Research paper

Thirty years unmanaged green roofs: Ecological research and design implications

Chiara Catalano, Corrado Marcenò, Vito Armando Laudicina, Riccardo Guarino

A Università degli Studi di Palermo, Dipartimento Scienze Agrarie e Forestali, Italy
b Università degli Studi di Palermo, Dipartimento STEBICEF, Sezione Botanica, Italy
c Zürcher Hochschule für Angewandte Wissenschaften (ZHAW), Institut für Umwelt und Natürliche Ressourcen (IUNR), Switzerland
d Department of Botany and Zoology, Masaryk University, Czech Republic

HIGHLIGHTS

- 30 years study of unmanaged simple-intensive green roofs in Temperate climate.
- 120 Phytosociological relevés allowed an accurate assessment of vegetation dynamics.
- 32 species traits were used to investigate green roof ecosystem property variations.
- Spontaneous stress-tolerant and ruderal species out-competed most of the sown species.
- Unplanned plant colonisation should be accepted to develop resilient green roofs.

ARTICLE INFO

Article history:
Received 6 April 2015
Received in revised form 13 January 2016
Accepted 16 January 2016

Keywords:
Simple-intensive green roofs
Temperate ecosystems
Long term dynamics
Functional traits
Urban biodiversity

ABSTRACT

The variations in species composition and assemblage of unmanaged simple-intensive green roofs in Hannover, Germany, were investigated over a thirty year period, in order to assess the persistence of the initial seed mixture and to evaluate floristic changes. The roofs were greened in 1985 with soil-based turf rolls sown with a mixture of five grasses (Festuca rubra, Festuca ovina, Agrostis capillaris, Lolium perenne and Poa pratensis). Three sets of 120 phytosociological relevés, sampled in 1987, 1999 and 2014, have been compared to assess: (1) nestedness vs spatial turnover, (2) functional diversity and (3) the importance of vegetation dynamics on green roof performance and design. Results demonstrated that from 1987 to 1999 the species diversity increased and the species turnover prevailed over nestedness, due to the progressive niche occupation by new species. In contrast, from 1999 to 2014 species diversity remained steady, suggesting that nestedness prevailed over species turnover. The main driver of the observed functional changes was a shift towards relatively more theromoconic conditions. In terms of plant life strategies, the competitive species sown on the roof gradually gave way to stress-tolerant and ruderal species, along with a progressive increase in species with shortdistance seed dispersal strategies. It is concluded that: (a) to create resilient green roofs, spontaneous colonisation should be accepted and considered as a design factor; and (b) regional plant communities could serve as a model for seed recruitment and installations.

© 2016 Elsevier B.V. All rights reserved.

1. Introduction

Urban sustainability is one of the urgent challenges of the 21st century (Wu, 2014), since more than 50% of the world’s population live in urban areas, and this figure is estimated to reach 66% by 2050 (UNDESA, 2014). Continuously spreading cities and the growth of intensive agriculture are the major causes of habitat loss and fragmentation worldwide (Grimm et al., 2008). However urban green spaces can play a key role in biodiversity conservation (Goddard, Dougill, & Benton, 2010) and enhance urban ecosystem resilience (Colding, 2007). In particular, green roofs can partially compensate for the loss of green areas by replacing impervious surfaces, contributing to an increase in urban biodiversity (Brenneisen, 2003, 2006). In fact, by replicating specific habitat features and conditions, these artificial biotopes can host native flora and fauna in
relatively undisturbed stands where plants, insects and birds can become established (Köhler, 2006; Kadas, 2006; Baumann, 2006).

The first known study on the biotic colonisation of green roofs dates back to 1940, when Kreh (1945) listed the plant species colonising some tar-paper-gravel roofs in Stuttgart, Germany. This roofing technique was developed at the beginning of 19th century in Silesia and consisted of a combination of tar and four layers of paper covered by a mixture of gravel and sand (Köhler and Poll, 2010). In Kreh’s study (1945), species were categorised according to the following functional group: bryophytes, CAM (Crassulacean Acid Metabolism) species and therophytes, substrate depth preferences (5–20 cm), pollination and dispersal strategies.

