The short-term effects of wildfire on heathland flowering plant communities as a resource for insect pollinators: a case study at Killarney National Park, Ireland

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Abstract:

Fire is a globally important driver of ecosystem diversity and functioning, yet the frequency and behaviour of wildfire has become increasingly unpredictable as a result of climate change, posing risks to ecosystems, societies, and economies. Post-fire plant and animal community dynamics are often shaped by ecological, landscape, and fire characteristics, but as risk factors change, understanding community responses, rather than individual species, is critical for predicting post-fire trajectories and informing management.

Temperate heathlands are of cultural, ecological, and economic importance in Europe but face heightened wildfire risk due to changing land management and climate. In April 2021, a wildfire burned extensive areas of mixed heathland within Killarney National Park, Ireland. Here, we used a paired burnt–unburnt experimental design to quantify the effects of this event on floral community composition and the availability of food resources for pollinating insects in heathland habitats.

Results demonstrated a decrease in the number of insect pollinated flowers (floral units) during the first year post-fire, despite an overall floral unit increase driven by certain wind-pollinated species. However these effects became weaker two years post-fire, with increased compositional similarity between Fire and Control communities and an increase in insect-pollinated floral units, although responses varied seasonally. Some species occurred more frequently in Fire or Control sites, indicating distinct post-fire “winners” and “losers”. While there is evidence of post-fire floral recovery, proportional increases in certain fire-favouring species suggest potential shifts in community structure and resource availability for pollinators following wildfire. This has implications for the resilience of this landscape under altered fire-regimes and potential cascading effects on pollination services.

Keywords:

Post-fire recovery, floral resources, disturbance ecology, peatland, blanket bog, species richness, community ecology, ecosystem resilience

Introduction

Fire is an important driver of global biodiversity and ecosystem services (Scott et al. 2013). However its effects are often context or taxa dependant (Pastro et al. 2011), fire behaviour being influenced by a combination of climate (Wang et al., 2021), ignition sources (Syphard et al., 2025), and fuel characteristics (Hargrove et al., 2000). Certain fire characteristics can drive heterogeneous disturbance regimes, sustaining ‘healthy’ fire-prone ecosystems (Kelly & Brotons 2017), particularly when emulating historic patterns (Stephens et al., 2014; Lindenmayer et al., 2025). However in recent years changes in fire extent and behaviour have outpaced our understanding of wildfire across various landscapes and ecological functions (Kumar & Kumar, 2022; Carbone et al., 2025). Mid- to high-latitude regions are experiencing increasing fire occurrence (Miller et al., 2025) while European fire seasons have lengthened, with the annual burned area exceeding the 17-year average in each of the past five years (European Commission Joint Research Centre, 2025). Consequently there is growing concern that increased wildfire risk due to climate and land-use changes, could threaten ecosystems, economies and societies (DellaSala et al., 2022; Xu et al., 2020), while changes in fire frequency, severity, or distribution, may also reduce ecosystems resilience (Fernandez-Garcia et al., 2020; Banza et al., 2019; Schumann, 2020). Understanding a) the impacts on plant and animal species compositional changes and b) their post-fire interactions and recovery is key to mitigate these potential negative impacts.

Post-fire communities are heavily influenced by individuals that persist or escape disturbance. In plants, this involves on-site propagation and dispersal from off-site (Abella et al., 2009), with vegetation cover often re-establishing faster than composition (Abela, 2009; Kelly et al., 2023). Fire self-reinforcing processes can lead to pyrophyllic vegetation dominance, or pyrophobic vegetation can also benefit from habitat fragmentation (Gowda et al., 2019). Post-fire responses depend on pre-fire landscape composition (Day et al., 2017), climate (Chitale, 2019) and fire characteristics such as behaviour, severity and height (Rodríguez-Trejo et al., 2019), which in turn alter animal species distributions and resource availability (Fontaine et al., 2009; Griffiths and Brook, 2014; Nimmo et al., 2019). Anticipating the complex effects of fire is vital for effective landscape management strategies (Jolly et al., 2022; Tania Marisol González et al., 2022) and maintaining ecosystem functioning in affected areas. Plant–pollinator interactions represent a key example, as fire-driven changes in floral resource availability and habitat structural succession can influence pollinator activity and plant reproductive processes, yet these dynamics remain poorly integrated into empirical assessments of ecosystem functioning (Brown et al., 2016). Floral species and pollination are of particularly high priority for ecological conservation (Vanbergen, 2013) with 85% of wild flowering plants and 70% of cultivars relying on it for sexual reproduction (Artamendi et al., 2025). Consequently disruption to these interactions has implications for plant reproductive cycles and ecological community trajectories with knock-on effects for a range of important ecological processes, food security, plant and animal products (Breeze et al., 2011; Potts et al., 2016).

Increasingly unpredictable fire patterns across Europe, including 2022 and 2023 ranking among the five worst wildfire seasons since the beginning of the century (Miller et al., 2025), highlight the need for more robust quantitative understanding into responses to fire (Fernandez-Anez et al., 2021). Research largely focuses on fire-prone Mediterranean ecosystems, while wildfires in traditionally less fire-prone temperate conservation areas across northwest Europe remain less understood (Kelly et al., 2018; San-Miguel-Ayanz et al., 2022). However, Northern Europe's lengthening wildfire seasons are catalysing a shift in wildfire dynamics characterised by increasing fire occurrence, size and intensity in regions historically considered low-risk (European Commission Joint Research Centre, 2025; European Commission, 2026; Miller et al., 2025). Projections of hotter, drier summers in Britain and Ireland (Blenkinsop & Fowler, 2007; Fealy & Murphy, 2009), compounded by unique drivers such as vegetation type and moisture conditions, create increasingly fire-vulnerable systems (Albertson et al., 2010; Kelly et al., 2020). This is particularly significant given that these islands hold a disproportionately large share of global peat, and heathlands (Thom et al., 2016; Smith et al., 2023). Management changes in such habitats are already

driving dramatic declines and species compositional shifts (Shackelford et al., 2015; Davies et al., 2022), where severe fires can result in increased greenhouse gas emissions, biodiversity losses, and water quality degradation (Belcher et al., 2021; Xu et al., 2020; Belillas & Roda, 1993). As historical fire records suggest trends across Britain and Ireland are more closely aligned with each other than with broader global patterns (Hawthorne et al. 2018), understanding post-fire responses in these systems is critical for effective future management.

Heathlands are globally significant ecosystems (Thompson & Brown, 1992) with ecological and cultural importance dating back to Neolithic times (Harris et al., 2011). As nutrient-poor, open habitats dominated by low-growing evergreen sclerophyllous plants (Specht, 1979; Løvschal & Damgaard, 2022), they support species of high conservation value (Usher and Thompson, 1993; Thompson et al., 1995; Webb et al., 2010), key ecosystem functions including carbon storage (Fagundéz, 2013; Lunt et al., 2019), providing services for leisure, climate change mitigation and flood control (Holden et al., 2007; Harenda et al., 2018). Heathland characteristics vary with soil moisture, altitude, and exposure. In Ireland, they occur across a range of substrates, from sandy coastal systems to upland calcareous dry heath, but are frequently associated with peatland habitats, forming wet heath–bog mosaics across large areas of the landscape (Fossitt, 2000; Perrin, 2024; NPWS, 2025). Peatlands cover 20% of Ireland's land area, storing 53% of the national soil carbon stock, but the majority of these systems are now considered degraded or disturbed (O'Connell et al., 2014; Eaton et al., 2008). As semi-natural systems, heathlands are maintained in intermediate successional states by natural and anthropogenic disturbance (Damgaard et al., 2024), but drainage, land-use change, and management intensification are driving declines (Thompson et al., 1995; Turetsky et al., 2014; Garcia et al., 2013; De Graaf et al., 2009). Irish heathlands are no exception, with national declines reported since 2000, overall conservation status, structure and function all being assessed as "unfavourable-bad", and with fire identified as a key threat (NPWS, 2025). Losses are often compounded by altered community composition and increased nitrogen deposition, which favour grasses and mosses at the expense of heather regeneration (Fartmann et al., 2015). With the growing threat of wildfire, understanding how disturbance shapes these ecological communities is critical, as their dynamic nature can lead to complex responses to environmental change (Vandvik et al., 2005).

Periodic intervention is crucial for rejuvenating heathlands, conserving biodiversity (Garcia et al., 2013; Bullock and Pakeman, 1997), and preventing afforestation (Lisken-Kleinmans, 1998; Webb, 1998). In Atlantic heathland, biomass removal such as grazing, sod cutting, and prescribed burning (Walmsley et al., 2021) is traditionally aimed at promoting

heterogeneity (Velle et al., 2014; Gimingham, 1993), although context-dependent trade-offs exist, especially in nutrient-poor landscapes (Newton et al., 2009; Wilkie, 2007). Prescribed burning has sparked debate (Gorden et al. 2025; Lee et al., 2013), particularly over deep peat (Douglas et al., 2015), as potential benefits for grazers or managed game species such as grouse, which hold cultural and management value for some UK stakeholders (Martin et al., 2013; Welch, 1984) may not align with impacts on other species (Tharme et al., 2001) or ecosystem services (Robertson et al., 2017). Consequently, existing fire research often centres on peat-forming and game species (Kelly et al., 2018) in prescribed burns of heather moorland (Farage et al., 2009; Fielding et al., 2024). Yet fire impacts vary with vegetation age and structure (Davies et al., 2009; Brys et al., 2005; Noble et al., 2019), and there remains a limited understanding of post-fire landscape dynamics (Davies et al., 2010), including plant-animal interactions important for ecosystem functioning (Nicholson & Egan, 2020; Banza et al., 2021). In particular, comparatively little is understood about how post-fire responses in vegetation communities influence pollination networks, as well as the species diversity and functioning they support. With rising concerns over uncontrolled fire, addressing these gaps is essential to ensure the long-term sustainability and ecological value of these habitats.

Background and Aims

The aim of this study was to assess the ecological impacts of the large-scale 2021 wildfire in Killarney National Park, Ireland's oldest national park and a UNESCO Biosphere Reserve. Initial estimates suggested that 2,500–3,000 ha had been burned (Department of Housing, Local Government and Heritage, 2021). As one of the most extensive fires recorded in the region in recent decades this prompted a National Parks and Wildlife Service (NPWS) commissioned initiative to study the impact of the fire on biodiversity. Using a burned–unburned matched-pair experimental design in the two years immediately post-fire, we compared mixed heathland sites directly affected by the wildfire to unburned control plots of similar habitat type, relief, and altitude. This occurred across a mosaic of wet heath (HH3) and upland blanket bog (PB2) habitats (Fossitt, 2000), corresponding to Annex I habitat types "Northern Atlantic wet heaths with *Erica tetralix*" (4010) and "Active blanket bogs" (7130) under the EU Habitats Directive classification (NPWS, 2025).

We counted floral units for all species, but focused our analysis on insect-pollinated (entomophilous) plant species to assess the direct effects of fire on flowering plants and floral resource availability, as well as the indirect consequences for dependent flower visitors. Floral unit counts were used as both a rapid and functionally relevant measure of

plant response, providing a direct index of pre-zygotic reproductive effort, specifically plant investment in pollinator attraction and reward. By capturing variation in flowering phenology and productivity, floral units allowed this study to link post-fire vegetation recovery with potential ecosystem function through pollination networks and addressed the following hypotheses:

H1) Floral resource abundance and diversity will be initially reduced in burned sites relative to unburnt controls due to species-specific susceptibility to fire and variation in post-fire growth responses. We expect that trends will differ between entomophilous flora and the wider floral community due to differences in post-fire strategies, growth rates and faster flowering of certain faster growing anemophilous (wind pollinated) species.

H2) The effects of burning on floral resource availability will vary seasonally and between years, with initial reductions in burned sites followed by partial recovery over time, reflecting differences in species' flowering phenology and post-fire recovery trajectories. We predict that trends will differ between entomophilous flora and the wider floral community due to the slower recovery time and initial post-fire floral reduction of some slower growing, woody entomophilous (insect-pollinated) flora in the park, including heather species which can take several years to recover post-fire.

H3) Variation in entomophilous floral units will drive quantitative differences in nectar resource availability for flower visitors, with burned sites in pairs exhibiting initially reduced nectar availability due to slower recovery of key nectar producing heather species.

