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Careful plant choice can deliver more biodiverse vertical greening (green façades)

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Careful plant choice can deliver more biodiverse vertical greening (green façades)

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A R T I C L E I N F O

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A B S T R A C T

Plants growing against walls (green façades) are an important part of urban greening. We report on an experiment that used a set of replicated mini model building plots designed to quantify and compare potential biodiversity benefits associated with three plant species commonly grown as green façades in temperate climates: *Hedera helix* (common ivy) (either as a straight species, or a mix with *H. helix* 'Glacier'), *Parthenocissus tricuspidata* (Virginia creeper) and *Pileostegia viburnoides* (climbing hydrangea). We assessed the relative abundance of invertebrates collected from green façades in Reading (UK), over two growing seasons. The abundance of invertebrates increased with wall vegetation depth and cover, where considerably more invertebrates were collected from vigorous/deeper leaf wall cover by *Hedera helix* compared to the other treatments. A combination of two ivy taxa showed a higher invertebrate abundance compared to *H. helix* alone. The experiment demonstrates that green façades on buildings provide resources for invertebrates; the more vegetation resources there are, and the more varied they are, the more invertebrates are supported. It is clear that green façades can add to the value of invertebrate abundance on buildings and that plant choice is a strong determining factor to that value.

1. Introduction

Urban green infrastructure is a mixture of variable landscape types including parks, domestic gardens, street trees, school playgrounds, cemeteries, railway embankments and other green corridors, green roofs and walls/façades. These provide a range of benefits to the urban environment which include regulation of ambient temperatures and humidity, reduced likelihood of localised flooding, trapping of gaseous and particulate pollution, food production, biodiversity and cultural/wellbeing services (for example being used for physical activity, social interaction and relaxation) (Cameron et al. 2012; Cameron et al. 2014; Pérez and Perini 2018, Blanusa et al. 2019; Jones et al. 2022).

Green façades (climbing or self-standing plants growing with minimal support up and along building walls), together with more engineered living wall systems, can provide these green infrastructure benefits as a part of urban design particularly where ground space is at a premium (Perini et al. 2011). Some quantification of a range of these ecosystem services provided by green façades has been made (for overviews see Perini et al. 2011; Manso et al. 2021; Jones et al. 2022) including: thermoregulation and relative humidity (Cameron et al. 2014; Thomsit-Ireland et al. 2020), air quality improvement (Pugh et al. 2012; Hellebaut et al., 2022), increased property value and other economic benefits (Collins et al. 2017). Green façades have the additional advantage of a relatively low cost compared to more engineered wall systems (Manso et al. 2021). Despite this information the implementation of green façades (both in new build and as retrofit) remains limited. It had been suggested that this is rarely due to lack of potential sites for implementation, but rather the ownership, management, and economic issues (Gantar et al., 2022). There could also be an issue of designers, planners and decision-makers often facing conflicting or incomplete information on the effectiveness of green façades to deliver multiple benefits and possible drawbacks (Hall et al., 2014). It is therefore important that the information about the ability of this form of green infrastructure to provide a suite of co-benefits is available (Perini et al. 2011). One aspect that may help maximise the uptake and investment into green infrastructure is the high value that the public attributes to two services in particular: how those green spaces make them feel (wellbeing aspects) and the impact they have on biodiversity (Blanusa

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et al. 2021, Samus et al., 2022). The latter has also been widely recognised as helping to improve sustainability of the urban environment (Mayrand and Clergeau, 2018). Furthermore, the invertebrate abundance and diversity that potentially occupy these green walls and façades will play central role to maintaining the ecological function and health of urban green infrastructure (Kotze et al., 2022). Consequently, there is value in robust, evidence-based guidance on the impact of green wall/façades' planting choices for supporting biodiversity, in addition to the other environmental services, highlighting the importance to the practitioners and urban residents in maintaining these green spaces (Kotze et al., 2022).

It is often stated that green walls and façades are beneficial for biodiversity (e.g. Elgizawy, 2016) although quantified evidence has been highlighted as lacking (Matt et al. 2015; Manso et al. 2021; Mayrand and Clergeau, 2018). In some cases it has even been suggested that the biodiversity value of green façades is low (e.g. Jones et al. 2022). However, although no differences between plant species was presented, a study of green façades in Staffordshire UK (dominated by *Hedera*, *Pyracantha*, *Jasminum* and *Parthenocissus*) showed that they were used for forage and nesting for a variety of birds throughout the year, but particularly shelter and forage in winter (Chiquet et al. 2013). Furthermore, green façades support greater invertebrate abundance from a range of taxonomic and feeding guilds (functional groups) compared to bare walls in North America and northern Europe, with increased invertebrate abundance corresponding to increasing amounts of vegetation and vegetation complexity as well as habitat availability adjacent to the walls (Matt et al. 2015; Madre et al. 2015). The type of vertical greening can also affect faunal composition. In Paris, France, the communities of beetle (Coleoptera) and spider (Araneae) on green façades (climbing plants) were similar to that found on hot dry xerothermophilous habitats similar to cliffs, whereas modular panel and felt-layer green walls tended to have a fauna similar to that found around waterfalls (Madre et al. 2015). The green façades in the Paris study related to only one species of plant, *Parthenocissus tricuspidata* (Siebold & Zucc.); the North American study (Matt et al. 2015) did not specify the species of plant in the results.