Modern green roofs can be classified as intensive, extensive and simple-intensive (German guidelines; FLL, 2008). Extensive green roofs consist of a shallow substrate ranging from 6 to 15 cm, planted or sown with drought tolerant plant species, and require low maintenance; intensive green roofs consist of a >20 cm thick substrate (normally top-soil), planted with woody and/or herbaceous species, and generally require irrigation and high maintenance; and simple-intensive green roofs can be seen as an intermediate roof type, consisting of 15–20 cm thick substrate (including top-soil), hosting perennial grasses and tall herbaceous species, and requiring medium maintenance.

Several studies of spontaneously colonised tar-paper-gravel, simple-intensive as well as extensive green roofs in central Europe, have described the recurrent plant communities thriving on different depths and kinds of substrate (Darius and Drepper, 1984; Thommen, 1986; Borchardt, 1994). These studies found that on 5–8 cm gravel roofs, stress tolerant species (Sedo-Scleranthetea) are enhanced while greater depths favour edural species (Artemisietea vulgaris and/or Stellarietea mediae) and competitive species (Molinio-Arthonatheretea and Festuco Brometea) (Bornkamm, 1961; Bossler & Suszka, 1988). Moreover, humus accumulation, nutrient supply and water holding capacity were identified as the main environmental drivers for plant establishment and community dynamics over time.

Recently, plant functional traits including Grime’s CSR strategies (Grime, 1974, 2001) and life forms, have been used to predict green roof ecosystem services and identify suitable plant species (Nagase and Dunnett, 2010; Lundholm, Maclvor, MacDougall, & Ranalli, 2010; Van Mechelen, Dutoit, Kattge, & Hermy, 2014).

Despite the importance of long-term data in providing adequate planning recommendations (Rowe, Getter, & Durrman, 2012), few studies have examined green roof dynamics for more than a decade (Krüger, 1999; Köhler, 2006; Köhler and Poll, 2010). Köhler & Poll (2010) assessed the effects of growing media on the vegetation quality and species richness of roofs in Berlin over a time span ranging from 13 to 48 years. Krüger (1999, 2001) instead focused on the changes in species composition over 12 years on the roofs of an eco-settlement in Hannover previously investigated by Ackermann and Vahle (1987).

The present study revisited the research site investigated by Ackermann and Vahle (1987) and Krüger (1999, 2001) to examine the composition of the plant community over a thirty year period. Where the main goal of previous studies was the phytosociological description of the vegetation, with the recognition of different facies (characterized by the dominance of a given species) and typologies, the current study focuses on whole roof communities.

We hypothesised that species composition and assemblage on unmanaged green roofs would have changed over the course of thirty years. Specific aims were: (1) to assess if such changes were due to nestedness (species loss) or to turnover (species replacement), (2) to determine changes in species and functional diversity over time and (3) to assess the importance of vegetation dynamics on green roof performance and design.

2. Materials and methods

2.1. Study area

The study area consisted of 15 simple-intensive green roofs of the Waldorf School in the eco-settlement “Laaber Wiesen” in Hanover (Germany, 52°22'N, 9°43'E; 55 m a.s.l.), built between 1983 and 1985 on land formerly cultivated for rye. 9 km away from the city centre. The area lies north of the city park Eilenriede, near Laher Wald, at the southern edge of the Bothfeld district. Along the northern side, the eco-settlement is adjacent to farmland, whereas the other sides neighbour the city conurbation.

The local climate, according to the Köppen-Geiger classification, is warm-temperate, fully humid (Kottek, Grieser, Beck, Rudolf, & Rubel, 2006). The roofs of the eco-settlement were designed by Boockhoo & Rentrop architects and by the landscape architect Gustav Störzer on the basis of the Grassdach-System-Minke roofing technique (Fig. 1) (Minke and Witter, 1983; Minke 2000). This technology was conceived for sloped roofs (5–25°) and consists of a wooden structure sealed with a root resistant, waterproof PVC membrane and a mixture of local topsoil and light aggregates overlapped by a readymade turf carpet (Rollrasen). The investigated roofs had an inclination of 25°, and were elevated 4–7 m from the ground. Although differences in exposure and shade cast by trees could have locally influenced the roof vegetation, the effect of these variables were not investigated in the current study since we were interested in temporal changes in species composition, rather than in spatial variation. The substrate consisted of a mixture of topsoil/expanded clay (liapor) in a 1:1 ratio, 8 cm thick, plus another 8 cm in a 2:1 ratio. The turf rolls were prepared next to the settlement on plastic films to prevent root penetration into the ground. Ten centimetres of topsoil was sown with commercial seeds of Festuca rubra (50%), Festuca ovina (25%); Agrostis capillaris (5%); Lolium perenne (5%); Poa pratensis (15%) and installed on the roofs after 6 months. Our investigation focused on the roofs of the Waldorf School (Fig. 2), since they were left to the natural succession, in contrast to the rest of the settlement, where turfs were periodically irrigated, fertilised and mown as was originally intended (Krüger, 1999, 2001). Since their installation, the roofs of the Waldorf School have been surveyed twice: in 1987 (Ackermann and Vahle, 1987) and in 1999 (Krüger, 1999), allowing the presented long-term vegetation study and a realistic performance assessment.

