

DOCTORAL (PHD) DISSERTATION

**PHYTOTOXICITY OF ATMOSPHERIC PARTICULATE MATTER**

**Selenge Tumurbaatar**

Doctoral School of Chemistry and Environmental Sciences

DOI:10.18136/PE.2026.975

*Supervisor:*

**Dr Nóra Julianna Kovács**

**Dr Katalin Eszter Hubai**

**University of Pannonia**  
Department



**Veszprém**

**2025**

PHYTOTOXICITY OF ATMOSPHERIC PARTICULATE MATTER

The thesis was prepared for the award of a doctoral degree (PhD) within the framework of the  
Doctoral School of Chemistry and Environmental Sciences at University of Pannonia

in the discipline of Environmental Sciences

written by: Selenge Tumurbaatar

Supervisors: Dr Nóra Julianna Kovács, Dr Katalin Eszter Hubai

I recommend the dissertation for acceptance: yes / no .....  
supervisor

I recommend the dissertation for acceptance: yes / no .....  
supervisor

I recommend the dissertation for the peer review.  
.....  
Chair of the DDHC

The PhD-candidate has achieved .....% at the public debate.

The composition of the Final Examination Committee:

Chair: .....

Reviewers: .....

Members: .....

Veszprém,

.....  
chair of the committee

Qualification of degree: .....

Veszprém,

.....  
Chair of the UDHC

## Content

List of Figures .....	1
List of Tables .....	2
Abbreviations .....	3
Absztrakt .....	5
Abstract .....	7
Хураангуй .....	9
1. Main objectives .....	11
2. Introduction .....	12
2. 1. Atmospheric particulate matter .....	12
2. 2. Traffic-related air pollution .....	13
2. 3. Illegal waste burning .....	14
2. 4. Exposure to polycyclic aromatic hydrocarbons .....	15
2. 5. Heavy metals in the atmosphere .....	19
2. 6. Phytotoxicity of atmospheric pollutants .....	19
2. 7. Mechanisms of phytotoxicity .....	21
2. 7. 1. Oxidative stress in plants .....	21
2. 7. 2. Photosynthetic pigments and air pollution: A serious concern .....	23
2. 7. 3. Phytotoxicity testing .....	24
2. 7. 4. The OECD vegetative vigour test .....	25
2. 7. 5. Bioaccumulation of polycyclic aromatic hydrocarbons in plants .....	27
3. Assessing Risk of Emissions Generated During Illegal Waste Burning: Phytotoxicity and Bioaccumulation .....	30
3. 1. Materials and methods .....	30
3. 1. 1. Sampling and sample preparation .....	30
3. 1. 2. Plant material used .....	30
3. 1. 3. Cultivation and treatment of test plants .....	32
3. 1. 4. Storage, preservation, preparation for analyses .....	33
3. 1. 5. Chemical analyses of plant samples .....	33
3. 1. 6. Measurement of biomass .....	34
3. 1. 7. Photosynthetic pigment measurement .....	34
3. 1. 8. Peroxidase measurement .....	35
3. 1. 9. Statistical approach .....	35
3. 2. Results and discussion .....	36
3. 2. 1. Composition of the PM extract .....	36
3. 2. 2. Phytotoxicity assessment .....	39
3. 2. 3. Bioaccumulation and bioconcentration of PAHs .....	43

3. 2. 4. Limitations of the study .....	46
<b>4. Phytotoxicity testing of emissions from illegal waste burning based on the OECD Terrestrial Plant Test.....</b>	<b>47</b>
4. 1. Materials and methods.....	47
4. 1. 1. Sampling and sample preparation .....	47
4. 1. 2. Plant material used.....	47
4. 1. 3. Experimental procedures.....	49
4. 2. Results and discussion.....	49
4. 2. 1. Composition of the PM extract .....	49
4. 2. 2. Phytotoxicity .....	52
4. 2. 3. Individual sensitivity .....	53
4. 2. 4. Sensitivity of different end-points .....	55
<b>5. Can landraces better cope with environmental stress?.....</b>	<b>61</b>
5. 1. Materials and methods.....	61
5. 1. 1 Sampling and sample preparation .....	61
5. 1. 2. Plant material used.....	61
5. 1. 3. Experimental procedures.....	62
5. 2. Results and discussion.....	63
5. 2. 1. Composition of the PM extract .....	63
5. 2. 2. Phytotoxicity Assessment .....	64
5. 2. 3. Discussion.....	69
5. 2. 3. 1. Comparison of end-points.....	70
5. 2. 3. 2. Comparison of varieties used .....	73
<b>6. Contribution of atmospheric fallout to the soil-root-leaf transfer of PAHs in higher plants</b>	<b>75</b>
6. 1. Materials and Methods .....	75
6. 1. 1. Sampling and sample preparation .....	75
6. 1. 2. Plant material used.....	75
6. 1. 3 Experimental procedures.....	75
6. 2. Results and Discussion .....	76
6. 2. 1. Composition of the PM extract .....	76
6. 2. 2. Discussion.....	78
<b>7. Conclusions .....</b>	<b>83</b>
<b>8. New scientific points.....</b>	<b>84</b>
<b>Acknowledgments.....</b>	<b>85</b>
<b>9. References .....</b>	<b>86</b>
<b>List of publications .....</b>	<b>112</b>

# List of Figures

<b>Figure 1.</b> The major sources of PAHs ((Di Vaio et al. 2016, Dat and Chang 2017, and Yang et al. 2021) .....	16
<b>Figure 2.</b> US EPA priority pollutant PAH compounds.....	17
<b>Figure 3.</b> Schematic diagram summarising potential exposure routes for LMW and HMW PAHs, including both foliar and soil uptake. ....	18
<b>Figure 4.</b> Steps of the phytotoxicity testing procedure .....	33
<b>Figure 5.</b> Reduction in biomass of treated lettuce plants compared to the control. ....	40
<b>Figure 6.</b> Changes in peroxidase activity in (a) lettuce and (b) mustard .....	42
<b>Figure 7.</b> Accumulation pattern of different molecular weight PAH groups in the test plants. ....	45
<b>Figure 8.</b> Average <b>a.)</b> biomass in comparison <b>b.)</b> Average length of leaves to the control <i>A. rosea</i> L.,	55
<b>Figure 9.</b> Average <b>a.)</b> biomass in comparison and <b>b.)</b> length of leaves in to the control <i>M. jalapa</i> comparison to the control.....	55
<b>Figure 10.</b> Average biomass in comparison to the control <i>G. aristata</i> .....	56
<b>Figure 11.</b> Average biomass in comparison to the control <i>C. Pepo</i> L., .....	56
<b>Figure 12.</b> Average length of leaves in comparison to the control <i>I. purpurea</i> .....	56
<b>Figure 13.</b> Typical necrotic spots on <i>A. rosea</i> L., .....	57
<b>Figure 14.</b> Typical necrotic spots on <i>I. purpurea</i> .....	57
<b>Figure 15.</b> <i>G. aristata</i> POD level.....	58
<b>Figure 16.</b> <i>C. officinalis</i> POD level .....	58
<b>Figure 17.</b> <i>I. purpurea</i> POD level .....	58
<b>Figure 18.</b> Chlorophyll-b ratio in <i>M. jalapa</i> .....	59
<b>Figure 19.</b> Carotene content in <i>M. jalapa</i> .....	59
<b>Figure 20.</b> Consecutive steps of sampling and sample preparation. ....	61
<b>Figure 21.</b> Biomass changes in the ‘Roma’ variety relative to the control .....	64
<b>Figure 22.</b> Biomass changes in the ‘Mobil’ variety relative to the control.....	65
<b>Figure 23.</b> Biomass changes in the ‘Lugas’ variety relative to the control.....	65
<b>Figure 24.</b> Changes in carotene content in the ‘Roma’ variety relative to the control.....	66
<b>Figure 25a.</b> POD activity (nmol x (min x mg protein) in the ‘Roma’ variety .....	67
<b>Figure 25b.</b> POD activity (nmol x (min x mg protein) in the ‘Mobil’ variety.....	67
<b>Figure 25c.</b> POD activity (nmol x (min x mg protein)) in ‘Tápiószelei’ .....	68
<b>Figure 25d.</b> POD activity (nmol x (min x mg protein)) in ‘Gulácsi’ .....	68
<b>Figure 26.</b> Concentrations of PAHs with varying molecular weights in soil and treated plant compartments, illustrating the soil-root-aerial parts transfer pathway .....	78
<b>Figure 27.</b> Concentrations of various molecular weight PAHs in soil and plant compartments, illustrating the soil-to-root-to-aerial parts pathway .....	78
<b>Figure 28.</b> Accumulation of total PAHs across various compartments.....	79

## List of Tables

<b>Table 1.</b> Concentration of PAHs in the waste PM10 extract.....	36
<b>Table 2.</b> Heavy metal concentrations in the PM <sub>10</sub> extract .....	38
<b>Table 3.</b> Summary of phytotoxicity test results. + indicates a significant effect; 0: effect recorded was non-significant.....	39
<b>Table 4.</b> Concentration of PAHs in the test plants and calculated BCF values.....	43
<b>Table 5.</b> Concentration of PAHs in the PM10 extract.....	50
<b>Table 6.</b> Concentration of heavy metals in the PM10 extract .....	51
<b>Table 7.</b> Summary of recorded effects for each endpoint and species tested: '+' indicates a significant effect, while '0' denotes a non-significant effect.....	52
<b>Table 8.</b> Morphology of 5 different tomato varieties/landraces.....	61
<b>Table 9.</b> Description of test groups. WW: Well-watered; RW: Reduced watering. PM-: no treatment; PM+: treatment with the PM extract .....	62
<b>Table 10.</b> Concentration of PAHs in the aerosol extract .....	63
<b>Table 11.</b> Response of the tested varieties .....	72
<b>Table 12.</b> Concentrations of PAHs in the aerosol extract (expressed in $\mu\text{g L}^{-1}$ ) and in the treated samples (expressed in $\mu\text{g kg}^{-1}$ ). PAHs prioritized by the US EPA are indicated in bold. S: soil treatment; A: aerial parts treatment. ....	77

# Abbreviations

(B(a)P - benzo(a)pyrene

BCF – bioconcentration factor

CAT - catalase

CO - carbon monoxide

CLRTAP - long-range trans-boundary air pollution

DTT - dithiothreitol

GPX - glutathione peroxidase

HMW - high molecular weight

LEZ - low emission zone

LMW – low molecular weight

LOQ - limit of quantification

LOD - limit of detection

NO<sub>x</sub> - nitrogen oxides

PAHs – polycyclic aromatic hydrocarbons

PE - polyethylene

PET - polyethylene terephthalate

PM – particulate matter

PM<sub>2.5</sub> – particulate matter 2.5 micrometers or less in diameter

PM<sub>10</sub> - particulate matter 10 micrometers or less in diameter

POD - peroxidase

POPs - persistent organic pollutants

PP - polypropylene

PSN – photosynthetic

PS - polystyrene

PUR - polyurethane

PVC - polyvinyl chloride

ROS - reactive oxygen species

SOD - superoxide dismutase

SO<sub>2</sub> - sulfur dioxide

TMB - tetramethylbenzidine

UFP – ultrafine particle

VEC - vehicle exhaust catalyst

VOC - volatile organic compound

## Absztrakt

A növények, mint helyhez kötött élőlények, folyamatosan kitettek a légköri szennyező anyagok hatásainak, amelyek lehetnek gázhalmazállapotúak, illetve részecskékhez kötöttek. Két jelentős antropogén eredetű forrás emelendő ki, melyek a vegetációs időszakban hatással lehetnek a növényekre: a közúti közlekedés, valamint a háztartási hulladék nyílt téri égetése. A közlekedés elsősorban a policiklusos aromás szénhidrogének (PAH-ok) kibocsátásáért felelős, amelyek toxikus tulajdonságaik révén környezeti és egészségügyi kockázatot is jelentenek. A hulladékégetés során – a felhasznált anyagok összetételétől függően – mind PAH-ok, mind nehézfémek kerülhetnek a légkörbe.

Kutatásom célja az volt, hogy ellenőrzött laboratóriumi körülmények között, az OECD 227. számú vizsgálati irányelve (Terrestrial Plant Test: Vegetative Vigour Test) alkalmazásával tanulmányozzam a közlekedési eredetű részecskeszennyezés, valamint a háztartási hulladék nyílt téren történő égetése során keletkező levegőszennyező komponensek fitotoxikus hatásait.

A vizsgálatok az alábbi négy fő témakört érintettem:

- I. Öt különböző paradicsomfajta/helyi fajta, mint például a „Roma”, a „Mobil” és a „Lugas” (kereskedelmi forgalomban kapható fajták), valamint a „Gulácsi” és a „Tápiószelei” (regionális helyi fajták) érzékenységet hasonlították össze a légszennyezés és a szárazság okozta stressz együttes hatásával a paradicsomra (*Lycopersicon esculentum* Mill.). A vízstressz hatása kifejezettebb volt, míg a PM-kivonattal történő kezelés csak a három kereskedelmi forgalomban kapható fajtánál váltott ki reakciót. Általánosságban megállapítható, hogy a helyi fajták nagyobb ellenállást mutattak az abiotikus stresszrel szemben, a kereskedelmi forgalomban kapható „Roma” fajta mutatta a legnagyobb, míg a regionális „Gulácsi” fajta a legkisebb érzékenységet az egyes és a kombinált kezelésekre.
- II. Megállapítottam a fajspecifikus felhalmozódási potenciált a saláta (*Lactuca sativa* L.) és a fehér mustár (*Sinapis alba* L.) növények összehasonlításával, amelynek eredményeként a saláta lényegesen magasabb bioakkumulációt mutatott az alacsony molekulatömegű (LMW) PAH-ok tekintetében az ehető részeiben.
- III. A légköri policiklusos aromás szénhidrogének (PAH-ok) nedves lerakódásának szimulálásával a rukola (*Eruca sativa* Mill., Brassicaceae család) esetében a PAH-ok légköri felvétele bizonyult a fő útvonalnak, különösen a nagyobb molekulatömegű PAH-ok esetében. Az eredmények azt is kimutatták, hogy mind a

talaj-légtér részek, mind a légtér részek-gyökér átvitelek alacsonyak, különösen a HMW PAH-ok esetében.

- IV. A kommunális hulladékok kísérleti égetése során keletkező kibocsátásokra adott fitotoxikus reakciókat két általánosan alkalmazott tesztnövény (*Lactuca sativa* és *Sinapis alba*) és hat dísnövény (Gaillardia aristata, Calendula officinalis, *Alcea rosea annua*, *Ipomoea purpurea*, *Mirabilis jalapa* és *Cucurbita pepo*) összehasonlításával értékelték. Erős fajspecifikus érzékenységet tapasztaltak, a *L. sativa* és a *M. jalapa* reagált a legerőteljesebben, ami összhangban áll a közzétett tanulmányokkal.
- V. A dízelkibocsátás és a hulladékégetés során keletkező PM-minták esetében a növekedés gátlása és a peroxidáz enzimaktivitás bizonyult a legérzékenyebb végpontnak, ami összhangban áll a közzétett tanulmányokkal.

Összefoglalásként megállapítható, hogy az alkalmazott tesztprotokoll hatékony eszköznek bizonyult a légköri eredetű PAH-ok és nehézfémek nedves ülepedésének, valamint ezek fitotoxikus hatásainak vizsgálatára. A kísérletek értékes információkkal szolgáltak a közlekedési eredetű kibocsátások, illetve a háztartási hulladék égetéséből származó légszennyezés növényélettani hatásairól.

# Abstract

Plants, being sessile organisms, are continuously exposed to atmospheric contaminants, both gaseous and particle-bound.

During the vegetation period, two main anthropogenic sources should be taken into consideration, vehicular traffic and open burning of domestic waste. Traffic is mainly responsible for the emission of polycyclic aromatic hydrocarbons (PAHs), a group of chemicals known to possess toxic properties. Depending on the composition of the waste burned, emissions of both heavy metals and PAHs should be anticipated.

The aim of my research was to investigate the phytotoxic responses in case of traffic related particulate matter (PM) and open waste burning generated air pollutant components under controlled (laboratory) conditions, applying a standard protocol, namely the No. 227 OECD GUIDELINE FOR THE TESTING OF CHEMICALS: Terrestrial Plant Test: Vegetative Vigour Test. The following examinations were performed:

- I. Sensitivity of five different tomato varieties/landraces such as 'Roma', 'Mobil' and 'Lugas' (commercially available varieties) as well as 'Gulácsi' and 'Tápiószelei' (regional landraces) was compared to the combined effect of air pollution and drought stress on tomato (*Lycopersicon esculentum* Mill.). Water stress had more pronounced effect while treatment with the PM extract triggered response only in case of the three commercial varieties. In general, higher resistance of landraces toward abiotic stress was established, the commercial variety 'Roma' showing the highest while the regional landrace 'Gulácsi' showing the lowest sensitivity to individual and combined treatments.
- II. I established species-specific accumulation potential comparing lettuce (*Lactuca sativa* L.) and white mustard (*Sinapis alba* L.) plants, lettuce showing substantially higher bioaccumulation of low molecular weight (LMW) PAHs in edible parts.
- III. Simulating wet deposition of atmospheric polycyclic aromatic hydrocarbons (PAHs) in rocket (*Eruca sativa* Mill., family Brassicaceae), atmospheric uptake of PAHs proved the main pathway, especially for higher molecular weight PAHs. Results also demonstrated that both soil-aerial parts and aerial parts-root transfers are low, especially in case of HMW PAHs.

- IV. Phytotoxic responses to emissions generated during experimental burning of municipal waste were evaluated comparing two commonly applied test plants (*Lactuca sativa* and *Sinapis alba*) and six ornamental plants (*Gaillardia aristata*, *Calendula officinalis*, *Alcea rosea annua*, *Ipomoea purpurea*, *Mirabilis jalapa* and *Cucurbita pepo*). Strong species-specific sensitivity was experienced, *L. sativa* and *M. jalapa* being the most responsive, in concordance with reported studies.
- V. In case of diesel emission and waste burning generated PM samples, growth inhibition and peroxidase enzyme activity proved the most responsive end-points, in concordance with reported studies.

Based on these results, it can be concluded that the protocol proved an important tool to assess wet deposition of atmospheric PAHs and heavy metals. Important information could be collected about the potential phytotoxicity of both traffic-related emissions and PM generated by open burning of domestic waste.

## Хураангуй

Ургамлууд нь амьд организмын хувьд хийн болон тоосонцор агаарын бохирдолд үргэлж өртөж байдаг.

Ургамлын ургалтынэ хугацаанд хоёр үндсэн хүний үйл ажиллагаанаас үүдэлтэй эх үүсвэрийг анхаарах шаардлагатайв Үүнд: автотээврийн хөдөлгөөн болон ахуйн хог хаягдлыг ил задгай шатаах явдал багтана. Тээврийн хэрэгсэл нь хорт шинж чанартай гэж мэдэгдсэн химийн нэгдлүүдийн бүлэг болох полицикл ароматик нүүрстөрөгч (ПАХс)-ийн ялгарлын гол эх үүсвэр болдог. Шатааж буй хог хаягдлын найрлагаас шалтгаалан хүнд металл болон ПАХс-ийн ялгарлыг тооцоолох шаардлагатай. Миний судалгааны зорилго нь автотээврийн хөдөлгөөнөөс үүсэх тоосонцор (ПМ) болон хог хаягдлыг ил задгай шатаах үеп үүсэх агаарын бохирдуулагч бүрдлүүдийн ургамалд үзүүлэх хорт нөлөөллийфг стандарт протоколын дагуу, хяналттай (лабораторийн) нөхцөлд судлах явдал байв. Үүнд: ОИСД-ийн Химийн бодисын турших 227 дугаар удирдамж болох “Газрын ургамлын туршилт: Ургамлын өсөлтийн чадварын туршилт”-ыг ашигласанв Дараах шинжилгээнүүдийг хийсэн:

гаарын бохирдол болон ган гачиг гэсэн хосолсон хүрээлэн буй орчны стрессийн нөлөөнд улаан лооль (*Lycopersicon esculentum*)-ын хариу үйлдлийг үнэлэхдээ 3 төрлийн худалдааны сорт болон бүс нутгийн 2 уламжлалт сортыг харьцуулан судалсан. Үүнд: ‘Gulácsi’ нэртэй уламжлалт сорт багтсан бөгөөд энэ нь хамгийн тэсвэртэй нь болох нь тогтоогдсон. Энэ нь уур амьсгалын өөрчлөлтийн нөлөөнд зохицсон, нарийн сонгосон таримал ургамлын сортуудыг ашиглахын чухлыг харуулж байна.

уршилтанд сонгосон загвар хог хаягдлыг шатаах замаар гарган авсан тоосонцор (ПМ) дээжийн ургамалд хот нөлөөллийг өргөн хэрэглэгддэг хоёр туршилтын ургамал болох салат навч (*Lactuca sativa*) болон цагаан гич (*Sinapis alba*)-д үнэлэв. Мөн ижил төстэй дээжийг Унгарын цэцэрлэгжүүлэлтэд өгрөн хэрэглэгддэг зургаан төрлийн гоёлын ургамалд туршсан. Хоёр туршилтын үр дүнгээс харахад ялгарсан бохирдуулагч нь ургамалд хортой шинж чанартай болох нь батлагдсан боловч ургамлын зүйлээс хамаарах мэдрэмтгийн байдал ихээхэн ялгаатай байв.

агаан гич (*Sinapis alba* L.) ургамал дээр дизель түлшний утааны ханд ашиглан туршилт хийсэн. Туршилтын явцад полицикл ароматик нүүрстөрөгч (ПАХ)-ийн агууламжийг ургамлын янз бүрийн хэсгүүдэд (иш, навч, үрэнд) хэмжиж,

биоконцентрацийн коэффициентууд (BCF) мөн тооцоолсон. Үрийн түвшинд хүртэл мэдэгдэхүйц хуримтлал ажиглагдсан нь авто замын ойролцоо таримал ургамал тариалах нь эрсдэл дагуулж болзошгүйг хариуулж байна. Ерөнхийдөө бүх ургамлын эдүүд өндөр молекулт ПАХ нэгдлүүдийг хуримтлуулах өндөр хандлагатай байв.

жил төрлийн хандыг руколо (*Eruca sativa* Mill.) ургамалд ашиглаж, ПАХ нэгдлүүдийн дамжих замыг дуурайлган судалсан. Үүнд, агаар-газрын дээрх хэсэг-үндэс гэсэн замаар нэг хэсэгт, хөрс-үндэс-газрын дээрх хэсэг гэсэн замаар нөгөө хэсэгт туршилт хийв. Агаар-газрын дээрх хэсэг-үндэс замын хувьд ПАХ нэгдлүүд газрын дээрх хэсгүүдэд мэдэгдэхүйц хэмжээгээр хуримтлагдсан ч үндэс рүү дамжих нь харьцангуй бага байв. Харин хөрс-үндэс-газрын дээрх хэсэг гэсэн замын хувьд өндөр молекулт ПАХ нэгдлүүд ихэвчлэн үндэст хуримтлагдаж байгааг үр дүн харуулсан. Ерөнхийдөө руколо ургамал нь өндөр молекулт ПАХ нэгдлүүдийг мэдэгдэхүйц хэмжээгээр хуримтлуулах чадвартай болох нь тогтоогдсон.

Эдгээр үр дүнд үндэслэн, агаарын бохирдлын ПАХ болон хүнд металлуудын нойтон тунадасны нөлөөг үнэлэхэд ашигласан протокол нь чухал хэрэгсэл болох нь батлагдлаа. Аьтотээврийн хөдөлгөөнтэй холбоотой ялгарлууд болон ахуйн хог хаягдлыг ил кадгай шатаах замаар үүссэн тоосонцор (ПМ)-ын ургамалд үзүүлэх боломжит хорт нөлөөний талаар чухал мэдээлэл цуглуулах боломжтой байв.

# 1. Main objectives

Plants, being sessile organisms, have to endure the chemical stress posed by air pollutants. Considering potential sources, winter heating will probably affect only coniferous species, while traffic-related emissions and open burning of waste will expose herbaceous and deciduous species both in natural and agricultural ecosystems.

During my research the main goal was to investigate the potential phytotoxic effects of these emissions, based on a standard protocol, the No. 227 OECD GUIDELINE FOR THE TESTING OF CHEMICALS: Terrestrial Plant Test: Vegetative Vigour Test. As the Guideline follows a step-by-step and clearly defined test protocol, individual assays are repeatable and reproducible, fulfilling basic quality assurance criteria. Repeatability and reproducibility make it possible to compare the sensitivities of different plant species, which will in turn provide basic data for extrapolating these results to wider plant communities.

Both traffic-related and waste burning emissions contain significant amount of polycyclic aromatic hydrocarbons. As such, phytotoxicity test results were intended to compare with analytical measurements, to determine if the distribution pattern of PAHs might support potential toxicity of the samples.

*Specific objectives were as follows:*

- i. To evaluate the combined effect of air pollution and drought stress comparing the sensitivity of different tomato varieties/ landraces using diesel emission samples, in order to assess the possible impact of road traffic and drought on agricultural ecosystems
- ii. To follow under laboratory conditions the transfer of PAHs via wet deposition assessing both the air-serial parts and the soil-root-aerial parts pathways, using rocket (*Eruca sativa* Mill.) as model species.
- iii. To determine in a chronic study if diesel emissions specifically occurring in roadside environments can deteriorate crop quality via accumulation of polycyclic aromatic hydrocarbons.
- iv. To evaluate the potential phytotoxic effects of waste burning comparing two widely accepted test species, *Eruca sativa* Mill. and *Lactuca sativa* L. (lettuce).
- v. To evaluate the potential phytotoxic effects of waste burning comparing selected ornamental plants commonly used in European (Hungarian) horticulture.

## 2. Introduction

### 2. 1. Atmospheric particulate matter

Particulate matter refers to solid and liquid particles suspended in the ambient air. Particulate matter is a mixture of particles emitted directly into the air (primary particulate matter) and particles formed in the air from chemicals that convert gaseous pollutants (secondary particulate matter) (Popoola et al. 2018). These particles are generated from both natural and anthropogenic sources into the environment (Dat and Chang 2017). Natural sources of PM include wildfires, volcanic eruptions, and dust storms, while anthropogenic sources predominantly consist of vehicular emissions, industrial discharges, and combustion of fossil fuels. The composition of particulate matter can vary widely depending on the source; for instance, combustion-related particulates often contain organic compounds and heavy metals.

The United States Environmental Protection Agency (US EPA) classified atmospheric PM according to their aerodynamic diameter sizes, namely fine  $PM_{2.5}$  (0-2.5  $\mu m$ ), and coarse  $PM_{10}$  (2.5-10  $\mu m$ ). In the atmosphere, fine PM dominates in the surface area, while coarse PM dominates through their weight. Both fine and coarse PM respond to changes in atmospheric mechanisms such as humidity, precipitation, and wind, and their deposition characteristics are changing (Grantz et al. 2003). The European Union has determined threshold values for 10  $\mu m$  diameter ( $PM_{10}$ ) and for the 2.5  $\mu m$  ( $PM_{2.5}$ ) particles (European Parliament 2008). In Hungary, the estimated annual emission of  $PM_{10}$  is 47 000 t while that of  $PM_{2.5}$  is 33 000 t (data from 2023, [https://www.ksh.hu/stadat\\_files/kor/hu/kor0017.html](https://www.ksh.hu/stadat_files/kor/hu/kor0017.html)).

The significance of atmospheric particulate matter cannot be overstated. PM has profound effects on humans, other living organisms, and the natural environment (Zhang et al. 2018). Efforts to monitor and control atmospheric particulate matter have been implemented globally, with regulations aimed at reducing PM emissions from transportation and industry. Various measures, including improved fuel standards, the promotion of cleaner technologies, and stricter emissions controls, have shown to be effective in reducing ambient PM levels. For instance, European Union (EU) Directives legalized the values guidelines for air pollutants and then the Convention on Long-Range trans-boundary air pollution (CLRTAP) and protocols correlated to emission reductions. Therefore, most common effective approaches are prohibition of solid fuel use in homes, low emission zones (LEZs), and the introduction of vehicle exhaust catalysts (VECs) (Jonidviai Jafari et al. 2021).

Considering the impact on vegetation, particles might cause direct physical damage, by abrasion or by blocking the stomata of plants. However, chemical risk is more obvious as these particles might carry potentially toxic compounds, polycyclic aromatic hydrocarbons and heavy metals are the most widely discussed (Kelly & Fussel 2012, Kim et al. 2013). With the decrease in size, the hazard grows as smaller particles will have larger relative surface area; as such, they can bind relatively higher amounts of potentially toxic pollutants (Samek et al. 2017).

The majority of European herbaceous plants (both native and crop species) are exposed to PM-posed risk during the vegetation period, which only partially overlaps with the heating season, the major contributor of air polluting agents. During the vegetation period, however, two main anthropogenic activities have to be taken into consideration: traffic and open burning of domestic waste.

## **2. 2. Traffic-related air pollution**

Diesel engines are a prominent source of air pollution, contributing significantly to the deterioration of air quality in urban and rural environments. The combustion of diesel fuel generates a range of harmful pollutants, including nitrogen oxides (NO<sub>x</sub>), particulate matter (PM), carbon monoxide (CO), and volatile organic compounds (VOCs). These substances pose serious risks to public health and the environment (Fayyazbakhsh et al. 2022). Exposure to diesel emissions has been linked to numerous human health issues. The effects of diesel emission pollution extend beyond human health. Nitrogen oxides, another by product of diesel combustion, contribute to the formation of ground-level ozone. The release of nitrogen oxides plays a significant role in the formation of smog, which can harm ecosystems, reduce crop yields, and impair visibility. Additionally, particulate matter negatively affects soil and water quality, leading to further environmental degradation (Montano et al. 2025). which can impair lung function and aggravate asthma and other respiratory diseases. Moreover, the carcinogenic components of diesel exhaust, such as benzene and formaldehyde, increase the risk of developing cancer (Bai et al. 2015).

In response to the growing concern over diesel emissions, governments and organizations around the world have implemented various regulations aimed at reducing emissions from diesel vehicles and machinery. These measures include stricter emission standards, incentives for cleaner technologies, and the promotion of alternative fuels, such as electric or hydrogen-powered vehicles. To further mitigate the impact of diesel emissions,

communities are also encouraged to invest in improved public transportation systems, implement urban planning strategies that reduce traffic congestion, and promote active transportation options like cycling and walking (Jonidi Jafari et al. 2021).

Due to technological improvements and more stringent regulations, PM emission of diesel-powered vehicles has shown a diminishing tendency in Europe, but it still poses a considerable risk (Matthias et al. 2020). Diesel emissions are major contributors of PAHs (Caumo et al. 2022). The International Agency for Research on Cancer (IARC) classified diesel engine exhaust as carcinogenic to humans by the (IARC 2014).

Diesel exhaust emissions might diminish; however, road transport will remain an important contributor of air pollution, via the so-called non-exhaust emissions such as road dust resuspension, tyre wear and road abrasion, and brake wear. They are the major sources of potentially toxic heavy metals such as Cr, Pb, Zn or Cu, all of well-known toxic potentials (Rienda et al. 2023).

### **2. 3. Illegal waste burning**

Illegal waste burning is an alarming practice that has serious implications for both public health and the environment. This activity involves the combustion of waste materials in unauthorized locations, which can release toxic pollutants into the air, soil, and water systems. As populations grow and waste management systems become overburdened, illegal burning has become more prevalent, posing a significant threat to communities across the globe. The pollutants emitted from illegal waste burning can lead to a variety of health issues, including respiratory diseases, cardiovascular problems, and even cancer. Harmful substances like dioxins, furans, and particulate matter are released during combustion, affecting both those who engage in the burning and nearby residents. Vulnerable populations, such as children and the elderly, are particularly at risk (Triassi et al. 2015).

In addition to health effects, illegal waste burning contributes to environmental degradation. The practice releases greenhouse gases, contributing to climate change and impacting air quality. Furthermore, it can contaminate local soil and waterways, harming ecosystems and wildlife (Vaverková et al. 2019).

