ (not Raleigh Park)

Of the two restoration plots that did not show an apparent shift in plant community, LV_Rest1 was classified as S4 and had been subject to arson attacks in 2006 and 2014 when the area was dense reed (J Webb, personal communication 17 August 2021). It also has a history of nitrate pollution (Webb 2016b, Harte 2020) enriching the peat with nutrients and accelerating establishment of vigorous *Phragmites australis*. Nevertheless, biodiversity did increase with mean species per quadrat doubling from 4.2 to 8.4 by 2021. CV_Rest1 also retained the OV26 NVC community according to MAVIS, but mean species per quadrat had doubled from 6 in 2017 to 12 in 2021, a clear signal that restoration was shifting the plot towards a more biodiverse plant community.

Whilst a substantial increase in species richness and plant community is recorded, none of the restoration plots matched the NVC M13 *Schoenus nigricans-Juncus subnodulosus* mire plant community found in the LV_Benchmark plot. They had some of the Sedge species associated with M13 (in particular *Carex lepidocarpa, Carex panicea* and the ubiquitous *Carex flacca*), some of the floriferous herbs (*Pedicularis palustris* and *Mentha aquatica)* but the key species *Schoenus nigricans* and *Molinia caerulea* characterising M13 (Elkington *et al.,* 2001) were wholly absent from all restoration plots. Interestingly Wheeler (2002) states that *Anagalis tenella, Parnassia palustris* and *Pedicularis palustris* are not found in the Oxfordshire M22 fen meadow but these species were established and thriving in 3, 3 and 5 of the restoration plots respectively.

A further point to consider is that the NVC matches were relatively low, which Morris (2002) also found for reed bed communities at Cothill Fen. This may be partly explained by the difference between the NVC methodology which uses large $4m^2$ quadrats deliberately selecting areas of similar vegetation as opposed to the methods of Morris, Webb (2019a, 2019b and 2019c) and Snowdon (2017) and this project which employed many small samples together totalling $2.5m^2$ but randomly collected. An example of this was the lack of Carex acutiformis and Carex disticha in the restoration plots, even though these are a supposedly characteristic species of M22a (Elkington *et al.,* 2001). No *Juncus subnodulosus* was found in any Raleigh Park restoration plots, supposedly the defining species for this plant community. Nevertheless, Morris (2002) considered the data generated provided cover abundance values sufficient for meaningful matching against NVC communities. Diack *et al.,* 2013 also note that the

M22 plant community is very valuable but can be very species-rich, which corresponds with the findings of this study.

Perhaps the effect of restoration to date was to create a hybrid between the M13 and M22a plant communities, including the species easier to establish by donor seed but lacking those that needed very specific abiotic conditions, have poor germination or relied on clonal reproduction, which Hall *et al.,* 2010 found a key strategy for Carex re-establishment in degraded fens. With the plots under restoration for just 7 years or less it may take a longer time, possibly decades, for M13 communities to develop.

The restoration plots tended to fit the low sward characteristic of M13 but this was primarily a function of the tight cutting of reed in the early summer in the Chilswell and Lye Valley plots rather than a characteristic of the vegetation, except in the areas where *Pedicularis palustris* had taken over and completely supressed it. The prolonged grazing at Raleigh Park did create a noticeably lower sward but this was dominated by patches of taller *Juncus inflexus*, *Glyceria* and *Epilobium spp* rather than Carex.

If success was narrowly defined in terms of restoring all the plots to the target reference ecosystem of M13 mire, then the Wild Oxford project has not yet achieved this goal. This may not be possible if the peat substrate is rich in nutrients, as Wheeler (2002) found substrate fertility explaining why M13 forms instead of M22a when base-rich mineral groundwater and high water tables are held constant. Strouh *et al.,* (2012) reported that even after 60 years, formerly farmed, degraded fens located immediately adjacent to the extent lowland alkaline Wicken Fen could not recover their desired key vegetative components. Instead, a new wetland ecosystem had developed which did not fit conventional NVC plant communities, but still had substantial ecological value which would likely enrich given sufficient time. Defining plant communities into simple categories that can be differentiated from others is a human construct and ignores the reality that no habitat or ecosystem is the same and this uniqueness is an aspect of biodiversity to be valued (Moreno-Mateos, *et al.,* (2015).

It may be impossible to replicate a theoretical model plant community and so it is more pragmatic not to consider this a rigid goal but a guide for ecological restoration as suggested by Clewell and Aronson (2013). The differences between all the restoration plots tend to support this; biodiversity has increased, but in unpredictable ways with

Chilswell's restoration plots supporting substantial plant and huge floristic diversity whilst LV_Rest1 has remained stubbornly slow in shifting plant community but still supported very high floral units of *O. lachenalia* and hosted the greatest numbers of hoverflies and flies of any restoration plot. Even with sites so close together it is unclear why *Eleocharis uniglumis* and *Carex dioica* are only found in Raleigh Park and Lye Valley respectively.

4.1.2 Floral and Nectar resource

Fen ecosystem restoration in Chilswell and Lye Valley has increased their value for invertebrate pollinators. However, the project has very likely underestimated this value; stemming firstly from missing nectar sugar estimates and secondly because pollen resource could not be quantified. *Filipendula ulmaria,* and *Hypericum tetrapterum* were both abundant and produced substantial floral units within restoration plots, but in the absence of a published estimate of mg nectar sugar per floral unit they did not contribute to plot nectar resource estimates; this potentially introduced a negative bias to plots rich in *Hypericum tetrapterum* at Lye Valley.

There is some uncertainty with *Pedicularis palustris,* where the nectar value for *Rhinanthus minor* was substituted as a proxy. The proxy nectar value of 109µg seems low compared to *Vicia cracca* (484µg). The latter has very similar flower morphology and both species were observed by the FIT Counts to be predominantly visited by bumble bees. Given that *Pedicularis palustris* is exclusively pollinated by bumble bees (Macior, 1993) logically it must carry a reasonable nectar and/or pollen reward. As such an abundant plant a small increase in nectar value could substantially uplift the plot overall nectar value and produce a stronger correlation between restoration and resource utility for flying insect pollinators.

Turning to the second factor in undervaluing potential fen pollinator resources, pollen is a key source of protein for invertebrate pollinators, providing essential amino acids to bumble bees (Ceulemans *et al.,* 2017). Species in the *Solanum* genus have double the mean protein concentration of *Cirsium spp* (Pamminger *et al.,* 2019) and bumble bees were recorded utilising *Solanum dulcamara*, yet its nectar contribution to the plot resource value is insignificant. However, pollen values could only be found for 12 of the 63 species in flower and so this this resource could not be quantified. It seems likely the floriferous restoration plots offered a diversity of pollen resources which were attractive to a range of different invertebrate pollinators and by omitting pollen, the total floral resource for invertebrates has been substantially undervalued.

The poor floral and nectar resource in the Raleigh Park restoration plots can be explained by two factors. Firstly, the species that produced 72% of floral units (*Pedicularis palustris, Galium uliginosum, Oenanthe lachenalii* and *Jacobaea erucifolia*) and 53% of the nectar (*Jacobea erucifolia, Lythrum salicaria, Oenanthe lachenalii, Pedicularis palustris* and *Vicia cracca*) in Chilswell and Lye Valley were absent from Raleigh Park (apart from non-flowering *Pedicularis palustris* and *Oenanthe lachenalii* in RP_Rest4).

