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Alkaline habitat for vegetated roofs? Ecosystem dynamics in a vegetated roof with crushed concrete-based substrate

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ABSTRACT

Using local and recycled materials is a sustainable way to establish a vegetated roof. In order to understand how the roof ecosystem functions and returns ecosystem services, it is important to study vegetation, soil organisms and runoff quality. We established a vegetated roof experiment based on a substrate containing lightweight crushed concrete, an alkaline side product from a concrete factory, mixed with compost. This five-year experiment in southern Finland tested how planting method (pre-grown vegetation mats vs. pot planting), compost content (20% vs. 40%, fresh volume), and substrate depth affect the cover and diversity of plants, the abundance of soil animals and the quality of runoff. Although the substrate had a high pH (7.3–11.8), many vascular plants were able to survive and establish viable populations. The planting method had a strong effect on plant diversity and the cover of individual species because the vegetation mats became dominated by the invasive, non-native *Phedimus hybridus*. Establishment with pot plants in turn provided bare ground that was colonised by spontaneous non-invasive species. This resulted in higher diversity, and a more even distribution of species. The amount of compost had only a weak impact on vegetation, whereas high pH generally reduced plant abundance and diversity. The concentrations of total phosphorus and total nitrogen in runoff were low as compared to values reported from many other vegetated roofs, were not affected by compost content and decreased over time. In summary, the high-pH substrate based on recycled materials is an environmentally responsible choice, suitable for a wide variety of plants, even rare and endangered species.

1. Introduction

Vegetated roofs, also called green roofs, are multifunctional nature-based solutions (NBS). They are promoted globally in cities as part of green infrastructure to offer solutions to problems such as flooding, urban heat and loss of biodiversity. Vegetated roofs can reduce and detain urban runoff, alleviate urban heat via evapotranspiration, abate noise and air pollution, sequester carbon and offer habitat for species (Berndtsson, 2010; Oberndorfer et al., 2007). However, there is still limited understanding on how to design these constructed ecosystems to promote optimal function and structure while being truly sustainable (Bozorg-Chenani et al., 2015; Lundholm, 2015). There are many factors that may cause a substantial environmental footprint. For example, vegetated roofs are often established using pre-grown vegetation mats and several artificial layers made of virgin materials, which all may be transported for long distances. Furthermore, vegetation mats may harbour non-native species (Páll-Gergely et al., 2014) and are also heavy to transport, resulting in high fuel consumption (Bozorg-Chenani et al., 2015; Nurmi et al., 2016). Mats are also fertilised during production to ensure dense vegetation cover (Emilsson et al., 2007; Mitchell et al., 2017). Leaching of nutrients, especially phosphorus (P) in the growing substrate, or from fertilizers, is a serious matter of concern (reviewed in Li and Babcock Jr., 2014; Mitchell et al., 2017; Jennett and Zheng, 2018). Indeed, the yearly nutrient load per unit area from vegetated roofs can exceed the load from agricultural systems, especially in terms of P export (Kuoppamäki and Lehvävirta, 2016; Mitchell et al., 2017). Thus, there is a need to find ways to establish sustainable vegetated roofs by using local, recycled materials that can

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retain nutrients.

Research on substrate components and plants for roofs, while improving the quality of runoff, is still relatively scanty (Vijayaraghavan, 2016). One way to improve P retention in vegetated roofs, or any NBS, could be to use substrate amendments, such as limestone, biochar or concrete (Erickson et al., 2007; Egemose et al., 2012; Kuoppamäki et al., 2016). P chemistry in soil and substrate solution is largely controlled by Ca, Fe and Al soil concentrations, which are closely related to soil pH (Jennett and Zheng, 2018). Concrete aggregates generally have a very high pH and Ca content and thereby a good capacity to absorb P. Egemose et al. (2012) suggested concrete as an effective and cheap residue from demolition sites to treat runoff from urban and agricultural areas in infiltration ponds and showed that the release of heavy metals was negligible. Thus, recycled concrete aggregate could offer an environmentally responsible roof substrate component (Bates et al., 2015a).

While concrete as a substrate component could be part of the solution to nutrient leaching, it is not clear whether it can support diverse vegetation due to its low moisture holding capacity (cf. Bates et al., 2015a) and high pH. To improve the moisture holding capacity of a substrate, concrete could be mixed with organic materials, such as compost (Bates et al., 2015a) that, however, may contain high amounts of soluble nutrients. It remains to be explored whether nutrient leaching from compost can be sufficiently prevented with concrete. Finally, even though different alkaline and sustainable substrate components are available, our understanding of the actual pH tolerance of many wild plant species is limited. Nevertheless, it is generally understood that plant species are differently adapted to soil pH and that in alkaline soil nutrients are not easily available to plants (e.g. Calvo-Polanco et al., 2017).

Calcium-rich, concrete-based substrate may offer possibilities for conservation of calcicole plants by creating additional habitats on vegetated roofs. Calcareous meadows are important for threatened vascular plants and are categorised as critically endangered habitats in Finland (Hyvärinen et al., 2019; Kontula and Raunio, 2018). Furthermore, spontaneously colonising species may contribute to the biodiversity of vegetated roofs (e.g. Aloisio et al., 2019; Lundholm, 2015), especially where bare patches offer open microsites for spontaneously arriving species. On the other hand, vegetated roofs may have potential to facilitate harmful biological invasions (e.g. Kinlock et al., 2016).

The main aim of our five-year experiment was to investigate whether lightweight crushed concrete, mixed with various amounts of compost, can be used as substrate on vegetated roofs, in terms of plant establishment and runoff quality. We also examined several substrate characteristics, such as organic matter content (OM) and the leaching of nutrients and metals, as well as the density of nematodes and enchytraeids. These animals are important for soil processes, including OM decomposition and nutrient cycling (Nielsen, 2019; Sulkava et al., 1996), and have been suggested as suitable bioindicators for soil health and quality (Pulleman et al., 2012).

In Fig. 1, we portray all our a priori assumptions and hypotheses regarding the relationships between the system components. Our hypotheses were as follows:

1) Increasing compost content increases plant cover and diversity while high pH may have a negative impact on most species. Planting method and substrate depth have species-specific effects, and therefore uniform hypotheses across the dataset are not possible. Finally, the amount of open ground increases the probability of spontaneously arriving species to get foothold. Therefore pot planting that results in more bare ground in the beginning than the dense pre-grown mats, also allows more biodiversity.
2) The leaching of nutrients in runoff is lower from the concrete-based substrate than from other substrates reported in the literature, is lower with less compost and decreases with time. We had competing hypotheses for the effect of planting method on nutrient leaching, implying different mechanisms: higher leaching of nutrients from the mats as compared to pot planting would indicate a surplus fertilisation during production that exceeds the uptake by plants, while lower leaching from the mats would indicate that high vegetation cover at planting is essential to decrease leaching.

2. Materials and methods

2.1. Study site and experimental design

We established a vegetated roof experiment in October 2013, just before the first snow, and monitored it for 5 years. The study site is located in the municipality of Hollola, southern Finland (N 60° 59′ 12.47″, E 25° 24′ 38.53″), on a rooftop (slope 1:48) of a concrete factory. 12 strips of recycled fabric were laid out as the bases of the experimental strips to provide friction for the substrate and to retain moisture on the roof (Figs. 2 and 3). The 11 m long and 1.6 m wide strips were 0.8 m apart, with a 1.7 minimum distance from the roof edge. The spaces between the experimental strips were filled with gravel (grain size 16–32 mm). Each strip was then covered with a 10 cm bed of reed (*Phragmites australis*) to serve as a drainage layer. Experiential knowledge suggests that it could serve as an environmentally responsible underdrain on vegetated roofs, instead of the commonly used moulded polystyrene. The reed originated from the nearby Lake Vesijärvi, where it is cut yearly as a restoration measure.

Two substrates were randomly assigned to the experimental strips. The substrate of six experimental strips contained 40% *v*/v garden waste compost, and that of the other six strips 20% v/v (hereafter, referred to as Mix40 and Mix20). Both mixtures contained 5% crushed spruce bark. The remaining 55% in Mix40 and 75% in Mix20, consisted of lightweight crushed concrete, a high-pH factory by-product containing expanded clay aggregate that was sieved through 20 mm. All strips were divided into an upper and lower part based on the support capacity of the roof, with 10 and 8 cm target substrate depths, respectively.

Vegetation was established with imported pre-grown vegetation mats (hereafter "mats") and local potted plants (hereafter "pots"), each randomly applied to six of the experimental strips. The mats included a 3 cm thick substrate, thus the depth of the experimental substrate (Mix20 or Mix40) below the mats was adjusted to achieve the total substrate depth of 10 or 8 cm. Pot strips received 668 plants (pot size 9 × 9 × 8 cm), located randomly in 28 × 4 grids with 40 cm distances between the plants (except the last strip had four plants less than the others). Both mat and pot strips received a complementary seed mixture (Appendix A, Table A.1). Each combination of treatments, two levels of compost content x two planting methods, was replicated three times.

Our aim was to study the success of plant species that are in decline and tolerant to high pH and calcium (Hyvärinen et al., 2019). However, our choice of test species was limited by the availability of plants. As the actual pH tolerance of wild species is often unknown, we also accepted species that are not listed as tolerant to high pH. For example, knowledge regarding *Antennaria dioica* and *Dianthus deltoides* is miscellaneous and suggests tolerance of low or high pH (e.g. Ellenberg et al., 1991; Hill et al., 1999). Thus, our experimental plants represented a wide variety of pH tolerances, as reported by practitioners and researchers.

The species planted in the vegetation mats (17 vascular plant species and 3 bryophytes), pot plants (14 species) and seeds (5 species in mats, 7 in pot strips) are listed in Appendix A, Table A.1. Two more species were added as pot plants in 2015, *Astragalus alpinus* ($n = 8$) and *Oxytropis campestris* ($n = 10$), on both mat and pot strips. Of the 32 planted vascular plant species, 14 were included in both mat and pot strips. Four of the species are listed as declining or threatened in Finland: *A. dioica* (near threatened, NT), *Dianthus arenarius* (endangered, EN), *D. deltoides* (NT) and *Galium verum* (vulnerable, VU) (Hyvärinen et al., 2019). In 2014, we weeded *Chenopodium album*, to

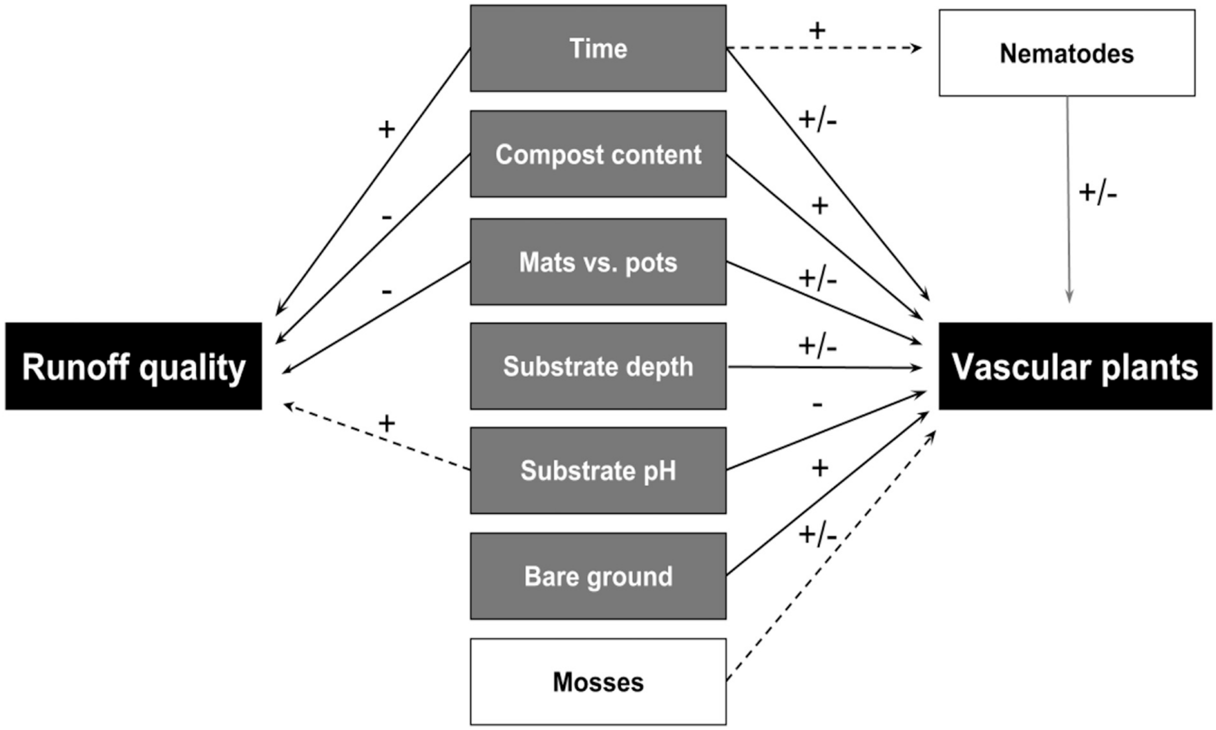

Fig. 1. A priori hypotheses regarding the relationships between the components of the vegetated roof ecosystem experiment in terms of runoff quality and the abundance and diversity of plants. +/− indicate positive and/or negative impacts of the studied factors (grey and white boxes) on the response variables (black boxes). Factors in white boxes or relationships indicated with dashed lines were monitored but not included in statistical models.