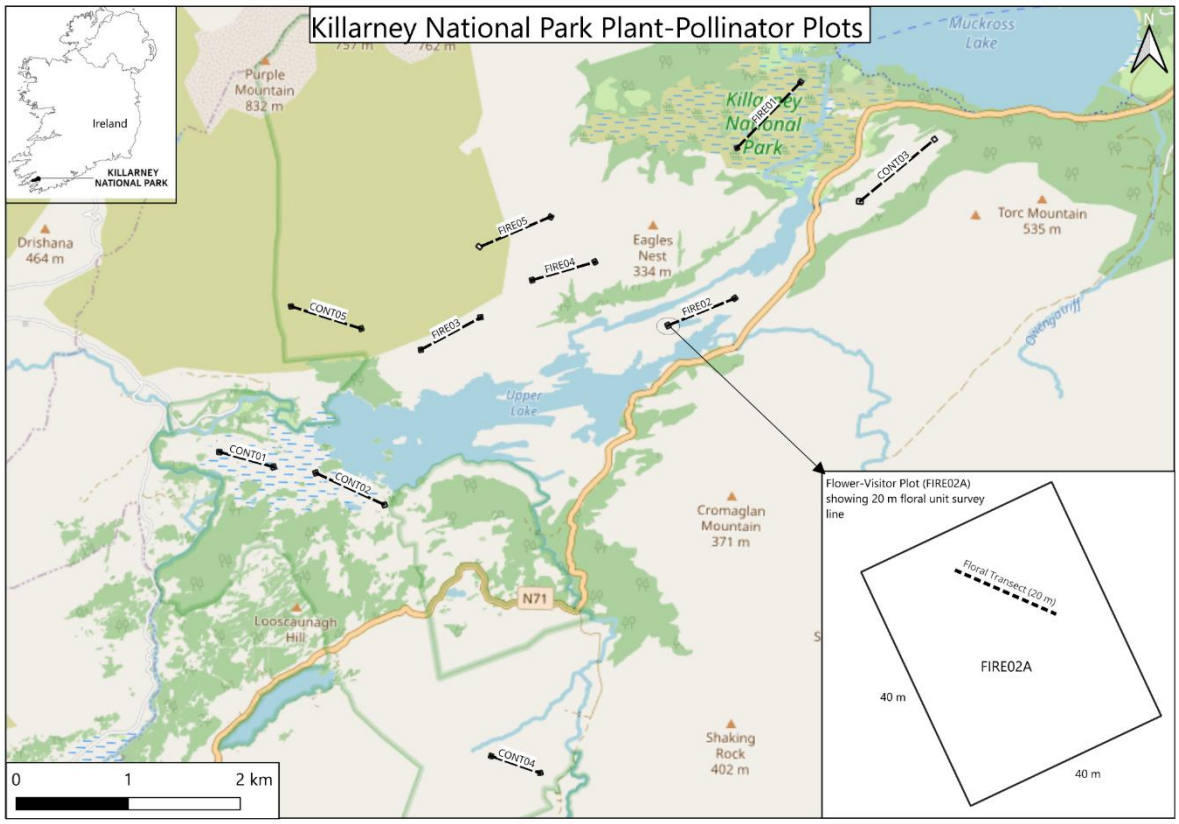
Materials and methods

Field site

The study was conducted in Killarney National Park, County Kerry, Ireland (52.02039° N, 9.58282° W), a 10,200Ha UNESCO Biosphere Reserve, EU Special Area of Conservation (SAC), EU Special Protection Area (SPA) and Ireland's oldest National Park, presented to the state in 1932 (Newman et al., 2014; NPWS, 2005, 2013, n.d.). Renowned for its high conservation value, the park supports a diverse range of habitats including peatlands, heathlands, lakes and the largest remaining tract of semi-natural woodland in Ireland (Hamilton et al., 2022). The park is shaped by the underlying Devonian sandstone and Carboniferous limestone geology (Kelly, 1981), combined with a temperate Atlantic climate, with cool summers, 15 °C July average and mild, wet winters, 6 °C February average (Whaley, 1985; Hawthorne et al., 2023). In April 2021, a predominantly heathland wildfire

burned approximately half one of the reserve's terrestrial habitats, while the last fire of this scale occurred in 1984 (Department of Housing, Local Government and Heritage, 2021; O'Sullivan, 1991), the blaze reflects a recent trend of increasing fire frequency and severity in the region (Perry et al., 2022; Carroll et al., 2021; Hawthorne et al., 2018).

To assess the effects of the wildfire on biodiversity, twenty 40 m² vegetation plots were established at either end of ten 1 km transects (established for bird surveys) located within Killarney's wet heath (HH3) and blanket bog (PB2) mosaics (Fossitt, 2000). Five ("Fire") transects, within areas burned during the April 2021 wildfire, were each paired with a corresponding unburned transect ("Control") of equivalent habitat type, altitude, slope (all <20°) and aspect. (Spatial information is provided in Map 1). Established during the Gittings (2021) post-fire bird report, the design provided ten Fire and ten matched Control plots, either end of five Fire and five Control transects. This paired natural-experiment approach enabled direct comparison of post-fire responses between treatments while accounting for landscape-scale environmental heterogeneity (Banza et al., 2021).



Map 1. Transect distribution and example floral transect within plots layout across Killarney National Park. For this study plots were placed at either end of both Control and Fire transects 1 – 5.

Floral resource surveys

To assess vegetation responses to wildfire and monitor floral resources across the flowering season, each 40 m² plot was surveyed three times per year: early summer (May), mid-summer (early July) and late summer (late August), over two years (2022 and 2023). To account for temporal and environmental variation, cloud cover and recent rainfall were recorded during each visit and time of day randomised. Surveys were conducted during dry daylight hours, avoiding high winds or rain to reduce potential effects on pollinator activity, as it was part of the wider study.

Ten 1 m² quadrats were placed every 2 m along a representative 20 m transect within each 40 m² plot (see Map 1). All open flowers were counted within each quadrat and specimens were identified in the field or collected for later identification. Floral units, in this case, an individual flower, or flowers within a single capitulum, being the distinct functional unit visited by a pollinator (Ollerton et al., 2007; Baldock et al., 2015; Faegri & Pijl, 1966) were recorded to quantify floral resource availability for insect flower visitors and assess plant reproductive responses to fire.

Nectar Variance Estimates

To assess whether variation in floral unit abundance drives quantitative differences in nectar availability for flower visitors we used measurements of total nectar sugar (µg) per flower per day and flowering duration (days) in the dataset of Baude et al. (2016) which documents plant traits, flowering phenology, flower density and nectar productivity at the flower scale for 260 common UK plant species. We focused on the dominant vegetation in our survey area, the heather species *Erica cinerea*, *Calluna vulgaris* and *Erica tetralix*. *E. cinerea* and *C. vulgaris*, being in the top four contributors to UK nectar provision (Baude et al. 2016). This focus allowed us to refine our estimates using empirical per-flower nectar data for these key taxa, while also investigating other species where empirical data was available. We then multiplied our floral unit counts with the available empirical data on species nectar sugar content (µg per flower per day) and flowering duration (days) to estimate total nectar production per species across their flowering period. Multiplying these figures by our recorded number of floral units per study treatment provided an estimate of nectar availability across species which accounted for the bulk of entomophilous floral units within our sites in each study year.

Statistical analysis

All analyses were performed in R 4.4.1 (R Core Team, 2024). We used generalised linear mixed-effects models (GLMMs) implemented in lme4 (Bates et al., 2015); a full package list can be found in Table S1 in the Supplementary Information. Data distributions were assessed visually and using diagnostic plots, and appropriate model families or transformations were applied where assumptions were not met. Count responses were modelled using Poisson or negative-binomial error distributions according to their mean–variance structure. Poisson models were used for specific seasonal analyses and species-richness counts when dispersion was acceptable, whereas negative-binomial models were used for pooled floral-unit abundance and other overdispersed count responses. Significance of fixed effects was evaluated using likelihood-ratio tests and single-term deletions, and post hoc comparisons were obtained using the emmeans package (Lenth & Piaskowski, 2026). Community composition was analysed using the vegan package (Oksanen et al., 2026).

Floral units

Floral-unit abundance was analysed using generalized linear mixed models (GLMMs). We primarily focused on the responses of entomophilous (insect-pollinated) species, modelling floral-unit counts using a Poisson error distribution appropriate for count data. Treatment (Fire vs. Control) and sampling season (Early, Mid, and Late summer, ordered by time since fire within each year) were included as fixed effects, and their interaction tested whether wildfire altered seasonal trajectories of floral availability. Site Pair was included as a random intercept to account for environmental variation between matched pairs. To evaluate temporal responses both within and between years, we also fitted a model that directly contrasted years. The same modelling framework was then applied to anemophilous (wind-pollinated) species; however, due to overdispersion in this dataset, these models were fitted using a negative binomial distribution.

Species diversity

Sampling completeness was assessed using species accumulation curves and incidence-based Chao2 richness estimation (Chao et al., 1984; Chiu et al., 2014) on sample season and year-pooled site × species presence–absence matrices. These analyses were used to estimate the number of potentially undetected species and evaluate whether sampling effort

was sufficient to characterise the entomophilous flora present within the study area. Species accumulation curves and richness estimators indicated that sampling was sufficient to characterise dominant patterns in community composition and floral resource availability, although some rare species may have remained undetected (see Figure S5, Supplementary Information).

Floral species richness was analysed using Poisson GLMMs with the same fixed and random effects structure as above. Shannon diversity and Pielou evenness metrics were then calculated for each Site. Variation in these was modelled using Gaussian mixed models with Treatment and Sampling season as fixed effects and Site as a random intercept. Models were fitted using REML, and residual structure was assessed using DHARMa (Hartig, 2022). Significance of fixed effects was evaluated using likelihood-ratio tests (drop1 in base R) and Wald chi-square tests using the Anova function in the car package (Weisberg & Fox, 2019). Interaction terms (Treatment × Sampling season) were used to test whether Fire and Control sites followed distinct temporal trajectories. Estimated marginal means were obtained using emmeans to assess treatment differences over time.

Community composition

To test the effects of wildfire on overall community composition, Bray–Curtis dissimilarities derived from site × species floral-unit matrices were calculated with floral units summed across sampling periods within each year. Treatment effects were tested using the vegan package (Oksanen et al., 2026) using Analysis of Similarity (ANOSIM) with permutations restricted within matched site pairs to reflect the paired sampling design. To assess whether differences detected by ANOSIM could be influenced by unequal within-treatment variability in community composition, PERMDISP was used to test whether this variability differed between Fire and Control pairs. Any species contributing to compositional differences between treatments were identified using SIMPER. Non-metric multidimensional scaling (NMDS) ordinations were produced using square-root transformed, Wisconsin-standardised data, and plotted (Figure S4, Supplementary Information) using ggplot2 (Wickham, 2016).

Plant–pollinator resource metrics

Total nectar sugar per species (Baude et al., 2016) was analysed separately for 2022 and 2023 using GLMMs with a Gamma distribution and log link. Treatment was included as a fixed effect and Flowering species as a random intercept to account for interspecific variation

in nectar production. Plots were paired during field sampling to reduce landscape-level heterogeneity between burned and unburned areas; however, pair identity was not included in the final models because nectar estimates were analysed at the treatment–species level and the small number of observations limited the complexity of the random-effects structure. Random-intercept and random-slope structures were compared, with the simpler retained where more complex models failed to converge.

Analysing combined floral species dataset

Analyses were repeated for the full floral dataset to evaluate whether post-fire entomophilous floral responses reflected broader floral trends, or whether there was variation in responses when considering anemophilous (wind pollinated) species with no nutritional resource for flower visitors (see Figure S6 - S11, Supplementary Information). The same modelling framework and diagnostic checks were repeated. When specifically assessing floral unit differences separate models were fitted for each sampling season across years, using a negative binomial error distribution to account for overdispersion in the dataset.

Results

Entomophilous Floral Unit Abundance

Fire altered the abundance of floral resources but also their seasonal occurrence across two years post-fire. Aggregated across seasons, floral unit abundance of entomophilous (insect pollinated) species differed strongly between years (see Figure 1). Fire sites produced substantially fewer floral units than controls in 2022 (GLMM: $z = -34.4$, $p < 0.001$). The magnitude of this difference declined but was still significant in 2023 (GLMM: $z = -8.6$, $p < 0.001$). A significant Treatment × Year interaction indicated differing post-fire responses between years ($\chi^2_1 = 745.6$, $p < 0.001$).

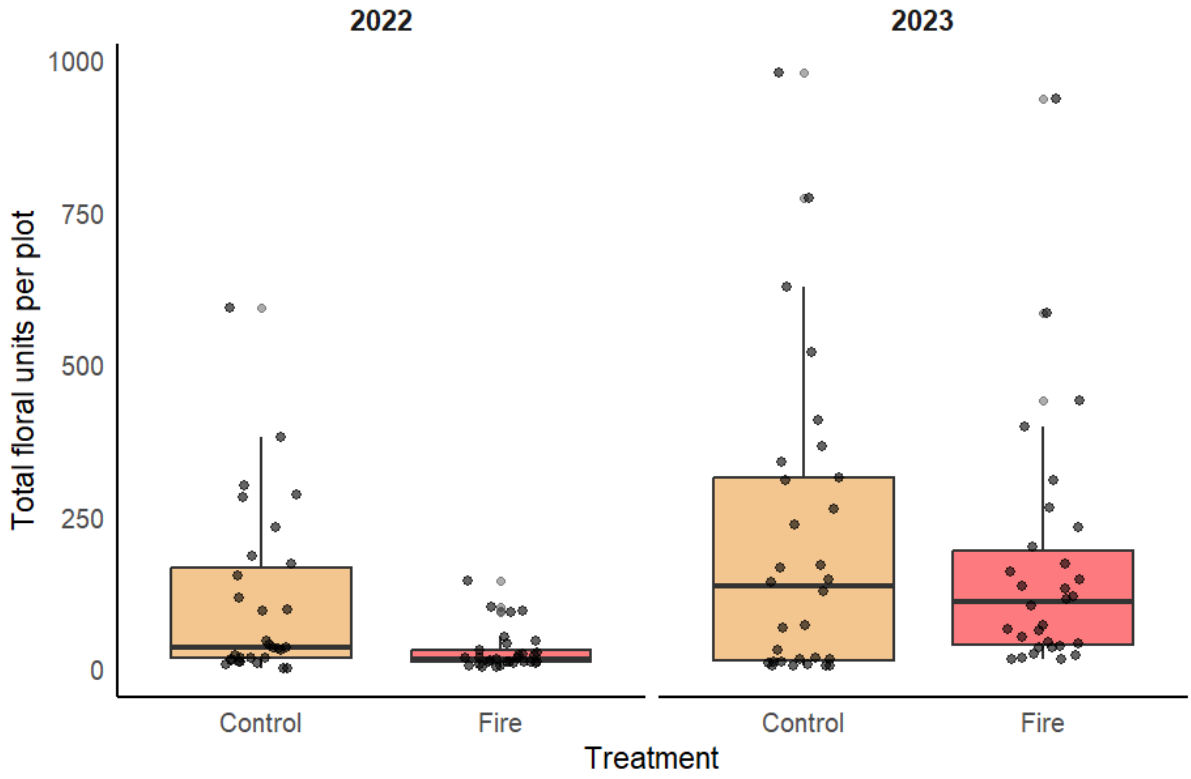


Figure 1. Distribution of total floral units per plot across all sampling seasons for Control and Fire treatments in 2022 and 2023. Each point represents one plot sample, illustrating the aggregated number of floral units recorded in each sample season across the entire year. Boxplots show the median and interquartile range; jittered points indicate variation between plots. Colours distinguish treatment groups.