Secondly, there is a statistically significant reduction in floral and nectar resource in the plots that were grazed, all in Raleigh Park. This was very obvious visually between sites (Figures 4.1 and 4.2) and within Raleigh Park in comparison to previous years (Figure 4.3 and 4.4). In 2021 the cows were kept on-site far longer than previous years until mid-August, browsing both the palatable plants and flowers, especially the orchid *Dactylorhiza fuchsii*. Consequently, *Cirsium palustre* became the dominant flowering plant and source of nectar, presumably being unpalatable to cows and providing 87% of all nectar in the Raleigh Park restoration plots, more than double the proportion it provided in Chilswell and Lye Valley.

Figure 4.1: ***Cirsium palustre*** **provides only floral resource in RP_Rest2**

Figure 4.2: Abundant *Pedicularis palustris, Jacobaea erucifolia* and *Lythrum salicaria* in CV_Rest2 (both photographs taken July 2021

Figure 4.3: Abundant *Filipendula ulmaria* and *Lotus pedunculatus* in RP_Rest4, July 2019 after cattle removed

Figure 4.4: High intensity grazing with cattle on-site in peak flowering season resulted in no floral units in RP_Rest4, July 2021

Looking at control plots, some had high floral resource provision, notably RP_Ctl1 which offered very high nectar value far in excess of the Raleigh Park restoration plots. This was primarily due to the presence of *Impatiens gladulifera* which has 7668μg nectar sugar per floral unit (Baude *et al.,* 2016) and a species absent elsewhere following proactive removal. This plot is fenced off and cows do not have access, so had this been grazed, as for the other control RP_Ctl2, it is very likely the floral and nectar resource would have been low. High nectar provisioning was also recorded at CV_Ctl2 after 1 year of restoration, primarily explained by the high nectar producing *Eupatorium cannabinum* offering an estimated 57017μg per capitulum (Baude *et al.,* 2016, Timberlake *et al.,* 2019) and 86% of the nectar in the plot. It was unclear why it was present in this plot but entirely absent from CV_Ctl1, in which *C.Sepium* was the lone floral resource. One explanation is that the site may have experienced some unrecorded past management intervention or event which enabled it to establish against the dominance and heavy shading of *Phragmites australis*.

4.1.3 Flying insect pollinators

Visits by flying insect pollinators seem to broadly reflect the findings for floral and nectar resource, implying that restoration is benefiting pollinators in Chilswell and Lye Valley. Where the evidence differs is that Raleigh Park still appears valuable, although the relationship is not statistically significant. This might simply reflect that *Cirsium palustre* is such an important plant for pollinators that even at low floral unit abundance it still attracts flying insect pollinators in numbers.

Pollinator visits seem to be a function of both flower species, time of flowering and predicted nectar value but also vary by insect group. That the invasives *Cirsium arvense* and *Impatiens glandulifera* attracted the first and fourth highest mean flying insect visits, yet are generally considered to be undesirable and are actively targeted for removal in restoration is a paradox. The decisions behind ecosystem restoration activities are multi-faceted and more nuanced by human judgements on what plant or invertebrate groups should and shouldn't benefit than perhaps acknowledged. If a fen restoration goal was just to increase ecosystem services for human benefit, the preference of Honey Bees for *Lythrum salicaria, Epilobium hirsutum and Impatiens glandulifera* might hypothetically justify encouraging these floristic species over others such as *Oeanthe lachenalii* (the most visited species overall and particularly value for hoverflies and flies) to boost economic value from honey products.

The low insect visits, floral units and species in flower at LV_Bench plot is in contrast to the Chilswell and Lye Valley plots. It raises the question as to whether the restoration plots represent a stage in a transition to the less floristic M13 community and whether this transitional stage is actually more desirable and more biodiverse when factoring in invertebrate pollinators? This highlights the risk of not thinking of biodiversity in holistically, as pollinators are just one part of the rich fenland invertebrate community which the project was unable to explore. An invertebrate survey of the Raleigh Park fens (Gregory, 2021) found 235 invertebrate species of which the majority, 138 species, were not involved in pollination or consuming nectar and pollen even as a small part of their diet (excluding all beetles, solderflies, hoverflies, bees, wasps and allies). Similarly, at nearby Hinksey Heights Alkaline fen, 66% of recorded invertebrates did not feed on nectar or pollen resources (FHT 2019). On balance, flying insect pollinators are a part indicator for invertebrate biodiversity rather than an absolute measure.

4.2 Effectiveness of fen ecosystem restoration activities

The analysis in section 3.8 has demonstrated the restoration activities undertaken through the Wild Oxford project have statistically significant relationships with biotic variables and they have increased plant community species richness, floral nectar resources and flying insect pollinator visits. These are summarised in Table 4.2, collectively proving the null hypothesis “There is no relationship between the application of different ecological restoration techniques and biodiversity” to be false.

Table 4.2: Summary of how restoration activities influenced biodiversity measures across all sites

Biotic Variable (Dependent)	Restoration activity variable (independent)		
	Correlations & PCA		Linear Regression Modelling and ANOVA (statistically significant)
	Positive correlation	Negative correlation	
Plant community species richness	• Intensity • duration • hand collected seed (weak) • grazing (weak)		Intensity, seed and grazing predicted to **add 1.17** species/quadrat per unit of restoration activity
Floral units	• brown hay • hand collected seed • intensity (weak)	• grazing	Intensity, seed and grazing predicted to **add 8.94** floral units/quadrat per unit of restoration activity
Nectar	• brown hay • hand collected seed • intensity (weak)		Seed and grazing predicted to **add 0.91** mg nectar/quadrat per unit of restoration activity (this includes a reduction of nectar by -2.31mg from grazing)
Flying insects	• Brown hay (hoverflies and flies only) • Intensity (weak bumble bees, hoverflies and flies only)		Brown Hay application predicted to **add 1.17** insect visits/FIT quadrat per unit of restoration activity

4.2.1 Intensity and Duration

The intensity at which restoration is conducted seems to be a stronger influence on species richness and floral units than duration. To some extent it is hard to separate the two as over time the variable for intensity will increase, but plots that had 3 cuts a year (Chilswell and Lye Valley) had a more pronounced response than in Raleigh Park where only 2 a year occurred and none in 2020 due to the pandemic (Appendix 1). This reflects what is reported in the literature; Middleton (2006b), Klimkowska *et al.,* (2010), Nielsson (2015) and Ross *et al.,* (2019) found repeated mowing and scrub removal essential pre-requisites for rich fen rehabilitation and maintenance. Menichino *et al.,* (2016) also recommend that in the absence of grazing, M13 communities have frequent mowing with alternating early and late seasonal cuts to increase and maintain species richness through reducing herb height, halting succession to shrubby and woodland communities and creating bare ground for recolonisation. Sundberg (2012) achieved a doubling in species richness in a previously degraded rich fen through Shrub removal and mowing after 6 years, with areas exposed to additional intensive mowing having more abundant target species.

Removal of biomass serves two purposes; firstly to lower the fertility of the peat by removing nutrients and secondly to knock back the vigour of grasses, reeds and tall herbs, reducing herb height and letting in light which the more specialist fenland species require to establish and flower (Middleton, 2006b). Based on experience of restoring the Oxfordshire alkaline fens, Webb (2020b) states that timing of cuts in plots dominated by *P.australis* seem to be important, with an early spring cut at ground level followed by an early summer cut above the emerging herb layer key to weakening reeds when they are growing most vigorously and drawing upon the energy and resources in their rhizomes. This technique is employed at both Chilswell and Lye Valley and this project has provided evidence of its efficacy. By contrast the autumn cut & collect is simply removing cellulose and dead plant matter containing relatively little nutrients, but important to remove ground litter and create opportunities for seedlings to emerge next spring and the shorter sward plant community plants to establish (J Webb 2021, person communication 17 August).