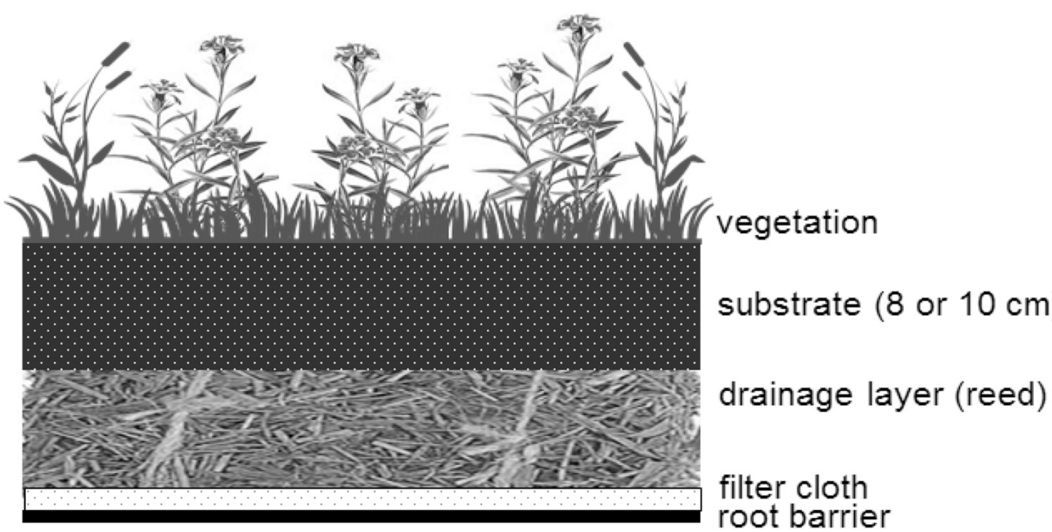

Fig. 2. The layers of the experimental vegetated roof. As there was no filter fabric between the substrate and the drainage layer, in reality they were partially mixed.

avoid it becoming dominant. After that, occasional individuals of *Epilobium adenocaulon, Epilobium ciliatum*, *Erigeron canadensis* and *Taraxacum* spp. were removed.

To enable runoff sampling from each strip, in 2014 they all were equipped with an individual gutter at the edge of the roof, where runoff was diverted via 12 separate downpipes into 200 l containers (Fig. A.1). Despite the 0.8 m gravel-covered space between the strips and the 1:48 slope, in 2015 we noticed that there was no difference in the water quality between Mix20 and Mix40. To ensure no sideward movement of water, a plastic wall was mounted in the middle of the gravel between the strips in summer 2016. As there still was no difference between the two Mixes, we assumed that sideward movement of water was negligible and that runoff drained into the containers as planned. Thus, we included runoff data from before 2016 in the analysis. When accounting for the space between the experimental strips, the “catchment” of one strip comprised of 17.6 m^2 vegetated roof area and 14.1 m^2 gravel area i.e. in total 56% of each “catchment” was vegetated.

2.2. Substrate quality and soil animals

We carried out an a priori quality check for hazardous leachates in the substrate and found that the leachate from the substrate did not contain too elevated concentrations of trace elements (Appendix B). When establishing the experiment, substrates were randomly sampled to have 1000 ml composite samples from both mixtures. These samples were treated in a RETSCH sieve shaker (AS 200 basic). The grain

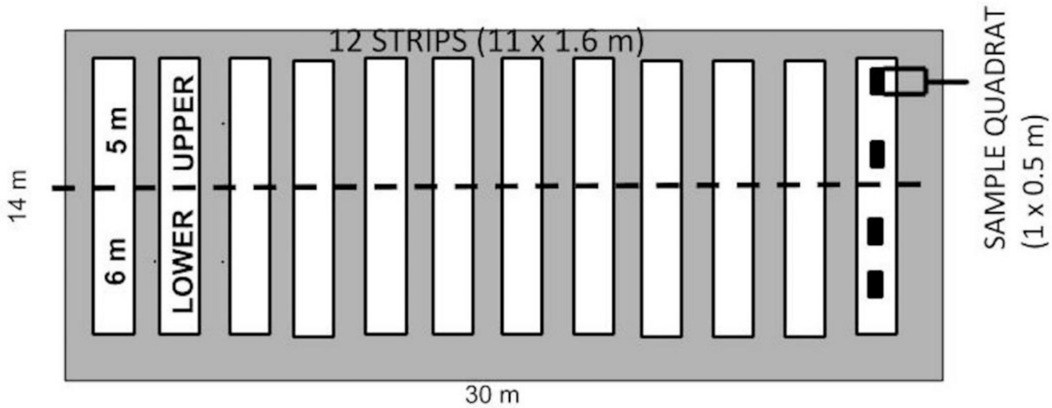

Fig. 3. Set-up of the vegetation and substrate depth inventories on 12 experimental strips (11 m × 1.6 m). The target substrate depth was 10 cm on the 5 m long upper part of the strip and 8 cm on the 6 m long lower part. Vegetation was inventoried in four randomly located sample quadrats (0.5 × 1 m) on each strip; two on the upper and two on the lower part of the strip.

fractions were weighed to obtain the particle size distribution, which in Mix20/Mix40, was 1/0% < 0.5 mm, 35/10% 0.5–2.0 mm, 19/30% 2.0–4.0 mm and 45/60% 4.0–20.0 mm. WHC was determined by subtracting the weight of oven dry (105 °C, 24 h) samples from water saturated substrate samples (n = 3). pH was measured after extraction with 1:5 (v/v) soil:distilled water (ISO 10390 standard, pH meter WTW inoLab pH 720) (n = 2) and soil organic matter content (n = 3) as the loss on ignition (LOI, 4 h, 550 °C; Radojević and Bashkin, 2006). At the beginning of the experiment, the pH of the Mix20 and Mix40 were 11.7 and 11.0, and LOI% for Mix20 4.4% and Mix40 6.1%. In the mat treatments, pH was measured separately from the substrate of the mats and from the crushed concrete substrate and the mean pH was weighted by the thickness of the two layers. According to the manufacturer, the density of lightweight concrete is on 800–1200 kg/m^3. Porosity was not possible to measure exactly, but just by visual inspection the product is highly porous (see photo in Appendix B).

To analyse the density of nematodes and enchytraeids, three soil cores (depth 6 cm, diameter 3 cm) were randomly collected from each strip in 2016 and 2017. In the mat strips, the upper substrate layer and layer below the mat were sampled separately. Samples per strip were pooled, mixed properly, and nematodes were extracted from ca. 10 g and enchytraeids from ca. 30 g of fresh, non-sieved substrate using the wet funnel methods by Sohlenius (1979) and O'Connor (1955), respectively. The number of nematodes and enchytraeids was counted under microscope with 10–40 magnification.

2.3. Runoff

Samples for runoff quality were collected twice in 2014 and once a year in 2015–2018 from the discharge from each experimental strip (see 2.1.). pH was determined using Metler Delta 340. Total nutrient concentration was determined after oxidising in an autoclave at 120 °C for 30 min. Total phosphorus (TP) was measured spectrophotometrically using the molybdate blue method (ISO 6878: 2004). High Performance Liquid chromatography (HPLC) was used to determine total nitrogen (TN) with 0.04 M NaCl as an eluent (ISO 29441:2010). Samples collected in June 2017 were analysed also for PO_4 and dissolved metals (analysis described in *2.2.*). In addition, runoff from adjacent bitumen roof and rainfall were sampled for pH determination in 2017.

2.4. Vegetation inventories

At each experimental strip, vascular plants were identified using stratified random sampling in September 2014 (only mats), August 2016 and August 2017 (see Appendix, Fig. A.2). Every study year, two 0.5 m^2 sample quadrats were randomly positioned on the upper and two on the lower part of the strip), each quadrat at least 1 m apart from another one (Fig. 3). Transversally, the sample quadrats were located in the middle of the strip. All vascular plants were identified to species level, except *Hieracium* and *Taraxacum* that were identified to genus level. The nomenclature follows Hämet-Ahti et al. (1998) and FinBIF (2019).

The percentage cover was estimated for each species, the moss layer and bare ground (scale: 0.5, 1, 2, 3…10, 15, 20… 90, 91, 92… 99, 100%). To complement the findings in the quadrats, we searched the rest of the strip for species (presence only). In 2014, only species presence (in pot strips as survival of each planted plant and in mat strips as presence of species in sample quadrats) was determined because cover estimates in four sample quadrats would not give a representative view of the newly established pot strips (with distance between pot plants being 40 cm). Substrate depth was measured each year using a metal stick piercing into the substrate at five points, at the centre and in each corner of the sample quadrats.

2.5. Weather

Weather data for 2014–2017 was obtained from the Finnish Meteorological Institute (2019) from a station in the city of Lahti, 13 km from our research site. Summer (June–August) 2014 mean air temperature was 0.5 °C warmer than the long-term (1981–2010) average 15.5 °C. Summers 2015 and 2017 were 1.1 °C and 1.4 °C colder than average, respectively, while summer 2016 was average (15.5 °C). The summers of 2014–2016 were less rainy than average (222 mm precipitation), with total rainfall varying from 170 mm to 211 mm, while the cool summer of 2017 was more rainy with 237 mm precipitation.

2.6. Data analyses

2.6.1. Runoff data

We used LMM (lme4 package in R; Bates et al., 2015b) to estimate the effects of planting method, substrate composition, time, and their three-way interaction, on runoff pH, TN and TP. Note that interaction with time will allow for different responses in different strips through time, and thus even tackled the unlikely theoretical event of water mixing from adjacent strips into the containers the first year, before mounting the solid waterproof separation of the strips. TN data was log-transformed to improve normality. Time was measured as the number of days since the beginning of the experiment and it was added to the models as a continuous variable. Also the quadratic term of time was added to the model to allow non-linearity. The strip ID was included in the models as a random term to account for repeated measures. We used backward stepwise model selection based on p- and AIC-values to simplify the models (see 2.7.1). Due to the complexity of the full model, we also conducted LMMs with only one main explanatory variable at a time. In addition, we ran the models with repeated measures ANOVA to examine the robustness of the results of our full LMM. The results were similar, thus we show the results only for the full models.