2.2. Vegetation data

A database of 138 species × 120 phytosociological relevés was created using TURBOVEG software (Hennekens and Schaminée, 2001), 33 of which were sampled between July and November 1987 (Ackermann and Vahle, 1987), 23 between May and June 1999 (Krüger, 1999), 64 between June and July 2014. In all cases, plot size ranged from 1 to 4 m². All relevés were sampled following the phytosociological method of the Zürich-Montpellier School (Braun-Blanquet, 1964). In addition to the species list and their respective cover values, each relevé included the following attributes: exposure, slope, total cover of grass and cryptogamic layer. Taxonomical nomenclature was standardised using The Plant List (http://www.theplantlist.org/, accessed in November 2014). All relevés were georeferenced via the Google Maps interface of the TURBOVEG software and then exported in Quantum GIS vers. 1.8.0-Lisboa (Fig. 2).

2.3. Species traits

In order to analyse the vegetation data, 32 plant species traits were considered, grouped into the following categorical (c) or ordinal (o) functional units: (1c) cholorogy, (2c) life form, (3c)
(2) one of the following five life forms, according to the Raunkiaer’s classification: chamaephyte, hemiepiphyte, phanerophyte, geophyte and therophyte, drawn from Guarino et al. (2010);

(3) one of the following five seed dispersal strategies: anemochory, autochory, barochory, zoochory (including epizoochory, endozoochory and myrmecochory), drawn from Guarino et al. (2010);

(4) one of the following three life strategies (Grime, 1974, 2001; Frank & Klotz, 1990): Competitor, Ruderal and Stress-tolerant, drawn from BIOFLOR (Klotz et al., 2002);

(5b) one of the following EIVs, based on Ellenberg et al. (1992): light (L), temperature (T), continentality (C), soil moisture (F), soil reaction (R), soil nitrogen (N), drawn from BIOFLOR (Klotz et al., 2002);

(6t) one of the following five hemeroby degrees (Hill, Roy, & Thompson, 2002; Walz and Stein, 2014): oligohemerobic, mesohemerobic, ß-euhemerobic, α-euhemerobic, polyhemerobic, drawn from BIOFLOR (Klotz et al., 2002);

(7a) an urbaniety value (U), expressing the species’ affinity to urban environments on a scale from 1 to 5: from urbanophobic (1) to urbanophilic (5), drawn from BIOFLOR (Klotz et al., 2002).

2.4. Substrate chemical analyses

In July 2014, fourteen representative plots (in terms of exposure, orientation and thickness) were selected. In each plot, three replicates of substrate cores were sampled to assess their chemical properties. Substrate samples were air dried and then sieved at Φ 2 mm. Total organic carbon (TOC) and total nitrogen (TN) were determined on pulverised substrate samples by the Walkley–Black dichromate oxidation method (Nelson and Sommers, 1996) and by Kjeldahl digestion (Bremmer, 1996), respectively. Soil reaction was measured in distilled water using a soil/solution ratio of 1:2.5 (w/v) and a glass membrane electrode. Chemical properties in 1985 (LUFÁ, 1985) were compared with those from 2014 using paired t-tests.

