JEE
ISSN: 2288-1220
https://doi.org/10.5141/jee.25.033
Journal of Ecology and Environment (2025) 49:19

RESEARCH

Epiphytic lichen as a bioindicator of air pollution across selected urban, suburban, and rural areas in Malaysia

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ARTICLE INFO
Received April 24, 2025
Revised July 24, 2025
Accepted July 24, 2025
Published on August 19, 2025

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Background: Lichens are widely recognized as bioindicators of air pollution due to their sensitivity to atmospheric contaminants. However, species-level responses in Southeast Asia, particularly in Malaysia, remain underreported. This study investigates the relationship between lichen diversity and ambient air pollution across three land-use types in Peninsular Malaysia: urban (Shah Alam), suburban (Jerantut), and rural (Kuala Selangor).
Results: Lichen samples were collected from 10 *Mimusops elengi* trees per site using 20 cm × 30 cm quadrats and identified morphologically and chemically. Air quality data (PM_{10}, $PM_{2.5}$, NO_2, SO_2, O_3, and CO) from 2023 were obtained from the Department of Environment Malaysia. Urban sites exhibited the highest pollution levels, while suburban areas had the lowest. Interestingly, the rural site showed moderately high pollution levels, possibly due to local sources such as open burning, nearby markets and coastal aerosol. Correspondingly, the suburban site recorded the highest lichen diversity and index of atmospheric purity (IAP = 37.83). A strong negative correlation between IAP and PM_{10} ($r = -0.957$) underscores the impact of fine particulates on lichen community structure. Environmental variables such as bark pH, temperature, and humidity also influenced species distribution but played a secondary role compared to air pollutants.
Conclusions: This study provides evidence that epiphytic lichen communities respond clearly to varying levels of atmospheric pollution, supporting their use as low-cost, effective bioindicators in tropical regions. While lichen sampling was conducted as a one-time event, the results remain ecologically meaningful due to lichens' long-term integration of environmental conditions. These findings offer a valuable baseline for future ecological assessments and contribute to the development of long-term lichen-based air quality monitoring frameworks in Malaysia.

Keywords: air pollution, bioindicators, diversity, index of atmospheric purity, lichens

Introduction

Rapid urbanization and industrial growth are exacerbating environmental challenges globally, particularly air pollution. Airborne pollutants such as particulate matter (PM_{10}, $PM_{2.5}$), nitrogen dioxide (NO_2), sulphur dioxide (SO_2), ozone (O_3), and carbon monoxide (CO) pose significant threats to both human health and biodiversity (Belguidoum et al. 2022). These pollutants originate from industrial emissions, vehicular exhaust, and combustion processes, leading to air quality degradation and ecosystem disturbances especially in urban areas (Barzeghar et al. 2022). Epiphytic lichens are among the organisms most impacted by air pollution, serving as key indicators in ecological monitoring due to their exceptional sensitivity to changes in atmospheric contaminants.

Lichens, symbiotic relationships between fungi and algae or cyanobacteria, are widely recognized as bioindicators of air quality because they directly absorb atmospheric nutrients and pollutants. Unlike vascular plants, lichens lack protective features such as cuticles or stomata, making them particularly vulnerable to airborne contaminants in their surroundings (Shelyakin et al. 2024; Thakur et al. 2024). Their physiological traits allow them to reflect long-term air quality conditions in a given environment.

Air pollution levels vary significantly between urban, suburban, and rural areas, leading to different ecological conditions. Urban areas typically have the highest concentrations of pollutants due to industrial activities, vehicle emissions, and dense populations (Castells-Quintana et al. 2021). Suburban and rural sites generally experience lower pollution levels, allowing for greater lichen diversity and richness (Lucheta et al. 2019). This pollution gradient influences lichen species composition and abundance, with pollution-tolerant species persisting in contaminated areas while sensitive species disappear.

Numerous studies have documented shifts in lichen community composition in response to air pollution and climate factors (Abas et al. 2019; Ismail et al. 2017). Lichens are particularly suitable for assessing pollution gradients, as their presence, diversity, and abundance reflect pollution intensity over time. Different species exhibit varying tolerance thresholds, making them useful for identifying areas of low and high pollution exposure (Harikrishna and Mukherjee 2024; Vitali et al. 2019). Their accessibility and long lifespan make lichens a cost-effective and reliable tool for tracking long-term pollution trends (Benítez et al. 2019).

The index of atmospheric purity (IAP) is a widely used metric for quantifying the impact of air pollution on lichen communities. The IAP relies on phytosociological indicators to assess air quality, particularly in urban areas. This method assumes that the presence and growth of epiphytic lichens are primarily influenced by air pollution levels (Tanona and Czarnota 2020). The IAP expresses air quality as an index based on lichen diversity and incorporates an ecological value for each species, reflecting their sensitivity or tolerance to atmospheric contaminants (Silprasit et al. 2023).

Developed by LeBlanc and Sloover (1970), higher IAP values indicate better air quality, characterized by a greater diversity of pollution-sensitive lichens, whereas lower values suggest pollution stress and biodiversity loss. Previous studies have shown that urban areas generally exhibit lower IAP values and reduced lichen diversity due to higher pollutant concentrations, while suburban and rural sites display higher IAP values and support more diverse lichen communities (Jayalal et al. 2016; Yatawara and Dayananda 2019).

Despite growing interest in lichen-based air pollution monitoring, studies in Malaysia that examine lichen diversity across urban–rural gradients remain limited. Malaysia's rapidly changing landscape and diverse vegetation make it an important setting for community-level assessments of lichen response to environmental quality. However, baseline data on epiphytic lichen communities across different land-use types are still scarce.

This study investigates lichen diversity and community composition across selected urban, suburban, and rural sites in Peninsular Malaysia and examines their relationship with ambient air pollution. In addition to long-term air quality data, we recorded on-site environmental parameters such as bark pH, temperature, and humidity to provide ecological context. The results contribute important baseline information for future biomonitoring efforts and support the application of lichen community patterns as indicators of air quality in tropical environments.

Materials and Methods

Study area

This study was conducted in three selected areas, each representing a different land-use category based on classification by the Department of Environment (DOE) Malaysia from 2017 to the present. The urban site was located in Shah Alam, Selangor; the suburban site in Jerantut, Pahang; and the rural site in Kuala Selangor, Selangor. Table 1 shows the selected sampling sites and their coordinates.

Site selection was based on proximity to Continuous Air Quality Monitoring (CAQM) stations, which provide long-term, standardized pollutant data. These stations are part of a national DOE network of 51 CAQM sites across residential, industrial, and traffic-related areas, designed to detect variations in air quality that may affect health and the environment (Department of Environment Malaysia 2010). Figure 1A shows a CAQM monitoring station, while Figure 1B shows the national CAQM network in Peninsular Malaysia. A detailed map of the study locations is provided in Figure 2.