Governments and organizations are responding to this issue with increased regulation and public awareness campaigns aimed at educating communities about the dangers of illegal

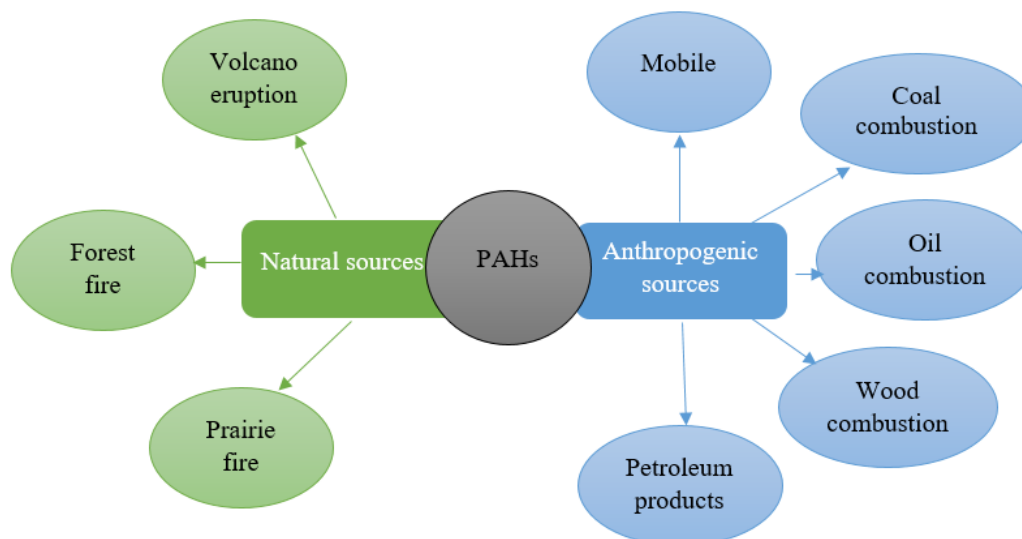
waste burning. Proper waste management practices, such as recycling and safe disposal methods, are essential to mitigate this problem. To combat illegal waste burning effectively, it requires a collaborative approach involving local authorities, environmental groups, and the public. Reporting suspicious burning activities and advocating for better waste management policies can make a significant difference in addressing this critical issue. Community engagement is crucial in finding solutions to this growing problem.

In Hungary, open burning of domestic waste is prohibited while open burning of garden waste is considered legal in pre-defined circumstances. However, open burning of domestic waste still occurs, frequently together with garden (green) waste. It generates significant emission of particulate matter (PM), especially PM<sub>2.5</sub> and finer fractions (Krecl et al. 2021). Depending on the type of waste burned, emissions contain potentially toxic heavy metals and polycyclic aromatic hydrocarbons (PAHs) (Schmidl et al. 2008).

## **2. 4. Exposure to polycyclic aromatic hydrocarbons**

Polycyclic aromatic hydrocarbons (PAHs) are organic chemical pollutants that consist of at least two fused rings. The particulate matter absorbs PAHs from the atmosphere, which are easily transported into the atmosphere through turbulence, participating in long-term air pollution. So, PAHs are becoming a significant environmental and human health concern (Wu et al. 2021).

In the atmosphere, PAHs can originate from both natural and anthropogenic sources. (Figure 1.) Natural sources include volcanic eruptions and forest fires, while human activities encompass vehicle emissions, industrial processes, and the burning of coal and oil (Di Vaio et al. 2016, Dat and Chang 2017, Yang et al. 2021).



**Figure 1.** The major sources of PAHs ((Di Vaio et al. 2016, Dat and Chang 2017, and Yang et al. 2021)

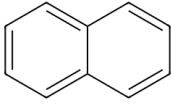
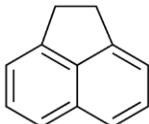
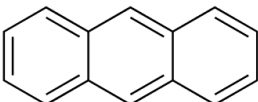
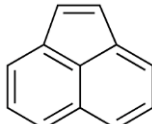
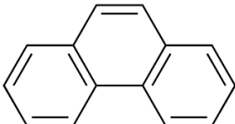
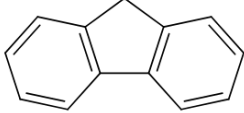
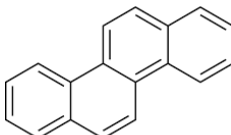
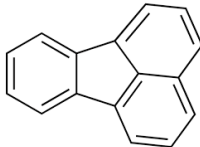
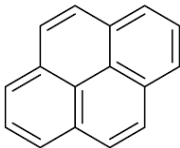
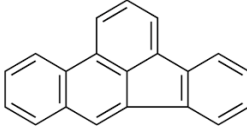
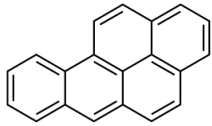
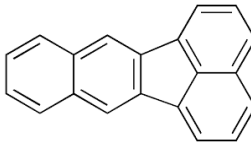
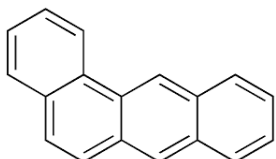
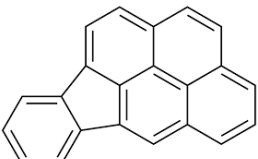
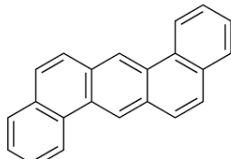
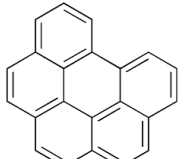
Once released into the atmosphere, PAHs can undergo various transformation processes. They can be transported over considerable distances by wind currents, leading to accumulation far from their original sources. Their persistence in the environment is also noteworthy, as some PAH compounds can remain in the atmosphere for extended periods before being deposited back to the surface via precipitation or sedimentation (He et al. 2014)

Monitoring and regulating PAH levels in the atmosphere is crucial for safeguarding public health and ensuring environmental protection. Effective strategies to mitigate PAH emissions include the implementation of cleaner combustion technologies, stricter regulations on industrial emissions, and the promotion of alternative energy sources. Ongoing research is essential to enhance our understanding of PAH behavior in the atmosphere, assess their health risks, and develop innovative methods for pollution control (Li et al. 2024).

As awareness of the dangers posed by PAHs continues to grow, it is imperative that governments, industries, and communities work together to reduce their impact on the environment and public health. (Abdel-Shafy & Mansour 2016).

As PAHs have reportedly toxic properties, the US Environmental Protection Agency (EPA) identified 16 priority PAHs (EPA-PAH) in the 1970s (Keith 2015) (Figure 2.), which are frequently found in environmental monitoring samples: acenaphthene, acenaphthylene, anthracene, fluoranthene, fluorene, naphthalene, phenanthrene, pyrene, benz[*a*]anthracene,

benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, benzo[*ghi*]perylene, benzo[*a*]pyrene, chrysene, dibenz[*a,h*]anthracene, and indeno[1,2,3-*cd*]pyrene. Srogi (2007) identified the so-called Car-PAHs, namely benzo(a)anthracene, chrysene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(a)pyrene (B(a)P), dibenzo(a,h)anthracene, indeno(1,2,3-cd)pyrene and benzo(g,h,i)perylene. There is a list of 15 EU priority PAHs which are recommended for monitoring by the Scientific Committee on Food (SCF): benz[*a*]anthracene, benzo[*b*]fluoranthene, benzo[*j*]fluoranthene, benzo[*k*]fluoranthene, benzo[*ghi*]perylene, benzo[*a*]pyrene, chrysene, cyclopenta[*cd*]pyrene, dibenzo[*a,h*]anthracene, dibenzo[*a,e*]pyrene, dibenzo[*a,h*]pyrene, dibenzo[*a,i*]pyrene, dibenzo[*a,l*]pyrene, indeno[1,2,3-*cd*]pyrene, and 5-methylchrysene (Zelinkova and Wenzl 2015).

			
Naphthalene	Acenaphthene	Anthracene	Acenaphthylene
			
Phenanthrene	Fluorene	Chrysene	Fluoranthene
			
Pyrene	Benzo[ <i>b</i> ]fluoranthene	Benzo[ <i>a</i> ]pyrene	Benzo[ <i>k</i> ]fluoranthene
			
Benzo[ <i>a</i> ]anthracene	Indeno[1.2.3- <i>cd</i> ]pyrene	Dibenz[ <i>a,h</i> ]anthracene	Benzo[ <i>ghi</i> ]perylene

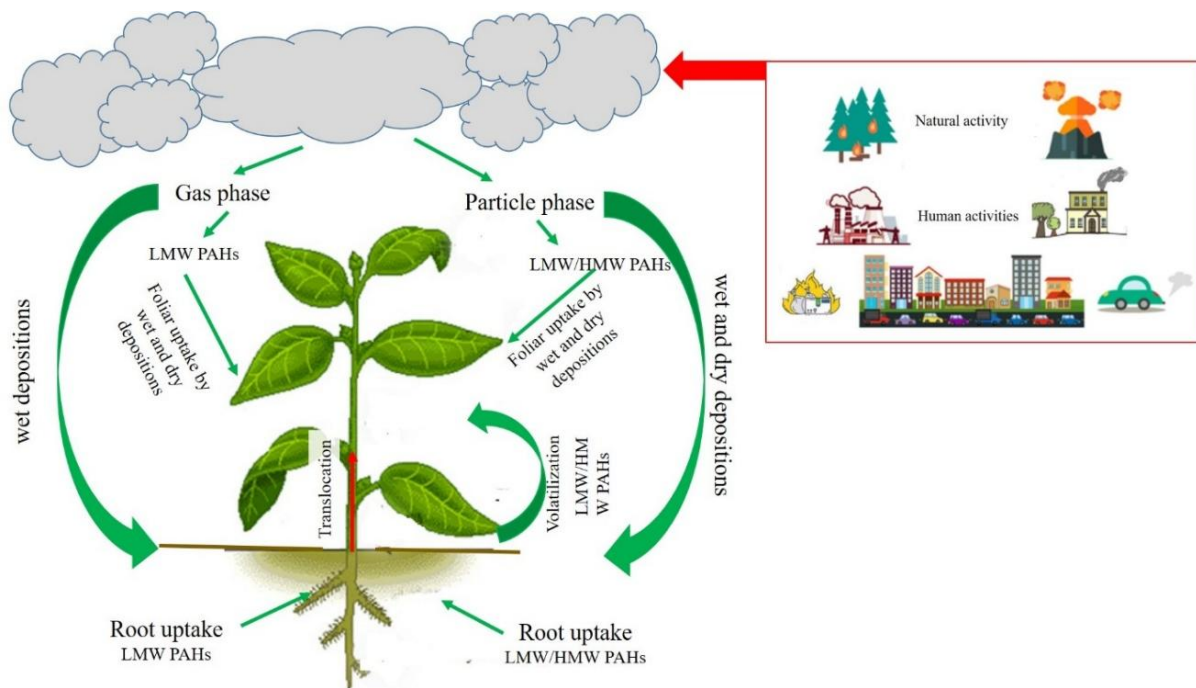
**Figure 2.** US EPA priority pollutant PAH compounds

In the atmosphere, low molecular weight PAHs (which consist of two-four aromatic fused rings) typically occur in gaseous form, higher molecular weight PAHs (with more than four rings) typically bound to particles (Liu et al. 2022). Higher plants are typically exposed to both gaseous and particle-bound compounds, via different pathways.

Foliar uptake has been demonstrated to present the major PAH exposure pathway for plants (Tao et al. 2006). Gaseous PAHs can directly pass through the stomata (Huang et al. 2018) or can enter the plants' interior via diffusion through the wax and cuticular lamellae (Lehndorff and Schwark, 2004). Also, gaseous PAHs can be washed out by precipitation, this process is called wet deposition (Smith & Jones 2000).

Exposure by particles is called dry deposition, meaning that particles are settled on plant surfaces. Particles can enter directly through the stomata or PAHs will desorb from these particles and transferred to the cuticular wax (Wang et al. 2017).

Atmospheric PAHs (mostly particle-phase) can also reach the soil, establishing an additional air-soil-root exposure route for plants (Krauss et al. 2005). However, as HMW PAHs will typically adsorb to soil particles, uptake from the soil mainly works for LMW PAHs (Jiao et al. 2023). Figure 3. summarises possible exposure routes for both LMW and HMW PAHs.



**Figure 3.** Schematic diagram summarising potential exposure routes for LMW and HMW PAHs, including both foliar and soil uptake.

## **2. 5. Heavy metals in the atmosphere**

Heavy metals, often defined as metallic elements with a high density that are toxic or poisonous at low concentrations, play a significant role in environmental pollution, particularly in the atmosphere (Popoola et al. 2018). Common heavy metals include Cr, As, Ni, Pb, Zn, Cu, V and Cd are carcinogenic, Cd and As are potentially mutagenic and Pb and Hg are fetal toxic (Duan & Tan 2013).

Heavy metals can enter the atmosphere through various sources, including industrial processes, vehicular emissions, and the burning of fossil fuels. For example, coal-fired power plants are known to release significant quantities of mercury and arsenic, while mining operations can disseminate lead and cadmium into the air. Additionally, waste incineration often contributes to the atmospheric levels of these toxic metals.

Heavy metals not only pose a risk to human health but also have detrimental effects on the environment. They can accumulate in soil and water, affecting wildlife and disrupting ecosystems. Higher concentrations of heavy metals can lead to bioaccumulation in food chains, causing toxicity in species that consume contaminated plants or animals (Zaynab et al. 2022).

Both traffic-related non-emission sources and open burning of waste contribute to atmospheric heavy metal levels. Depending on the composition of waste, significant amounts of Cd, Zn, Sb can be released (Kováts et al. 2022, 2023). Cr, Pb, S, Zn and V were found at relatively high concentrations in road dust (Rienda et al. 2023).

## **2. 6. Phytotoxicity of atmospheric pollutants**

Phytotoxicity refers to the adverse effects of substances on plant growth and development, often manifested as inhibited root and shoot growth, chlorosis, and leaf necrosis. Research has shown that PAHs can significantly impact plant life, leading to a condition known as phytotoxicity. The degree of phytotoxicity can vary widely depending on the specific PAH compound, its concentration, exposure duration, and the plant species involved. Common symptoms of PAH-induced phytotoxicity include inhibited seed germination, reduced root and shoot growth, chlorosis, and even plant mortality (Jajoo 2017). The mechanisms behind the phytotoxic effects of PAHs are manifold. These compounds can disrupt physiological processes in plants, including photosynthesis, respiration, and nutrient uptake.

The hydrophobic nature of PAHs allows them to strongly adsorb onto soil particles, affecting their bioavailability and consequently influencing how plants interact with their surrounding environment. Furthermore, the uptake of PAHs through roots can cause oxidative stress in plants, leading to the production of reactive oxygen species (ROS), which can damage cellular structures and interfere with metabolic pathways. Certain plant species exhibit varying levels of resilience to PAH exposure, making some more suitable for phytoremediation strategies aimed at reducing PAH concentrations in contaminated soils. (Hubai et al. 2022)

In conclusion, understanding the phytotoxicity of PAHs is crucial for developing effective environmental management practices and remediation strategies. Continued research on this topic will provide valuable insights into mitigating the adverse effects of PAHs on plant life and ensuring ecological balance.

Heavy metals such as lead (Pb), cadmium (Cd), arsenic (As), mercury (Hg), and chromium (Cr) are prominent contributors to contamination and can impair plant physiological processes (Terekhova et al. 2021). When atmospheric fallout is addressed, cadmium, lead, copper and zinc are most frequently discussed (Hu et al. 2013). While Cd and Pb are ultimately toxic, Zn and Cu are essential elements and their toxicity will depend on their concentration (Hough et al. 2004).

## **2. 7. Mechanisms of phytotoxicity**

The mechanism by which PAHs and heavy metals exert their toxic effects on plants can involve several pathways:

1. Inhibition of growth: Heavy metals can interfere with nutrient uptake by competing with essential minerals, leading to deficiencies that stunt growth.
2. Oxidative stress: The presence of heavy metals can induce the generation of reactive oxygen species (ROS), which can damage cellular structures, including lipids, proteins, and nucleic acids.
3. Alteration of photosynthesis: Heavy metals can disrupt the chlorophyll synthesis pathway, diminishing photosynthetic efficiency, which is vital for plant energy production.
4. Enzyme inhibition: Many heavy metals can inhibit the activity of crucial enzymes involved in metabolic processes, further impairing plant health and growth. (Asati et al. 2016)

### **2. 7. 1. Oxidative stress in plants**

Oxidative stress is a critical condition that affects the health and functioning of plants. It arises when there is an imbalance between the production of reactive oxygen species (ROS) and the plant's ability to detoxify these harmful compounds or repair the resulting damage. This imbalance can be triggered by various environmental stressors, including extreme temperatures, drought, salinity, and pathogen attack. Under normal circumstances, plants produce ROS as a part of their metabolic processes. However, when the levels of these reactive molecules exceed the plant's antioxidant defense capabilities, oxidative stress ensues. This can lead to cellular damage, affecting key components such as lipids, proteins, and nucleic acids, ultimately impairing plant growth and development. The response of plants to oxidative stress involves both avoidance and tolerance mechanisms. Plants employ a range of antioxidants, including enzymes like superoxide dismutase (SOD), catalase, and peroxidases, to mitigate the effects of ROS. Additionally, non-enzymatic antioxidants, such as ascorbate (vitamin C), glutathione, and flavonoids, play vital roles in protecting plant cells from oxidative damage. Research into oxidative stress in plants is paramount due to its implications for agriculture and food security. Understanding how plants cope with oxidative stress can lead to the development of strategies to enhance crop resilience against abiotic and biotic stresses. These strategies may include breeding for stress-resistant varieties or utilizing biostimulants that enhance the plant's

antioxidative capacity. In conclusion, oxidative stress represents a significant challenge for plant health, influencing productivity and sustainability in agriculture. Continued research in this field is essential to develop solutions that can safeguard plant integrity in the face of increasing environmental stressors. (Hubai et al. 2021).

Wu et al. (2017) examined the chemical composition and toxicity of fine PM emitted during the combustion of petrol and diesel fuels and found significant correlation between PAHs concentrations and intracellular ROS generation. In an experiment of Liu et al. (2009), treatment with phenanthrene resulted in increased levels of H<sub>2</sub>O<sub>2</sub> in *Arabidopsis thaliana* seedlings.

Antioxidant enzymes play a crucial role in the defense mechanisms of plants, serving not only to mitigate oxidative stress but also to maintain overall cellular health. These enzymes are vital in protecting plant cells from damage caused by reactive oxygen species (ROS), which are produced as byproducts of metabolic processes, especially during environmental stress conditions like drought, salinity, and extreme temperatures.

### **Types of antioxidant enzymes**

The primary antioxidant enzymes found in plants include:

1. **Superoxide Dismutase (SOD):** This enzyme catalyzes the dismutation of superoxide radicals into hydrogen peroxide and oxygen. SOD is often the first line of defense against oxidative stress in plant cells.
2. **Catalase (CAT):** Catalase plays a critical role in breaking down hydrogen peroxide into water and oxygen, thereby preventing potential cellular damage.
3. **Peroxidase (POD):** Peroxidases contribute to the detoxification of hydrogen peroxide and are involved in various biochemical processes, including cell wall lignification and hormone signaling.
4. **Glutathione Peroxidase (GPX):** This enzyme utilizes glutathione as a substrate to reduce peroxides, thus protecting against oxidative damage.

Antioxidant enzymes are essential for maintaining the redox balance within plant cells, enhancing their ability to withstand abiotic stresses. Increased activity of these enzymes is often correlated with improved plant resilience. For instance, during drought conditions, plants exhibit elevated levels of SOD and CAT, highlighting their role in stress adaptation. Moreover,

the expression of antioxidant enzymes can be influenced by various factors such as light intensity, temperature, and nutrient availability. Understanding how these enzymes operate opens up new avenues for improving crop tolerance, which is imperative in the face of changing climate conditions (Bhaduri & Fulekar 2012).

The study of antioxidant enzymes in plants is vital for the development of stress-resistant crops, ultimately contributing to food security and sustainable agriculture. Continued research in this field may lead to innovative strategies for enhancing plant resilience against environmental challenges. By exploring the intricate roles of these enzymes, we can harness their potential to improve agricultural practices and ensure the health of ecosystems.

Antioxidant enzymes have been long in use to assess air pollution-related plant damage (Keller, 1974). POD was found reliable tool to assess the condition of both heavy-metal stressed experimental plants (Jaskulak et al. 2018) and plants under PAH exposure as well (Liu et al. 2009). Wei et al. (2014) reported the concentration-dependent increase in POD concentrations when phenanthrene was applied. Fluoranthene showed a similar triggering effect on POD activity (Tomar & Jajoo 2015). Some heavy metals typically occurring in atmospheric samples have also shown direct effect on ROS production such as Cd (Akinyemi et al. 2017) or Zn (Chemingui et al. 2019).

### **2. 7. 2. Photosynthetic pigments and air pollution: A serious concern**

Photosynthetic pigments, primarily chlorophylls, carotenoids, and phycobilins, play a vital role in the process of photosynthesis, enabling plants, algae, and certain bacteria to convert light energy into chemical energy. These pigments not only contribute to the energy production of ecosystems but also help in oxygen generation, which is critical for the survival of aerobic life forms on Earth. However, the increasing levels of air pollution pose significant threats to these essential pigments, subsequently impacting global ecosystems. Air pollutants, such as nitrogen oxides (NO<sub>x</sub>), sulfur dioxide (SO<sub>2</sub>), and particulate matter, can adversely affect the health of photosynthetic organisms. For instance, high levels of nitrogen compounds can lead to nutrient imbalances in aquatic environments, promoting algal blooms that deplete oxygen in water bodies and affect marine life. Additionally, the presence of polycyclic aromatic hydrocarbons (PAHs) and heavy metals in the air can accumulate in plant tissues, interfering with pigment synthesis and function. Research indicates that air pollution can cause structural damage to chlorophyll molecules, impairing their ability to absorb light effectively. This

degradation reduces the overall efficiency of photosynthesis, which can have cascading effects on plant growth, agricultural yields, and, ultimately, food security. Furthermore, as less carbon dioxide is sequestered by affected plants, the compounding effect on climate change becomes evident (Joshi & Swami 2009).

Decreased levels of photosynthetic pigments chlorophyll-a and b were already used as potential tools already in the 1970's (Rabe & Kreeb 1979). Damage to the photosynthetic machinery can be indicated by the reduction in the levels of PSNI and PSNII photochemical activity (Kreslavski et al. 2017), or reduction in net photosynthetic rate and of Rubisco activity (Tomar et al. 2019).

### **2. 7. 3. Phytotoxicity testing**

Phytotoxicity tests are essential evaluations used to determine the toxic effects of chemicals on plants. The primary objective of phytotoxicity testing is to safeguard plant health and ensure ecological balance. By understanding how specific substances affect plants, researchers can make informed decisions about product use and regulatory standards. Reliable testing methods and transparent reporting are crucial for compliance and environmental protection.

Phytotoxicity tests follow standardized protocols to ensure reliability and reproducibility. Key steps include (Barral & Paradelo 2011):

- Selection of test species: Choosing sensitive species that are representative of the ecological contexts where the substance may be applied.
- Application of test substances: Carefully administering the chemicals in controlled conditions to avoid contamination and variability.
- Data collection and analysis: Systematically observing and recording plant responses, followed by statistical analysis to interpret the results.

## **Types of phytotoxicity tests**

1. **Seed germination tests:** These tests evaluate the effects of chemicals on seed sprouting and early development. Typically, seeds are grown in a controlled setting with varying concentrations of the test substance to determine the rate of germination and the health of seedlings.
2. **Growth tests:** This type examines the impact of substances on the growth and biomass accumulation of plants. It often involves monitoring parameters such as height, leaf area, and overall vigor over a specified growth period.
3. **Bioassays:** These are broader assessments that may involve multiple plant species to evaluate relative toxicity levels. Bioassays help identify the potential risks posed by new agricultural chemicals to diverse ecosystems.

### **2. 7. 4. The OECD vegetative vigour test**

The No. 227 OECD GUIDELINE FOR THE TESTING OF CHEMICALS: Terrestrial Plant Test: Vegetative Vigour Test (OECD 2006) was originally introduced to assess the phytotoxicity of general chemicals, biocides and crop protection products (Boutin et al. 2012; Carpenter et al. 2013). It is a critical method used to assess the growth potential and overall health of various plant species. This protocol is particularly significant for evaluating the effects of environmental conditions and agricultural practices on plant development.

The primary aim of the Vegetative Vigour Test is to determine the growth capacity of test plant species under controlled conditions. This assessment plays an essential role in ecological studies, agricultural research, and regulatory decision-making. By understanding how plants respond to different external factors, researchers can make informed choices about crop management, environmental conservation, and sustainability practices. It is considered integral to the safety assessment of new agrochemicals and genetically modified organisms. By establishing how these products affect plant growth and vigour, stakeholders can better understand potential impacts on ecosystems and food production systems (Mendes da Silva & Andrade–Vieira 2025)

Based on the step-by-step protocol laid out by the Guideline, Kováts et al. (2017) made some modifications to assess the phytotoxic potential of airborne particles. The major modification was using repeated treatments. While spraying plants is a single event, wet

deposition may occur frequently, even lasting for longer periods, depending on the nature of the precipitation. The study applied three consecutive treatments on cucumber (*Cucumis sativus* L.) test plants, assessing potential phytotoxicity of urban samples collected in Budapest. At low doses applied, the extract triggered stimulative effect both in mass and shoot length in comparison to the control, most possibly due to the nutrient content of the sample. At higher doses, however, necrosis appeared.

Another modification to the original protocol was the introduction of new end-points. The Guideline suggests phytotoxicity assessments based on growth impairment (biomass and/or shoot reduction) and physical symptoms. In order to get a more comprehensive characterisation of possible phytotoxic responses, these end-points were supplemented by biochemical markers such as concentration of photosynthetic pigments such as chl-a, chl-b, carotene, the antioxidant enzyme peroxidase and total protein content (Kováts et al. 2020). This study assessed seasonal pattern of rural atmospheric PM samples. In concordance with literature data, highest toxicity was experienced in case of samples collected in colder months. The addition of these new end-points could differentiate between ecological effects as both biomass and protein concentration reflected the higher toxicity of cold seasons, while photosynthetic pigments seemed to indicate the nutrient content of the samples. Similar, so-called ‘Janus-faced’ effects were experienced in a similar study of Hubai et al. (2021). This phenomenon is anticipated in cases where the sample contains both nutrients and toxic compounds. At low concentrations/doses, nutrients are supposed to mask the toxic effect at low concentrations.

In a more comprehensive study, aqueous extract simulating emissions of diesel-powered vehicles was used to compare the sensitivity of different, common European roadside plants (Kováts et al. 2021). The batch of test plants included the following species: wild parsnip (*Pastinaca sativa* L.), common daisy (*Bellis perennis* L.), common thistle (*Cirsium arvense* L.) Scop., common sowthistle (*Sonchus oleraceus* L.), common dandelion (*Taraxacum officinale* F.H. Wigg.), soapwort (*Saponaria officinalis* L.), goosefoot (*Chenopodium album* L.), white clover (*Trifolium repens* L.), Mediterranean sage (*Salvia aethiopsis* L.), hoary plantain (*Plantago media* L.), common sorrel (*Rumex acetosa* L.), meadow buttercup (*Ranunculus acris* L.) and wood avens (*Geum urbanum* L.) The major finding of the study was that a relatively significant portion of the test plants showed resistance to traffic-related air pollution.

Hubai et al. (2021) applied similar diesel exhaust sample to test if essential oil production in the medicinal plant sweet basil (*Ocimum basilicum* L.) can be used as a measure of chemical stress. Linalool production proved a reliable marker.

Another line of studies assessed the potential phytotoxicity of emissions generated during burning of domestic waste. Hubai et al. (2022) demonstrated clear concentration-effect relationship in analysing samples generated during controlled combustion of different plastic waste types such as polyvinyl chloride (PVC), polyurethane (PUR), polypropylene (PP), polystyrene (PS) and polyethylene (PE). All plastic waste burning samples resulted in measureable plant stress indicated by the elevated levels of peroxidase enzyme activity.

### **2. 7. 5. Bioaccumulation of polycyclic aromatic hydrocarbons in plants**

The extent to which plants absorb PAHs will depend on multiple factors, including the type of plant species, the concentration of PAHs in the environment, and environmental conditions such as pH, temperature, and moisture. Certain plant species have demonstrated a higher efficiency in the uptake and accumulation of PAHs such as oak and pine (Huang et al. 2018), making them potential bioindicators of environmental contamination. Research indicates that the bioaccumulation of PAHs can negatively affect plant growth, physiology, and overall health. The presence of these hydrocarbons can disrupt photosynthetic functions, reduce biomass, and lead to a decline in reproductive success.

Furthermore, plants that accumulate significant levels of PAHs can enter the food chain, potentially posing risks to herbivores and, subsequently, higher trophic levels, including humans (Esmaeili et al. 2021). Considering human risk, dietary uptake proved the major exposure route of PAHs (Ramesh et al. 2004). PAH content of vegetables can reach very high levels: Tesi et al. (2021) for example reported that in some parts of Nigeria, some vegetables could be unsuitable for consumption due to their high PAH, especially benzo(a)pyrene content.

Early studies already reported bioaccumulation of atmospheric PAHs in vegetables grown in industrial areas (Voutsas et al. 1996; Kipopoulou et al. 1999). Atmospheric uptake was also demonstrated in vegetables grown in the vicinity of thermal power plants (Khillare et al. 2012).

A wide range of vegetables have been tested for accumulation of PAHs. Nadal et al. (2009) used chard (*Beta vulgaris*) collected from residential area, industrial and unpolluted in

Tarragona County (Spain) to assess PAH exposure. Xiong et al. (2017) reported the uptake of 15 PAHs in cabbage. Amato-Lourenco et al. (2016) demonstrated the accumulation of PAHs in *Brassica oleracea* var. *acephala* (collard greens) and *Spinacia oleracea* (spinach) in 10 community gardens in Sao Paulo (Brasil) affected by vehicular emissions. Spinach has proven a good accumulator: Jia et al. (2018) reported concentrations of 16 PAHs in vegetables collected from near industrial areas of Shanghai with the highest values in spinach (*Spinacia oleracea* var.) (223.3-458.0  $\mu\text{g kg}^{-1}$ ), followed by Chinese cabbage (*Brassica rapa* var.), (206.0-348.1  $\mu\text{g kg}^{-1}$ ), Shanghai green cabbage (*Brassica chinensis* var.), (206.4-284.7  $\mu\text{g kg}^{-1}$ ), and finally lettuce (*Lactuca sativa* L. (132.0-319.2  $\mu\text{g kg}^{-1}$ ). Different Brassica species are widely used: Mo et al. (2009) assessed the concentration of 16 PAHs in samples from Pearl River Delta, South China and found the following values: *Brassica parachinensis* (flowering Chinese cabbage) 438  $\mu\text{g kg}^{-1}$ , *Brassica chinensis* (paitsai) 950  $\mu\text{g kg}^{-1}$  and *Brassica juncea* (mustard) 1790  $\mu\text{g kg}^{-1}$ .

While most studies assess accumulation of PAHs in crops collected for human consumption, relatively less publications assess bioaccumulation under controlled conditions. The OECD 227 Guideline was also adopted for treatment of test plants used for bioaccumulation studies. Teke et al. (2020) treated *L. sativa* with the extract of urban PM samples and assess the accumulation pattern of PAHs after a 21-day exposure period. Strong correlation was established between BCF and molecular weight of PAH compounds, showing the dominance of the lower molecular weight (LMW) PAH. Naphthalene had the highest concentration, in line with other bioaccumulation studies (Busso et al. 2018).

Kováts et al. (2022) carried out a comparative study to test bioaccumulation of diesel emission related PAHs using a relatively large batch of vegetables, including *Lactuca sativa* L. 'Kobak' (lettuce) (Family Asteraceae); *Eruca sativa* Mill. (rocket) (Family Brassicaceae); *Lepidium sativum* L. (garden cress) (Family Brassicaceae); *Apium graveolens* var. *secalinum* Alef. (leaf celery) (Family Apiaceae); *Valerianella locusta* L. (corn salad) (Family Caprifoliaceae); *Beta vulgaris* subsp. *vulgaris* convar. *cicla* L. 'Lukullus' (chard) (Family Amaranthaceae); *Spinacia oleracea* L. 'Matador' (spinach) (Family Amaranthaceae) and *Ocimum basilicum* 'Compact' (basil) (Family Lamiaceae). Accumulation pattern in most of the tested vegetables showed some similarity, with the dominance of lower molecular weight (LMW) PAH compounds. Chard, spinach and corn salad, however, were efficient in accumulating 6-ring PAHs as well.

Hubai et al. (2022) compared PAH accumulation patterns in *Plantago lanceolata* L. test plants experimentally treated with diesel emission extract samples vs. plants collected in the field, involving high-trafficked sites. These patterns showed good similarity.

## 3. Assessing Risk of Emissions Generated During Illegal Waste Burning: Phytotoxicity and Bioaccumulation

### 3. 1. Materials and methods

#### 3. 1. 1. Sampling and sample preparation

An aqueous extract of PM<sub>2.5-10</sub> was prepared according to Corrêa et al. (2016). PM<sub>2.5-10</sub> samples were collected from waste burning. Waste samples were selected to represent the average composition of domestic waste reported by the (Hungarian) National Statistics Agency. Similar ratios have been reported in an international context as well (Krecl et al. 2021).

