Urban Ecosystems (2026) 29:212
https://doi.org/10.1007/s11252-026-02066-5

RESEARCH

Comparative growth, biodiversity and cooling effects of native and adaptive Miyawaki microforests and a constructed wetland in a urban park (Reggio Emilia, Italy)

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Received: 10 March 2026 / Accepted: 7 July 2026

Abstract

Urban greening is increasingly promoted within European and international regulations as *'Nature-Based Solutions'* (NBS) concept, to enhance biodiversity and mitigate the urban heat island (UHI) effect. Within the LIFE CityAdap3 project, the city of Reggio Emilia (Italy) implemented "Climate-Friendly Parks" interventions across public green areas. This study focuses on one of these green areas, Marco Biagi Park, where multiple NBS were established between 2021 and 2022, including two Miyawaki microforests —a Native Forest (NF) and an Adaptive Forest (AF) —, a wetland system, rural hedgerows and trees rows. An environmental monitoring program was conducted to evaluate: (i) growth performance and structural development of the two microforests through mortality rate, relative growth rates (RGR_H and RGR_{DBH}) and slenderness ratio (SR); (ii) floristic dynamics and ecological composition of the wetland area; and (iii) microclimatic regulation across different NBS sites and surrounding built-up areas. Linear mixed-effects models were applied to detect spatial and interannual patterns. Three years after planting, both microforests showed very low mortality and comparable overall growth, confirming successful establishment. However, differences in growth allocation emerged: AF species invested relatively more in diameter growth and structural stability, while NF species displayed greater variability and more height-oriented development. Both forests exhibited a decline in SR over time, a typical adjustment under high-density conditions. The wetland showed a rapid floristic enrichment (+69% species richness in one year), with early functional zonation and an increase in Mediterranean chorotypes, suggesting a dynamic community assembly in response to urban environmental conditions. Microclimatic results showed consistently cooler and more humid conditions within microforests compared to built-up areas, while differences between NF and AF were limited and influenced by interannual variability. Overall, the findings highlight and quantify the capacity of integrated blue–green infrastructures to enhance biodiversity, support rapid vegetation development and mitigate urban heat, supporting the combined use of native and climate-adaptive species in future urban afforestation strategies.

Keywords Microforests · Urban park · Biodiversity · Microclimate · Nature-based solution

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Published online: 20 July 2026

Abbreviations

DBH	Diameter at Breast Height [cm]
%M	Mortality Rate [%]
SVF	Sky View Factor [%]
RGR	Relative Growth Rate [yr^{-1}]
RGR_H	Relative Growth Rate Based on Height (H) [yr^{-1}]
RGR_{DBH}	Relative Growth Rate Based on DBH [yr^{-1}]
SR	Slenderness Ratio (H/DBH) [ratio]
H	Total Height [m]
T_{air}	Air Temperature [°C]
T_{soil}	Soil Temperature [°C]
RH_{air}	Relative Air Humidity [%]
SWC	Soil Water Content [%]
CI	Confidence Interval

NBS types and site codes

NBS	Nature-Based Solutions
AF	Adaptive Forest
NF	Native Forest
AF_{int}/AF_{ext}	Adaptive Forest, internal / external measurement site
NF_{int} / NF_{ext}	Native Forest, internal / external measurement site
Sh	Shrubs row
Tr	Street trees
Sh_Tr	Shrubs and street trees rows (considered together)
Ext	External built-up areas

Introduction

Urban environments play a key role in major international and European strategies and regulations aimed at promoting biodiversity and restoring natural resources (Caprile 2022; MASE, 2023). In accordance with Agenda 2030, the International Union for Conservation of Nature (IUCN, 2020), followed by political governance bodies, introduced the concept of *'Nature-Based Solutions'* (NBS). This term refers to all those actions aimed at the socio-environmental improvement of communities, carried out through the protection and restoration of ecosystems and promoting the sustainable use of natural resources. At European level, 'EU Forest Strategy for 2030' aims to increase the quantity and quality of forests present in the community territory, including cities, as part of the broader European strategy for biodiversity (Commissione Europea, 2021). This calls for a multidisciplinary, evidence-based approach to design sustainable solutions that balance economic growth, social equity and environmental protection. Plant diversity serves as a key point in the design of NBS: plants, indeed, generate and support a wide range of ecosystem services (ES), which go beyond the ornamental value that has historically been attributed to greenery in cities (Isbell et al. 2011). Aware of these potentialities, plant ecologist A. Miyawaki claimed that the process of regeneration of green spaces in residential and industrial areas must not and cannot be limited to a mere decorative intervention (Miyawaki 2008). The relative lack of vegetation cover, the impervious surfaces and high density of buildings, enhance the overheating phenomenon leading to the formation of the Urban Heat Island (UHI), where urban temperatures exceed those of nearby rural areas (Despini et al. 2021; Synnefa et al. 2008). The direct and indirect benefits deriving from the presence of urban greenery are multiple and differentiated, included in two macro categories (Perini and Magliocco 2014): environmental services (such as microclimatic mitigation, carbon sequestration and storage, capture of atmospheric particulate matter, soil stabilization and permeabilization) and cultural services (mental and physical well-being, socialization, creativity, etc.) (Berglihn 2021; Santunione et al. 2024). The presence of trees in urban spaces allows to reduce the air temperature heat stress through shading and evapotranspiration processes influenced by factors such as leaf morphology, plant density, plant species and age and season (Qiu et al. 2013; Schmidt 2010).

The present research was realized in the context of LIFE CityAdap3 project (https://www.lifecityadap3.eu) started in 2021, including the city of Reggio Emilia (Po Valley, Italy) beyond six Spanish cities. The project's pilot actions facilitated the planting of over 2,850 new trees across public parks in urban and peri-urban areas of Reggio Emilia with the primary objective of mitigating the microclimate and counteracting the formation of UHI effect. Four parks were chosen based on their varying structural characteristics and levels of public usage. Among them, Marco Biagi Park is the one on which this study focuses. The activity, titled "*Climate-Friendly Parks*" integrated the so called "landscape-environmental devices" within urban green areas: they adhere to the principles of NBS and include (1) two types of microforests, planted using the Miyawaki method, (2) constructed wetlands, (3) shrubs rows and (4) street trees rows.

The main landscape-environmental devices introduced in the city parks of Reggio Emilia are the microforests, defined as small forested areas of between 100 and 200 m^2 (Schirone et al. 2025), in which numerous trees and shrub species have been planted according to the principles of the method developed by the plant ecologist Akira Miyawaki in the 1980s (Miyawaki 2008). This method involves a careful selection of plant species and the choice of samplings planted at a very young stage with a high plant density (from 2 to 4 per m^2), trying to simulate the process of natural plant succession, from pioneer species to climax species of mature forests. The most relevant

feature concerns the timing of this succession: in a natural environment this can take even a few centuries (Van Der Maarel 2005), while with Miyawaki technique the first significant results can be observed after just ten years. The method prioritizes the criterion of autochthony in species selection, according to the concept of *'potential natural vegetation'*, which identifies the mature vegetative state that a system would reach without human intervention and depends strongly on the climatic and eco-pedological conditions of the planting site (Chiarucci et al. 2010). The second NBS involved in this study could be defined according to the Ramsar Classification system which includes ponds, canals, drains, and artificial lakes among human-made wetlands. In urban contexts, these elements can be integrated into constructed wetland systems, which have been widely described and applied as NBS (Alikhani et al. 2021). They can provide biodiversity opportunities as well as water quality improvement, flood protection, climate mitigation and wildlife support (Pedersen et al. 2019).

Shrubs rows can be regarded as hedgerow-like landscape elements, consistent with the traditional lowland woody field margins of the Po Valley (Sitzia 2007), where linear woody vegetation plays an important role in ecological connectivity and biodiversity support. They were traditionally used as element of delimitation of the boundaries of cultivated fields or meadows, providing at the same time important activities of windbreak and stabilization of the banks (Pasini 2019). According to FAO, street trees are one of the main components of urban forests, together with trees in parks, gardens, and other urban green spaces (FAO, accessed on 22 April 2026). As also reported by other authors (Marando et al. 2019), they consist of linearly arranged trees species typically planted through human activity. Their potential beneficial effect is multiple: by providing shade to particularly sunny areas and represent an important tool for improving air quality through the capture of airborne particulate matter (PM) (Santunione et al. 2024).

Although NBS are increasingly promoted for urban climate adaptation, quantitative assessments that consider plant biodiversity, early forest structural development and microclimatic regulation remain limited. The cooling mechanisms provided by vegetation are well documented, however there is still a lack of quantitative data on the spatial extent of temperature mitigation effects derived from field-based measurements rather than satellite imagery (Ouyang et al. 2023; Saini et al. 2025). In addition, evidence on the contribution of individual NBS, included a blue infrastructure to the regulation of ES remains limited (Ouyang et al. 2023). Moreover, to date, no studies have simultaneously assessed biodiversity enhancement and microclimatic regulation in urban areas enriched with multiple types of NBS. Specifically, the available literature on the ES provided on Miyawaki forests is still very limited and it is mainly focused on vegetation establishment and growth dynamics (Schirone et al. 2011, 2025). A few recent studies have reported potential biodiversity enhancement associated with the introduction of microforests (Morales et al. 2026), although the authors themselves underline that the available evidence is still scarce and that further empirical data are needed. Other studies highlight the innovative afforestation approach introduced by Miyawaki but also point to a lack of empirical data (Gill 2024). Furthermore, no published data are available on direct empirical comparisons between different typologies of Miyawaki microforests established under the same urban environmental conditions.

This study addresses these gaps through an innovative side-by-side comparison of two contrasting microforests — one composed of native species and the other including adaptive species — located within the same urban park. Furthermore, the study expands beyond green infrastructure alone by assessing the contribution of a constructed wetland area within the park. By integrating multi-year ecological monitoring with microclimatic measurements, this research contributes to support evidence-based urban planning and more effective implementation of NBS.

The present study implemented an environmental monitoring program, starting one year after the introduction of the NBS and continuing over the subsequent two years (2024–2025). The overall aim is to provide a quantitative assessment of selected ecological and microclimatic benefits associated with their implementation.

Specifically, the objectives of this study are:

(1) to evaluate the growth performance and early structural development in two different Miyawaki microforests over two consecutive years, comparing native and adaptive species assemblages in relation to species-specific traits;
(2) to analyze the dynamics of floristic biodiversity and the ecological role of the constructed wetland area placed within the M. Biagi Park;
(3) to quantify the microclimatic effects of the implemented green and blue infrastructures during two consecutive warm seasons (2024–2025), assessing their contribution to heat stress mitigation.