The selected sites also reflect different land-use classifications defined by the DOE. Shah Alam, the capital of Selangor, is a densely populated urban center (population ~438,745; Department of Statistics Malaysia 2023) covering

Table 1 Selected study site and their coordinates

Station	Classification	Sampling site	Latitude	Longitude
Shah Alam	Urban	SK TTDI Jaya	03° 06′ 16.98″ N	101° 33′ 22.39″ E
Jerantut	Suburban	SMK Jerantut	03° 56′ 54.09″ N	102° 21′ 59.87″ E
Kuala Selangor	Rural	SM Sains Kuala Selangor	03° 19′ 16.70″ N	101° 15′ 22.47″ E

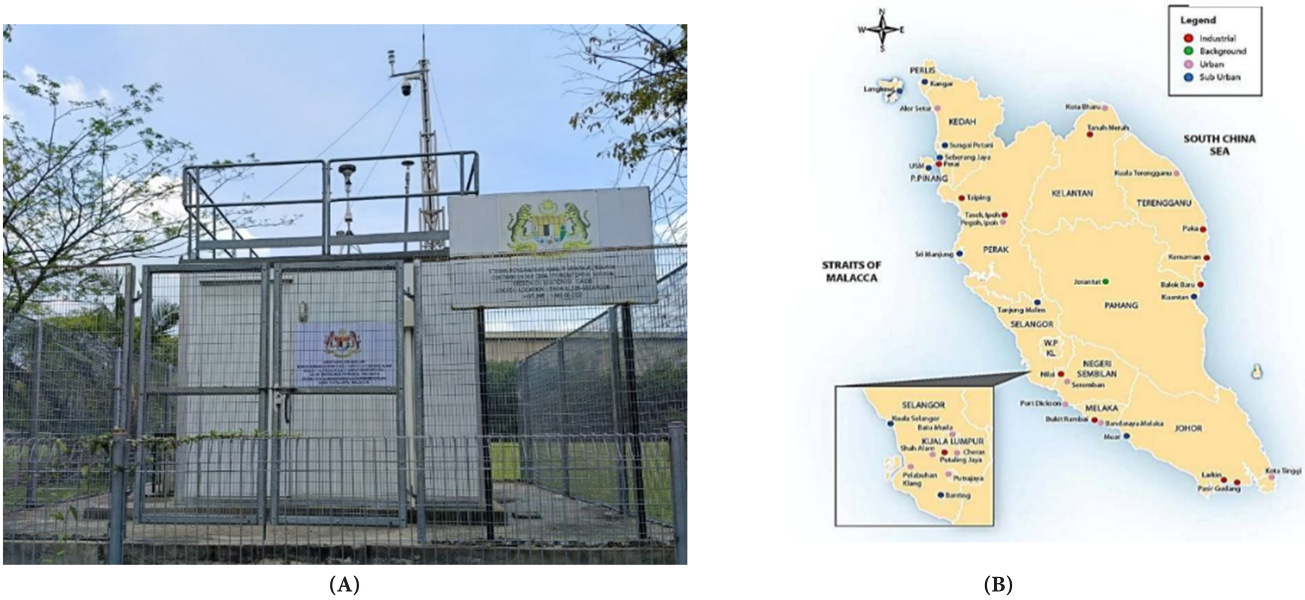

Fig. 1 (A) Shah Alam CAQM Station and (B) CAQM location in Peninsular Malaysia (Department of Environment Malaysia 2010). CAQM: Continuous Air Quality Monitoring.

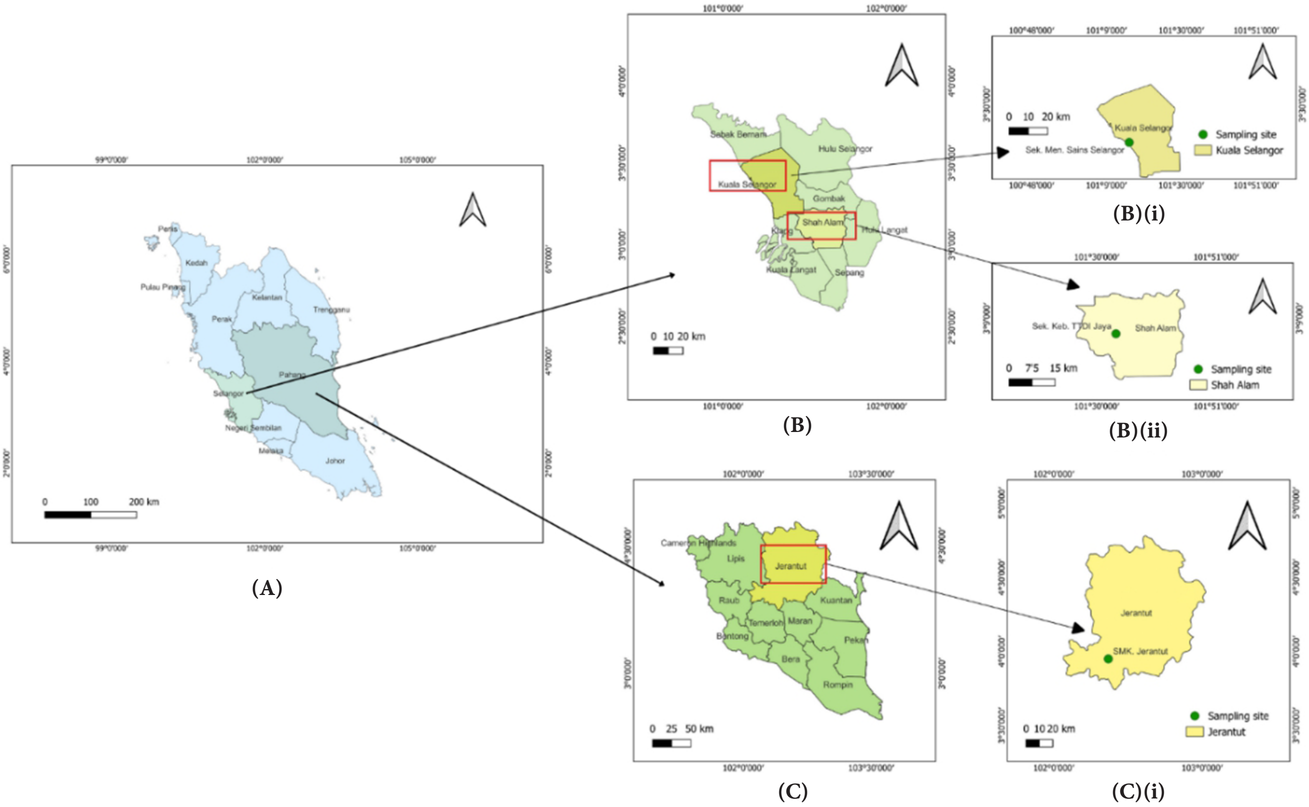

Fig. 2 Maps show the sampling sites (QGIS Desktop 3.34.10, 2024). (A) Peninsular Malaysia, (B) Selangor State—(B)(i) Kuala Selangor District, (B)(ii) Shah Alam District, and (C) Pahang State—(C)(i) Jerantut District.

290.3 km^2 with annual rainfall of ~2,374 mm. Jerantut, a suburban town in central Pahang (~96,006 residents), serves as a gateway to Taman Negara and spans 7,563 km^2 with rainfall of ~2,093 mm. Kuala Selangor, a rural district covering 1,195 km^2, has a population of ~171,266 and annual rainfall of ~2,062 mm (Department of Statistics Malaysia 2023).

In addition to land-use classification, the immediate surroundings of each sampling location were noted, as these micro-scale environmental conditions may influence pollution exposure and bioindicator responses. The urban site in Shah Alam was situated in a busy commercial and residential area with nearby factories and industrial zones. The suburban site in Jerantut was located in a quiet residential neighborhood, surrounded by houses and a school with minimal traffic or commercial activity. In contrast, the rural site in Kuala Selangor was located near the coast, adjacent to open areas, small shops and a night market, with visible signs of rubbish burning activity.