Entomophilous floral-unit abundance also varied seasonally, differing in trajectory between treatments (GLMM: Treatment × Ordered Sample Season: $\chi^2_5 = 130.99$, $p < 0.001$). In the first post-fire sample season (early summer 2022), abundance on Fire plots did not differ significantly from Controls ($\beta = 0.27 \pm 0.17$ SE, $p = 0.11$), but by mid and late summer 2022, floral production was substantially lower in Fire plots (mid-summer: $z = -5.3$, $p < 0.001$; late-summer: $z = -4.8$, $p < 0.001$). In early summer 2023, floral abundance on Fire plots exceeded Controls ($z = 2.8$, $p = 0.005$), declined to similar levels in mid-summer ($z = 1.2$, $p = 0.22$), and was significantly lower by late-summer ($z = -3.1$, $p = 0.002$). Overall, floral units exhibited strong seasonal, non-linear trends post-fire, with substantial shifts in floral outputs between years (see Figure 2).

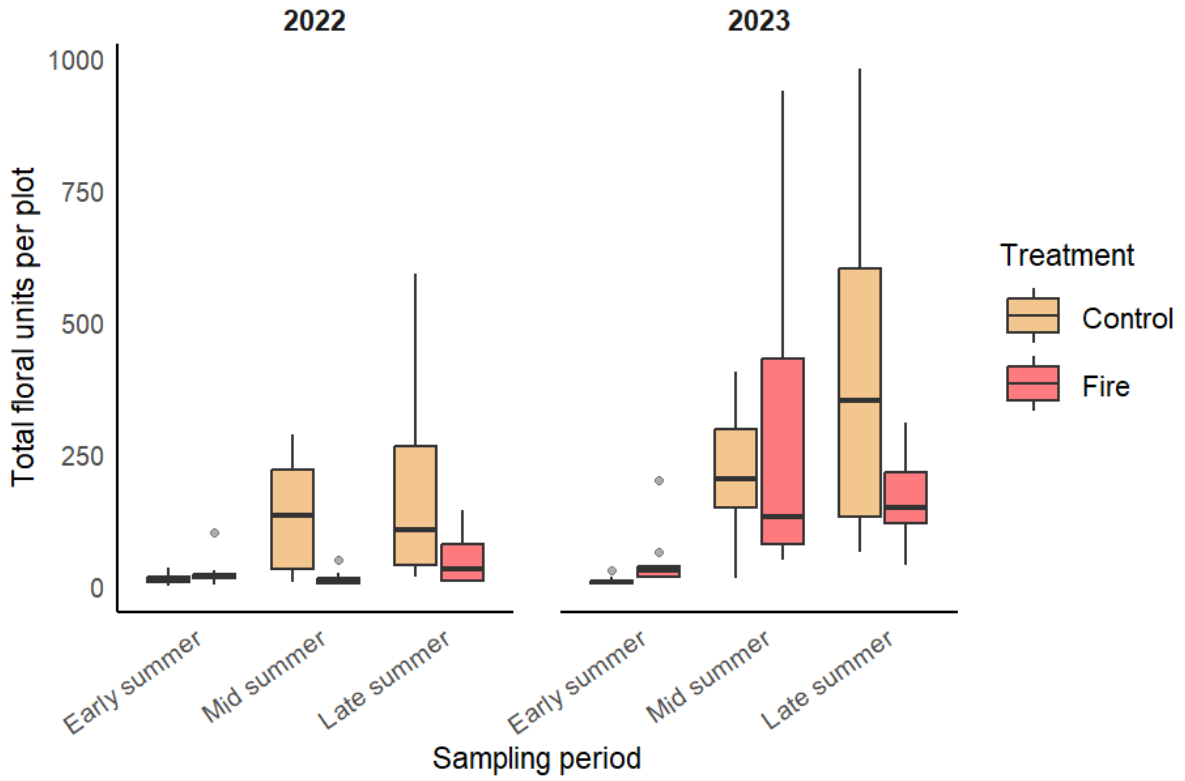


Figure 2. Plot level floral unit counts for Control and Fire treatments across Early, Mid, and Late summer sampling periods in 2022 and 2023. Boxplots show the distribution of floral units per plot within each sampling period, separately for each year. Jittered points indicate plot-level variation, colours distinguish treatment groups.

Entomophilous Flowering Species Diversity

Entomophilous species richness did not differ between Fire and Control plots (GLMM: Estimate = –0.00 ± 0.11 SE, z = 0.00, p = 1.00) and did not significantly vary across sampling periods (χ^2_5 = 5.64, p = 0.34), indicating that fire did not reduce the number of flowering species available to flower visitors (see Figure 3). Richness estimator Chao2 suggested that some locally rare species may have remained undetected (observed richness = 15 species; estimated richness = 24.5 ± 9.8 SE), although the species accumulation curve became increasingly shallow with additional sites surveyed (see Figure S5, Supplementary information), indicating that sampling captured the majority of entomophilous flowering species and that any unrecorded taxa were likely either rare, spatially restricted or had a particularly narrow flowering window. These results suggest that fire did not reduce entomophilous species richness during the two years following fire (see

Figure 3), despite changes in the abundance (see Figure 2) and composition (see Figure 4) of floral resources.

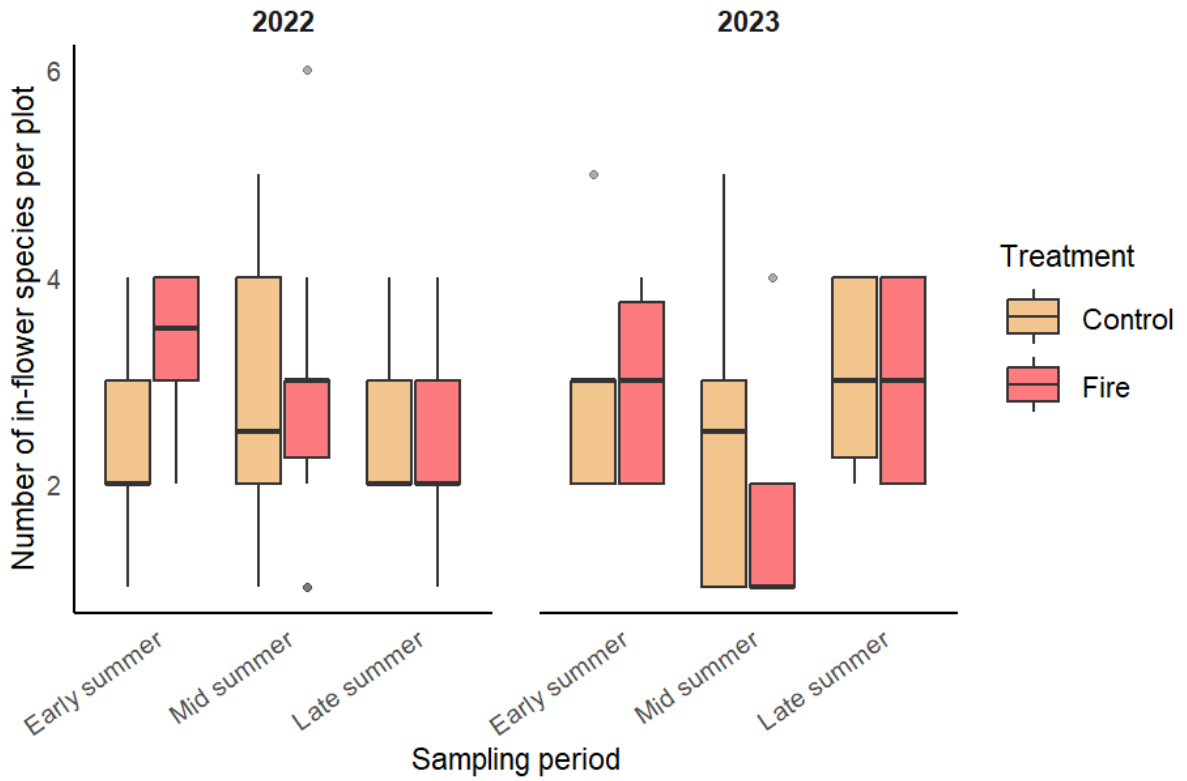


Figure 3. Seasonal patterns of the number of in-flower entomophilous species per plot in Control and Fire treatments across Early, Mid, and Late summer in 2022 and 2023. Boxplots show the median and interquartile range for the number of species in flower for each sampling period, with points representing individual treatment plots within each season. Years are delineated by plot panels to illustrate annual differences in flowering phenology and post-fire flowering trends. Colours differentiate Control and Fire treatments.

Entomophilous Community differences:

ANOSIM revealed significant compositional differences in entomophilous floral communities between Control and Fire plots in 2022 (ANOSIM: R = 0.353, p = 0.035), when permutations were restricted within matched pairs. SIMPER analysis suggested that a small number of species accounted for most of the observed dissimilarity, with the most influential taxa exhibiting strong fire-driven floral declines (see S2, Supplementary Information). *Calluna vulgaris* (Ling Heather) was the strongest contributor, accounting for 42.6% of total dissimilarity (166 Control vs 31 Fire units; SIMPER p = 0.085). *Erica tetralix* (Cross-leaved

Heath) contributed a further 36.4% (121 Control vs 27 Fire units; SIMPER p = 0.005). Several additional species were also reduced in Fire plots, including *Erica cinerea* (Bell Heather; 10.6 Control vs 0.1 Fire units; SIMPER p = 0.015) and *Potentilla erecta* (Tormentil; SIMPER p = 0.005). Altogether, the five strongest species contributors accounted for 94.2% of Control–Fire dissimilarity, indicating that early post-fire floral shifts were driven primarily by strong reductions in key heathland species.

In 2023, community differentiation between treatments remained significant but was weaker (ANOSIM: R = 0.126, p = 0.035), with the lower R statistic indicating greater compositional overlap compared to the previous year (see Figure 4). SIMPER analysis suggested that several of the same taxa continued to contribute to Control–Fire dissimilarity, although species responses became more variable. *Erica tetralix* (Cross-leaved Heath) became more florally abundant in Fire plots (256 Control vs 359 Fire units) and was the largest contributor to dissimilarity (SIMPER: 43.9%, p = 0.620), although its contribution varied substantially among sites. *Erica cinerea* (Bell Heather) remained less abundant in Fire plots (31 Control vs 1 Fire units; SIMPER p = 0.050). *Calluna vulgaris* (Ling Heather) also remained reduced in Fire plots (301 Control vs 76.6 Fire units; SIMPER p = 0.080), although its proportional contribution was smaller than in the previous year. Several forb species showed positive fire responses in the second year, including *Polygala serpyllifolia* (Milkwort; SIMPER p = 0.005) and *Pedicularis sylvatica* (Lousewort; SIMPER p = 0.040). Overall, several key taxa continued to contribute to compositional differences between treatments, but species-specific responses were more variable in direction and magnitude in 2023, alongside greater overall compositional similarity between Fire and Control plots. This pattern is suggestive of community reorganisation and early signs of post-fire recovery.

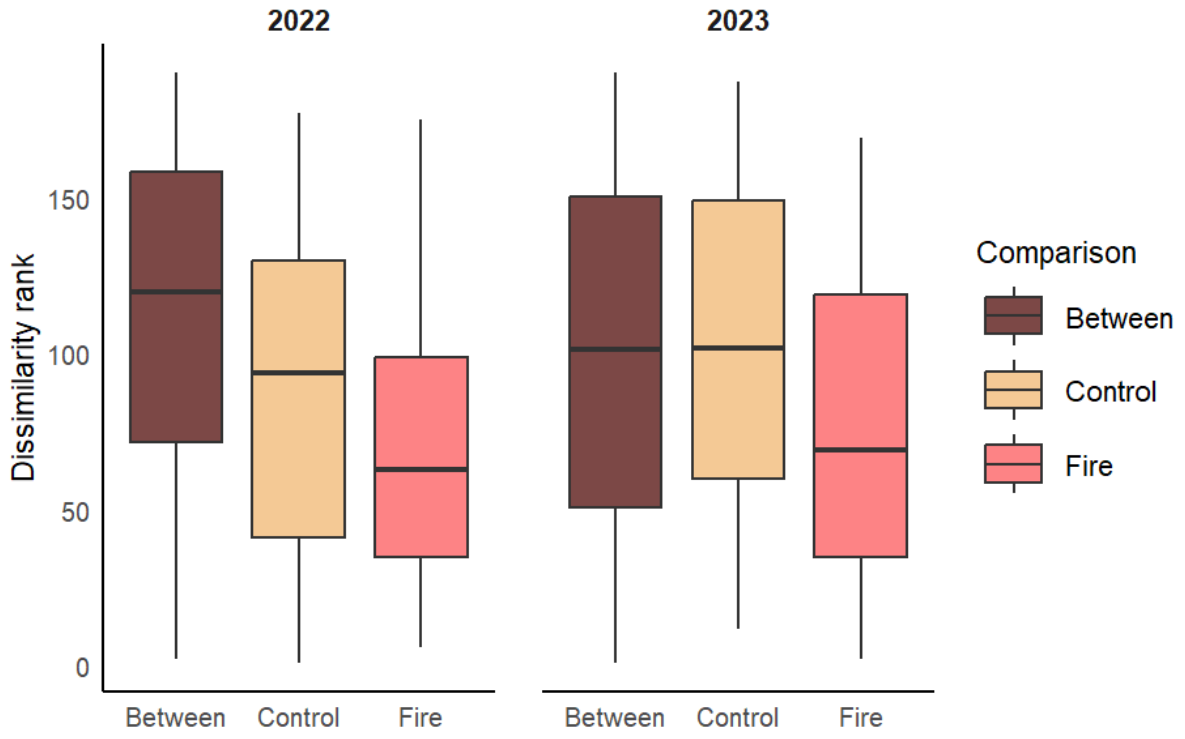


Figure 4. Boxplots show the distribution of ranked Bray–Curtis dissimilarities within and between treatments in each year. Higher ranks indicate greater compositional dissimilarity. In both panels, the boxes illustrate the interquartile range, horizontal lines show medians, and whiskers capture the overall spread of ranks, illustrating the magnitude and variability of compositional differences between treatments.