Materials and methods

Study area

The area of the study, M. Biagi Park, is in the northeastern part of the city of Reggio Emilia (Emilia Romagna Region, Italy), within Santa Croce district (44.705033, 10.655217).

The park is highly frequented by residents and its location, with a medium-to-high morphological sensitivity to summer heatwaves (Comune di Reggio Emilia, 2020), makes it an ideal site for evaluating the microclimatic mitigation benefits of urban greening initiatives. The park covers an area of 13,000 m² and is equipped with a children playground, small walkable paths surrounded by grassy spaces and rows of trees marking its perimeter. *Carpinus betulus* L. and *Celtis australis* L. lines the northern boundary, while *Tilia platyphillos* Scop. is found along the southeast edge. The park is bordered by a residential complex to the southeast, an industrial area to the west, an irrigation canal and a high-traffic road to the north. The maps of the park before and after the introduction of NBS are reported in Fig. 1.

The NBS introduced in M. Biagi Park through Life City-Adapt3 project are described one by one.

Two types of microforests (Fig. 2) were established according to the plant species involved and their primary function:

- A *Native Forest* (NF) composed of trees and shrub species typical of the Po Valley.
- An *Adaptive Forest* (AF) that includes also Mediterranean species considered suitable for future climatic conditions.

AF and the NF have been placed at a distance of about 20 m from each other, each one covered an area of 150 m² and initially included 450 planted individuals corresponding to a planting density of 3 plants/m^2, established between late 2021 and early 2022. At the time of planting, sampling height ranged between 50 and 80 cm. The soil was deeply decompacted (up to 50–60 cm) to enhance aeration, nutrient availability, and water retention, thereby enhancing soil conditions for rapid root development and the high-density planting typical of the Miyawaki method. An irrigation system was provided only during the first year after plantation to support establishment.

Fig. 1 Schematic representation of the study area comparing the pre-intervention (**a**) and the post-intervention (**b**) conditions. The maps highlight the spatial distribution of the Nature-based Solutions (NBS) implemented within the park

Fig. 2 Aerial photos captured by DJI Mavic Mini drone during summer 2024 on two microforests areas: (**a**) Adpative microforest (AF) and (**b**) Native microforest (NF)

The second key NBS introduced in M. Biagi Park is a constructed wetland realized in July 2023 on the northern border of the park (Pedersen et al. 2019). It is constituted by a pond of about 200 m^2 and a drainage canal 60 m long approximately, linked to the pond through a submerged pump (Fig. 3). The water comes from a canal managed by "Consorzio di Bonifica", which passes close to the park, receiving also agricultural waters. After the pond had been filled, five hydrophyte species were introduced: *Nymphaea alba* L., *Nasturtium officinale* W. T. Aiton, *Callitriche palustris* L., *Mentha aquatica* L., and *Nuphar lutea* (L.) Sm.

Two groups of shrubs were also introduced in the area: the first beside the pond and the second near the playground, for a total of 96 plants, mainly represented by native species such as *Frangula alnus* Mill., *Euonymus europaeus* L., *Berberis vulgaris* L., *Sambucus nigra* L., *Cornus mas* L., *Cornus sanguinea* L., and *Corylus avellana* L.

Lastly, three street trees rows were planted, consisting of 15 individuals of local linden (*Tilia platyphyllos* Scop.), spaced 3 m apart.

The entire green area is managed under a low-input maintenance regime designed to preserve its ecological functions while ensuring usability. In particular, the microforests were irrigated only during the first year after planting, and no mowing was carried out within these areas. From the second year, the growth of herbaceous vegetation within microforests declined markedly because of canopy development and the associated increase in shading. The constructed wetland canal has been left under fully natural conditions, with no removal of the annual above-ground dry biomass of emergent plants, whereas the pond is cleaned twice a year to control algal overgrowth. The herbaceous vegetation in the semi-wet area and along its outer perimeter is mown twice annually. In the rest of the park, the grass is managed under a reduced mowing regime, with two cuts per year, to allow herbaceous species to complete their phenological cycle; only selected perimeter strips are mown four times a year to maintain accessibility for visitors.

Fig. 3 Aereal photo of wetland area within M. Biagi Park (Reggio Emilia): pond on the left side, canal develops towards the right

Miyawaki microforests growth performance analysis

The identification of trees and shrubs species over the two microforests, and over the entire M. Biagi Park, was performed using dichotomous keys and each individual was assigned a unique identification code for an accurate counting across all landscape-environmental devices. Two allometric parameters were measured for each specimen: (1) tree/shrub height (H) measured in [m] using a tape or a clinometer (Forestry Pro II, Nikon) and (2) Diameter at Breast Height (DBH) measured at 130 cm from the soil on the main trunk with a tape.

After the census, the growth performance of the two microforests was assessed using the following indices:

- Mortality Rate (% M), was assessed for species in the microforests over the years 2024–2025 using the formula:

$$\%M = \left(\frac{number\ of\ death}{number\ of\ individual\ planted}\right) \times 100$$

- Relative Growth Rate (RGR), was calculated for the species in the microforests for the period 2024–2025 (annual H and annual DBH growth), using the following formula:

$$RGR = \left(\frac{1\mathrm{n}\ M\ 2 - 1\mathrm{n}\ M1}{t\ 2 - t\ 1}\right)$$

where M1 and M2 are the morphometric parameters at the first and the second time point, respectively (total height [m] and DBH [cm]) and t1 and t2 are the reference years, in our case 2024 and 2025, respectively. RGR calculated with height data have been reported as RGR_H, while the RGR calculated with DBH were reported as RGR_{DBH}.

- Slenderness Ratio (SR), an index of tree growth and stability (Wang et al. 1998), was calculated for the species in the microforests for the period 2024–2025, using the following formula [m]:

$$SR = \frac{H}{DBH}$$

Constructed wetland site and biodiversity dynamics assessment

Floristic surveys on constructed wetland area in M. Biagi Park have been carried out for 2 consecutive years (2024 and 2025), with surveys conducted twice a week, during spring (from 1st of April) to late summer (end of July). Taxonomic

identifications were performed using botanical dichotomous keys, along with the Acta Plantarum website (https://www.actaplantarum.org/) and the PlantNet smartphone app (https://identify.plantnet.org/it), then confirmed using "Flora d'Italia" (Pignatti 2017). The census of herbaceous species and hydro-hygrophilous species was conducted using the quadrats sampling method. A 1 m² quadrat subdivided in a grid of 100 smaller squares of 10 cm² each, was employed. Within each quadrat, all plant species were identified to the lowest possible taxonomic level and their percentage cover was visually estimated. A total of 21 quadrats (1 × 1 m²) were placed along the canal and the pond perimeter, and the list of herbaceous species was recorded. Chorological and Biological Spectra (based on S. Pignatti 1982) were calculated to provide an in-depth understanding of species composition, ecological characteristics and species turnover between the years 2024 and 2025.

Microclimatic measurements

The microclimatic conditions of urban greening at M. Biagi Park were evaluated through on-site measurements of the following parameters:

- Air temperature (T_{air}) at 150 cm height.
- Soil temperature (T_{soil}) at 10 cm depth.
- Relative air humidity (RH) at 150 cm height.
- Soil Water Content (SWC) at 10 cm depth.

The measurements were carried out two/three times per week during spring and summer period of two consecutive years (2024 and 2025), specifically from the first days of May and until the beginning of September. The measurements were carried out only on days with clear skies and no (or poor) ventilation, between 12:00 and 14:00 h, to maximize the impact of the radiative regime on the local microclimate, and thus the microclimate mitigation related to the microforests. A previous research indeed highlights that NBS tend to exhibit their strongest cooling effects during peak daytime and hot periods (Wei et al. 2025).

Both T_{air} and RH were measured using portable instruments (PCE-320 InfraRed Psychrometer) at two points in the microforests (one inside and one at the border), at one point next to the shrubs row, at three points near the constructed wetland (pond, south bank and north bank of the canal) and at one point between the street trees rows. T_{air} and RH were also recorded at three locations along the industrial and built-up area east of the park, using buffer distances of approximately 100 m (Ext1), 200 m (Ext2), and 300 m (Ext3) from the park centroid (defined in QGIS), to highlight potential differences between the green area and the surrounding urbanized context. All on-site measurement points are shown in Fig. 4a and b. This standardized sampling strategy was adopted to capture peak thermal stress conditions; however, the results reflect daytime mitigation potential rather than full diurnal and nighttime buffering capacity.

Measurement of T_{air} and RH were taken at 150 cm above ground level, with K-type thermocouple sensor shielded from direct sunlight, while soil parameters were recorded at 10 cm depth using a using a portable probe (PCE-SMM1 Soil Moisture Meter), with three replicates collected for each air and soil measurement session to ensure statistical robustness.

For each monitoring site, the Sky View Factor (SVF) was acquired using a hemispherical fisheye lens mounted on a Fujifilm X-T3 camera. Images were captured in 1:1 format with the sun low on the eastern horizon, in accordance with the standard requirements for SVF acquisition. The SVF

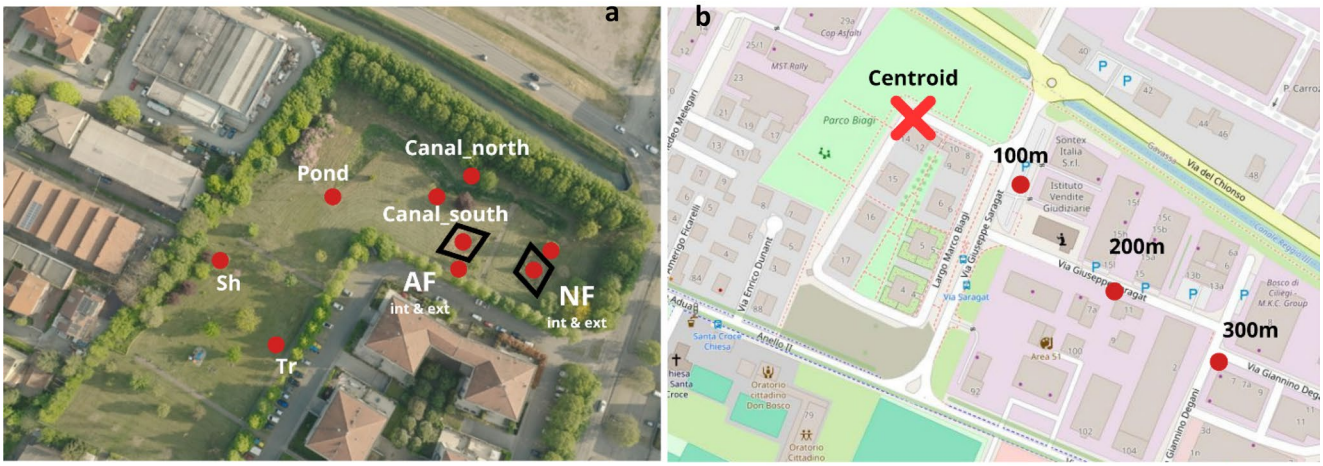

Fig. 4 (**a**) Localization of T_{air} and RH, T_{soil} and SWC measurement sites within the park: a total of 12 sites were selected, including 2 in the Adaptive microforest (internal and external), 2 in the Native microforest (internal and external), 1 in the pond area, 2 along the canal (north and south sides), 1 between the street trees rows, and 1 in the shrub row, (**b**) Localization of T_{air} and RH measurement points along the buffer distances of 100 m (Ext1), 200 m (Ext2) and 300 m (Ext3) from the park centroid

was then quantified through image analysis using ImageJ, selecting the circular area of each photo and calculating the Region of Interest (ROI) represented by sky openness among sampling locations. SVF should be considered when interpreting microclimatic differences among park sites, as it affects radiative exchange by controlling both daytime solar exposure and nighttime longwave cooling (Svensson 2004). Meteorological data of the park were also recorded by a meteorological station installed within M. Biagi Park, sided the pond, which collected data continuously and provided them by a web platform available online (http://cbec.ectoss.com:88/?display=Parco%20Biagi).