Lichen sampling and identification

Lichen samples were collected from ten mature trees of the same species, *Mimusops elengi*, at each site. Sampling was conducted during the dry season to minimize seasonal variability: in April 2024 for urban sites, June 2024 for suburban sites, and August 2024 for rural sites. The selection of trees was based on their accessibility, structural stability, proximity to pollution sources (such as roads or developed areas), and their location within a 500-meter radius of the nearest CAQM station.

To minimize habitat-related variability, all selected trees had a minimum diameter of 80 cm at breast height and shared similar bark texture. Sampling was performed on the side of the trunk facing the dominant pollution source. A 20 × 30 cm flexible plastic quadrat marked with grid lines (Fig. 3) was used to standardize sampling and was positioned 1–3 meters above ground level. Lichen thalli within each grid were carefully removed using a ceramic knife or chisel, ensuring intact collection for accurate identification. Samples were stored in labeled paper envelopes and air-dried before identification.

Lichen identification was performed using both a stereomicroscope and compound light microscope, focusing on morphological characteristics such as thallus form, reproductive structures, and surface features. To support morphological determination, standard chemical spot tests were conducted to detect the presence of secondary metabolites. The K test (10%–25% potassium hydroxide), C test (freshly prepared calcium hypochlorite), KC test (K followed by C), and P test (1%–5% para-phenylenediamine in ethanol) were applied, with color reactions recorded immediately.

Basic environmental parameters were also recorded at each sampling point. Bark pH was determined by mixing 0.5 g of bark powder with 10 mL of deionized water, shaking intermittently for one hour and measuring the pH of the filtered solution using a Mettler Toledo MP 230 pH meter. Ambient temperature and relative humidity were measured in situ at the time of sampling using a UNI-T UT333S Digital Temperature and Humidity Meter.

Air pollutant data

Air quality data for PM_{10}, $PM_{2.5}$, NO_2, SO_2, O_3, and CO were obtained from the DOE Malaysia through its network of CAQM stations. These stations provide standardized, long-term measurements of key atmospheric pollutants for environmental monitoring and public health reporting, including the development of the air pollutant index (API).

For this study, monthly mean concentrations from the year 2023 were compiled to represent recent air quality conditions prior to lichen sampling, which was conducted in 2024. Although the DOE provides pollutant data across multiple years, 2023 was selected as it was the most recent complete dataset available at the time of analysis. Air quality data for 2024 were not yet published in the DOE's Environmental Quality Report when this study was finalized.

Each study site was matched with its nearest CAQM station. The Shah Alam (urban) and Jerantut (suburban) stations recorded data for all six pollutants: PM_{10}, $PM_{2.5}$, NO_2, SO_2, O_3, and CO. In contrast, the Kuala Selangor station

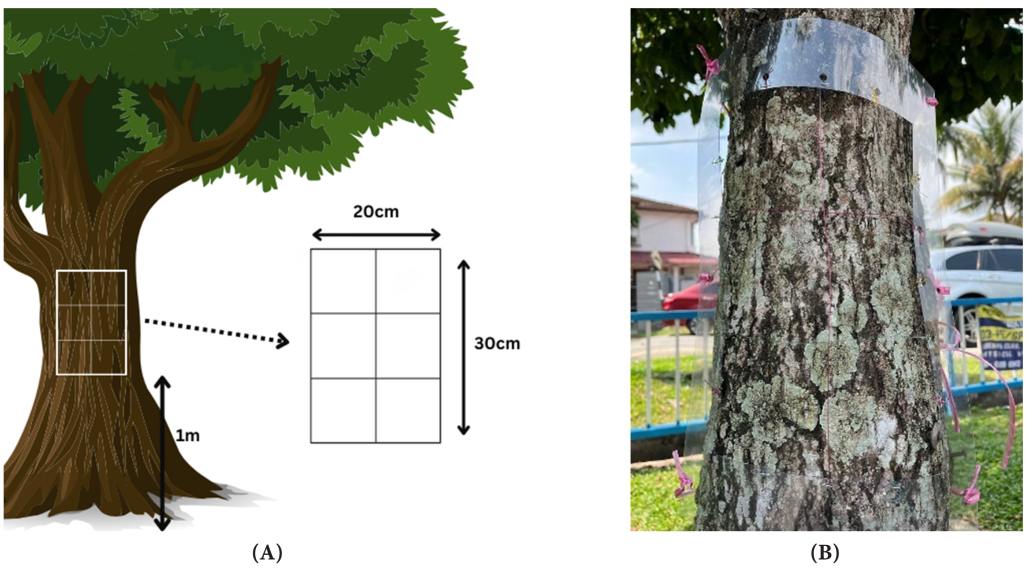

Fig. 3 Sampling method using 20 × 30 cm quadrat placed on the tree bark. (A) Illustration of quadrat placement 1 m above the ground. (B) Photograph of the quadrat used.

(rural) reported only PM_{10} and $PM_{2.5}$ levels, consistent with DOE monitoring practices in low-emission areas, where particulate matter typically dominates the API.

Although lichen sampling was conducted as a one-time event, the use of 2023 air pollutant data is ecologically justified. Lichens are slow-growing, long-lived organisms that passively accumulate airborne pollutants over time. Therefore, a single sampling point can reliably reflect the effects of chronic air pollution, particularly when interpreted alongside recent data from standardized air quality monitoring networks.

Determination of IAP and lichen diversity

Phytosociological data from each sampling site were used to calculate the IAP following the method proposed by LeBlanc and Sloover (1970). The IAP serves as a composite measure of air quality based on the presence, distribution, and ecological behavior of lichen species on host trees. The index was calculated using the formula:

$$IAP = \sum_{i}^{n} \frac{Q \times f}{10}$$

where n is the total number of lichen species recorded at the site, Q is the ecological index of a species (defined as the average number of species that coexist with it across all trees), and f is the number of sampled trees on which the species was present. The factor of 10 is used to scale the index for easier interpretation.

Air quality levels based on IAP values were interpreted using the classification system proposed by Conti and Cecchetti (2001), as shown in Table 2.

To further assess lichen community structure, two diversity indices were calculated: the Shannon–Wiener index (H') and the Simpson index (D). The Shannon–Wiener index accounts for both species richness (total number of species) and evenness (distribution of individuals among species). It was calculated using the formula:

$$H' = -\sum_{i=1}^{S} p_i \ln(p_i)$$

where S is the total number of species, and p_1 is the proportion of individuals of species i relative to the total number of individuals.

The Simpson index (D) reflects species dominance by estimating the probability that two individuals randomly selected from a sample belong to the same species. It was calculated as:

$$D = 1 - \frac{\sum n(n-1)}{N(N-1)}$$

where n is the number of individuals of species and N is the total number of individuals across all species. Lower D values indicate higher diversity, as dominance by one or a few species is reduced. These indices offer complementary perspectives on lichen diversity and are useful in interpreting the ecological responses of lichen communities to varying levels of atmospheric pollution.