2.5. Species change

To compare species composition and relative abundance over time, the whole data set was imported into JUICE software (Tichý, 2002) and relevés were grouped according to the year: 1987 (group 1), 1999 (group 2) and 2014 (group 3). Based on presence/absence data and down-weighting of rare species, a non-metric multidimensional scaling (NMDS) ordination of the species composition of the three groups was performed using R software (version 2.9.0; R Development Core Team, 2009). Since NMDS is a measure of dissimilarity based on a monotonic transformation where the rank order and the distances between points of the original correlation matrix are preserved in the ordination (Austin, 1976; Kenkel & Orlóci, 1986; Whittaker, 1987), it represents an ideal tool to assess the spatial turnover. In order to measure the percentage differences between the considered groups, the Mann–Whitney U similarity was measured on presence/absence data. In this specific case, the presence/absence method was adopted instead of the square root data transformation to discard the influence of the species percentage cover. The total number of species (species pool) per group (year of survey) was calculated together with the average species richness per plot and the species pool sizes were compared by means of accumulation curves. Moreover, to visualise how the species–richness varied across increasing number of plots, a sample-based rarefaction curve was computed (Colwell, Mao, Chang, 2004; Jiménez-Alfaro, Fernández-Pascual, Díaz González, Pérez-Haase, & Ninot, 2012). Relative frequency (RF) of diagnostic species (Φ > 0.20; Chytrý, Tichý, Hoit, & Botta-Dukát, 2002), was
calculated on a 0–1 scale, as a factor of a given species occurrence (N) on the total number of relevés for each group. Sørensen index was calculated to determine the $\beta$-diversity, using presence/absence data and bootstrap procedure with 500 iterations.

2.6. Functional diversity

To assess the shift in mean species trait values and their dissimilarity, a community-weighted mean (CWM) of each trait was calculated using FunctDiv (Lepš, De Bello, Lavorel, & Berman, 2006). CWM values are weighted by the relative abundance of species (Garnier et al., 2004). Species with mixed strategies and/or hemeroby (see Table S1 for details) were assigned multiple traits. The obtained values for each group were organised in a traits/plot matrix, and a non-parametric Wilcoxon test was used to evaluate the differences between the years 1987–2014, 1987–1999 and 1999–2014 (pairwise comparison). The analysis was performed in SPSS software 22.0.

3. Results

3.1. Species change

Species composition changed significantly among the three survey years (Fig. 3), particularly between the years 1987–2014, with a Mann–Whitney U percentage difference of 76.32% (z-statistics 44.9, $p < 0.001$). The lowest difference was detected between the years 1999–2014 (Mann–Whitney U: percentage difference 36.59%, z-statistics 18.94, $p < 0.001$) and intermediate values were obtained between the years 1987–1999 (Mann–Whitney U: percentage difference 68.82%, z-statistics 23.38, $p < 0.001$).

The total species richness increased from 1987 to 2014, with more species detected in 2014 ($N=80$ in 64 relevés), than in 1999 ($N=70$ in 23 relevés) and 1987 ($N=67$ in 33 relevés). Although the sampling effort differed among the three survey years, the rarefaction curve (Fig. 4) showed that the cumulative number of vascular plants at the 23rd relevé was higher in the 1999 group, with 70 estimated species, followed by the 1987 group, with 61 species, and by the 2014 group, with 58 species. The mean species richness per plot ($\alpha$ diversity) decreased from 1987 (14.1 ± 9.1) to 2014 (10.8 ± 3.7), reaching the maximum in 1999 (14.3 ± 5.8). Furthermore, in 2014 the highest value of exclusive species was recorded, with 37 (26.8%) species, whereas 21 (15.2%) and 27 (19.5%) exclusive species were recorded in 1999 and 1987, respectively. Only 23 (16.6%) species were in common among all three years, while 14 (10.1%) species were in common between the years 1999–2014; 11 (7.9%) between the years 1987–1999 and only 5 (3.6%) between the years 1987–2014. The $\beta$ diversity, instead, increased from 1987 (0.49 ± 0.03) to 1999 (0.63 ± 0.03) and then it remained constant until 2014 (0.65 ± 0.02).
Concerning the relative frequency (RF) of the diagnostic species, 10 of them were in common in all three groups; 15 were exclusively found in 1987, 9 in 1999 and 12 in 2014 (Table 1).