2.6.2. Vegetation data

We used R version 3.5.2 (R Core Team, 2018) to perform all statistical analyses. To give a comprehensive overview of the vegetation development, we determined the frequency and relative frequency of each species based on all the species found in the strips (Appendix A, Table A.2). We also present species' frequency, relative frequency and average cover (%) in the sample quadrats in 2016 and 2017 (Appendix A, Table A.3). Furthermore, we calculated species richness (S), Shannon-Wiener diversity index (H′), Simpson diversity index (1-D) and Pielou's evenness ($J = H'/\ln(S)$) per sample quadrat (vegan package; Oksanen et al., 2018). For these values, we tested the differences of means among years and between planting methods with paired *t*-test or with non-parametric Wilcoxon test if the variable was not normally distributed.

To evaluate species' performances, we classified them into calcicoles and non-calcicoles, i.e. those specifically associated with calcareous soils and those not (Kontula and Raunio, 2018; NatureGate, 2019). Furthermore, we categorised the species based on Ellenberg values for soil pH into those found in mainly acid soils with Ellenberg values 1–3, moderately acid to moderately basic soils with values 4–6 and in basic soils with values 7–9 (Ellenberg et al., 1991; Hill et al., 1999). If multiple Ellenberg values were reported, we used the maximum value.

For community level analysis, we used non-metric multidimensional scaling (NMDS) with Bray-Curtis dissimilarity index (vegan package; Oksanen et al., 2018). We included year, planting method, compost content, substrate depth, substrate pH and bare ground, fitted them onto the ordination, and assessed their statistical significance based on permutations. We ran NMDS for combined 2016 and 2017 data, to determine whether sample quadrats differed between the years plus performed separate ordinations for both planting methods.

We used generalized linear mixed models (GLMM, MASS package in R; Venables and Ripley, 2002) to determine the factors affecting diversity indices and the cover of individual species across 2016 and 2017. Explanatory variables were the same as above, with the interactions of year*substrate depth and year*compost content. Neither mosses nor nematodes were included as they correlated with other predictors. Sample location nested within the strip's upper or lower part was included as a random factor to account for repeated measures and spatial dependence of the data points. The error distributions in the models were Poisson for species richness, and quasibinomial for evenness, Simpson's diversity index and species covers. For the Shannon-Wiener index we used a linear mixed model (LMM) with normal error distribution (normality tested with Shapiro-Wilk test). Model validation for all models was conducted with residual plots. In addition, obvious outliers were omitted from the data. No random effects were included in the graphs showing the predicted values. We performed backward stepwise model selection to simplify the models based on *p*-values using the following procedures:

- First, the interaction term with the highest p-value $> .2$ was removed and removal of interaction terms continued until only interactions with p-values $\leq .2$ remained.
- Then, pH and the amount of bare ground were removed step-wise if their p-values were ≥ 0.2, starting with the variable with the highest p-value.
- Finally, planting method, year, compost content and substrate depth were removed if their p-values were ≥ 0.2 and if they were not included in any retained interaction, starting with the variable with the highest p-value.

We individually tested those 13 species that occurred in at least 30 sample quadrats and at least four strips. Covers of species that did not meet these criteria were pooled into three groups: planted, spontaneous non-native invasive (called simply invasive hereafter) and spontaneous non-invasive species (hereafter spontaneous) (Finnish Invasive Alien Species Portal, 2019; Nobanis, 2018; Norwegian Biodiversity Information Centre, 2018). We considered a species invasive if it was categorised as invasive or potentially invasive in Scandinavia or the Baltic countries in any of the sources.

For three species, we excluded strips with zero observations and reran the analysis, because the model failed to converge or was biased. Due to the high number of explanatory variables in our full model, we also ran reduced models with only the main treatments, planting method and compost content, as explanatory variables, and year nested within the strip as a random variable. As the results of the reduced models supported the results of the full models, we only show the latter ones.

3. Results

3.1. Substrate properties

The mean pH of the substrate was 11.4 in 2013, prior to installation of the experiment, and by 2017 it had decreased by roughly 3 units (Table 1). Even though the substrate in pre-grown mats had a lower pH (7.7 ± 0.1) than the crushed concrete substrate below the mats (8.3 ± 0.2; measured in 2016–2017), the overall substrate pH did not differ between mat and pot strips (Table 1; $t = 1.64$, $p = .128$). There was no difference in the WHC (ca. 66%) of the two substrates nor between mat and pot strips (Table 1). The mean LOI% (across 2013, 2014, 2016; 2 outliers omitted from Mix20) was $5.7 \pm 1.6\%$ for Mix20 and $6.5 \pm 1.6\%$ for Mix40.

3.2. Soil animals

The number of nematodes per strip varied from 0 to 392 ind./g FW (i.e. 0 to 19,6 million ind./m^2, Table 1). The mean number of nematodes was ca. 45 times higher in 2017 than in 2016 with extremely high variation both between and within the planting methods (Table 1). Mats had 3.5 times more nematodes than pot strips but due to the high variation the difference was not statistically significant ($t = -1.82$, $p = .096$). In the mat strips ca. 75% of the nematodes occurred in the mat layer and only one fourth in the substrate below the mats. Enchytraeids were not found in any of the samples. The amount of compost had no impact on nematode number.

3.3. Runoff quality

Mean runoff pH varied between 8.1 and 8.5, with a significant curvilinear relationship with time (Table 2). Rainfall pH was 6.1 and runoff from the adjacent bitumen roof had pH 7.7. Runoff from strips with Mix20 and Mix40 had an average pH 8.13 and 8.08, respectively (Table 2). TP concentrations in runoff decreased over time, while TN concentrations first increased and then started to decrease (Fig. 4, Table 2). TP concentrations at the beginning of the experiment were 0.15–0.25 mg/l and decreased to 0.04–0.06 mg/l at the end while TN concentrations decreased from 2.5–3.9 mg/l to 0.9–1.0 mg/l. The initial total nutrient concentrations were quite close to dissolved NO_x and PO_4 concentrations of 5.6 and 0.3 mg/, respectively, according to the leachate test made of fresh substrate mixture before establishing the experiment (Table B.1). TN concentrations in runoff were higher from pot strips (mean across years = 2.83 ± 1.56 mg/l) than from mat strips (2.18 ± 1.14 mg/l). The amount of compost did not affect total nutrient concentrations (Table 2). In summer 2017 mean PO_4

Table 1
Summary statistics of substrate properties per year and planting method (pot vs. mat strips). The first values show mean ± SD and the second values below show min-max. Initial substrate was sampled prior to the experiment, whereas measurements in 2014–2017 were taken from substrate core samples of the experimental strips on the roof. Results for the planting method are calculated across the 2014–2017 datasets. WHC = water holding capacity, OM = organic matter.

		Initial	Year			Planting method	
	Variable	**substrate**	**2014**	**2016**	**2017**	**Mat**	**Pot**
Included in models	**Substrate depth cm****		10.3 ± 1.54 7.7–14.3	9.9 ± 1.5 7.4–14.3	10.1 ± 1.7 7.1–14.2	9.9 ± 1.4 7.4–13.6	10.1 ± 1.8 7.1–14.3
	Bare ground %**			3.4 ± 5.3 0–20	2.2 ± 3.3 0–15	0.4 ± 1.1 0–6	5.2 ± 5.2 0–20
	Substrate pH*	11.4 ± 0.4 11.0–11.8		7.8 ± 0.5 7.3–9.2	8.6 ± 0.4 7.6–9.0	8.1 ± 0.7 7.6–9.2	8.5 ± 0.4 7.3–9.0
Not included in models	**Mosses %** †**		73.5 ± 11.7 45–90	80.4 ± 13.2 3–90	25.7 ± 16.8 0–65	50.3 ± 34.5 0–90	55.8 ± 26.9 0–90
	WHC %*		66.7 ± 7.6 51–82			69.2 ± 9.4 51–82	64.2 ± 3.7 58–69
	OM %*	6.6 ± 1.9 4.4–9.7	6.8 ± 1.3 5.0–8.9	5.7 ± 1.7 3.6–9.8		5.63 ± 1.2 3.6–8.5	6.9 ± 1.8 3.7–9.8
	Nematodes ind./g FW*			1.2 ± 1.1 0.0–3.1	55 ± 91.1 1.6–391.5	73.8 ± 124.0 0.0–391.5	10.8 ± 14.5 0.3–46.9

* = strips, ** = sample squares, † = in 2014 studied only in mats.

concentrations in runoff were 0.08 ± 0.01 and 0.09 ± 0.02 mg/l in mats and pots, respectively. This means that ca. 63% of the P in runoff was in dissolved form.

Lead and cadmium concentrations in runoff samples taken in 2017 were below the level set for a reliable quantitation value (0.0006 and 0.0002 µg/l, respectively) in all samples. Mean (± SE) concentrations of other metals were: arsenic 3.7 ± 0.3, chromium 0.9 ± 0.1, copper 10.3 ± 0.7, cobalt 0.40 ± 0.04, nickel 2.9 ± 0.2 and zinc 3.6 ± 0.4 µg/l.

3.4. Vegetation

3.4.1. Success of planted species

In the inventories, we found 28 of the 32 planted or sown species (Appendix A, Table A.2), 15 of which were detected every year. The cover of planted species was generally higher in pot than mat strips (Table 3).

Of the 17 species included only in the pre-grown vegetation mats (Appendix A.1), five species increased in frequency, and spread to the pot strips: *Hylotelephium ewersii*, *Poa alpina*, *Phedimus spurius*, *Sedum album* and *S. sexangulare* (Fig. 5; Appendix A.2). Five species were sown as seeds in the mats, of which only *Thymus serpyllum* and *Viola tricolor* occurred after four years. The sown species had a greater success in the pot strips (Appendix A, Table A.1 and A.2; Fig. 5). The occurrence of most of the 16 pot plant species decreased over time (Appendix A, Table A.2), and five of them were not detected after the first year (Appendix A, Table A.2; Fig. 5).

In 2016 and 2017, *Festuca ovina, Sedum acre, P. alpina* and *S. sexangulare* were among the most abundant planted species, the first two species occurred mostly in pot strips, and the last two occurred mostly in mats (Fig. 5; Appendix A, Table A.3). In 2016, *T. serpyllum* was the most abundant planted species but in 2017 its abundance was low (Appendix A, Table A.2 and A.3). In Fig. 5, we list the species from those that failed to successful ones (for details see Appendix A, Table A.3).

75% of the 12 planted species categorised as calcicoles (Appendix A, Table A.2 and A.3) were present in 2016 and/or 2017 compared to 70% survival of the 20 non-calcicoles. *F. ovina, S. acre, S. album* and *T. serpyllum* were among the most successful calcicoles, whereas *Poa compressa* and *Centaurea jacea* were not detected after 2014. When averaged across the years 2016–2017, vegetation cover for planted calcicoles was higher (20%) compared to other planted species (12%) even though the initial amounts of planted calcicoles were clearly lower than the amounts of non-calcicoles. The mat strips were planted with less calcicole species (7) than non-calcicoles (12), only three of the eight seed-sown species were calcicoles, and the pot plants included 282 individuals of calcicoles vs. 404 non-calcicoles (Appendix, Tables A1 and A2). None of the three planted species associated with acidic soils were detected in 2016 or 2017 (Ellenberg 1–3; Appendix A, Table A.2 and A.3), whereas 82% and 72% of the species associated with neutral and alkaline soils still occurred.