Here we report on an innovative experiment that used a set of replicated plots designed to quantify and compare a variety of potential benefits associated with three plant species commonly grown as green façades in temperate climates, whose thermoregulation and moisture retention qualities have been previously reported (Thomsit-Ireland et al. 2020). The plants in the experiment were chosen as species that are currently commonly grown as green façades in the UK (Chiquet et al. 2013) and included evergreen and deciduous species. By making minor changes to methodology that has previously been used to compare invertebrate abundance and diversity of different planting regimes our aim was to test whether, or to what extent, those plants commonly used as green facades differ in the abundance of invertebrates supported. As a reliable representation of diversity (Salisbury et al. 2020) we report on the relative abundance of invertebrates on these green façades and discuss the potential benefits they have for invertebrate abundance in the urban context.

2. Methods

2.1. Construction, plot layout and plants

The experiment was carried out using twenty model 'buildings' at a field in Berkshire, UK. The 'buildings' were constructed in 2014–5, their dimensions 60 × 50×50 cm (hwd), using standard clay bricks arranged in a stretcher bond. The buildings were built on a concrete grey slab footing (68.2 ×50×4 cm) with a 1200 gauge (0.3 mm) damp proof membrane laid upon the concrete slab. The buildings were spaced by 1.5 m (N–S) and 1.75 m (E-W); the ground between was covered in standard horticultural woven black polypropylene cover (100 g/m^2), supressing all unwanted ground plant growth. At the base of each model 'building' 0.2 m of soil/ground was left exposed to enable the planting of experimental plants. A model building 'roof' was added consisting of Metsã Wood oriented strand board (60 ×60×1.1 cm) topped with 1.3 mm bitumen shed felt.

In 2014 four plant treatments were planted around model buildings (each replicated four times, so 16 vegetated buildings), along with four bare 'control' buildings. Treatments were allocated with restricted randomisation. The plant treatments were: (i) *Hedera helix* (common ivy); (ii) a 50%−50% mix of *H. helix* L. and *H. helix* 'Glacier' (referred to further in the text as *H. helix* 'Glacier' combination) (iii) *Parthenocissus tricuspidata* 'Veitchii' (Siebold & Zucc.) Planch (Virginia creeper); and (iv) *Pileostegia viburnoides* (climbing hydrangea) *Hook.f. & Thomson* (Fig. 1). Two plants of a single species/treatment were planted 0.25 m apart on each wall of the buildings, eight plants in total per building. By the first sampling date in 2016 all plants were well established, giving near 100% coverage of wall area. *Hedera helix* and *P. viburnoides* are evergreen so they provided cover year-round, whereas *P. tricuspidata* buildings were bare branches in the period mid-November to mid-April and foliated otherwise. Plants were pruned once per year, typically in early November at the base of the building, ensuring full building coverage, but no sprawling further away.

2.2. Invertebrate sampling

Invertebrates were sampled between spring (April) and early autumn (October) on ten occasions between July 2016 and July 2018. A Vortis suction sampler was used to sample plant-inhabiting invertebrates (Arnold, 1994, Burkard Manufacturing Co. Limited, Rickmansworth, Herts, UK). The nozzle was held approximately 5–10 cm into the vegetation in the centre of each 'building' wall for 10 s on each side (40 s per building), where no plant material was present the nozzle was held approximately 5–10 cm from the wall surface. This was at approximately 45° angle to the operator. Whilst the Vortis suction sampler is operated at best efficiency placed directly on the ground (90 degrees) (Mommertz et al. 1996) the vertical nature of the vegetation being sampled meant this was not possible. The Vortis sampler has previously been successfully used at less than maximum efficiency and provided the methodology is standardised it is a method that enables treatment comparisons (Salisbury et al. 2017). On all sampling occasions vegetation was dry to the touch and all buildings were sampled between 09:30 and 12:00, the order of recording being completely randomised. All samples were taken by the same operator to reduce bias.

The collection tube was removed from the Vortis immediately after sampling each building, and stored in a cool box in the field. In the laboratory, to reduce catch activity the tubes were cooled to 4–10 °C for approximately 20 min. On the day of sampling a pooter was used to separate invertebrates from plant debris. Further identification of the invertebrates was carried out at a later date after the catch was killed by freezing for 24 h (−20 °C).