4.2.2 Brown Hay and Seed

The application of brown hay and seed have explained increases in all the biotic variables, in particular seed for plant species richness, floral units, nectar and brown hay for insect counts. Some plants clearly returned from the seedbank, for example C*arex distans* found only in CV_Rest1 which had no brown hay or seed applied and has a long seedbank life (Webb, 2019a). The sedges *Carex flacca, Carex panicea* and *Carex lepidocarpa* re-established in the Chilswell and Lye Valley research plots where brown hay was not applied probably also came from the seedbank, having longevity of 20+, 15-20 and 6+ years respectively (Schütz 2000).

Whilst Sedge seeds can remain viable a long time if undisturbed in the soil (Schütz 2000), the literature suggests this is more often not the case in fen restoration, for example the *Carex* spp meadow seedbank had largely disappeared from former wetlands after 16 years of farming (Wang *et al.,* 2017) whilst van der Valk (1999) found that after 6 months germination of *Carex spp* was poor when grown ex-situ. Bossuyt and Honnay (2008) reviewed 103 seedbank studies and found those of marsh ecosystems to be dominated by low diversity and relatively persistent seeds but after

5 years of degradation, restoration relying on the seedbank alone was unlikely to be successful. The exception for marshlands were *Carex spp* and *Juncus spp*, the latter having great longevity and abundance, remaining viable for a long time. This may explain how *Juncus spp* appeared in the plots not receiving brown hay, especially obvious in the restoration plots at Raleigh Park.

Whilst *Hypericum tetrapterum*, *Lythrum salicaria, Scrophularia auriculata* were all present in low numbers in the study sites prior to restoration (*Lythrum salicaria* was not found in Raleigh Park), *Oenanthe lachenalii, Parnassia palustris and Vicia cracca* were absent and *Pedicularis palustris* found in only 1 quadrat at Lye Valley. They seem very likely to have established due to their re-introduction by seed with *Oenanthe lachenalia, Parnassia palustris* and *Vicia cracca* still restricted to their re-introduction plot CV_Rest2 at Chilswell Valley. *Galium uliginosum* and *Succisa pratensis* occurred in very low numbers (1 and 4 quadrats respectively) in the adjacent CV_Rest1 plot which received no hay or seed. *Succisa pratensis* has very short-lived seed (Webb, 2019b) and can only have returned to Chilswell Valley through the restoration activities. This mirrors the findings of Hall *et al.,* (2010) that herbaceous flowering plants did not return from the seedbank in a fen restored from *Tyhpa* dominance, whilst Rasran *et al.,* (2007) found after five years degraded fen meadows receiving no hay saw the return of peatland peats in less than 5% of quadrats, compared to 28% of quadrats in plots receiving hay.

The statistically significant difference between plots not receiving donor seed suggests that recovery from the seedbank alone likely cannot be relied upon; re-introduction by seed and brown hay has been a fundamental driver of successful restoration. With the remnant oxfordshire fens being so small, fragmented and isolated, deliberate re-introduction by seed may be the only way that species richness and genetic diversity can be maintained. As the majority are no longer grazed, large herbivores cannot move seeds between them, which Webb (2019b) and Middleton *et al.,* (2006a) indicated would historically have been an important mechanism for seed dispersal via hooves and fur (Middleton *et al.,* 2006b).

Application of brown hay and seed served another important role; the deliberate re-introduction of *Pedicularis palustris,* a hemiparasite and ecosystem engineer proven to reduce biomass and height of dominant *Carex* and *Juncus* (Decleer et al., 2013). Webb (2020b) evidenced its effectiveness in the Chilswell and Lye Valley research plots by substantially reducing the height of *Pedicularis australis* and as a biennial, then creating open patches of bare ground for colonisation by other wetlands upon its death. The project is unable to disentangle the extent which the application of *Pedicularis palustris* influenced biodiversity in comparison to the other management variables. However, two pieces of evidence suggest it is a very significant component in restoration; firstly, the substantial uplift in species richness, floral units and nectar in the Chilswell and Lye Valley restoration plots in comparison to Raleigh Park, where it had only recently been applied in one plot. Secondly, large patches of *Anagallis tenella*, and *Triglochin palustris* and *Galium uliginosum* were only found in CV_Rest2 where *P.palustris* had previously flowered and died, supporting the assertion of Webb (2020b) that in creating bare ground, it is an efficient and important mechanism by which lost fen plant communities can re-establish.

Figures 4.5 – 4.7 provide a visual illustration through summer 2021 of *Pedicularis palustris* supressing reed growth and a comparison to the CV_Ctl1 plot (Figure 4.8). By mid-August *Pedicularis palustris* was almost 1m tall and had formed a floriferous sward totally outcompeting *Phragmites australis* (Figure 4.9).

Figure 4.5: CV_Rest2, early June 2021. *Pedicularis palustris* in centre

Figure 4.6: CV_Rest2, early July 2021. Scarcely any reeds, many flowers

Figure 4.7: CV_Rest2, mid-August 2021, setting seed

Figure 4.8: CV_Ctl1, monoculture of *Phragmites australis*, mid-August 2021

Figure 4.9: Floristically diverse sward in CV_Rest2 with *Pedicularis palustris, Jacobaea* erucifolia and *Eupatorium cannabinum*, early August 2021

4.2.3 Grazing

Grazing appeared to explain some of the increased species richness at Raleigh Park, probably through supressing rank grasses, reducing competition from tall ruderal herbs and preventing succession to Willow Carr (Merriam *et al*., 2018) and mirrors the increase in species observed by Groome and Shaw (2015) in low intensity grazed plots on a lowland mire. Preventing succession to wet woodland may be especially important at Raleigh Park, where the main fen, plots RP_Rest1 and RP_Rest2, had until very recently been Willow Carr and abundant seedlings and stump regrowth were recorded in 58% and 65% of quadrats respectively in 2021 vs just 20% and 28% in 2017. Middleton (2013) found evidence that cattle hoofprints may actually create the opportunities for germination of tree seedlings which become problematic when grazing ceases as they can develop into a tree canopy and shade out the short fen plant communities.

It does not explain the rich diversity of the Chilswell and Lye Valley restoration plots, nor the LV_Bench target plot as these were not grazed. Furthermore, Raleigh Park had significantly lower floral diversity and numbers due to the cows preferentially eating the flowers of palatable herbaceous plants (also found by Groome and Shaw, 2015). Substantial poaching and upturned turves were also observed in the wettest areas of the main fen, which may be positive for creating germination opportunities, but not if the propagules of the target plant community are unavailable as cattle have prevented reproduction and reduced seed resource. Groome and Shaw (2015) found this type of trampling to be particularly harmful for *Sphagnum* bog mosses, in spite of the benefits for the wider plant community, which they emphasise were only achieved by low density, carefully managed grazing. This was not observed at Raleigh Park. The timing of grazing late into the summer in 2021 is not just problematic for invertebrate pollinators having less nectar and pollen resource but also the numerous phytophagous invertebrates that feed on green plants (Denton, 2014).

A potential future problem of grazing may be to inadvertently accelerate the dominance of *Juncus inflexus* which the cows did not eat. Table 4.3 shows that in the grazed plots of Raleigh Park is it becoming dominant with substantial increase in abundance since restoration began in 2017. By comparison it is scarcely present in all the ungrazed plots. As more palatable species are preferentially targeted by cattle, they diminish in abundance with consequent expansion of *Juncus inflexus* over an increasing area (Mark *et al.,* 2020). Somewhat ironically, clumps of *Juncus inflexus* did shelter flowering plants from grazing (Figure 4.10).