The comparison between NBS and surrounding built-up areas was conducted using an observational field-based approach under real urban conditions. This approach allows the assessment of relative microclimatic differences associated with different NBS types while accounting for natural variability in urban environmental conditions.

Statistical analysis

All the statistical analyses were carried out with R software 4.3.3.

The results of slenderness ratio (SR) were analyzed through a linear mixed model was performed (LMM). Model assumptions were verified prior to interpretation (normality of residuals and homoscedasticity). The model was estimated using restricted maximum likelihood (REML) in R (packages: *lme4*, *lmerTest*). The data exhibit a hierarchical structure, with observations grouped by species; to account for the resulting non-independence among observations within species and interspecific variability in SR, species was included as a random intercept. The objectives of this statistical approach are:

(1) Evaluate temporal changes in slenderness ratio (SR) in young afforestation systems and test whether these trends differ between AF and NF. The applied model considers SR as the response variable, with fixed effects for year (2024 vs. 2025), microforest type (AF vs. NF), and their interaction.
(2) To complement the model, results by assessing individual and species-level differences in SR between 2024 and 2025, identifying the species which show the most pronounced shifts in SR.

Microclimatic evaluation was carried out applying a linear mixed-effects model (LMM) for each of the measured microclimatic variable, using the *lmer4* function in R packages, separately for each variable (T_{air}, RH, T_{soil}, SWC).

The data had a nested structure, with repeated measurements collected across sites on different sampling days. To account for the non-independence of observations collected on the same date, sampling date was included in all analyses as a random effect, specified as a random intercept grouped by date. Site was included as a fixed effect. Model assumptions were assessed by checking normality of residuals and homoscedasticity.

In each of these cases, estimated marginal means (EMMs) and pairwise contrasts (with Tukey correction) between sites within years were obtained using *emmeans* package in R software and results were visualized with boxplots highlighting significant differences. For all the analysis, we considered a significance level of $\alpha = 0.05$.

Results

Plants growth performance in Miyawaki microforests

The mortality rate occurred between 2024 and 2025, respectively after 2 and 3 years after plantation, is very low in both types of forest. In NF, 198 surviving individuals were recorded in 2024 and 196 in 2025. Only four individuals died between the two monitoring periods: two *Quercus* trees (*Q. robur* L. and *Q. petraea* (Matt.) Liebl.), one *Morus alba* L., and one *Carpinus betulus* L. Additionally, two new individuals were detected in the 2025 survey: *Prunus avium* L. and *Rhamnus cathartica* L. (Table 1). The most abundant species were *Q. robur* (more than 30 individuals), *Acer campestre* L. and *M. alba* (each with more than 20 individuals), followed by *Fraxinus excelsior* L. (22 individuals). Interestingly, three specimens of *Koelreuteria paniculata* Laxm., — a species not included in the original planting list, largely cultivated for ornamental use and naturalized in Emilia Romagna Region were also observed, likely having colonized the area spontaneously.

In the AF, a total of 170 individuals were recorded in 2024 and 169 in 2025. Only one mortality event was observed between the two surveys, involving a juvenile *Quercus frainetto* Ten. (Table 1). Notably, a *Ginkgo biloba* L. tree, not included in the original planting plan, was identified in this microforest, probably inserted afterwards by a private citizen. *Q. frainetto* was the most abundant species, with over 60 individuals recorded. Only two species—*F. excelsior* and *Ostrya carpinifolia* Scop. — have more than 20 individuals.

RGR values were comparable between the two microforest types (Fig. 5). Between 2024 and 2025, trees height in NF averagely increased by 36.8%, while DBH averagely increased by 75.0%. In AF, trees height averagely increased by 62.5% and DBH averagely by 90.9% over the same period. Thus, although mean values of H and DBH in 2025

Table 1 Species identified in each microforest (NF and AF), with the number of individuals per species recognized in each monitoring year. Species were also classified according to their typical growth habit

	Adaptive microforest			Native microforest		
	Species	2024	2025	Species	2024	2025
Dominant Tree Component	*Quercus frainetto*	66	65	*Prunus avium*	10	11
	Fraxinus excelsior	27	27	*Fraxinus excelsior*	22	22
	Salix alba	1	1	*Quercus petraea*	4	3
	Acer monspessulanum	2	2	*Quercus robur*	36	35
	Celtis australis	2	2	*Populus alba*	1	1
				Tilia platyphyllos	10	10
Dominated Tree Component	*Quercus ilex*	6	6	*Laurus nobilis*	1	1
	Fraxinus oxycarpa	7	7	*Fraxinus ornus*	11	11
	Ostrya carpinifolia	24	24	*Acer campestre*	27	27
	Alnus glutinosa	1	1	*Morus alba*	26	25
	Acer campestre	15	15	*Malus sylvestris*	6	6
	Prunus mahaleb	4	4	*Carpinus betulus*	13	12
	Morus alba	2	2			
Shrub Component	*Hippophea rhamnoides*	2	2	*Rosa canina*	5	5
	Cotinus coggygria	2	2	*Prunus spinosa*	6	6
	Tamarix gallica	4	4	*Cornus sanguinea*	6	6
	Cornus sanguinea	2	2	*Euonymus europaeus*	7	7
	Rhamnus cathartica	1	1	*Frangula alnus*	4	4
				Rhamnus cathartica	0	1
Species not expected	*Ginkgo biloba*	1	1	*Koelreuteria paniculata*	3	3
	Total	**170**	**169**	**Total**	**198**	**196**

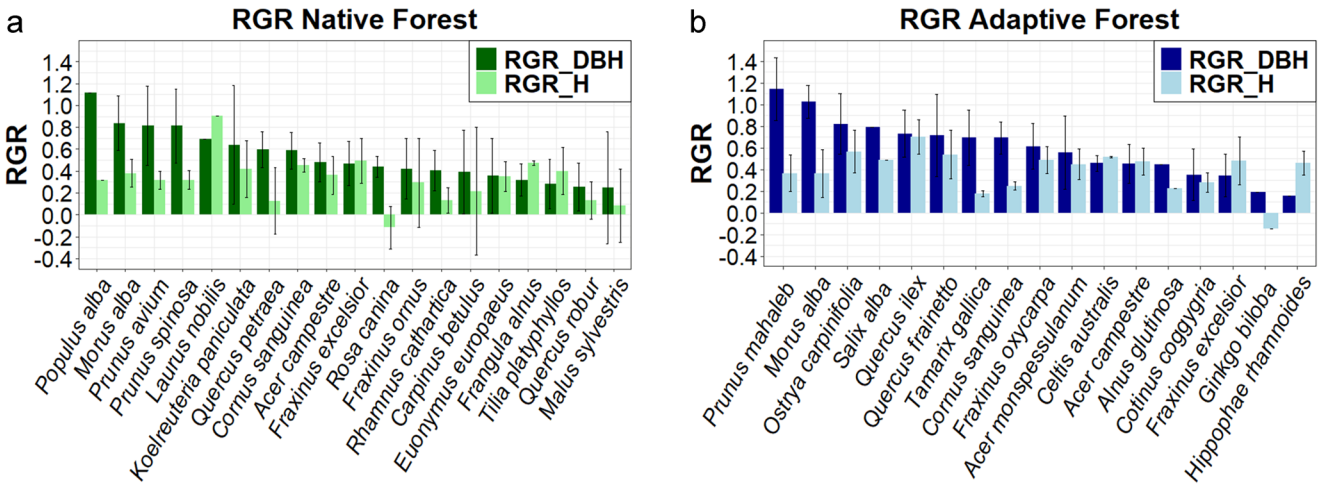

Fig. 5 Mean RGRs (RGR_H and RGR_{DBH}) calculated for each species identified in NF (**a**) and AF (**b**) referred to the growth from 2024 to 2025. Standard deviations (when the individuals of each species were at least 2) are reported

were comparable between the two microforests, AF appears to have performed slightly better because it started from lower values in 2024 and reached similar levels by 2025, indicating a greater overall increment over the monitoring period.

In NF, RGR_H reached always positive values apart for *Rosa canina* L., which showed a high variability among individuals. *Laurus nobilis* L., a species with a typical Mediterranean distribution but widely used in urban contexts within temperate regions, exhibited the highest RGR_H reaching 0.90. The highest RGR_{DBH} was recorded for *Populus alba* L., represented in NF by a single individual with a RGR_{DBH} = 1.12. *M. alba* displayed a relatively high RGR_{DBH} 0.84 ± 0.36 compared to its moderate RGR_H (0.38 ± 0.08).

In AF, RGR_H were generally moderate, with *Quercus ilex* L. (0.70 ± 0.16), *Q. frainetto* (0.55 ± 0.22) and *O. carpinifolia* (0.55 ± 0.19) as the highest values. The highest RGR_{DBH}

were recorded for *Prunus mahaleb* L. (1.14 ± 0.29) and *M. alba* (1.02 ± 0.15).

Variability in RGR was generally higher in NF species compared to AF species, as indicated by standard deviations (Fig. 5).