Table 2 Air quality levels based on the IAP (Conti and Cecchetti, 2001)

Pollution level	IAP range	Air quality classification
Level A	$0 \leq IAP \leq 12.5$	Very high level of pollution
Level B	$12.5 < IAP \leq 25$	High level of pollution
Level C	$25 < IAP \leq 37.5$	Moderate level of pollution
Level D	$37.5 < IAP \leq 50$	Low level of pollution
Level E	$IAP > 50$	Very low level of pollution

IAP: index of atmospheric purity.

Statistical analysis

Spatial maps of the study sites were generated using QGIS Desktop version 3.34.10. All statistical analyses were conducted using IBM SPSS Statistics version 30.0. Prior to hypothesis testing, data were assessed for normality using the Shapiro–Wilk test and for homogeneity of variances using Levene's test.

Differences in air pollutant concentrations across the three land-use types were analyzed using one-way ANOVA, followed by Tukey's post hoc test for multiple pairwise comparisons. Where appropriate, independent sample t-tests were applied to compare specific variables between two groups.

Lichen species diversity indices, including the Shannon–Wiener index and Simpson index, were calculated using PAST software version 4.03 (University of Oslo, Norway). A heatmap showing Pearson's correlation coefficients (r) between environmental variables and IAP across the three sampling sites was generated using GraphPad Prism version 10.5.0. All statistical results are reported as mean ± standard error of the mean, and significance was determined at the 0.05 level unless stated otherwise.

Results

Comparison of pollutant data

To ensure temporal alignment with the one-time lichen sampling conducted in 2024, comparative analyses were based on air pollutant data from the most recent available year (2023), as provided by the DOE. Figure 4 presents the differences in PM_{10} and $PM_{2.5}$ concentrations across urban, suburban, and rural site.

For PM_{10}, the rural site recorded the highest mean concentration (32.76 ± 2.59 $\mu g/m^3$), followed by the urban (26.67 ± 2.10 $\mu g/m^3$) and suburban (21.96 ± 1.72 $\mu g/m^3$)

sites. A one-way ANOVA indicated a significant difference across sites ($F = 6.248$, $p = 0.005$), and Tukey's post hoc test confirmed that the rural site had significantly higher PM_{10} levels than the suburban site ($p = 0.004$). However, differences between urban and rural, and between urban and suburban sites, were not statistically significant.

For $PM_{2.5}$, the highest concentration was observed in the urban site (22.07 ± 2.02 $\mu g/m^3$), followed by the rural (18.33 ± 1.46 $\mu g/m^3$) and suburban (13.69 ± 1.48 $\mu g/m^3$) sites. ANOVA revealed a significant difference ($F = 6.273$, $p = 0.005$), with post hoc analysis indicating that $PM_{2.5}$ levels in the urban site were significantly higher than in the suburban site ($p = 0.003$). No significant differences were found between the urban and rural sites, or between rural and suburban sites.

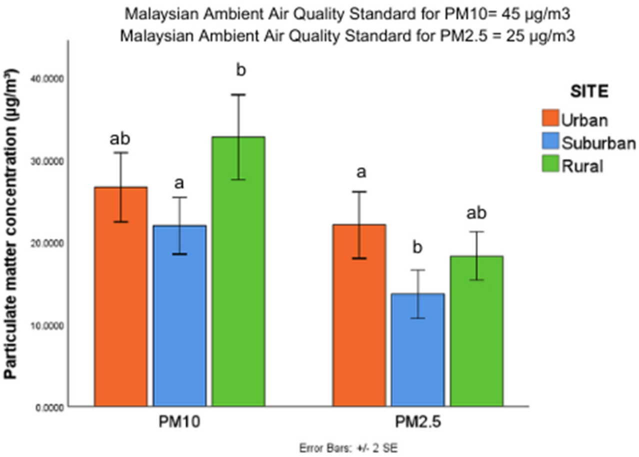

Fig. 4 PM_{10} and $PM_{2.5}$ concentrations.

Figure 5 shows the 2023 concentrations of SO_2, NO_2, CO, and O_3 in urban and suburban areas. The rural site was excluded from this comparison due to unavailable data for these pollutants.

The mean SO_2 concentration was significantly higher in the urban area (0.0014 ± 0.0001 ppm) than in the suburban area (0.0010 ± 0.0001 ppm), with an independent t-test confirming this difference ($t(22) = 4.161$, $p < 0.001$).

NO_2 levels exhibited the most substantial disparity, with the urban area recording a much higher mean concentration (0.0149 ± 0.0005 ppm) compared to the suburban area (0.0028 ± 0.0005 ppm), yielding a highly significant result ($t(22) = 23.396$, $p < 0.001$). Similarly, CO concentrations were significantly higher in the urban site (0.7282 ± 0.0350 ppm) than in the suburban site (0.4346 ± 0.0350 ppm) ($t(22) = 8.405$, $p < 0.001$).

In contrast, O_3 concentrations did not differ significantly between the urban (0.0184 ± 0.0016 ppm) and suburban (0.0162 ± 0.0016 ppm), ($t(22) = 1.432$, $p = 0.166$), indicating comparable O_3 levels in both environments.