Table 4.3: Abundance of *Juncus inflexus* in the study sites

Grazed plot	Abundance of *Juncus inflexus* (as % of quadrats)	
	2017	**2021**
RP_Ctl2	Data unavailable	30%
RP_Rest1	7.5%	23%
RP_Rest2	10%	45%
RP_Rest3	Data unavailable	53%
RP_Rest4	Data unavailable	83%
Ungrazed plot	**2017**	**2021**
CV_Ctl1	Data unavailable	0%
CV_Ctl2	Data unavailable	0%
CV_Rest1	0%	2.5%
CV_Rest2	0%	0%
LV_Ctl	Data unavailable	2.5%
LV_Rest1	2.5%	0%
LV_Rest2	0%	2.5%
LV_Bench	Data unavailable	0%
RP_Ctl1	Data unavailable	0%

Overall, the project findings tend to support the views of Middleton *et al.,* (2006b, 2013) and Stammel *et al.,* (2003) that high grazing pressure on fens and fen meadows has a negative effect on biodiversity, reducing species diversity and degrading soil.

Figure 4.10: *Stellaria graminea* and *Cerastium fontanum* protected from grazing in Raleigh Park by unpalatable *Juncus inflexus*

4.2.4 Raising the water table by rewetting

Rewetting, by blocking artificial drainage channels to slow the rate at which groundwater leave the sites with the objective of raising the year-round water table towards the surface, is an important fen ecosystem restoration technique (Klimkowska *et al.,* 2007 and 2010). Whilst it was employed in different ways across all sites, it was not possible to separate this variable out from other restoration measures within the timeframe of the project.

4.3 Influence of abiotic factors on ecosystem restoration

The main findings are summarised in Table 4.4, suggesting that abiotic factors do influence the plant communities, but it is less clear how they interact with restoration. Statistically significant relationships were found with DWT and some minerals across all sites but not for nitrates and with spurious model outputs for phosphates (as explained in section 3.8.10). Stronger relationships were found for DWT and nitrates when sites were examined individually. Overall the project has determined that the null hypothesis “There is no relationship between abiotic factors, biodiversity and ecological restoration actions” is false. This is chiefly in the role that DWT and Calcium play in respectively maintaining the high water table and in limiting Phosphate, key conditions that M13 and M22c stress tolerant plant communities require (Boyer and Wheeler 1989, Wheeler 2002, Rozbrojova, and Hajek 2008)

Table 4.4: Summary of abiotic variables influencing research plot biodiversity

Biotic Variable (Dependent)	Abiotic variable (independent)		
	Correlations & PCA		Linear Regression Modelling and ANOVA (statistically significant)
	Positive correlation	Negative correlation	
Plant community species richness	• Iron (weak)	• DWT (weak)	DWT, Calcium and Potassium predicted to **add 0.12** species/quadrat per unit (note that DWT predicts species fall by -0.109 per cm of DWT rise to ground surface)
Floral units	• Iron (strong) • Peat (weak)	• Nitrates (weak) • Potassium (weak) • pH (weak)	
Nectar	• Iron (moderate) • Peat (weak)	• Nitrates (weak) • Phosphates (weak) • Potassium (weak)	
Flying insects		• Calcium (weak)	

4.3.1 Water Table Depth

The mean annual WTD for LV_Bench is consistent with the values reported by Wheeler (2002) for M13 *Schoenus nigricans-Juncus subnodulosus* mire and the generally high mean WTD of the other plots are consistent with Alkaline fens (Large et al., 2007 Diack *et al.,* 2013), except for perhaps RP_Rest3 which had extreme variability.

The apparent link between lower mean WTD and increased plot species richness may appear counter-intuitive but might be explained by two possible factors. Firstly, 5 of the 6 plots with below average species richness and a combined mean annual WTD of -3.3cm were control plots. As discussed in section 4.1 and 4.2 and by the weak modelled contribution of WTD to species richness, the low biodiversity in these plots is most likely to be explained by absence of restoration activities not WTD. Secondly, 5 of the 8 plots with above average species richness were M22a fen-meadow which Wheeler (2002) found to be characterised by a mean annual WTD of -16.1cm. By contrast half the species poor control plots were S4 reed swap or S26 reed fen which generally have wetter, high water tables year-round (Rodwell 1997, Large *et al.,* 2007).

WTD seems to have quite strong influence in predicting species richness at Lye Valley; this may simply be a function of the low diversity LV_Ctl having a much higher water table year and little seasonal variation in comparison to the other more biodiverse plots. Chilswell Valley also seems more influenced by DWT with both restoration plots having markedly lower summer water tables. M22a tend to occur in sites with low summer water tables (Wheeler, 2002), which seems to be reflected in the hydrology in CV_Rest2, LV_Rest2 and RP_Rest4 and could support the NVC match in these plots being closer to M22a than M13.

Restoration in all sites included activities designed to retain clean groundwater on-site, reduce peat erosion and at Chilswell and Lye Valley prevent nutrient enrichment from contaminated surface water. The effect of these measures and whether they changed water flows at or immediately below the ground surface could not be assessed in Chilswell and Lye Valley as they took place before the dipwells were installed. However, the effect of blocking the artificial drain in RP_Rest3 in January 2021 was captured by the dipwell measurements (Figures 3.13 and 3.14). The result was dramatic, with WTD remaining at ground level for nearly six months until June 2021 when it started to drop steadily, although not to the extremes of late summer 2020. This might have been due to cattle damaging the log dams and water escaping more quickly, rather than reduced groundwater flows from changes in rainfall which was not observed in the other plots.

It was beyond the scope of the project to consider the effect of climate on groundwater flows. However, at just 0.4km^2 the very small catchment of Raleigh Park, at least one-third urbanised with sealed surfaces, suggests it may be the most vulnerable to climate disruption and volatility from droughts; in 2018 a major drought resulted in the spring-fed stream and pond disappearing and the main fen partially drying out (Webb, 2019c). The catchments of Chilswell and Lye Valley are more than twice the size and potentially better buffered against future worsening extremes of drought. Lye Valley's catchment is almost entirely urbanised which may reduce the volume of rainwater infiltrating into the soil and recharging the groundwater flows, whilst increasing the speed and volume of surface run-off directed through the fen by the storm drains (Webb, 2019b). The substantial seasonal variation in dipwell temperatures in Raleigh Park (Figure 3.15) may potentially be a consequence of the small catchment and shallower groundwater flows between sites. Chilswell Valley has the most natural, un-urbanised catchment which might explain why water temperature variation is less than Lye Valley.

The very wet winter of 2020/21 was followed by a damp, cool spring and relatively low summer evapotranspiration. Surface water pools in the LV_Bench plot remained wet all summer in 2021, the first time this was observed for decades which Webb assigns to the 6-month lag between rain falling, filling the aquifer as groundwater and emerging in the spring-fed fens (2021, personal communication 28 September). This may have given the false impression that the fen groundwater supply is more reliable and the plant and invertebrate community more robust to climate change and future extreme weather events than in reality.

4.3.2 Nutrients

Phosphate levels were low and within bounds expected for Calcareous fens where they are limited by co-precipitation with Calcium (Boyer and Wheeler 1989, Emsens *et al.,* 2017). Only in three plots, LV_Ctl, CV_Ctl2 and RP_Rest3, did nitrate exceed the mean concentration of 0.69 mg/L reported by Hájek *et al.,* (2002) for clean, tufa-forming Alkaline fens. Whilst not a factor influencing restoration across all sites combined, modelling did predict a statistically significant effect of nitrates reducing species richness in both Lye Valley and Chilswell Valley when sites were examined individually. This supports previous investigations into water quality in Lye Valley where high nitrate levels of 7.2 – 27.7mg/L were reported in springs above the valley slope and to the North of the LV_Rest2 plot by Harte (2020) and 5-10mg/L nitrate by Webb (2016b). Moderate nitrate pollution was also recorded in the LV_Ctl and LV_Rest1 plots by Webb (2016b) and up to 10mg/L in the Lye Brook immediately above the main fen by Lamberth (2007). The source pollution was assigned to building works, leaking sewers and mains water (Harte, 2020, Webb 2016b and 2019b).