2.3. Vegetation depth co-variate

Visual estimations of the proportion of wall covered by vegetation indicated that vegetation depth was a better comparative measure of vegetation resource, as with the exception of the no plant treatment walls were in almost all cases completely colonised by the plants (as shown in Fig. 1b). This is similar to the estimates of Thomsit-Ireland et al. (2020) made on the plots in 2015 and 2016. The depth (cm) of vegetation was measured at each recording event. Two measurements were made per wall of each vegetated building, approximately 0.3 m up from the ground.

2.4. Invertebrate identification

Invertebrates were identified to a level that enabled allocation to ecological functional group of the stage sampled. In addition to a total

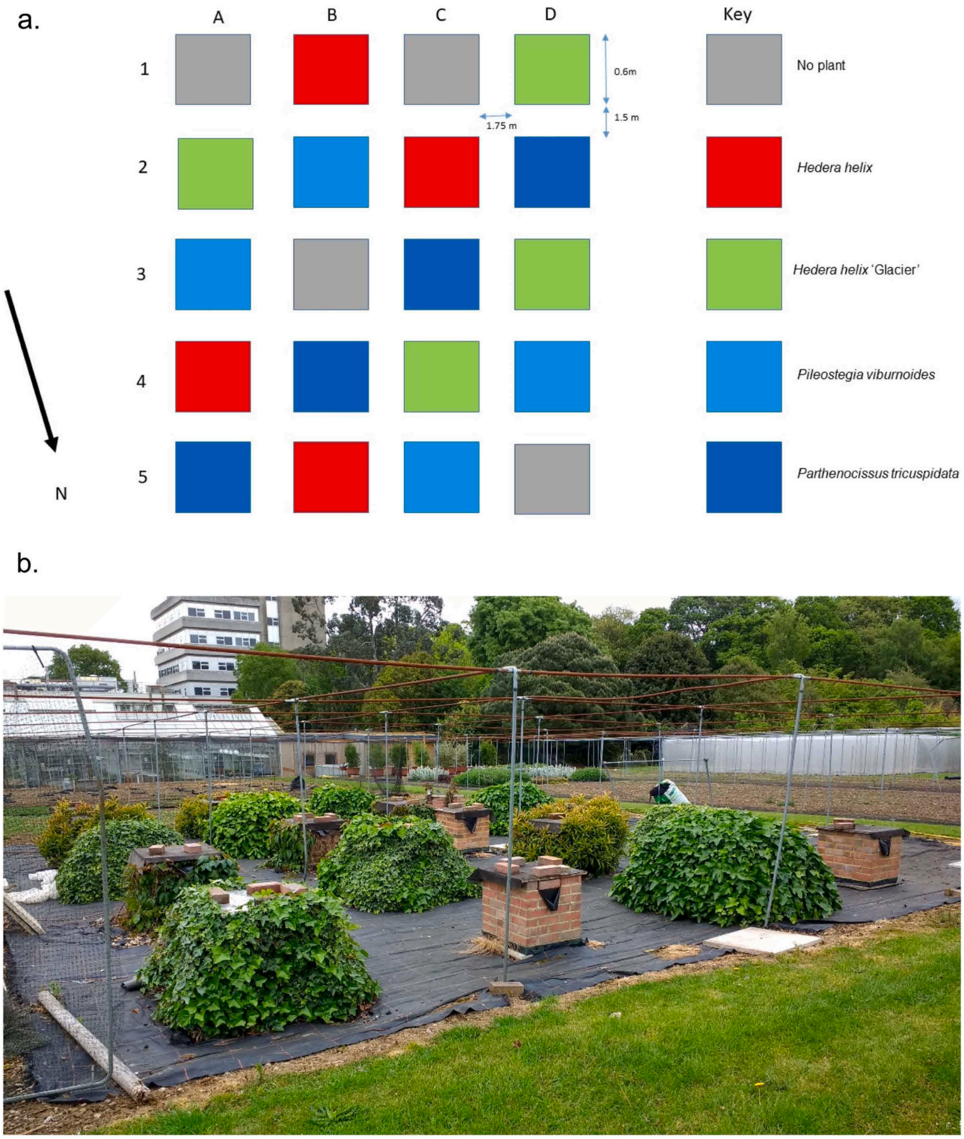

Fig. 1. Green wall plot layout at the University of Reading. a. Schematic layout. b. Image of the plots in May 2017 bottom left corner is plot D1 *Hedera helix 'glacier'*.

count (all invertebrates, excluding ants, Hymenoptera: Formicidae due to their colonial nature) functional group allocations were: herbivores (excluding nectar and pollen feeders), predators and detritivore/fungivore (Detritivore). Predators were those that feed primarily by killing and consuming prey; Detritivores were those likely to be feeding on dead/decomposing material or associated fungi. Slugs were excluded from the functional groups but not the total due to many species being herbivorous and detritivores (Rowson et al. 2014) and identification to species was not feasible. Two subgroups of predators were classified: the parasitoid Hymenoptera (parasitica), and spiders (predator: Araneae). A taxonomic subgroup of detritivores, the springtails (Detritivore:Collembola), was also included. The allocations of taxa to these functional groups are indicated in Supplementary Resource 1 Data.