Considering the SR evaluation (Fig. 6) the model revealed a significant overall decline in 2025 compared to 2024 (β = −24.9 cm/cm, SE = 4.98, $p < 0.01$), indicating a general trend toward reduced SR over time. The SR trend is consistent with early growth dynamics under high-density planting, where height and diameter growth rates adjust rapidly during initial stand development (Kontogianni et al. 2011; Wang et al. 1998).

The type of microforest also had a significant effect: NF exhibits significant higher mean SR values than AF (β = 22.85, SE = 6.85, $p < 0.001$). This indicates that individuals in NF tended to growth more to height than to stem diameter, resulting in a more slender architecture. This pattern is consistent with the species-level RGR results and suggests a lower growth performance in NF when the height–diameter relationship is considered. The year × microforest type interaction was not significant (*p-value* = 0.67), suggesting that the SR temporal decline was consistent across both microforest types. At the species level, most taxa exhibited a decrease in mean SR between 2024 and 2025, supporting the general pattern identified by the model. *M. alba* shows higher SR in NF than in AF, especially in 2024, indicating a more slender architecture in NF. *P. spinosa* (only in NF) has high SR in 2024 and lower SR in 2025, consistent with early vertical growth. By contrast, *Fraxinus* spp. and *Salix alba* L. (mainly in AF) fall in a lower SR range, suggesting earlier structural adjustment.

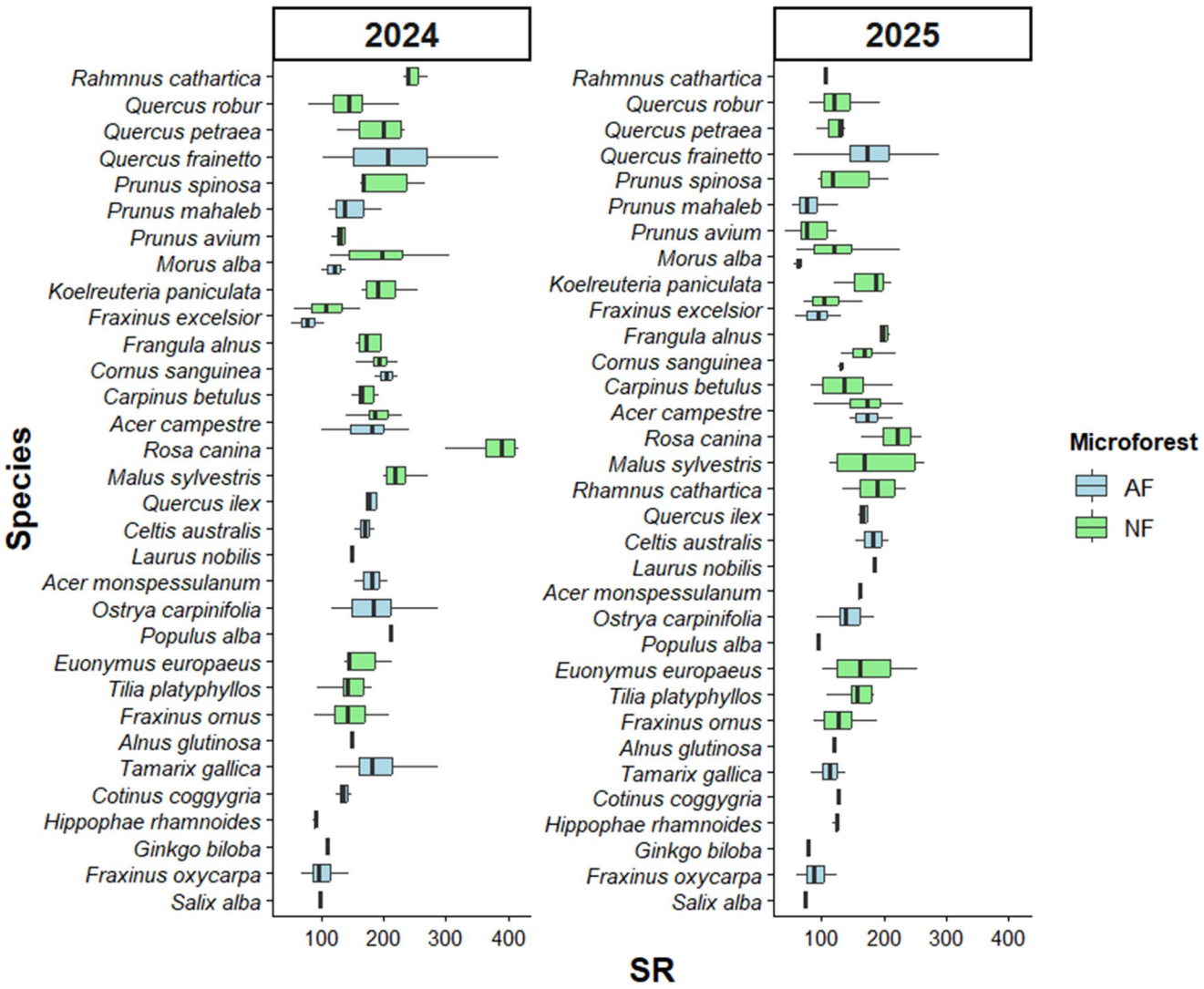

Fig. 6 Slenderness evaluation (SR): mean SR rates for species within NF and AF, referred to the growth period 2024–2025

Biodiversity assessment on constructed wetland

The floristic survey on the constructed wetland area resulted in the identification of 38 and 57 vascular herbaceous plant taxa in 2024 and 2025, respectively, associated with the pond, the canal and adjacent banks and shores. The species list is reported in Table 2.

In 2024, the pond hosted the five introduced hydrophytes and one newly established species (*Ranunculus aquatilis* L.), while vegetation on the pond and canal banks was more diverse than within the canal itself. A total of 15 species were identified on the banks of the pond and 17 on the canal's banks, while only 7 species were recorded directly within the canal itself, including *Limniris pseudacorus* (L.) Fuss, *Carex pendula* Huds, and *Sparganium erectum* L.

In 2024 the most frequent life form was the scapose hemicryptophytes (H scap, 18.4%), followed by caespitose hemicryptophytes (H caesp) and scapose therophytes (T scap) which account for 15.8% of species (Fig. 7a). The most represented chorotype is the Paleo-temperate/subcosmopolite, followed by euorasiatic (26.3%, Fig. 7c). By 2025, the aquatic flora of the pond had expanded to include also *Typha latifolia* L. and *Lemna minor* L., both absent in 2024. In 2025, on the banks of the canal *Lysimachia vulgaris* appears, while *Sparganium erectum* disappeared. Within the canal, *Typha latifolia* formed a nearly monospecific stand extending approximately 25 m, with

Table 2 All plants species recorded *in* and *around* the wetland area of the M. Biagi Park, reported by sampling site. A comparison between 2024 and 2025 is provided using "–" to indicate absence and "+" to indicate presence

Species	Position	2024	2025	Species	Position	2024	2025	Species	Position	2024	2025
Alnus glutinosa	Canal shores	+	+	*Avena fatua*	Pond banks	-	+	*Bellis perennis*	Canal banks	+	+
Prunus cerasifera var. pissardii		+	+	*Calepina irregularis*		-	+	*Bromus arvensis*		-	+
Salix alba		+	+	*Cerastium glomeratum*		+	-	*Calepina irregularis*		-	+
Ulmus minor		+	+	*Cichorium intybus*		-	+	*Carex pendula*		+	+
Callitriche palustris	Pond	+	+	*Convolvulus arvensis*		+	+	*Chaerophyllum hirsutum*		-	+
Lemna minor		-	+	*Crepis vescicaria*		-	+	*Cichorium intybus*		-	+
Mentha aquatica		+	+	*Epilobium* sp.		-	+	*Convolvulus arvensis*		+	+
Nasturtium officinalis		+	+	*Euphorbia helioscopia*		+	-	*Crepis vesicaria*		-	+
Nuphar lutea		+	+	*Geranium dissectum*		+	+	*Dactylis glomerata*		-	+
Nymphaea alba		+	+	*Malva neglecta*		-	+	*Festuca* sp.		+	+
Ranunculus aquatilis		+	+	*Malva sylvestris*		+	-	*Ficaria verna*		+	+
Typha latifolia		-	+	*Mentha aquatica*		-	+	*Geranium dissectum*		+	+
Carex pendula	Canal	+	+	*Lamium purpureum*		+	-	*Malva sylvestris*		+	+
Limniris pseudacorus		+	+	*Lactuca virosa*		-	+	*Mentha aquatica*		+	+
Lysimachia vulgaris		-	+	*Lolium perenne*		+	+	*Lolium perenne*		+	+
Lythrum salicaria		+	+	*Lotus pedunculatus*		-	+	*Lotus pedunculatus*		-	+
Mentha aquatica		+	+	*Onobrychis viciifolia*		-	+	*Phalaris arundinacea*		-	+
Sparganium erectum		+	-	*Papaver rhoeas*		+	+	*Poa* sp.		+	+
Typha latifolia		+	+	*Poa* sp.		+	+	*Potentilla reptans*		+	+
Urtica dioica		-	+	*Potentilla reptans*		+	+	*Primula vulgaris*		+	+
Valeriana officinalis		+	+	*Rumex sp.*		+	+	*Ranunculus velutinus*		-	+
				Sonchus arvensis		+	-	*Rumex* sp.		+	+
				Silene latifolia		-	+	*Sonchus oleraceus*		+	+
				Sonchus oleraceus		-	+	*Sorghum halepense*		-	+
				Taraxacum officinale		+	+	*Symphytum officinale*		+	+
				Trifolium hybridum		-	+	*Taraxacum officinale*		+	+
				Trifolium pratense		-	+	*Tulipa sylvestris*		-	+
				Trifolium repens		+	+	*Veronica persica*		+	+
				Veronica persica		+	+				