More positively Higson (2020) and Webb (2016b) confirmed very low levels of nitrates in the pools on the East side of the fen closest to the LV_Bench plot. This matches the findings of this study and its clean, very low nutrient, base-rich water is as important to maintaining its M13 plant community as the regular mowing and biomass removal

(Wheeler, 2002, Emsens *et al.,* 2017). Wheeler (2002) suggests abandoned M13 plant communities can survive scrub and ruderal invasion a long time without intervention due to the very low nutrient and high water table abiotic conditions they have adapted too.

Whilst the project did not record nitrate to be a problem in the Chilswell or Lye Valley restoration plots, it seems probable that nitrate pollution and its arson history is a factor in the lower species richness in the LV_Rest1 plot, despite the intensity of restoration and frequency of brown hay and seed application. The low measurements may have been explained by strong groundwater flows following the wet winter 2020-21 causing greater rates of flushing through the nitrates than in previous years as low water table and drought conditions tend to concentrate pollutant levels (J Webb 2021, personal communication 28 September). The role of atmospheric nitrogen deposition on the research plots is unknown but high rates of 18-23kgN/ha/yr are experienced around Oxford (CEH, 2016) and Jani *et al.,* (2020) found that it was the largest source of Nitrate in urban stormwater runoff. Fens are nitrogen sinks and may be more buffered than other terrestrial ecosystems (Lamberth 2007) but it seems probable atmospheric nitrogen is another long-term negative pressure on the low nutrient Alkaline fen plant community through eutrophication and acidification (Plantlife, 2017).

It will be interesting to observe how the CV_Ctl2 plot develops under restoration over time and whether its elevated nitrate levels found in winter 2021 will slow the restoration process to rich fen as experienced in Lye Valley. Webb has reported high nitrate loads of 5-10mg/L in the tufa spring of CV_Rest2 over successive years (2016a, 2019a) which was presumed to be from agricultural fertilisers (Webb, 2019a), however these were not detected in the CV_Rest2 dipwell by this study. Alkaline fens are phosphate limited (Wheeler, 2002, Rozbrojova, and Hajek 2008, Emsens *et al.,* 2017) and Webb (2016a) theorised that the high nitrate levels would not necessarily be a problem due to phosphate being unavailable through tufa formation. This seems correct as CV_Rest2 is now the most biodiverse of all restoration plots.

4.3.3 Minerals

Calcium concentration was high in all plots with an overall mean of 170.5mg/L, confirming source of strongly base-rich groundwaters. This is more than twice the mean of 66.8mg/L recorded by Snowdon in 2017 at the SAC designated Parsonage Moor, where calcium levels have fallen by more than half since 1975 and eutrophication from high nitrate levels resulted in a negative shift from M13 to less diverse tall herb communities (2017). Rozbrojova, and Hajek 2008 state that concentrations of 100mg/L are extremely high and important in maintaining low productivity communities. Harte (2020) and Higson (2020) also found substantially lower mean Calcium concentrations in the LV_Rest1/LV_Rest2 and LV_Benchmark plots at 58.0 mg/L and 52.8mg/L respectively, compared to 159mg/L and 137mg/L recorded in this study. Snowdon and Harte/Higson took 4 and 8 Calcium samples respectively as opposed to only 1 in this study which might explain the difference. However, levels of Sodium recorded by Snowdon (2017), Harte (2020) and Higson (2020) are very similar, so it does seem likely that this study detected genuinely higher Calcium concentrations. Whether this would translate until stronger protection from eutrophication than at Parsonage Moor by limitation of Phosphate due to the abundant calcium is unknown.

The CV_Rest2 plot has a strong spring, forming a very substantial tufa deposit extending down the valley slope several metres wide at the base and at least 0.25m thick (Figure 4.11). Hájek *et al.,* (2002) stated that a mean calcium concentration of 231mg/L in upwelling groundwater is needed for tufa to form in fens, with CV_Rest2 and CV_Ctl2 having calcium approaching these levels. Tufa-rich springs were features of all the study sites but especially large and frequent throughout Chilswell Valley, suggesting clean, calcium-rich groundwater. This may explain why the restoration plots had the highest biodiversity, as the high calcium levels limit phosphate and buffer against eutrophication from nitrates (Boyer and Wheeler 1989, Rozbrojova, and Hajek 2008). CV_Ctl2 has the highest Calcium concentration of all plots might best be targeted next for restoration, having the ability to limit phosphate and thereby make the slightly elevated levels of nitrate in the plot less problematic.

The CV_Rest2 plot also recorded by far the highest concentration of Iron. This was still just one-third the 690mg/L threshold at which Iron was found by Emsens *et al.,* (2017) to trap Phosphate and make it available for plant growth with negative consequences for rare low nutrient plant communities; however they report that the characteristics of Alkaline fens limit Phosphate plant availability regardless of Iron content. Iron is also less toxic to wetland plants with sedges and other monocotyledons being more tolerant (Rozbrojova, Z. and Hajek, M. (2008).

Figure 4.11 Calcium-rich tufa spring in CV_Rest2

The positive contribution of Potassium to species richness by modelling is unclear but could be related to concentrations being higher in the Raleigh Park restoration plots which had high levels of species diversity. Levels of Potassium were low in the other plots, but the overall mean for all sites matched that for Parsonage Moor (Snowdon, 2017).

4.4 Wild Oxford - Ecosystem Restoration Success

The uplift in biodiversity achieved by the Wild Oxford is substantial and statistically significant. More broadly, Wild Oxford has achieved the broader principles of ecosystem restoration as defined by and Gann *et al.,* (2019) through:

- altering physical biological attributes, by raising water table through removing Willow carr, damming drainage channels and re-introducing species;
- proactive stakeholder engagement, through working with local communities and “Friends of” groups to ensure the wishes of locals are identified and incorporated into the restoration activities;
- using a range of ecological knowledge employed in novel ways, as demonstrated by the use of *Pedicularis palustris* as an ecosystem engineer to parasitise *Phragmites australis*, reducing its dominance and creating the conditions necessary for the seedlings of target fen species to establish

(Decleer 2013, Webb 2020b) and targeting the correct wetland species suited to alkaline, low nutrient, mineral rich peat; and

- perhaps most importantly, restoring natural processes either through the direct browsing of large herbivores or their simulation by the cutting and removal of biomass (Middleton, 2006b)

4.4.1 Learning points for further research

In hindsight a better experimental design would be to simultaneously collect baseline data on the plant community, floral units, nectar resource, flying insects and abiotic variables from the restoration and control plots prior to restoration activities. This would have provided certainty that the restoration plots were not already in a more favourable ecological condition or higher than the controls. In practice, conservation work has very limited budgets and in-depth monitoring may lose out in competition for funding.

The presence of Judy Webb's data from 2017 provides significant reassurance that the findings are robust, albeit some discrepancy in that the first surveys at Chilswell were 3 years after restoration began and there is no data from before the Willow Carr was removed from Raleigh Park, which marked the true start of the project. There are practical reasons for this including time, resources and the fact that some plots were virtually inaccessible due to the density of reed and willow scrub, whilst RP_Rest3 was impenetrable bramble scrub (J Webb, personal communication, 17 August 2021).