Plant–pollinator nectar resource

Following floral unit trends, wildfire significantly reduced nectar sugar availability in the first year after fire (see Figure 5), supported by dramatic nectar reduction across five key species which account for the bulk of nectar provisioning in this habitat (Baude et al., 2016). In 2022, Fire plots contained substantially less nectar sugar than unburned controls (GLMM; LRT: $\chi^2 = 14.42$, d.f. = 1, $p < 0.001$). Raw nectar sugar totals were markedly lower in Fire plots than Controls in 2022 (Figure 5), consistent with model-estimated reductions in nectar availability following wildfire. By 2023, nectar sugar availability remained lower in Fire plots, although the effect was weaker than in 2022 (GLMM; LRT: $\chi^2 = 4.19$, d.f. = 1, $p = 0.041$). Although back-transformed estimated marginal means indicated lower nectar sugar availability in Fire plots, by 2023 raw treatment totals were comparatively closer, reflecting species-level variation incorporated within the GLMM structure and suggesting partial recovery of nectar production two years post-wildfire.

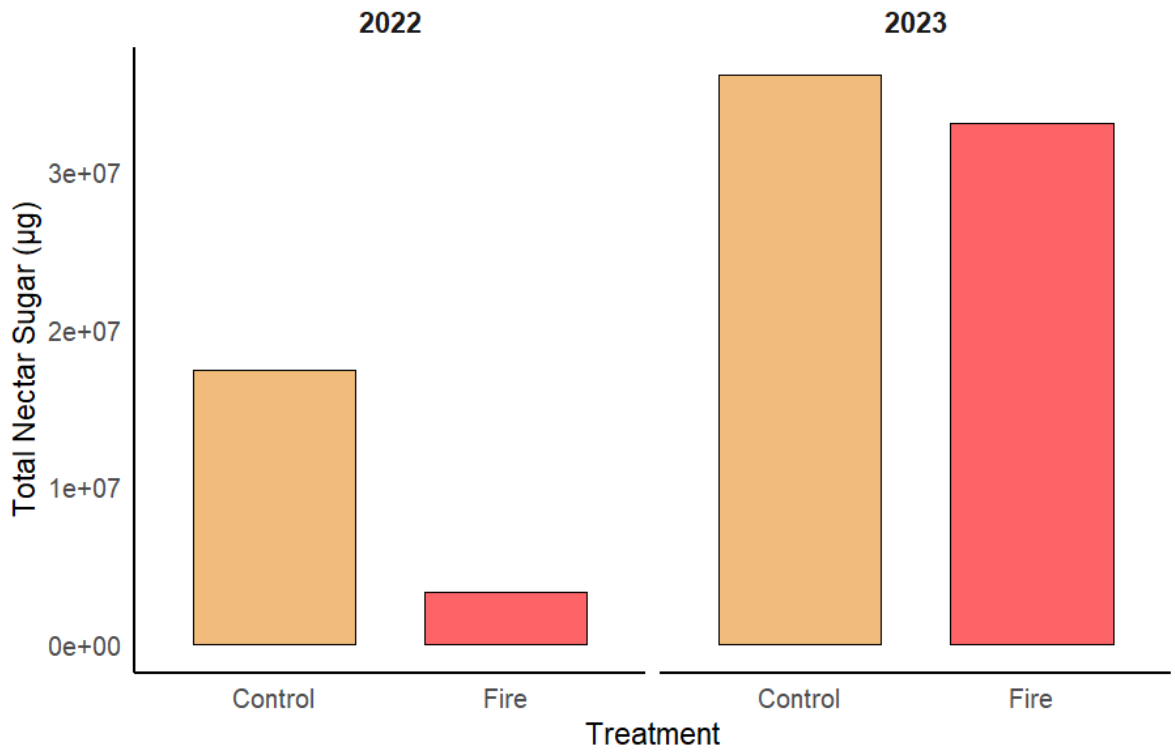


Figure 5. Bar plots illustrating total nectar sugar (µg) produced across a subset of key entomophilous plant species and additional species with quantitative nectar data available in Control and Fire plots for 2022 and 2023. Values represent the summed nectar sugar across plot per treatment–year combination, illustrating overall differences in nectar resource availability. Nectar production was lower in burned plots in 2022, but treatment differences diminished in 2023. Colours represent treatment (Control vs Fire).

Across both years, flowering species showed substantial interspecific variation in nectar sugar content, reflected in the random-intercept standard deviations (SD = 2.69 in 2022; SD = 2.32 in 2023). These random effects capture consistent differences among species in their average nectar production, independent of treatment. Random-slope GLMMs allowing species-specific fire responses failed to converge reliably, indicating insufficient information to estimate variation in treatment effects among species (see Figure 6 for cross-species nectar differences). We therefore retained the simpler random-intercept models, providing stable estimates of the overall fire effect while accounting for species-level variation.

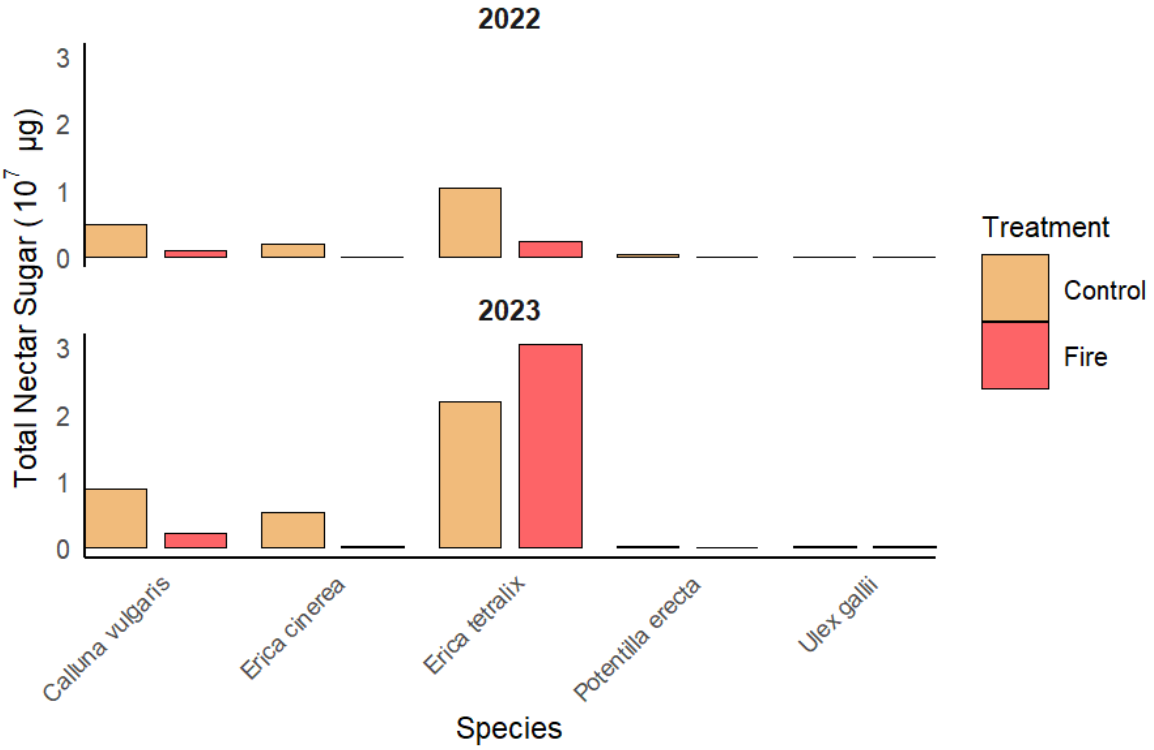


Figure 6. Bar plot illustrating total nectar sugar production for flowering species in Control and Fire plots in 2022 and 2023. Bars show the total estimated nectar sugar (µg × 10^7) contributed by each species under each treatment. Colours distinguish Control and Fire treatments and facets separate years to illustrate temporal changes in nectar availability following fire. Species-level totals integrate flower density, flowering duration, and measured nectar sugar per flower. This plot summarises treatment differences in nectar resources without aggregating species, highlighting which key taxa and additional contributors with empirical data available, account for the majority of nectar provisioning.

Full flowering dataset trends

Results differed when including anemophilous (wind-pollinated) flora (for species list and results see Supplementary information), wildfire had a significant positive impact on floral units (see Figure S7) in 2022 (GLMM: β = 0.61 ± 0.08 SE, z = 7.84, p < 0.001), while in 2023 fire exhibited a smaller, negative effect on floral units (β = −0.15 ± 0.07 SE, z = −2.03, p = 0.043). This was mirrored in the number of species in flower increasing during 2022 (GLMM: β = 0.31 ± 0.12 SE, z = 2.58, p = 0.010) but had no significant effect across 2023 (GLMM: β = –0.05 ± 0.11 SE, z = –0.45, p = 0.65), although seasonal trends varied (see Figure S7-S8, Supplementary information). Shannon diversity varied significantly across sampling periods (LMM: χ^2 = 11.47, d.f. = 5, p = 0.043) but did not differ overall between Fire and Control plots

(LMM: $\chi^2 = 0.91$, d.f. = 1, $p = 0.34$). Community evenness also varied across sampling periods (LMM: $\chi^2 = 17.54$, d.f. = 5, $p = 0.0036$) and was significantly lower in Fire than Control plots overall (LMM: $\chi^2 = 5.54$, d.f. = 1, $p = 0.019$).

Similarly, ANOSIM results demonstrated significant compositional differences in full floral communities between treatments in 2022 (ANOSIM: $R = 0.514$, $p = 0.005$), with some evidence that Control communities showed greater compositional variability than Fire communities (PERMDISP: $F = 2.845$, $p = 0.055$) (see Figure S9, Supplementary information). By 2023, compositional separation between treatments was considerably weaker, with greater overlap between Fire and Control communities (ANOSIM: $R = 0.136$, $p = 0.050$).

SIMPER analysis indicated that 2022 dissimilarity was driven primarily by a strong post-fire increase in floral units of the anemophilous *Schoenus nigricans* (Black Bog-rush), alongside reduced floral abundance of the entomophilous heathland shrub *Calluna vulgaris* (Ling Heather) within Fire sites. Anemophilous *Rhynchospora alba* (White Beak-sedge) also contributed substantially to treatment dissimilarity, although with high variability among sites. By 2023, compositional differences were weaker and distributed across multiple species. Fire sites were again characterised by greater floral abundance of *Schoenus nigricans*, alongside increased floral units of *Erica tetralix* (Cross-leaved Heath) and to a lesser extent, *Rhynchospora alba*. Although several additional species also differed between treatments, their responses were less consistent between sites, contributing to increasing overlap between Fire and Control communities. In contrast, Control sites retained greater floral abundance of certain mature entomophilous heathland species *Calluna vulgaris* (Ling Heather) and *Erica cinerea* (Bell Heather), alongside the anemophilous grass *Molinia caerulea* (Purple Moor-grass).

Overall, Fire treatment saw a marked flush of total floral units during 2022, driven by increases in certain anemophilous species, alongside compositional differences from Control communities. By 2023, total floral-unit abundance in Fire plots had fallen below Control levels, while compositional separation between treatments had weakened considerably, indicating greater compositional similarity between Fire and Control communities (see S6-S11, Supplementary Information).

Discussion

Wildfire altered multiple facets of floral resources in Killarney's temperate heathland, with consequences for pollinator foraging. In partial support of H1, entomophilous floral-unit

abundance was reduced in burned sites, but species richness remained unchanged, indicating that fire altered resource quantity rather than the number of species available. Instead, shifts in community structure, including reduced evenness and changing dominance patterns, distinguished burned plots. In line with H2, these effects varied across seasons and between years, with non-linear and treatment-specific recovery trajectories. Finally, consistent with H3, variation in entomophilous floral units translated into differences in nectar resource availability, highlighting the importance of changes in floral abundance in shaping post-fire pollinator resources. In contrast, analyses of the full floral community revealed a transient post-fire increase in floral abundance and species number in 2022, followed by weaker or negative effects in 2023 (see Figure S6-S8, Supplementary Information). These patterns were driven by short-term floral increases in anemophilous species, emphasising contrasting recovery dynamics between the two pollination types.