Experimental burning was performed in a domestic cast-iron residential stove (type: *Servant S114*, rated at: 5 kW) connected to a 11.8 m high lined stack with inner diameter of 150 mm (Hoffer et al. 2020). The average combustion temperature was 310 °C (±56 °C), however, temperature moved in a relatively wide range between 200-450 °C. In case of plastic samples it was occasionally as high as 600 °C. PM<sub>10</sub> samples were collected on quartz filters (Advantec QR-100 Ø150 mm).

Each filter was cut into small pieces and placed in beaker with 1000 mL high purity water for 24h at room temperature and stirred several times. The extract was filtered through 0.45 µm pore size filter (GN-6 Membrane, 0.45 µm Hydrophilic mixed cellulose esters) and stored in 40 ml vials at -20 °C until use.

#### 3. 1. 2. Plant material used

White mustard

- Latin name: *Sinapis alba* L.
- Taxonomy: Brassicaceae
- Life cycle: Annual
- Raunkiaer classification: Therophyte  
(<https://floraveg.eu/taxon/overview/Sinapis%20alba>)
- Morphology: Leaves alternate, long, loosely branched, irregularly toothed, petiolate, hairy on both sides. Flowers small, four-petaled yellow, cruciform; stamens tetradynamous; pistil bicarpellate. Fruit a loose, round-ribbed achene, swollen with

seeds, with a long, bulbous beak, and a pod distributed on the leaf axils. The seeds are globose, yellowish. (Bones & Rossiter 2006).

*S. alba* L. (white or yellow mustard, also known as *Brassica hirta*) is native to the Mediterranean region. It is a winter-spring crop that can be grown for a short season of about 85- 95 days and it grows up to 100 cm high (Boscaro et al. 2018). It is found worldwide as a weed and as a cultivated species (Mitrović et al. 2020). It is drought-resistant and therefore suitable for production in warm, dry soil areas. It is often grown in rotation with cereal crops as a leaf, seed, or green manure crop. Variety selection is based on various factors, including expected price, potential return, and agronomic characteristics (Boscaro et al. 2018).

The crop is grown from seeds, and it contains biologically active compounds. The most important is a glucosinolate called sinalbin, phydroxybenzyl glucosinolate. It gives the seeds a bitter taste. The leaves are rich in vitamins, minerals, and fiber. It is used as a flavoring in mixed salads potherb. (Abbasi et al. 2011)

## Lettuce

- Latin name: *Lactuca sativa* L.
- Taxonomy: Asteraceae
- Life cycle: annual
- Raunkiaer classification: Hemicryptophyte, Therophyte  
(<https://floraveg.eu/taxon/overview/Lactuca%20sativa>)
- Morphology: The leaf is curled and basal leaves are oblong-ovate, with a lanceolate base. The inflorescence (capitulum, head) consist of 7-15 (35) yellow flowers (florets). Inflorescence 10-15 mm long, cylindrical; involucre bracts narrower than wide, pale green, white-margined, erect at maturity. The fruit (achen) has up to 5 to 7 lateral ribs, a beak, and a white pappus. Its length (including the beak) is 6-8 mm, and its color is white, cream, gray, brown, or black.

The genus *Lactuca* L. comprises about 100 species and is divided into some sections and subsections. *Lactuca sativa* L. is classified in section *Lactuca*, subsection *Lactuca*. *L. serriola* L. and other related species, for instance *L. virosa* L., *L. saligna* L. and *L. altaica* Fisch. & Mey., are also classified in that subsection (De Vries 1997).

Lettuce is one of the most common leafy vegetables. It is cultivated both outside and in the glasshouse. Nowadays the consumption of lettuce has spread tremendously throughout

the world due to its high nutritive value and medicinal importance. The species *L. sativa* is characterized by a high genetic diversity resulting from its polyphyletic origin and a complex domestication process. Lettuce is source of vitamins and contains many minerals including calcium, phosphorus, iodine, iron, copper, and arsenic (Noumedem et al. 2017).

### **3. 1. 3. Cultivation and treatment of test plants**

Test plants were cultivated and treated according to the protocol specified by the No. 227 OECD GUIDELINE FOR THE TESTING OF CHEMICALS: Terrestrial Plant Test: Vegetative Vigour Test. The Guideline has been adopted to test the potential phytotoxic effects of atmospheric PM and to assess bioaccumulation of atmospheric PM-bound PAHs (Teke et al. 2020).

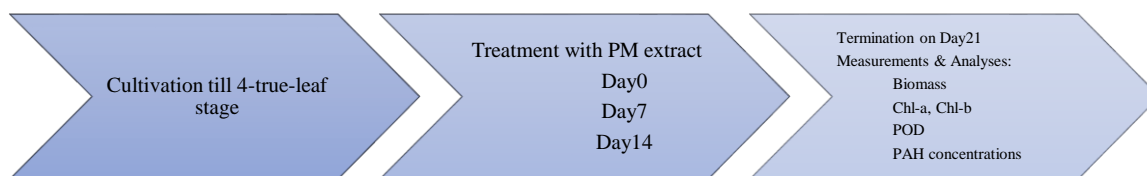
#### **Cultivation**

Test plants were cultivated in 15cm containers with 3 plants per pot, and 10 containers were used per each treatment. Plants were grown in commercial soil (pH:  $6.8 \pm 0.5$ ; N (m/m%): min 0.3; P<sub>2</sub>O<sub>5</sub> (m/m%): min 0.1; K<sub>2</sub>O (m/m%): min 0.3). Plants were kept in a glasshouse during cultivation and treatments, and environmental conditions were set according to the prescriptions of the Guideline. (temperature:  $22 \text{ }^{\circ}\text{C} \pm 10^{\circ}\text{C}$ ; humidity:  $70\% \pm 25\%$ ; photoperiod: minimum 16h light; light intensity:  $350 \pm 50 \text{ } \mu\text{E/m/s}$ ).

#### **Treatment of test plants**

Plants were sprayed and treated with the PM extract on the above-ground parts. CONXIN Q1P CX01-380 portable electric paint spray gun was used for spraying, and the treatment application volume was 5mL per pot. During PM treatment, each pot of soil was covered to avoid contamination.

According to the Guideline, the first treatment was applied when plants reached the 4 true leaf stage (Day0), followed by two additional treatments on Day7 and Day14. Steps of the test procedure are summarised in Figure 4.



**Figure 4.** Steps of the phytotoxicity testing procedure

### 3. 1. 4. Storage, preservation, preparation for analyses

After harvest, plant samples were taken immediately to the laboratory, washed with ionic load-free water, and frozen (-20°C) until analysis. Preparing the plant samples, a composite sample was prepared by using 30 test plants from each plant's part (leaves, stems, and seeds). Plant material was homogenized, and 10 g was grinded with 10 g anhydrous sodium sulphate in a ceramic mortar and was extracted with 20 mL n-hexane three times for 20 min with Ultra-sonic extraction. After that, 10 mL of acetone was added to the samples and was spiked with 100 µl of 0.01 µg mL<sup>-1</sup> deuterated PAH surrogate mixture (Naphtalene-d8, Acenaphthene-d10, Phenanthrene d10, Chrysene-d12, Benzo(a)pyrene-d12, Perylene-d12; Restek Corporation, USA). After the extraction the sample extract was concentrated in a dry nitrogen stream to 1 mL. All samples an additional solid phase silica gel and alumina oxide sample clean-up was performed.

### 3. 1. 5. Chemical analyses of plant samples

PAH concentrations in aerosol extract was measured by gas chromatographic-mass spectrometry, according to MSZ 1484-6:2003 standard (MSZ 1484-6:2003: Testing of waters. Determination of polycyclic aromatic hydrocarbons (PAH)). The analysis of accumulated PAHs in plant sample was carried out based on MSZ EN 15527:2009 (Characterization of waste. Determination of polycyclic aromatic hydrocarbons (PAH) in waste using gas chromatography mass spectrometry (GC/MS)).

The analysis of accumulated PAHs in plant samples was carried out based on MSZ EN 15527:2009 (Characterization of waste. Determination of polycyclic aromatic hydrocarbons (PAH) in waste using gas chromatography mass spectrometry (GC/MS). HP-6890 gas chromatograph was coupled to an HP 5973 (Agilent Technologies, Palo-Alto, USA) quadrupole mass spectrometer (low-resolution single MS) in this measurements.

Measurements were performed in harmony with the laboratory accreditation to ISO/IEC17025:2018 and followed the laboratory's internal quality management system guidelines. The LOD (Limit of detection) values of accumulated PAHs in plant samples were between 0.01  $\mu\text{g kg}^{-1}$  to 0.015  $\mu\text{g kg}^{-1}$  and in aerosol extract between 0.00015 to 0.0003  $\mu\text{g L}^{-1}$ . The LOQ (Limit of quantification) values in plant samples were ranged from 0,031 to 0.05  $\mu\text{g kg}^{-1}$  and 0.00029 to 0.001  $\mu\text{g L}^{-1}$  in aerosol filter water extract samples.

### 3. 1. 6. Measurement of biomass

According to the OECD GUIDELINE FOR TESTING, the biomass measurement were directly performed at the end of the test. Biomass measurement should be conducted alongside a visual symptoms of phytotoxic effects, including chlorosis, necrosis, wilting, and morphological deformation of leaves and stems. Biomass can be determined by measuring the final shoot weight—preferably the dry weight—obtained by harvesting the shoot at the soil surface and drying it at 60 °C until a constant weight is reached; alternatively, shoot height may be recorded as a complementary parameter.

### 3. 1. 7. Photosynthetic pigment measurement

Photosynthetic pigment contents were measured using the spectrophotometric method (Bag et al., 2012). The sample was 0.2 g from each plant's leaf. (10-10 plants in each group) The sample was homogenized with 15 mL of 80% acetone (Fisher Chemicals,  $\geq 99.8\%$ , ACS; VWR International Ltd., Debrecen Hungary), and centrifuged at 4500 rpm for 10 min (Mistral 2000 MSE, DJB Labcare Ltd. Newport Pagnell, Buckinghamshire, UK). After that, the supernatant was separated and diluted with 80% acetone to a final volume of 25 mL. Absorbance was measured at 470 nm, 647 nm, and 670 nm for photosynthetic pigment measurement using a UV-VIS spectrophotometer. (Metertech SP8001, ABL&E-JASKO Hungary Budapest, Hungary).

The following equations were used for the final calculation:

$$Chl - a = [9.78 \times 10^{-6} - 0.99 \times 10^{-6}] [V/1000 \times W]$$

$$Chl - b = [21.4 \times 10^{-6} - 465 \times 10^{-6}] \times [V/1000 \times W]$$

$$Carotene = [4.69 \times 10^{-4} - 0.268 \times (5.13 \times 10^{-6} + 20.41 \times 10^{-6})] \times [V/1000 \times W]$$

Where E: extinction values at wavelengths; V: final volume (25 mL); W: mass of sample (0.2 g)

### 3. 1. 8. Peroxidase measurement

The sample was made from 0.2 g of each plant's leaf, which was frozen and homogenized in an ice-cold mortar and pestle in phosphate buffer (50 mmol L<sup>-1</sup>, pH 7) containing 1 mmol L<sup>-1</sup> EDTA and 0.5 mmol L<sup>-1</sup> Phenylmethylsulfonyl fluoride (PMSF). After preparing, these samples were centrifuged for 20 min at 15.000 x g at 4 °C. Peroxidase activity (A 654) was measured according to the protocol described in Imberty et al. with minor modifications.

Tetramethylbenzidine (TMB) peroxidase activity (A 654) was measured following the protocol described in Imberty (Imberty et al, 1984) with minor modifications.

### 3. 1. 9. Statistical approach

The determinants of the accumulated PAH amounts in different plant parts and the PAH concentration of the extract were determined using Pearson's method. Based on analytical results, BCF were assessed comparing PAH concentrations in the seed/extract, leaf/extract, and stem/extract. The following equation was used (Kacálková and Tlustoš 2011):

$$BCF = \frac{\text{PAH concentration in plant sample } [\mu\text{g} / \text{kg}]}{\text{PAH concentration in the PM extract } [\mu\text{g} / \text{L}]}$$

The statistical analyses were performed by using the R 4.0.0 program (<http://cran.r-project.org/src/base/R-4/R-4.0.0.tar.gz>) programme ggfortify package (<https://CRAN.R-project.org/package=ggfortify>) and Rcmdr package. The analysis were performed using one-way and two-way ANOVA or Kruskal-Wallis chi-squared followed by the Tukey post hoc test. Statistical significance was defined as  $p \leq 0.05$ .

## 3. 2. Results and discussion

### 3. 2. 1. Composition of the PM extract

A total of 19 polycyclic aromatic hydrocarbons (PAHs) were identified in the extract, including all 16 priority PAHs designated by the U.S. Environmental Protection Agency (EPA) (Table 1.). These compounds are classified by the EPA as presenting the greatest environmental risk. (Keith, L. H. 2015)

**Table 1.** Concentration of PAHs in the waste PM<sub>10</sub> extract

Number of Rings	PAHs	PM <sub>10</sub> Extract (µg L <sup>-1</sup> )
two rings	Naphthalene	0.931
	2-methyl-naphthalene	0.377
	1-methyl-naphthalene	0.323
three rings	Acenaphthylene	7.41
	Acenaphthene	4.65
	Fluorene	6.13
	Phenanthrene	47.2
	Anthracene	8.97
four rings	Fluoranthene	22
	Pyrene	19.3
	Benanthracene	1.55
	Chrysene	2.9
	Benzo(b)fluoranthene	1.03
	Benzo(k)fluoranthene	0.38
	five rings	Benzo(e)pyrene
Benzo(a)pyrene		0.603
Dibenzo(a,h)anthracene		0.379
six rings	Indeno1.2.3CD-Pyrene	0.108
	Benzo(g,h,i)perylene	0.477
<b>Total PAHs</b>		<b>125</b>

The predominant PAHs in the extract were phenanthrene (47.2 µg L<sup>-1</sup>), fluoranthene (22 µg L<sup>-1</sup>), and pyrene (19.3 µg L<sup>-1</sup>). Given their potential sources, their presence cannot be considered source-specific, as various combustion activities can produce these compounds. Kwarteng et al. (2022) reported elevated levels of phenanthrene, fluoranthene, and pyrene during monitoring of occupational exposure to PM<sub>2.5</sub>-bound PAHs at a waste recycling site in

Ghana. Phenanthrene also emerged as a dominant PAH in studies characterizing emissions from green waste burning (Noblet et al. 2021) and the open burning of agricultural debris (Kakareka & Kukharchyk 2003).

The five-ring dibenzo(a,h)anthracene and six-ring benzo(g,h,i)perylene were detected at relatively high concentrations  $0.379 \mu\text{g L}^{-1}$  and  $0.477 \mu\text{g L}^{-1}$ , respectively. One possible source of these compounds is polyurethane (PU); Kováts et al. (2021) reported similarly elevated levels when PU was burned under controlled conditions. The concurrent presence of benzo(a)pyrene and indeno(1,2,3-cd)pyrene suggests a contribution from waste tire materials. Capozzi et al. (2017) investigated PAH accumulation in *Robinia pseudoacacia* leaves in areas with sporadic waste burning and attributed the presence of these PAHs to tire combustion. Similarly, Mendes et al. (2022) analyzed PAH emissions from waste tire burning at varying temperatures, detecting a broad spectrum of PAHs. They observed a decline in total PAH concentration with increasing temperature, from 18.83 mg PAH per kg of fuel burned at 650 °C to 1.91 mg PAH per kg at 900 °C.

Among the potentially toxic heavy metals, antimony (Sb) and zinc (Zn) were detected in significant concentrations (Table 2.). A likely source of Sb is the polyethylene terephthalate (PET) present in the burned waste. PET is widely used in food and beverage packaging (Filella 2020) and constitutes a substantial portion of household waste (Eriksen & Astrup 2019). Antimony trioxide ( $\text{Sb}_2\text{O}_3$ ) is commonly used as a catalyst in the production of PET, resulting in residual antimony levels that can reach up to  $300 \text{ mg kg}^{-1}$  (Chapa-Martínez et al. 2016). Sb has been shown to exhibit phytotoxic effects; for example, Zhu et al. (2022). Various barium compounds are utilized in plastics, with barium sulfate ( $\text{BaSO}_4$ ) being the most prevalent, serving as an inert white filler and extender (Turner & Filella 2020). While barium is not essential for plant growth, it can be toxic when present at elevated levels (de Souza Cardoso & Monteiro 2021).

**Table 2.** Heavy metal concentrations in the PM<sub>10</sub> extract

Heavy metal	Waste PM10 Extract ( $\mu\text{g L}^{-1}$ )
Zn	219
Ba	23
Sb	6.2
Cu	<5.00
Mo	<2.00
Ni	<2.00
Al	<10.0
B	<10.0
As	<1.000
Ag	<1.000
Co	<1.000
Cr	<1.000
Pb	<1.000
Se	<1.000
Sn	<1.000
Cd	<0.500
Na	<0.500
Hg	<0.200

Zinc (Zn) was detected in PM<sub>10</sub> samples during a comparative study, in which various plastic materials were burned under controlled conditions. Zn concentrations ranged from 8.23 to 78.8  $\mu\text{g kg}^{-1}$ , with polyvinyl chloride (PVC) producing the highest levels (Kováts et al. 2022). Similarly, Valavanidis et al. analyzed particulate soot emissions from the controlled combustion of six different plastics-including PVC, low- and high-density polyethylene, polystyrene, polypropylene, and polyethylene terephthalate-reporting the presence of potentially toxic heavy metals. Compared to other samples, emissions from PET burning contained notably high levels of aluminum (Al), lead (Pb), copper (Cu), nickel (Ni), and zinc (Zn). Hama et al. (2021) also reported elevated concentrations of Zn and Pb in PM<sub>2.5</sub> samples collected from solid waste burning activities. Similarly, Peter et al. (2018) detected a broad spectrum of toxic heavy metals-including arsenic (As), cadmium (Cd), copper (Cu), chromium (Cr), nickel (Ni), lead (Pb), selenium (Se), and zinc (Zn)-in PM<sub>2.5</sub> samples from a municipal waste dumpsite affected by both accidental and intentional fires. Heavy metals were even identified in firewater samples collected from experimentally burned waste dumps (Kucerova et al. 2023).

Experimental studies have demonstrated that zinc (Zn) can trigger antioxidative responses (Akanbi-Gada et al. 2019). However, in soils contaminated with heavy metals, Zn has been identified as the primary contributor to phytotoxic effects (Beyer et al. 2013).

### 3. 2. 2. Phytotoxicity assessment

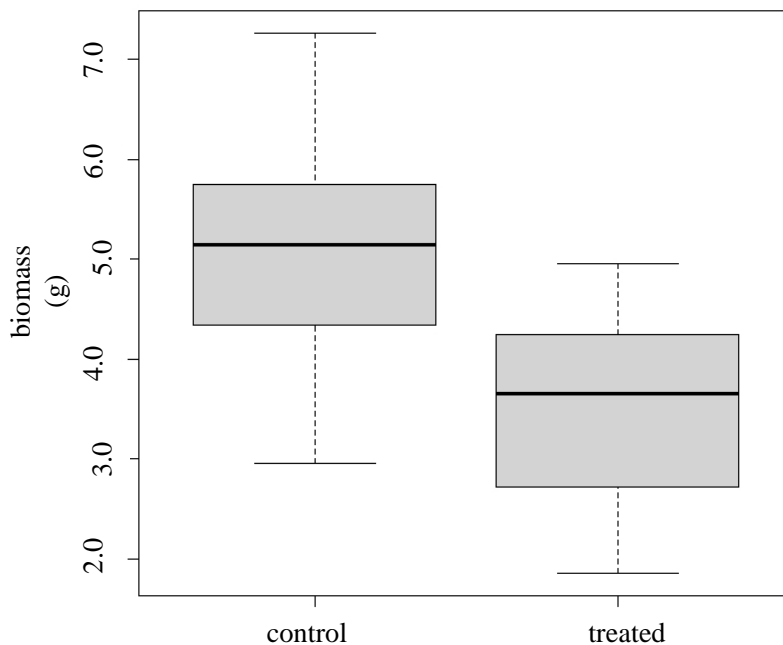
Table 3. summarizes the phytotoxicity test results, comparing the two test plants. *Lactuca sativa* exhibited greater sensitivity, with three test endpoints indicating toxic effects.

**Table 3.** Summary of phytotoxicity test results. + indicates a significant effect; 0: effect recorded was non-significant.

Standard	End-Point	<i>Lactuca sativa</i>	<i>Sinapis alba</i>
OECD 227	Biomass	+	0
	Chl-a	0	0
	Chl-b	+	0
	Carotene	0	0
	POD	+	+

## Decrease in biomass

Regarding biomass reduction following treatment, lettuce exhibited a significant response ( $Df = 1, F = 16.43, p = 0.000385$ ) (Figure 5.), whereas mustard showed no significant response ( $Df = 1, F = 0.156, p = 0.694$ ).



**Figure 5.** Reduction in biomass of treated lettuce plants compared to the control.

Reduced biomass in treated groups appears to be a common indicator of the harmful effects of air pollution. Kováts et al. assessed the phytotoxicity of urban  $PM_{2.5}$  extracts on 13 common roadside plants and identified biomass reduction as the most sensitive and distinguishing response (Kováts et al. 2021). Biomass was identified as the most sensitive endpoint in the study by Storch-Böhm et al., where test plants were exposed to diesel engine exhaust. Pavlík et al. exposed *Lactuca sativa* plants to atmospheric particulate matter suspensions and observed various symptoms, including reduced biomass in treated plants.

Exposure to individual PAHs has also been shown to inhibit plant growth in several studies. Foliar application of anthracene led to reduced biomass and decreased photosynthetic efficiency in lettuce plants (Wieczorek & Wieczorek 2007). Similarly, treatment with anthracene and benzo(k)fluoranthene caused biomass reduction in celery plants (Wieczorek et al. 2015). The phytotoxicity of atmospheric phenanthrene has been associated with decreased

biomass in clover (Desalme et al. 2011). The foliar exposure to phenanthrene resulted in a concentration-dependent decline in growth, photosynthesis, and chlorophyll content (Ahammed et al. 2012).

### **Chlorophyll-a and chlorophyll-b content**

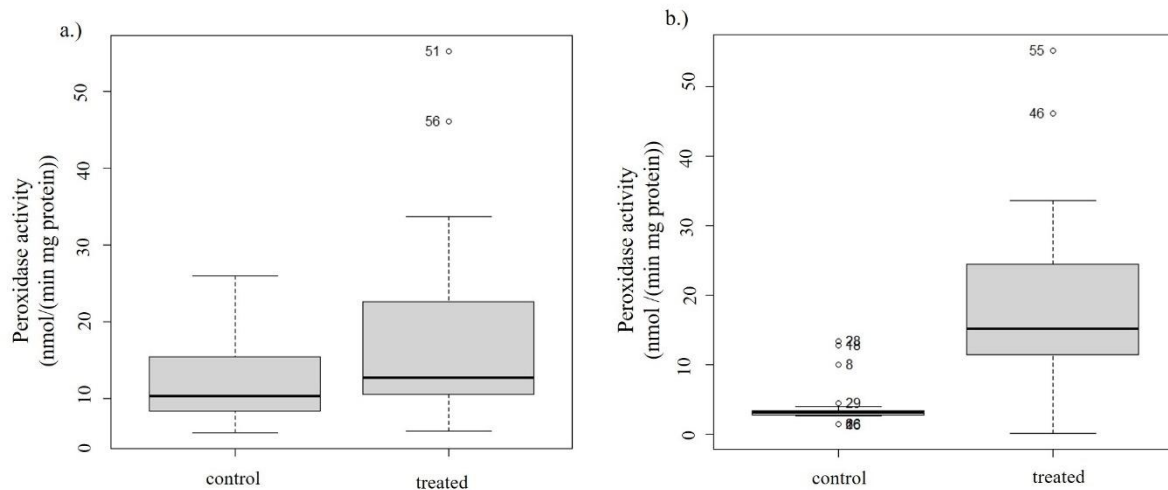
As shown in Table 4 was observed in chlorophyll-b levels in lettuce (Df = 1, F = 3.609, p = 0.0701). White mustard, however, did not exhibit any significant response. A reduction in chlorophyll content at polluted sites may result from inhibited chlorophyll biosynthesis, damage to chloroplasts, or increased chlorophyll degradation (Paull et al. 2021). These effects subsequently lead to decreased photosynthetic rates and overall productivity (Kaur & Nagpal 2017).

Reduction in photosynthetic pigment content is often regarded as a key indicator of air pollution effects. Total chlorophyll content has frequently been used to assess plant tolerance to air pollution across numerous studies (Karmakar & Padhy 2019; Zhang et al. 2016; Zhang et al. 2020; Bharti et al. 2018). However, different photosynthetic pigments can exhibit varying, species-specific sensitivities without a consistent pattern. For example, Giri et al. evaluated the impact of air pollution on *Azadirachta indica*, *Nerium oleander*, *Mangifera indica*, and *Dalbergia sissoo*, finding that in *N. oleander*, chlorophyll a (Chl-a) decreased more than chlorophyll b (Chl-b), whereas in *M. indica*, the opposite trend was observed (Giri et al. 2013). A similar study involving *Azadirachta indica*, *Nerium oleander*, *Mangifera indica*, and *Ficus bengalensis* found that chlorophyll a (Chl-a) experienced greater reductions than chlorophyll b (Chl-b) at polluted sites for *A. indica*, *N. oleander*, and *F. bengalensis*. However, *M. indica* again exhibited a different response, consistent with previous findings. Interestingly, carotenoid reductions were comparable across all species. Additionally, Joshi and Swami (Joshi & Swami 2009) reported species-specific differences in the sensitivity of Chl-a and Chl-b when examining the effects of traffic-related air pollution on six tree species.

In our study, white mustard did not exhibit significant changes in any photosynthetic pigments. In contrast, Joshi et al. observed alterations in all pigments when white mustard samples were collected from clean and urban/industrial sites in India. However, their study involved exposure over several months, which is considerably longer than the 21-day period used in our experiment (Joshi et al. 2009).

## Peroxidase activity

Peroxidase activity increased significantly in both test plants (Figure 6.). The differences between control and treated groups were statistically significant, with lettuce showing  $Df = 1$ ,  $F = 6.76$ ,  $p = 0.0112$ , and mustard showing  $Df = 1$ ,  $F = 49.99$ ,  $p = 1.63 \times 10^{-9}$ .



**Figure 6.** Changes in peroxidase activity in (a) lettuce and (b) mustard

In our study, increased peroxidase (POD) activity was observed. Changes in POD activity are widely recognized as a sensitive and general indicator of plant stress (Joshi et al. 2009). Previous research has utilized this biomarker to assess the effects of air pollution on plants (Keller 1974). Both atmospheric PAHs and heavy metals have been shown to induce oxidative stress by promoting the overproduction of reactive oxygen species (ROS), which in turn activates antioxidant enzymes like peroxidases (Molina Delgado & Segura 2021).

Previous studies have identified peroxidase (POD) activity as the most sensitive endpoint in plant stress experiments evaluating the phytotoxic effects of both heavy metals (Jaskulak et al. 2018) and PAH exposure (Liu et al. 2009). Our findings align with these observations; as increased POD activity has also been reported as a common response to toxic compounds associated with particulate matter (PM) (Rai 2016).

### 3. 2. 3. Bioaccumulation and bioconcentration of PAHs

All PAHs detected in the extract (Table 4) were found to accumulate in the leaves of both mustard and lettuce plants. Table 7 presents their concentrations in the PM extract and in plant tissues, along with the calculated BCF.

**Table 4.** Concentration of PAHs in the test plants and calculated BCF values.

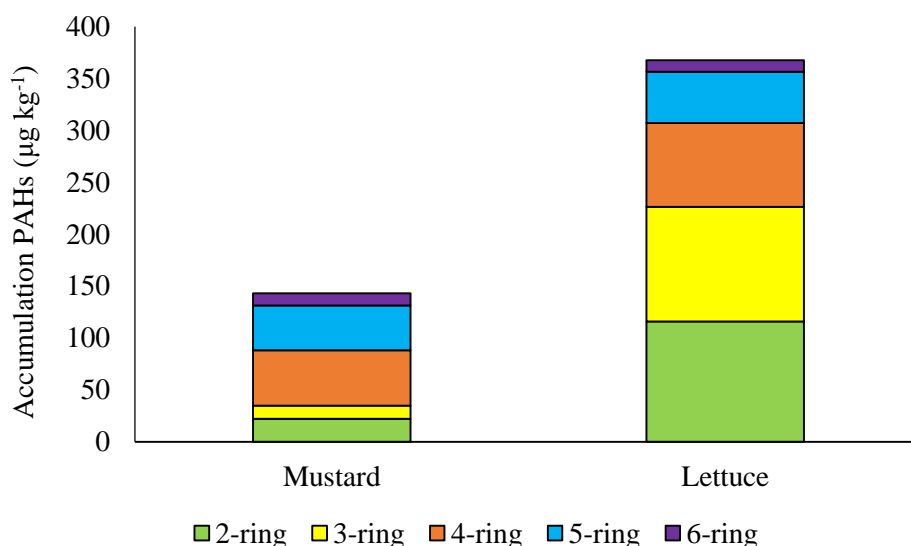
PAHs	Treated Mustard ( $\mu\text{g L}^{-1}$ )	BCF	Treated Lettuce ( $\mu\text{g L}^{-1}$ )	BCF
Naphthalene	19.5	20.9	102	109.6
2-methyl-naphthalene	1.4	3.7	8.7	23.1
1-methyl-naphthalene	1.1	3.4	5.3	16.4
Acenaphthylene	2.2	0.3	3.58	0.5
Acenaphthene	0.8	0.2	2.22	0.5
Fluorene	2.5	0.4	7.8	1.3
Phenanthrene	4.5	0.1	88	1.9
Anthracene	2.8	0.3	8.8	1.0
Fluoranthene	14.3	0.7	37.5	1.7
Pyrene	10.2	0.5	25.2	1.3
Benzanthracene	6.3	4.1	15.8	10.2
Chrysene	22.4	7.7	2.14	0.7
Benzo(b)fluoranthene	16.9	16.4	17.8	17.3
Benzo(k)fluoranthene	4.6	12.1	7.7	20.3
Benzo(e)pyrene	8.9	32.8	8	29.5
Benzo(a)pyrene	4.1	6.8	8	13.3
Dibenzo[a,h]anthracene	8.7	23.0	7.85	20.7
Indeno1.2.3CD-Pyrene	2.6	24.1	2.6	24.1
Benzo(g,h,i)perylene	9.3	19.5	8.5	17.8
<b>Total PAHs</b>	<b>143.1</b>		<b>367</b>	

Between the two test plants, lettuce demonstrated a higher accumulation capacity, with total PAH concentrations reaching  $367 \mu\text{g kg}^{-1}$ , compared to  $141.3 \mu\text{g kg}^{-1}$  in mustard. Lettuce's large leaf surface and thin cuticle make it particularly susceptible to the foliar deposition of contaminants. Its exceptional bioaccumulation potential has been well documented (Schreck et al. 2012; Schreck et al. 2013)

In a laboratory-scale accumulation study, lettuce plants treated with diesel exhaust  $\text{PM}_{2.5}$  extract accumulated all 18 PAHs present in the sample (Teke et al. 2020). Field studies have also shown that lettuce typically accumulates a broad spectrum of PAHs (Kováts et al. 2021).

Although mustard is a commonly used test species, limited information is available regarding its bioaccumulation potential. Other members of the Brassicaceae family have been referenced in the literature, but comparative studies generally report lower accumulation capacities than lettuce. For instance, Chen et al. measured concentrations of 16 PAHs in home gardens impacted by straw burning, finding levels ranging from  $208.7$  to  $269.5 \mu\text{g kg}^{-1}$  in lettuce, while lower concentrations were recorded in cabbage, ranging from  $114.5$  to  $175.1 \mu\text{g kg}^{-1}$ . Chinese cabbage showed PAH accumulation levels comparable to those of lettuce, ranging from  $200.8$  to  $295.1 \mu\text{g kg}^{-1}$ . In a study by Jia et al., various vegetables collected near industrial areas of Shanghai revealed the highest PAH concentrations in romaine (up to  $458.0 \mu\text{g kg}^{-1}$ ), followed by Chinese cabbage (up to  $348.1 \mu\text{g kg}^{-1}$ ), lettuce (up to  $319 \mu\text{g kg}^{-1}$ ), and Shanghai green cabbage ( $284 \mu\text{g kg}^{-1}$ ). In a comparative study by Kováts et al. (2022) rocket (*Eruca sativa* Mill.) exhibited lower PAH accumulation than lettuce when foliar uptake from an aqueous diesel aerosol extract was tested across different plant species.