Table 4.5 summarises potential for improving the methodology and further research that could be undertaken to strengthen the objective evaluation of ecological restoration in Valley-head Alkaline fens.

Table 4.5: Recommended adjustments to the methodology and possible future research for evaluating Alkaline fen ecological restoration techniques

Adjustment to methodology	Purpose	Effort
Use the NVC survey method to sample the plant community	Improve accuracy of matching to NVC plant communities	Low
Test a wider range of restoration techniques, including plug planting of species difficult to establish from seed and that mainly propagate vegetatively, a characteristic of some plant groups like Sedges that are sensitive to irregular seasonal flood pulses (Hall *et al.,* 2010).	This technique is a new addition to the restoration activities with *Menyanthes trifoliata* and *Eriophorum angustifolium* planted in Lye Valley spring 2021 and *Blysmus compressus* in Raleigh Park, summer 2021 (J Webb, personal communication 23 June 2021)	Low
Sampling nitrates and phosphates monthly and undertaking laboratory analysis in-house using the Brookes facilities	Provide a more detailed understanding of nutrient conditions in the plots at much better value for money.	Medium
Further research	**Purpose**	**Effort**
Use the propagules, plant matter and pollen preserved within the peat to try to reconstruct the plant community of sites before they were degraded.	Use the records of plant community to influence the target species assemblages for future ecosystem restoration efforts, particularly relevant in the case of species re-introduction	High
• Repeat sampling of the plant community in all plots every 2 years over a long-term timeframe. • Maintain weekly WTD measurements in dipwells	• Establish a robust time-series record of the plant community and track any further changes e.g. reducing or ceasing restoration • Determine the changes in WTD as a result of climate variation	High
• Continue monitoring WTD RP_Rest3 for 5-10 years • Sample the plant community in RP_Rest3 every 2 years	• Determine the long-term effect of rewetting restoration activities on WTD in RP_Rest3 • Evaluate the effect of re-wetting on a severely degraded fen	High
Direct sampling of invertebrates through sweep netting and malaise traps in each plot	Understand the utility of the control and restoration plots by the full range of invertebrate groups, not just flying insect pollinators	High
Study the quantitative effects over a longer time-frame of *Pedicularis palustris* on species richness, germination and nursery conditions influencing successful establishment of target Alkaline fen plant community and influence of abiotic factors.	Specific detail of the effects on Alkaline fens to build upon the qualitative study of Webb (2020) would be invaluable resource for effectively directing future restoration	Medium
Determine real-world nectar and pollen values by collecting samples in the field and using laboratory analysis for species missing from the Agriland databased (Baude *et al.,* 2016) and develop a new database for pollen values.	• Provide greater accuracy and precision to the estimation of nectar provision by Alkaline fens. • Incorporate estimates of pollen provision into the assessment of value for invertebrates	High

5. Appendices

Appendix 1: History of Ecological Restoration at the Study Sites

Plot	Management history
CV_Ctl_1	None
CV_Ctl_2_20	None
CV_Ctl_2_21	Cut Winter 2020 then May & July 2021
CV_Rest_1_E_21	First cut 30 August 2014, then 3 times a year
CV_Rest_2_W_21	1 dumpy bag brown hay added October 2017, hand collected seed (from LV Ecobench plot) of: DBS, greater bird's foot trefoil, purple loosestrife, tufted vetch, parsley water dropwort, meadow sweet, marsh lousewort, yellow loosestrife & common valerian added once a year
LV_Ctl	None
LV_Rest_1(A)	First cut Jan 2017, then July & Oct 2017, 3 times a year 2018 then 4 from 2019 onwards. Brown Hay added Oct 2018, 2019 and 2020. From 2017 annually hand collected seed of DBS, greater bird's foot trefoil, tufted vetch, parsley water dropwort, meadow sweet, marsh lousewort, yellow loosestrife & common valerian
LV_Rest_2(S)	First cut July 2016, then Oct 2016, then three reed cuts annually May, July and Oct 2017 and 2018 (surveyed 18 Oct before last cut). 2019, 2020 & 2021 additional cut Feb-April. No green hay or seed applied
LV_EcoBench	Annually since Oct 1990 with reciprocating mower, plus summer targeting of reed with shears 2018, 2019 & 2020. Brown hay and hand collected seed donor site.
RP_Ctl_1 (T)	Fenced, no access for cows, no management other than pulling Impatiens gladulifera
RP_Ctl_2 (N)	No management, but grazed annually during for 3 months summer since 2016, 2021 for at least 6 months
RP_Rest_1(T)	Grazing as above, Nov 2016 scrub removal, tree removal and first veg cut Oct 2017. Then two cuts (winter/spring) 2018, 2019, none in 2020 due to pandemic, partially cut in 2022
RP_Rest_2(P)	Grazing as above, Nov 2016 scrub removal, tree removal and first veg cut Oct 2017. Then two cuts (winter/spring) 2018, 2019, none in 2020 due to pandemic, partially cut in 2024
RP_Rest_3(C)	Initial scrub clearance Nov 2017 and first veg cut May 2018, Oct 2018. Then twice during 2019, no cuts 2020 due to pandemic. One veg cut, full scrub removal and drainage ditch blocked Jan 2021
RP_Rest_4(B)	Scrub clearance and first cut Oct 2018. Cut with Rytec 2019, 2020. One veg cut, full scrub removal and drainage ditch blocked Feb 2021. Small amounts of hand collected Marsh Lousewort, Yellow Rattle, DBS and some GoP on half plot in 2019 and 2020

Provided by Judy Webb 17 August 2021

Chilswell Valley

First fen restoration cutting and raking was on fen below boardwalk on 30.08.2014. I remember only reed and greater bindweed as the flora present at that time.

Monitored squares - One square (west) was planned to have additions, the other (east) just cutting and raking

Since 2014 then there have been 2-3 reed/veg cuts a year (if three, then May July & Oct , if two then June & Oct) by Wild Ox vols and occasionally OCC Thursday vols. At start cut reed burned on site in metal cradle, ashes removed, later reed merely piled at slope bottom. Subsequently this old piled reed used in damming/re-wetting restoration.

One builders bag of hay from the LVNF east side (your benchmark plot) added in Oct 2017 ONLY to the westernmost square (only ONE hay addition event)

From 2017 every year, hand collected seed (from LV) of: DBS, greater bird's foot trefoil, purple loosestrife, tufted vetch, parsley water dropwort, meadow sweet, marsh lousewort, yellow loosestrife & common valerian added ONLY to the westernmost square. The yellow loosestrife never showed up as a plant (a subject of my interest as to why as germinates at home in compost in pots well but has not germinated from seed spread in any fen restoration yet) rest of introduced seed successfully established in vegetation.

Both squares have had hand pulling/digging of: pendulous sedge, Michaelmas daisy, willow seedlings (vast numbers) plus I have pulled greater plantain when recording squares.

Lye Valley North Fen

More complicated story:

West bank was reed dominated with greater bindweed (and nettle at south end in Arson area) until 2011. Before this historically the OCC used to cut a system of firebreaks through the reed middle (to limit spread of fires, arson, see attached aerials showing burns - harvested from the web) until they ceased having manpower/time to do this – this cutting actually preserved some biodiversity in the top section of sewer hatch square, Juncus subnodulosus benefitted and survived here so did not need to come back from seed bank). Aerial photo shows the fire breaks as green in dead reed – rush preserved here. Note rush stays green and can grow all winter, reed effectively disappears all winter as visible live plant, new shoots only emerge late April/beginning of May. So there was a lot of dead combustible reed stems and thatch all through April, during Easter hols and that is mostly when burns happened. 'Arson square' Burnt to my knowledge in 2006 and again in 2014. Before that countryside officer Anthony Roberts told me it was an almost annual occurrence on that west bank, never on east fen.