3.4.2. Diversity and spontaneous species

Despite the highly alkaline substrate, altogether 71 plant species (*Hieracium* and *Taraxacum* identified to genus level) were found in the strips including 7 succulents, 5 grasses, 2 woody plants and 57 forbs

Table 2
The GLMM results regarding the effects of sampling time, planting method and compost content on pH and the concentrations of total phosphorus and total nitrogen in runoff from experimental strips during 2014–2018. Time is measured as days since the beginning of the experiment, including the quadratic term time^2. Only variables retained in the final models are shown ($p < .2$), with $p < .05$ bolded.

	pH			Total nitrogen (mg/l)			Total phosporus (mg/l)		
	EST	SE	p	EST	SE	p	EST	SE	p
Intercept	8.43	0.04	< **0.001**	1.43	0.07	< **0.001**	0.22	0.02	< **0.001**
Planting method	−0.02	0.04	0.628	0.17	0.07	**0.044**	0.04	0.02	0.124
Compost content	−0.09	0.04	**0.037**						
Time	−5.30E-06	2.91E-05	**0.008**	−4.16E-04	5.43E-05	< **0.001**	−9.61E-05	9.84E-05	< **0.001**
Time^2	0.32	0.13	**0.020**	−0.60	0.25	**0.018**	−0.08	0.04	0.079
Planting method * Compost content	0.11	0.06	0.059						

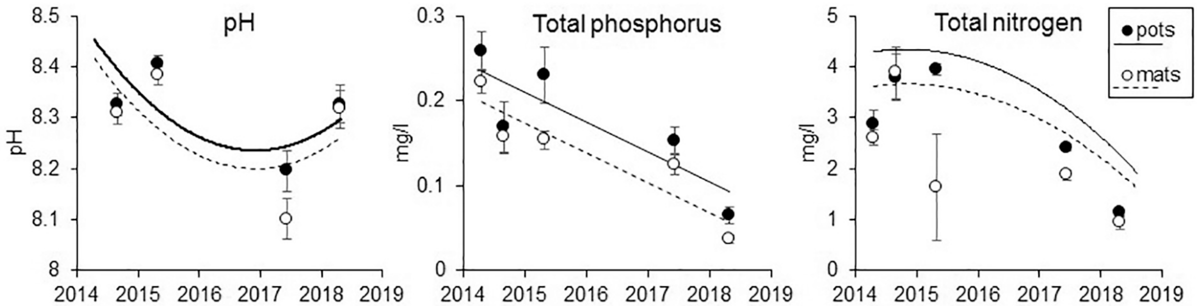

Fig. 4. Average (± SE) pH and concentrations of total phosphorus and total nitrogen in runoff from vegetated roofs established with pot plants or vegetation mats during the 5 years of the experiment (no sampling in 2016).

(Appendix A, Table A.2). The species number increased with time, and more species were identified in pot than mat strips (Table 3). Altogether, 44 species (59%) arrived spontaneously, but only seven of these occurred every year. Pot strips had more spontaneous species than mat strips (Table 3), the most common ones being *Trifolium repens*, *Cerastium fontanum* and *Vicia hirsuta* (Appendix A, Table A.2).

Six invasive species were detected. *Phedimus hybridus* had by far the greatest cover, especially in 2017 in mats (Appendix A, Fig. A.2, Table A.3), where it also negatively affected the species diversity indices (Table 3; Appendix A, Table A.2 and Table A.3). Other invasive species occurred in 79% of the sample quadrats, the most common ones being *Echium vulgare*, *Senecio viscosus* and *Epilobium adenocaulon* (Appendix A, Table A.2).

Species' categorisation based on their calcium affinity vs. Ellenberg pH values gave divergent results. Among the invasive and spontaneous species, *E. vulgare* was the only calcicole, but there were 28 species with high Ellenberg pH values. Most frequent of the latter were *C. album, Crepis tectorum, E. vulgare, Erigeron acris, P. hybridus* and *S. viscosus*.

3.4.3. Plant community structure

According to NMDS with mat and pot strips in 2016 and 2017, differences in species composition were mostly caused by the planting method ($r^2 = 0.707$, $p < .001$). The r^2 values of the other statistically significant factors were low: for year $r^2 = 0.076$ ($p = .002$; Fig. 6) and for substrate pH $r^2 = 0.055$ ($p = .073$). There was less bare ground in mat than pot strips ($r^2 = 0.333$, $p < .001$). Vegetation was much more uniform in the mat strips, dominated by *Phedimus* and *Sedum* species, mixed with *P. alpina* and *T. repens*. The pot strips were abundant with meadow species, e.g. *D. deltoides*, *T. serpyllum* and *V. tricolor*, together with *S. acre*.

The separate NMDSs for mat and pot strips showed that vegetation composition varied between 2016 and 2017 (pot $r^2 = 0.416$, $p < .001$; mat $r^2 = 0.191$, $p < .001$, Fig. 6). In the mat strips, substrate pH influenced the community structure ($r^2 = 0.208$, $p < .049$), while we found no effect of environmental variables in the pot strips. Especially *Erigeron canadensis*, *Pilosella officinarum* and *V. tricolor* were scarce at high pH.

3.4.4. The responses of plants to environmental variables

pH was an important explanatory variable for the cover of individual species, groups and species diversity – it was retained in 14 of the 20 GLMMs and had a negative coefficient in 12 of them (Table 4).

Table 3
Summary statistics of vascular plant species cover and diversity in the sample quadrats per study year and planting method (mat or pot strips). We compared the group means separately for year and planting method using t-test (and Wilcoxon test). Significant differences ($p < .05$) are bolded.

	Year	Planting method			
	2014	2016	2017	Mat	Pot
Total number of species					
Total nro species STRIPS	35	51	54	48	61
Total nro species QUADRATS		40	41	30	48
Species richness, mean ± SD and min-max					
All species		**11 ± 2.8**	**8.4 ± 2.4**	10.1 ± 3.4	9.3 ± 2.2
		7–18	2–8	2–10	2–8
Planted species		**6.7 ± 1.4**	**4.8 ± 1.4**	**6.2 ± 1.9**	**5.3 ± 1.3**
		4–9	2–8	2–10	2–8
Spontaneous and invasive species		4.5 ± 1.9	**3.6 ± 1.7**	3.9 ± 2.0	4.2 ± 1.8
		1–10	1–9	1–10	1–9
Spontaneous non-invasive species		3.4 ± 1.6	2.9 ± 1.7	**2.7 ± 1.7**	3.5 ± 1.6
		1–8	0–7	0–8	1–7
Invasive species		**1.1 ± 0.7**	**0.8 ± 0.5**	**1.2 ± 0.4**	**0.7 ± 0.8**
		0–3	0–2	1–2	0–3
Vegetation coverage, mean ± SD					
All species		68 ± 18	66 ± 26	**81 ± 12**	**53 ± 22**
Planted species		**39 ± 24**	**22 ± 17**	**20 ± 10**	**41 ± 26**
Spontaneous and invasive species		**36 ± 32**	**48 ± 34**	**71 ± 20**	**13 ± 13**
Spontaneous non-invasive species		**7 ± 7**	**12 ± 13**	**7 ± 7**	**12 ± 13**
Invasive species		**29 ± 29**	**36 ± 37**	**64 ± 18**	**1 ± 3**
Diversity indices, mean ± SD					
H´ (Shannon-Wiener)		1.35 ± 0.34	1.22 ± 0.48	**1.09 ± 0.36**	**1.47 ± 0.39**
D (Simpson)		0.58 ± 0.15	0.55 ± 0.20	**0.48 ± 0.15**	**0.66 ± 0.16**
Evenness		0.57 ± 0.15	0.57 ± 0.18	**0.52 ± 0.12**	**0.63 ± 0.16**

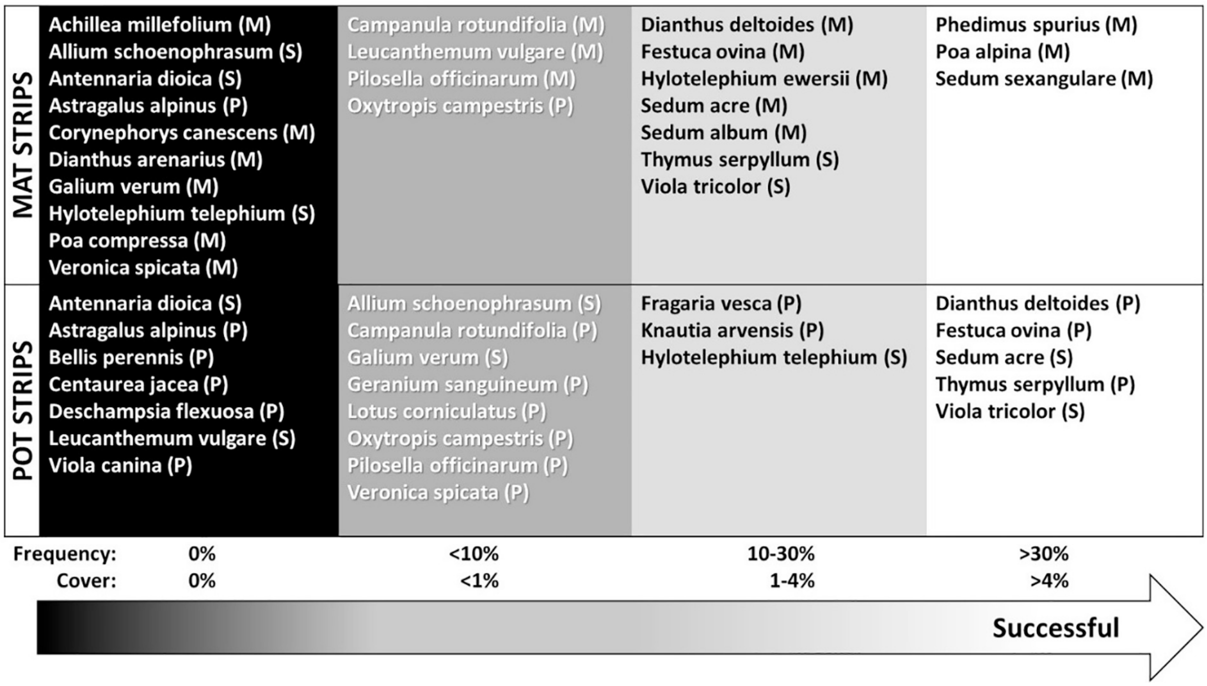

Fig. 5. All planted species are listed from failed to successful species based on their frequency and cover in 2016–2017. Species that were not observed in 2016 or 2017 in the sample quadrats were categorised as failed (black panel). The rest of species were categorised according to their frequency and cover, shown as percentage values below the panels. M = pre-grown in vegetation mats, P = planted as a pot plant, S = added as seeds.

Only *F. ovina* and *T. serpyllum* increased with increasing soil pH (Fig. 7). Although compost content was retained in 8 models, it was statistically significant only in 2 of them, so there was no strong overall effect of compost content (Table 4).

Planting method was retained in 13 models. Ten of the coefficients indicated higher cover or diversity in pot than mat strips. The amount of bare ground, retained in 12 models, was predominantly negatively associated with individual species (4 of the 5 significant models), and positively with the cover of spontaneous and invasive species and diversity indices (Table 4, Fig. 8). Substrate depth was retained in 17 models (significant in 12). Three succulent species grew best at a substrate depth $<$ 10 cm, whereas other species and diversity tended to respond positively to thicker substrate (Fig. 9). Substrate depth often had an interaction with year (Table 4, Fig. 9). For instance, thick substrate favoured invasive species (excluding *P. hybridus*) in 2016, but their cover was low across substrate depths in 2017. In fact, the most influential environmental variable for species cover was year, retained in 19 models, with 14 negative or positive statistically significant effects (Table 4).

4. Discussion

A lightweight crushed concrete-based substrate of roughly 10 cm 1) supported a highly variable soil nematode community but excluded a main soil animal guild (enchytraeids), 2) produced relatively low concentrations of nutrients in the runoff, even with a high compost content

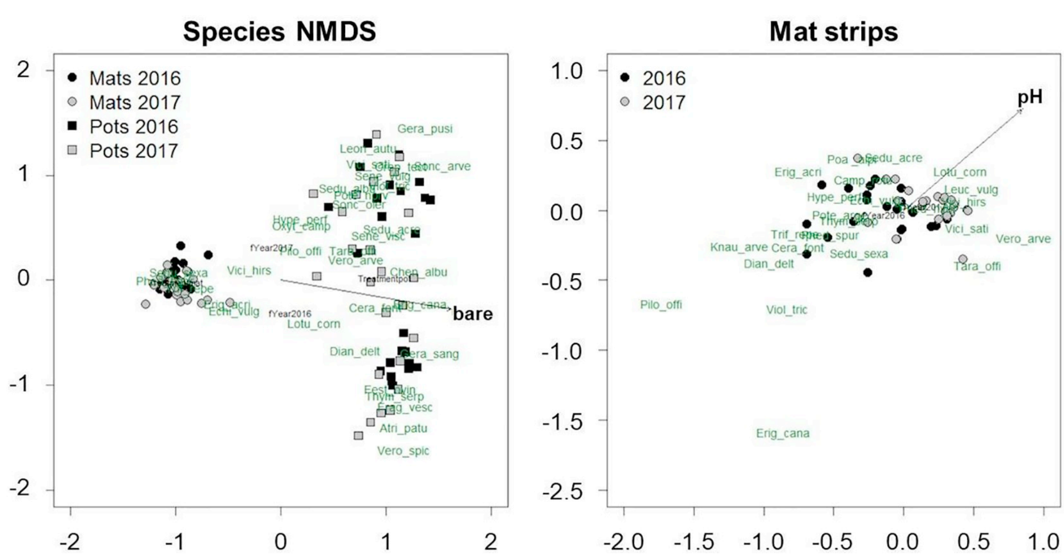

Fig. 6. NMDS ordination plots with all sample quadrats, and quadrats in mat strips only. The dots and squares display the quadrats by year and planting method; lines represent the direction and strength of environmental variables with $p < .05$; and species are shown with abbreviated names.