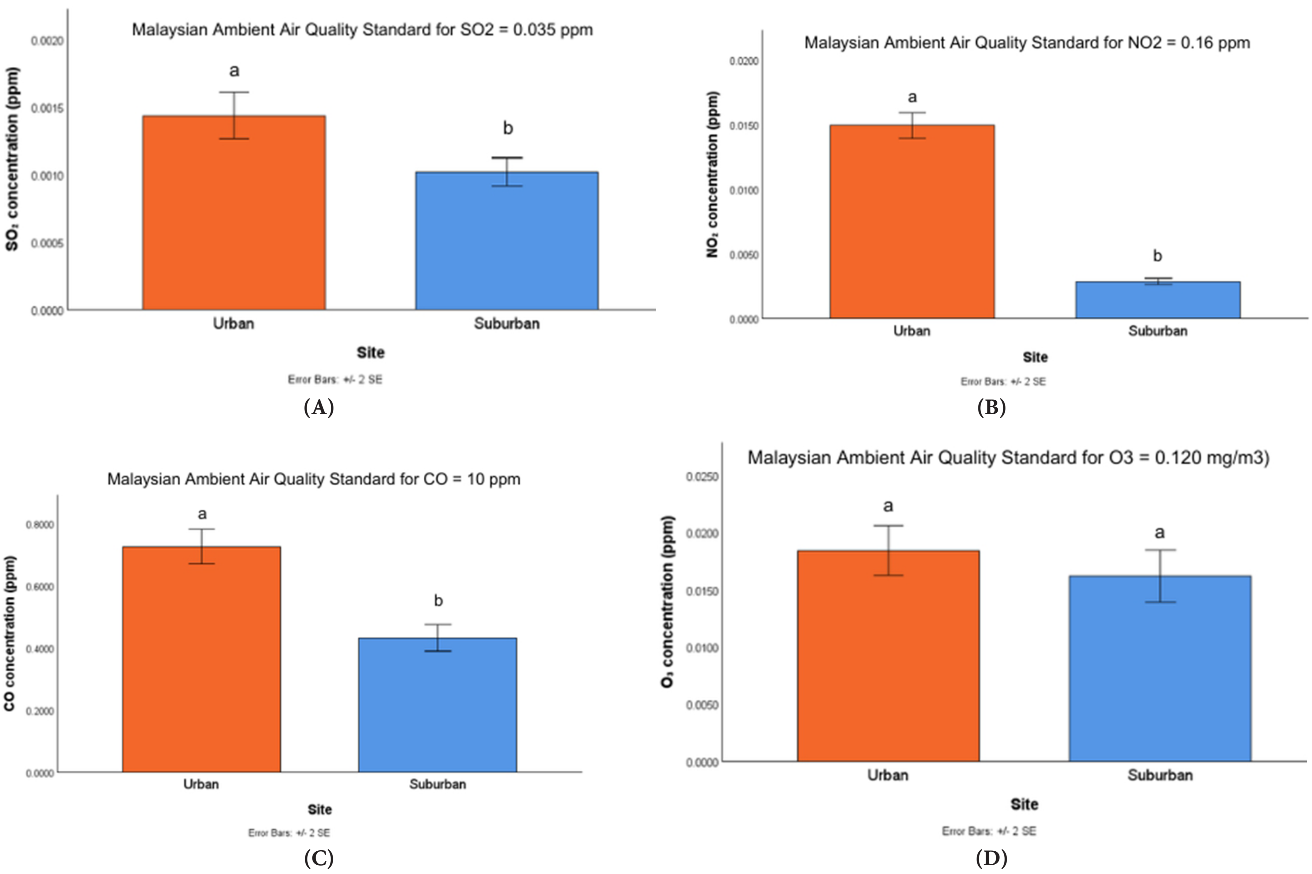

Fig. 5 Concentration of (A) SO_2, (B) NO_2, (C) CO_3 and (D) O_3 between urban and suburban site.

Lichen diversity

A total of 16 lichen species, representing 12 genera and 11 families, were identified across the three study sites. The family Graphidaceae was the most represented, contributing four species. The lichen assemblage was dominated by crustose forms, with 13 crustose and three foliose species recorded.

Species richness varied across sites. The urban site exhibited the highest richness (12 species), followed by the suburban site (10 species), and the rural site (7 species). Crustose lichens were prevalent at all sites, particularly in the urban area, where they accounted for 10 out of 12 recorded species. A summary of species presence and absence is provided in Table 3.

Figure 6 illustrates the abundance of species in each site. In terms of total individuals, the suburban site recorded the highest number (1,327 individuals, or 44.06% of all individuals), followed by the rural (972 individuals, 32.27%) and urban (713 individuals, 23.67%) sites. Several species were commonly encountered across all sites, including *Dirinaria applanata*, *Phlyctis argena*, and *Arthonia albovirescens*.

Species composition varied across the three sites. In the suburban site, dominant species included *Graphis scripta* (30.60%), *P. argena* (19.59%), and *Lecidea lithophila* (16.13%). The urban site was dominated by *D. applanata* (31.42%), *Graphis casiella* (22.86%), and *Buellia erubescens* (17.53%). In contrast, the rural site showed a marked domi-

Table 3 Lichen species in each site

Thallus	Families	Genera	Species	Urban	Suburban	Rural
Crustose	Graphidaceae	Graphis	*Graphis librata*	+	–	–
			Graphis scripta	+	+	–
			Graphis intricans	–	+	+
			Graphis casiella	+	–	–
	Arthoniaceae	Arthonia	*Arthonia albovirescens*	+	+	+
	Phlyctidaceae	Phlyctis	*Phlyctis argena*	+	+	+
	Lecideaceae	Lecidea	*Lecidea lithophila*	–	+	–
	Psoraceae	Protoblastenia	*Protoblastenia rupestris*	–	+	–
	Caliciaceae	Amandinea	*Amandinea punctata*	+	–	–
		Buellia	*Buellia erubescens*	+	–	+
		Pyxine	*Pyxine cocoes*			
	Lecanoraceae	Lecanora	*Lecanora helva*			
			Lecanora laxa			
	Chrysothricaceae	Chrysothrix	*Chrysothrix candelaris*	+	+	–
	Lepariaceae	Lepraria	*Lepraria incana*	+	–	+
	Leprocaulaceae	Leprocaulon	*Leprocaulon microscopicum*	+	–	–
Foliose	Parmeliaceae	Parmotrema	*Parmotrema tinctorum*	–	+	–
	Caliciaceae	Dirinaria	*Dirinaria applanata*	+	+	+
			Dirinaria picta	+	+	+
Species richness				12 lichens (10 crustose, 2 foliose)	10 lichens (7 crustose, 3 foliose)	7 lichens (5 crustose, 2 foliose)

+: present; –: absent.

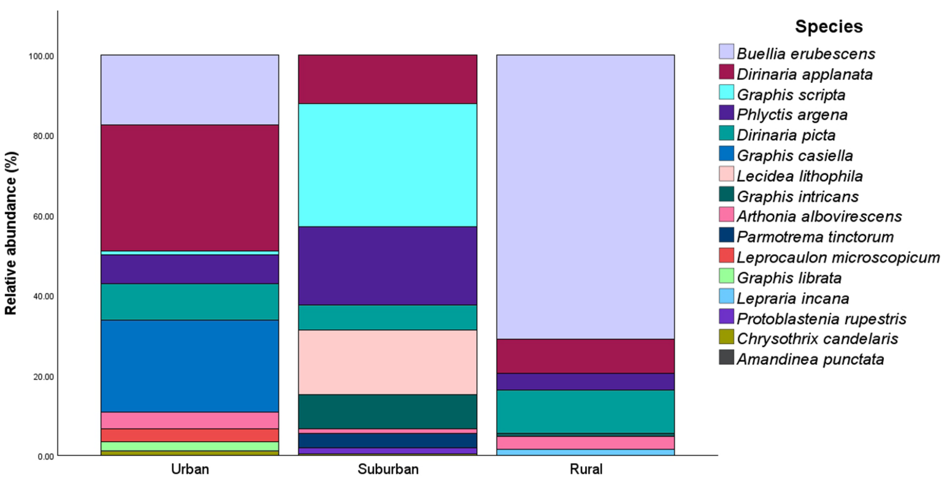

Fig. 6 Relative species abundance in urban site, suburban site and rural site.

nance of *B. erubescens* (70.88%), with lesser contributions from *Dirinaria picta* (10.80%) and *D. applanata* (8.64%). Differences in community structure were evident across sites, as indicated by metrics of dominance, richness, and evenness (Table 4).

The suburban site exhibited the highest species diversity and balance, with the lowest dominance value (D = 0.1862), highest Simpson's index (D' = 0.8138), and greatest Shannon diversity (H' = 1.863). Evenness was also highest in this site (E = 0.7156), indicating a more balanced distribution of individuals across species.

The urban site showed moderately high diversity (D' = 0.7985, H' = 1.825), though evenness was slightly lower (E = 0.6204). In contrast, the rural site had the lowest diversity and evenness. This site showed a high dominance index (D = 0.5169), with the lowest values for D' (0.4831), H' (1.060), and E (0.4125), indicating that a few species dominated the community.

These findings highlight that lichen communities in suburban environments tended to be more diverse and balanced compared to both urban and rural settings.

Correlation between IAP and environmental variables

The IAP, derived from lichen community composition and abundance, serves as an indirect but ecologically relevant measure of air quality, with higher values indicating lower levels of atmospheric pollution.