Cutting and raking started on the West bank in a small area at the north end SSSI just down from Heath Close in 2011/2012 with Carl Whitehead and the Thursday

OCC vols. I remember ash felling first scything and raking off dead reed in winter. **BUT this did not go much south and did not reach your squares**!

Sewer hatch square cutting and raking started with Wild Oxford & OCV vols and FoLV in July 2016. See photos attached – you can see what the vegetation was like before any work in the May photo and as the OCV young woman is felling it later in July.

This area then continued with cutting and raking with various cutting methods - OCC reciprocating mower used by Carl or various volunteers scything – summer cutting of green reed started as well as autumn dead reed cutting. Shading grey & crack willows coppiced. From 2017 probably three reed cuts a year – May, July and Oct. I set up my recording squares in 2017.

Cutting/raking reed progressed southwards on the west side slowly and we did not reach the **Arson square** area until 2017, when first cutting of the old dense dead reed and nettle started in **Jan 2017**. It was awful heavy hard work – photo attached shows Rich scything in that burned area in Jan 2017. Then it had two more cuts that year – Jul and Oct, so the three cuts a year in place from then.

For the last three years, 2019, 2020&2021 **both Arson Square and Sewer hatch square** have been **mown by Rod** with the Cobra mower at first reed emergence date – from end Feb through April and in 2020 he did a second mow in early May. All in order to knock reed back as soon as it gets up to around 40cm tall. This gets it down before it is big enough to scythe and he can cover a big area in a short time.

Benchmark square east side fen is currently normally only cut/raked once in October (OCC with reciprocating mower plus OCV & FoLV rakers) BUT for the last 3 years, FoLV vols have done a July cut and rake of the Lye brookside strip and the base of the bank to Peat Moors strip (i.e. either side of benchmark square). This has removed rank reed vegetation midsummer. Then an all over cut of the East side fen with reciprocating mower happens in October. In addition, for the last three years I have sent volunteers walking through the east side fen middle with shears, in midsummer, chopping off reed as low as they can from late June- July-Aug. Targeted removal of reed by hitting it at its most vulnerable time, whilst preserving other flowers. That is why there is hardly any reed now in the really good short middle section of your benchmark square. When the OCC first started reciprocating mower reed cutting in October with risings removed on this east side in late 1980s-early 1990s it was heavily reed dominated all over the east side. This work started only just in time to prevent so many species disappearing.

Arson Square additions: From 2017 **every year**, hand collected seed of: DBS, greater bird's foot trefoil, tufted vetch, parsley water dropwort, meadow sweet, marsh lousewort, yellow loosestrife & common valerian added ONLY to the Arson square, not to Sewer hatch square. Brown hay from centre east side N fen AND from Centre South Fen, added to ONLY the Arson Square in Oct 2018. This was repeated in 2019 and 2020 and will happen again in 2021. Can't see a lot of evidence of the success of all this hay transfer – no doubt due to the long history of damage to this square and the nutrient enrichment from contaminated springs at the top of the slope.

Raleigh Park

Add cow grazing to all below, but it would take more time than I have to work out when and how long each time:

Restoration work started with Wild Oxford vols in **top fen above ponds** in 2016. For the first year it was scrub and tree removal, didn't get on to cutting and raking until 2017 in the top fen below the barbed wire fence at first.

OCC removed some willows in the fen and around the ponds in 2017. Since then I estimate it has averaged only two cuts a year with risings removed. The Rytec has tried to get in to cut the north side of this fen but regularly gets stuck.

Fen below causeway - Scrub clearance started in 2017. Cutting and raking operated from May 2018. Since then cut/rake twice a year, roughly. Log damming of central drain started in a minor way by vols with Carl 2019. the most recent cut and rake was from Rod with brushcutter, before he could start major log damming and re-wetting in spring 2020..

Bridleway fen - Marsh lousewort & yellow rattle & DBS seeds introduced fen in a small way in 2019 at the lower end (source LV). Also added in late 2019 here were purple loosestrife seeds and the dregs from my bag of grass of Parnassus seed that I had been spreading in Lye Valley that autumn. More marsh lousewort seed from LV added here in 2020. NO hay from LV. Bridleway fen can be cut and collected by the Rytec, scything does not usually happen because of this.

Ash tree fen – Has had scything and raking probably since 2017. In Jan 2021 I spread some more yellow rattle seed from Milham ford plus tufted vetch seed from LV in this area.

<u>Removals</u>: Pendulous sedge dug out of a fen below causeway, but it has come back from seed bank with a vengeance. I have pulled young willows and alders from all fens at every opportunity plus have pulled and encouraged other vols to pull Typha latifolia greater reedmace seedlings, mainly in top fen below fence.

<u>Note The restoration of all three sites has been bedevilled by the occurrence of :</u>

1) unwanted species returning from the seed bank and needing removal, so have been pulled/dug from squares: pendulous sedge, bittersweet, bramble & hard rush the main culprits (vast bank of dormant long-lived seed in peat, activated by disturbance) although water figwort was a bit of a seed bank return problem at one point, it had a flush of return then dominance reduced. Rough stalked meadow grass and creeping bent may also have long lived seed and these have arisen and taken over in some areas and very difficult to remove.

2) unwanted species arriving via windblown seed and needing removal, so have been pulled/dug from squares: willows, ash, sycamore, Norway maple, Buddleia, birch, creeping thistle, Michaelmas daisy, Canadian fleabane, ragwort…

Green / Brown Hay

In your project squares. No green hay from Lye Valley used at all – only old brown October hay (there is a difference in seed amount contained, Oct hay contains much less seed than if it had been specifically harvested in July, green, with more seed). Brown hay is a waste product and would other wise have been dumped on brook banks and used in bankside restoration where it will form new peat. Green hay is usually specifically collected for seed spreading purposes.

No brown LV hay at all put at Raleigh Park, only one load of LV brown hay at Chilswell and several applications of brown hay at Lye Valley from east fen on the Arson square only. But much more to say.

RP has had very limited spreading of hand collected seed and 90% on the bridleway fen only.

Appendix 2: Species Nectar Sugar Value Database

Supplied as a separate excel spreadsheet

Appendix 3: Dipwell Core Stratigraphy

Supplied as a separate pdf file

Appendix 4: Plant Community, floral unit and nectar resource raw data

Supplied as 3 separate excel files for Chilswell Valley, Lye Valley and Raleigh Park