Significant differences were observed between the two plant species in both total PAH accumulation and accumulation patterns (Figure 7). For low molecular weight (LMW) PAHs, mustard accumulated  $22 \mu\text{g kg}^{-1}$  of 2-ring PAHs,  $12.8 \mu\text{g kg}^{-1}$  of 3-ring PAHs, and  $53.2 \mu\text{g kg}^{-1}$  of 4-ring PAHs. In contrast, lettuce accumulated substantially higher amounts:  $116 \mu\text{g kg}^{-1}$ ,  $110.4 \mu\text{g kg}^{-1}$ , and  $80.4 \mu\text{g kg}^{-1}$ , respectively. However, when comparing the accumulation of high molecular weight (HMW) PAHs, the values were quite similar between the two species. Mustard accumulated  $43.2 \mu\text{g kg}^{-1}$  of 5-ring PAHs, while lettuce accumulated  $49.35 \mu\text{g kg}^{-1}$ . For 6-ring PAHs, mustard accumulated  $11.9 \mu\text{g kg}^{-1}$  and lettuce  $11.1 \mu\text{g kg}^{-1}$ .



**Figure 7.** Accumulation pattern of different molecular weight PAH groups in the test plants.

Most studies agree that bioaccumulation is generally more efficient for low molecular weight (LMW) PAHs. However, different plant species exhibit varying bioaccumulation capacities. For instance, Teke et al. reported a strong correlation between PAH BCFs and molecular weights in lettuce (Teke et al. 2020), whereas mustard showed higher bioconcentration for high molecular weight (HMW) PAHs (Hubai et al. 2024). In a study by Chen et al. assessing the impact of biomass burning in home gardens, the proportion of LMW to HMW PAHs was 65.5% to 34.5% in lettuce, 80% to 20% in cabbage, and 66.8% to 33.2% in Chinese cabbage (Chen et al. 2018).

The accumulation of PAHs in vegetables may also present risks to human health, particularly since dietary intake is considered the primary route of exposure compared to inhalation (Bansal & Kim 2015). The International Agency for Research on Cancer has evaluated the potential carcinogenicity of 16 polycyclic aromatic hydrocarbons commonly found in food, as reviewed by Sampaio et al. (2021). According to this classification, benzo(a)pyrene is carcinogenic to humans; benzanthracene, benzo(b)fluoranthene, benzo(k)fluoranthene, and dibenzo[a,h]anthracene is probably carcinogenic; and chrysene, along with the six-ring compounds indeno[1,2,3-cd]pyrene and benzo[ghi]perylene, are possibly carcinogenic to humans. These compounds were detected in measurable amounts in the tested plants (Table 5). For example, the concentration of the probably carcinogenic benzo(b)fluoranthene was  $16.9 \mu\text{g kg}^{-1}$  in mustard and  $17.8 \mu\text{g kg}^{-1}$  in lettuce.

### **3. 2. 4. Limitations of the study**

While open burning of domestic waste is a global issue, the composition of the waste varies by region. The model waste used in our burning experiment was based on Hungarian statistical data, where plastics comprised the largest proportion (15%), followed by textiles (13%) and treated wood (12%). Similar dominance of plastics has been reported in studies from South Africa (Wang et al. 2013) and Nepal (Das et al. 2018). In contrast, a study from the United States also included glass (e.g., bottles) and metals (e.g., cans) to simulate so-called barrel burning (Gullett et al. 2001). The composition of the waste burned significantly affects the nature and quality of the emissions produced. Our study focused on compounds with well-established phytotoxic effects, such as PAHs and heavy metals. However, other research highlights substances that pose potential risks to human health, including polychlorinated dibenzo-p-dioxins and dibenzofurans (Sidhu et al. 2005) as well as endocrine disruptors (Song et al. 2019). A major limitation of our study is the current lack of data on the scale of this issue-specifically, how many households in Hungary engage in this illegal practice-which represents a significant knowledge gap for both scientists and policymakers.

## 4. Phytotoxicity testing of emissions from illegal waste burning based on the OECD Terrestrial Plant Test

### 4. 1. Materials and methods

#### 4. 1. 1. Sampling and sample preparation

Model waste was selected as described in 3.1.1. Similarly, experimental burning and sample preparation procedures followed the same steps.

#### 4. 1. 2. Plant material used

Great blanketflower

- Latin name: *Gaillardia aristata* Pursh.
- Taxonomy: Asteraceae
- Life cycle: Short-lived perennial
- Raunkiaer classification: Hemicryptophyte
- Morphology: It has large achenes and long hairy pappus. The height is around 65 cm.

The great blanketflower originates from North America and is planted worldwide (Süle et al. 2023). It is also cultivated for ornamental purposes, and there are reported more than 30 species. *Gaillardia* has antioxidant properties, respectively, anti-inflammatory, antibiotic, and anticancer (Buleandra et al. 2015).

Pot marigold

- Latin name: *Calendula officinalis* L.
- Taxonomy: Asteraceae
- Life cycle: annual
- Raunkiaer classification: Therophyte (Botta-Dukát et al. 2023)
- Morphology: The stems are hairy and reaching 30-60 cm height. The flowers are bright orange or yellow, 4-7 cm in diameter, and leaves are simple, alternate, lanceolate to oblong, with slightly toothed or entire margins and glandular hairs.

*Calendula officinalis* L. (Asteraceae) is commonly cultivated in Europe and North Africa. This plant is one of the annual ornamental plants, but it is significantly used for pharmaceutical and cosmetic purposes (Szakiel et al. 2005). The plant contains polyphenols (e.g., p-hydroxybenzoic, salicylic, vanillic, caffeic, gallic acids), flavonoids (acylated flavonoid-O-

glycosides, methoxylated flavonoids), amino acids, alkaloids, carotenoids, saponins, tannins, high molecular weight polysaccharides, and triterpenoid monoesters (Rigane G et al. 2013).

### Hollyhock

- Latin name: *Alcea rosea L.*
- Taxonomy: Malvaceae
- Life cycle: annual or perennial
- Raunkiaer classification: Hemicryptophyte (Botta-Dukát et al. 2023)
- Morphology: Hollyhock has large leaves, colored flowers and stems that are fibrous.

Hollyhock originates from the Sichuan province in southwestern China. This plant is used in many industries, clothing industry, food products, and papermaking. In the agriculture industry, it has ability to tolerate and accumulate heavy metals, also it has been used for phytoremediation of cadmium-contaminated soils (Wang et al. 2013). Hollyhock also has medical benefits and is widely used traditional medicine in the middle East countries (Sadeghi et al. 2024).

### Tall morning-glory

- Latin name: *Ipomoea purpurea (L.) Roth*
- Taxonomy: Convolvulaceae
- Life cycle: annual and perennial
- Raunkiaer classification: Therophyte  
(<https://floraveg.eu/taxon/overview/Ipomoea%20purpurea>)
- Morphology: It has large and brightly coloured flowers.

Morning glory originates from the United States of America, belongs to the Convolvulaceae family. It has been used for medicinal and nutritional purposes, and it grows across tropical and subtropical countries (Anbazhakan et al. 2025).

### Four o'clock

- Latin name: *Mirabilis jalapa L.*
- Taxonomy: Nyctaginaceae
- Life cycle: perennial

- Raunkiaer classification: Hemicryptophyte, Geophyte  
(<https://floraveg.eu/taxon/overview/Mirabilis%20jalapa>)
- Morphology: The leaves are pointed, flowers in group of three, flowers with five green bracelets, usually yellow. Height is usually 0.6 to 0.9 m tall and wide. The fruit is a coriaceous obovoid, root shaped tuber.

*Mirabilis jalapa* L., known as a four o'clock (Nyctaginaceae), is a large and herbaceous plant grown in throughout in India and Pakistan. This plant has a health beneficial activity including dysentery, diarrhea, muscle pain and abdominal colic (Rozina 2016). This plant showed that is rich in many active compounds including triterpenes, proteins, flavonoids, alkaloids, and steroids. Therefore, it has been reported comparison with terpenoid and flavonoids fractions from other parts of the plant, better antioxidant capacity and antimicrobial activity of these fractions from the tuberous root of *Mirabilis jalapa* L. (Gogoi et al. 2016)

#### Ornamental gourd

- Latin name: *Cucurbita pepo* L.
- Taxonomy: Cucurbitaceae
- Life cycle: annual
- Raunkiaer classification: Therophyte  
(<https://floraveg.eu/taxon/overview/Cucurbita%20pepo>)
- Morphology: This plant is creeping or climbing plants with 5 angled stems up to 15 m long. The branches are 6-24 cm long, the leaves are simple, 20-30 cm long.

### 4. 1. 3. Experimental procedures

Treatment of test plants, preservation of the samples, phytotoxicity testing and chemical analyses were performed as described in 3.1.3 until 3.1.9.

## 4. 2. Results and discussion

### 4. 2. 1. Composition of the PM extract

According to the US EPA, 16 priorit PAHs were measured in the testing PM extract. (Table 5). These are designated by the US Environmental Protection Agency (EPA) as creating the highest environmental risk (reviewed by Keith 2015).

**Table 5.** Concentration of PAHs in the PM<sub>10</sub> extract

PAHs	Molecular weight [g mol <sup>-1</sup> ]	Waste PM <sub>10</sub> extract (µg L <sup>-1</sup> )
Naphthalene	128.17	0.477
Acenaphthylene	152.19	0.740
Acenaphthene	154.21	1.19
Fluorene	166.22	0.515
Phenanthrene	178.23	9.66
Anthracene	178.23	1.33
Fluoranthene	202.25	12.0
Pyrene	202.25	9.4
Benzanthracene	228.29	0.772
Chrysene	228.30	0.734
Benzo(b)fluoranthene	252.31	0.270
Benzo(k)fluoranthene	252.31	0.082
Benzo(e)pyrene	252.32	0.098
Benzo(a)pyrene	252.32	0.079
Dibenzo[a,h]anthracene	278.35	0.026
Indeno1.2.3CD-Pyrene	276.33	0.096
Benzo(g,h,i)perylene	276.30	0.062
<b>Total PAHs</b>		<b>37.5</b>

In the extract, the prevailing PAHs were fluoranthene (12.0 µg L<sup>-1</sup>), phenanthrene (9.66 µg L<sup>-1</sup>), and pyrene (9.4 µg L<sup>-1</sup>). These PAHs were measured in very high concentrations during experimental combustion of the PS sample and were also typical in PU samples. (Kováts et al 2022). Kwarteng et al. (2022) reported that PM<sub>2.5</sub>-associated PAHs at a waste recycling site in Ghana had high concentrations of all these compounds. The phytotoxic of fluoranthene, phenanthrene, and pyrene has been confirmed by numerous studies. (e.g., Ahammed et al. 2012a, b; Desalme et al. 2011).

**Table 6.** Concentration of heavy metals in the PM<sub>10</sub> extract

Heavy metal	Waste PM <sub>10</sub> extract ( $\mu\text{g L}^{-1}$ )
Zn	$104 \mu\text{g L}^{-1}$
Cu	$31.3 \mu\text{g L}^{-1}$
Cd	$15.3 \mu\text{g L}^{-1}$
Sb	$4.67 \mu\text{g L}^{-1}$
Na	$2.71 \mu\text{g L}^{-1}$
Mo	$<2.00 \mu\text{g L}^{-1}$
Ni	$<2.00 \mu\text{g L}^{-1}$
Al	$<10.0 \mu\text{g L}^{-1}$
B	$<10.0 \mu\text{g L}^{-1}$
Ba	$<10.0 \mu\text{g L}^{-1}$
As	$<1.000 \mu\text{g L}^{-1}$
Ag	$<1.000 \mu\text{g L}^{-1}$
Co	$<1.000 \mu\text{g L}^{-1}$
Cr	$<1.000 \mu\text{g L}^{-1}$
Pb	$<1.000 \mu\text{g L}^{-1}$
Se	$<1.000 \mu\text{g L}^{-1}$
Sn	$<1.000 \mu\text{g L}^{-1}$
Hg	$<0.200 \mu\text{g L}^{-1}$

In the extract (Table 6) Zn was determined highest concentration ( $104 \mu\text{g L}^{-1}$ ), followed by Cu ( $31.3 \mu\text{g L}^{-1}$ ), Cd ( $15.3 \mu\text{g L}^{-1}$ ) and Sb ( $4.67 \mu\text{g L}^{-1}$ ). "Zinc (Zn) is an essential metal that supports various biological functions in plants, but it becomes toxic at higher concentrations. (Long et al. 2003). Zinc (Zn), which is commonly detected in particulate matter (PM) during open waste burning, was identified as the predominant metal in emissions from the Okhla landfill site in India. (Kumar et al. 2018).

In a study conducted in China, Wang et al. (2017) also reported zinc (Zn) as the dominant metal in emissions from open waste burning (OWB), with an estimated total emission of 1790.70 tonnes in 2013.

High concentrations of zinc (Zn) were detected in samples obtained from the experimental burning of waste tyres, while cadmium (Cd) was also present but at significantly lower levels. (Shakya et al. 2006). Controlled combustion of waste samples with different

compositions revealed that zinc (Zn) and copper (Cu) were the predominant heavy metals in the resulting PM<sub>2.5</sub> emissions. (Cheng et al. 2020). Ramadan et al. (2025) reported that zinc (Zn), copper (Cu), and cadmium (Cd) were the most concentrated heavy metals in PM samples generated from plastic burning, while Kováts et al. (2022) evaluated the toxicity of PM

p  
r  
o  
d  
u  
c  
e  
d  
c  
o  
m  
m  
o  
n  
l  
y  
u  
s  
e  
d  
a  
s  
t  
r  
a  
c  
e  
r  
s  
f  
o  
r  
a  
e  
r  
o  
s  
o  
l  
s  
o  
r  
i  
g  
i  
n  
a  
t  
i  
n  
g  
f  
r  
o  
m  
p  
l  
a  
s  
t  
i  
c  
-  
e  
n  
r  
i  
c  
h  
e  
d  
w  
a  
s  
t  
e  
b  
u  
r  
n  
i  
n  
g  
.K

Kumar et al. (2015) identified arsenic (As), cadmium (Cd), antimony (Sb), and tin (Sn) as tracers for aerosols originating from plastic-enriched waste burning. Kováts et al. (2023) observed high Sb concentrations during the controlled combustion of PET bottles. In their study, both polycyclic aromatic hydrocarbons (PAHs) and heavy metals in PM samples were analyzed, with Sb levels ranging from 6.9 to 16.9 mg kg<sup>-1</sup>. The presence of antimony trioxide (Sb<sub>2</sub>O<sub>3</sub>), commonly used as a catalyst in the manufacture of PET, may account for these concentrations. Chapa-Martínez et al. (2016) reported residual Sb contents in PET reaching up to 300 mg kg<sup>-1</sup>, suggesting that the PET content in the burned waste was likely the main source of elevated Sb levels. Given phytotoxicity of antimony, Zhu et al. (2022) further examined its biological effects and reported its antioxidative capacity in plants.

#### 4. 2. 2. Phytotoxicity

Phytotoxic effects, including biomass reduction, concentrations of photosynthetic pigments, and peroxidase activity, are summarized in Table 7. Visible symptoms are also presented in the table; however, all endpoints-except for the visible symptoms—are evaluated relative to the control group.

**Table 7.** Summary of recorded effects for each endpoint and species tested: '+' indicates a significant effect, while '0' denotes a non-significant effect

		<b>Biomass</b>	<b>Chl-a</b>	<b>Chl-b</b>	<b>Car</b>	<b>POD</b>	<b>Visible symptoms</b>
1	<i>G. aristata</i>	+	-	-	-	+	
1	<i>C. officinalis</i>	-	-	-	-	+	Day8: 20% mortality Necrotic spots
e	<i>A. rosea L.,</i>	+	-	-	-	-	Day 21: Necrotic spots
d	<i>I. purpurea</i>	-	-	-	-	+	Day 21: Necrotic spots
	<i>M. jalapa</i>	+	-	+	+	-	
	<i>C. pepo L.,</i>	+	-	-	-	-	

c  
o  
m  
b  
u  
s

### 4. 2. 3. Individual sensitivity

The only *Mirabilis* (*M. jalapa*) exhibited statistically significant responses in three endpoints: biomass and two photosynthetic pigments (chlorophyll b and carotenoid). Four other species showed significant effects in two endpoints each: *Gaillardia* (*G. aristata*) in biomass and peroxidase (POD) activity; *Calendula* (*C. officinalis*) in POD activity and visible symptoms; *Alcea* (*A. rosea* L.) in biomass and visible symptoms; and *Ipomoea* (*I. purpurea*) in POD activity and visible symptoms. Lastly, *Cucurbita* (*C. pepo* L.) displayed a significant phytotoxic effect only in biomass reduction.

Considering the number of positive responses—defined as effects significantly different from the control—*Mirabilis jalapa* demonstrated the greatest sensitivity among all species tested. This ornamental species is relatively well-studied in the literature for its heavy metal accumulation potential (Petriccione et al. 2013) and has been suggested as a candidate for phytoremediation of cadmium (Cd) and tin (Sn)-contaminated soils through phytoextraction (Liu et al. 2018; Liu et al. 2021; Wang and Liu, 2014). Although the Cd concentration in the treatment extract was relatively high ( $15.3 \mu\text{g L}^{-1}$ ), no studies have specifically addressed its phytotoxic effects on *M. jalapa*. Additionally, *M. jalapa* has been evaluated for its potential in the phytostabilization of lead (Pb) (Wang et al. 2018). While this study demonstrated effective bioaccumulation, phytotoxic responses such as reduced biomass were also observed.

The bioremediation potential of *Mirabilis jalapa* has been demonstrated for petroleum-contaminated soils (Peng et al. 2009). However, Tabassum et al. (2016) reported notable stress responses in *M. jalapa* during a phytoremediation study involving used engine oil contamination, including biomass reduction, decreased chlorophyll content, and elevated superoxide dismutase activity. Despite these findings, the species has been infrequently used in direct bioindication studies. Bergweiler et al. (2008) highlighted its sensitivity to air pollutants such as ozone, which induced chlorosis in the plant.

*A. rosea* L., was tested for its phytoremediation potential in Cd-contaminated soils. Cd stress significantly decreased plant growth and both chlorophyll and carotenoid contents, oxidative stress was also experienced (Khan et al. 2022). In the study of Wu et al. (2018), considerable growth impairment was found in Cd-exposed *A. rosea* L., Liu et al. (2008a, 2008b), on the other hand, found that both *A. rosea* and *C. officinalis* were good hyperaccumulators of Cd and Pb, without showing growth inhibition. Kaya and Gülser (2018)

also report that it had more powerful Cd tolerance and accumulation capacity than other ornamental plants.

*Calendula officinalis* has been identified as a potent hyperaccumulator of cadmium (Cd), as demonstrated by Liu et al. (2022), who observed no growth impairment in the test plants. Similarly, Goswami and Das (2016) evaluated the copper (Cu) phytoremediation potential of *C. officinalis*, reporting good accumulation capacity without growth inhibition; however, significant declines in chlorophyll and carotenoid pigment contents, alongside increased lipid peroxidation, indicated oxidative stress. Phytotoxic responses of *Gaillardia aristata* have been seldom investigated, with most studies focusing on herbicide-induced toxicity (e.g., Henry et al., 2023). Liu et al. (2012) reported that *G. aristata* could be effectively utilized for the phytoremediation of petroleum-contaminated soils, although phytotoxic symptoms were not assessed in their study.

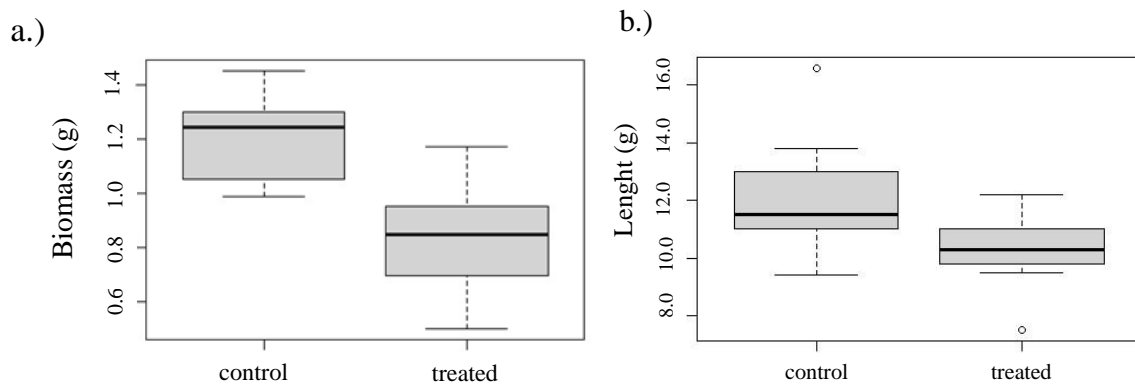
Kumar and Saritha (2014) investigated the potential of *Ipomoea purpurea* for phytoremediation in heavy metal-contaminated soils. While the species demonstrated metal accumulation capacity, phytotoxic responses such as reductions in chlorophyll a (chl-a) and chlorophyll b (chl-b), along with upregulation of antioxidative enzymes, were observed. Similarly, *Cucurbita pepo* L., has been reported to possess good heavy metal accumulation capacity (Tamez et al., 2019).

The aforementioned studies highlight the accumulation and phytoremediation potential of the ornamental plants examined in our study (reviewed by Khan et al., 2021), primarily focusing on the soil–root–plant exposure pathway. It is well-documented that soils surrounding waste disposal and open burning sites, particularly e-waste areas, can be heavily contaminated with metals (Arya et al. 2021; Liu et al. 2021; Kolawole et al. 2023). However, to our knowledge, this study directly assesses exposure to atmospheric pollution and uptake of contaminants via foliar absorption.

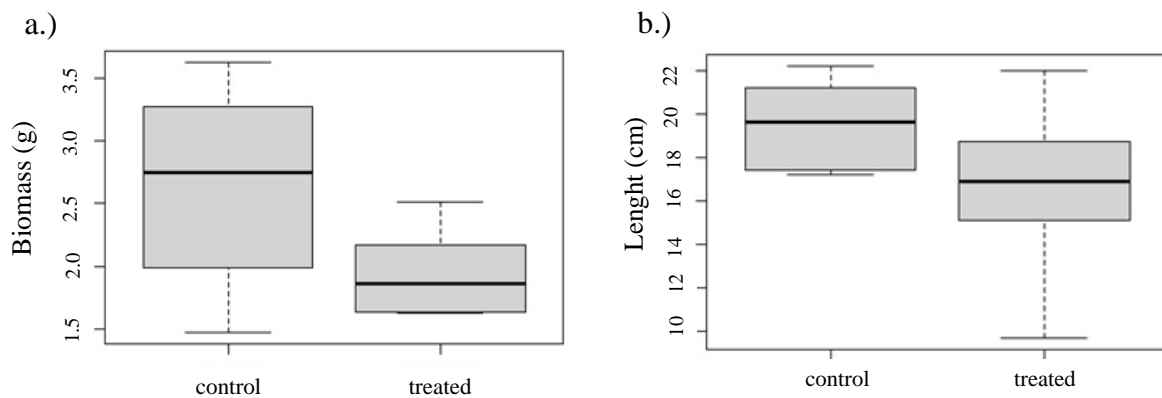
#### 4. 2. 4. Sensitivity of different end-points

##### Growth inhibition

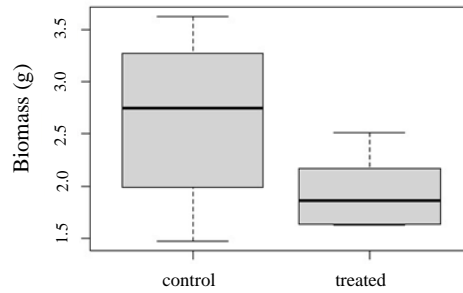
Growth inhibition was indicated by decreased biomass relative to the control and, where appropriate, by reduced shoot length. Significant responses were observed in *Gaillardia aristata*, *Alcea rosea L.*, *Mirabilis jalapa*, and *Cucurbita pepo L.* Statistically significant differences between the average biomass of control and treated groups were confirmed by ANOVA: *A. rosea L.*, ( $F = 20.966$ ,  $p = 0.00023$ ; Figure 8a), *M. jalapa* ( $F = 5.491$ ,  $p = 0.0315$ ; Figure 9a), *G. aristata* ( $F = 7.601$ ,  $p = 0.0130$ ; Figure 10), and *C. Pepo L* ( $F = 7.888$ ,  $p = 0.0116$ ; Figure 12). Additionally, significant differences were detected between treated and control groups in fresh leaf biomass and leaf length for *A. rosea L.*, ( $F = 5.599$ ,  $p = 0.029$ ; Figure 8b), *M. jalapa* ( $F = 4.895$ ,  $p = 0.041$ ; Figure 9b), and *I. purpurea* ( $F = 12.399$ ,  $p = 0.0023$ ; Figure 11).



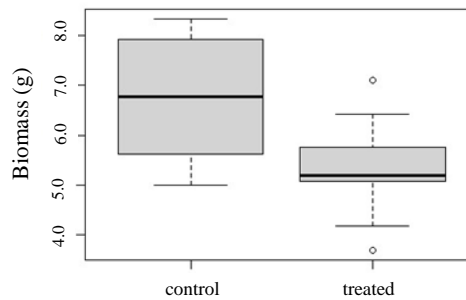
**Figure 8.** Average a.) biomass in comparison b.) Average length of leaves to the control *A. rosea L.*,



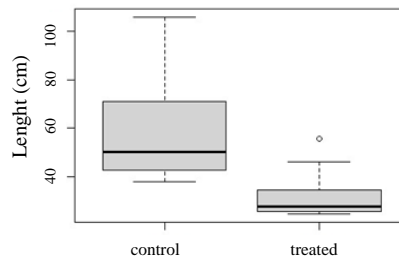
**Figure 9.** Average a.) biomass in comparison and b.) length of leaves in to the control *M. jalapa* comparison to the control



**Figure 10.** Average biomass in comparison to the control *G. aristata*



**Figure 11.** Average biomass in comparison to the control *C. Pepo L.*,



**Figure 12.** Average length of leaves in comparison to the control *I. purpurea*

Growth inhibition, assessed as reduced biomass and/or shoot length, is an endpoint originally recommended by the Guideline. Significant biomass reduction was observed in four species: *Gaillardia aristata*, *Alcea rosea L.*, *Mirabilis jalapa*, and *Cucurbita pepo L.*, while reduced shoot length was noted in *A. rosea L.*, and *M. jalapa*. A broad body of literature reports growth inhibition as a common response to exposure to selected polycyclic aromatic hydrocarbons (PAHs), including fluoranthene (e.g., Oguntimehin et al. 2008), phenanthrene (Ahammed et al. 2012), and anthracene (Wieczorek & Wieczorek, 2007), as well as to complex

environmental mixtures such as simulated diesel emissions (Honour et al. 2009; Bell et al. 2011).

Reduced growth is often considered an ultimate response to stressful environmental conditions (Ali et al. 2016), resulting from various underlying mechanisms. Desalme et al. (2011) reported biomass reduction in phenanthrene-treated white clover, attributing it to decreased net assimilation rate, lower carbon assimilation, and reduced chlorophyll content. In comparative studies, biomass was identified as the most sensitive endpoint when assessing phytotoxic effects of diesel emissions on common roadside plants (Kováts et al. 2021). Similarly, Storch-Böhm et al. (2020) evaluated diesel emission effects on *Guabiroba* (*Campomanesia xanthocarpa* O. Berg.) and found that endpoint sensitivities ranked as follows: peroxidase activity > biomass/chlorophyll content > protein levels > leaf injury.

### Visible symptoms

The occurrence of visible symptoms-such as chlorosis, necrosis, and discoloration-is another endpoint originally included in the Guideline. In *Calendula officinalis*, 20% mortality was observed on Day 8, followed by the development of necrotic spots. Necrosis was also recorded in *Alcea rosea* L., (Figure 13) and *Ipomoea purpurea* (Figure 15), although these symptoms appeared later, on Day 21.



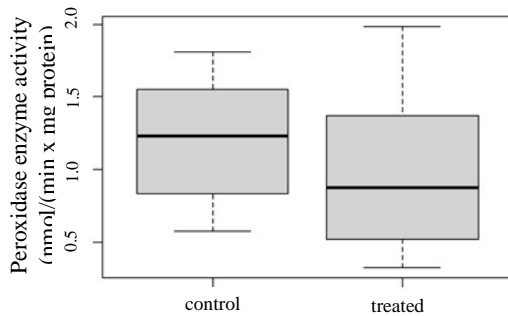
**Figure 13.** Typical necrotic spots on *A. rosea* L.,



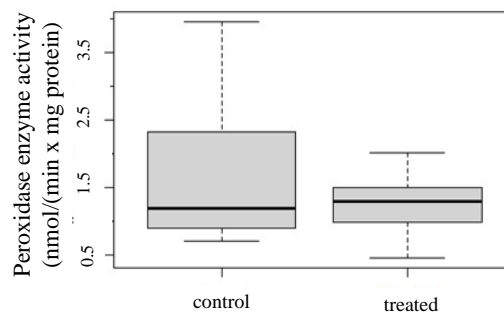
**Figure 14.** Typical necrotic spots on *I. purpurea*

## Peroxidase activity

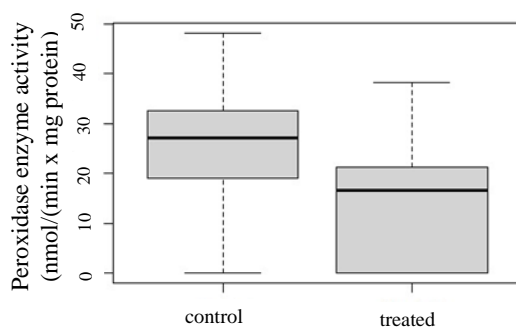
Peroxidase activity showed significant responses in three of the tested species. Statistically significant differences were observed in *Gaillardia aristata* (ANOVA  $F = 4.223$ ,  $p = 0.0444$ ), *Calendula officinalis* (ANOVA  $F = 4.474$ ,  $p = 0.0390$ ), and *Ipomoea purpurea* (ANOVA  $F = 11.194$ ,  $p = 0.0014$ ) (Figure 15, 16, and 17).



**Figure 15.** *G. aristata* POD level



**Figure 16.** *C. officinalis* POD level

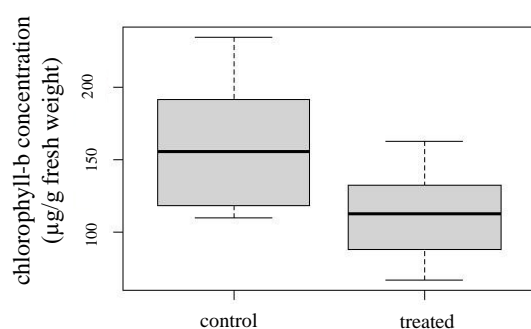


**Figure 17.** *I. purpurea* POD level

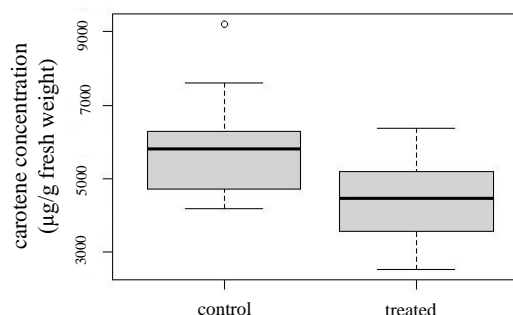
Depending on the composition of the waste, particulate matter (PM) generated during waste burning contains heavy metals and polycyclic aromatic hydrocarbons (PAHs), both of which can induce the formation of reactive oxygen species (ROS). The overproduction of ROS is believed to underlie the development of direct phytotoxic effects (Oguntimehin and Sakugawa, 2008). In the study by Vreeland et al. (2016), the redox activity of roadside trash burning emissions was assessed using the dithiothreitol (DTT) assay, revealing extremely elevated redox activity across all samples.

The level of oxidative stress can be evaluated by measuring peroxidase (POD) concentration, an enzyme primarily responsible for scavenging hydrogen peroxide (H<sub>2</sub>O<sub>2</sub>) (Bansal et al. 2016). Changes in POD activity respond to chemical stress induced by both heavy metals and PAHs. Junior et al. (2024) reported that heavy metals—including Cd, Zn, and Cu—commonly found in solid waste leachate, trigger antioxidative stress responses. Similarly, individual PAHs such as fluoranthene (Zezulka et al. 2013), fluorene (Arikan et al. 2023), phenanthrene (Ahammed et al. 2012a), and pyrene (Ahammed et al. 2012b), as well as their mixtures, have been shown to elevate POD levels (Krzyszczak et al. 2022). Moreover, POD proved a sensitive biomarker when extracts from particulate matter generated by combustion of various plastics—such as PVC, PUR, PP, PS, and PE—were examined (Hubai et al. 2022).