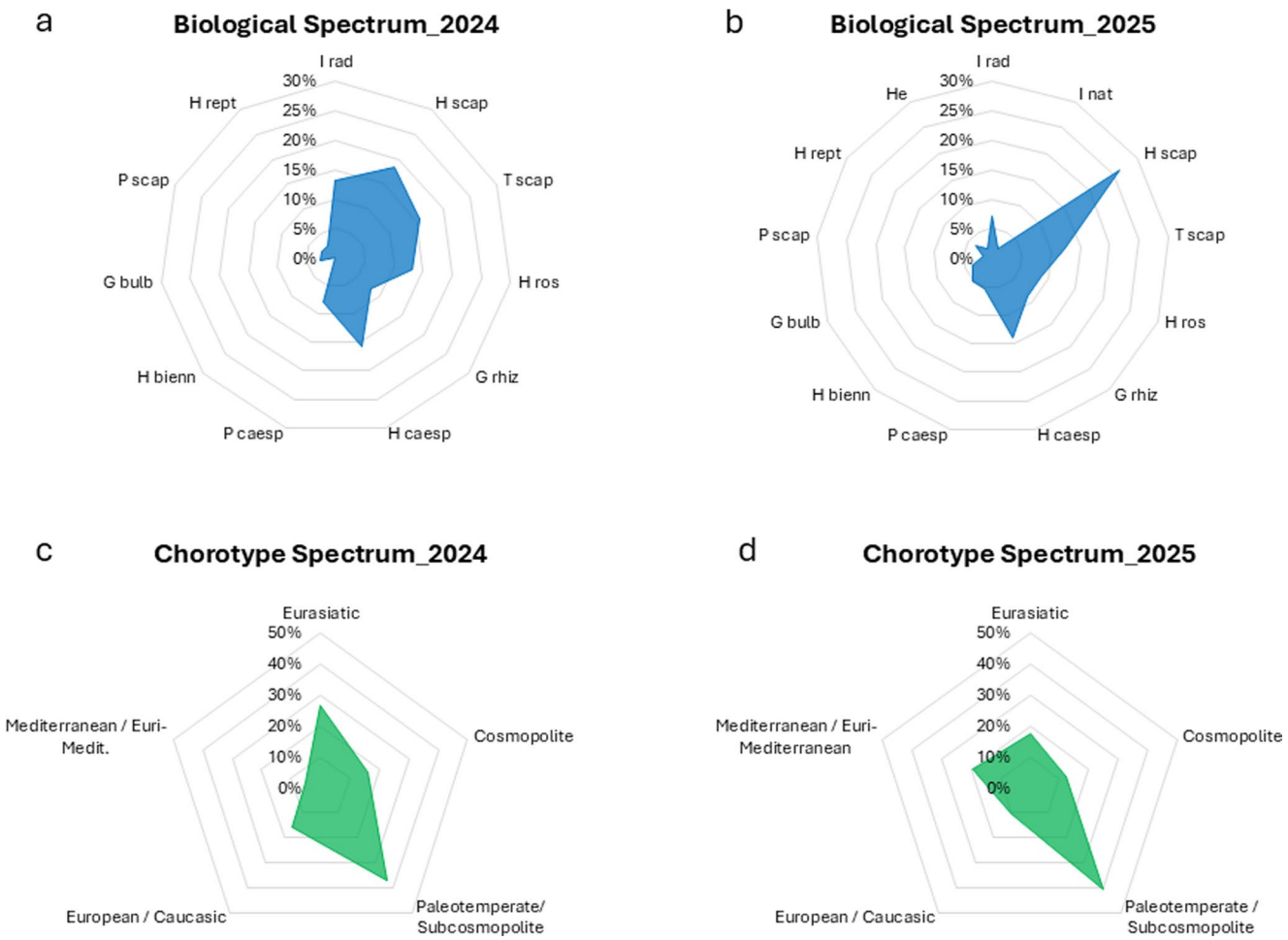

Fig. 7 (**a**) Biological spectra of constructed wetland area in M. Biagi Park, in 2024 and (**b**) 2025; abbreviations of Raunkiaer biological forms are reported as follows: I rad = rooting hydrophytes; H scap = scapose hydrophytes; T scap = scapose therophytes; H ros = rosulate hemicryptophytes; G rhiz = rhizomatous geophytes; H caesp = caespitose hemicryptophytes; P caesp = caespitose phanerophytes; H bienn = biennial hemicryptophytes; G bulb = bulbous geophytes; P scap = scapose phanerophytes; H rept = reptant hemicryptophytes; (**c**) chorological spectra of humid area in M. Biagi Park in 2024 and (**d**) 2025

scattered individuals of *Limniris pseudacorus*, *Lythrum salicaria* L. and *Carex pendula*. Along the pond and canal banks, 39 species were recorded in 2025 (16 species more than the previous year), including several ruderal or synanthropic species as *Veronica persica* Poir., *Sonchus oleraceus* L., *Poa* sp., *Trifolium repens* L., *Cichorium intybus L.,* and *Rumex* sp., along with several members of Fabaceae, Asteraceae, Poaceae, and Brassicaceae (Table 2). Woody riparian vegetation along the shores of water bodies remained limited in 2025, and consisting of only 7 individuals: *Alnus glutinosa* (L.) Gaertn., *Salix alba* L., *Ulmus minor* Mill.: all planted in 2022, plus pre-existing *Prunus cerasifera* Ehrh. var. pissardii.

The 2025 was again dominated by scapose hemicryptophytes, (26.8%, H scap), followed by caespitose hemicryptophytes (14.3%, H caesp) and scapose therophytes (12.3%, T scap) (Fig. 7b). As for chorotypes (Fig. 7d), the Paleotemperate/Subcosmopolite group remained the most represented one. Interestingly, Mediterranean and Euri-mediterranean type increases up to 19.6% vs. the 5.2% from the previous year.

Microclimate on-site analysis

All the outcomes of microclimate analysis are strongly dependent from climate conditions, which have been recorded by the meteorological station present within the park. The daily mean T_{air} from April 1st to October 1st of each year (2024, 2025) has been reported in Fig. 8. A Welch two-sample t-test indicated significantly higher air temperatures during July and August 2024 compared to the same months in 2025 (p-value < 0.05), while June 2025 was warmer than June 2024 (p-value < 0.05).

Microclimatic conditions varied significantly across sites and between years, as shown by the linear mixed-effects model (Table 3). In 2024, NF showed the lowest values T_{air}, while the external areas (Ext) reached the highest values (30.1 °C)

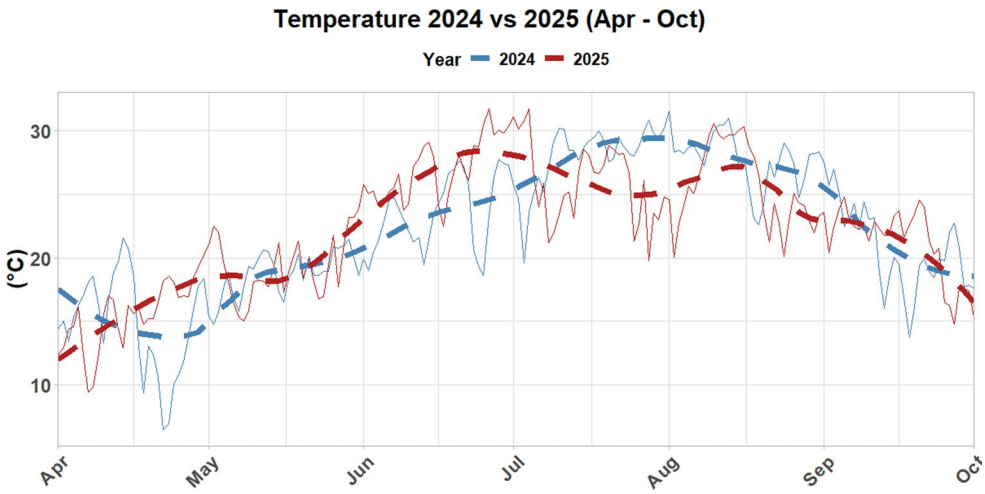

Fig. 8 Seasonal evolution of air temperature (°C) calculated as daily mean between April and October, comparing 2024 and 2025. Dashed lines indicate the overall smoothed trend

Table 3 Estimated marginal means (EMMs) of air temperature (T_{air}), relative air humidity (RH), soil temperature (T_{soil}) and soil water content (SWC) obtained from the linear mixed-effects model (LMM) in 2024. Site was included as a fixed effect, while the measurement data was included as a random effect. For each site, the estimated mean and its lower and upper confidence intervals (CI) are reported. NF=Native microforest (mean of 2 points), AF= Adaptive microforest (mean of 2 points); Sh_Tr= shrubs and street trees rows (mean of 2 points); pond, Ext= external sites (mean of 3 points), canal (mean of 2 points)

site	2024_Tair			2024_RHair		
	EMMs	lower.CI	upper.CI	EMMs	lower.CI	upper.CI
AF	28.3	27.6	29.1	52.3	50.2	54.3
NF	27.3	26.6	28.1	54.6	52.6	56.7
Canal	27.8	27.1	28.5	51.1	49.1	53.2
Pond	28.9	28.2	29.6	49.5	47.4	51.5
Sh_Tr	29.1	28.4	29.8	47.8	45.8	49.9
Ext	30.1	29.3	30.8	45.2	43.2	47.3
	2024_Tsoil			**2024_SWC**		
site	EMMs	lower.CI	upper.CI	EMMs	lower.CI	upper.CI
AF	20.4	19.6	21.1	21.7	20.0	23.4
NF	20.5	19.7	21.2	23.5	21.8	25.2
Canal	22.7	21.9	23.5	20.9	19.2	22.7
Pond	24.0	23.2	24.7	18.5	16.8	20.2
Sh	22.0	21.2	22.7	25.8	24.1	27.5
Tr	22.8	22.0	23.5	21.6	19.9	23.3

(Fig. 9a). RH followed an opposite pattern, with higher values at AF and NF and progressively lower RH at pond, Sh_Tr and Ext (Fig. 9b). T_{soil} was lowest in the two microforests (NF and AF), intermediate on shrubs site, and highest at the pond (Fig. 9c). SWC showed a different pattern, with the highest values at the shrubs site and relatively high values at NF, while the pond and canal recorded the lowest values (Fig. 9d). Pairwise comparisons of T_{air} and RH in 2024 confirmed significant differences among several sites, especially between certain NBS and the external areas, suggesting that the potential air-cooling effect of NBS varies across the park.

The linear mixed model outputs (EMMs) on microclimatic parameters collected in 2025 are reported in Table 4. A small variation of T_{air} among sites is reported (Fig. 10a), with the highest values in Ext sites, similar to pond and Sh_Tr, and lowest in NF, AF and canal. RH follows an opposite scheme (Fig. 10b): this parameter seems to be higher in AF, NF and canal compared to pond, Sh_Tr and Ext. During the same year, T_{soil} differed significantly among sites (Fig. 10c): pond and Tr showed the highest values, canal was intermediate, while AF, NF, and shrubs sites were characterized by lower T_{soil}. SWC showed the strongest differentiation (Fig. 10d). The shrubs site exhibited significantly higher moisture levels compared to all other sites (p-value < 0.05). NF showed relatively high values, while the pond and Tr were among the driest conditions.