Appendix 5: Full MAVIS NVC match

CHILSWELL VALLEY	Description	% Match
CV Ctl 1 (East)	S4a Phragmites australis swamp and reed-beds, Phragmites australis sub-community	53.0
	S4 Phragmites australis swamp and reed-beds	50.6
	S26b Phragmites australis-Urtica dioica tall-herb fen, Arrhenatherum elatius sub-community	39.9
CV Ctl 2 (West) - 2020	S26 Phragmites australis-Urtica dioica tall-herb fen	48.2
	S26d Phragmites australis-Urtica dioica tall-herb fen, Epilobium hirsutum sub-community	46.1
	OV26 Epilobium hirsutum community	45.0
CV Ctl 2 (West)	S26 Phragmites australis-Urtica dioica tall-herb fen	48.5
	OV26 Epilobium hirsutum community	47.7
	OV26b Epilobium hirsutum community, Phragmites australis-Iris pseudacorus sub-community	47.2
CV Rest 1 (East) 2017	OV26 Epilobium hirsutum community	38.2
	S26 Phragmites australis-Urtica dioica tall-herb fen	37.5
	S4 Phragmites australis swamp and reed-beds	35.0
CV Rest 1 (East)	OV26 Epilobium hirsutum community	41.6
	M22a Juncus subnodulosus-Cirsium palustre fen-meadow, typical sub-community	40.7
	OV26b Epilobium hirsutum community, Phragmites australis-Iris pseudacorus sub-community	39.5
CV Rest 2 (West) 2017	S26d Phragmites australis-Urtica dioica tall-herb fen, Epilobium hirsutum sub-community	37.4
	S26 Phragmites australis-Urtica dioica tall-herb fen	36.9
	S4 Phragmites australis swamp and reed-beds	36.2
CV Rest 2 (West)	M22a Juncus subnodulosus-Cirsium palustre fen-meadow, typical sub-community	50.7
	M22 Juncus subnodulosus-Cirsium palustre fen-meadow	46.2
	SD15d Salix repens-Calliergon cuspidatum dune-slack community, Holcus lanatus-Angelica sylvestris sub-community	42.5

LYE VALLEY	Description	% Match
LV Ctl Plot	OV26 Epilobium hirsutum community	47.4
	S26 Phragmites australis-Urtica dioica tall-herb fen	44.6
	M27b Filipendula ulmaria-Angelica sylvestris mire, Urtica dioica-Vicia cracca sub-community	43.3
LV Eco Bench	M13 Schoenus nigricans-Juncus subnodulosus mire	52.6
	M13a Schoenus nigricans-Juncus subnodulosus mire, Festuca rubra-Juncus acutiflorus sub-community	51.0
	M13c Schoenus nigricans-Juncus subnodulosus mire, Caltha palustris-Galium uliginosum sub-community	49.3
LV Rest 1 2017	S26 Phragmites australis-Urtica dioica tall-herb fen	43.9
	S4 Phragmites australis swamp and reed-beds	39.9
	OV26 Epilobium hirsutum community	39.8
LV Rest 1	S4 Phragmites australis swamp and reed-beds	45.9
	OV26 Epilobium hirsutum community	45.2
	S25a Phragmites australis-Eupatorium cannabinum tall-herb fen, Phragmites australis sub-community	44.9
LV Rest 2 2017	S4 Phragmites australis swamp and reed-beds	50.0
	S26 Phragmites australis-Urtica dioica tall-herb fen	47.0
	OV26 Epilobium hirsutum community	45.8
LV Rest 2	M22a Juncus subnodulosus-Cirsium palustre fen-meadow, typical sub-community	47.0
	S25a Phragmites australis-Eupatorium cannabinum tall-herb fen, Phragmites australis sub-community	43.5
	S25 Phragmites australis-Eupatorium cannabinum tall-herb fen	43.1

RALEIGH PARK	Description	% Match
RP Ctl 1	OV26 Epilobium hirsutum community	39.6
	S26d Phragmites australis-Urtica dioica tall-herb fen, Epilobium hirsutum sub-community	35.1
	OV26a Epilobium hirsutum community, Juncus effusus-Ranunculus repens sub-community	33.9
RP Ctl 2	OV26 Epilobium hirsutum community	51.2
	OV26a Epilobium hirsutum community, Juncus effusus-Ranunculus repens sub-community	45.0
	S26 Phragmites australis-Urtica dioica tall-herb fen	40.5
RP Rest 1 2017	OV26 Epilobium hirsutum community	42.7
	OV26a Epilobium hirsutum community, Juncus effusus-Ranunculus repens sub-community	39.0
	M22a Juncus subnodulosus-Cirsium palustre fen-meadow, typical sub-community	35.1
RP Rest 1	M22a Juncus subnodulosus-Cirsium palustre fen-meadow, typical sub-community	41.4
	OV26 Epilobium hirsutum community	40.3
	OV26a Epilobium hirsutum community, Juncus effusus-Ranunculus repens sub-community	38.6
RP Rest 2 2017	OV26 Epilobium hirsutum community	46.3
	OV26a Epilobium hirsutum community, Juncus effusus-Ranunculus repens sub-community	42.5
	S18 Carex otrubae swamp	39.7
RP Rest 2	M22a Juncus subnodulosus-Cirsium palustre fen-meadow, typical sub-community	44.3
	M22 Juncus subnodulosus-Cirsium palustre fen-meadow	41.4
	OV26 Epilobium hirsutum community	40.9
RP Rest 3	OV26 Epilobium hirsutum community	46.3
	M22a Juncus subnodulosus-Cirsium palustre fen-meadow, typical sub-community	45.4
	OV26a Epilobium hirsutum community, Juncus effusus-Ranunculus repens sub-community	44.5
RP Rest 4	M22a Juncus subnodulosus-Cirsium palustre fen-meadow, typical sub-community	51.7
	OV26c Epilobium hirsutum community, Filipendula ulmaria-Angelica sylvestris sub-community	49.1
	M22 Juncus subnodulosus-Cirsium palustre fen-meadow	48.6

Appendix 6: FIT Count raw data

Supplied as a separate excel spreadsheet

Appendix 7: Dipwell Manual Readings raw data

Supplied as a separate excel spreadsheet

Data logger raw data very large – available upon request

Appendix 8: Conductivity raw data

Supplied as a separate excel spreadsheet

Appendix 9: pH raw data

Supplied as a separate excel spreadsheet

Appendix 10a: Peat depth summary by plot

CHILSWELL VALLEY	Peat Depth (cm)			
	Average	Median	Minimum	Maximum
CV Control 1 - East	67	68	45	93
CV Control 2 - West	83	93	40	100
CV Rest 1 - East	88	90	65	100
CV Rest 2 - West	72	80	35	90
OVERALL OF 4 PLOTS	**78**	**80**	**35**	**100**

LYE VALLEY	Peat Depth (cm)			
	Average	Median	Minimum	Maximum
LV Ctrl	62	65	30	85
LV Rest 1 West (Arson)	61	60	45	100
LV Rest 2 East (Sewer)	84	87	40	100
LV Eco-bench	76	80	55	100
OVERALL OF 4 PLOTS	**71**	**74**	**30**	**100**

RALEIGH PARK	Peat Depth (cm)			
	Average	Median	Minimum	Maximum
RP Ctl 1	32	33	13	45
RP Ctl 2	25	25	10	40
RP Rest 1	29	30	5	60
RP Rest 2	44	45	20	70
RP Rest 3	5	0	0	15
RP Rest 4	32	30	15	65
OVERALL OF 6 PLOTS	**28**	**30**	**5**	**70**

Appendix 10b: Peat depth raw data – Chilswell Valley

Supplied as a separate excel spreadsheet

Appendix 10c: Peat depth raw data – Lye Valley

Supplied as a separate excel spreadsheet

Appendix 10d: Peat depth raw data – Raleigh Park

Supplied as a separate excel spreadsheet

Appendix 11: R-Code Linear Regression Modelling / ANOVA species richness and restoration

Provided as a separate R Markdown file

Appendix 12: R-Code Linear Regression Modelling / ANOVA floral units and restoration

Provided as a separate R Markdown file

Appendix 13: R-Code Linear Regression Modelling / ANOVA floral units and restoration

Provided as a separate R Markdown file

Appendix 14: R-Code Linear Regression Modelling / ANOVA Flying insect pollinators and restoration

Provided as a separate R Markdown file

Appendix 15: R-Code Linear Regression Modelling / ANOVA Abiotic Variables and restoration

Provided as a separate R Markdown file

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