Table 4
The results obtained by generalized linear mixed effect model separately for individual species, species groups and diversity indices in sample quadrats (0.5 × 1 m^2). Groups of species were classified as planted, spontaneous non-invasive and spontaneous invasive non-native species (see Table A.2 in the appendix). For species denoted with *, experimental strips with zero occurrences of that particular species were omitted from the analysis. The results include all coefficients that were retained in the model. Significant *p*-values are bolded. EST = coefficient, SE = standard error. If the interaction was retained, the estimates for individual variables are shown in grey. Planting method refers to strips established with vegetation mats or pot plants (both amended with seeds), with the mat strips used as a baseline; NA indicates that this variable was not included in the models as the species occurred only in the mats. For compost content, Mix20 was used as a baseline.

Species	Intercept		Planting method			Compost content			Year			Depth	
	EST	SE	EST	SE	p	EST	SE	p	EST	SE	p	EST	SE
Species models													
Dianthus deltoides	−8.31	0.91	0.86	0.37	**0.044**				−8.65	0.65	**0.002**	0.42	0.09
Festuca ovina	−10.35	3.38	2.73	0.68	**0.003**	−1.56	0.40	**0.004**	−4.72	2.20	**0.035**	−0.14	0.10
Hylotelephium ewersii *	10.34	6.86	−2.00	0.78	**0.042**	0.84	0.56	0.182	1.04	1.07	0.337	−0.32	0.14
Phedimus hybridus	0.20	0.18	NA	NA					0.75	0.17	< **0.001**		
Phedimus spurius *	−9.07	1.27							−0.93	0.22	< **0.001**	0.17	0.08
Poa alpina	−1.40	0.84	−1.90	0.33	< **0.001**				0.27	0.16	0.102	−0.16	0.08
Sedum acre	2.63	2.79	3.79	0.60	< **0.001**							−0.41	0.11
Sedum album	4.15	2.85							5.67	2.16	**0.011**	−0.12	0.20
Sedum sexangulare *	4.06	5.88				−0.14	0.45	0.763	0.44	0.87	0.616		
Thymus serpyllum	−15.57	4.31	3.93	0.85	**0.001**				8.79	1.71	< **0.001**	−0.04	0.15
Trifolium repens	−6.87	1.71	−3.08	0.74	**0.002**				−1.97	0.36	< **0.001**	0.40	0.15
Vicia hirsuta	4.08	3.21							−0.75	1.78	0.676	−0.49	0.17
Viola tricolor	8.26	3.09	2.48	0.71	**0.006**				1.14	0.20	< **0.001**	−0.16	0.12
Species group models													
Planted	−3.51	2.63	4.30	0.56	< **0.001**				−2.80	1.02	**0.007**	−0.07	0.13
Spontaneous non-invasives	8.72	2.23	1.27	0.36	**0.006**	−0.01	0.32	0.975	0.55	0.23	**0.020**		
Invasive	−9.21	1.64				1.15	0.46	0.031	3.75	1.90	0.056	0.59	0.15
Diversity models													
Species richness	3.81	0.57				0.13	0.07	0.079	−0.12	0.08	0.159	0.03	0.02
Shannon-Wiener	3.60	0.74	0.36	0.13	**0.013**	0.10	0.13	0.440	1.06	0.48	**0.030**	0.11	0.03
Simpson	4.07	1.31	0.74	0.21	**0.007**	0.07	0.21	0.733	1.76	0.86	**0.044**	0.18	0.06
Evenness	2.36	1.14	0.66	0.19	0.066				2.20	0.73	**0.004**	0.14	0.05

Species	Depth	pH			Bare ground (%)			Year * Depth			Year * Compost content		
	p	EST	SE	p	EST	SE	p	EST	SE	p	EST	SE	p
Species models													
Dianthus deltoides	< **0.001**							0.45	0.21	**0.031**			
Festuca ovina	0.153	0.83	0.44	0.063				0.30	0.21	0.154			
Hylotelephium ewersii *	**0.024**	−1.70	0.89	0.062							1.55	0.80	0.058
Phedimus hybridus													
Phedimus spurius *	**0.049**				−0.62	0.13	< **0.001**						
Poa alpina	0.060												
Sedum acre	**0.003**	−0.63	0.31	**0.047**									
Sedum album	0.540	−1.02	0.38	**0.009**				−0.48	0.20	**0.022**			
Sedum sexangulare *		−1.14	0.75	0.135	−0.29	0.11	**0.011**				1.04	0.53	0.059
Thymus serpyllum	0.784	1.08	0.54	0.050	−0.49	0.10	**0.000**	0.51	0.15	**0.001**			
Trifolium repens	**0.010**				0.17	0.12	0.146						
Vicia hirsuta	**0.006**	−0.78	0.41	0.060	−0.41	0.08	< **0.001**	0.27	0.17	0.126			
Viola tricolor	0.197	−1.43	0.35	< **0.001**	0.22	0.07	**0.001**						

(*continued on next page*)

Table 4 (*continued*)

Species	Depth	pH			Bare ground (%)			Year * Depth			Year * Compost content		
	p	EST	SE	p	EST	SE	p	EST	SE	p	EST	SE	p
Species group models													
Planted	0.591	−0.58	0.30	0.060	−0.24	0.06	**0.001**	0.30	0.10	**0.004**			
Spontaneous non-invasives		−1.55	0.27	**0.001**	0.23	0.06	**0.001**				1.56	0.35	**< 0.-001**
Invasive	**< 0.001**				0.43	0.09	**< 0.001**	−0.47	0.17	**0.007**	−1.22	0.63	0.059
Diversity models													
Species richness	0.135	−0.23	0.07	**0.001**									
Shannon-Wiener	**0.003**	−0.41	0.01	**< 0.001**	0.04	0.02	0.058	−0.10	0.04	**0.024**	0.22	0.14	0.117
Simpson	**0.006**	−0.68	0.16	**< 0.001**	0.07	0.04	0.071	−0.17	0.08	**0.034**	0.5	0.25	0.050
Evenness	**0.012**	−0.42	0.14	**0.003**	0.08	0.04	**0.035**	−0.19	0.07	**0.008**			

in the substrate and 3) enabled diverse meadow-type plant communities. Contrary to our hypothesis, the amount of compost in the substrate (20 vs. 40% *v*/v) had no significant effect on the abundance of most plant species, the diversity of the plant community, nor nutrient concentrations in the runoff.

4.1. Soil animals

Extreme variation in nematode abundance between treatments and sampling times and the total lack of enchytraeids indicate that the below-ground faunal community was highly unstable. Despite its significant role in soil processes, knowledge on soil fauna in vegetated roofs is scarce and focuses mainly on microarthropods (Rumble and Gange, 2013). Nematode densities in grasslands (Boag and Yeates, 1998) and dry meadows (Háněl, 2016) are typically around 10^6 m^2, but vary according to weather conditions. We found a comparable abundance in the mat strips following a rainy period in 2017. Nematodes and enchytraeids are affected by edaphic soil properties like pH, OM content and moisture (Nielsen, 2019), and Rumble and Gange (2013) reported that soil microarthropods in vegetated roof substrates were sensitive to drought. Drought was a likely reason for the lower end abundances in our experiment as well, indicated by the much lower nematode abundance in the drier year 2016 as compared to 2017. Also high pH may have reduced soil fauna and even excluded enchytraeids that prefer acidic conditions (Sulkava et al., 1996). This assumption is supported by the finding that in the mat treatments 75% of the nematode population was in the mat substrate that had a lower pH than the substrate below the mats. The abundance of soil organisms is likely to increase with increasing OM due to improved soil structure and WHC (Nielsen, 2019), but we did not find differences between Mix20 and Mix40. Nematodes and enchytraeids play an important role in decomposition of organic matter and in nutrient cycling and thus indirectly also in plant growth and provision of ecosystem services. Therefore, their role in the functioning of vegetated roofs is worth further studies.

4.2. Moderate concentrations of nutrients in runoff

The concentrations of both TP and TN in runoff decreased over the 5-year study. This was an expected result given that we did not add any nutrients. Obvious reasons for the decrease include a reduced mineralisation of the compost material releasing N, and a reduced amount of P bound to substrate particles over time, as well as uptake of both nutrients by biota. The slightly elevated TN concentrations measured one year after establishment were probably caused by an increase in soil OM decomposition rate, a common process after fresh OM input or mechanical treatment of soil (Kuzyakov, 2010).

Nutrient concentrations in runoff from our experiment fall into the lower half of the wide range of values found in runoff from vegetated roofs, with 0.6–6.8 mg/l for TN and 0.01–4.39 mg/l for TP (reviewed in Berndtsson, 2010; Li and Babcock Jr., 2014). At the beginning of our experiment, nutrient concentrations were close to the upper threshold values set for stormwater in Stockholm, Sweden (TN 5.0 mg/l and TP 0.2 mg/l; Stockholm Vatten, 2001), while at the end they were below the low thresholds (1.25 and 0.1 mg/l). Yet to water bodies downstream, nutrient loads are much more important than concentrations. Actual loads depend on local climate that can be highly variable between years. They depend also on vegetation and many other factors. Nevertheless, to make a rough estimate, assuming that our vegetated roof retains half of annual precipitation (meta-analysis by Mentens et al. (2006) for median roof substrate of 100 mm) means that the yearly TP and TN loads would be 45 mg/m^2 and 1240 mg/m^2, respectively. This is less than one third of what leached from another experiment established likewise but with crushed brick in the substrate instead of crushed concrete (Kuoppamäki and Lehvävirta, 2016). TP load from our roof was only 7–24% of that measured by Mitchell et al. (2017) in widely used vegetated roofs with 10 cm deep substrate in the

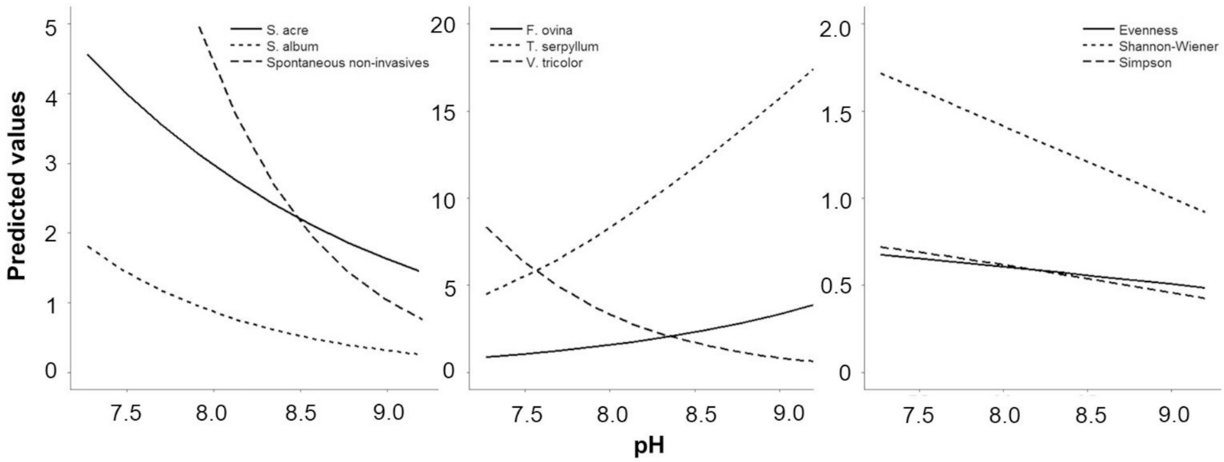

Fig. 7. Examples of the predicted covers (%) of species a) planted in mats only, b) planted or seeded in mats and pots, and c) species that were not planted, in relation to amount of bare ground, obtained from GLMMs.