2.5. Data analysis

Following methodology that created similar abundance data (Salisbury et al. 2017), analysis was carried out by Linear Regression in Genstat 22nd edition (VSN International, 2022). The number of individual invertebrates (abundance) per 'building' per 40 s Vortis sampling occasion, c, was transformed logarithmically to $y = \log_{10}(c+1)$. A covariate of mean vegetation depth (cm) was calculated for each building at each sampling event. For modelling, the no plant data was removed as there was a very low catch only and it did not make biological sense to include for this reason, similar to Madre et al. (2015) who also excluded uncovered walls from analysis. For each sampling occasion three linear regression models of transformed abundance, y, on vegetation depth, v, were compared: a single line through all four plant treatments; four parallel lines allowing the intercept to vary with treatment; and four separate lines allowing both slope and intercept to vary with treatment. Standard partial *F*-tests on two degrees of freedom were used to discriminate between models (Perry, 1982; Hawes et al. 2003). The mean vegetation depth was 29.4 cm. For ease of interpretation, estimated values of back-transformed invertebrate abundance on plants, at this mean value of v = 29.4 cm, were computed from the best-fitting regression model.

3. Results

A total of 1703 invertebrates were collected from a wide range of taxonomic and functional groups (Table 1) over the ten sampling occasions. Ants (Hymenoptera: Formicate) made up 139 specimens and due to their colonial nature these were excluded from analysis. Eight specimens were collected from the no plant controls (functional group

Table 1
Summary of fitted regressions of total abundance of invertebrate functional groups on vegetation depth for four plant treatments as green façades.

Functional group	n	P intercept	P slope	Best fitting model	d.f. for best fitting model	*F* for best fitting model	Estimated intercept (s)				Estimated slope	Estimated abundance at typical values of vegetation depth			
							Hedera helix	*H. helix* 'Glacier'	*Parthenocissus tricuspidata*	*Pileostegia viburnoides*		*H. helix*	*H. helix* 'Glacier'	*P. tricuspidata*	*P. viburnoides*
All invertebrate (excluding Hymenoptera: Formicidae)	1556	**< 0.001**	0.12	Parallel lines	4, 139	16.65	0.80	0.87	0.41	0.47	0.008	10.77	12.63	4.40	5.11
Herbivore	181	**< 0.001**	0.92	Parallel lines	4, 139	14.14	0.21	0.28	0.05	-0.02	0.004	2.10	2.48	1.46	1.23
Predator	385	**0.001**	0.26	Parallel lines	4, 139	5.55	0.26	0.33	0.11	0.11	0.007	3.02	3.51	2.13	2.12
Predator (Araneae)	135	**< 0.001**	0.15	Parallel lines	4, 139	7.22	0.35	0.40	0.15	0.24	-0.003	1.84	2.08	1.16	1.44
Predator (Parasitica)	185	0.60	0.63	One line	1, 143	31.53	-0.09				0.010	1.63			
Detritivore	467	**< 0.001**	0.38	Parallel lines	4, 139	8.96	0.60	0.59	0.20	0.33	0.0003	4.06	4.02	1.60	2.20
Detritivore (Collembola)	367	**< 0.001**	0.59	Parallel lines	4, 139	8.78	0.64	0.55	0.15	0.41	-0.003	3.49	2.89	1.14	2.08
Unclassified	520	**< 0.001**	0.37	Parallel lines	4, 139	5.77	0.30	0.23	0.15	0.07	0.008	3.33	2.88	2.39	1.96

Differences between intercepts were tested by partial F-tests (with probability P intercepts); similarly for slopes (with probability P slopes); all F-statistics have 2, m degrees of freedom where m> 100
From these tests the best-fitting model was determined: possible outcomes were either three lines of different slope and intercept, three parallel lines with different intercepts, a single line (no difference between the treatments) or no line (no significant relationship with canopy cover). The estimated intercepts and slopes are fitted values from the best-fitting model. Estimated abundance (natural scale) for typical values of vegetation depth is a back transformed estimate of total abundance for a vegetation depth of 29.4 cm.

breakdown 1 herbivore, 3 predators, 1 detritivore, 3 undetermined) the low number justifying excluding this data from the analysis. Of the 1556 remaining invertebrates, identification enabled functional group classification of 1033 (66%) specimens, 181 (12%) were confirmed as primarily herbivores, detritivores made up 467 (30%), predators consisted 385 (25%) of the catch. Of the detritivores, 367 (24%) were springtails (Collembola). Spiders (predators (Araneae)) consisted of 135 (9%) individuals and Hymenoptera (parasitica) 185 (12%). Of the 523 (34%) specimens where a functional group was not allocated 20 were slugs (Gastropoda), 224 (14%) mites (Arachnida), 261 (17%) Diptera (true fly) adults, one mirid bug nymph (Hemiptera: Miridae) and three lacewing adults (Neuroptera). Six specimens were identified as pollinators and eight as omnivores. Supplementary Resource 1 data provides full details of identifications and functional groups.