Wildfire often increases plant and floral abundance (LaManna, 2021; Banza et al., 2021), yet plant responses remain variable due to contrasting physiological and phenological fire adaptations (Rowe, 1983; Poulos, 2018) and increasing fire vulnerability in many landscapes (Keeley et al., 2011; Moritz et al., 2014). In Killarney, increases in total floral units reflected this general pattern, but most gains derived from anemophilous species rather than flowers offering nectar or pollen rewards (Potts et al., 2003). Fire-stimulated flowering is well described in Mediterranean, arid and tropical floras (Lamont & Downes, 2011; Fidelis & Zirondi, 2021; Capitanio & Carcaillet, 2008) but remains poorly documented in temperate heathlands, yet it is central to plant reproduction and pollinator food webs (Miller et al., 2011; Kevan & Viana, 2003). The distinction between vegetative recovery and floral reward availability is therefore crucial when evaluating post-fire ecosystem function, as post-fire increases in nutrient availability may enhance primary productivity and invertebrate resources in early successional habitats (Brüggeshemke et al., 2025).

Floral diversity typically declines with time after fire before recovering (Potts et al., 2003; Banza et al., 2021). Here, the number of species in flower increased one year post-fire but entomophilous richness remained unchanged, indicating that richness alone may mask important structural shifts. Fire initially reduced community evenness and produced compositionally similar flowering communities among Fire plots, followed by convergence with controls in year two; patterns broadly consistent with secondary succession (Capitanio & Carcaillet, 2008). The pyrodiversity hypothesis predicts that biodiversity may increase under a range of fire regimes (Ulyshen et al., 2022) and species trajectories were likely shaped by the relatively consistent fire intensity across Killarney (Gittings, 2021), but recovery also depends on seed-bank persistence, moisture and nutrient availability (Baker & Shinneman, 2004; Abela, 2009) and interactions involving herbivores (Murphy et al., 2018;

Kluth et al., 2024), seed dispersers (Saracino et al., 1997; José Benedicto-Royuela et al., 2024) and pollinators (Garcia et al., 2018). Therefore, while richness did not decline, shifts in relative abundances indicate meaningful changes in resource structure experienced by flower visitors.

Future post-fire trajectories remain difficult to predict as temperate heathlands show highly variable responses to fire, driven more by fire behaviour than fire presence (Keeley et al., 2011). Divergent fire-adaptive traits (Pausas et al., 2015) and selective pressures (Lamont & Downes, 2011) can also create feedback loops, vegetation shifts altering fuels and future fire behaviour (Meyn et al., 2007). Recovery is further shaped by life-history traits (Carbone et al., 2019), pre-fire stand structure (Davies et al., 2009), fire history (Kelly et al., 2023) and shifts in fire seasonality (Tangney et al., 2022). Killarney's low to moderate fire intensity, typical of moist, herb-dominated heathlands (Flannigan et al., 2013), often favours resprouting when fires are infrequent (Clarke et al., 2015), while the presence of facultative seeders and external colonisers (Pausas & Keeley, 2014) can generate diverse trajectories. Because this study quantified floral presence, slower-growing species such as dwarf shrubs may require multiple years to reach flowering maturity, resulting in inherently lagged responses.

Post-fire shifts in species composition reflected strong species-specific trends, consistent with concerns that frequent burning may disadvantage some resprouters (Fairman et al., 2019) or promote short-lived, productive species (Dooley et al., 2023), including Molinia dominance in some heathlands (Brys et al., 2005). Killarney's floral assemblage, composed of perennial species, is typical of Atlantic heathland, contrasting with global trends where low-reward annuals often dominate immediately post-fire (Potts et al., 2003). Early increases in wind-pollinated species such as *Schoenus nigricans* and *Rhynchospora alba* altered community structure but provided no nectar benefit. Several insect-pollinated forbs (*Narthecium ossifragum, Pinguicula grandiflora, Polygala serpyllifolia)* increased in year two, although gains in these infrequently occurring species, likely did not offset large reductions in nectar-rich heathers.

Heathers, including *Calluna vulgaris* and *Erica cinerea*, are major nectar providers in Atlantic heathlands and the UK generally (Baude et al., 2016) and have declined across Europe due to nutrient inputs and land-use change (Aerts & Heil, 1993; Ramil Rego et al., 2013; Fagúndez, 2013). Although heathlands often show resilience to fire (Camac et al., 2013; Myercough & Clark, 2007), repeated burning may favour ruderal species (Clement & Touffet, 1990), due to slow regeneration in older heath stands (Davies et al., 2010) and in obligate resprouters generally (Stokes et al., 2004; Hobbs & Gimingham, 1984). In *Calluna vulgaris*,

post-fire recovery is typically driven by a combination of vegetative resprouting and seedling establishment, resulting in gradual re-development of dominance following an initial reduction in cover (Mallik & Gimingham, 1985). Killarney's heather species showed divergent trajectories after fire: *Erica cinerea* recovered weakly, *Calluna vulgaris* increased but remained suppressed, whereas after an initial reduction, *Erica tetralix* exhibited fire-stimulated flowering in year two. These asynchronous responses reflect differing life histories, fire-adaptive strategies (Pausas & Keeley, 2014) and environmental drivers, including moisture, temperature and pollinator-mediated variation (Day Briggs & Anderson, 2025; Basnett et al., 2025). Corresponding with entomophilous community analyses indicating initial treatment differences driven by heather loss, followed by partial floral convergence.

Compared with vegetation composition patterns, nectar responses globally remain less understood. Evidence from other systems shows highly variable species-level nectar responses to fire. Nectar and nectar-foraging taxa diversity often peak immediately after burning (Potts et al., 2003), yet species differ markedly in whether fire increases or decreases nectar production (Alves-Silva & Del-Claro, 2013; Geest & Baum, 2022). Our finding of reduced nectar availability but high interspecific variability reflects this pattern, and community-level assessments likely mask additional intraspecific trends. Fire-stimulated flowering in *Erica tetralix* partly compensated for losses from other heathers, but slower-growing taxa may require more time to re-enter flowering phases. Early regeneration commonly favours low-reward flowers before shifting to perennial, higher-reward species (Potts et al., 2003), and Killarney showed a similar early dominance of anemophilous taxa. Post-fire pollinator provisioning can rely on a narrow subset of host plants (Rivers and Ulyshen, 2026), while fire season and time since burning influence nectar landscapes and pollinator behaviour (Leone et al., 2023). Because nectar and pollen reward structures may take several years to stabilise and can be altered by repeated fire (Shafer et al., 2025), longer-term monitoring at Killarney would be beneficial.

Finally, the decline in dissimilarity between Fire and Control plots between years provides further evidence for recovery, although post-fire increases in shrubs and bryophytes may occur at the expense of peat-forming Sphagnum species critical for carbon storage and water retention (Kelly et al., 2023). Short-term increases in wind-pollinated species do not yet indicate a shift towards a persistent, low-nectar sward characteristic of frequently burned systems (Brys et al., 2005; Dooley et al., 2023), though invasive floral species and increased grazing by invasive deer may modify long-term trajectories. Environmental heterogeneity and seed-dispersal dynamics also shape distributions (Saracino et al., 1997), but the paired design controlled for major differences in slope, moisture and habitat structure (Fossitt,

2000). Surveys covered peak regional flowering months (May–September), when most floral activity occurs (Baude et al., 2016), supporting high sampling completeness (see Figure S5, Supplementary Information).

By quantifying floral resources, this study highlights the implications of post-fire shifts for pollination and reproductive dynamics (Day Briggs & Anderson, 2025; Artemendi et al., 2025), addressing current research gaps and providing an empirical baseline for post-disturbance resource availability and bottom-up drivers of ecological recovery. These shifts define the energetic landscape available to flower visitors and provide a foundation for future plant–pollinator network analyses (Bascompte & Jordano, 2007; Tylianakis et al., 2010; Memmott et al., 2004). This creates potential for the integration of interaction data, network structure and community resilience (Bascompte & Scheffer, 2023; Windsor et al., 2021; Pocock et al., 2012) within a resource-based framework.

Conclusions

Wildfire significantly affected multiple components of floral resources in Killarney's temperate heathland, with implications for flower visitors during at least two years post-fire. Although total floral output increased after burning, gains were dominated by wind-pollinated species, while key entomophilous taxa, particularly heathers, declined sharply. As heathers are major regional nectar providers, these shifts likely constrained pollinator resources in the first post-fire year. Nevertheless, increases in several insect-pollinated forbs and reduced community uniformity by year two indicate a rapid post-fire recovery trajectory.

Overall, this study demonstrates that evaluating post-fire floral resources, rather than vegetation structure alone, is valuable for understanding resource availability, particularly for insect flower visitors and their services in fire-affected habitats. Further work examining the impacts of plant-pollinator network structure and complexity is ongoing, using the benefits of DNA-metabarcoding for scalable multi-site comparisons within an existing framework to generate highly resolved interaction data (Evans & Kitson, 2020). Species-specific responses highlight that post-fire trajectories are heterogeneous. Some slower maturing species may require several years to re-enter flowering phases, whereas some forbs and *Erica tetralix* showed evidence of more rapid flower stimulation. Convergence of post-fire floral communities towards control treatment communities suggests that recovery is underway and could be rapid, but longer-term dynamics and whether the system returns to pre-fire structure may depend on factors including future fire behaviour, grazing pressure, and climatic conditions in the park.

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Data Availability

Data and code used in this manuscript are openly available via Zenodo: 10.5281/zenodo.22274978 (Howells et al., 2026).

Author Contributions

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Christina Maloney: Investigation (supporting). **Dom Boka:** Investigation (supporting). **Alannah Warren:** Investigation (supporting), Data curation (supporting). **Rachel Dowling:** Investigation (supporting). **Siún Ní Cheallaigh :** Investigation (supporting). **Jaime Maidwell:** Investigation (supporting). **Joanne Monaghan:** Investigation (supporting). **Darren M. Evans:** Conceptualization (lead), Formal analysis (supporting), Methodology (equal), Project administration (equal), Writing - review and editing (equal), Supervision (lead).

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Supplementary Information for: *The short-term effects of wildfire on heathland flowering plant communities as a resource for insect pollinators: a case study at Killarney National Park, Ireland*

Contents

S12 - Table illustrating floral units for all floral species across treatment types and years

S13 - Reference table denoting pollination mode and floral rewards of plant species

Statistical package reference list

Table S1. R packages, versions, purposes and references used in statistical analyses and figure preparation.

Package	Version	Citation	Purpose
base / stats	4.4.1	R Core Team (2024). *R: A language and environment for statistical computing.* R Foundation for Statistical Computing, Vienna, Austria.	Core statistical functions, data summaries, Poisson GLMs and likelihood-ratio tests.
tidyverse	2.0.0	Wickham, H. et al. (2019). Welcome to the tidyverse. *Journal of Open Source Software*, **4**(43), 1686.	General data manipulation and workflow management.
dplyr	1.2.1	Wickham, H., François, R., Henry, L., Müller, K. & Vaughan, D. (2026). *dplyr: A Grammar of Data Manipulation.* R package version 1.2.1.	Data cleaning, filtering, grouping, transformation and summarisation.
tidyr	1.3.2	Wickham, H., Vaughan, D. & Girlich, M. (2025). *tidyr: Tidy Messy Data.* R package version 1.3.2.	Data reshaping, including long/wide transformations and construction of community matrices.
stringr	1.5.1	Wickham, H. (2023). *stringr: Simple, Consistent Wrappers for Common String Operations.* R package version 1.5.1.	String manipulation during nectar-data preparation.
forcats	1.0.0	Wickham, H. (2023). *forcats: Tools for Working with Categorical Variables (Factors).* R package version 1.0.0.	Factor reordering for species-level figures.

ggplot2	4.0.3	Wickham, H. (2016). *ggplot2: Elegant Graphics for Data Analysis.* Springer-Verlag New York.	Statistical graphics and figure production.
wesanderson	0.3.7	Ram, K. & Wickham, H. (2023). *wesanderson: A Wes Anderson Palette Generator.* R package version 0.3.7.	Custom colour palettes for treatment and comparison figures.
lme4	2.0.1	Bates, D., Mächler, M., Bolker, B. & Walker, S. (2015). Fitting linear mixed-effects models using lme4. *Journal of Statistical Software*, **67**(1), 1–48.	Mixed-effects modelling of floral units, species richness, diversity and nectar resources.
glmmTMB	1.1.9	Brooks, M.E. et al. (2017). glmmTMB balances speed and flexibility among packages for zero-inflated generalized linear mixed modeling. *The R Journal*, **9**(2), 378–400.	Testing a zero-inflated negative-binomial model alternative.
DHARMa	0.4.6	Hartig, F. (2022). *DHARMa: Residual Diagnostics for Hierarchical (Multi-Level/Mixed) Regression Models.* R package version 0.4.6.	Simulation-based residual, dispersion and model-fit diagnostics.
emmeans	2.0.2	Lenth, R.V. & Piaskowski, J. (2026). *emmeans: Estimated Marginal Means, aka Least-Squares Means.* R package version 2.0.2.	Estimated marginal means, back-transformed predictions and treatment contrasts.
car	3.1-5	Fox, J. & Weisberg, S. (2019). *An R Companion to Applied Regression.* 3rd ed. Sage, Thousand Oaks, CA.	Type-II Wald tests for fixed effects in diversity models.
e1071	1.7-14	Meyer, D., Dimitriadou, E., Hornik, K., Weingessel, A. & Leisch, F. (2023). *e1071: Misc Functions of the Department of Statistics, Probability Theory Group (Formerly: E1071), TU Wien.* R package version 1.7-14.	Calculation of skewness of raw and log-transformed nectar-sugar data during model assessment.

vegan	2.7-3	Oksanen, J. et al. (2026). *vegan: Community Ecology Package.* R package version 2.7-3.	ANOSIM, PERMDISP, SIMPER, Bray–Curtis community analyses, NMDS ordination, Shannon diversity and species richness.