Among the three sites, the suburban site recorded the highest IAP value (37.83), followed by the rural site with 27.07, and the lowest was at the urban site with 24.53. Based on Conti and Cecchetti (2001) classification, the suburban site falls within Level D (37.5 < IAP ≤ 50), indicating low pollution. The rural site falls within Level C (25 < IAP ≤ 37.5), corresponding to moderate pollution, while the urban site is classified as Level B (12.5 < IAP ≤ 25), reflecting high pollution conditions.

To better understand the influence of local conditions on lichen community patterns, microclimatic variables were measured at each site. These include bark pH, temperature, and relative humidity. Table 5 presents the mean values of these environmental factors across the urban, suburban, and rural sites.

To assess how these local environmental conditions may influence lichen diversity and IAP, a Pearson correlation analysis was conducted. The results, presented as a heatmap in Figure 7, illustrate the relationships between IAP values and selected environmental parameters across the study sites. The analysis revealed several strong associations, indicating that IAP is influenced by both air pollutants and local environmental conditions.

Table 4 Dominance index (D), Simpson's index (D'), Shannon–Wiener diversity index (H'), and index of evenness (E)

Index	Value index		
	Urban	Suburban	Rural
Dominance (D)	0.2015	0.1862	0.5169
Simpson (D')	0.7985	0.8138	0.4831
Shannon (H')	1.825	1.863	1.06
Evenness (E)	0.6204	0.7156	0.4125

Table 5 Mean ± standard error of the mean of bark pH, temperature, and relative humidity across study sites

Site	Bark pH	Temperature (°C)	Relative humidity (%)
Urban	3.85 ± 0.10	30.03 ± 0.49	67.85 ± 3.10
Suburban	8.33 ± 0.16	34.94 ± 0.68	63.87 ± 2.09
Rural	6.66 ± 0.19	31.15 ± 0.51	68.60 ± 1.91

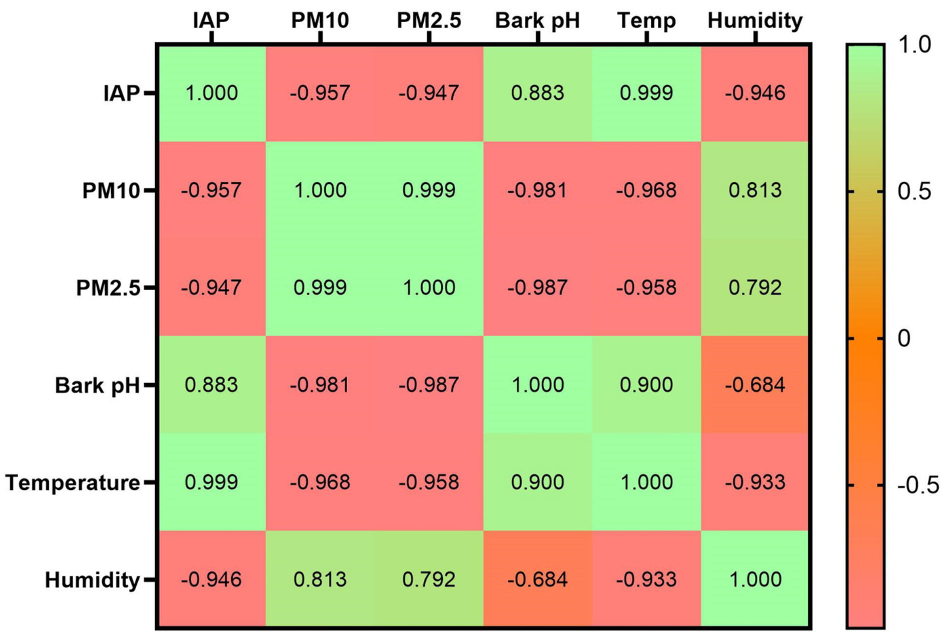

Fig. 7 Heatmap showing Pearson's correlation coefficients (*r*) between environmental variables and the index of atmospheric purity (IAP) across three sampling sites. Color gradient represents the strength and direction of correlation: green indicates strong positive correlation (*r* = +1) and red indicates strong negative correlation (*r* = –1).

Correlation analysis revealed a very strong negative relationship between IAP and particulate matter concentrations, specifically PM_{10} (r = −0.957) and $PM_{2.5}$ (r = −0.947), indicating that higher particulate concentrations are associated with lower IAP values due to reduced lichen diversity in more polluted environments. A strong positive correlation was also observed between IAP and bark pH (r = 0.883), with higher pH values associated with higher IAP scores.

Additionally, IAP was positively correlated with temperature (r = 0.999) and negatively correlated with relative humidity (r = −0.946), suggesting that local microclimatic factors also influence lichen community structure and air purity scores. The analysis highlights that IAP is primarily shaped by atmospheric pollutant concentrations, particularly PM_{10} and $PM_{2.5}$, underscoring its relevance as a lichen-based index of air quality. Nonetheless, strong correlations with bark pH, temperature, and relative humidity suggest that local environmental conditions may also influence lichen community structure and thereby may indirectly affect IAP values.

Discussion

Air quality analysis across the three sampling sites revealed marked spatial variation in both particulate and gaseous pollutants, with urban areas exhibiting the highest concentrations of $PM_{2.5}$, NO_2, SO_2, and CO. These elevated levels are attributed to high traffic volume, dense population, and industrial activities. In contrast, suburban sites recorded the lowest pollutant levels, reflecting a less industrialized landscape with reduced anthropogenic pressure.

Interestingly, in 2023, PM_{10} concentrations were highest in the rural site, surpassing both urban and suburban sites. This irregularity likely reflects episodic and localized influences, such as open biomass burning in agricultural fields and resuspension of coarse particles intensified by land–sea breeze circulation typical of Kuala Selangor's coastal geography (Asmat et al. 2018; Department of Environment Malaysia 2023). However, this does not contradict the long-term trend, where DOE annual reports show that urban areas consistently exhibit the highest mean concentrations of most pollutants over multiple years, reinforcing their status as chronically polluted environments.

More notably, rural sites consistently exhibited higher pollutant concentrations than suburban sites across multiple years, which deviates from the typical expectation that air pollution decreases progressively from urban to suburban to rural areas. This unexpected trend may be explained by the specific environmental and anthropogenic characteristics of the rural sampling location in Kuala Selangor. The site was situated near the coastline and surrounded by open areas, small-scale commercial premises and a night market. During field visits, instances of open rubbish burning were observed in the vicinity, potentially contributing to elevated levels of airborne particulates. In contrast, the suburban site was located within a quiet residential area characterized by low traffic density, the absence of commercial activity, and no visible combustion sources, thereby supporting its consistently lower pollutant concentrations.

These air pollution patterns strongly influenced lichen community structure and composition. The urban site, despite its higher pollution load, exhibited the greatest species richness, likely due to the dominance of pollution-tolerant taxa. However, this richness did not translate into greater ecological balance, as diversity and evenness metrics were lower compared to the suburban site. The rural site, though less polluted in the long term, was dominated by a few abundant species, notably *B. erubescens*, resulting in lower overall richness and evenness.

Crustose lichens made up over 80% of the identified species, highlighting their adaptive advantages under polluted or stress-prone conditions. Their tightly adherent growth form, structural resilience, and tolerance to desiccation and surface pollutants make them well-suited for colonizing bark in urban environments. Conversely, more pollution-sensitive foliose species were rare and mostly confined to the suburban site. This is consistent with previous studies (Das et al. 2020; Tumur et al. 2025), which reported crustose lichens as the most pollution-tolerant growth form compared to foliose and fruticose lichens.