Changes in the mean vegetation depth and mean abundance per plot between sampling occasions, are shown in Fig. 2 for each of the four treatments, excluding 2016 as only one measurement was taken in that year, in July. These include data for the 'no plant' treatment, indicating the low values further justifying exclusion from the analysis. Measurements of plant depth indicated that across all plant treatments the amount of vegetation peaked in late summer 2018 (August-September) and was lowest as the beginning of the recording season (April-May) (Fig. 2.1). *Parthenocissus tricuspidata* gave consistently lower values than the other treatments which were comparable in the depth of vegetation. The pattern of abundance of invertebrates across the plant treatments was similar, with peaks in April and September 2017. Abundance declined on all plant treatments from April to June in 2017 and 2018. In 2017 a peak in abundance was observed in September (Fig. 2.2). An exception is high abundance of invertebrates recorded on *P. tricuspidata* in July 2018.

For total abundance, abundance of the three primary functional groups (Herbivore, Predator, Detritivore) and abundance of those not allocated to a functional group, the best fit model was three parallel lines with different intercepts for the four treatments (Table 1) and there was a positive linear dependence with vegetation depth. This indicates that regardless of vegetation depth, the multiplicative differences between the treatments, measured as proportions, remained constant. There were however exceptions amongst the subgroups, both Predator (Araneae) and Detritivore (Collembola) gave a negative linear dependence with vegetation depth. In addition, Hymenoptera (parasitica) showed a linear dependence with vegetation depth but no difference between the treatments.

The observed data and fitted relationships for the 1556 (total minus ants) invertebrates collected are shown in Fig. 3; estimates of regression coefficients and significance tests are presented in Table 1. Considerably more invertebrates were collected from the two *Hedera helix* (33% *H. helix* and 38% *H. helix* 'Glacier' combination) treatments compared to *P. tricuspidata* (13%) and *Pileostegia viburnoides* (16%) treatments, equating to more than double the number of invertebrates at typical values of vegetation depth. As the best-fitting model is four parallel lines, with abundance at any given value of cover highest in the *Hedera* treatments and lower in the *P. tricuspidata* and *P. viburnoides* treatments it follows that a greater amount of vegetation would be required to achieve the same invertebrate abundance as the *Hedera* treatments. Whilst the difference in abundance supported between the *Hedera* treatments and the other plant treatments is most prominent, it is noticeable that the combination of two ivy taxa (*H. helix* 'Glacier' combination) consistently gave a higher abundance than one ivy taxon alone, for total abundance being 17% higher.

Relatively similar proportions to total invertebrate abundance were observed in all three primary functional groups (Herbivore, Predator, Detritivore, Figs. at supplementary resource 2 additional results) analysed, with *Hedera* treatments achieving 28–38% of catch compared to 20% or less for the *P. tricuspidata* and *P. viburnoides* treatments, being as low as 13% for detritivores on the *P. tricuspidata* treatment. The only exception to this general pattern was the subgroup Hymenoptera

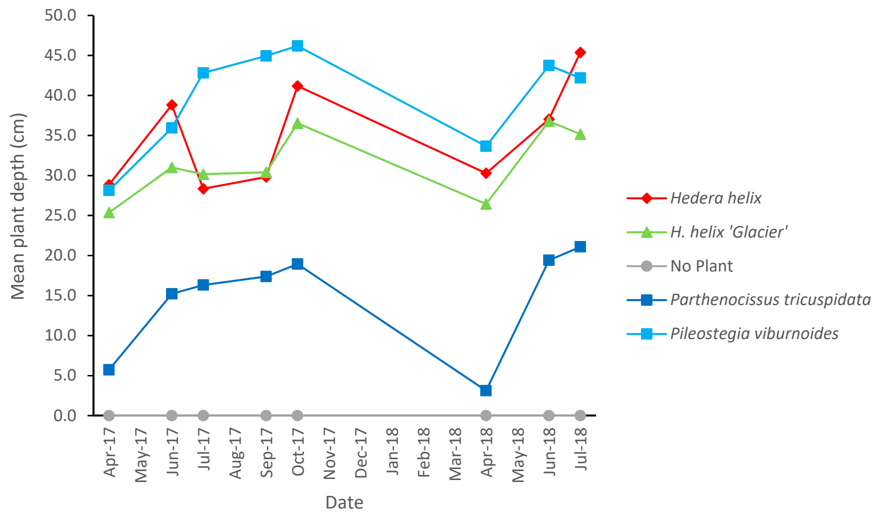

Fig. 2.1. Mean vegetation depth of green façades on the experimental buildings in 2017 and 2018.