Mean SVF values ranged from 10,4% to 52,4%, with the lowest values recorded within and alongside the NF, while the highest values were observed in the pond area and at the external sites. Figure 11 shows the fisheye photographs for most sampling locations, together with the processed images derived from image analysis and the corresponding SVF (%) values for sampling sites. The reported SVF values represent the mean calculated across the monitored sites associated

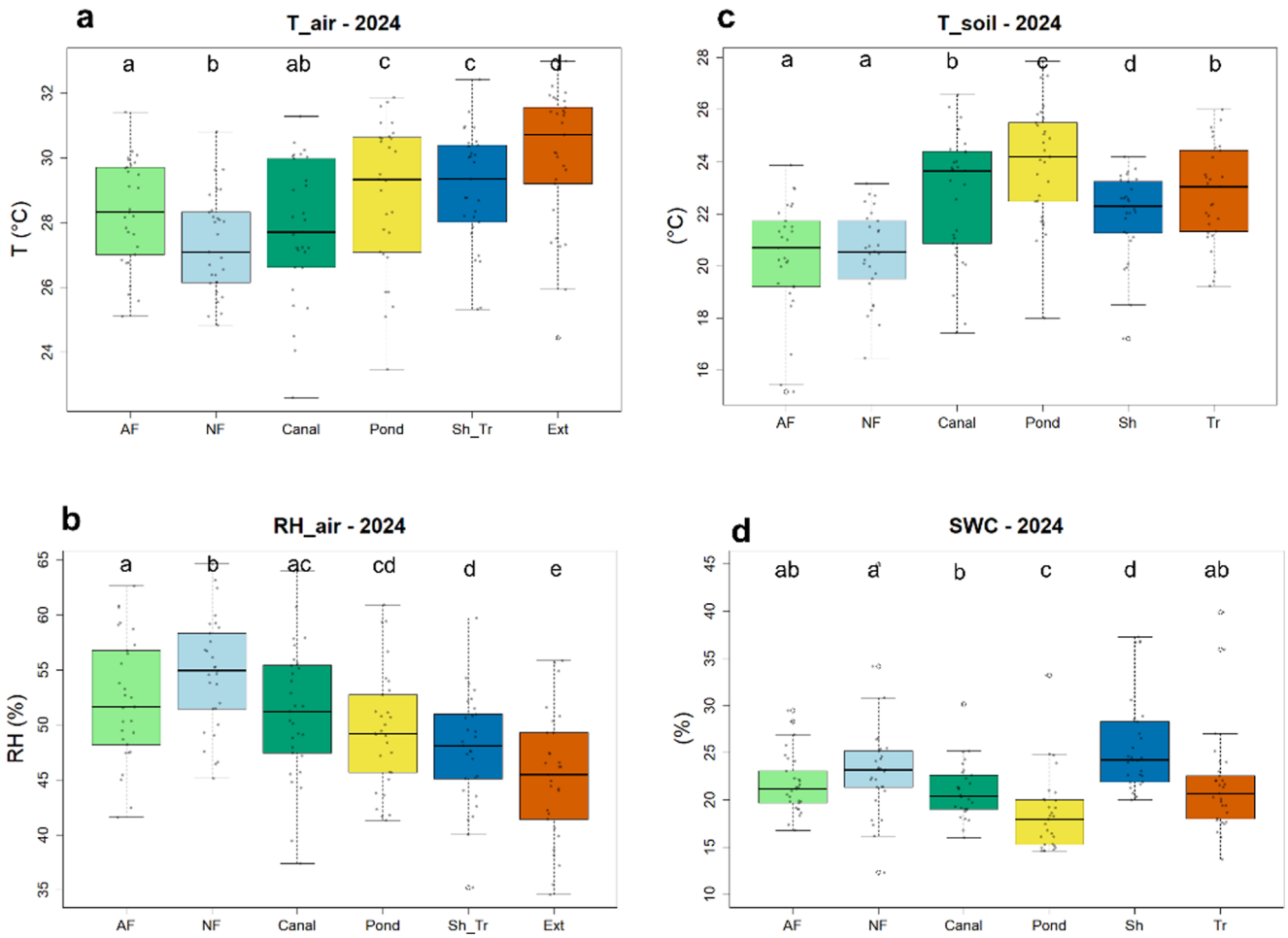

Fig. 9 Boxplot of average microclimatic parameters recorded in 2024; (**a**) T_{air} (°C); (**b**) RH (%); (**c**) T_{soil} (°C); (**d**) SWC measured. NF=Native microforest (mean of 2 points), AF= Adaptive microforest (mean of 2 points); Sh_Tr= shrubs and street trees rows (mean of 2 points); pond, Ext= external sites (mean of 3 points), canal (mean of 2 points). Different letters mark significant differences ($p < 0.05$)

Table 4 Estimated marginal means (EMMs) of air temperature (T_{air}), relative air humidity (RH), soil temperature (T_{soi}) and soil water content (SWC) obtained from the linear mixed-effects model (LMM) in 2025. *Site* was included as a fixed effect, while the measurement date was included as a random effect. For each site, the estimated mean and its lower and upper confidence intervals (CI) are reported. NF=Native microforest (mean of 2 points), AF= Adaptive microforest (mean of 2 points); Sh_Tr= shrubs and street trees rows (mean of 2 points); pond, Ext= external sites (mean of 3 points), canal (mean of 2 points)

site	2025_Tair			2025_RHair		
	EMMs	lower.CI	upper.CI	EMMs	lower.CI	upper.CI
AF	28.2	27.3	29.2	46.9	44.5	49.2
NF	28.3	27.3	29.2	47.1	44.8	49.5
Canal	28.6	27.6	29.5	47.4	45.0	49.7
Pond	29.5	28.5	30.4	44.6	42.3	47.0
Rh_Tr	29.6	28.6	30.5	43.5	41.2	45.9
Ext	30.1	29.1	31.0	43.3	40.9	45.7
	2025_Tsoil			**2025_SWCsoil**		
site	EMMs	lower.CI	upper.CI	EMMs	lower.CI	upper.CI
AF	22.9	22.0	23.8	15.6	14.3	16.8
NF	22.7	21.8	23.6	17.1	15.8	18.3
Canal	23.4	22.5	24.3	16.5	15.3	17.8
Pond	24.5	23.6	25.4	14.6	13.3	15.8
Sh	22.8	21.9	23.7	22.4	21.1	23.6
Tr	24.6	23.7	25.5	14.1	12.8	15.3

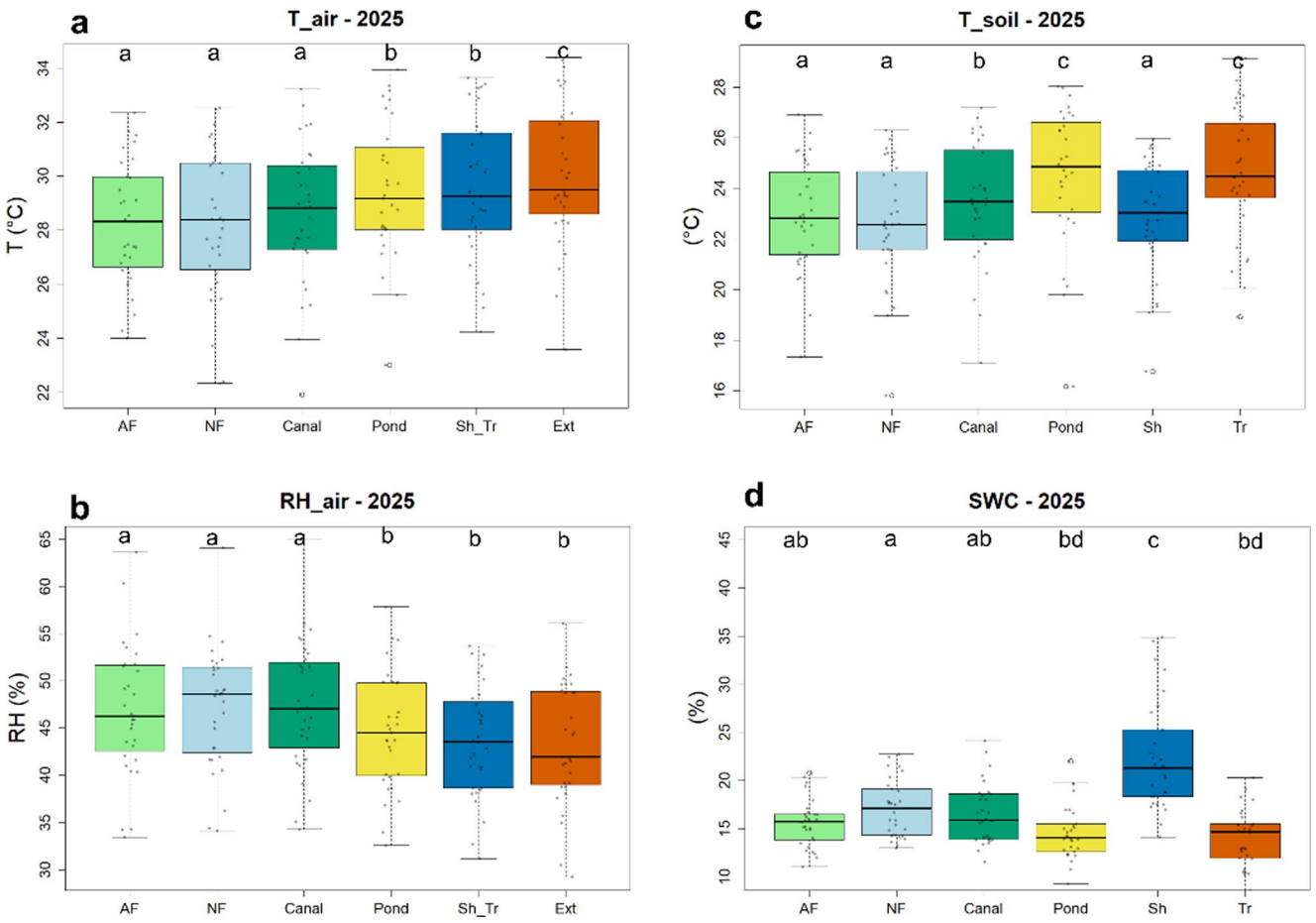

Fig. 10 Boxplot of average microclimatic parameters recorded in 2025; (**a**) T_{air} (°C); (**b**) RH (%); (**c**) T_{soil} (°C); (**d**) SWC measured. NF=Native microforest (mean of 2 points), AF= Adaptive microforest (mean of 2 points); Sh_Tr= shrubs and street trees rows (mean of 2 points); pond, Ext= external sites (mean of 3 points), canal (mean of 2 points). Different letters mark significant differences ($p<0.05$)

with each NBS type, as illustrated in Fig. 4. Regression analysis (Fig. 12) identified a clear positive relationship between SVF and T_{air}, with coefficients of determination (R^2) equal to 0.683 in 2024 and 0.777 in 2025. These findings indicate that increasing SVF was associated with increasing T_{air}. At the same time, the fisheye photographs indicate that the contribution of surrounding buildings to sky obstruction was limited at monitored sites, as they accounted for only a small fraction of the visible sky hemisphere. Indeed, most of the shaded area was covered by trees and other vegetation.