Note: Package versions correspond to those used for data processing, statistical analysis and figure preparation. Full software citations are provided within the table and are therefore not repeated in the Supplementary References (code and data available at 10.5281/zenodo.22274978).

Entomophilous species figures

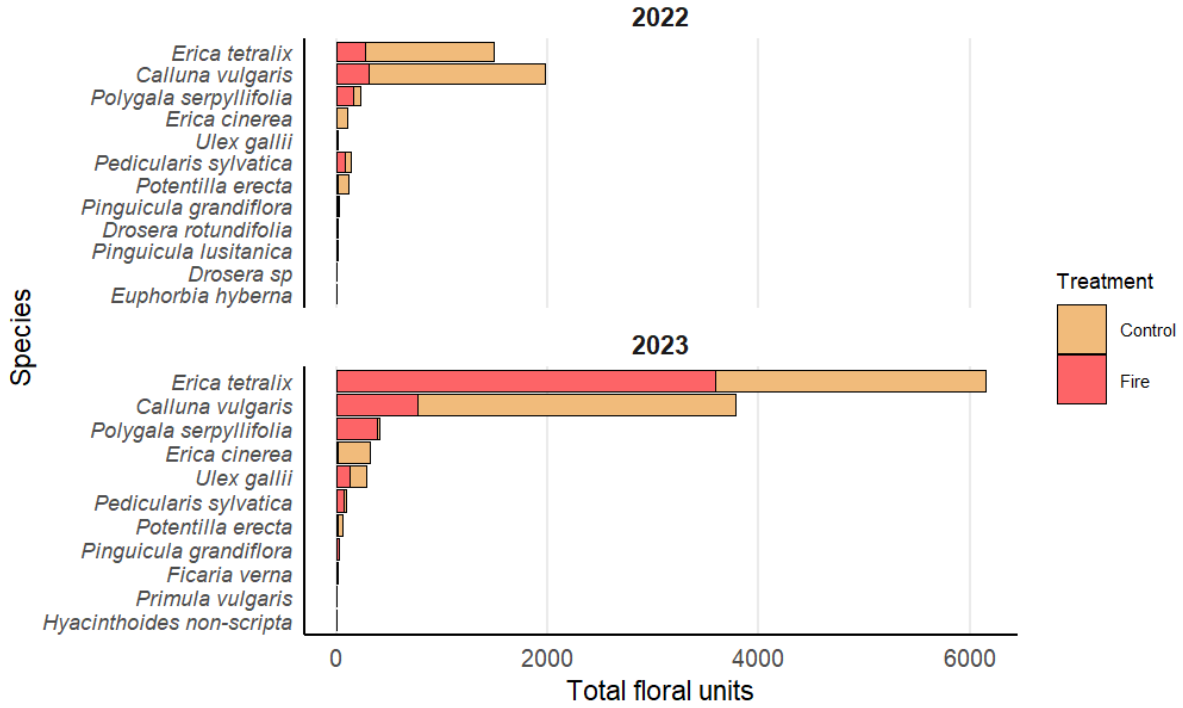

Figure S2. Species-level floral unit totals for entomophilous plants in 2022 and 2023. Stacked bars illustrate the total number of floral units per species (aggregated across all plots) in Control and Fire treatments, with taxa labelled by binomial nomenclature on the y-axis and ordered from highest to lowest floral abundance. Colours distinguish treatments.

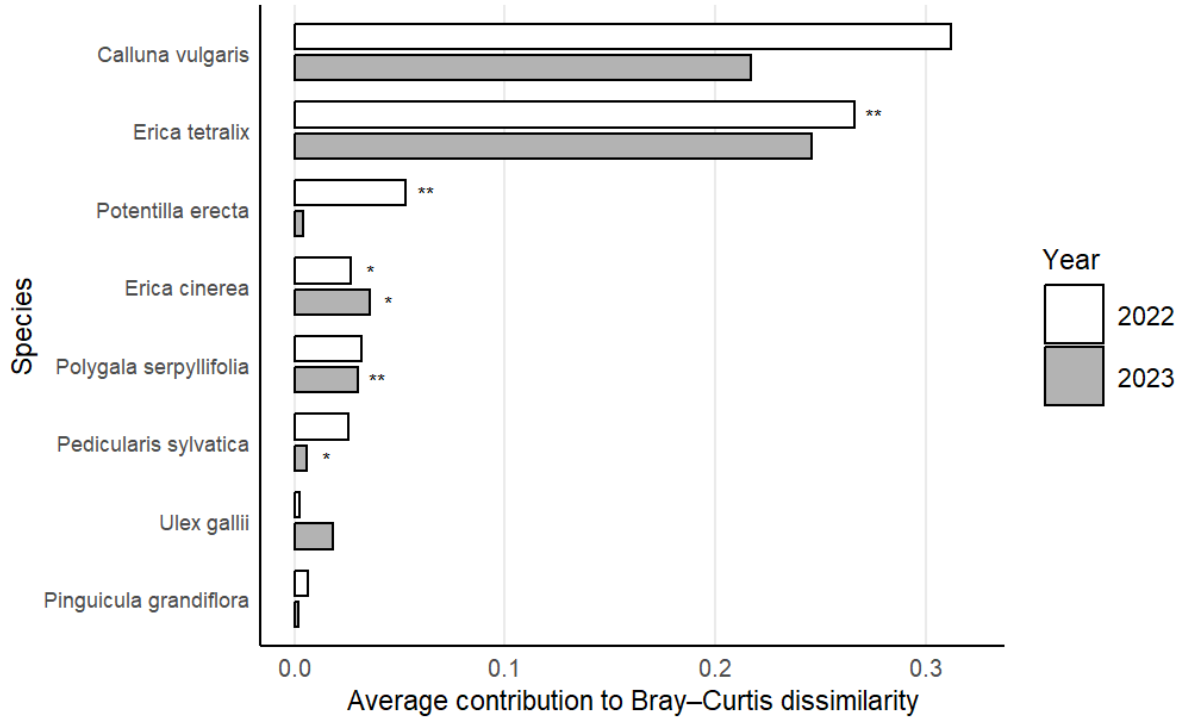

Figure S3. Horizontal bar plots illustrating the average contribution of the eight most influential entomophilous species to Bray–Curtis dissimilarity between Control and Fire communities in 2022 and 2023, based on SIMPER analysis. White bars represent 2022 and grey bars represent 2023. Species are ordered according to their maximum contribution across both years. Asterisks indicate permutation-based SIMPER significance levels (* $p \leq 0.05$; ** $p \leq 0.01$; *** $p \leq 0.001$), with permutations restricted within matched pairs.

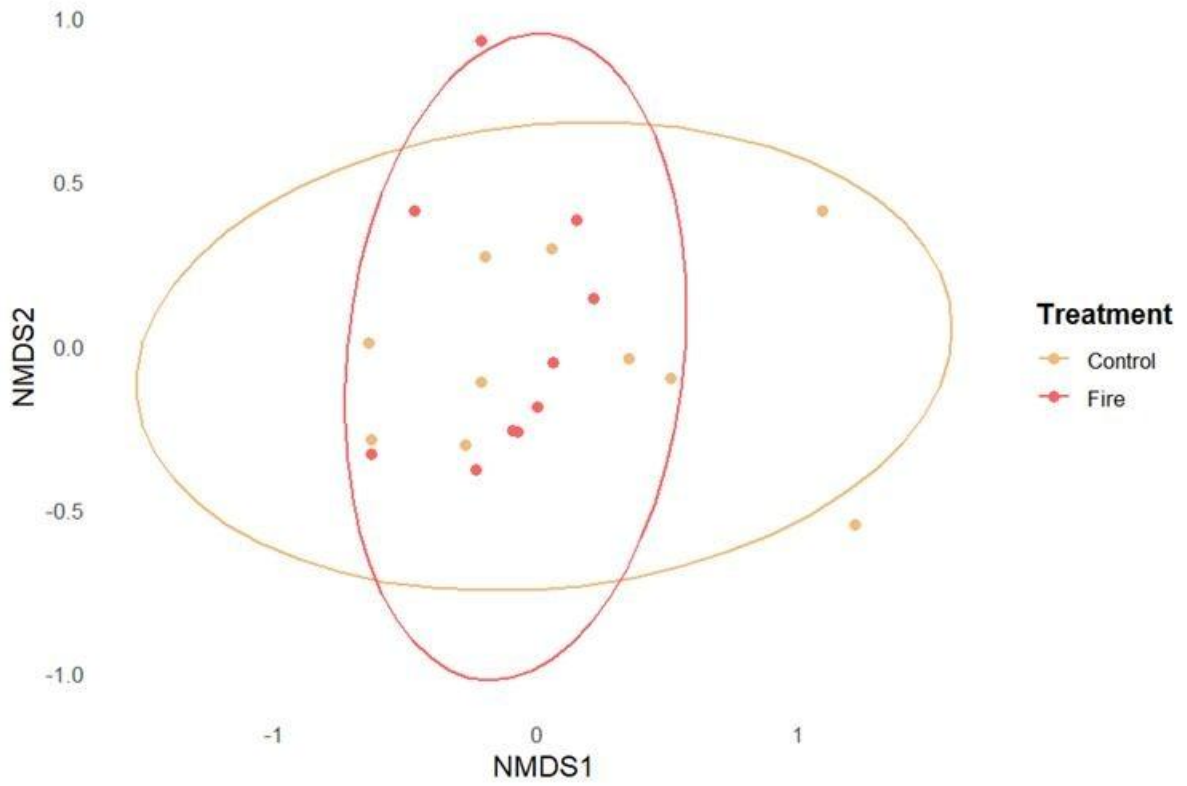

Figure S4. Non-metric multidimensional scaling (NMDS) ordination of pooled entomophilous floral communities across Control and Fire plots using Bray–Curtis dissimilarities derived from square-root transformed, Wisconsin-standardised floral-unit matrices. Square-root transformation reduced the influence of highly abundant species, while Wisconsin standardisation reduced the influence of highly abundant species and differences in total floral abundance among sites, improving comparability of community composition within the ordination. Points represent individual sites and are coloured by treatment, with ellipses illustrating 95% confidence intervals around treatment centroids. The ordination demonstrates partial compositional separation between Fire and Control communities, indicating that wildfire altered the structure of insect-pollinated floral assemblages. However, overlap between treatment ellipses suggests substantial shared community composition (stress = 0.10) and supports evidence of increasing convergence between treatments in the second year post-fire.

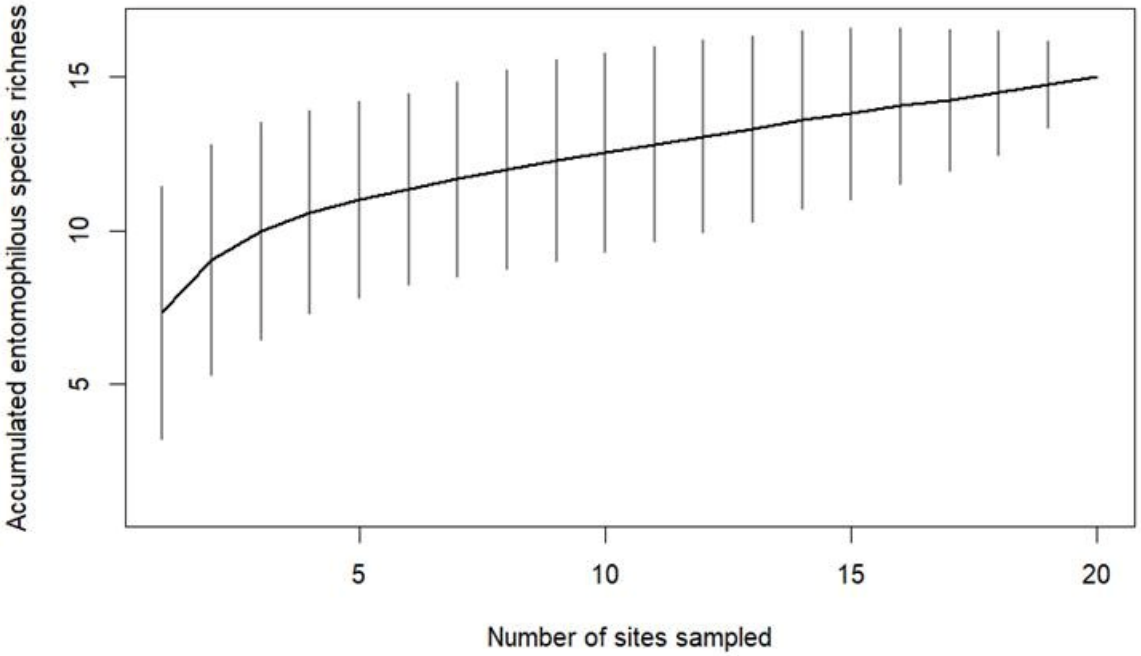

Figure S5. Species accumulation curve for entomophilous flowering species, with samples pooled across seasons and years, from the 20 study sites in Killarney National Park. The curve shows the mean cumulative number of species detected with increasing numbers of sites sampled (randomised site order; 1000 permutations) using a site × species incidence (presence–absence) matrix. Species richness increased rapidly at low sampling effort before showing diminishing returns with additional sites, indicating that most commonly occurring entomophilous species were detected within the first c. 10–12 sites, with relatively few additional species recorded thereafter. Vertical bars represent standard deviations among random permutations and reflect spatial heterogeneity in species occurrence across sites. Incidence-based richness estimation suggested the potential presence of additional undetected rare species (observed richness = 15 species; estimated richness = 24.5 ± 9.8 SE). However, the increasingly shallow accumulation curve indicates high sampling completeness and suggests that the surveyed communities captured a substantial proportion of the entomophilous flora present within the study area.