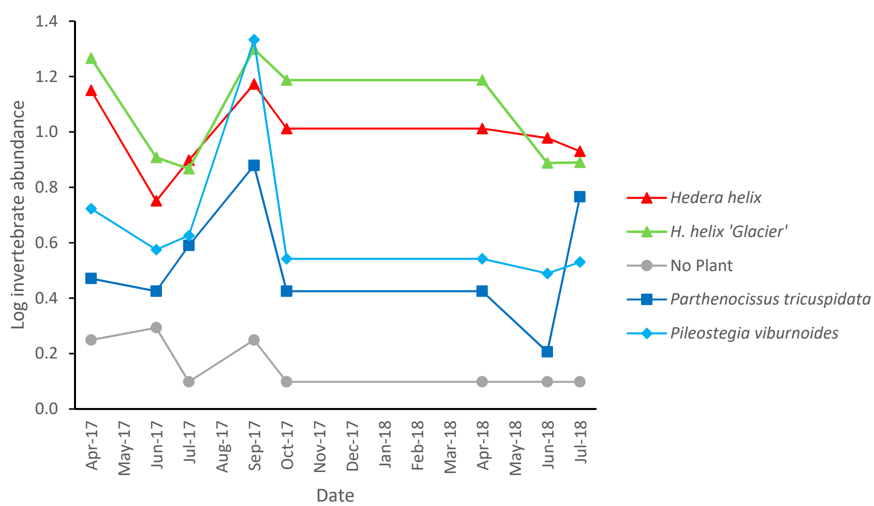

Fig. 2.2. Mean invertebrate abundance (Log) recorded by Vortis suction sampler from green façades on experimental buildings in 2017 and 2018.

(parasitica) where there was no difference between the treatments. In most functional groups tested where there was a difference between the treatments, *P. viburnoides* gave similar results to *P. tricuspidata* with a 4% or less difference. The exceptions were detritivores where the percentage caught on *P. tricuspidata* was 6% lower than on *P. viburnoides* and Detritivore (Collembola) where the difference was 10%.

4. Discussion

Data from our 2-year experiment support the notion that green façades on buildings provide resources for invertebrates, and that the more vegetation there is (i.e. the greater the vegetation depth) the greater the abundance supported. This is to be expected as the positive response of invertebrate abundance to increasing vegetation resource seen in our results is well known; for example it has previously been observed from green façades in Washington DC, USA and Paris, France (Madre et al. 2015; Matt et al. 2015), as well as in experimental plots, gardens and the wider environment (Salisbury et al. 2017; Smith et al. 2006; Dennis et al. 1998). This adds further evidence that supporting invertebrate abundance is a clear benefit of green façades in addition to bird life (Chiquet et al. 2013), improved thermoregulation, moisture retention, sound dampening and economic value (Perini et al. 2011; Cameron et al. 2014; Collins et al. 2017; Thomsit-Ireland et al. 2020; Manso et al. 2021; Jones et al. 2022).

Uniquely, this experiment demonstrates that invertebrate abundance is determined in part by plant species used as green façades, with some species supporting significantly higher abundance than others. Here we have shown that common ivy (*Hedera helix*) supported a greater invertebrate abundance than *Parthenocissus tricuspidata* and *Pileostegia viburnoides.* There are many other plant species grown as green façades including climbing species in the genera *Actinidia, Campsis, Clematis, Hydrangea*, *Jasminum, Lonicera, Pyracantha, Vitis* and *Wisteria* and many others that may be suited and others not yet considered (Köhler, 2008; Chiquet et al. 2013). It is noteworthy that a combination of two ivy taxa (*H. helix* 'Glacier' in combination with a straight species *H. helix*) showed a higher invertebrate abundance compared to *H. helix* alone. This hints that a mixture of plants may support a greater abundance than a single cultivar or species, as has been shown for example in grassland and garden (backyard) habitats (Dennis et al. 1998, Norton et al. 2019; Sperling and Lortie (2010)). A further consideration is that *H. helix* is the only plant in this experiment considered native to Britain (Stace, 2010) and therefore possibly supported more invertebrates. For garden plants, the amount of vegetation (rather than region of origin) has been shown to have the greatest effect on invertebrate abundance, however 'nativeness' does also have influence (Salisbury et al. 2017).