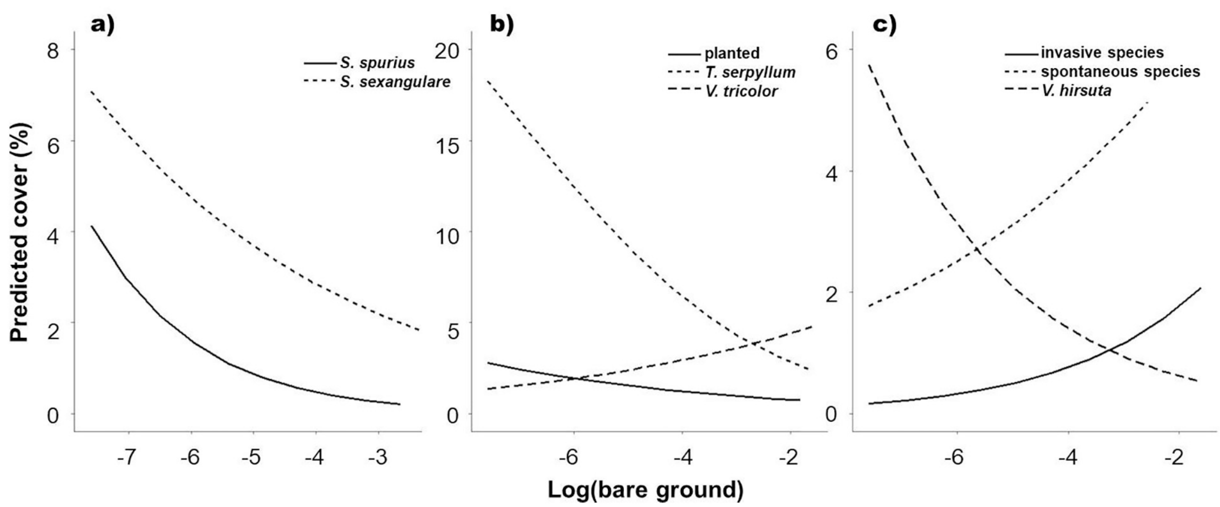

Fig. 8. Predicted values for a,b) cover (%) of plant species and groups and c) diversity indices in relation to substrate pH, based on the GLMMs (LMM for the Shannon-Wiener index). Only statistically significant relationships are shown (see Table 4).

U.S. Compared to P levels in runoff from other vegetated roofs and cultivated fields, our experimental roof turned out to be successful. Yet, the load of P was still 40% of, or equal to, levels in urban stormwater and the load of N was four times higher than from stormwater (Valtanen et al., 2014).

The concentrations of PO_4, the biologically active form of P, were ca. 0.1 mg/l, only 4–11% of levels measured by Mitchell et al. (2017). Their runoff pH was close to neutral (ca. 6.9–7.2), while in our experiment slightly acidic (pH 6) rainwater turned into alkaline (pH 8.1–8.4) runoff - a major change as the scale is logarithmic. Concrete retains P because the high pH and high Ca concentration immobilise P (cf. Jennett and Zheng, 2018). Runoff pH has even been suggested as an indicator of the ability of crushed concrete to retain P (Egemose et al., 2012).

TN in runoff was higher from pot than mat strips, thus supporting the hypothesis that a dense vegetation cover in the mat strips efficiently utilised N, while the sparsely vegetated pot strips left more N in a leachable form. Also Kuoppamäki and Lehvävirta (2016) measured less TN in runoff from roofs installed with mats compared to those installed with pot plants and seeds. These differences can hardly be explained by fertilisation of the pots that introduce very little soil onto roofs (here only one tenth of that in the mats).

While re-use of materials is recommendable to reduce environmental footprint in green construction, the concern is that recycled materials may act as sources of hazardous trace elements (Jennett and Zheng, 2018). Obviously, a priori tests to find dangerous leachates are necessary when screening for new substrates. Only materials that pass this test, should be further tested for their capacity to support flora and fauna and to produce ecosystem services. In terms of trace elements, the recycled lightweight concrete based substrate used in our experiment seems environmentally friendly.

4.3. Vegetation

4.3.1. Plants survive on alkaline substrate

In our experiment, the planted calcicoles and species that tolerate high pH grew better, but several non-calcicoles and species reported to avoid high pH levels also did well. Only one calcicole species, *E. vulgare*, spontaneously arrived on our roof, but based on the Ellenberg values two thirds of the spontaneous species found on the roof tolerate alkaline soil. Furthermore, none of the three planted species with acidic soil affinities survived (*A. dioica*, *D. flexuosa*, *C. canescens*; Appendix A, Table A.2.; Ellenberg et al., 1991; Hill et al., 1999). In summary, species that occur in calcareous habitats, such as *S. acre*, *S. sexangulare*, *S. album* and *F. ovina*, or tolerate high pH, such as *Knautia arvensis*, *P. alpina* (e.g. Ellenberg et al., 1991; Kontula and Raunio, 2018) can be recommended for concrete-based substrates. Furthermore, some species with a wide pH tolerance could be suitable. For example, *T. serpyllum* usually occurs

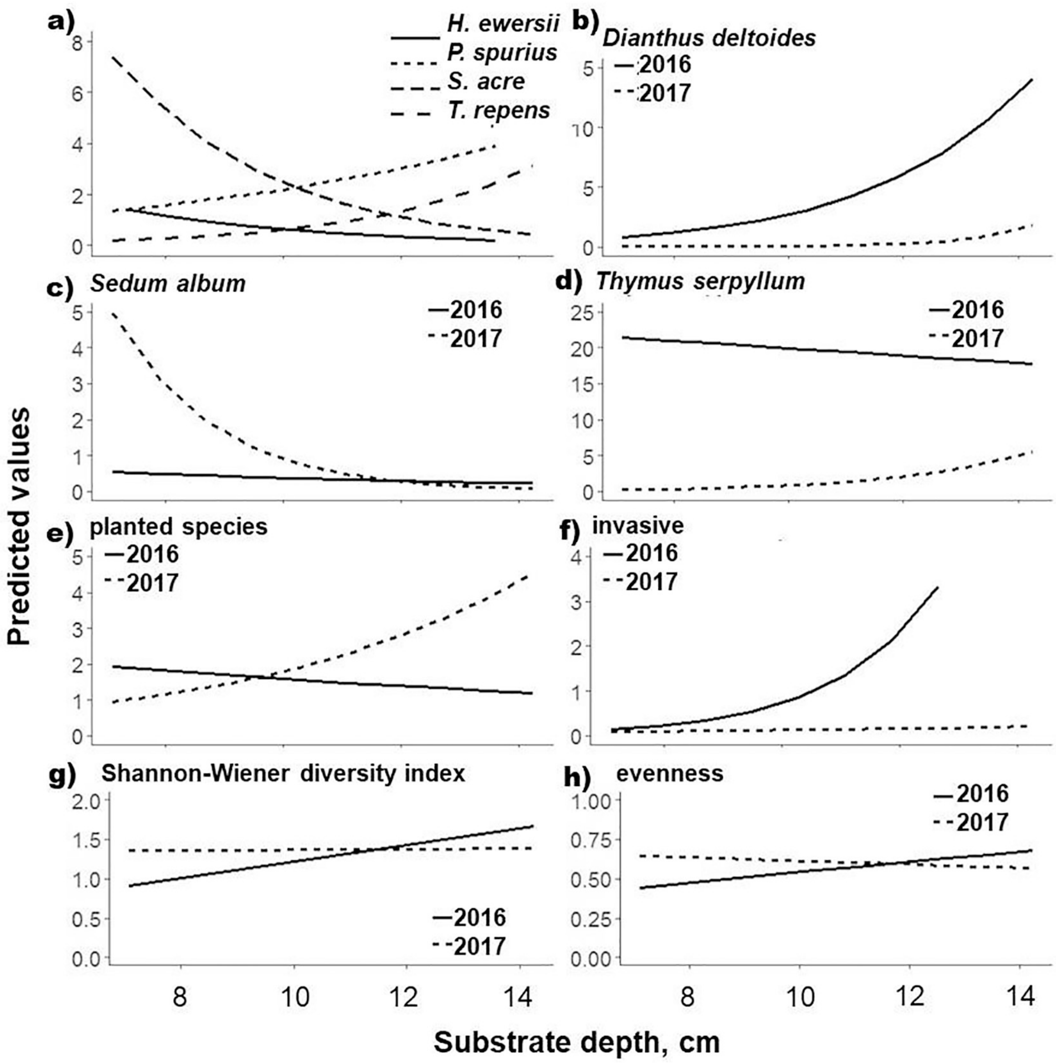

Fig. 9. Predicted values from GLMMs for the cover (%) of selected species and groups, and diversity indices in relation to substrate depth (cm). For readability, different scales are used on y axis and species are shown separately when interaction with year was retained in the model. Values for the Shannon-Wiener index from LMM.

in moderately acid soils (Ellenberg et al., 1991; Hill et al., 1999), but according to Pigott (1955), it tolerates a wide pH range, and even alkalic conditions like it does in our experiment. Likewise, *D. deltoides*, despite having an affinity to low pH based on Ellenberg et al. (1991), survived in our alkaline substrate, which is in line with observations by e.g. Stroh (2014). Also *V. tricolor*, a non-calcicole associated to neutral soils (Appendix, Table 2.A.), was very successful.

Yet some species with affinities to alkaline and calcareous soils did not survive in our experiment (*C. jacea, P. compressa*; Ellenberg et al., 1991; Hill et al., 1999; Kontula and Raunio, 2018). *C. jacea* did not survive in another roof experiment in Finland either (Gabrych et al., 2016). *P. compressa* was planted only in the pre-grown mats, and as it did not spread to the nearby open pot strip communities, it is not likely that it was just outcompeted by other species in the mats, rather, the roof conditions were too harsh for it.

A very low or a very high soil pH limits species richness, but between pH 6–8 the impact seems to be dependent on the species pool (Pärtel, 2002; Tyler, 1994). A high pH limits those plant species that cannot tolerate high Ca concentrations and low Fe and P availabilities (Tyler, 1994). Our models showed that only two species were more abundant at higher pH: both are associated with calcareous habitats, *F. ovina* having alkaline and *T. serpyllum* neutral Ellenberg pH values. Five species with neutral to alkaline Ellenberg values showed no response to pH, while six more species with similar Ellenberg values gave a positive response to decreasing pH (Table 4). The high substrate pH (7.3–9.2 in 2016–2017) obviously limits acidicole species but is not high enough to favour only calcicole species.

Guidelines recommend pH around neutral for vegetated roofs, e.g. FLL (2018) suggests pH 6.0–8.5. Contrary to uniform rules, target pH should be determined based on the species' requirements and the wanted habitat. Calcareous habitats with high pH support rich plant communities (Pärtel, 2002) and could be created on man-made substrates (Auniņš et al., 2013). In fact, Znamenskiy et al. (2006) argued that in Europe, there are more species, also rare ones, associated with high than low pH habitats. Consistently with this and our results, Bates et al. (2015a) showed that alkaline (pH 8) roof substrate supported numerous meadow species in the UK. Even initial pH > 11 of our substrate enabled the growth of several plant species.