Species-specific responses further illustrate ecological filtering along pollution gradients. Tolerant species like *D. picta* and *G. casiella* were dominant in urban sites, where they withstand exposure to NO_2, SO_2, and vehicular emissions. The high abundance of *D. picta* in the urban area aligns with previous findings by Khairuddin et al. (2017), which reported its resistance to traffic-related pollutants.

In contrast, species such as *Parmotrema tinctorum* and *L. lithophila*, known to be sensitive to atmospheric pollutants, were found exclusively in the suburban site. The occurrence of *P. tinctorum* only in this less polluted area aligns with its known sensitivity to air pollution (Raimundo-Costa et al. 2024), further highlighting the suburban site as a favorable habitat for pollution-sensitive species.

Moreover, the distribution of *P. argena* and *G. scripta*, more abundant in suburban areas, aligns with their known sensitivity to nitrogenous pollutants such as NO_2, which were lower in those environments (Okon et al. 2024).

IAP varied across the sites in a pattern consistent with pollution levels and lichen community diversity. The suburban site recorded the highest IAP value, reflecting its cleaner air and balanced species composition. The urban site, characterized by the highest levels of fine particulates and gaseous pollutants, showed the lowest IAP, indicating significant ecological stress. The rural site fell in between, with a moderate IAP value.

Particulate matter, particularly $PM_{2.5}$, showed a strong negative correlation with IAP, highlighting its role in inhibiting lichen growth. Smaller particles are more likely to penetrate lichen tissues, clog gas exchange structures, and impair photosynthesis (Nawaz et al. 2025), contributing to lower IAP scores and reduced lichen diversity in polluted areas. Bark pH also emerged as a significant factor, with more alkaline substrates, particularly in suburban areas, being associated with higher IAP values and richer lichen assemblages (McDonald et al. 2017). This pattern is consistent with the sensitivity of many species to bark acidification caused by SO_2 and other acidic pollutants (Chrabąszcz and Mróz 2017).

Microclimatic conditions also influenced lichen responses. IAP was positively associated with temperature, possibly due to enhanced metabolic activity and colonization in warmer sites (Colesie et al. 2018; Di Nuzzo et al. 2022). Conversely, a negative correlation was observed with relative humidity, which may reflect indirect effects such as increased growth of pollution-tolerant species or competitive exclusion under prolonged moist conditions (López et al. 2016).

Despite providing valuable insights, this study has certain limitations. Sampling was conducted only once per site and involved a limited number of trees, which may constrain the generalizability of the results. However, lichens are slow-growing organisms that integrate atmospheric conditions over time, making one-time sampling ecologically meaningful. The use of 2023 pollutant data is justified as it represents the most recent available conditions impacting lichen communities. Future studies should consider multi-seasonal sampling, increased spatial coverage and the inclusion of additional variables such as canopy structure and wind exposure to strengthen lichen-based air quality assessments.

Overall, these findings reinforce the value of epiphytic lichens as sensitive and integrative bioindicators of ecological air quality. Their community structure responds clearly to variations in air pollution, particularly particulate matter, while also reflecting influences from bark pH and microclimatic conditions. Metrics such as the IAP enhance this assessment by quantifying complex ecological responses across spatial gradients, especially in diverse land-use settings like those in Malaysia.

Conclusions

This study demonstrates that air pollution plays a critical role in shaping lichen diversity across different land-use types. The suburban site, which recorded the lowest pollutant concentrations, exhibited the highest IAP and species richness. In contrast, the urban site, characterized by elevated levels of PM, NO_2, SO_2, and CO, showed the lowest IAP and lichen diversity. A total of 16 species were identified, predominantly crustose forms, with tolerant species such as *D. picta* and *G. casiella* dominating urban environments, while sensitive species like *P. tinctorum* were found only in the suburban site. Strong negative correlations between IAP and particulate concentrations (PM_{10} and $PM_{2.5}$) show the detrimental impact of airborne particulates on lichen health. Additionally, environmental factors such as bark pH, temperature, and relative humidity also influenced species composition, reflecting the interplay between pollution stress and microhabitat conditions. These findings reinforce the value of integrating lichen-based indices with abiotic data to improve ecological air quality assessments. Importantly, this study offers baseline data on lichen diversity and pollution gradients in Peninsular Malaysia, contributing to regional bioindicator frameworks. Future research should focus on long-term monitoring and experimental approaches to investigate how specific pollutants affect lichen physiology and how these effects interact with local habitat factors such as substrate pH, canopy cover, and moisture availability.

Abbreviations

NO_2: Nitrogen dioxide
SO_2: Sulphur dioxide
O_3: Ozone
CO: Carbon monoxide
IAP: Index of atmospheric purity
DOE: Department of Environment
CAQM: Continuous Air Quality Monitoring
API: Air pollutant index

Acknowledgements

We would like to express our gratitude to the Ministry of Higher Education (MOHE) and Universiti Teknologi MARA (UiTM) for their support. We also extend our appreciation to the Malaysian Department of Environment (DOE) for providing the essential air quality data.

Authors' contributions

NSAH conducted data collection, performed data analysis and wrote the manuscript. AI conceived the research idea, supervised the study and reviewed the manuscript. MTL provided expertise on atmospheric pollutants and contributed to data interpretation. FP assisted with statistical and ecological analysis. RZ facilitated access to environmental data and provided regulatory insights. SK contributed to the discussion on environmental impact. FB provided input on chemical analysis and manuscript revision.

Funding

This research was funded by the Ministry of Higher Education (MOHE) under the Fundamental Research Grant Scheme (FRGS) (FRGS/1/2023/WAB02/UITM/02/3) and Universiti Teknologi

MARA (UiTM) (600-RMC/FRGS 5/3 (139/2023)).

Availability of data and materials
The air quality data used in this study was requested from the Malaysian Department of Environment (DOE) through https://btm.doe.gov.my/permohonandata/udara.

Ethics approval and consent to participate
Not applicable.

Consent for publication
Not applicable.

Competing interests
The authors declare that they have no competing interests.

References

Abas A, Sulaiman N, Adnan NR, Aziz SA, Nawang WNSW. Using lichen (Dirinaria sp.) as bio-indicator for airborne heavy metal at selected industrial areas in Malaysia. EnvironmentAsia. 2019;12(3):85-90. https://doi.org/10.14456/ea.2019.48.

Asmat A, Tarmizi SNM, Zakaria NH. Seasonal particulate matter (PM_{10}) concentration in Klang Valley, Malaysia. Int J Eng Technol. 2018; 7(3):162-7. https://doi.org/10.14419/ijet.v7i3.11.15953.

Barzeghar V, Hassanvand MS, Faridi S, Abbasi S, Gholampour A. Long-term trends in ambient air pollutants and the effect of meteorological parameters in Tabriz, Iran. Urban Clim. 2022;42:101119. https://doi.org/10.1016/j.uclim.2022.101119.

Belguidoum A, Haichour R, Lograda T, Ramdani M. Biomonitoring of air pollution by lichen diversity in the urban area of Setif, Algeria. Biodiversitas. 2022;23(2):970-81. https://doi.org/10.13057/biodiv/d230240.