3.2. Functional diversity

A pairwise comparison of functional diversity among years found that for most of traits there were differences over time. Out of 32 traits, 23 traits differed significantly between the years 1987–2014, 18 between the years 1987–1999 and 14 between the years 1999–2014 (Table 2). Between the years 1987–2014, the following traits displayed a significant variation: Boreal and Central-European species decreased while Exotic, Eurimediterranean and Eurasian species increased. The distribution frequency of all life forms changed: hemicyryptophytes and geophytes decreased whereas chamephytes, phanerophytes and therophyte increased. All the seed dispersal strategies changed significantly: anemochore species decreased while autochore, barochore and zoochore species increased. EIVs varied significantly, with the exception of R and N: in particular, C, M, and T decreased, while I increased. With regard to life strategies, competitor species decreased significantly, while ruderal species increased. Hemerobry values showed that α-euhemerobic decreased while β-euhemerobic and oligohemerobic species increased.

Considering the significant variations observed between the years 1987–1999, Boreal species decreased while Eurasian and Eurimediterranean species increased. Life forms varied as well: hemicyryptophytes and geophytes decreased while therophytes increased. Regarding the seed dispersal strategies, anemochory decreased whereas barochory and zoochory increased. The following EIVs decreased: C, M, R, N. Competitor species decreased while ruderal and stress tolerant ones increased. Oligohemerobic species increased and urbanity decreased.

The significant variations observed between the years 1999–2014 indicated a decline of Boreal, Central-European and Cosmopolitan species, while Eurasian, Eurimediterranean and Exotic increased. Concerning the life form, only phanerophytes and chamephytes increased significantly. In regard to the seed dispersal strategies, anemochore and zoochore species decreased while autochore and barochore species increased. With regards to EIVs, only R increased. Life strategies and hemeroby did not show any significant variation, while urbanity displayed a slightly significant increase.

3.3. Substrate parameters

The chemical properties of the substrate sampled in 1985 were 36 g kg$^{-1}$ total organic carbon (TOC), 2.6 g kg$^{-1}$ total nitrogen (TN) and 4.5 pH, whereas those determined on substrates sampled in 2014 were (means ± standard deviation) 31.0 ± 3.0 g kg$^{-1}$ of TOC, 1.7 ± 0.2 g kg$^{-1}$ of TN and pH of 5.4 ± 0.5. The absence of significant shifts in chemical properties between 1985 and 2014 was congruent with the absence of significant variation of the edaphic EIVs (N, R) between 1987 and 2014 (Table 2). Unfortunately, no chemical data of the substrate are available for the year 1999 when, according to the EIVs, a slight acidification of the substrate could have occurred.

4. Discussion

The hypothesis that species composition and assemblage changed during 30 years was confirmed by the estimation of species pools, α- (species richness per plot) and β-diversity (species
diversity within group) and diagnostic species per group. These changes were due to both spatial turnover (species replacement) and nestedness of assemblage (species loss) (Wright and Reeves, 1992; Ulrich, Almeida-Neto, & Gotelli, 2009). Since in the first decade (1987–1999) the species richness per plot and the species diversity per group increased, species turnover was more important than species loss, due to the progressive occupation of empty niches. In contrast, during the years 1999–2014 species richness per plot decreased and species diversity per group remained steady, revealing that nestedness prevailed on turnover. Our results showed that only few of the species included in the initial seed mixture were able to establish themselves permanently.

The CWM values of the considered plant traits are useful descriptors of the roof ecosystem dynamics (Garnier et al., 2004).

Since ecosystem functioning is influenced more by the functional diversity than by the species richness (Diaz et al., 2007), and the functional diversity changed significantly between our three survey years, we expect that stability, productivity, nutrient balance and resilience of the green roofs also changed over the last 30 years (Mason, MacGillivray, Steel, & Wilson, 2003). The main driver of these changes was a shift towards relatively more thermic conditions, revealed not only by the significant increase of Eurosiberian and Eurasian species but also by the decrease of hemicryptophytes in favour of therophytes and, consequently, by the significant variation of the EIVs related to temperature and edaphic humidity.

With regard to plant dispersal, two years after the construction of the roof (1987) anemochory dominated. This means that the competitive species originally sown gradually decreased, leaving space and resources to the ruderal ones. This may be a result of the ability of wind-dispersed ruderal species to colonise empty niches, which progressively became available (Grime, 2001). A greater number of ruderal species was recorded in 1999 (Table 2) after which, stress-tolerant ones gained space, and were at their most common in 2014. Along with that, the progressive increase in barochory and autochory illustrated a shift in the succession towards short-distance dispersal species. Furthermore, the establishment of ant colonies probably affected the vegetation dynamics (Guarino, Ferrario, & Mossa, 2005) as the observed increased incidence of zoochorous (myrmecochorous) species in 1999–2014 would suggest.