Discussion

Plants growth performance in Miyawaki microforests

A very low mortality rate was observed between 2024 and 2025 in both microforests, suggesting that most individuals successfully passed the most critical post-planting establishment phase and are now better adjusted to local site conditions. This pattern is consistent with the general expectation that mortality is highest shortly after planting and tends to decrease once trees have established functional root systems and acclimated to local biotic and abiotic conditions (Hilbert et al. 2019; Wattenhofer and Johnson 2021). Previous knowledge on microforests showed that Miyawaki method can accelerate tree establishment and ecological succession in Mediterranean sites, promoting rapid stand development (Schirone et al. 2011).

The side-by-side comparison between Native (NF) and Adaptive (AF) microforests established under identical environmental and management conditions provides a unique opportunity to evaluate species-related functional effects from site-driven variability. Despite the overall similarity in mean growth between NF and AF, species-level RGRs indicate context-dependent responses likely driven by micro-environmental conditions, stand structure and species traits. Among genus *Quercus* - the most abundant in both

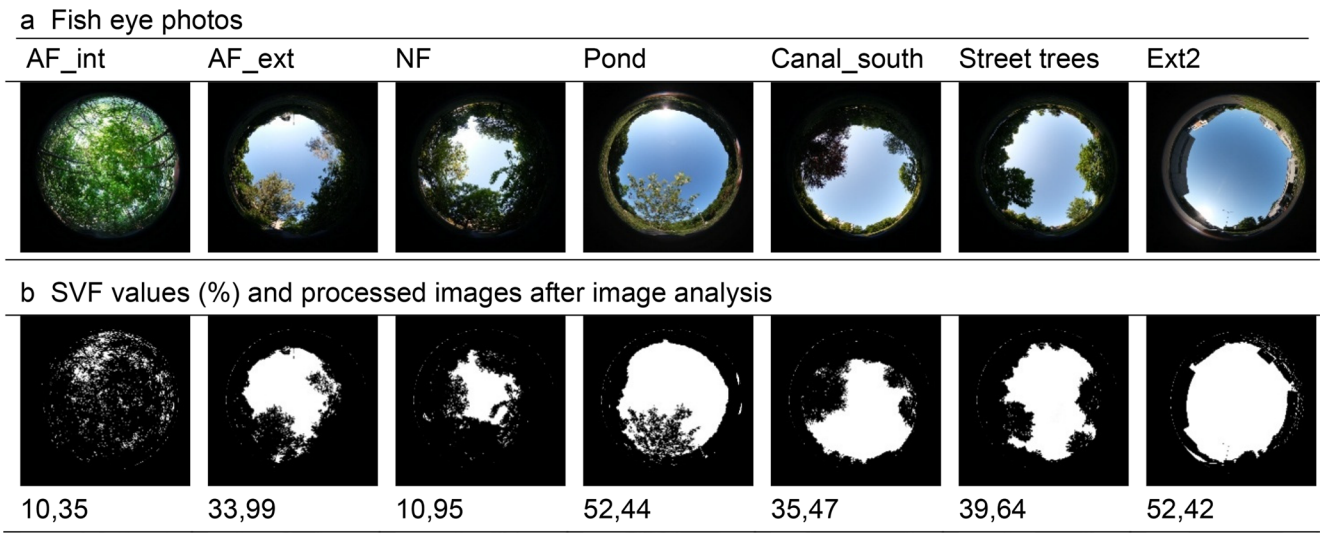

Fig. 11 **a**) Fisheye photographs acquired at most monitored sites. **b**) Corresponding processed images used SVF calculation. The reported SVF values are expressed as mean values for the sampling sites illustrated in Figure 4. As an example, the SVF values for both the internal and external sites of the Adaptive microforest (AF), together with the corresponding images, are reported as representative of canopy cover within the microforest

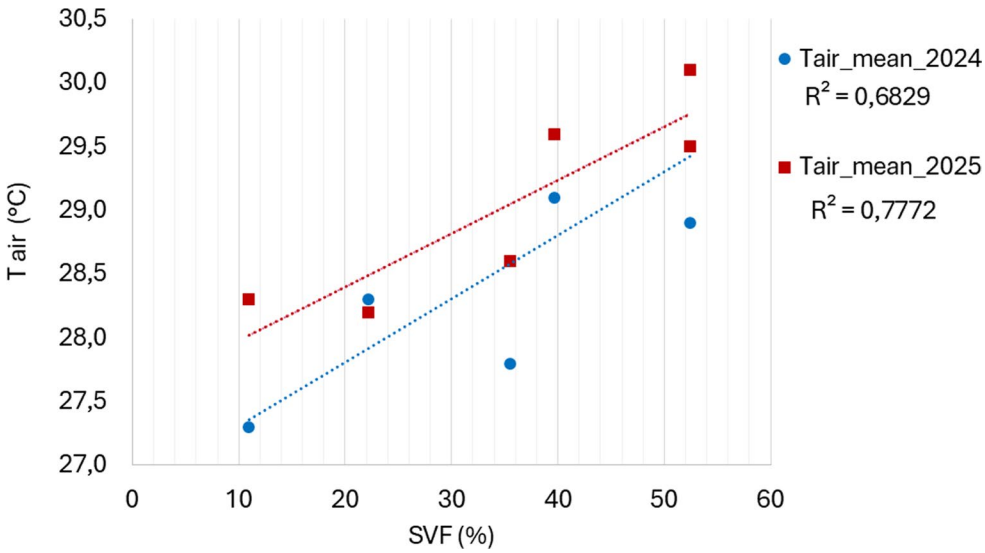

Fig. 12 Scatter plot of mean T_{air} (°C) versus SVF (%) for the 2024 and 2025 datasets. The values shown in the plot represent mean SVF and T_{air} values calculated for each site as follows: NF=Native microforest (mean of 2 points), AF= Adaptive microforest (mean of 2 points); Sh_Tr= shrubs and street trees rows (mean of 2 points); pond, Ext= external sites (mean of 3 points), canal (mean of 2 points)

microforests - AF oaks (*Q. ilex*, *Q. frainetto*) showed comparatively high diameter growth (RGR_{DBH} > 0.7), whereas NF oaks (*Q. robur*, *Q. petraea*) exhibited lower RGR_{DBH} (RGR_{DBH} < 0.6) and higher inter-individual variability. Although this pattern differs from studies reporting that *Q. ilex* grows more slowly and/or is less resilient than temperate oaks in other contexts (Früchtenicht et al. 2021), it may reflects favourable establishment conditions in AF and adaptive responses (Julio Camarero 2019). These outcomes may highlight the role of competition, local climate and microclimate in shaping oak performance (Gea-Izquierdo et al. 2009; Lamonica et al. 2020). In contrast, several NF species displayed higher inter-individual variability and a tendency toward more height-oriented growth, consistent with competitive light acquisition strategies typical of mesophilous temperate assemblages (Castagneri et al. 2022). Shared species further support this plasticity in growth allocation between microforests (*Morus alba*, *Cornus sanguinea*, *Acer campestre*). From a biogeographical perspective, the relatively strong performance observed in AF for euri-Mediterranean and sub-Mediterranean taxa (e.g., *O. carpinifolia*, *Q. ilex*, *Q. frainetto*), especially in terms of diameter increment, suggests their good suitability under Po Valley conditions and may indicate potential to cope with ongoing warming trends (Vezzoli et al. 2015; Zullo et al. 2019). Thus, in this experimental study, Miyawaki microforests exhibited high RGRs, supporting the effectiveness of dense, multispecific planting in promoting rapid and early biomass accumulation

in urban environments. The SR analysis complemented this result, indicating a general interannual SR decline in both systems. This trend is coherent with early stand development under high density, where trees progressively allocate more growth to diameter (Kontogianni et al. 2011; Wang et al. 1998). The non-significant year × microforest interaction suggests similar temporal trends in both microforests. At the species level (Fig. 5), variability in architectural traits was evident, with some species showing greater height growth than others—often observed in dominant species such as *Fraxinus excelsior*, *Tilia platyphyllos* and *Celtis australis*, which showed a positive change in the SR from 2024 to 2025. Such conditions are typical of Miyawaki-style afforestation systems (Nagajyothi et al. 2025) where high planting density promotes rapid early growth and enhances variability among species and individuals during establishment (Ezenwenyi and Chukwu 2017).

Biodiversity assessment on constructed wetland

The floristic composition of constructed wetland area suggests the development of a micro-ecosystem with a clear vegetation zonation along hydrological gradients. The presence of pond banks and a drainage canal create a ecotone that supports high plant diversity (Mitsch W.J., 2015). The coexistence of rooting and floating macrophytes (e.g. *Nymphaea alba*, *Lemna minor*), rhizomatous helophytes (e.g. *Typha latifolia*, *Limniris pseudacorus*) and mesophilous ruderal taxa (e.g. *Poa* sp., *Veronica sp.*, *Trifolium* sp.) reflects a transition from aquatic to terrestrial habitats driven by fluctuating water availability and disturbance.

Despite being completed only at the end of 2023, the wetland showed notable species richness by the second year of monitoring, consistent with values reported for comparable water-based NBS in Italy and other urban environments (Cannucci et al. 2025; Martín Muñoz et al. 2023).