Benítez Á, Medina J, Vásquez C, Loaiza T, Luzuriaga Y, Calva J. Lichens and bromeliads as bioindicators of heavy metal deposition in Ecuador. Diversity. 2019;11(2):28. https://doi.org/10.3390/d11020028.

Castells-Quintana D, Dienesch E, Krause M. Air pollution in an urban world: a global view on density, cities and emissions. Ecol Econ. 2021;189:107153. https://doi.org/10.1016/j.ecolecon.2021.107153.

Chrabąszcz M, Mróz L. Tree Bark, a valuable source of information on air quality. Pol J Environ Stud. 2017;26(2):453-66. https://doi.org/10.15244/pjoes/65908.

Colesie C, Büdel B, Hurry V, Green TGA. Can Antarctic lichens acclimatize to changes in temperature? Glob Chang Biol. 2018;24(3):1123-35. https://doi.org/10.1111/gcb.13984.

Conti ME, Cecchetti G. Biological monitoring: lichens as bioindicators of air pollution assessment--a review. Environ Pollut. 2001;114(3):471-92. https://doi.org/10.1016/s0269-7491(00)00224-4.

Das K, Goel N, Baweja P, Rani A, Uniyal PL. Lichens as bioindicators and biomonitoring agents. Environ We Int J Sci Tech. 2020;15:131-44.

Department of Environment Malaysia. 2010. Malaysia Environmental Quality Report 2010. https://www.doe.gov.my/wp-content/uploads/2021/10/Air-Qulaity.pdf. Accessed 21 Mar 2025.

Department of Environment Malaysia. 2023. Malaysia Environmental Quality Report 2023. https://storage.data.gov.my/environment/eqr_2023.pdf. Accessed 21 Mar 2025.

Department of Statistics Malaysia. 2023. Kawasanku dashboard. https://open.dosm.gov.my/dashboard/kawasanku. Accessed 21 Mar 2025.

Di Nuzzo L, Giordani P, Benesperi R, Brunialti G, Fačkovcová Z, Frati L, et al. Microclimatic alteration after logging affects the growth of the endangered lichen *Lobaria pulmonaria*. Plants (Basel). 2022;11(3):295. https://doi.org/10.3390/plants11030295.

Harikrishna S, Mukherjee S. Biomonitoring with lichens: evaluating heavy metal concentrations, environmental factors, and lichen diversity in Bengaluru, Karnataka, India. Environment and Ecology. 2024;42(1A):213-9. https://doi.org/10.60151/envec/ZLOV2341.

Ismail A, Marzuki SD, Mohd Yusof NB, Buyong F, Mohd Said MN, Sigh HR, et al. Epiphytic terrestrial algae (Trebouxia sp.) as a biomarker using the free-air-carbon dioxide-enrichment (FACE) system. Biology (Basel). 2017;6(1):19. https://doi.org/10.3390/biology6010019.

Jayalal U, Oh SO, Park JS, Sung JH, Kim SH, Hur JS. Evaluation of air quality using lichens in three different types of forest in Korea. Forest Sci Technol. 2016;12(1):1-8. https://doi.org/10.1080/21580103.2014.1003983.

Khairuddin NA, Muhammad N, Hashim NH, Yusof H, Jusoh S, Abas A, et al. Identification of most tolerant lichen species to vehicular traffic's pollutants at Batu Pahat area. AIP Conf Proc. 2017;1891(1):020078. https://doi.org/10.1063/1.5005411.

LeBlanc SCF, Sloover JD. Relation between industrialization and the distribution and growth of epiphytic lichens and mosses in Montreal. Can J Bot. 1970;48(8):1485-96. https://doi.org/10.1139/b70-224.

López LGC, Medina EAS, Peña AM. Effects of microclimate on species diversity and functional traits of corticolous lichens in the Popayan Botanical Garden (Cauca, Colombia). Cryptogam Mycol. 2016; 37(2):205-15. https://doi.org/10.7872/crym/v37.iss2.2016.205.

Lucheta F, Koch NM, Käffer MI, Riegel RP, de Azevedo Martins SM, Schmitt JL. Lichens as indicators of environmental quality in southern Brazil: an integrative approach based on community composition and functional parameters. Ecol Indic. 2019;107:105587. https://doi.org/10.1016/j.ecolind.2019.105587.

McDonald L, Van Woudenberg M, Dorin B, Adcock AM, McMullin RT, Cottenie K. The effects of bark quality on corticolous lichen community composition in urban parks of southern Ontario. Botany. 2017; 95(12):1141-9. https://doi.org/10.1139/cjb-2017-0113.

Nawaz S, Riaz B, Naseer F. Impact of air borne particulate matter on plants, climate, ecosystems, and human health: a comprehensive review. The Research of Medical Science Review. 2025;3(1):68-81.

Okon OG, Archibong BF, Antia UE, Rhouma A, Okon JE, Mbong EO, et al. Lichen diversity, substrate preference and environmental dynamics as indicators of air quality. Duts J Pure Appl Sci. 2024;10(2c):176-87. https://doi.org/10.4314/dujopas.v10i2c.17.

Raimundo-Costa W, Ferreira DC, Anhê ACBM, Senhuk APMdS. The use of *Parmotrema tinctorum* (Parmeliaceae) as a bioindicator of air pollution. Rodriguésia. 2024;72:e01872019. https://doi.org/10.1590/2175-7860202172090.

Shelyakin MA, Zakhozhiy IG, Dalke IV, Malyshev RV, Golovko TK. Effects of dust pollution on photosynthesis and respiration parameters of lichens in the bauxite mine area. Russ J Plant Physiol. 2024;71(4): 116. https://doi.org/10.1134/S1021443724605536.

Silprasit K, Rimlumduan T, Phowana N, Thummajitsakul S. Index of atmospheric purity (IAP) related to potential ecological risk indexes (RI) of heavy metals accumulation in urban area. Curr Appl Sci Technol. 2023;24(1):e0254863. https://doi.org/10.55003/cast.2023.254863.

Tanona M, Czarnota P. Index of atmospheric purity reflects the ecological conditions better than the environmental pollution in the Carpathian forests. J Mt Sci. 2020;17(11):2691-706. https://doi.org/10.1007/s11629-020-6266-1.

Thakur M, Bhardwaj S, Kumar V, Rodrigo-Comino J. Lichens as effective bioindicators for monitoring environmental changes: a comprehensive review. Total Environment Advances. 2024;9:200085. https://doi.org/10.1016/j.teadva.2023.200085.

Tumur A, Mamut R, Seaward MRD. Diversity and elevational levels of lichens in Western Tianshan National Nature Reserve in Xinjiang, China. Diversity. 2025;17(2):102. https://doi.org/10.3390/d17020102.

Vitali M, Antonucci A, Owczarek M, Guidotti M, Astolfi ML, Manigrasso M, et al. Air quality assessment in different environmental scenarios by the determination of typical heavy metals and Persistent Organic Pollutants in native lichen *Xanthoria parietina*. Environ Pollut. 2019;254:113013. https://doi.org/10.1016/j.envpol.2019.113013.

Yatawara M, Dayananda N. Use of corticolous lichens for the assessment of ambient air quality along rural-urban ecosystems of tropics: a study in Sri Lanka. Environ Monit Assess. 2019;191(3):179. https://doi.org/10.1007/s10661-019-7334-2.