4.3.2. Spontaneous species: benefit and nuisance to diversity

Several plant species spontaneously colonised our experiment: almost 60% of all species arrived spontaneously, and one third of those were found every study year. This suggests that many of these species became a stable part of the community and is in line with other studies reporting spontaneously arrived species on vegetated roofs (Aloisio et al., 2019; Catalano et al., 2016). For example, Catalano et al. (2016)

reported over 10 times more spontaneously arrived than planted species on roofs established with five grasses, showing a dramatic change over time. In line with Catalano et al. (2016), we conclude that spontaneous colonisation should be part of vegetated roof design because a vegetation composition with only pre-selected plants may often require much more maintenance than one with a spontaneously developing component. The type of species that get a foothold can be moderated by the substrate quality. In our study the soil solution nutrient levels were not low compared to native grasslands (Tilman et al., 1996) - a situation that may favour fast growing species that are capable of high and fast N uptake, such as *C. album* that was weeded from our experiment during the first year.

While spontaneous native plants contribute to biodiversity value, vegetated roofs may be colonised by non-native invasive species that reduce species richness (e.g. Kinlock et al., 2016). Our results suggest a high competitive capacity of the non-native, invasive *P. hybridus* that efficiently reproduces vegetatively (Norwegian Biodiversity Information Centre, 2018). In our experiment it was presumably introduced as a contaminant in the vegetation mats and became highly dominant in the mat strips. It likely suppressed other species, such as *F. ovina* and *D. deltoides* that were more abundant on pot than mat strips, although they were planted on both. Similarly, *V. tricolor* and *H. telephium*, both sown to pot and mat strips, were more successful in the pot strips.

Although *P. hybridus* is not currently considered invasive in Finland (Finnish Invasive Alien Species Portal, 2019), it is invasive in the neighbouring countries, Sweden and Norway (Nobanis, 2018; Norwegian Biodiversity Information Centre, 2018). Such species may disperse from roofs to surrounding areas and threaten native diversity (Jauni and Ramula, 2015). As imported vegetation mats may offer a pathway for non-native species (Kinlock et al., 2016; Páll-Gergely et al., 2014), we recommend locally produced native plant species. Yet, we also found several invasive species for which the importance of different dispersal mechanisms onto the roof remains unknown. Hypothetically, vegetated roofs could receive invasive species from their immediate surroundings, or further away via contamination during production of substrate and plants.

4.3.3. Effects of planting method on vegetation

The initial species composition has long-term effects on community dynamics, ecosystem function, and biodiversity on roofs (Aloisio et al., 2019). We also found the planting method to strongly affect vegetation composition and diversity. Although the total number of species did not differ between mat and pot strips, the diversity indices and NMDS showed that the mat strips had a more uniform vegetation than pot strips.

Pre-grown vegetation mats are often used to create immediate vegetation cover on roofs, but we also found seeding and pot plants to be a suitable method for establishing roof vegetation. Overall, pot strips offered better growth conditions and diversity. This is good news as pre-grown vegetation mats are expensive and may spread harmful non-native species (Nurmi et al., 2016; Páll-Gergely et al., 2014).

Two species naturally occurring in open habitats, *V. tricolor* and *T. serpyllum*, responded positively to the amount of bare ground. The lower total vegetation cover with open microsites in pot strips also likely enabled the establishment of viable populations from seeds for *H. telephium* (see 4.3.2.). We conclude that, similar to ground-level ecosystems (e.g. Valentin et al., 2017), enriching vegetation with additional seeding to vegetation mats may be challenging, and that competition rather than facilitation by other plants affected seedling success in our experiment (cf. Fibich et al., 2013).

4.3.4. Impacts of compost content and substrate depth on vegetation

Contrary to our hypothesis, we did not find an impact of the amount of compost on vegetation. One reason might be that to avoid excessive complexity we could not include an interaction of the establishment method with the compost content in the models. Thus, the strong

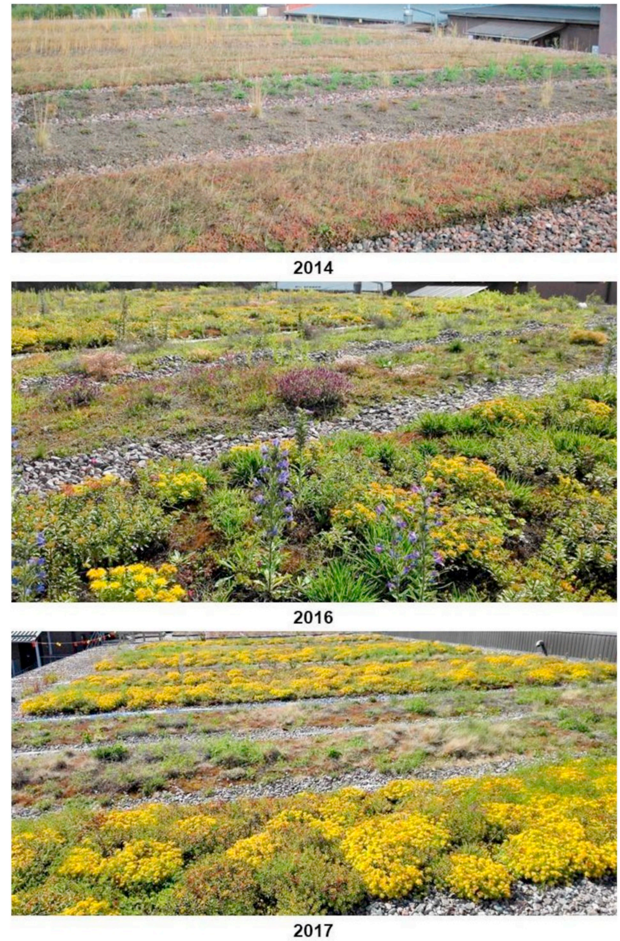

Fig. 10. General overview of the vegetated roof experiment in photos taken at the time of plant inventories in 2014, 2016 and 2017.

dominance of *P. hybridus* in the mats may have obscured the pattern. Actually, in the pot strips 27 species (out of 52) were more abundant on Mix40 than on Mix20, 10 species were equally abundant and 15 species were less abundant on Mix40. Nagase and Dunnett (2011) suggested that 10% OM by volume is optimal for a stable plant growth as a higher content may induce the growth of weeds and invasive species, and promote lush growth that may result in damage during drought. However, only minor weeding was needed in our experiment and we found no evidence of too lush growth: *C. album* was the only weed removed in 2014, and then on, only occasional individuals of *C. canadensis*, *E. adenocaulon*, *E.ciliatum* and *Taraxacum* spp. were removed once a year during inventories. On growing media made with crushed tile, crushed brick or Lytag®, Graceson et al. (2014) found that increasing the amount of composted green waste from 20% (*v*/v) to 30% (v/v) led to increases in the biomass of sedum, which then outcompeted forbs. Our raw data also suggests that in the pot strips our native sedums (*S. album* and *S. sexangulare*) were more abundant with the 40% compost content.

The weak effect of compost content in our experiment might also be partly explained by the highly alkaline habitat where most of the available P is strongly absorbed by concrete-based calcium oxides and hydroxides. Therefore, P was likely the limiting nutrient for plant growth, so any impact of additional N in compost was probably prevented. Finally, given the very similar LOI in the two compost treatments, the difference was probably too small to cause differences in vegetation.

Consistent with previous findings (e.g. Gabrych et al., 2016; Heim and Lundholm, 2014), our results indicate that substrate depth affects plant species abundance and diversity. As previously reviewed by Oberndorfer et al. (2007), substrate depths from 7 to 14 cm, like measured in our study, can support functionally diverse vegetation. In general, thin substrates maintain succulents and mosses, while thicker substrates support meadow plants and high species richness (e.g. Gabrych et al., 2016). Our results presented a similar trend as three succulents (*S. acre*, *S. album* and *H. ewersii*) grew better on lower substrate depth (< 10 cm), whereas other species grew better on thicker substrate. A variable substrate depth, like in our experiment, offers a diversity of niches and coexistence of species associated with different conditions and thus diverse communities (Heim and Lundholm, 2014).

The effect of substrate depth may be associated with annual variation of weather. Thick substrates can retain moisture longer than shallow substrates so the impact of depth may become apparent during drought. For instance, the abundance of *D. deltoides* and spontaneous invasive species increased with increasing substrate depth in 2016, an average year including periods of drought. Yet, in the more rainy and cool year 2017 no such effect was detected.

5. Conclusions

There are many rare, endangered or declined populations of calcicoles due to originally limited habitats and habitat loss. We showed that vegetated roofs can support biodiversity by offering space for native species, also for rare ones like the near-threatened *D. deltoides.* An alkaline concrete-based substrate with pH above 8 seems suitable for many plant species, even those not identified as calcicoles. Yet, vegetated roofs established with imported plant material may serve as pathways for invasive species. Therefore, we recommend the use of native plant species and local products. Crushed lightweight concrete with green waste compost were both locally produced, recycled materials and therefore environmentally responsible choices.

A great benefit of alkaline, concrete based substrate in vegetated roofs is the reduced leaching of phosphorus (P), which is highly important for the protection of adjacent water bodies. However, as the total load of P was comparable to that in stormwater from city centre, there is still a need to improve nutrient retention in vegetated roofs.

The high emergence of spontaneous species, the failure of many planted species to establish and the high variation in plant cover, nematode density and runoff quality showed that vegetated roofs are dynamic elements. Their changing appearance (Fig. 10) and spontaneous colonisation should be recognised as factors when designing vegetated roofs. In such highly dynamic ecosystems, studies should aim at collecting long-term data, as determining properties of the system based on short-term measurements is dodgy.

Declaration of Competing Interest

None.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecoleng.2020.105970.

References

Aloisio, J.M., Palmer, M.I., Tuininga, A.R., Lewis, J.D., 2019. Plant colonization of green roofs is affected by composition of established native plant communities. Front. Ecol. Environ. 6, 238. https://doi.org/10.3389/fevo.2018.00238.

Auniņš, A., Auniņa, L., Bambe, B., Eņģele, L., Ikauniece, S., Kabucis, I., Laime, B., Lārmanis, V., Rēriha, I., Rove, I., Rūsiņa, S., Sniedze-Kretalova, R., Strāķe, S., 2013. European Union Protected Habitats in Latvia, Interpretation Manual. Latvian Fund for Nature, Ministry of Environmental Protection and Regional Development. www.daba.gov.lv/upload/File/Publikacijas/ROKASGR_biotopi_EN.pdf.

Bates, A.J., Sadler, J.P., Gresswell, R.B., MacKay, R., 2015a. Effects of recycled aggregate growth substrate on green roof vegetation development: a six year experiment. Landsc. Urban Plan. 135, 22–31. https://doi.org/10.1016/j.landurbplan.2014.11.010.

Bates, D., Maechler, M., Bolker, B., Walker, S., 2015b. Fitting Linear Mixed-Effects Models using lme4. J. Stat. Softw. 67, 1–48. https://doi.org/10.18637/jss.v067.i01.

Berndtsson, J.C., 2010. Green roof performance towards management of runoff water quantity and quality: a review. Ecol. Eng. 36, 351–360. https://doi.org/10.1016/j.ecoleng.2009.12.014.

Boag, B., Yeates, G.W., 1998. Soil nematode biodiversity in terrestrial ecosystems. Biodivers. Conserv. 7, 617–630. https://doi.org/10.1023/A:1008852301349.

Bozorg-Chenani, S., Lehvävirta, S., Häkkinen, T., 2015. Life cycle assessment of layers of green roofs. J. Clean. Prod. 90, 153–162. https://doi.org/10.1016/j.jclepro.2014.11.070.

Calvo-Polanco, M., Zhang, W.S., Macdonald, S.E., Señorans, J., Zwiazek, J.J., 2017. Boreal forest plant species responses to pH: ecological interpretation and application to reclamation. Plant Soil 420, 195–208. https://doi.org/10.1007/s11104-017-3356-0.