Unexpectedly, zoochory and hemochory played a more important role than wind which may be related to the habitat filtering as provided by settlements. In fact, dispersal by man and animals may express species-specific preferences i.e. animals may prefer locations with already established biocenosis (fertile surfaces) rather than roads or pavements (sealed surfaces). Wind, instead, is normally channelled along streets, buildings and in general sealed surfaces increasing the probability that anemochorous species will land on unfertile grounds (Knapp et al., 2008).

As a matter of fact, green roofs can serve not only as extra fertile surfaces (not sealed) where plant species can grow in urban environments, but also as places where they may thrive and build a viable population.

There remain other factors that could have affected community dynamics: (a) the influence of the seed bank persisting in the substrate, which contained a relevant percentage of local topsoil, (b) the possible influence of random human visits (e.g. for maintenance purposes) which may have accidentally introduced seeds from neighbouring areas, and (c) the effect of climate change on the observed shifts in life strategy. Indeed, since the edaphic conditions remained almost steady over the thirty year period and the selected roofs were not maintained, environmental factors might have been the most influential. Throughout the last century, Central Europe has experienced a remarkable increase in mean temperatures and the last decade in particular (2005–2014) was the warmest on record (EEA, 2015). In Germany specifically, there has been a strong increase in air temperature and between 1988–2000, almost all years had warmer annual means than the average (Chmielewski, Müller, & Bruns, 2004). This trend has been even more substantial in urban agglomerations due to the heat-island effect which favours the establishment of xeric species coming from warmer regions (Sukopp & Wurzel, 2003).
Moreover the turf-roll construction and the inclination of the roofs was responsible for certain abiotic conditions. The greening took place on the ground and after six months, the grown grass-mats were installed on the sloped roofs. This resulted in the change of several growing conditions such as moisture (from damp, due to the effect of the plastic film used to prevent the root penetration into the soil, to drained, due to the roof slope and used substrate mixture) and exposure as the roofs were facing north, south, east and west. Moreover, the inclination caused a slight shift of the substrate and thus an alteration of the initial homogeneous thickness: on the ridge, the substrate varied from 5–10 cm whereas the depth at the gutter ranged from 20–25 cm. These effects became visible after several years: the survey conducted in 1987 still reflected the initial conditions, while in 1999 and 2014 the decline of competitor species in favour of ruderal and stress tolerant species became evident.

5. Conclusions

Species composition and assemblage changed dramatically over 30 years: from five species sown in 1985, over 10 times more species were recorded in 1987 (67), in 1999 (70) and in 2014 (80), with only 23 of them in common across the whole data set. This suggests that tailored seed mixtures rarely possess the ability to create stable communities without high maintenance (irrigation, fertilisation and weeding). Therefore, if the aim is to develop resilient plant communities on green roofs, spontaneous colonisation should be accepted and considered as a design factor.

We believe that screening regional flora, the recurrent combinations of plant species could serve as a model for seed recruitment and installation on green roofs (Catalano, Guarino, & Brenneisen, 2013). In fact, plants thriving in similar conditions to those of the roofs but not belonging to the regional nor to the local species pool (sensu Zobel, Maarel, & Dupré, 1998) can become successfully established but may fail to enhance habitat connectivity in urban areas.

From a monitoring perspective, plant functional traits prove to be a good means to assess and interpret species change over time. With reference to CSR strategies, the most successful plants in our study were the stress-tolerant species (which have the capacity to maximise limited resources) followed by the ruderal species (which have the capacity to maximise resources in disturbed conditions). These species were better adapted to green roof conditions and outcompeted the sown ones.

Acknowledgements

This research was partially funded by the German Academic Exchange Services (DAAD-codes nr. A/12/87640 and nr. 91562407-50015559) and it was possible thanks to Dipl. ing. Dagmar Krüger and Dipl. ing. Andreas Ackermann and Dip. ing. Christof Vahle, who shared their results and original relevés in 1999 and 1987.