Full Dataset Results (including anemophilous species)

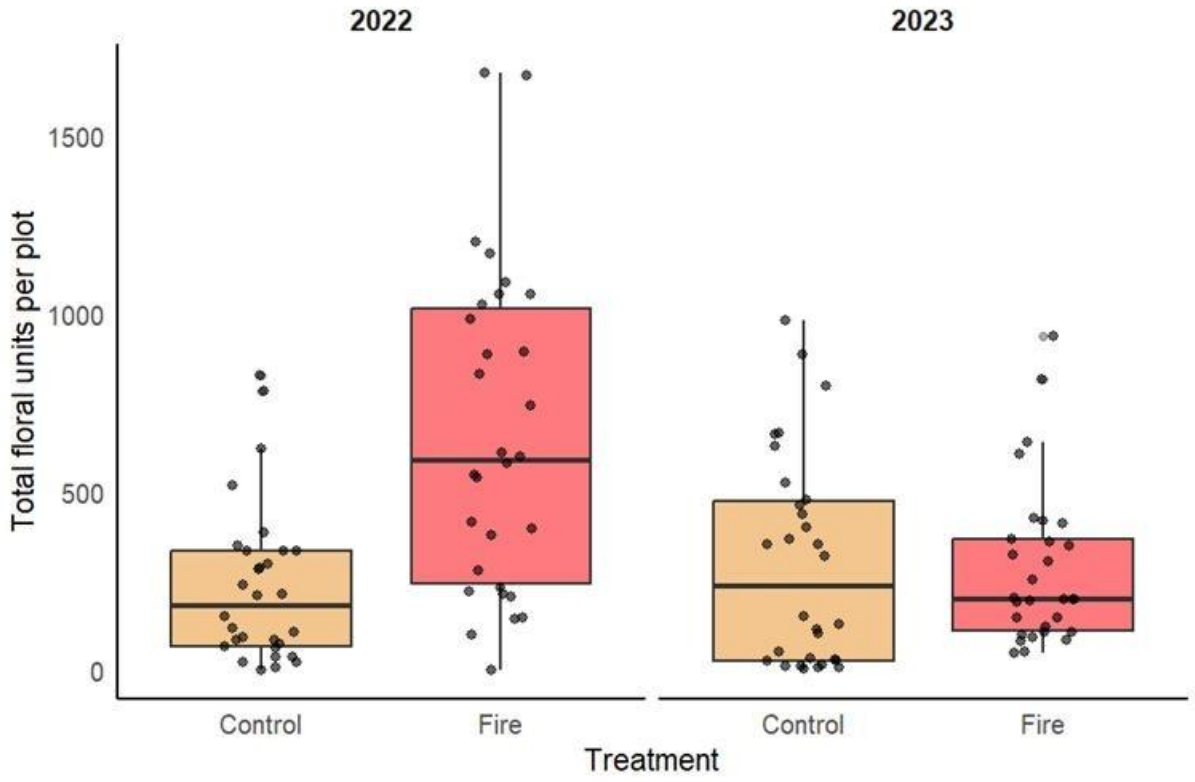

Figure S6. Box plot illustrating distribution of total floral units per plot aggregated across all sampling seasons for Control and Fire treatments across full floral dataset in 2022 and 2023. Each point represents one plot, illustrating the aggregated number of floral units recorded across the entire sampling year. Boxplots show the median and interquartile range; jittered points indicate variation between plots. Colours distinguish treatment groups.

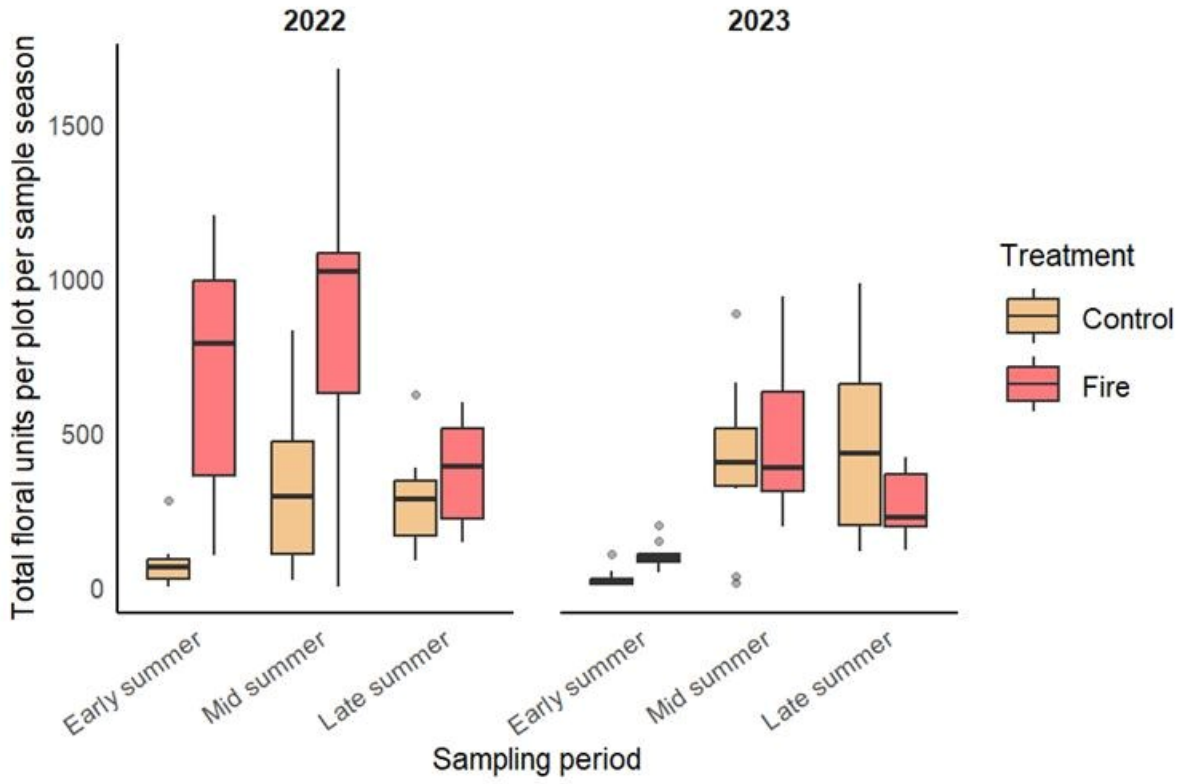

Figure S7. Box plot illustrating seasonal plot level floral unit counts for Control and Fire treatments in full floral dataset across Early, Mid, and Late summer sampling periods in 2022 and 2023. Boxplots show the distribution of floral units per plot within each sampling period, separately for each year. Jittered points indicate plot-level variation, colours distinguish treatment groups.

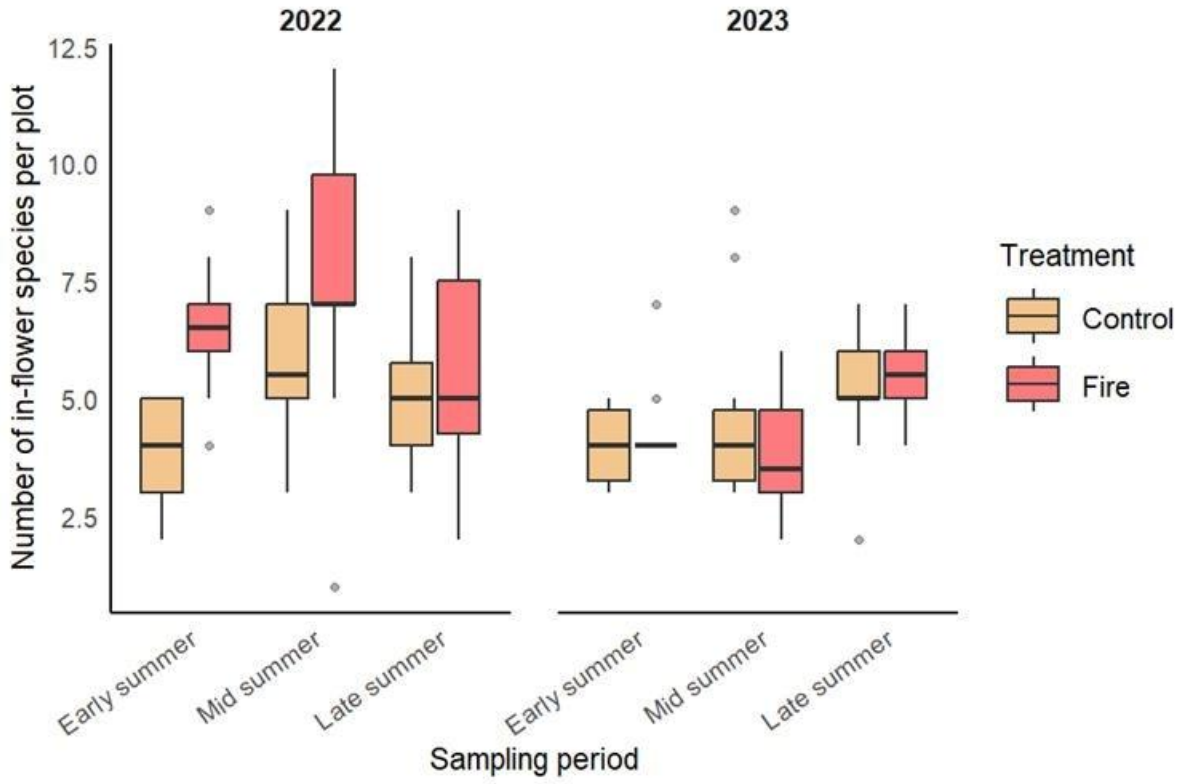

Figure S8. Box plot illustrating seasonal plot level counts of the number of species in flower for Control and Fire treatments in full floral dataset across Early, Mid, and Late summer sampling periods in 2022 and 2023. Boxplots show the distribution of flowering species per plot within each sampling period, separately for each year. Jittered points indicate plot-level variation, colours distinguish treatment groups.

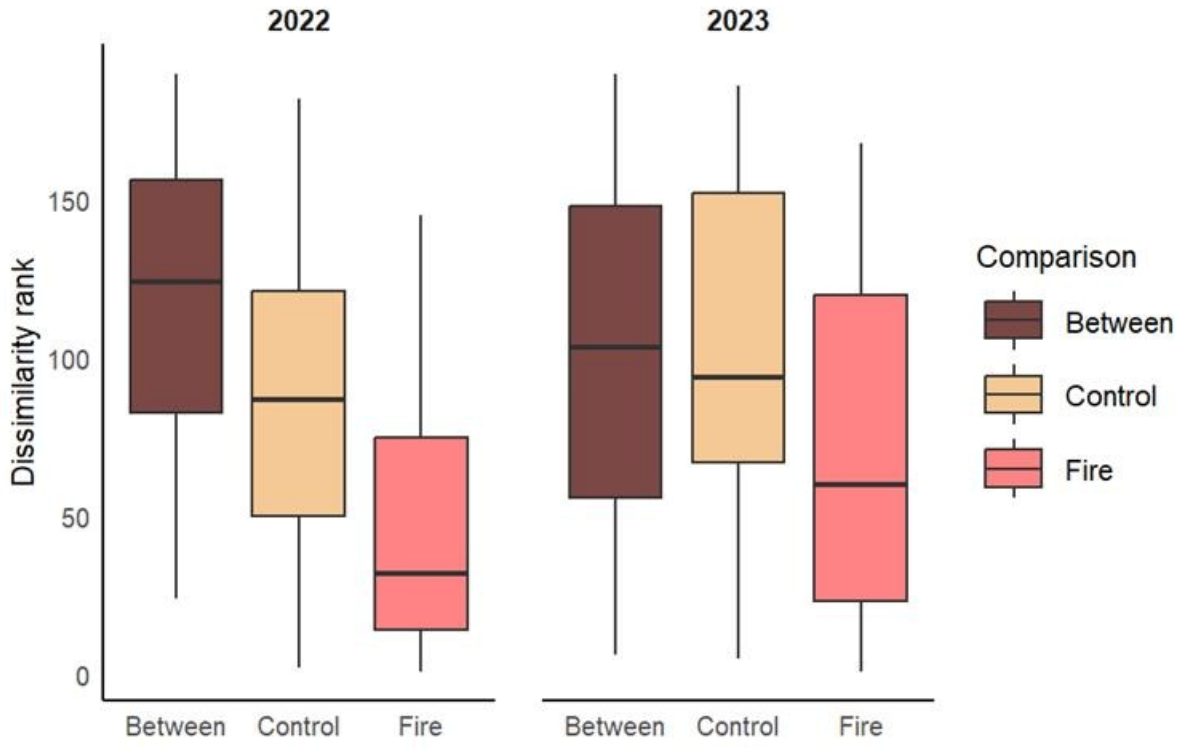

Figure S9. Boxplots showing the distribution of ranked Bray–Curtis dissimilarities within and between treatments across the full floral dataset in each year. Higher ranks indicate greater compositional dissimilarity. In 2022, Fire and Control plots were strongly separated (ANOSIM: R = 0.514, p = 0.005), with non-overlapping median ranks and spreads indicating substantial between-group differences. In 2023, group separation was weaker (ANOSIM: R = 0.136, p = 0.050), with the lower R statistic indicating greater compositional similarity between Fire and Control plots than in 2022. In both panels, the boxes illustrate the interquartile range, horizontal lines show medians, and whiskers capture the overall spread of ranks, illustrating the magnitude and variability of compositional differences between treatments.