### Photosynthetic pigments



**Figure 18.** Chlorophyll-b ratio in *M. jalapa*



**Figure 19.** Carotene content in *M. jalapa*

The contents of chlorophyll-a, chlorophyll-b, and carotenoids were compared between control and treated groups for each species. Statistically significant decreases in chlorophyll b and carotenoid levels were observed only in *Mirabilis jalapa*, which, as noted previously, exhibited the highest sensitivity (ANOVA  $F = 6.635$ ,  $p = 0.0196$  for chlorophyll b; Figure 18; and ANOVA  $F = 5.610$ ,  $p = 0.0300$  for carotenoids; Figure 19). Photosynthetic pigments are critical for plant metabolism, and their reduction is often associated with inhibited plant growth (Rai et al. 2016). Pigment levels have long been used as biomarkers for monitoring air quality. For instance, Rabe and Kreeb (1979) reported decreased chlorophyll-a and b levels in plants exposed to ambient air pollution. Numerous studies have also documented reduced total chlorophyll content in plants growing in polluted environments, such as urban biotopes

(Nadgórska–Socha et al. 2017) and areas affected by heavy traffic emissions (Joshi and Swami 2009).

However, these habitats are exposed to a complex mixture of pollutants, including gaseous toxicants such as ozone, SO<sub>2</sub>, and NO<sub>x</sub> (Joshi et al. 2009). This implies that polycyclic aromatic hydrocarbons (PAHs) and heavy metals-commonly measured and evaluated as particulate matter (PM)-bound components-occur in varying concentrations in ambient air. Consequently, biomonitoring studies assess the combined effects of all pollutants present, in addition to these two groups. This complexity may partially explain the relatively lower sensitivity of photosynthetic pigments observed in the present study.

Notably, in the sole sensitive species, *Mirabilis jalapa*, chlorophyll-b and carotenoid levels showed significant changes while chlorophyll-a did not, a differential sensitivity that is supported by other studies. Borowiak et al. (2018) similarly reported greater sensitivity of chlorophyll-b in *Lolium multiflorum* exposed to ambient air. Additionally, higher sensitivity of carotenoids relative to chlorophylls was observed in roadside plants exposed to diesel emissions (Kováts et al. 2021). Comparative studies further confirm that different plant species exhibit variable degrees of reduction in photosynthetic pigment concentrations when exposed to polluted environments (Zhang et al. 2016; Uka et al. 2021).

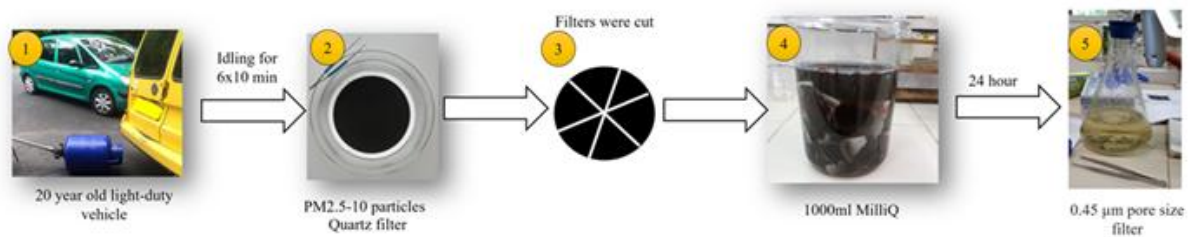
## 5. Can landraces better cope with environmental stress?

### 5. 1. Materials and methods

#### 5. 1. 1 Sampling and sample preparation

For phytotoxicity testing, aqueous extract of Diesel exhaust PM<sub>2.5-10</sub> was sampled as follows: a 20-year-old, light-duty vehicle (environmental standard: Euro3) was operated at idling for 6 x 10 minutes. Sampling was performed using a KÁLMÁN PM<sub>2.5</sub> sampler (flow rate 32 m<sup>3</sup> h<sup>-1</sup>). A closed premise was placed approximately 1 meter from the tailpipe, emitted particles were collected on quartz filters (Whatman QMA, diameter 150 mm).

Sample preparation was carried out as described above. (Figure 20)



**Figure 20.** Consecutive steps of sampling and sample preparation.

#### 5. 1. 2. Plant material used

Tomato

- Latin name: *Lycopersicon esculentum* Mill.
- Taxonomy: Solanaceae
- Life cycle: annual
- Raunkiaer classification: Therophyte  
(<https://floraveg.eu/taxon/overview/Lycopersicon%20esculentum>)

Tomato is a popular vegetable, the second most important vegetable crop after potatoes and has over 4000 varieties. Plants are easily grown in glasshouse and the fruit does not contain

t  
h  
e

**Table 8.** Morphology of 5 different tomato varieties/landraces

t  
o  
x  
i

	Roma	Mobil	LugasF1	Tápiószelei	Gulácsi
<b>Type of growth</b>	determinate	indeterminate	indeterminate	indeterminate	Indeterminate potato-leaf morphology, due to the potato leaf mutation (Kessler et al. 2001)
<b>Fruit form</b>	Cylindrical (egg or plum shaped)	slightly flattened	Cylindrical (egg or plum shaped)	slightly flattened	ribbed, slightly flattened
<b>Average mass</b>	60 g	130 g	65 g	46 g	237 g

### 5. 1. 3. Experimental procedures

Cultivation of the test plants was performed according to the OECD Guideline as described previously. In order to represent water stress conditions, two levels of water availability were selected based on the study of Lin et al. (2021): high water availability (200 mL day<sup>-1</sup>) (well-watered, WW) and low water availability (50 mL day<sup>-1</sup>) (reduced watering, RW). Test groups are given in detail in Table 9.

**Table 9.** Description of test groups. WW: Well-watered; RW: Reduced watering. PM-: no treatment; PM+: treatment with the PM extract

	TG1 (control)	TG2	TG3	TG4
Roma	WW/PM-	RW/PM-	WW/PM+	RW/PM+
Mobil	WW/PM-	RW/PM-	WW/PM+	RW/PM+
Lugas	WW/PM-	RW/PM-	WW/PM+	RW/PM+
Tápiószelei	WW/PM-	RW/PM-	WW/PM+	RW/PM+
Gulácsi	WW/PM-	RW/PM-	WW/PM+	RW/PM+

Preservation of the samples, chemical analyses and phytotoxicity testing were performed as described in 3.1.4, 3.1.5, and 3.1.7.

Phytotoxicity testing involved an additional end-point, measurement of the surface of stomata. In addition, surface of the stomata was also measured. Stomatal surface area was determined using 3-3 leaves of each test plant (10 plants per treatment group). Leaves were prepared according to Bruzzese and Hasan (1983). Photos were taken using a Nikon Eclipse

E600 microscope equipped with QImaging Retiga R3 camera. The width and length of stomata were measured by Ocular software, 50 stomata per treatment for each variety were used. The stomatal surface was calculated as follows:

$$S = \pi * \frac{a}{2} * \frac{b}{2}$$

where  $S$  is the surface of the stomata ( $\mu\text{m}^2$ ),  $a$  is the length ( $\mu\text{m}$ ) and  $b$  is the width ( $\mu\text{m}$ ).

## 5. 2. Results and discussion

### 5. 2. 1. Composition of the PM extract

Measured PAH concentrations are presented in Table 10, with PAHs grouped based on their number of rings.

**Table 10.** Concentration of PAHs in the aerosol extract

	PAHs	Concentration ( $\mu\text{g L}^{-1}$ )
two rings	Naphthalene	0.001
	2-methyl-naphthalene	0.394
	1-methyl-naphthalene	0.367
three rings	Acenaphthylene	0.12
	Acenaphthene	0.052
	Fluorene	0.472
	Phenanthrene	0.448
	Anthracene	0.058
four rings	Fluoranthene	0.074
	Pyrene	0.081
	Benzoanthracene	0.032
	Chrysene	0.013
five rings	Benzo(b)fluoranthene	0.064
	Benzo(k)fluoranthene	0.036
	Benzo(e)pyrene	0.055
	Benzo(a)pyrene	0.043
	Dibenzo(a,h)anthracene	0.045
six rings	Indeno(1,2,3CD)Pyrene	0.033
	Benzo(g,h,i)perylene	0.035
<b>Total PAHs</b>		<b>2.43</b>

## 5. 2. 2. Phytotoxicity Assessment

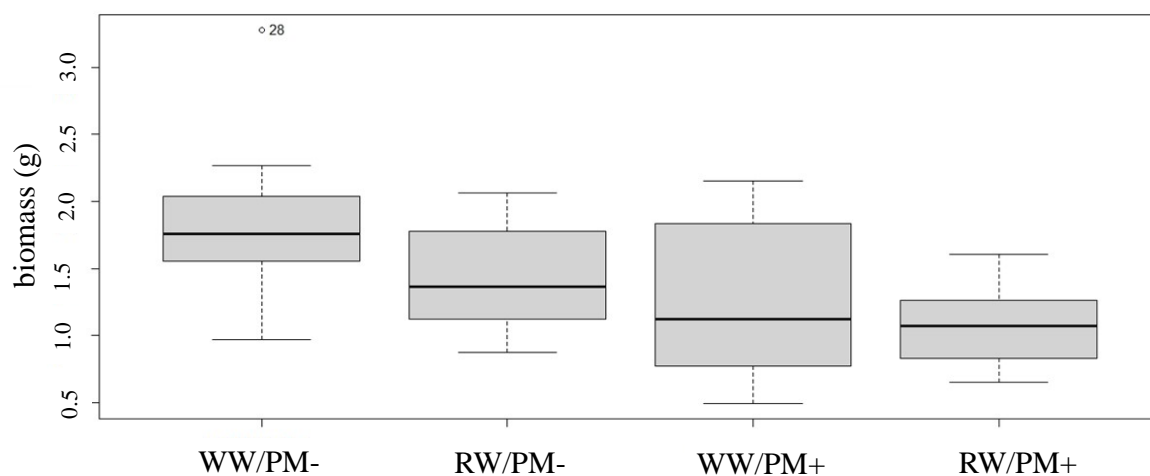
### Changes in biomass

Compared to the control group (adequate water supply without any treatment), three of the tested varieties - ‘Roma’, ‘Mobil’, and ‘Lugas’—showed significant differences in biomass (Figure 21, 22, 23). The Kruskal-Wallis test indicated significant effects for each: ‘Roma’ ( $\chi^2 = 8.85$ ,  $df = 3$ ,  $p = 0.0313$ ), ‘Mobil’ ( $\chi^2 = 15.76$ ,  $df = 3$ ,  $p = 0.0013$ ), and ‘Lugas’ ( $\chi^2 = 10.98$ ,  $df = 3$ ,  $p = 0.0118$ ). For the ‘Roma’ variety, post hoc Neményi tests revealed significant differences between the control and WW/PM+ treatment ( $p = 0.0409$ ), as well as between the control and RW/PM+ treatment ( $p = 0.0125$ ).

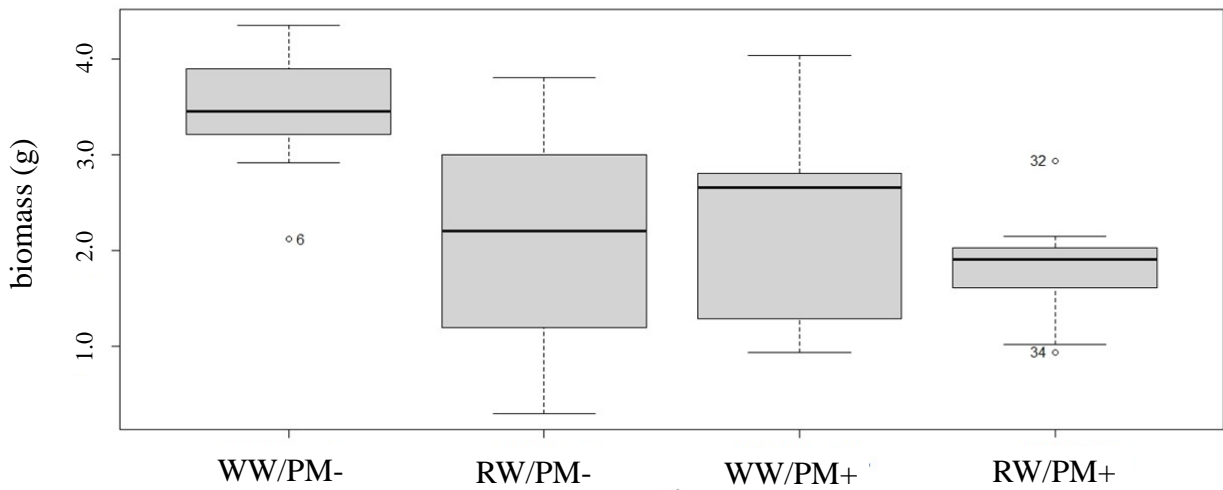
The ‘Mobil’ landrace exhibited a significant response to all treatments, with the Neményi test indicating statistically significant differences for RW/PM- ( $p = 0.0269$ ), WW/PM+ ( $p = 0.0483$ ), and RW/PM+ ( $p = 0.0008$ ).

The ‘Lugas’ variety showed a marginally significant difference between the control and the WW/PM+ treatment (Neményi test:  $p = 0.0941$ ), and a statistically significant response to the combined treatment of reduced water supply and PM (RW/PM+) (Neményi test:  $p = 0.0084$ ) (Figure 23).

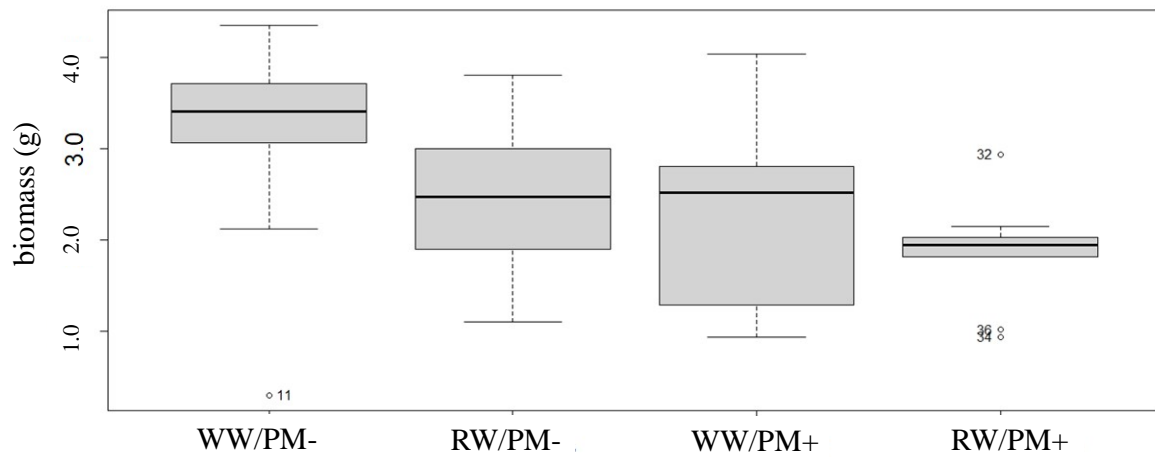
No statistically significant differences were observed for the ‘Gulácsi’ and ‘Tápószelei’ landraces.



**Figure 21.** Biomass changes in the ‘Roma’ variety relative to the control



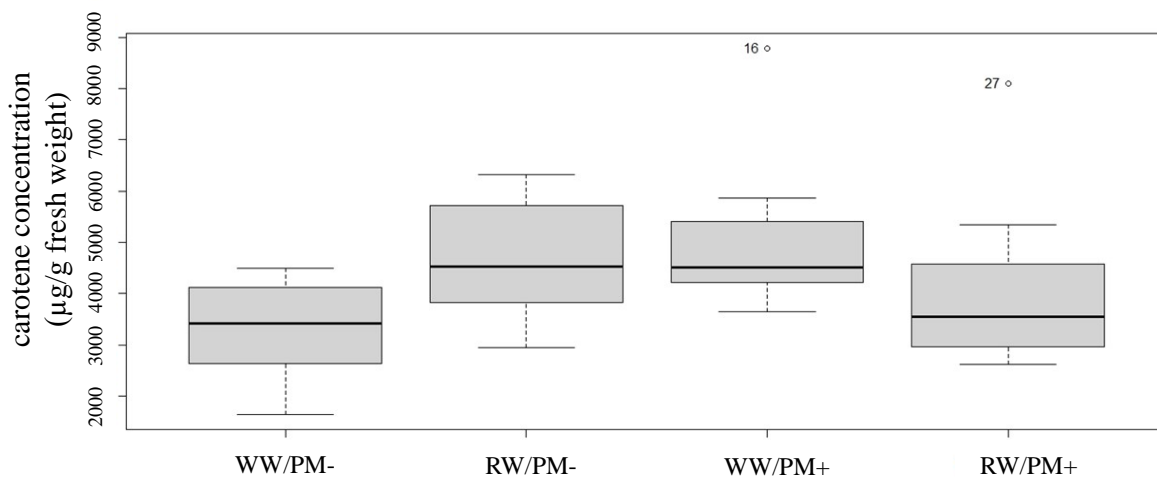
**Figure 22.** Biomass changes in the 'Mobil' variety relative to the control



**Figure 23.** Biomass changes in the 'Lugas' variety relative to the control

## Photosynthetic pigments

No significant differences were observed in chlorophyll-a content between treated plants and the control. Similarly, chlorophyll-b concentrations did not differ significantly between treated and control plants across all tested varieties. Carotene concentration showed a significant overall difference between the control and treated samples in the ‘Roma’ variety (Kruskal-Wallis  $\chi^2 = 8.20$ ,  $df = 3$ ,  $p = 0.0421$ ); however, post hoc tests (Tukey and Neményi) did not identify any pairwise comparisons as statistically significant (Figure 24). The other tested varieties exhibited no significant responses.

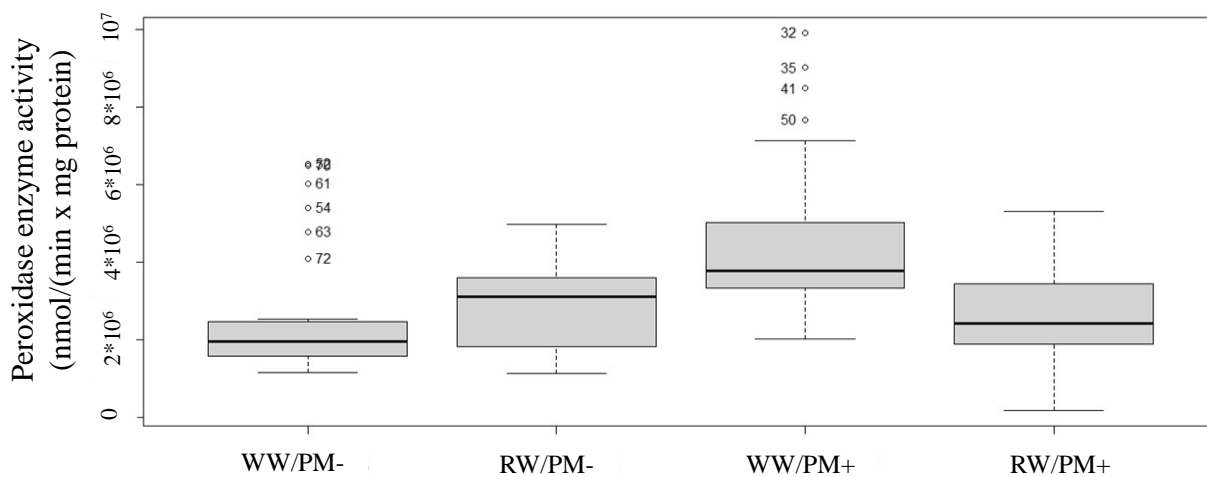


**Figure 24.** Changes in carotene content in the ‘Roma’ variety relative to the control

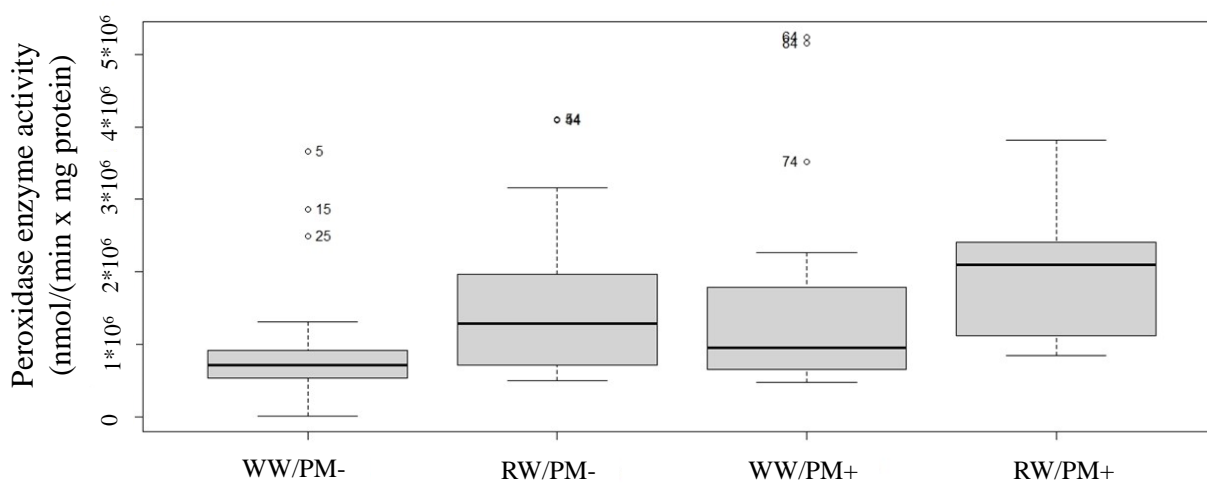
## Peroxidase activity

POD activity (nmol x (min x mg protein)) showed significant differences in the following varieties: ‘Roma’ (ANOVA:  $F = 11.09$ ,  $p = 2.14 \times 10^{-6}$ ), ‘Mobil’ (ANOVA:  $F = 5.43$ ,  $df = 3$ ,  $p = 0.00159$ ), ‘Tápiószelei’ (ANOVA:  $F = 5.92$ ,  $p = 0.000864$ ), and ‘Gulácsi’ (ANOVA:  $F = 10.47$ ,  $p = 5.16 \times 10^{-6}$ ). For the ‘Roma’ variety, POD activity differed significantly across all treatment groups compared to the control (Tukey HSD test: control vs. RW/PM-,  $p = 1.01 \times 10^{-8}$ ; control vs. RW/PM+,  $p = 0.0287$ ; control vs. WW/PM+,  $p = 1.0 \times 10^{-7}$ ) (Figure 25a). For the ‘Mobil’ variety, a significant response was observed in the RW/PM- treatment group ( $p = 0.0868$ ) and a highly significant response in the RW/PM+ group ( $p < 0.001$ ) (Figure 25b). In the ‘Tápiószelei’ variety, a significant difference was found in the RW/PM- group (Tukey HSD:  $p = 0.00041$ ) (Figure 25c). The ‘Gulácsi’ variety showed a significant increase in POD

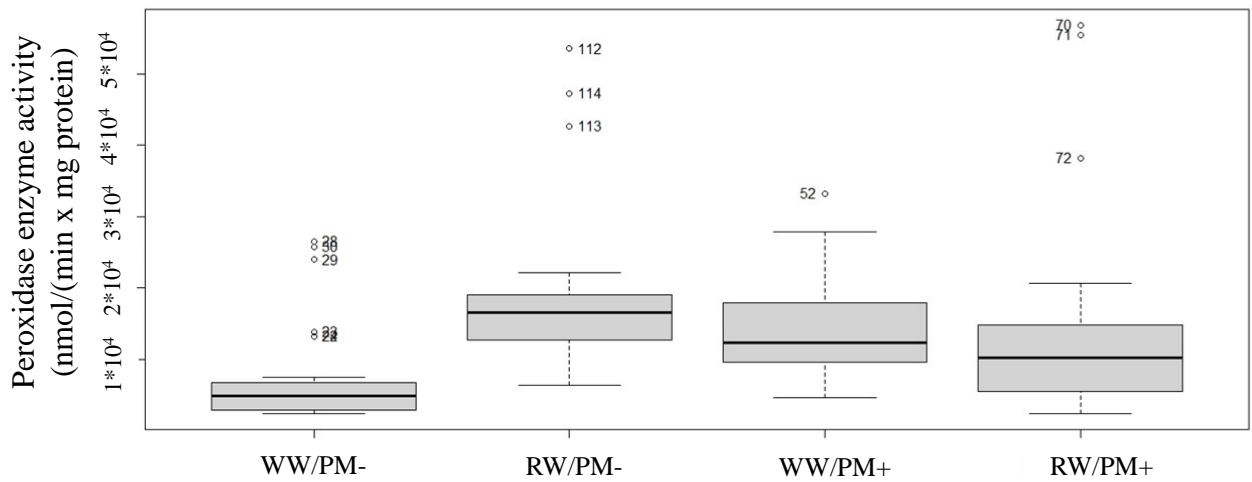
activity in both the RW/PM- (Tukey HSD:  $p = 0.00102$ ) and RW/PM+ (Tukey HSD:  $p = 0.0000031$ ) treatment groups (Figure 25d).



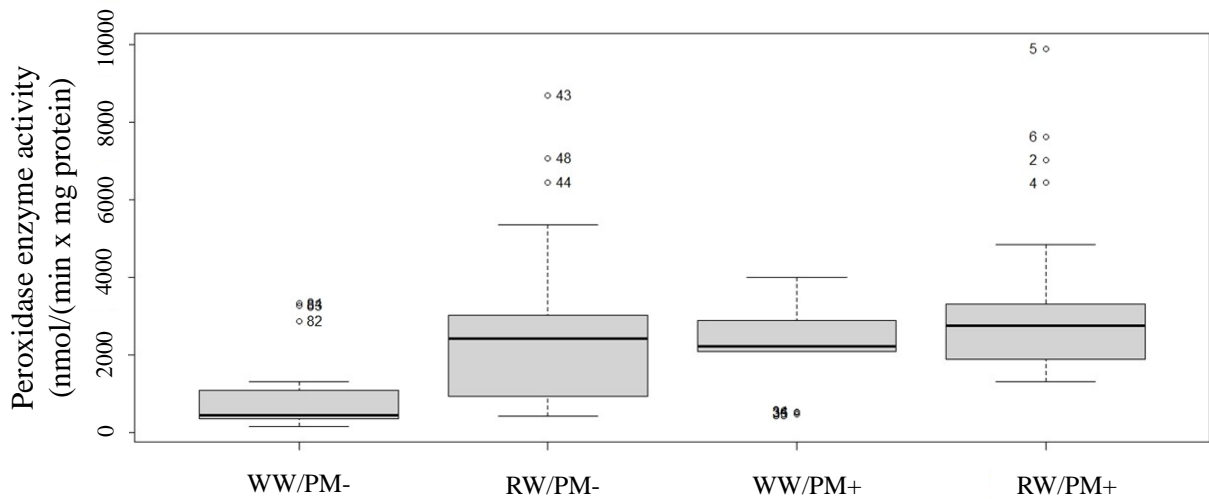
**Figure 25a.** POD activity (nmol x (min x mg protein) in the 'Roma' variety



**Figure 25b.** POD activity (nmol x (min x mg protein) in the 'Mobil' variety



**Figure 25c.** POD activity (nmol x (min x mg protein)) in ‘Tápiószelei’



**Figure 25d.** POD activity (nmol x (min x mg protein)) in ‘Gulácsi’

### Stomata

The surface area of stomata showed significant responses (Tukey HSD) across all tested varieties, with effects varying depending on the treatment and variety. In ‘Roma’, a significant increase was observed only under water stress conditions (WW/PM<sup>-</sup> vs. RW/PM<sup>-</sup>:  $p < 0.0001$ ; WW/PM<sup>-</sup> vs. WW/PM<sup>+</sup>:  $p = 0.972$ ; WW/PM<sup>-</sup> vs. RW/PM<sup>+</sup>:  $p < 0.0001$ ). In contrast, ‘Mobil’ exhibited significant increases in stomatal area across all treatment groups (WW/PM<sup>-</sup> vs. RW/PM<sup>-</sup>:  $p < 0.0001$ ; WW/PM<sup>-</sup> vs. WW/PM<sup>+</sup>:  $p = 0.0001$ ; WW/PM<sup>-</sup> vs. RW/PM<sup>+</sup>:  $p <$

0.0001). A similar pattern was observed in 'Tápiószelei' (WW/PM<sup>-</sup> vs. RW/PM<sup>-</sup>:  $p < 0.0001$ ; WW/PM<sup>-</sup> vs. WW/PM<sup>+</sup>:  $p < 0.0001$ ; WW/PM<sup>-</sup> vs. RW/PM<sup>+</sup>:  $p < 0.0001$ ).

For the 'Lugas' variety, only the reduced water supply combined with PM treatment showed a significant increase compared to the control, although this was not statistically significant (RW/PM<sup>+</sup>:  $p = 0.4344$ ). In contrast, the 'Gulácsi' variety exhibited the opposite trend, with a significant decrease observed in the treatment groups subjected to water scarcity (WW/PM<sup>-</sup> vs. RW/PM<sup>-</sup>:  $p < 0.0001$ ; WW/PM<sup>-</sup> vs. WW/PM<sup>+</sup>:  $p = 0.086$ ; WW/PM<sup>-</sup> vs. RW/PM<sup>+</sup>:  $p = 0.001$ ).

### 5. 2. 3. Discussion

The impact of drought on crop species is increasingly severe due to global climate change (Ma et al. 2022). Therefore, understanding the tolerance mechanisms of crop plants—such as tomato—and developing drought-tolerant varieties is of critical importance (Çelik et al. 2021).

The sensitivity of the five tested varieties is summarized in Table 4 based on the endpoints evaluated in this study. For biomass, the 'Roma' variety showed a significant decrease in both the WW/PM<sup>+</sup> and RW/PM<sup>+</sup> treatment groups. The 'Mobil' variety exhibited significant responses across all treatment groups. In the case of 'Lugas', a significant decrease was observed in the RW/PM<sup>+</sup> group, along with a marginally significant effect in the WW/PM<sup>+</sup> group.

Among the photosynthetic pigments, only the 'Roma' variety showed a significant change in carotene content when compared to the control, specifically in the WW/PM<sup>+</sup> and RW/PM<sup>-</sup> treatment groups. Although chlorophyll content is widely used as an indicator of pollution-induced damage (e.g., Singh et al. 2023), no significant changes were observed in any treatment group in this study.

The treatment applied in this study was designed to simulate the wet deposition of particulate matter (PM) and PM-bound potentially toxic compounds. PM can enter plant leaves directly through the stomata, carrying associated pollutants. In this way, plants absorb contaminants deposited on leaf surfaces via both dry and wet deposition processes (Shahid et al., 2021). This uptake is facilitated by the small size of PM, with the effect likely being most pronounced for fine particles such as PM<sub>2.5</sub> (particles with a diameter of 2.5  $\mu\text{m}$  or smaller) (Gao et al. 2021).

The total PAH content of the PM extract was  $2.43 \mu\text{g L}^{-1}$ , with the 3-ring compounds fluorene ( $0.472 \mu\text{g L}^{-1}$ ) and phenanthrene ( $0.448 \mu\text{g L}^{-1}$ ) being the most prevalent. Atmospheric phenanthrene has been shown to exert detectable phytotoxic effects on tomato, as reported by Ahammed et al. (2012b).

Stomatal surface area changes showed mixed trends across the tested varieties. In ‘Roma’, ‘Mobil’, ‘Lugas’, and ‘Tápiószelei’, treatment-specific increases were observed. In contrast, the ‘Gulácsi’ variety exhibited a statistically significant decrease in the RW/PM– and RW/PM+ treatment groups, suggesting a potential effect of water scarcity.

In response to air pollution, a reduction in stomatal pore surface area is often observed as a means to enhance stomatal closure efficiency. Wuytack et al. (2010) reported significantly smaller stomatal pores in white willow growing in urban environments compared to those in rural areas. Similarly, Kardel et al. (2010) found that plantain (*Plantago lanceolata*) exhibited reduced stomatal pore size, supporting its use as an indicator of urban air quality. In contrast, Kulshreshtha et al. (2009) reported an increase in stomatal area in roadside plants such as *Bougainvillea* species exposed to high levels of particulate matter pollution. Similarly, Şevik et al. (2017) observed larger stomata in *Pyracantha coccinea* leaves collected from high-traffic areas.