Catalano, C., Marcenò, C., Laudicina, V.A., Guarino, R., 2016. Thirty years unmanaged green roofs: Ecological research and design implications. Landsc. Urban Plan. 149, 11–19. https://doi.org/10.1016/j.landurbplan.2016.01.003.

Egemose, S., Sønderup, M.J., Beinthin, M.V., Reitzel, K., Hoffmann, C.C., Flindt, M.R., 2012. Crushed concrete as a phosphate binding material: a potential new management tool. J. Environ. Qual. 41, 647–653. https://doi.org/10.2134/jeq2011.0134.

Ellenberg, H., Weber, H.E., Düll, R., Wirth, V., Werner, W., Paulissen, D., 1991. Zeigerwerte von Pflanzen in Mitteleuropa. 3. Aufl. Scr. Geobot. 18, 9–166.

Emilsson, T., Berndtsson, J.C., Mattsson, J.E., Rolf, K., 2007. Effect of using conventional and controlled release fertiliser on nutrient runoff from various vegetated roof systems. Ecol. Eng. 29, 260–271. https://doi.org/10.1016/j.ecoleng.2006.01.001.

Erickson, A.J., Gulliver, J.S., Weiss, P.T., 2007. Enhanced sand filtration for storm water phosphorus removal. J. Environ. Eng. 133, 485–497.

FinBIF, 2019. Finnish Biodiversity Information Facility. https://laji.fi/en.

Finnish Invasive Alien Species Portal, 2019. http://www.vieraslajit.fi (accessed 15 October 2019).

Finnish Meteorological Institute, 2019. Temperature and Precipitation Statistics from 1961 Onwards. https://en.ilmatieteenlaitos.fi/statistics-from-1961-onwards.

FLL, 2018. Green Roof Guidelines – Guidelines for the Planning, Construction and Maintenance of Green Roofs. Landscape Development and Landscaping Research Society e.V., Bonn.

Gabrych, M., Kotze, J., Lehvävirta, S., 2016. Substrate depth and roof age strongly affect plant abundances on sedum-moss and meadow green roofs in Helsinki, Finland. Ecol. Eng. 86, 95–104. https://doi.org/10.1016/j.ecoleng.2015.10.022.

Graceson, A., Monaghan, J., Hall, N., Hare, M., 2014. Plant growth responses to different growing media for green roofs. Ecol. Eng. 69, 196–200. https://doi.org/10.1016/j.ecoleng.2014.03.067.

Hämet-Ahti, L., Suominen, J., Ulvinen, T., Uotila, P., 1998. Retkeilykasvio. 4. reformed edition. The Finnish Museum of Natural History Helsinki, pp. 656 (In Finnish).

Háněl, L., 2016. Soil nematodes in alpine meadows of the Tatra National Park (Slovak Republic). Helminthologia 54, 48–67. https://doi.org/10.1515/helm-2017-0005.

Heim, A., Lundholm, J., 2014. The effects of substrate depth heterogeneity on plant species coexistence on an extensive green roof. Ecol. Eng. 68, 184–188. https://doi.org/10.1016/j.ecoleng.2014.03.023.

Hill, M.O., Mountford, J.O., Roy, D.B., Bunce, R.G.H., 1999. Ellenberg's Indicator Values for British Plants. ECOFACT Volume 2 Technical Annex. Institute of Terrestrial Ecology, Huntingdon.

Hyvärinen, E., Juslén, A., Kemppainen, E., Uddström, A., Liukko, U.M., 2019. Suomen lajien uhanalaisuus – Punainen kirja 2019. In: The 2019 Red List of Finnish Species. Ministry of the Environment & Finnish Environment Institute, Helsinki, . http://hdl.handle.net/10138/299501; (web archive link, 15 October 2019). https://punainenkirja.laji.fi (accessed 15 October 2019).

Jauni, M., Ramula, S., 2015. Meta-analysis on the effects of exotic plants on the fitness of native plants. Perspect. Plant Ecol. 17, 412–420. https://doi.org/10.1016/j.ppees.2015.06.002.

Jennett, T.S., Zheng, Y., 2018. Component characterization and predictive modeling for green roof substrates optimized to adsorb P and improve runoff quality: a review. Environ. Pollut. 237, 988–999. https://doi.org/10.1016/j.envpol.2017.11.012.

Kinlock, N.L., Schindler, B.Y., Gurevitch, J., 2016. Biological invasions in the context of green roofs. Isr. J. Ecol. Evol. 62, 32–43. https://doi.org/10.1080/15659801.2015.1028143.

Kontula, T., Raunio, A., 2018. Threatened habitat types in Finland 2018. Red list of habitats. In: Part II: Descriptions of Habitat Types. Finnish Environment Institute and Ministry of the Environment, Helsinki, pp. 661–757. http://urn.fi/URN:ISBN:978-952-11-4819-4 (In Finnish).

Kuoppamäki, K., Lehvävirta, S., 2016. Mitigating nutrient leaching from green roofs with biochar. Landsc. Urban Plan. 152, 39–48. https://doi.org/10.1016/j.landurbplan.2016.04.006.

Kuoppamäki, K., Hagner, M., Lehvävirta, S., Setälä, H., 2016. Biochar amendment in the green roof substrate affects runoff quality and quantity. Ecol. Eng. 88, 1–9. https://doi.org/10.1016/j.ecoleng.2015.12.010.

Kuzyakov, Y., 2010. Priming effects: Interactions between living and dead organic matter. Soil Biol. Biochem. 42, 1363–1371. https://doi.org/10.1016/J.Soilbio.2010.04.003.

Li, Y., Babcock Jr., W., 2014. Green roofs against pollution and climate change. A review. Agron. Sustain. Dev. 34, 695–705. https://doi.org/10.1007/s13593-014-0230-9.

Lundholm, J.T., 2015. The ecology and evolution of constructed ecosystems as green infrastructure. Front. Ecol. Evol. 3, 106. https://doi.org/10.3389/fevo.2015.00106.

Mentens, J., Raes, D., Hermy, M., 2006. Green roofs as a tool for solving the rainwater runoff problem in the urbanized 21st century? Landsc. Urban Plan. 77, 217–236. https://doi.org/10.1016/j.landurbplan.2005.02.010.

Mitchell, M.E., Matter, S.F., Durtsche, R.D., Buffam, I., 2017. Elevated phosphorus: dynamics during four years of green roof development. Urban Ecosyst. 20, 1121–1133. https://doi.org/10.1007/s11252-017-0664-3.

Nagase, A., Dunnett, N., 2011. The relationship between percentage of organic matter in substrate and plant growth in extensive green roofs. Landsc. Urban Plan. 103, 230–236. https://doi.org/10.1016/j.landurbplan.2011.07.012.

NatureGate, 2019. http://www.luontoportti.com/suomi/en/.

Nielsen, U.N., 2019. Soil Fauna Assemblages. Cambridge University Press, Cambridge.

Nobanis, 2018. European Network on Invasive Alien Species. https://www.nobanis.org/.

Norwegian Biodiversity Information Centre, 2018. The Alien Species List of Norway – ecological risk assessment 2018. Retrieved (data) from https://www.biodiversity.no/alien-species-2018 (accessed 15 October).

Nurmi, V., Votsis, A., Perrels, A., Lehvävirta, S., 2016. Green Roof Cost-Benefit Analysis: special Emphasis on Scenic Benefits. J. Benefit Cost Anal. 7, 488–522. https://doi.org/10.1017/bca.2016.18.

Oberndorfer, E., Lundholm, J., Bass, B., Coffman, R.R., Doshi, H., Dunnett, N., Gaffin, S., Köhler, M., Liu, K.K.Y., Rowe, B., 2007. Green roofs as urban ecosystems: ecological structures, functions and services. BioScience 57, 823–833. https://doi.org/10.1641/B571005.

O'Connor, F.B., 1955. Extraction of enchytraeid worms from a coniferous forest soil. Nature 175, 815–816.

Oksanen, J.Blanchet, F.G.Friendly, M.Kindt, R.Legendre, P.McGlinn, D.Minchin, P.R.O'Hara, R.B.Simpson, G.L.Solymos, P.Stevens, M.H.Szoecs, E.Wagner, H. Vegan: Community Ecology Package. R Package Version 2: 5–3. https://CRAN.R-project.org/package=vegan.

Páll-Gergely, B., Kyrö, K., Lehvävirta, S., Vilisics, F., 2014. Green roofs provide habitat for the rare snail (Mollusca, Gastropoda) species Pseudotrichia rubiginosa and Succinella oblonga in Finland. Memoranda Soc. Fauna Fl. Fenn. 90, 13–15.

Pärtel, M., 2002. Local plant diversity patterns and evolutionary history at the regional scale. Ecology 83, 2361–2366. https://doi.org/10.1890/0012-9658(2002)083[2361:LPDPAE]2.0.CO;2.

Pigott, C.D., 1955. Thymus, L. J. Ecol. 43, 365–387.

Pulleman, M., Creamer, R., Hamer, U., Helder, J., Pelosi, C., Pérès, G., Rutgers, M., 2012. Soil biodiversity, biological indicators and soil ecosystem services — an overview of European approaches. Curr. Opin. Environ. Sustain. 4, 529–538. https://doi.org/10.1016/j.cosust.2012.10.009.

R Core Team, 2018. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/ (accessed 15 October 2019).

Radojević, M., Bashkin, V.N., 2006. Practical Environmental Analysis, 2nd ed. RSC Publishing, Cambridge.

Rumble, H., Gange, A.C., 2013. Soil microarthropod community dynamics in extensive green roofs. Ecol. Eng. 57, 197–204. https://doi.org/10.1016/j.ecoleng.2013.04.012.

Sohlenius, B., 1979. A carbon budget for nematodes, rotifers and tardigrades in a Swedish coniferous forest soil. Holarct. Ecol. 2, 30–40. https://doi.org/10.1111/j.1600-0587.1979.tb00679.x.

Stroh, P.A., 2014. *Dianthus deltoides* (L.) Maiden Pink. In: Species Account. BSBI, . https://bsbi.org/wp-content/uploads/dlm_uploads/Dianthus_deltoides_species_account.pdf.

Sulkava, P., Huhta, V., Laakso, J., 1996. Impact of soil faunal structure on decomposition and N-mineralisation in relation to temperature and moisture in forest soil. Pedobiologia 40, 505–513.

Tilman, D., Wedin, D., Knops, J., 1996. Productivity and sustainability influenced by diversity in grassland ecosystems. Nature 379, 718–720.

Tyler, G., 1994. A new approach to understanding the calcifuge habit of plants. Ann. Bot-London 73, 327–330. https://doi.org/10.1006/anbo.1994.1038.

Valentin, K.H., Schafer, D., Kleinebecker, T., Fischer, M., Prati, D., Hölzel, N., 2017. Enriching plant diversity in grasslands by large-scale experimental sward disturbance and seed addition along gradients of land-use intensity. J. Plant Ecol. 10, 581–591. https://doi.org/10.1093/jpe/rtw062.

Valtanen, M., Sillanpää, N., Setälä, H., 2014. The effects of urbanisation on runoff pollutant concentrations, loadings and their seasonal patterns under cold climate. Water Air Soil Pollut. 225, 1977. https://doi.org/10.1007/s11270-014-1977-y.

Vatten, Stockholm, 2001. Klassificering av dagvatten och recipienter samt riktlinjer för reningskrav - Del 2 Reningskrav och Dagvattenklassificering. Stockholm Vatten, Stockholm (In Swedish).

Venables, W.N., Ripley, B.D., 2002. Modern Applied Statistics with S. Fourth Edition. Springer, New York.

Vijayaraghavan, K., 2016. Green roofs: a critical review on the role of components, benefits, limitations and trends. Renew. Sust. Energ. Rev. 57, 740–752. https://doi.org/10.1016/j.rser.2015.12.119.

Znamenskiy, S., Helm, A., Pärtel, M., 2006. Threatened alvar grasslands in NW Russia and their relationship to alvars in Estonia. Biodivers. Conserv. 15, 1797–1809. https://doi.org/10.1007/s10531-004-6680-7.