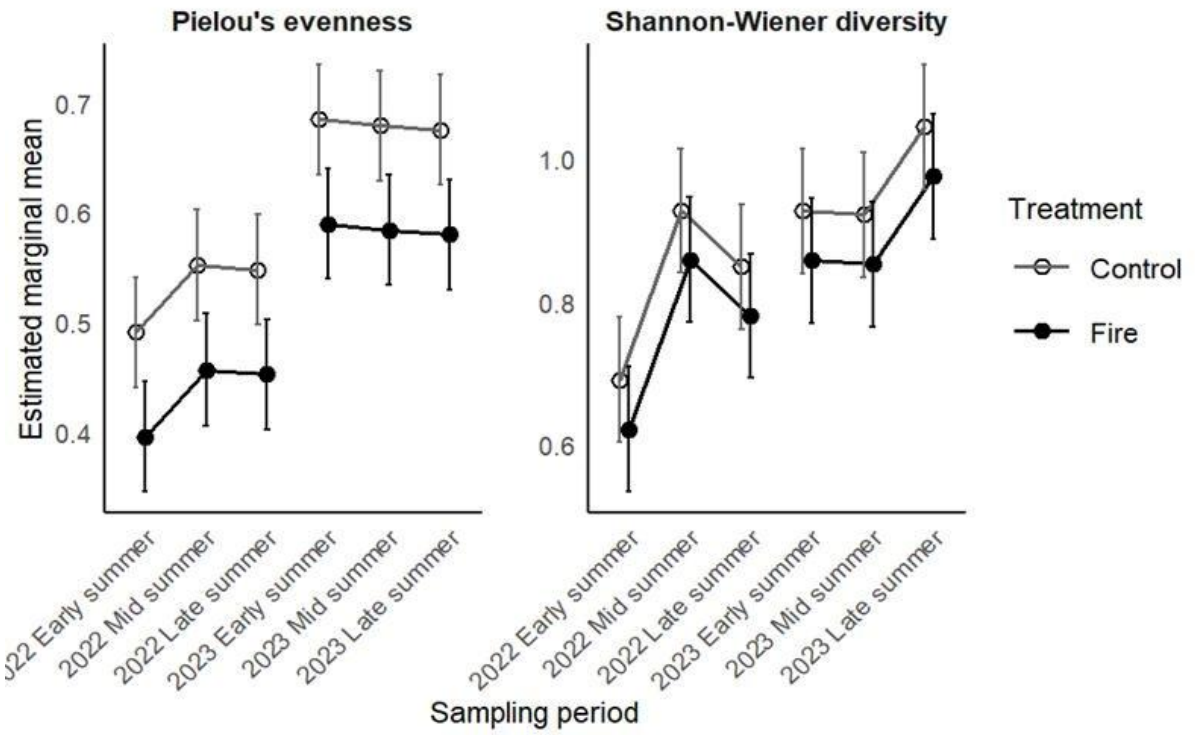

Figure S10. Estimated marginal means of Pielou's evenness (left) Shannon-Wiener diversity index (right) for full flowering communities across sampling periods in 2022 and 2023**.** Points show model-estimated means (± SE) from LMMs, with Control sites depicted using open circles and grey lines, and Fire sites with filled black circles and black lines. Lines connect sampling periods within each year to illustrate seasonal trajectories, the break in lines reflect the break between monitoring years. Facets display diversity metrics on independent y-axes, highlighting how fire altered community structure across years.

Table S11. Table illustrating Shannon Diversity index, number of species and Pielou's J score for average species evenness, per treatment and between seasons in 2022 and 2023 across the full floral dataset with data pooled. Trends described below.

	Fire			***Control***		
Season	***Shannon index***	***Total Species***	***Species Evenness***	***Shannon index***	***Total Species***	***Species Evenness***
2022						
Early Summer	*1.78*	*11*	*0.742*	*1.72*	*10*	*0.746*
Mid - Summer	*2.12*	*17*	*0.747*	*2.02*	*16*	*0.730*
Late Summer	*1.83*	*13*	*0.712*	*1.78*	*11*	*0.743*
2023						
Early Summer	*1.42*	*8*	*0.683*	*1.97*	*11*	*0.822*
Mid - Summer	*1.32*	*10*	*0.572*	*1.85*	*12*	*0.743*
Late Summer	*1.68*	*10*	*0.729*	*1.79*	*11*	*0.747*

During 2022 (year following wildfire) the total number of species in flower were consistently higher across sampling seasons in "Fire" sites (*11, 17, 13*) compared to the "Control" treatment (*10, 16, 11*), as were the diversity index scores and "mid summer" evenness scores. Post-burn communities did however illustrate a similar but lower evenness score during "Early" and "Late" summer 2022 when compared to control sites (see Figure 5).

Overall diversity and evenness metrics were higher in the "Control" sites during 2023 (two years post-fire). Seasonal trends also appear to differ when compared to the previous year. The number of species in-flower is higher in "Control" sites (*11, 12, 11*) compared to "Fire" sites (*8, 10, 10*) across the three sampling periods. Species diversity is higher in the "Control" sites across all sample seasons (1.97, 1.85, 1.79) compared to "Fire" treatments

(1.42, 1.32, 1.68). Species evenness was also higher in the “Control” sites compared to the “Fire” sites across 2023, with a largest difference accounted for by the low mid-summer evenness score across Fire treatments.

Reference tables for recorded floral species

Table S12. Table illustrating floral units for all floral species across treatment types and years

Flowering species	**Binomial Nomenclature**	**Control 2022**	**Fire 2022**	**Control 2023**	**Fire 2023**
Bell Heather	*Erica cinerea*	106	1	310	10
Black Bog-rush	*Schoenus nigricans*	887*	11475*	548	1402
Bluebell	*Hyacinthoides non-scripta*	0	0	0	0
Bog Asphodel	*Narthecium ossifragum*	33	124	2	14
Bog Myrtle	*Myrica gale*	16	4	58	0
Common Cotton-grass	*Eriophorum angustifolium*	26	101	9	12
Cross-leaved Heath	*Erica tetralix*	1213	273	2558	3589
Deergrass	*Trichophorum cespitosum*	366	1267	112	0
Hare's-tail Cotton-grass	*Eriophorum vaginatum*	14	155	8	13
Irish spurge	*Euphorbia hyberna*	1	0	0	0
Large-flowered Butterwort	*Pinguicula grandiflora*	10	14	8*	19*
Lesser Celandine	*Ficaria verna*	0	0	8	0
Ling Heather	*Calluna vulgaris*	1664*	311*	3010*	766*

Lousewort	*Pedicularis sylvatica*	54	86	19	73
Milkwort	*Polygala serpyllifolia*	75	155	30	383
Pale Butterwort	*Pinguicula lusitanica*	3	9	0	0
Primrose	*Primula vulgaris*	0	0	0	0
Purple Moor-grass	*Molinia caerulea*	229	278	198	58
Round-leaved Sundew	*Drosera rotundifolia*	4	11	0	0
Rush sp	*Juncus sp*	3	31	0	0
Sharp-flowered Rush	*Juncus acautiflorus*	115	76	28*	1*
Soft Rush	*Juncus effusus*	2	47	8	0
St Patrick's Cabbage	*Saxifraga spathularis*	1	0	0	0
Star sedge	*Carex echinata*	0	4	0	0
Sundew sp	*Drosera sp*	0	1	0	0
Tormentil	*Potentilla erecta*	106	12	38	17
Unknown	n/a	0	1	0	0
Western Gorse	*Ulex gallii*	10	2	164	121
White Beak-sedge	*Rhynchospora alba*	2044	5437	1923	2014

Table S13. Reference table denoting predominant pollination mode and floral rewards of plant species recorded in Killarney National Park heath–bog plots. Species are classified according to their predominant reproductive or pollination mode, with floral rewards identified as nectar and/or pollen where applicable.

Species	**Predominant pollination mode**	**Floral reward**	**Reference(s)**
Erica cinerea (Bell Heather)	Entomophilous	Nectar & pollen	Proctor, M., Yeo, P. & Lack, A. (1996). *The Natural History of Pollination*. HarperCollins, London; Baude, M. et al. (2016). Historical nectar assessment reveals the fall and rise of floral resources in Britain. *Nature* 530, 85–88; Stace, C.A. (2019). *New Flora of the British Isles*. 4th ed. C&M Floristics, Suffolk.
Schoenus nigricans (Black Bog-rush)	Anemophilous	None	Wragg, P.D. & Johnson, S.D. (2011). Transition from wind pollination to insect pollination in sedges: experimental evidence and functional traits. *New Phytologist* 191, 1128–1140; Stace (2019).
Hyacinthoides non-scripta (Bluebell)	Entomophilous	Nectar & pollen	Proctor et al. (1996); Baude et al. (2016); Stace (2019).
Narthecium ossifragum (Bog Asphodel)	Entomophilous	Pollen only	Jacquemart, A.-L. (1996). Selfing in *Narthecium ossifragum* (Melanthiaceae). *Plant Systematics and Evolution* 203, 99–110; Proctor et al. (1996); Stace (2019).
Myrica gale (Bog Myrtle)	Anemophilous	None	Skene, K.R., Sprent, J.I., Raven, J.A. & Herdman, L. (2000). *Myrica gale* L. *Journal of Ecology* 88, 1079–1094; Stace (2019).
Eriophorum angustifolium (Common Cotton-	Anemophilous	None	Wragg & Johnson (2011); Stace (2019).

grass)			
Erica tetralix (Cross-leaved Heath)	Entomophilous	Nectar & pollen	Proctor et al. (1996); Baude et al. (2016); Stace (2019).
Trichophorum cespitosum (Deergrass)	Anemophilous	None	Wragg & Johnson (2011); Stace (2019).
Eriophorum vaginatum (Hare's-tail Cotton-grass)	Anemophilous	None	Wragg & Johnson (2011); Stace (2019).
Euphorbia hyberna (Irish Spurge)	Entomophilous	Nectar	Proctor et al. (1996); Baude et al. (2016); Stace (2019).
Pinguicula grandiflora (Large-flowered Butterwort)	Entomophilous	Nectar	Heslop-Harrison, Y. (2004). *Pinguicula* L. *Journal of Ecology* 92, 1071–1118; Proctor et al. (1996); Stace (2019).
Ficaria verna (Lesser Celandine)	Entomophilous	Nectar & pollen	Proctor et al. (1996); Baude et al. (2016); Stace (2019).
Calluna vulgaris (Ling Heather)	Entomophilous	Nectar & pollen	Proctor et al. (1996); Baude et al. (2016); Stace (2019).
Pedicularis sylvatica (Lousewort)	Entomophilous	Nectar & pollen	Proctor et al. (1996); Stace (2019).
Polygala serpyllifolia (Milkwort)	Entomophilous	Nectar & pollen	Proctor et al. (1996); Stace (2019).

Pinguicula lusitanica (Pale Butterwort)	Entomophilous, with facultative self-pollination	Nectar	Heslop-Harrison (2004); Proctor et al. (1996); Stace (2019).
Primula vulgaris (Primrose)	Entomophilous	Nectar & pollen	Proctor et al. (1996); Baude et al. (2016); Stace (2019).
Molinia caerulea (Purple Moor-grass)	Anemophilous	None	Taylor, K., Rowland, A.P. & Jones, H.E. (2001). *Molinia caerulea* (L.) Moench. *Journal of Ecology* 89, 126–144; Stace (2019).
Drosera rotundifolia (Round-leaved Sundew)	Entomophilous/ autogamous	None	Murza, G.L. & Davis, A.R. (2003). Comparative flower structure of three species of sundew (*Drosera anglica*, *Drosera linearis*, and *Drosera rotundifolia*) in relation to breeding system. *Canadian Journal of Botany* 81, 1129–1142; Stace (2019).
Juncus spp.	Predominantly anemophilous	None	Huang, S.-Q., Xiong, Y.-Z. & Barrett, S.C.H. (2013). Experimental evidence of insect pollination in Juncaceae, a primarily wind-pollinated family. *International Journal of Plant Sciences* 174, 1219–1228; Stace (2019).
Juncus acutiflorus (Sharp-flowered Rush)	Anemophilous	None	Huang et al. (2013); Stace (2019).
Juncus effusus (Soft Rush)	Predominantly anemophilous	None	Huang et al. (2013); Stace (2019).
Saxifraga spathularis (St Patrick's Cabbage)	Entomophilous	Nectar & pollen	Proctor et al. (1996); Stace (2019).
Carex echinata (Star Sedge)	Anemophilous	None	Wragg & Johnson (2011); Stace (2019).

Drosera spp. (Sundew sp.)	Unclassified	—	Stace (2019).
Potentilla erecta (Tormentil)	Entomophilous	Nectar & pollen	Proctor et al. (1996); Baude et al. (2016); Stace (2019).
Ulex gallii (Western Gorse)	Entomophilous	Pollen	Stokes, K.E., Bullock, J.M. & Watkinson, A.R. (2003). *Ulex gallii* Planch. and *Ulex minor* Roth. *Journal of Ecology* 91, 1106–1124; Proctor et al. (1996); Stace (2019).
Rhynchospora alba (White Beak-sedge)	Predominantly anemophilous	Pollen	Villa-Machío, I., Zamora, J.C., Sandoval-Sierra, J.V., Blanco-Pastor, J.L., Fernández-Mazuecos, M. & Jiménez-Mejías, P. (2022). Insect pollination in temperate sedges? A case study in *Rhynchospora alba* (Cyperaceae). *Plant Biosystems* 156, 196–202; Stace (2019).

Pollination mode represents the predominant reproductive pathway reported in the cited literature for the species or associated taxonomic group, it is not necessarily the exclusive pathway. Floral rewards indicate resources potentially available to floral visitors. Unidentified *Drosera* records were not assigned a pollination mode or floral reward because reproductive traits could not be reliably attributed without species-level identification.

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