Invertebrate abundance is one benefit of vertical greening and other

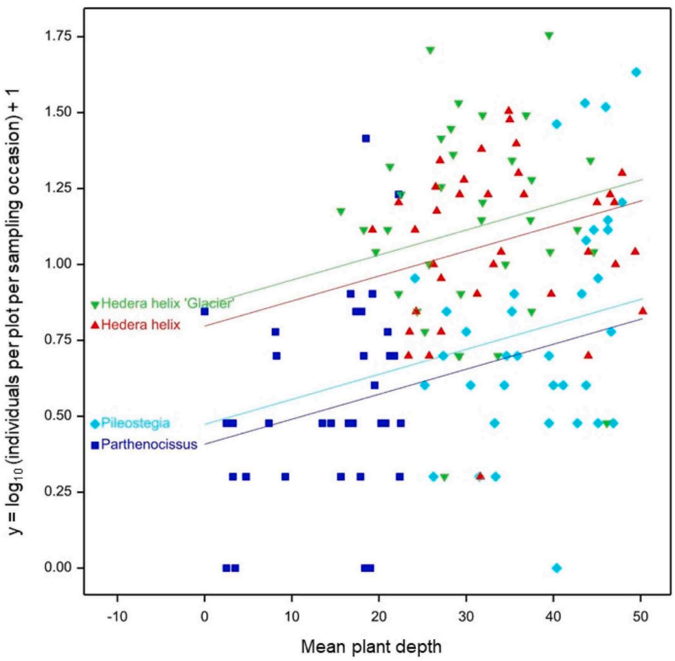

Fig. 3. Dependence of total abundance of invertebrates (excluding Hymenoptera: Formicidae) on vegetation depth. Observed data and fitted regressions for total abundance ($n = 1556$), over all sampling occasions combined. *Hedera helix* red upward triangles, *H. helix* 'Glacier' green downward triangles; *Parthenocissus tricuspidata* dark blue, squares; *Pileostegia viburnoides* light blue diamonds. Estimated intercepts and slope are shown in Table 1. The fitted regressions differ ($F4, 139 = 16.65$, $P < 0.0001$).

green infrastructure but it should not be considered in isolation and ideally decisions on planning and planting should not be based on single-issue factors (Jones et al. 2022). So although any plant used may increase invertebrate abundance other factors such as thermoregulation and pollution reduction effects (e.g. particulate matter) (e.g. Köhler 2008; Perini et al. 2011; Thomsit-Ireland et al. 2020) should be considered. In the case of the plants used in this experiment multiple benefits are known. Prior measurements on the same experimental set up (Thomsit-Ireland et al. 2020) indicated that *Hedera* had the greatest thermoregulation effect compared to the other species, although all plants reduced daily variation in relative humidity. Therefore the invertebrate abundance information presented here further supports *Hedera* as having greater benefits than other plants in this experiment. Use of *Hedera* however requires some practical considerations and education of building and landscape professionals. Ivy is battling the stigma of damaging wall surfaces and that can in some situations be the case (due to the strong attachment of ivy aerial roots onto materials such as damaged plaster) (e.g. Viles et al., 2011). But there are ways in which either ivy can be managed well, for example through the application of colourless deterrent paints onto wall surfaces which prevent or weaken ivy attachment (Thomsit-Ireland et al., 2016). Also, by planting ivy around buildings constructed of materials known to be unsusceptible to damage by aerial roots (e.g. solid bricks or concrete) or by using *Hedera* species known to produce a weaker attachment (e.g. *H. hibernica* as opposed to *H. helix*) (Thomsit-Ireland et al. 2016).

This research was carried out in south east England and applicability to other parts of the world and plants used there would benefit from investigation, in addition to suitability of plants as climate change affects growing conditions. Other vertical greening systems would also benefit from investigation, for example, it is possible the results do not apply to living walls which are either modular or continuous but typically not rooted in the ground (Francis and Lorimer, 2011, Peirini at al. 2011, Manso et al. 2021). There are also indications that different vertical greening systems support different assemblages of invertebrates (Madre et al. 2015). It is noteworthy that the direct green façade system used here may be the most economically sustainable (Perini and Rosasco, 2013).

4.1. Sampling methods and experimental limitations

The limitations of using small model buildings are discussed in detail by Thomsit-Ireland et al. (2020). In their view, although the small buildings may have overestimated the impact of planting on thermoregulation, the replication in approach enabled a comparison and sense of direction of how plant species differ when applied as a form of vertical greening. This can also be applied to invertebrate abundance. Whilst elements of scale (height and width of walls and corresponding plant cover), would benefit from additional research the replicated nature of this work gives a comparative indication of the value of green façades for invertebrate abundance. With invertebrate abundance there are also additional elements that should be considered, including wall green façade size (leaf area/canopy density), direction the walls face (aspect), connectivity (e.g. distance from other buildings) and management (Jones et al. 2022). Here we have shown that even small model buildings with green facades can support a range of invertebrates and results such as those of Matt et al. (2015) indicate that larger walls will support greater abundance.

Suction sampling is an established method of recording invertebrates from plant material and has proven to be effective for sampling vertical green surfaces (Madre et al. 2015). The use of the Vortis suction sampler in a manner that does not maximise the equipment's efficiency, such as in this experiment, were discussed in Salisbury et al. (2017). In summary, provided a measure of vegetation structure is included as a covariate it is possible to compare the abundance of invertebrates between treatments. Total abundance however, will be underestimated as whilst most invertebrates are collected in the first few seconds, not all are captured (Brook et al. 2008; Sanders and Entling, 2011; Zentane et al. 2016).