### **5. 2. 3. 1. Comparison of end-points**

Table 10 summarizes the responses of each variety and landrace. Storch-Böhm et al. (2022) evaluated the sensitivity of various endpoints in assessing the impact of air pollution on native Brazilian plants and established the following order of sensitivity: peroxidase > biomass  $\approx$  chlorophyll. In our study, peroxidase activity and biomass proved to be the most responsive indicators. Peroxidase activity showed significant increases in seven treatment groups and non-significant changes in two, while biomass exhibited significant decreases in six treatment groups and a non-significant response in one.

Peroxidase (POD) is widely recognized as a general indicator of plant stress (Czégény and Rácz 2023), responding to both drought conditions (Csiszár et al. 2012) and airborne pollution (Lucas et al. 2019). In our study, POD activity exhibited a diverse pattern. For the ‘Roma’ variety, elevated POD levels were observed across all treatment groups, indicating sensitivity to both water stress and PM exposure.

The ‘Mobil’ and ‘Lugas’ varieties appeared more responsive to PM treatment. ‘Mobil’ showed a statistically significant increase in POD activity in the WW/PM+ treatment group. In ‘Lugas’, POD levels were elevated under the same treatment, although the difference compared to the control was only marginally significant. Among the two landraces, increases in POD activity were observed solely in response to water stress: ‘Tápiószelei’ showed elevated POD levels in the RW/PM– group, while ‘Gulácsi’ exhibited increases in both the RW/PM– and RW/PM+ treatment groups.

Biomass reduction was observed in all treatment groups for the ‘Mobil’ variety. In ‘Roma’ and ‘Lugas’, biomass was more responsive to PM treatment, with significant decreases occurring in the WW/PM+ and RW/PM+ groups. Specifically, ‘Lugas’ showed a significant decrease in the RW/PM+ group and a marginally significant response in the WW/PM+ group.

In most studies, biomass reduction has been reported as the most sensitive indicator of stress, sometimes even more so than biochemical endpoints (Storch-Böhm et al. 2020). In our previous research, biomass was the most responsive endpoint when assessing the phytotoxic effects of atmospheric particulate matter on native plants (Kováts et al. 2021). Indeed, the OECD Guideline 227 considers biomass and physical damage symptoms as sufficient endpoints for toxicity evaluation (Kováts et al. 2017). The reduced growth rate can be partly attributed to damage at various structural levels of the photosynthetic apparatus (e.g., Oguntimehin et al. 2010). Additionally, drought may negatively impact photosynthesis by impairing stomatal function (Zhou et al. 2017), which can ultimately lead to decreased yield.

Chlorophyll pigments showed no significant response, whereas carotene was responsive to all treatment types in the ‘Roma’ variety. This lack of chlorophyll sensitivity contrasts with most findings reported in the literature. Early studies used decreased chlorophyll a and b levels as indicators of air pollution (Rabe and Kreeb, 1979). Similarly, Storch-Böhm et al. (2020) observed a significant reduction in total chlorophyll content in leaves experimentally exposed to diesel engine exhaust.

**Table 11.** Response of the tested varieties

## Roma

	<b>Biomass</b>	<b>Chl-a</b>	<b>Chl-b</b>	<b>Carotene</b>	<b>POD</b>	<b>Stomata</b>
TG2 RW/PM-	No difference	No difference	No difference	Increase	Increase	Increase
TG3 WW/PM+	Decrease	No difference	No difference	Increase	Increase	No difference
TG4 RW/PM+	Decrease	No difference	No difference	Increase	Increase	Increase

## Mobil

	<b>Biomass</b>	<b>Chl-a</b>	<b>Chl-b</b>	<b>Carotene</b>	<b>POD</b>	<b>Stomata</b>
TG2 RW/PM-	Decrease	No difference	No difference	No difference	Increase (marginally significant)	Increase
TG3 WW/PM+	Decrease	No difference	No difference	No difference	Increase	Increase
TG4 RW/PM+	Decrease	No difference	No difference	No difference	No difference	Increase

## Lugas

	<b>Biomass</b>	<b>Chl-a</b>	<b>Chl-b</b>	<b>Carotene</b>	<b>POD</b>	<b>Stomata</b>
TG2 RW/PM-	No difference	No difference	No difference	No difference	No difference	No difference
TG3 WW/PM+	Decrease (marginally significant)	No difference	No difference	No difference	Increase (marginally significant)	No difference
TG4 RW/PM+	Decrease	No difference	No difference	No difference	No difference	Increase

## Tápószelei

	<b>Biomass</b>	<b>Chl-a</b>	<b>Chl-b</b>	<b>Carotene</b>	<b>POD</b>	<b>Stomata</b>
TG2 RW/PM-	No difference	No difference	No difference	No difference	Increase	Increase
TG3 WW/PM+	No difference	No difference	No difference	No difference	No difference	Increase
TG4 RW/PM+	No difference	No difference	No difference	No difference	No difference	Increase

## Gulácsi

	<b>Biomass</b>	<b>Chl-a</b>	<b>Chl-b</b>	<b>Carotene</b>	<b>POD</b>	<b>Stomata</b>
TG2 RW/PM-	No difference	No difference	No difference	No difference	Increase	Decrease
TG3 WW/PM+	No difference	No difference	No difference	No difference	No difference	No difference
TG4 RW/PM+	No difference	No difference	No difference	No difference	Increase	Decrease

### 5. 2. 3. 2. Comparison of varieties used

Comparative studies on the tolerance of tomato landraces to atmospheric pollution, particularly particulate matter (PM), are relatively scarce. Regarding chemical stress, Saleem et al. (2015) reported that the 'Roma' variety was more sensitive to soil fluoride contamination than other tested varieties. Similarly, Jaja and Odoemena (2004) found 'Roma' to be more susceptible to lead (Pb) and copper (Cu) pollution during germination and growth assessments. Daresta et al. (2015) utilized the 'Roma' variety to evaluate the phytotoxic effects of atmospheric PM, growing plants on PM<sub>10</sub> collected from quartz fiber filters. Compared to the control, significant differences were observed in root growth. Reactive oxygen species (ROS) concentrations were measured in both control and treated roots, with significantly higher ROS levels detected in the treated samples, indicating that PM exposure stimulated ROS production. Additionally, the harmful effects of PM<sub>10</sub> on photosynthetic activity were evident, as the ratios of Chl a/Chl b and total chlorophyll (Chl a + b) to carotenoid content decreased in the treated plants. The 'Roma' variety was also employed as a model species to evaluate the applicability of the OECD protocol for testing PM-bound phytotoxicity (Kováts et al. 2020; Hubai et al. 2021). In the current comparative study, 'Roma' demonstrated the greatest sensitivity, exhibiting the highest number of significant responses (see Table 4).

There is more information available regarding the drought resistance of local landraces. Xu et al. (2017) compared the heat shock tolerance of different tomato varieties and identified significant differences. Guida et al. (2017) experimentally demonstrated the relative drought resistance of local Italian landraces under conditions without irrigation. Additionally, Landi et al. (2017) evaluated drought resistance in traditional Italian tomato landraces compared to commercial cultivars. Under drought stress, commercial cultivars exhibited constitutive activation of the ROS scavenging system due to increased oxidative stress, whereas traditional landraces experienced lower levels of oxidative stress. The study concluded that this difference may partly reflect a more efficient ROS scavenging mechanism in the traditional landraces.

In general, commercial cultivars are considered more drought-sensitive than landraces (Zdravkovic et al. 2013). In our study, the 'Gulácsi' landrace demonstrated the greatest drought resistance, showing the fewest responses—only elevated POD levels under water deficit conditions. It is worth noting that this was the only variety in our study to exhibit a reduction in stomatal surface area in response to water scarcity. Additionally, the leaves of this variety displayed a 'potato leaf' morphology. However, further research is needed to clarify any

potential relationship between leaf morphology and environmental stress resistance in tomato varieties Ahammed et al. (2012b). The results obtained for the three commercial varieties in this study further support their sensitivity to atmospheric pollution.

## **6. Contribution of atmospheric fallout to the soil-root-leaf transfer of PAHs in higher plants**

### **6. 1. Materials and Methods**

#### **6. 1. 1. Sampling and sample preparation**

Diesel emission sample was prepared as described in 5.1.1.

#### **6. 1. 2. Plant material used**

Rocket

- Latin name: *Eruca sativa* (L.) Mill.
- Taxonomy: Brassicaceae
- Life cycle: annual
- Raunkiaer classification: Therophyte
- Morphology: This plant has a lance-shaped leaf and four-petalled cream-coloured flowers. Reaching a height of up to 1 m, the leaves are 20 cm long.

This annual plant grows in late spring and early summer. The leaves are usually used in salads, sauces, or steamed as greens (Blažević & Mastelić 2008). Rocket seeds are also used in oil production. Rocket contains carotenoids, vitamin C, fibers, flavonoids, glucosinolates, and folic acid (Barillari et al. 2005). Generally, rocket is rich of vitamins A, C, and K, as well as thianin, riboflavin, niacin, vitamin B6 (pyridoxine), and pantothenic acid (Grami et al. 2024).

#### **6. 1. 3 Experimental procedures**

Treatment of test plants, preservation of the samples, phytotoxicity testing and chemical analyses were performed as described in 5.1.3.

## 6. 2. Results and Discussion

### 6. 2. 1. Composition of the PM extract

A total of 19 PAHs were identified in the PM extract. Table 11 presents the concentrations of accumulated PAHs in various plant compartments and in the soil following direct application of the extract to the soil. These included the 16 priority PAHs listed by the US EPA (Keith 2015), as well as the so-called 'car PAHs,' such as benzo(a)anthracene, chrysene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(a)pyrene, dibenzo(a,h)anthracene, indeno(1,2,3-cd)pyrene, and benzo(g,h,i)perylene (Srogi 2007).

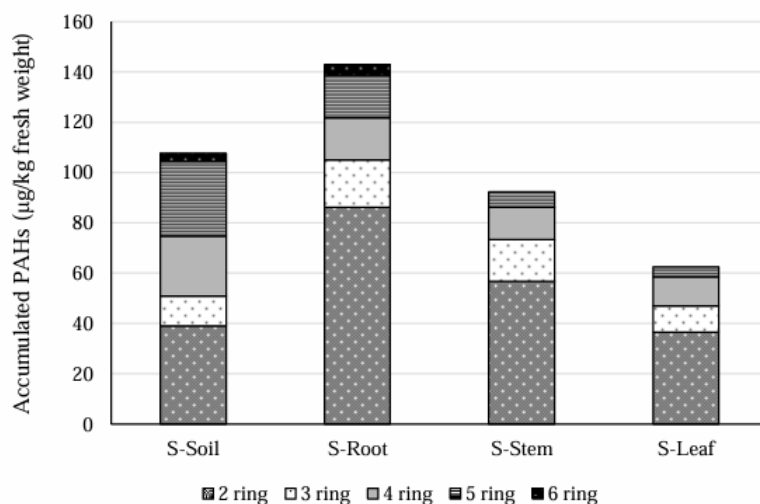
The extract showed a predominance of low molecular weight (LMW) PAHs. In addition to naphthalenes, the dominant compounds were the 3-ring PAHs fluorene ( $1.472 \mu\text{g L}^{-1}$ ) and phenanthrene ( $0.448 \mu\text{g L}^{-1}$ ). These compounds are commonly reported as dominant species in atmospheric samples (as reviewed by (Dat & Chang 2017)) and in bulk deposition (Ollivon et al. 2002). Phenanthrene (Phen) was identified as the dominant PAH in a study investigating the distribution of traffic-related contaminants in urban topsoils (Nikolaeva et al. 2017). It was also found to be a characteristic component in various diesel blends, as reported by de Souza (de Souza & Corrêa 2016).

In the treated soil, the predominant PAH was the 5-ring compound benzo(b)fluoranthene (B(b)f), with a concentration of  $11.7 \mu\text{g kg}^{-1}$ . Consistently, Qi et al. (Qi et al. 2023) identified B(b)f as the most prevalent PAH in urban environments, comprising 39.94% of total PAHs. Notably high concentrations of the 4-ring PAHs pyrene ( $9.2 \mu\text{g kg}^{-1}$ ) and chrysene ( $8.3 \mu\text{g kg}^{-1}$ ) were also detected. These compounds are commonly recognized as markers of urban soil contamination (Wang 2013).

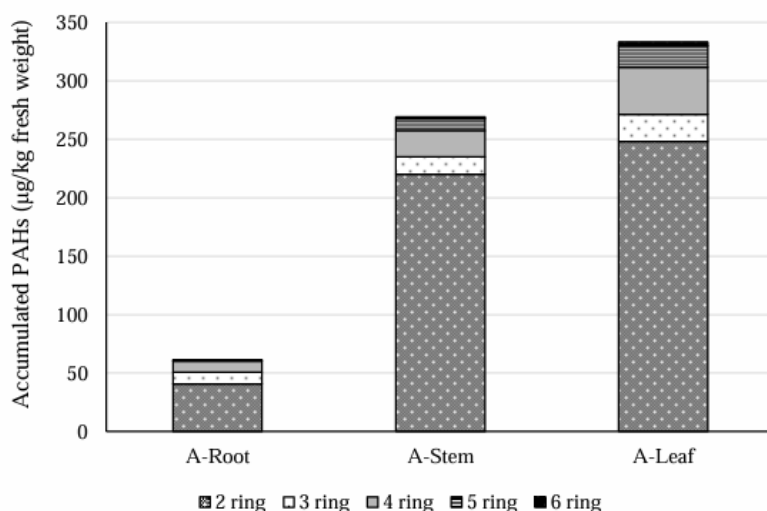
**Table 12.** Concentrations of PAHs in the aerosol extract (expressed in  $\mu\text{g L}^{-1}$ ) and in the treated samples (expressed in  $\mu\text{g kg}^{-1}$ ). PAHs prioritized by the US EPA are indicated in bold. S: soil treatment; A: aerial parts treatment.

	PAHs	Extract	S-Soil	S-Root	S-Stem	S-Leaf	A-Root	A-Stem	A-Leaf
2- rings	Naphthalene	0.491	32.6	78.5	50.2	30.4	32	208	234
	2-methyl-naphthalene	0.394	4.2	5.1	4.2	3.9	4.4	7.7	8.6
	1-methyl-naphthalene	0.367	2.1	2.5	2.3	2.2	4.1	4.4	5.5
3- rings	Acenaphthylene	0.12	1.4	1.9	1.6	0.7	0.8	0.5	5.2
	Acenaphthene	0.052	1.7	2	1.9	1.5	1.87	2.4	2.4
	Flourene	0.472	1.1	4.1	4.1	2.9	3.3	4.8	5.2
	Phenanthrene	0.448	5.4	7.8	7.5	4.9	2.9	6.4	8.6
4- rings	Anthracene	0.058	2.3	3.1	1.5	0.4	1.2	0.9	2
	Flouranthene	0.074	1.62	6.2	5.7	5.1	5.7	9.3	10.1
	Pyrene	0.081	9.2	3.5	2.9	2.9	1.8	5.1	9.1
	Benzoanthracene	0.032	4.7	4.7	2.6	2.5	1.7	5.6	8.1
	Chrysene	0.013	8.3	2.4	1.6	1.1	0.4	2.5	12.7
5- rings	Benzo(b)fluoranthene	0.064	11.7	4.6	2.8	2.2	1.1	3.2	8.4
	Benzo(k)fluoranthene	0.036	4.9	4.7	0.8	0.7	0	2.3	2.2
	Benzo(e)pyrene	0.055	5.6	2.7	1.6	1.1	0	1.8	4.3
	Benzo(a)pyrene	0.043	3.6	3	1	0	0	1.1	1.4
	Dibenzo(a,h)anthracene	0.045	4	1.9	0	0	0	1.5	2.3
6- rings	Indeno1.2.3CD-Pyrene	0.033	1.2	1.5	0	0	0	0.5	1
	Benzo(g,h,i)perylene	0.035	2.1	2.8	0	0	0	1.3	2.1
<b>Total PAHs</b>			<b>108</b>	<b>143</b>	<b>92.3</b>	<b>62.5</b>	<b>61.3</b>	<b>369</b>	<b>333</b>

Figure 26 illustrates the distribution of 2- to 6-ring PAHs across different compartments along the soil–root–aerial parts pathway. Figure 27 presents the ratios of various PAH isomers associated with the aerial parts–roots pathway.



**Figure 26.** Concentrations of PAHs with varying molecular weights in soil and treated plant compartments, illustrating the soil-root-aerial parts transfer pathway



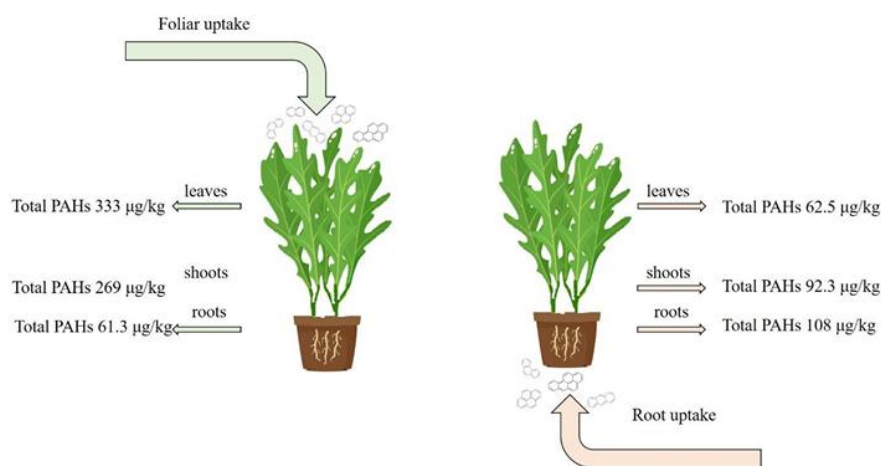
**Figure 27.** Concentrations of various molecular weight PAHs in soil and plant compartments, illustrating the soil-to-root-to-aerial parts pathway

### 6. 2. 2. Discussion

Following the three treatments on Days 1, 8, and 15, the total PAH concentration in the treated soil measured  $108 \mu\text{g kg}^{-1}$ , suggesting that wet deposition is a significant contributor to soil PAH levels.

Regarding the accumulation and potential transport of the total 19 PAHs, a distinct pattern was observed along both the soil-root-stem-leaf and leaf-stem-root pathways (Figure 28). In the soil-root-stem-leaf pathway, where the soil was treated with the extract, the total

concentrations were  $108 \mu\text{g kg}^{-1}$  in the soil,  $143 \mu\text{g kg}^{-1}$  in the roots,  $92.3 \mu\text{g kg}^{-1}$  in the stems, and  $62.5 \mu\text{g kg}^{-1}$  in the leaves. In the leaf-stem-root pathway, where the above-ground parts were treated with the extract, the highest concentration was detected in the leaves ( $333 \mu\text{g kg}^{-1}$ ), followed by a similarly high level in the stems ( $269 \mu\text{g kg}^{-1}$ ), and a lower concentration in the roots ( $61.3 \mu\text{g kg}^{-1}$ ), indicating a decreasing trend from leaves to stems to roots.



**Figure 28.** Accumulation of total PAHs across various compartments

Evaluating the soil-root-aerial parts pathway, the results indicate significant bioconcentration between the soil and roots, followed by a decreasing trend from roots to stems and leaves. This pattern aligns with existing literature. While earlier studies suggested that root uptake was the primary route for persistent organic pollutants (POPs) (Huelster et al. 1994), more recent research shows that the transport of PAHs from roots to above-ground plant parts is relatively limited (Wild et al. 2006; Wieczorek & Wieczorek 2007). However, some studies have not identified such clear pathways. For example, Jia et al. cultivated vegetables—including Shanghai green cabbage (*Brassica chinensis* var.) and Chinese cabbage (*Brassica rapa* var.)—in experimentally treated soils and found average concentrations of 16 PAHs to be  $204.7 \mu\text{g kg}^{-1}$  in the soil,  $83.0 \mu\text{g kg}^{-1}$  in the roots, and notably higher again in the leaves at  $93.9 \mu\text{g kg}^{-1}$  (Jia et al. 2021).

In the case of soil treatment, uptake and even bioconcentration were observed for low molecular weight (LMW) PAHs, specifically those with 2 and 3 rings, as well as for one 4-ring PAH, fluoranthene (Flt). Similarly, Gao and Zhu demonstrated the translocation of phenanthrene (Phen) and pyrene (Pyr) from roots to shoots in experimentally spiked soils (Gao & Zhu 2004). Imam et al. also reported that leafy vegetables grown in contaminated soils primarily accumulated LMW PAHs (Inam et al. 2016). In a related pot study, Zhang et al.

investigated PAH uptake in Chinese cabbage (*Brassica chinensis* L.), a close relative of rocket, and assessed the accumulation of PAHs with varying ring numbers. The study reported a negative correlation between PAH uptake from experimentally spiked soil and the molecular weight of the compounds, with uptake reaching 76.55% for low molecular weight (LMW) PAHs but only 6.05% for high molecular weight (HMW) PAHs (Zhang et al. 2021). Similarly, Cao et al. cultivated carrot (*Daucus carota* L.) and cabbage (*Brassica pekinensis* L.) in PAH-contaminated soils under both field and greenhouse conditions. Their findings revealed that LMW PAHs accumulated more readily in both plant species, as indicated by higher BCF values compared to HMW PAHs (Cao et al. 2024). Jiao et al. conducted a pot study to investigate the migration of PAH congeners in a soil–ryegrass exposure system. The study demonstrated that higher molecular weight PAHs ( $\geq$  benzo[a]anthracene) predominantly adsorbed to soil particles, thereby reducing their bioavailability and subsequent plant uptake (Jiao et al. 2023).

Statistical analysis revealed significant differences in PAH concentrations across plant compartments. For 3-ring PAHs, concentrations differed significantly between roots and leaves ( $t = 3.6327$ ,  $df = 4$ ,  $p = 0.02211$ ). Similarly, 4-ring PAHs showed significant differences between roots and leaves ( $t = 3.8903$ ,  $df = 3$ ,  $p = 0.03012$ ). In the case of 5-ring PAHs, significant differences were observed between roots and stems ( $t = 4.5771$ ,  $df = 4$ ,  $p = 0.01021$ ), as well as between roots and leaves ( $t = 6.041$ ,  $df = 4$ ,  $p = 0.003787$ ).

The concentrations of 6-ring PAHs were very similar in the soil and roots, with indeno[1,2,3-cd]pyrene (IP) measured at  $1.2 \mu\text{g kg}^{-1}$  in the soil and  $1.5 \mu\text{g kg}^{-1}$  in the roots, and benzo[ghi]perylene (B(g,h,i)p) at  $2.1 \mu\text{g kg}^{-1}$  and  $2.8 \mu\text{g kg}^{-1}$ , respectively. These compounds were not detected in the above-ground parts of the test plants (Figure 2). In a related study, Paraíba et al. cultivated corn in soil fertilized with wastewater sludge and evaluated PAH bioaccumulation in the grains. The study concluded that high molecular weight (HMW) PAHs have limited mobility through the transpiration stream and exhibit low accumulation potential in corn grains (Paraíba et al. 2010). A similar trend was observed by Jia et al. (2021), who reported that although experimentally treated soils contained a relatively high proportion of HMW PAHs (with 5-ring PAHs accounting for 20% and 6-ring PAHs for 85%), these compounds were not detected in the vegetables grown in those model soils (Jia et al. 2021).

Numerous studies have investigated the uptake of PAHs from contaminated soils and their subsequent distribution among different plant organs. In a systematic study, Tao et al. measured total PAH concentrations in various parts of rice plants cultivated at a PAH-

contaminated site. Contrary to expectations of a vertical concentration gradient, higher PAH levels were detected in plant parts with larger surface areas, such as leaf blades (Tao et al. 2006). While atmospheric PAHs can settle into the soil, their strong sorption in the root zone often limits their translocation to above-ground plant parts (Bogolte et al. 2007). Persistent organic pollutants (POPs), including PAHs, are generally believed to bind tightly to the root epidermis (Moeckel et al. 2009). The uptake of PAHs in the root zone is influenced by their lipophilicity, typically expressed as the octanol-water partition coefficient ( $\log K_{ow}$ ); thus, high molecular weight (HMW) PAHs tend to adsorb onto root surfaces rather than being absorbed and translocated (Fismes et al. 2002).

Several studies have examined PAH uptake by root vegetables. Zhang et al. investigated the atmospheric dust–soil–yellow carrot core pathway and found that yellow carrot (*Daucus carota* subsp. *sativus*) accumulated PAHs from the surrounding soil, with notable concentrations detected in the core tissue beneath the peels (Zhang et al. 2018).

In the case of atmospheric uptake of high molecular weight (HMW) PAHs via the air-leaf-stem-root pathway, a gradual decline in the concentration of individual PAHs was observed, mirroring the trend seen for the total 19 PAHs. This decrease was more pronounced for HMW PAHs (Figure 3). Statistically significant differences were observed in the concentrations of 4-ring PAHs between leaves and roots ( $t = 4.5244$ ,  $df = 3$ ,  $p = 0.02019$ ), as well as between stems and roots ( $t = 8.175$ ,  $df = 3$ ,  $p = 0.003829$ ). Similarly, when comparing 5-ring PAHs concentrations, significant differences were found between leaves and roots ( $t = 3.2911$ ,  $df = 4$ ,  $p = 0.03019$ ) and between stems and roots ( $t = 8.242$ ,  $df = 4$ ,  $p = 0.001182$ ).

In the leaves, the 2-ring Naphthalene showed the highest concentration,  $234 \mu\text{g kg}^{-1}$ . Of 3-ring PAHs, Phen was the dominant species, its concentration was  $8.6 \mu\text{g kg}^{-1}$ . In the study of Zhao et al. it was the most abundant PAH in *Salix matsudana* leaves, showing good consistency with atmospheric concentrations (Zhao et al. 2018). In the leaves, the 2-ring PAH Nap exhibited the highest concentration at  $234 \mu\text{g kg}^{-1}$ . Among the 3-ring PAHs, Phen was the most prevalent, with a concentration of  $8.6 \mu\text{g kg}^{-1}$ . This finding aligns well with the study by Zhao et al. where Phen was also the most abundant PAH in *Salix matsudana* leaves, reflecting atmospheric concentration patterns (Zhao et al. 2018). Yakovleva et al. reported similar findings, identifying Phen as the most abundant PAH in plants and atmospheric wet deposition when examining soil-plant systems in northern taiga biocenoses (Yakovleva et al. 2012).

6-ring PAHs were detected in both the leaves and stems. The concentration of IP was  $1 \mu\text{g kg}^{-1}$  in the leaves and  $0.5 \mu\text{g kg}^{-1}$  in the stems, while B(g,h,i)P measured  $2.1 \mu\text{g kg}^{-1}$  in the leaves and  $1.3 \mu\text{g kg}^{-1}$  in the stems. However, these compounds were not translocated to the roots. Similarly, 5-ring PAHs were found in the leaves and stems (Fig. 3), but except for B(b)f, they were not transported to the roots. The concentration of B(b)f clearly decreases from  $8.4 \mu\text{g kg}^{-1}$  in the leaves to  $3.2 \mu\text{g kg}^{-1}$  in the stems and  $1.1 \mu\text{g kg}^{-1}$  in the roots. These findings also highlight that the bioaccumulation of high molecular weight (HMW) PAHs should not be overlooked. Elevated PAH levels in internal leaf tissues are likely due to limited translocation and the relative persistence of these hydrophobic compounds (Desalme et al. 2011; Desalme et al. 2011).

## 7. Conclusions

Plants – both cropped species and native ones – are exposed to atmospheric pollution in many forms, gaseous compounds, particulate matter and potentially toxic contaminants which occur bound to particles. A wide range of anthropogenic sources contribute to the emission of these pollutants, in this work two major sources, traffic-related emissions and open burning of domestic waste have been addressed. Phytotoxic responses to PM-bound polycyclic aromatic hydrocarbons and heavy metals were assessed based on the No. 227 OECD GUIDELINE FOR THE TESTING OF CHEMICALS: Terrestrial Plant Test: Vegetative Vigour Test. The Guideline has been adopted to test the potential phytotoxic effects of atmospheric PM and in previous works proved applicable for the evaluation of atmospheric pollution. Based on our comparative studies, phytotoxicity of both traffic-related PM emissions and open burning of domestic waste have been supported. Phytotoxicity of the samples was evaluated based on different end-points, such as biomass reduction, photosynthetic pigments and the antioxidant enzyme peroxidase (POD). In concordance with literature review, biomass reduction and POD proved sensitive endpoints, however, photosynthetic pigments showed much lower sensitivity than reported by available studies. Sensitivity, however, proved strongly species-specific for both cropped species and commonly grown herbaceous plants.

## 8. New scientific points

1. Sensitivity of five different tomato varieties/landraces such as 'Roma', 'Mobil' and 'Lugas' (commercially available varieties) as well as 'Gulácsi' and 'Tápiószelei' (regional landraces) was compared to the combined effect of air pollution and drought stress on tomato (*Lycopersicon esculentum* Mill.). Water stress had more pronounced effect while treatment with the PM extract triggered response only in case of the three commercial varieties. In general, higher resistance of landraces toward abiotic stress was established, the commercial variety 'Roma' showing the highest while the regional landrace 'Gulácsi' showing the lowest sensitivity to individual and combined treatments.
2. The treatment procedure described in the No. 227 OECD GUIDELINE FOR THE TESTING OF CHEMICALS: Terrestrial Plant Test: Vegetative Vigour Test was also used in a modified form for bioaccumulation studies. White mustard (*Sinapis alba* L.) plants were experimentally treated with an extract of diesel exhaust, in a chronic test, applying weekly treatment over a 3-month exposure period. Results revealed that even seeds were affected, showing considerable accumulation of higher molecular weight PAHs. Based on the study, it can be assumed that traffic-related air pollution damages crop quality in fields situated nearby roads.
3. Simulating wet deposition of atmospheric polycyclic aromatic hydrocarbons (PAHs) in rocket (*Eruca sativa* Mill., family Brassicaceae), atmospheric uptake of PAHs proved the main pathway, especially for higher molecular weight PAHs. Results also demonstrated that both soil-aerial parts and aerial parts-root transfers are low, especially in case of HMW PAHs.
4. Phytotoxic responses to emissions generated during experimental burning of municipal waste were evaluated comparing two commonly applied test plants (*Lactuca sativa* and *Sinapis alba*) and six ornamental plants (*Gaillardia aristata*, *Calendula officinalis*, *Alcea rosea annua*, *Ipomoea purpurea*, *Mirabilis jalapa* and *Cucurbita pepo*). Strong species-specific sensitivity was experienced, *L. sativa* and *M. jalapa* being the most responsive, in concordance with reported studies.
5. In case of diesel emission and waste burning generated PM samples, growth inhibition and peroxidase enzyme activity proved the most responsive end-points, in concordance with reported studies.

## **Acknowledgments**

I would like to express my gratitude my supervisors Professor Nora Kováts, and Dr. Katalin Hubai for their guidance, cooperation, teaching, planting and understanding me during these academic years.

I would like to thank Dr. Bettina Eck-Varanka for her big help for during the laboratory work. Also, I am really grateful for the other Professors who taught me lectures and laboratory work.

Special thanks to University of Pannonia, and Hungarian government who supported and gave the opportunity during these years.

I am also so thankful to my family and friends who encouraged and supported me all the time.

## 9. References

- Abbasi, B. H.; Rashid, A.; Khan, M. A.; Ali, M.; Shinwari, Z. K.; Ahmad, N.; Mahmood, T. (2011): In vitro plant regeneration in *Sinapis alba* and evaluation of its radical scavenging activity. *Pakistan Journal of Botany*, 43, 21-27.
- Abdel-Shafy, H. I.; Mansour, M. S. (2016): A review on polycyclic aromatic hydrocarbons: Source, environmental impact, effect on human health and remediation. *Egyptian journal of petroleum*, 25(1), 107-123.
- Ahammed, G. J.; Wang, M. M.; Zhou, Y. H.; Xia, X. J.; Mao, W. H.; Shi, K.; Yu, J. Q. (2012a): The growth, photosynthesis and antioxidant defense responses of five vegetable crops to phenanthrene stress. *Ecotoxicology and environmental safety*, 80, 132-139.
- Ahammed, G. J.; Yuan, H. L.; Ogwen, J. O.; Zhou, Y. H.; Xia, X. J.; Mao, W. H.; Yu, J. Q. (2012b): Brassinosteroid alleviates phenanthrene and pyrene phytotoxicity by increasing detoxification activity and photosynthesis in tomato. *Chemosphere*, 86(5), 546-555.
- Ahammed, G. J.; Yuan, H. L.; Ogwen, J. O.; Zhou, Y. H.; Xia, X. J.; Mao, W. H.; Yu, J. Q. (2012c): Brassinosteroid alleviates phenanthrene and pyrene phytotoxicity by increasing detoxification activity and photosynthesis in tomato. *Chemosphere*, 86(5), 546-555.
- Akanbi-Gada, M. A.; Ogunkunle, C. O.; Vishwakarma, V.; Viswanathan, K.; Fatoba, P. O. (2019): Phytotoxicity of nano-zinc oxide to tomato plant (*Solanum lycopersicum* L.): Zn uptake, stress enzymes response and influence on non-enzymatic antioxidants in fruits. *Environmental Technology & Innovation*, 14, 100325.
- Akinyemi, A. J.; Faboya, O. L.; Olayide, I.; Faboya, O. A.; Ijabadeniyi, T. (2017): Effect of Cadmium Stress on Non-enzymatic Antioxidant and Nitric Oxide Levels in Two Varieties of Maize (*Zea mays*). *Bulletin of Environmental Contamination and Toxicology*, 98(6), 845–849.
- Alengebawy, A.; Abdelkhalek, S. T.; Qureshi, S. R.; Wang, M. Q. (2021): Heavy metals and pesticides toxicity in agricultural soil and plants. *Ecological risks and human health implications*. *Toxics*, 9(3), 42.