The repeated occurrence of *Mentha aquatica* and *Veronica persica* across several zones indicate active spontaneous colonization. The substantial increase of species richness from 2024 to 2025 (+69%) suggests a rapid community assembly process under urban conditions (Batzer 2014), although this pattern should not be interpreted uncritically as evidence of ecological maturity, as part of the observed richness is attributable to ruderal and disturbance-tolerant taxa typical of early successional stages. Canal banks are dominated by ruderal/nitrophilous species, typical of disturbed ecotones, which nonetheless contribute to stabilization and early successional structure (Biondi et al. 2012; Prach et al. 2001). The presence of plants belonging to Fabaceae (e.g., *Trifolium* spp., *Lotus pedunculatus*) may enhance soil fertility via nitrogen fixation (Reid et al. 2024). The vegetation differentiation observed along the hydrological gradient suggests that even small artificial wetlands can rapidly develop habitat heterogeneity (Bentley et al. 2022). These dynamics are consistent with those described by (Van Der Maarel 2005), which are typically rapid, heterogeneous, and strongly shaped by hydrological conditions and disturbance. Rather than convering toward a single stable vegetation type, small constructed wetlands tend to develop as dynamic mosaics of microhabitats, continuously reorganized by microsite variability, water-level fluctuations, and minor disturbances (Bai et al. 2012; Jiang et al. 2011). The marked increase in Mediterranean and Euri-Mediterranean chorotypes (+14.4%) suggests a directional shift in species composition, likely driven by warmer summer conditions in the Po Valley (Monteleone and Borzí 2024), and the high ecological plasticity of Mediterranean taxa under fluctuating hydrological regimes and disturbance (Biondi et al. 2012; Mitsch W.J., 2015). Although the monitoring period is still short, the rapid increase in Mediterranean chorotypes may represent an early signal of climate-related effects on plants community assembly in urban wetlands. Here, the presence of a constructed wetland can support relatively high biodiversity at early stages, as open-habitat, aquatic and marsh species may coexist. It should be considered the low-input maintenance regime, specifically intended to reduce disturbance to the ecological dynamics of the different NBS and of the site as an integrated system. Because external interventions were strongly limited, vegetation development was allowed to proceed with relatively low management pressure. However, this ecological pattern is placed within urban context, exposing it to a more fragile and unstable dynamics, possibly due to anthropogenic disturbance, resource fluctuations, eutrophication, which can increase vulnerability to invasions and drive shifts in vegetation composition (Van Der Maarel 2005). For this reason, continued monitoring will be essential to determine whether the current assemblage evolves toward a more specialized and functionally coherent wetland flora over time.

NBS on-site effects on microclimate

The persistence of warm conditions during mid-to-late summer 2024 likely amplified microclimatic differences among sites, making the role of local structural features more evident. In particular, T_{air} was strongly related to SVF (Fig. 12): sites with higher SVF, such as the pond and the street trees rows, reached higher T_{air}, whereas densely vegetated sites with lower SVF (AF, NF, and the shrubs row) maintained cooler and more humid conditions. This pattern is consistent with previous studies showing that, when shading is mainly provided by trees and vegetation, lower SVF is associated with lower air temperature due to canopy shading (Speak and Salbitano 2022; Yan et al. 2022). This

evidence may explain part of the microclimatic results: among the microforests, the cooling effect was more pronounced in NF than in AF. Indeed, NF got a temperature offset of approximately − 0.99 °C (p-value < 0.001) and RH offset of + 2.35% (p-value < 0.05) related to AF (Table 3). In 2024, all sites exhibited lower T_{air} than the external areas, with the largest reduction observed in NF (− 2.8 °C), followed by the canal (− 2.3 °C) and AF (− 1.8 °C), indicating a marked thermal differentiation under warmer conditions. In 2025, despite a relatively warm June, cooler conditions in July and August reduced overall thermal stress across the park, leading to weaker thermal gradients among sites. All NBS still showed lower T_{air} than the external areas, but the magnitude of ΔT_{air} was generally reduced—particularly for NF, the canal, pond, and Sh_Tr. Taken together, these results suggest that the cooling effect of vegetated areas is strongly modulated by both canopy shading, indirectly reflected by SVF, and background meteorological conditions, with stronger differences emerging during hotter periods (Meili et al. 2021). The microclimatic differences observed among the microforests and other NBS sites, may partly reflect differences in their average SVF values. As canopy structure develops, cooling benefits are expected to increase through both enhanced shading and higher evapotranspiration rates, mechanisms widely recognized as primary drivers of vegetation-induced temperature reduction (Gu et al. 2022; Wu et al. 2022). However, the limited differences between AF and NF did not translate into major differences in microclimatic performance, suggesting that at this early stage canopy density, position and planting configuration may be more important than species origin in determining cooling intensity (Michelsen-Correa and Scull 2005). The weaker effects observed for the shrub row and the pond are likely related to their more open structure, reflected in higher SVF values, and to the still limited size of trees and shrubs, which results in low canopy continuity and reduced shading (Soltanifard et al. 2025). For instance, *Tilia platyphyllos* individuals in the street trees rows were still relatively small in 2025 (mean height 4.3 ± 0.5 m), with non-overlapping crowns, while shrubs had not yet developed sufficient cover to provide substantial shade. Despite these limitations, lower T_{air} values compared to external areas confirm that each NBS already contribute to summer thermoregulation, in line with evidence reported in urban studies (Manteghi et al. 2015; Syafii et al. 2016). Considering the soil parameters, areas with a higher density of vegetation cover exert a greater influence on water dynamics at the ground level, likely through shading, reduced evaporation, and enhanced infiltration (Marrazzo and Raimondi 2025; Ni et al. 2019; Zhao et al. 2020). Both adaptive and native microforests exhibited the lowest T_{soil} among the environmental devices, with no significant differences between forest types (p-value > 0.05), likely due to the dense canopy reducing soil heating by intercepting solar radiation (Michelsen-Correa and Scull 2005). By contrast, SWC was slightly higher on shrubs soil than in the microforests, particularly in 2025, possibly because of differences in ground cover (Ng et al. 2016). In the shrubs site the ground was densely covered by spontaneous reptant species such as *Trifolium repens* and *Potentilla reptans* L., whereas in the microforests it was mainly covered by *Rumex* sp. or *Festuca* sp. or characterized by bare soil, which appear to be less effective at retaining soil moisture.

Limitations of the study

Among the main limitations of this study, it should first be noted that the research was conducted within a single urban park characterized by a highly specific environmental and management context. Consequently, the patterns observed here, including plant survival, growth dynamics, and interactions with the surrounding urban environment, should be regarded as context-dependent and cannot be generalized to all urban settings. Rather, their applicability is likely limited to sites with broadly similar climatic, geographical, and environmental conditions, particularly in terms of soil, hydrology, microclimate, and management regime. The findings on microforest growth and performance may be cautiously extended to other urban contexts sharing similar climatic and geographical characteristics (Po Valley region) as the study site is broadly representative of the area in terms of climate, urban structure, and park size. One of the main gaps this study aimed to address is the absence – to date- of studies fully comparable in terms of experimental design and territorial context, especially in the Po Valley, which further constrains direct comparison.

Similarly, the biodiversity results may be considered indicative of the early ecological development of small constructed wetland systems, a condition that may reasonably occur in comparable urban settings, as shown in other studies (Cannucci et al. 2025; Martín Muñoz et al. 2023).

By contrast, the microclimatic results appear to be more strongly context-dependent, as they are influenced by local spatial configuration, vegetation structure, and the characteristics of the surrounding built environment. At the same time, the present study provides a broader perspective by including soil-related measurements in addition to aboveground microclimatic variables; to our knowledge, such data are still lacking for this type of intervention within the same regional context. Moreover, previous studies on similar NBS have reported broadly consistent microclimatic effects across different urban contexts, suggesting that, despite the site-specific nature of this case study, some of the trends observed here may have wider relevance.

This work should be regarded as a representative empirical case study that contributes site-based evidence to the still limited quantitative assessments of NBS in Mediterranean or warm-temperate urban contexts. In this respect, our results are consistent with emerging studies on Miyawaki forests (Morales et al. 2026; Schirone et al. 2025) and urban green infrastructure in Italy, which similarly highlight the importance of local climate, vegetation structure, and design configuration in shaping ecological and microclimatic outcomes.

Conclusions

This study provides an integrated evaluation of structural development, floristic dynamics and microclimatic regulation in a multifunctional Nature-Based Solution system implemented at M. Biagi Park (Reggio Emilia, Italy). By combining two contrasting Miyawaki microforests (Native and Adaptive) with a newly established constructed wetland and other green devices, the research offers a controlled comparison of NBS components under identical urban conditions. The comparison between the Native Forest (NF) and the Adaptive Forest (AF) revealed broadly similar survival rates and overall growth performance three years after planting, confirming the effectiveness of high-density Miyawaki plantations in promoting rapid early establishment. However, differences emerged in biomass allocation and structural traits. AF species—particularly Mediterranean oaks—showed relatively greater relative diameter increment and lower slenderness ratios, suggesting earlier investment in mechanical stability and potentially more conservative functional strategies. In contrast, several NF species displayed higher slenderness and greater inter-individual variability, reflecting more height-oriented growth dynamics. These structural divergences, observed under identical site conditions, indicate that species functional traits influence early-stand architecture even when overall ecosystem services delivery remains comparable.

From a microclimatic perspective, both microforests consistently provided lower air and soil temperatures and higher relative humidity compared with the surrounding built-up areas. Differences between NF and AF were limited and varied between years, suggesting that, at this early development stage, canopy density and stand structure play a stronger role than species origin in determining cooling intensity.

The wetland component showed a dynamic biodiversity patterns. Within two years from its creation, species richness increased substantially, and a clear functional zonation along hydrological gradients became established. The coexistence of aquatic, helophytic and ruderal taxa reflects an active and rapid community assembly process. Notably, the marked increase in Mediterranean and Euri-Mediterranean chorotypes suggests a potential early signal of climatic filtering in this artificial urban ecosystem.

The results suggest that multifunctional NBS systems may offer early benefits in terms of vegetation establishment, biodiversity support, and summer thermal mitigation. However, these findings remain preliminary and should be interpreted with caution, given the short monitoring period. The observed differences between native and adaptive mocroforests point potentially relevant functional dynamics, but longer-term monitoring is needed to determine whether these early patterns will persist and result in differentiated resilience and ecosystem service provision over time.

Acknowledgements The Authors are also grateful to Daniela Mordacci and all the technical staff of the Municipality of Reggio Emilia for involving us in this project and for their availability and support during the experimental data collection. Financial contribution from the PRD2026 from the Department of Life Sciences, University of Modena and Reggio Emilia, Italy is gratefully acknowledged.

Author contributions E.S and G.S. conceived of the presented idea. E.S. developed the theory .E.S. verified the analytical methods. E.S. and F.R. supervised the findings of this work.G.S and F.Z and F.R. performed the computations.G.S. and F.Z. carried out the experiment. G.S and F.Z. wrote the manuscript.G.S and F.Z. and F.R. derived the models and analysed the data. F.Z. assisted with measurements and G.S. helped carry out the simulations. G.S and F.Z prepared the Figures.All authors discussed the results and contributed to the final manuscript.

Funding Open access funding provided by Università degli Studi di Modena e Reggio Emilia within the CRUI-CARE Agreement.

Data availability The data that support the findings of this study are available from the corresponding author upon reasonable request.

Declarations

Competing interests The authors declare no competing interests.

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