4.2. Responses of invertebrate functional groups to plant treatment

Abundance of the functional groups herbivores, predators and detritivores showed similar patterns to total invertebrate abundance, with significantly higher abundance on the *Hedera* treatments compared to *P. tricuspidata* and *P. viburnoides.* We can assume that these groups were deriving greater benefit from *Hedera* wall cover. The results indicate these are likely driven by plant resources (measured as plant depth) but it is possible they may have been affected by food (e.g. fungal hyphae or algal growths) or other resources such as plant structural differences between the plant treatments which were not quantified by the methodology used (e.g. attached but senesced leaves on *Hedera*). These results are similar to those found in previous works (Ballard et al. 2013; Salisbury et al. 2017). With herbivores, the primary food resource is likely to be available living plant material, and for predators invertebrate prey and for detritivores the amount of dead plant material. It is noticeable however, that detritivores overall had a very shallow slope and that the subgroup Collembola (79% of the detritivores) had a negative slope indicating decreasing abundance with increasing vegetation. The diet of Collembola species found on plants consists of fungal hyphae, spores or decaying vegetation with pollen grains, unicellular algae and moss (Christiansen (1964); Hopkin, 2007). Given this, it is unclear why their abundance decreased as amount of vegetation increased and future investigations with these invertebrates would benefit from gaining additional information on the availability of these resources. Predators (Araneae) also gave a negative slope (decreasing spider numbers with increased vegetation) as similar results to Salisbury et al. 2020); further investigation is required into spider requirements and densities especially as the habitat and resource needs of most spiders remain unknown (Bell et al. 2001). There were no differences between the plant treatments in the abundance of the other subgroup of the predators, Hymenoptera (parasitica); however, they followed the general pattern of increased abundance with increased vegetation. This is still likely to be due to the presence of more food resource (more invertebrate prey, honeydew or nectar for adults). It is possible that some of these patterns are a result of relatively low invertebrate numbers caught in this experiment (<200 for some functional groups). Further identification to species (beyond the scope of this work) may also provide insights, if information is available on precise biological requirements of each species. For example, Madre et al. (2015) found that the spider and beetle fauna associated with *P. tricuspidata* as a green façade was largely associated with hot dry xerothermophilous habitats similar to cliffs.

The lower numbers of herbivores on *P. tricuspidata* compared to *P. viburnoides* at similar levels of vegetation depth would benefit from winter measurements to elucidate. The only deciduous plant in this experiment was *P. tricuspidata;* perhaps as a consequence herbivore numbers may have been low at the beginning of each growing season before it was recolonised, whilst evergreen plants retained greater numbers of herbivores throughout the winter essentially starting the new season at a higher level.

5. Conclusions

The experiment has shown that green façades can provide a resource for plant-inhabiting invertebrates from a range of primary functional groups including herbivores, predators and detritovores. This indicates that this type of greening can support a good ecological balance of invertebrates, and therefore the function of these green façades; for example predators keeping herbivores in check and detritovores in nutrient recycling and avoiding the build-up of dead material. Invertebrate abundance will also support other parts of the food web such as birds (Chiquet et al. 2013). In addition, the greater the amount of vegetation, the higher the abundance of all the invertebrate functional groups, regardless of plant species. Invertebrate abundance is a good representation of diversity (Salisbury et al. 2020) therefore the value for biodiversity of green façades should be considered as an additional positive effect of this type of vertical greening. This will help private and public building owners maximize their space for wildlife as well as designers and developers to meet requirements such as those for the UK's biodiversity net gain strategy (GovUK, 2023) and local planning level to green neighborhoods (Aretis et al. 2015). Previous work has indicated that as with other ecosystem services (e.g. building cooling and insulation, Thomsit-Ireland et al. 2020), plant selection has an effect on the extent biodiversity support that is provided. Whilst this study only compared three plant types, it was clear that common ivy (*Hedera*) supported higher invertebrate abundance than *Parthenocissus tricuspidata* and *Pileostegia viburnoides*. Additionally, ivy grown as a mix of two taxa supported a higher abundance of invertebrates, indicating that plant mixes may provide greater resource. Establishing value in real life situations with case studies of existing and newly grown green façades could also provide insights that may help convince architects and planners to use these more widely. Whilst further experimentation and application is required into more plant species, plant mixtures, native versus non-native and deciduous versus evergreen plants, it is clear that green façades can add to the value of invertebrate abundance on buildings and that careful plant choice is a strong determining factor to that value.

CRediT authorship contributions statement

Salisbury A. designed and carried out a majority of the field and lab work (all invertebrate collection and identification) carried out analysis and did a majority of the work drafting the manuscript. Blanusa T. advised and was involved practically in all aspects of experimental design and management of the work in addition contributed significant amount text to the manuscript. Bostock H. provided significant horticultural insight and practical knowledge improving the application of the results and conclusions. Perry J. was a statistical consultant advising on appropriate design analysis and presentation of data and results and overview of the manuscript.

Author statement

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Declaration of Competing Interest

This work was entirely funded by the Royal Horticultural Society and there are no competing interests.

Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.ufug.2023.128118.

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