Ali, I.; Liu, B.; Farooq, M.; Ahsan, Islam. F. (2016): Toxicological effects of bisphenolA on growth and antioxidant defense system in *Oryza sativa* as revealed by ultrastructure analysis. *Ecotoxicology and Environmental Safety*, 124, 277–284.

Amato-Lourenco, L. F.; Moreira, T. C. L.; de Oliveira Souza, V.C.; Barbosa, F. Jr.; Saiki, M.; Saldiva, P. H. N.; Mauad, T. (2016): The influence of atmospheric particles on the elemental content of vegetables in urban gardens of Sao Paulo, Brazil. *Environmental Pollution*, 216, 125-134.

Anbzhakan, R.; Li, N. Q.; Poudel, B.; Gao, J. Y. (2025): In vitro regeneration of *Ipomoea purpurea* (L.) Roth via indirect organogenesis from seedling-derived internode explants. *Vegetos*, 1-8.

Arikan, B.; Yildiztugay, E.; Ozfidan-Konakci, C. (2023): Responses of salicylic acid encapsulation on growth, photosynthetic attributes and ROS scavenging system in *Lactuca sativa* exposed to polycyclic aromatic hydrocarbon pollution. *Plant Physiology and Biochemistry*, 203, 108026.

Arya, S.; Rautela, R.; Chavan, D.; Kumar, S. (2021): Evaluation of soil contamination due to crude E-waste recycling activities in the capital city of India. *Process Safety and Environmental Protection*, 152, 641-653.

Asati, A.; Pichhode, M.; Nikhil, K. (2016): Effect of heavy metals on plants: an overview. *International Journal of Application or Innovation in Engineering & Management*, 5(3), 56-66.

Bag, N.; Palni, L. M. S.; Chandra, S.; Nandi, S. K. (2012): Somatic embryogenesis in 'maggar'bamboo (*Dendrocalamus hamiltonii*) and field performance of regenerated plants. *Current science*, 1279-1287.

Bai, Y.; Brugha, R. E.; Jacobs, L.; Grigg, J.; Nawrot, T. S.; Nemery, B. (2015): Carbon loading in airway macrophages as a biomarker for individual exposure to particulate matter air pollution—A critical review. *Environment international*, 74, 32-41.

Bansal, P.; Verma, S.; Srivastava, A. (2016): Biomonitoring of air pollution using antioxidative enzyme system in two genera of family Pottiaceae (Bryophyta). *Environmental Pollution*, 216, 512-518.

- Bansal, V.; Kim, K. H. (2015): Review of PAH contamination in food products and their health hazards. *Environment international*, 84, 26-38.
- Barillari, J.; Canistro, D.; Paolini, M.; Ferroni, F.; Pedulli, G. F.; Iori, R.; Valgimigli, L. (2005): Direct antioxidant activity of purified glucoerucin, the dietary secondary metabolite contained in rocket (*Eruca sativa* Mill.) seeds and sprouts. *Journal of Agricultural and food chemistry*, 53(7), 2475-2482.
- Barral, M. T.; Paradelo, R. (2011): A review on the use of phytotoxicity as a compost quality indicator. *Dyn. Soil Dyn. Plant*, 5(2), 36-44.
- Bell, J. N. B.; Honour, S. L.; Power, S. A. (2011): Effects of vehicle exhaust emissions on urban wild plant species. *Environmental Pollution*, 159, 1984-1990.
- Bergweiler, C.; Carreras, H.; Wannaz, E.; Rodriguez, J.; Toselli, B.; Olcese, L.; Pignata, M. L. (2008). Field surveys for potential ozone bioindicator plant species in Argentina. *Environmental monitoring and assessment*, 138.
- Beyer, W. N.; Green, C. E.; Beyer, M.; Chaney, R. L. (2013): Phytotoxicity of zinc and manganese to seedlings grown in soil contaminated by zinc smelting. *Environmental Pollution*, 179, 167-176.
- Bhaduri, A. M.; Fulekar, M. H. (2012): Antioxidant enzyme responses of plants to heavy metal stress. *Reviews in Environmental Science and Bio/Technology*, 11, 55-69.
- Bharti, S. K.; Trivedi, A.; Kumar, N. (2018): Air pollution tolerance index of plants growing near an industrial site. *Urban climate*, 24, 820-829.
- Blažević, I.; Mastelić, J. (2008): Free and bound volatiles of rocket (*Eruca sativa* Mill.). *Flavour and fragrance journal*, 23(4), 278-285.
- Bogolte, B. T.; Ehlers, G. A. C.; Braun, R.; Loibner, A. P. (2007): Estimation of PAH bioavailability to *Lepidium sativum* using sequential supercritical fluid extraction—a case study with industrial contaminated soils. *European journal of soil biology*, 43(4), 242-250.
- Bones, A. M.; Rossiter, J. T. (2006): The enzymic and chemically induced decomposition of glucosinolates. *Phytochemistry*, 67(11), 1053-1067.

Borowiak, K.; Budka, A.; Hanć, A.; Kayze, D.; Lisiak, M.; Zbierska, J.; ... & Łopatka, N. (2018). Relations between photosynthetic pigments, macro-element contents and selected trace elements accumulated in *Lolium multiflorum* L. exposed to ambient air conditions. *Acta Biologica Cracoviensia series Botanica*, 60(1).

Boscaro, V.; Boffa, L.; Binello, A.; Amisano, G.; Fornasero, S.; Cravotto, G.; Gallicchio, M. (2018): Antiproliferative, proapoptotic, antioxidant and antimicrobial effects of *Sinapis nigra* L. and *Sinapis alba* L. extracts. *Molecules*, 23(11), 3004.

Botta-Dukát, Z.; Bartha, D.; Dancza, I.; Lukács, B. A.; Pinke, G. (2023): Adaptation of life form categorisation of Ellenberg and Mueller-Dombois to the Hungarian Flora. *Acta Botanica Hungarica*, 65(1-2), 1-34.

Boutin, C.; Aya, K.L.; Carpenter, D.; Thomas, P.J.; Rowland, O. (2012): Phytotoxicity testing for herbicide regulation: Shortcomings in relation to biodiversity and ecosystem services in agrarian systems. *Science of the Total Environment*, 415, 79–92.

Bruzzese, E.; Hasan, S. (1983): A Whole Leaf Clearing and Staining Technique for Host Specificity Studies of Rust Fungi. *Plant Pathol.*, 32: 335- 338

Buleandra, M.; Crisan, C. C.; Calinescu, I.; Zalaru, C.; Moldovan, Z.; David, I. G.; Badea, I. A. (2015): Rapid Analysis of the Volatile Components of *Gaillardia aristata* and *G. × grandiflora*. *Chemistry of Natural Compounds*, 51, 787-789.

Busso, I. T.; Tames, F.; Silva, J.A.; Ramos, S.; Homem, V.; Ratola, N.; Carreras, H. (2018): Biomonitoring levels and trends of PAHs and synthetic musks associated with land use in urban environments. *Science of the Total Environment*, 618, 93–100.

Cao, C.; Wu, Y. Y.; Lv, Z. Y.; Wang, J. W.; Wang, C. W.; Zhang, H.; Chen, H. (2024): Uptake of polycyclic aromatic hydrocarbons (PAHs) from PAH-contaminated soils to carrots and Chinese cabbages under the greenhouse and field conditions. *Chemosphere*, 360, 142405.

Capozzi, F.; Di Palma, A.; Adamo, P.; Spagnuolo, V.; Giordano, S. (2017): Monitoring chronic and acute PAH atmospheric pollution using transplants of the moss *Hypnum cupressiforme* and *Robinia pseudacacia* leaves. *Atmospheric Environment*, 150, 45-54.

- Carpenter, C.; Boutin, C.; Allison, J. E.; (2013): Effects of chlorimuron ethyl on terrestrial and wetland plants: Levels of, and time to recovery following sublethal exposure. *Environmental Pollution*, 172, 275-282.
- Caumo, S.; Yera, A. B.; Vicente, A.; Alves, C.; Roubicek, D. A.; Vasconcellos, P. (2022): Particulate matter-bound organic compounds: Levels, mutagenicity, and health risks. *Environmental Science and Pollution Research*, 29, 31293–31310.
- Çelik, Ö.; Ayan, A.; Meriç, S.; Atak, Ç. (2021): Comparison of tolerance related proteomic profiles of two drought tolerant tomato mutants improved by gamma radiation. *Journal of Biotechnology*, 330, 35-44.
- Chapa-Martínez, C. A.; Hinojosa-Reyes, L.; Hernández-Ramírez, A.; Ruiz-Ruiz, E.; Maya-Treviño, L.; Guzmán-Mar, J. L. (2016): An evaluation of the migration of antimony from polyethylene terephthalate (PET) plastic used for bottled drinking water. *Science of the total Environment*, 565, 511-518.
- Chemingui, H.; Smiri, M.; Missaoui, T.; Hafiane, A. (2019): Zinc Oxide Nanoparticles Induced Oxidative Stress and Changes in the Photosynthetic Apparatus in Fenugreek (*Trigonella foenum graecum* L.). *Bulletin of Environmental Contamination and Toxicology*, 102(4), 477–485
- Chen, Y.; Zhang, F.; Zhang, J.; Zhou, M.; Li, F.; Liu, X. (2018): Accumulation characteristics and potential risk of PAHs in vegetable system grow in home garden under straw burning condition in Jilin, Northeast China. *Ecotoxicology and environmental safety*, 162, 647-654.
- Cheng, K.; Hao, W.; Wang, Y.; Yi, P.; Zhang, J.; Ji, W. (2020): Understanding the emission pattern and source contribution of hazardous air pollutants from open burning of municipal solid waste in China. *Environmental Pollution*, 263, 114417.
- Corrêa, A. X. R.; Cotelle, S.; Millet, M.; Somensi, C. A.; Wagner, T. M.; Radetski, C. M. (2016): Genotoxicity assessment of particulate matter emitted from heavy-duty diesel-powered vehicles using the in vivo *Vicia faba* L. micronucleus test. *Ecotoxicology and Environmental Safety*, 127, 199–204.
- Csiszár, J.; Gallé, Á.; Horváth, E.; Dancsó, P.; Gombos, M.; Váry, Z.; Tari, I. (2012): Different peroxidase activities and expression of abiotic stress-related peroxidases in apical root segments

of wheat genotypes with different drought stress tolerance under osmotic stress. *Plant Physiology and Biochemistry*, 52, 119-129.

Czégény, G.; Rácz, A. (2023): Phenolic peroxidases: Dull generalists or purposeful specialists in stress responses?. *Journal of Plant Physiology*, 280, 153884.

Daresta, B. E.; Italiano, F.; de Gennaro, G.; Trotta, M.; Tutino, M.; Veronico, P. (2015): Atmospheric particulate matter (PM) effect on the growth of *Solanum lycopersicum* cv. Roma plants. *Chemosphere*, 119, 37-42.

Das, B.; Bhave, P. V.; Sapkota, A.; Byanju, R. M. (2018): Estimating emissions from open burning of municipal solid waste in municipalities of Nepal. *Waste management*, 79, 481-490.

Dat, N. D.; Chang, M. B. (2017): Review on characteristics of PAHs in atmosphere, anthropogenic sources and control technologies. *Science of the total environment*, 609, 682-693.

de Souza Cardoso, A. A.; Monteiro, F. A. (2021): Sulfur supply reduces barium toxicity in Tanzania guinea grass (*Panicum maximum*) by inducing antioxidant enzymes and proline metabolism. *Ecotoxicology and Environmental Safety*, 208, 111643.

de Souza, C. V.; Corrêa, S. M. (2016): Polycyclic aromatic hydrocarbons in diesel emission, diesel fuel and lubricant oil. *Fuel*, 185, 925-931.

De Vries, I. M. (1997): Origin and domestication of *Lactuca sativa* L. *Genetic Resources and Crop Evolution*, 44, 165-174.

Desalme, D.; Binet, P.; Bernard, N.; Gilbert, D.; Toussaint, M. L.; Chiapusio, G. (2011): Atmospheric phenanthrene transfer and effects on two grassland species and their root symbionts: A microcosm study. *Environmental and experimental botany*, 71(2), 146-151.

Di Vaio, P.; Coccoziello, B.; Corvino, A.; Fiorino, F.; Frecentese, F.; Magli, E.; Perissutti, E. (2016): Level, potential sources of polycyclic aromatic hydrocarbons (PAHs) in particulate matter (PM10) in Naples. *Atmospheric Environment*, 129, 186-196.

Duan, J.; Tan, J. (2013): Atmospheric heavy metals and arsenic in China: situation, sources and control policies. *Atmospheric Environment*, 74, 93-101.

Eck, J. V.; Kirk, D. D.; Walmsley, A. M. (2006): Tomato (*Lycopersicon esculentum*). *Agrobacterium Protocols*, 459-474.

Eriksen, M. K.; Astrup, T. F. (2019): Characterisation of source-separated, rigid plastic waste and evaluation of recycling initiatives: Effects of product design and source-separation system. *Waste Management*, 87, 161-172.

Esmaeili, A.; Knox, O.; Juhasz, A.; Wilson, S. C. (2021): Advancing prediction of polycyclic aromatic hydrocarbon bioaccumulation in plants for historically contaminated soils using *Lolium multiflorum* and simple chemical in-vitro methodologies. *Science of the Total Environment*, 772, 144783.

European Parliament (2008): Directive 2008/50/EC of the European Parliament and of the Council of 21 May 2008 on ambient air quality and cleaner air for Europe

Fayyazbakhsh, A.; Bell, M. L.; Zhu, X.; Mei, X.; Koutný, M.; Hajinajaf, N.; Zhang, Y. (2022): Engine emissions with air pollutants and greenhouse gases and their control technologies. *Journal of Cleaner Production*, 376, 134260

Filella, M. (2020): Antimony and PET bottles: Checking facts. *Chemosphere*, 261, 127732.

Fismes, J.; Perrin-Ganier, C.; Empereur-Bissonnet, P.; Morel, J. L. (2002): Soil-to-root transfer and translocation of polycyclic aromatic hydrocarbons by vegetables grown on industrial contaminated soils. *Journal of Environmental quality*, 31(5), 1649-1656.

Gao, P. P.; Xue, P. Y.; Dong, J. W.; Zhang, X. M.; Sun, H. X.; Geng, L. P.; Liu, W. J. (2021): Contribution of PM<sub>2.5</sub>-Pb in atmospheric fallout to Pb accumulation in Chinese cabbage leaves via stomata. *Journal of Hazardous Materials*, 407, 124356.

Gao, Y.; Zhu, L. (2004): Plant uptake, accumulation and translocation of phenanthrene and pyrene in soils. *Chemosphere*, 55(9), 1169-1178.

Giri, S.; Shrivastava, D.; Deshmukh, K.; Dubey, P. (2013): Effect of air pollution on chlorophyll content of leaves. *Current Agriculture Research Journal*, 1(2), 93-98.

Gogoi, J.; Nakhuru, K. S.; Policegoudra, R. S.; Chattopadhyay, P.; Rai, A. K.; Veer, V. (2016): Isolation and characterization of bioactive components from *Mirabilis jalapa* L. radix. *Journal of Traditional and Complementary Medicine*, 6(1), 41-47.

- Goswami, S.; Das, S. (2016): Copper phytoremediation potential of *Calandula officinalis* L. and the role of antioxidant enzymes in metal tolerance. *Ecotoxicology and environmental safety*, 126, 211-218.
- Grami, D.; Selmi, S.; Rtibi, K.; Sebai, H.; De Toni, L. (2024): Emerging role of *Eruca sativa* Mill. in male reproductive health. *Nutrients*, 16(2), 253.
- Grantz, D. A.; Garner, J. H. B.; Johnson, D. W. (2003): Ecological effects of particulate matter. *Environment international*, 29(2-3), 213-239.
- Guida, G.; Sellami, M. H.; Mistretta, C.; Oliva, M.; Buonomo, R.; De Mascellis, R.; Giorio, P. (2017): Agronomical, physiological and fruit quality responses of two Italian long-storage tomato landraces under rain-fed and full irrigation conditions. *Agricultural Water Management*, 180, 126-135.
- Gullett, B. K.; Lemieux, P. M.; Lutes, C. C.; Winterrowd, C. K.; Winters, D. L. (2001): Emissions of PCDD/F from uncontrolled, domestic waste burning. *Chemosphere*, 43(4-7), 721-725.
- Hama, S.; Kumar, P.; Alam, M. S.; Rooney, D. J.; Bloss, W. J.; Shi, Z.; Gupta, S. K. (2021): Chemical source profiles of fine particles for five different sources in Delhi. *Chemosphere*, 274, 129913.
- He, J.; Fan, S.; Meng, Q.; Sun, Y.; Zhang, J.; Zu, F. (2014): Polycyclic aromatic hydrocarbons (PAHs) associated with fine particulate matters in Nanjing, China: Distributions, sources, and meteorological influences. *Atmospheric Environment*, 89, 207-215.
- Henry, G.; Tucker, K.; McCurdy, J. (2023): Response of common wildflower species to postemergence herbicides. *HortScience*, 58(6), 660-665.
- Hoffer, A.; Jancsek-Turóczi, B.; Tóth, Á.; Kiss, G.; Naghiu, A.; Levei, E. A.; Gelencsér, A. (2020): Emission factors for PM 10 and polycyclic aromatic hydrocarbons (PAHs) from illegal burning of different types of municipal waste in households. *Atmospheric Chemistry and Physics*, 20(24), 16135-16144.

Honour, S.L.; Bell, J. N. B.; Ashenden, T. W.; Cape, J. N.; Power, S. A. (2009): Responses of herbaceous plants to urban air pollution: Effects on growth, phenology and leaf surface characteristics. *Environmental Pollution* 157, 1279–1286.

Hough, R.L.; Breward, N.; Young, S.D.; Crout, N.M.J.; Tye, A.M.; Moir, A.M.; Thornton, I. (2004): Assessing potential risk of heavy metal exposure from consumption of home-produced vegetables by urban populations. *Environmental Health Perspectives* 112, 215–221.

Hu, W.; Huang, B.; Shi, X.; Chen, W.; Zhao, Y.; Jiao, W. (2013): Accumulation and health risk of heavy metals in a plot-scale vegetable production system in a peri-urban vegetable farm near Nanjing, China. *Ecotoxicology Environmental Safety* 98, 303–309.

Huang, S.; Dai, C.; Zhou, Y.; Peng, H.; Yi, K.; Qin, P.; Zhang, X. (2018): Comparisons of three plant species in accumulating polycyclic aromatic hydrocarbons (PAHs) from the atmosphere: A review. *Environmental Science and Pollution Research*, 25, 16548-16566.

Hubai, K.; Kováts, N.; Eck-Varanka, B.; Tumurbaatar, S.; Teke, G. (2024): Accumulation of Atmospheric PAHs in White Mustard–Can the Seeds Be Affected?. *Bulletin of Environmental Contamination and Toxicology*, 112(5), 76.

Hubai, K.; Kováts, N.; Sainnokhoi, T. A.; Eck-Varanka, B.; Hoffer, A. (2022): Phytotoxicity of particulate matter from controlled burning of different plastic waste types. *Bulletin of Environmental Contamination and Toxicology*. 109(5), 852-858.

Hubai, K.; Kováts, N.; Sainnokhoi, T. A.; Teke, G. (2022): Accumulation pattern of polycyclic aromatic hydrocarbons using *Plantago lanceolata* L. as passive biomonitor. *Environmental Science and Pollution Research*, 29(5), 7300-7311.

Hubai, K.; Kováts, N.; Teke, G. (2021): Effects of urban atmospheric particulate matter on higher plants using *Lycopersicon esculentum* as model species. *SN Applied Sciences*, 3(9), 770.

Hubai, K.; Székely, O.; Teke, G.; Kováts, N. (2021): Is essential oil production influenced by air pollution in *Ocimum basilicum* L.?. *Biochemical Systematics and Ecology* 96, 104248.

Huelster, A.; Mueller, J. F.; Marschner, H. (1994): Soil-plant transfer of polychlorinated dibenzo-p-dioxins and dibenzofurans to vegetables of the cucumber family (Cucurbitaceae). *Environmental science & technology*, 28(6), 1110-1115.

IARC Working Group on the Evaluation of Carcinogenic Risks to Humans (2014): Diesel and Gasoline Engine Exhausts and Some Nitroarenes. IARC Monographs on the Evaluation of Carcinogenic Risks to Humans, 105, 9.

Imberty, A.; Goldberg, R.; Catesson, A. M. (1984): Tetramethylbenzidine and p-phenylenediamine-pyrocatechol for peroxidase histochemistry and biochemistry: two new, non-carcinogenic chromogens for investigating lignification process. *Plant Science Letters*, 35(2), 103-108.

Inam, E.; Ibanga, F.; Essien, J. (2016): Bioaccumulation and cancer risk of polycyclic aromatic hydrocarbons in leafy vegetables grown in soils within automobile repair complex and environ in Uyo, Nigeria. *Environmental Monitoring and Assessment*, 188, 1-9.

Jaja, E. T.; Odoemena, C. S. I. (2004): Effect of Pb, Cu and Fe compounds on the germination and early seedling growth of tomato varieties. *Journal of Applied Sciences & Environmental management*, 8 (2) 51 - 53

Jajoo, A. (2017): Effects of environmental pollutants polycyclic aromatic hydrocarbons (PAH) on photosynthetic processes. *Photosynthesis: structures, mechanisms, and applications*, 249-259.

Jaskulak, M.; Rorat, A.; Grobelak, A.; Kacprzak, M. (2018): Antioxidative enzymes and expression of *rbcL* gene as tools to monitor heavy metal-related stress in plants. *Journal of Environmental Management*, 218: 71-78.

Jia, J.; Bi, C.; Jin, X.; Zeng, Y.; Deng, L.; Wang, X.; Chen, Z. (2021): Uptake, translocation, and risk assessment of PAHs in contaminated soil-air-vegetable systems based on a field simulation experiment. *Environmental Pollution*, 271, 116361.

Jia, J.; Bi, C.; Zhang, J.; Jin, X.; Chen, Z. (2018): Characterization of polycyclic aromatic hydrocarbons (PAHs) in vegetables near industrial areas of Shanghai, China: Sources, exposure, and cancer risk. *Environmental Pollution*, 241, 750-758.

Jiao, S.; Hou, X.; Zhao, G.; Feng, Y.; Zhang, S.; Zhang, H.; Jiang, G. (2023): Migration of polycyclic aromatic hydrocarbons in the rhizosphere micro-interface of soil-ryegrass (*Lolium perenne* L.) system. *Science of The Total Environment*, 903, 166299.

- Jonidi Jafari, A.; Charkhloo, E.; Pasalari, H. (2021): Urban air pollution control policies and strategies: a systematic review. *Journal of Environmental Health Science and Engineering*, 19, 1911-1940.
- Joshi, N.; Chauhan, A.; Joshi, P. C. (2009): Impact of industrial air pollutants on some biochemical parameters and yield in wheat and mustard plants. *The Environmentalist*, 29, 398-404.
- Joshi, P. C.; Swami, A. (2009): Air pollution induced changes in the photosynthetic pigments of selected plant species. *Journal of Environmental Biology*, 30(2), 295-298.
- Junior, S. F. S.; da Silva, E. O.; Mannarino, C. F.; Correia, F. V.; Saggiaro, E. M. (2024): A comprehensive overview on solid waste leachate effects on terrestrial organisms. *Science of The Total Environment*, 915, 170083.
- Kacálková, L.; Tlustoš, P. (2011): The uptake of persistent organic pollutants by plants. *Central European Journal of Biology*, 6(2), 223–235.
- Kakareka, S. V.; Kukharchyk, T. I. (2003): PAH emission from the open burning of agricultural debris. *Science of the Total Environment*, 308(1-3), 257-261.
- Kardel, F.; Wuyts, K.; Babanezhad, M.; Wuytack, T.; Potters, G.; Samson, R. (2010): Assessing urban habitat quality based on specific leaf area and stomatal characteristics of *Plantago lanceolata* L. *Environmental Pollution*, 158(3), 788-794.
- Karmakar, D.; Padhy, P. K. (2019): Air pollution tolerance, anticipated performance, and metal accumulation indices of plant species for greenbelt development in urban industrial area. *Chemosphere*, 237, 124522.
- Kaur, M.; Nagpal, A. K. (2017): Evaluation of air pollution tolerance index and anticipated performance index of plants and their application in development of green space along the urban areas. *Environmental science and pollution research*, 24, 18881-18895.
- Kaya, I.; Gülser, F. (2018): Determining heavy metal contents of hollyhock (*Alcea rosea* L.) in roadside soils of a Turkish lake basin. *Polish Journal of Environmental Studies*, 27(5).
- Keith, L. H. (2015): The Source of U.S. EPA's Sixteen PAH Priority Pollutants. *Polycyclic Aromatic Compounds*, 35(2-4), 147–160.

- Keller, T. (1974): The use of peroxidase activity for monitoring and mapping air pollution areas. *European Journal of Forest Pathology*, 4(1), 11-19.
- Kelly, F. J.; Fussell, J. C. (2012): Size, source and chemical composition as determinants of toxicity attributable to ambient particulate matter. *Atmospheric Environment* 60, 504-526.
- Khan, A. A.; Wang, T.; Nisa, Z. U.; Alnusairi, G. S.; Shi, F. (2022): Insights into cadmium-induced morphophysiological disorders in *Althea rosea* Cavan and its phytoremediation through the exogeneous citric acid. *Agronomy*, 12(11), 2776.
- Khan, A. H. A.; Kiyani, A.; Mirza, C. R.; Butt, T. A.; Barros, R.; Ali, B.; Yousaf, S. (2021): Ornamental plants for the phytoremediation of heavy metals: Present knowledge and future perspectives. *Environmental Research*, 195, 110780.
- Khillare, P. S.; Jyethi, D. S.; Sarkar, S. (2012): Health risk assessment of polycyclic aromatic hydrocarbons and heavy metals via dietary intake of vegetables grown in the vicinity of thermal power plants. *Food and Chemical Toxicology*, 50(5), 1642–1652.
- Kim, K. H.; Jahan, S. A.; Kabir, E.; Brown, R. J. (2013): A review of airborne polycyclic aromatic hydrocarbons (PAHs) and their human health effects. *Environment International*, 60, 71-80.
- Kipopoulou, A.M.; Manoli, E.; Samara, C. (1999): Bioconcentration of polycyclic aromatic hydrocarbons in vegetables grown in an industrial area. *Environmental Pollution*, 106, 369-380.
- Kolawole, T. O.; Iyiola, O.; Ibrahim, H.; Isibor, R. A. (2023): Contamination, ecological and health risk assessments of potentially toxic elements in soil around a municipal solid waste disposal facility in Southwestern Nigeria. *Journal of Trace Elements and Minerals*, 5, 100083.
- Kováts, N.; Fábrián, V. A.; Hubai, K.; Diósi, D.; Sainnokhoi, T. A.; Békéssy, Z.; Teke, G. (2020): Seasonal differences in rural particulate matter ecotoxicity. *Aerosol Science and Engineering*, 4, 169-177.
- Kováts, N.; Horváth, E.; Eck-Varanka, B.; Csajbók, E.; Hoffer, A. (2017): Adapting the Vegetative Vigour Terrestrial Plant Test for assessing ecotoxicity of aerosol samples. *Environmental Science and Pollution Research*, 24, 15291–15298.

Kováts, N.; Hubai, K.; Diósi, D.; Hoffer, A.; Teke, G. (2022): Foliar uptake and accumulation of polycyclic aromatic hydrocarbons from diesel emissions. *Polycyclic Aromatic Compounds*, 42(9), 6124-6135.

Kováts, N.; Hubai, K.; Diósi, D.; Sainnokhoi, T. A.; Hoffer, A. (2021): Sensitivity of typical European roadside plants to atmospheric particulate matter. *Ecological Indicators*, 124, 107428.

Kováts, N.; Hubai, K.; Sainnokhoi, T. A.; Teke, G. (2021): Biomonitoring of polyaromatic hydrocarbon accumulation in rural gardens using lettuce plants. *Journal of Soils and Sediments*, 21, 106-117.

Kováts, N.; Hubai, K.; Sainnokhoi, T.A.; Eck-Varanka, B.; Hoffer, A.; Tóth, Á.; Kakasi, B.; Teke, G. (2022): Ecotoxic emissions generated by illegal burning of household waste. *Chemosphere*, 298,134263.

Kováts, N.; Hubai, K.; Sainnokhoi, T.A.; Eck-Varanka, B.; Hoffer, A.; Tóth, Á.; Teke, G. (2023): Ecotoxicity of PM10 emissions generated during controlled burning of waste PET. *Environmental Toxicology and Pharmacology*, 04118.

Krauss, M.; Wilcke, W.; Martius, C.; Bandeira, A.G.; Garcia, M. V., & Amelung, W. (2005). Atmospheric versus biological sources of polycyclic aromatic hydrocarbons (PAHs) in a tropical rain forest environment. *Environmental Pollution*, 135(1), 143-154.

Krecl, P.; de Lima, C. H.; Bosco, T. C. D.; Targino, A. C.; Hashimoto, E. M.; Oukawa, G. Y. (2021): Open waste burning causes fast and sharp changes in particulate concentrations in peripheral neighborhoods. *Science of the Total Environment*, 765, 142736.

Kreslavski, V. D.; Brestic, M.; Zharmukhamedov, S. K.; Lyubimov, V. Y.; Lankin, A. V. (2017): Mechanisms of inhibitory effects of polycyclic aromatic hydrocarbons in photosynthetic primary processes in pea leaves and thylakoid preparations. *Plant Biology*, 19(5), 683-688.

Kucerova, R.; Zavoral, M.; Mudrunka, J.; Takac, D.; Marcalikova, L. (2023): Analysis of Firewater Samples from Simulated Fires in Illegal Waste Dumps. *Engineering Proceedings*, 57(1), 14.

Kulshreshtha, K.; Rai, A.; Mohanty, C. S.; Roy, R. K.; Sharma, S. C. (2009): Particulate pollution mitigating ability of some plant species. *International Journal of Environmental Research*, 3(1), 137-142.

Kumar, G. H.; Saritha, K. V. (2014): Lead stress-induced changes of antioxidant enzymes and Biochemical Compounds in Selected Weeds and their contribution for phytoremediation. *International Journal of Biological Sciences*, 1(2), 9-17

Kumar, S.; Aggarwal, S. G.; Gupta, P. K.; Kawamura, K. (2015): Investigation of the tracers for plastic-enriched waste burning aerosols. *Atmospheric Environment*, 108, 49-58.

Kumar, S.; Aggarwal, S. G.; Sarangi, B.; Malherbe, J.; Barre, J. P.; Berail, S.; Donard, O. F. (2018): Understanding the influence of open-waste burning on urban aerosols using metal tracers and lead isotopic composition. *Aerosol and Air Quality Research*, 18(9), 2433-2446.

Kumar, S.; Krzyszcak, A.; Dybowski, M.; Joško, I.; Kusiak, M.; Sikora, M.; Czech, B. (2022): The antioxidant defense responses of *Hordeum vulgare* L. to polycyclic aromatic hydrocarbons and their derivatives in biochar-amended soil. *Environmental Pollution*, 294, 118664.

Kwarteng, L.; Devasurendra, A. M.; Laskaris, Z.; Arko-Mensah, J.; Nti, A. A. A.; Takyi, S.; Batterman, S. (2022): Occupational exposures to particulate matter and PM<sub>2.5</sub>-associated polycyclic aromatic hydrocarbons at the Agbogbloshie waste recycling site in Ghana. *Environment international*, 158, 106971.

Landi, S.; De Lillo, A.; Nurcato, R.; Grillo, S.; Esposito, S. (2017): In-field study on traditional Italian tomato landraces: The constitutive activation of the ROS scavenging machinery reduces effects of drought stress. *Plant physiology and biochemistry*, 118, 150-160.

Lawal, O. J.; Ayodele, A. E.; Chukwuka, K. S. (2007): Morphological studies in *Lycopersicon esculentum* Mill. lines in Southwestern Nigeria. *Journal of Biological Sciences*, 7(5), 737-744.

Lehndorff, E.; Schwark, L. (2004): Biomonitoring of air quality in the Cologne Conurbation using pine needles as a passive sampler—part II: polycyclic aromatic hydrocarbons (PAH). *Atmospheric Environment*, 38(23), 3793–3808.

- Li, X.; Shen, X.; Jiang, W.; Xi, Y.; Li, S. (2024): Comprehensive review of emerging contaminants: Detection technologies, environmental impact, and management strategies. *Ecotoxicology and Environmental Safety*, 278, 116420.
- Liu, H.; Weisman, D.; Ye, Y.; Cui, B.; Huang, Y. (2009): An oxidative stress response to polycyclic aromatic hydrocarbon exposure is rapid and complex in *Arabidopsis thaliana*. *Plant Science*, 176, 375–382.
- Liu, J. N.; Zhou, Q. X.; Sun, T.; Ma, L. Q.; Wang, S. (2008a): Growth responses of three ornamental plants to Cd and Cd–Pb stress and their metal accumulation characteristics. *Journal of hazardous materials*, 151(1), 261-267.
- Liu, J. N.; Zhou, Q. X.; Sun, T.; Ma, L. Q.; Wang, S. (2008b): Identification and chemical enhancement of two ornamental plants for phytoremediation. *Bulletin of Environmental Contamination and Toxicology*, 80, 260-265.
- Liu, J.; Jia, J.; Grathwohl, P. (2022): Dilution of concentrations of PAHs from atmospheric particles, bulk deposition to soil: a review. *Environmental Geochemistry and Health*, 44(12), 4219–4234.
- Liu, J.; Xin, X.; Zhou, Q. (2018): Phytoremediation of contaminated soils using ornamental plants. *Environmental Reviews*, 26(1), 43-54.
- Liu, R.; Jadeja, R. N.; Zhou, Q.; Liu, Z. (2012): Treatment and remediation of petroleum-contaminated soils using selective ornamental plants. *Environmental engineering science*, 29(6), 494-501.
- Liu, X.; Gu, S.; Yang, S.; Deng, J.; Xu, J. (2021): Heavy metals in soil-vegetable system around E-waste site and the health risk assessment. *Science of the Total Environment*, 779, 146438.
- Liu, Z.; Chen, M.; Lin, M.; Chen, Q.; Lu, Q.; Yao, J.; He, X. (2022): Cadmium uptake and growth responses of seven urban flowering plants: hyperaccumulator or bioindicator?. *Sustainability*, 14(2), 619.
- Long, X. X.; Yang, X. E.; Ni, W. Z.; Ye, Z. Q.; He, Z. L.; Calvert, D. V.; Stoffella, J. P. (2003): Assessing zinc thresholds for phytotoxicity and potential dietary toxicity in selected vegetable crops. *Communications in Soil Science and Plant Analysis*, 34(9-10), 1421-1434.

Lucas, J. A.; Gutierrez-Albanchez, E.; Alfaya, T.; Feo-Brito, F.; Gutiérrez-Mañero, F. J. (2019): Oxidative stress in ryegrass growing under different air pollution levels and its likely effects on pollen allergenicity. *Plant Physiology and Biochemistry*, 135, 331-340.

Ma, J.; Tang, X.; Sun, B.; Wei, J.; Ma, L.; Yuan, M.; Jiang, L. (2022): A NAC transcription factor, TaNAC5D-2, acts as a positive regulator of drought tolerance through regulating water loss in wheat (*Triticum aestivum* L.). *Environmental and Experimental Botany*, 196, 104805.

Matthias, V.; Bieser, J.; Mocanu, T.; Pregger, T.; Quante, M. (2020): Modelling road transport emissions in Germany—Current day situation and scenarios for 2040. *Transport Research part D: Transport and Environment*, 87, 102536.

Mendes da Silva, L.; Andrade-Vieira, L. F. (2025): Ecotoxicological bioassays with terrestrial plants: a holistic view of standards, guidelines, and protocols. *Journal of Toxicology and Environmental Health, Part B*, 28(3), 183-221.

Mentes, D.; Tóth, C. E.; Nagy, G.; Muránszky, G.; Póliska, C. (2022): Investigation of gaseous and solid pollutants emitted from waste tire combustion at different temperatures. *Waste Management*, 149, 302-312.

Mitrović, P. M.; Stamenković, O. S.; Banković-Ilić, I.; Djalović, I. G.; Nježić, Z. B.; Farooq, M.; Veljković, V. B. (2020): White mustard (*Sinapis alba* L.) oil in biodiesel production: A review. *Frontiers in plant science*, 11, 299.

Mo, C.-H., Cai, Q.-Y., Tang, S.-R., Zeng, Q.-Y. Wu, Q.-T. (2009): "Polycyclic Aromatic Hydrocarbons and Phthalic Acid Esters in Vegetables from Nine Farms of the Pearl River Delta, South China. *Archives of Environmental Contamination and Toxicology* 56, 181–189.

Moeckel, C.; Nizzetto, L.; Strandberg, B.; Lindroth, A.; Jones, K. C. (2009): Air– Boreal forest transfer and processing of polychlorinated biphenyls. *Environmental Science & Technology*, 43(14), 5282-5289.

Molina Delgado, L.; Segura, A. (2021): Biochemical and metabolic plant responses toward polycyclic aromatic hydrocarbons and heavy metals present in atmospheric pollution. *Plants*, 10(11), 2305.

Montano, L.; Baldini, G. M.; Piscopo, M.; Liguori, G.; Lombardi, R.; Ricciardi, M.; Motta, O. (2025): Polycyclic Aromatic Hydrocarbons (PAHs) in the Environment: Occupational Exposure, Health Risks and Fertility Implications. *Toxics*, 13(3), 151.

Nadal, M.; Schuhmacher, M.; Domingo J. L. (2009): Levels of PAHs in soil and vegetation samples from Tarragona County, Spain.” *Environmental Pollution* 132 (2004): 1-11.

Nadgórska–Socha, A.; Kandziora-Ciupa, M.; Trzęsicki, M.; Barczyk, G. (2017): Air pollution tolerance index and heavy metal bioaccumulation in selected plant species from urban biotopes. *Chemosphere*, 183, 471-482.

Nikolaeva, O.; Rozanova, M.; Karpukhin, M. (2017): Distribution of traffic-related contaminants in urban topsoils across a highway in Moscow. *Journal of Soils and Sediments*, 17, 1045-1053.

Noblet, C.; Besombes, J. L.; Lemire, M.; Pin, M.; Jaffrezo, J. L.; Favez, O.; Albinet, A. (2021): Emission factors and chemical characterization of particulate emissions from garden green waste burning. *Science of the Total Environment*, 798, 149367.

Noumedem, J. A. K.; Djeussi, D. E.; Hritcu, L.; Mihasan, M.; Kuete, V. (2017): *Lactuca sativa*. Medicinal Spices and Vegetables from Africa; Academic Press: Cambridge, MA, USA, 437-449.

OECD. (2006): Test No. 227: Terrestrial Plant Test: Vegetative Vigour Test, OECD Guidelines for the Testing of Chemicals, Section 2, OECD Publishing, Paris.

Oguntimehin, I.; Eissa, F.; Sakugawa, H. (2010): Negative effects of fluoranthene on the ecophysiology of tomato plants (*Lycopersicon esculentum* Mill): Fluoranthene mists negatively affected tomato plants. *Chemosphere*, 78(7), 877-884.

Oguntimehin, I.; Sakugawa, H. (2008): Fluoranthene fumigation and exogenous scavenging of reactive oxygen intermediates (ROI) in evergreen Japanese red pine seedlings (*Pinus densiflora* Sieb. et. Zucc.). *Chemosphere*, 72(5), 747-754.

Ollivon, D.; Blanchoud, H.; Motelay-Massei, A.; Garban, B. (2002): Atmospheric deposition of PAHs to an urban site, Paris, France. *Atmospheric Environment*, 36(17), 2891-2900.

Paraíba, L. C.; Queiroz, S. C. N.; Maia, A. D. H. N.; Ferracini, V. L. (2010): Bioconcentration factor estimates of polycyclic aromatic hydrocarbons in grains of corn plants cultivated in soils treated with sewage sludge. *Science of the total environment*, 408(16), 3270-3276.

Paull, N. J.; Krix, D.; Irga, P. J.; Torpy, F. R. (2021): Green wall plant tolerance to ambient urban air pollution. *Urban Forestry & Urban Greening*, 63, 127201.

Pavlík, M.; Pavlíková, D.; Zemanová, V.; Hnilička, F.; Urbanová, V.; Száková, J. (2012): Trace elements present in airborne particulate matter—Stressors of plant metabolism. *Ecotoxicology and Environmental Safety*, 79, 101-107.

Peng, S.; Zhou, Q.; Cai, Z.; Zhang, Z. (2009): Phytoremediation of petroleum contaminated soils by *Mirabilis jalapa* L. in a greenhouse plot experiment. *Journal of hazardous materials*, 168(2-3), 1490-1496.

Peter, A. E.; Nagendra, S. S.; Nambi, I. M. (2018): Comprehensive analysis of inhalable toxic particulate emissions from an old municipal solid waste dumpsite and neighborhood health risks. *Atmospheric pollution research*, 9(6), 1021-1031.

Petriccione, M.; Di Patre, D.; Ferrante, P.; Papa, S.; Bartoli, G.; Fioretto, A.; Scortichini, M. (2013): Effects of *Pseudomonas fluorescens* seed bioinoculation on heavy metal accumulation for *Mirabilis jalapa* phytoextraction in smelter-contaminated soil. *Water, Air, & Soil Pollution*, 224, 1-17.

Popoola, L. T.; Adebajo, S. A.; Adeoye, B. K. (2018): Assessment of atmospheric particulate matter and heavy metals: a critical review. *International Journal of Environmental Science and Technology*, 15, 935-948.

Qi, A.; Wang, P.; Lv, J.; Zhao, T.; Huang, Q.; Wang, Y.; Wang, W. (2023): Distributions of PAHs, NPAHs, OPAHs, BrPAHs, and ClPAHs in air, bulk deposition, soil, and water in the Shandong Peninsula, China: Urban-rural gradient, interface exchange, and long-range transport. *Ecotoxicology and Environmental Safety*, 265, 115494.

Rabe, R.; Kreeb, K. H. (1979): Enzyme activities and chlorophyll and protein content in plants as indicators of air pollution. *Environmental Pollution* 19, 119-137.

Rai, P. K. (2016): Impacts of particulate matter pollution on plants: Implications for environmental biomonitoring. *Ecotoxicology and environmental safety*, 129, 120-136.

Ramadan, B. S.; Rosmalina, R. T.; Syafrudin, Munawir, Khair, H.; Widiyanti, A.; Matsumoto, T. (2025): Potential exposure to metals-bound particulate during open burning of different waste materials. *Environmental Monitoring and Assessment*, 197(4), 495.

Ramesh, A.; Walker, S. A.; Hood, D.; Guill'en, M. D.; Schneider, K.; Weyand, E. H. (2004): Bioavailability and Risk Assessment of Orally Ingested Polycyclic Aromatic Hydrocarbons. *International Journal of Toxicology*, 23(5), 301–333.

Rienda, I. C.; Nunes, T.; Gonçalves, C.; Vicente, A.; Amato, F.; Lucarelli, F.; Alves, C. A. (2023): Road dust resuspension in a coastal Atlantic intermunicipal urban area with industrial facilities: Emission factors, chemical composition and ecotoxicity. *Atmospheric Research*, 294, 106977.

Rigane, G.; Younes, S. B.; Ghazghazi, H.; Salem, R. B. (2013): Investigation into the biological activities and chemical composition of *Calendula officinalis* L. growing in Tunisia. *International Food Research Journal*, 20(6), 3001.

Rozina, R. (2016): Pharmacological and biological activities of *Mirabilis jalapa* L. *International Journal of Pharmacology Research*, 6(5), 160-8.

Sadeghi, A.; Razmjoo, J.; Karimmojeni, H.; Baldwin, T. C. (2024): Differential responses of Hollyhock (*Alcea rosea* L.) varieties to salt stress in relation to physiological and biochemical parameters. *Scientific Reports*, 14(1), 8105.

Saleem, M.; Ahmad, M. N.; Khan, B. A.; Zia, A.; Ahmad, S. S.; Shah, H. U.; Qazi, M. I. (2015): Effects of soil fluoride on the growth and quality of two tomato varieties grown in Peshawar, Pakistan. *Fluoride*, 48(2), 174-178.

Samek, L.; Furman, L.; Mikrut, M.; Regiel-Futyra, A.; Macyk, W. (2017): Chemical composition of submicron and fine particulate matter collected in Krakow, Poland. Consequences for the APARIC project. *Chemosphere* 187, 430-439.

Sampaio, G. R.; Guizellini, G. M.; da Silva, S. A.; de Almeida, A. P.; Pinaffi-Langley, A. C. C.; Rogero, M. M.; Torres, E. A. (2021): Polycyclic aromatic hydrocarbons in foods: Biological effects, legislation, occurrence, analytical methods, and strategies to reduce their formation. *International Journal of Molecular Sciences*, 22(11), 6010.

Schmidl, C.; Bauer, H.; Dattler, A.; Hitzemberger, R.; Weissenboeck, G.; Marr, I.L.; Puxbaum, H. (2008): Chemical characterisation of particle emissions from burning leaves. *Atmospheric Environment*, 42(40), 9070–9079.

Schreck, E., Bonnard, R., Laplanche, C., Leveque, T., Foucault, Y., Dumat, C. (2012). DECA: a new model for assessing the foliar uptake of atmospheric lead by vegetation, using *Lactuca sativa* as an example. *Journal of Environmental Management*, 112, 233-239.

Schreck, E.; Laplanche, C.; Le Guédard, M.; Bessoule, J. J.; Austruy, A.; Xiong, T.; Dumat, C. (2013): Influence of fine process particles enriched with metals and metalloids on *Lactuca sativa* L. leaf fatty acid composition following air and/or soil-plant field exposure. *Environmental pollution*, 179, 242-249.

Şevik, H.; Çetin, M.; Yiğit, N.; Türkyılmaz, A.; Canbulat, S.; Belkayalı, N.; Kuşcu, İ. S. K. (2017): Determination of Plant Leaf Micro-Morphological Characters Depending on Traffic Density: Case Study of *Pyracantha coccinea*. *Journal of International Environmental Application and Science*, 12(3), 212-216.

Shahid, M.; Natasha, Dumat, C.; Niazi, N. K.; Xiong, T. T.; Farooq, A. B. U.; Khalid, S. (2021): Ecotoxicology of heavy metal (loid)-enriched particulate matter: foliar accumulation by plants and health impacts. *Reviews of Environmental Contamination and Toxicology Volume*, 253, 65-113.

Shakya, P. R.; Shrestha, P.; Tamrakar, C. S.; Bhattarai, P. K. (2006): Studies and determination of heavy metals in waste tyres and their impacts on the environment. *Pakistan Journal of Analytical and Environmental Chemistry*, 7(2), 70-76.

Sidhu, S.; Gullett, B.; Striebich, R.; Klosterman, J.; Contreras, J.; DeVito, M. (2005): Endocrine disrupting chemical emissions from combustion sources: diesel particulate emissions and domestic waste open burn emissions. *Atmospheric Environment*, 39(5), 801-811.

Singh, A. K.; Kumar, M.; Baudh, K.; Singh, A.; Singh, P.; Madhav, S.; Shukla, S. K. (2023): Environmental impacts of air pollution and its abatement by plant species: A comprehensive review. *Environmental Science and Pollution Research*, 30(33), 79587-79616.

Smith, K. E. C.; Jones, K. C. (2000): Particles and vegetation: implications for the transfer of particle-bound organic contaminants to vegetation. *Science of the Total Environment*, 246, 207-236.

Song, S.; Zhou, X.; Guo, C.; Zhang, H.; Zeng, T.; Xie, Y.; Sun, X. (2019): Emission characteristics of polychlorinated, polybrominated and mixed polybrominated/chlorinated dibenzo-p-dioxins and dibenzofurans (PCDD/Fs, PBDD/Fs, and PBCDD/Fs) from waste incineration and metallurgical processes in China. *Ecotoxicology and Environmental Safety*, 184, 109608.

Srogi, K. (2007): Monitoring of environmental exposure to polycyclic aromatic hydrocarbons: a review. *Environmental Chemistry Letters*, 5, 169-195.

Storch-Böhm, R. F.; Somensi, C. A.; Cotellet, S.; Deomar-Simões, M. J.; Poyer-Radetski, L.; Dalpiaz, F. L.; Radetski, C. M. (2020): Sensitivity of different parameters for selection of higher plants in urban afforestation: Exposure of Guabiroba (*Campomanesia xanthocarpa* O. Berg.) to diesel engine exhaust. *Environmental Pollution*, 265, 114675.

Storch-Böhm, R. F.; Somensi, C. A.; Testolin, R. C.; Rossa, Ü. B.; Corrêa, R.; Ariento-Neto, R.; Radetski, C. M. (2022): Urban afforestation: using phytotoxicity endpoints to compare air pollution tolerance of two native Brazilian plants Aroeira (*Schinus terebinthifolius*) and Cuvatã (*Cupania vernalis*). *Environmental Science and Pollution Research*, 29(37), 56579-56591.

Süle, G.; Miholcsa, Z.; Molnár, C.; Kovács-Hostyánszki, A.; Fenesi, A.; Bauer, N.; Szigeti, V. (2023): Escape from the garden: Spreading, effects and traits of a new risky invasive ornamental plant (*Gaillardia aristata* Pursh). *NeoBiota*, 83.

Szakiel, A.; Ruszkowski, D.; Janiszowska, W. (2005): Saponins in *Calendula officinalis* L.—structure, biosynthesis, transport and biological activity. *Phytochemistry Reviews*, 4, 151-158.

Tabassum, S.; Shahid, N.; Shafiq, M.; Mumtaz, M.; Arslan, M. (2016): The Oxidative Stress Response of *Mirabilis jalapa* to Exhausted Engine Oil (EEO) during Phytoremediation. *Polish Journal of Environmental Studies*, 25(6).

Tamez, C.; Hernandez-Molina, M.; Hernandez-Viezcas, J. A.; Gardea-Torresdey, J. L. (2019): Uptake, transport, and effects of nano-copper exposure in zucchini (*Cucurbita pepo*). *Science of The Total Environment*, 665, 100-106.

Tao, S.; Jiao, X. C.; Chen, S. H.; Liu, W. X.; Coveney Jr, R. M.; Zhu, L. Z.; Luo, Y. M. (2006): Accumulation and distribution of polycyclic aromatic hydrocarbons in rice (*Oryza sativa*). *Environmental Pollution*, 140(3), 406-415.

Teke, G.; Hubai, K.; Diósi, D.; Kováts, N. (2020): Assessment of foliar uptake and accumulation of airborne polyaromatic hydrocarbons under laboratory conditions. *Bulletin of Environmental Contamination and Toxicology*, 104, 444-448.

Terekhova, V. A.; Prudnikova, E. V.; Kiryushina, A. P.; Karpukhin, M. M.; Plekhanova, I. O.; Yakimenko, O. S. (2021): Phytotoxicity of heavy metals in contaminated podzolic soils of different fertility levels. *Eurasian Soil Science*, 54(6), 964-974.

Tesi, G. O.; Iniaghe, P. O.; Lari, B.; Obi-Iyeke, G.; Ossai, J. C. (2021): Polycyclic aromatic hydrocarbons (PAHs) in leafy vegetables consumed in southern Nigeria: concentration, risk assessment and source apportionment. *Environmental Monitoring and Assessment*, 193(7), 443.

Tomar, R. S.; Jajoo, A. (2015): Photomodified fluoranthene exerts more harmful effects as compared to intact fluoranthene by inhibiting growth and photosynthetic processes in wheat. *Ecotoxicology and Environmental Safety*, 122, 31–36.

Tomar, R.S.; Singh, B.; Jajoo, A. (2019): Effects of organic pollutants on photosynthesis. *Photosynthesis, productivity and environmental stress*, 1-26.

Triassi, M.; Alfano, R.; Illario, M.; Nardone, A.; Caporale, O.; Montuori, P. (2015): Environmental pollution from illegal waste disposal and health effects: A review on the “Triangle of Death”. *International Journal of Environmental Research and Public Health*, 12(2), 1216-1236.

Turner, A.; Filella, M. (2020): The influence of additives on the fate of plastics in the marine environment, exemplified with barium sulphate. *Marine Pollution Bulletin*, 158, 111352.

Uka, U. N.; Belford, E. J.; & Elebe, F. A. (2021). Effects of road traffic on photosynthetic pigments and heavy metal accumulation in tree species of Kumasi Metropolis, Ghana. *SN Applied Sciences*, 3, 1-12.

Valavanidis, A.; Iliopoulos, N.; Gotsis, G.; Fiotakis, K. (2008): Persistent free radicals, heavy metals and PAHs generated in particulate soot emissions and residue ash from controlled combustion of common types of plastic. *Journal of Hazardous Materials*, 156(1-3), 277-284.

Vaverková, M. D.; Maxianová, A.; Winkler, J.; Adamcová, D.; Podlasek, A. (2019): Environmental consequences and the role of illegal waste dumps and their impact on land degradation. *Land Use Policy*, 89, 104234.

Voutsas, D.; Grimanis, A.; Samara, C. (1996): Industrial area in relation to soil and air particulate matter. *Environmental Pollution*, 94(3), 325-335.

Vreeland, H.; Schauer, J. J.; Russell, A. G.; Marshall, J. D.; Fushimi, A.; Jain, G.; Bergin, M. H. (2016): Chemical characterization and toxicity of particulate matter emissions from roadside trash combustion in urban India. *Atmospheric Environment*, 147, 22-30.

Wang, J.; Ye, S.; Xue, S.; Hartley, W.; Wu, H.; Shi, L. (2018): The physiological response of *Mirabilis jalapa* Linn. to lead stress and accumulation. *International Biodeterioration & Biodegradation*, 128, 11-14.

Wang, S.; Liu, J. (2014): The effectiveness and risk comparison of EDTA with EGTA in enhancing Cd phytoextraction by *Mirabilis jalapa* L. *Environmental monitoring and assessment*, 186, 751-759.

Wang, X. S. (2013): Polycyclic aromatic hydrocarbons (PAHs) in particle-size fractions of urban topsoils. *Environmental earth sciences*, 70(6), 2855-2864.

Wang, X.; Firouzkouhi, H.; Chow, J. C.; Watson, J. G.; Carter, W.; De Vos, A. S. (2023): Characterization of gas and particle emissions from open burning of household solid waste from South Africa. *Atmospheric Chemistry and Physics*, 23(15), 8921-8937

Wang, Y.; Cheng, K.; Wu, W.; Tian, H.; Yi, P.; Zhi, G.; Liu, S. (2017): Atmospheric emissions of typical toxic heavy metals from open burning of municipal solid waste in China. *Atmospheric Environment*, 152, 6-15.

Wang, Y.; Zhao, S.; Chen, P.; Liu, Y.; Ma, Z.; Malik, W. A.; Chang, Y. (2023): Genetic diversity and population structure analysis of Hollyhock (*Alcea rosea* Cavan) using high-throughput sequencing. *Horticulturae*, 9(6), 662.

- Wei, H.; Song, S.; Tian, H.; Liu, T. (2014): Effects of phenanthrene on seed germination and some physiological activities of wheat seedling. *C. R. Biologies* 337, 95–100.
- Wieczorek, J. K.; Wieczorek, Z. J. (2007): Phytotoxicity and accumulation of anthracene applied to the foliage and sandy substrate in lettuce and radish plants. *Ecotoxicology and environmental safety*, 66(3), 369-377.
- Wieczorek, J.; Sienkiewicz, S.; Pietrzak, M.; Wieczorek, Z. (2015): Uptake and phytotoxicity of anthracene and benzo [k] fluoranthene applied to the leaves of celery plants (*Apium graveolens var. secalinum* L.). *Ecotoxicology and Environmental Safety*, 115, 19-25.
- Wild, E.; Dent, J.; Thomas, G. O.; Jones, K. C. (2006): Visualizing the air-to-leaf transfer and within-leaf movement and distribution of phenanthrene: further studies utilizing two-photon excitation microscopy. *Environmental science & technology*, 40(3), 907-916.
- Wu, D.; Zhang, F.; Lou, W.; Li, D.; Chen, J. (2017): Chemical characterization and toxicity assessment of fine particulate matters emitted from the combustion of petrol and diesel fuels. *Science of the Total Environment*, 605, 172-179.
- Wu, M.; Luo, Q.; Liu, S.; Zhao, Y.; Long, Y.; Pan, Y. (2018): Screening ornamental plants to identify potential Cd hyperaccumulators for bioremediation. *Ecotoxicology and environmental safety*, 162, 35-41.
- Wu, Y.; Shi, Y.; Zhang, N.; Wang, Y.; Ren, Y. (2021): Pollution levels, characteristics, and sources of polycyclic aromatic hydrocarbons in atmospheric particulate matter across the Hu line in China. A review. *Environmental Chemistry Letters*, 19(5), 3821-3836.
- Wuytack, T.; Verheyen, K.; Wuyts, K.; Kardel, F.; Adriaenssens, S.; Samson, R. (2010): The potential of biomonitoring of air quality using leaf characteristics of white willow (*Salix alba* L.). *Environmental monitoring and assessment*, 171, 197-204.
- Xiong, G. N.; Zhang, Y. H.; Duan, Y. H.; Cai, C. Y.; Wang, X.; Li, J. Y. (2017): Uptake of PAHs by cabbage root and leaf in vegetable plots near a large coking manufacturer and associations with PAHs in cabbage core. *Environmental Science and Pollution Research*, 24(23), 18953–18965.

- Xu, J.; Wolters-Arts, M.; Mariani, C.; Huber, H.; Rieu, I. (2017): Heat stress affects vegetative and reproductive performance and trait correlations in tomato (*Solanum lycopersicum*). *Euphytica*, 213, 1-12.
- Yakovleva, E. V.; Beznosikov, V. A.; Kondratenok, B. M.; Gabov, D. N. (2012): Bioaccumulation of polycyclic aromatic hydrocarbons in the soil-plant systems of the northern-taiga biocenoses. *Eurasian Soil Science*, 45, 309-320
- Yang, L.; Zhang, H.; Zhang, X.; Xing, W.; Wang, Y.; Bai, P.; Tang, N. (2021): Exposure to atmospheric particulate matter-bound polycyclic aromatic hydrocarbons and their health effects: a review. *International journal of environmental research and public health*, 18(4), 2177.
- Zaynab, M.; Al-Yahyai, R.; Ameen, A.; Sharif, Y.; Ali, L.; Fatima, M.; Li, S. (2022): Health and environmental effects of heavy metals. *Journal of King Saud University-Science*, 34(1), 101653.
- Zdravković, J.; Jovanović, Z.; Đorđević, M.; Girek, Z.; Zdravković, M.; Stikić, R. (2013): Application of stress susceptibility index for drought tolerance screening of tomato populations. *Genetika*, 45(3), 679-689.
- Zelinkova, Z.; Wenzl, T. (2015): The Occurrence of 16 EPA PAHs in Food – A Review. *Polycyclic Aromatic Compounds*, 35, 2-4, 248-284.
- Zežulka, Š.; Kummerová, M.; Babula, P.; Váňová, L. (2013); Lemna minor exposed to fluoranthene: growth, biochemical, physiological and histochemical changes. *Aquatic Toxicology*, 140, 37-47.
- Zhang, B.; Jiao, L.; Xu, G.; Zhao, S.; Tang, X.; Zhou, Y.; Gong, C. (2018): Influences of wind and precipitation on different-sized particulate matter concentrations (PM 2.5, PM 10, PM 2.5–10). *Meteorology and Atmospheric Physics*, 130, 383-392.
- Zhang, F.; Zhang, G.; Liao, X. (2021): Negative role of biochars in the dissipation and vegetable uptake of polycyclic aromatic hydrocarbons (PAHs) in an agricultural soil: Cautions for application of biochars to remediate PAHs-contaminated soil. *Ecotoxicology and Environmental Safety*, 213, 112075.

Zhang, P. Q.; Liu, Y. J.; Chen, X.; Yang, Z.; Zhu, M. H.; Li, Y. P. (2016): Pollution resistance assessment of existing landscape plants on Beijing streets based on air pollution tolerance index method. *Ecotoxicology and Environmental Safety*, 132, 212-223.

Zhang, W.; Zhang, Y.; Gong, J.; Yang, B.; Zhang, Z.; Wang, B.; Yue, K. (2020): Comparison of the suitability of plant species for greenbelt construction based on particulate matter capture capacity, air pollution tolerance index, and antioxidant system. *Environmental Pollution*, 263, 114615.

Zhang, Y.; Hou, D.; Xiong, G.; Duan, Y.; Cai, C.; Wang, X.; Liu, W. (2018): Structural equation modeling of PAHs in ambient air, dust fall, soil, and cabbage in vegetable bases of Northern China. *Environmental Pollution*, 239, 13-20.

Zhao, X.; He, M.; Shang, H.; Yu, H.; Wang, H.; Li, H.; Li, D. (2018): Biomonitoring polycyclic aromatic hydrocarbons by *Salix matsudana* leaves: a comparison with the relevant air content and evaluation of environmental parameter effects. *Atmospheric Environment*, 181, 47-53.

Zhou, R.; Yu, X.; Ottosen, C. O.; Rosenqvist, E.; Zhao, L.; Wang, Y.; Wu, Z. (2017): Drought stress had a predominant effect over heat stress on three tomato cultivars subjected to combined stress. *BMC plant biology*, 17, 1-13.

Zhu, Y.; Li, Z.; Shen, J.; Wu, K.; Zhao, P.; Wu, Z.; Feng, R. (2022): Toxicity of different forms of antimony to rice plants: Photosynthetic electron transfer, gas exchange, photosynthetic efficiency, and carbon assimilation combined with metabolome analysis. *Journal of Hazardous materials*, 437, 129433.

## List of publications

### Papers related to the dissertation

Kováts, N.; Hubai, K.; Eck-Varanka, B.; **Tumurbaatar, S.**; Teke, G. (2024): Accumulation of atmospheric PAHs in white mustard – can the seeds be affected?. *Bulletin of Environmental Contamination and Toxicology*, 112, 76. DOI: <https://doi.org/10.1007/s00128-024-03895-w> (**impact factor: 2.7; Q2**)

**Tumurbaatar, S.**; Kováts, N.; Hubai, K. (2024): Phytotoxicity Testing of Atmospheric Polycyclic Aromatic Hydrocarbons. *Atmosphere*, 15(9), 1143. DOI: <https://doi.org/10.3390/atmos15091143> (**impact factor: 2.5; Q2**)

**Tumurbaatar, S.**; Kováts, N.; Eck-Varanka, B.; Teke, G.; Hubai, K. (2025): Assessing Risk of Emissions Generated During Illegal Waste Burning: Phytotoxicity and Bioaccumulation. *Atmosphere*, 16(2), 164. DOI: <https://doi.org/10.3390/atmos16020164> (**impact factor: 2.5; Q2**)

Hubai, K.; Eck-Varanka, B.; **Tumurbaatar, S.**; Teke, G.; Kováts, N. (2025): Contribution of Atmospheric Fallout to the Soil–Root–Leaf Transfer of PAHs in Higher Plants. *Sciences*, 15(8), 4407; DOI: <https://doi.org/10.3390/app15084407> (**impact factor: 2.5; Q2**)

**Tumurbaatar, S.**; Kováts, N.; Eck-Varanka, B.; Hubai, K. (2025): Combined effects of air pollution and drought stress in tomato landraces. Submitted to *Heliyon*.

Hubai, K.; Kováts, N.; **Tumurbaatar, S.**; Eck-Varanka, B.; Hoffer, A.; Tóth, Á.; Teke, G. (2025): Phytotoxicity testing of emissions from illegal waste burning based on the OECD Terrestrial Plant Test. To be submitted.

### Conference attendance

#### Poster

Can landraces better cope with environmental stress?

Authors: Nora Kováts, Katalin Hubai, Bettina Eck-Varanka, Selenge Tumurbaatar

Conference title: Closed cycles and the Circular Society 2023, The power of ecological engineering. Time: 1-5 October, 2023. Chania, Greece