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Research article

Plant biomass responses to PFAS exposure: A meta-analysis with implications for phytoremediation in terrestrial and aquatic systems

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ABSTRACT

Given the widespread distribution of per- and polyfluoroalkyl substances (PFAS) in the environment and their tendency to bioaccumulate in living organisms, it is crucial to understand PFAS-plant interactions. However, the literature is still highly fragmented. In this meta-analysis, we compiled a dataset comprising 776 observations from 68 studies, covering 12 perfluoroalkyl acids (PFAAs) and 45 plant species. Biomass was selected as a widely used and informative endpoint for assessing plant responses to PFAAs exposure, stratified into whole-plant (WB), aboveground (AB), and belowground (BB) biomass. PFAAs' phytotoxicity generally manifested as a reduction in WB and AB. However, the observed phenotype was heavily influenced by experimental conditions, particularly exposure concentration. Conversely, BB tended to increase in aquatic species, particularly *E. crassipes* and *B. articulata*, following exposure to PFAAs. The information provided in this meta-analysis can inform the development and optimisation of phytoremediation strategies for PFAS removal or sequestration, and, on the other hand, help identify more sensitive species for ecotoxicology studies addressing the negative effects of PFAS on plants.

1. Introduction

Per- and poly-fluoroalkyl substances (PFAS) are synthetic fluorinated compounds that include more than 7 million distinct chemicals (Schymanski et al., 2023). Perfluoroalkyl acids (PFAAs) are non-polymeric PFAS that comprise perfluorocarboxylic acids (PFCAs) and perfluorosulfonic acids (PFSAs), and they are among the most commonly used classes of PFAS (Brahana et al., 2024). The presence of the C-F bond confers PFAS their unique chemical and physical properties, including high stability, hydrophobicity, and oleophobicity, which have favoured their use in a wide range of consumer products (Glüge et al., 2020). PFAS are considered emerging contaminants and are of great concern due to their high environmental mobility, their ability to bioaccumulate in organisms, and their transfer along food chains (Brunn et al., 2023). They are ubiquitous in the environment and have been detected in soil, surface water, groundwater, rainwater, and air (Evich et al., 2022). Depending on the distance from potential sources of dispersion, including fluorochemical production plants and waste management sites (Gebbink and van Leeuwen, 2020; Hasnine et al., 2025), it is estimated that a significant proportion of European soil may exceed soil contamination levels of more than 5 μg kg^{-1} (Moghadasi et al., 2023), and prediction maps identify high levels of ecological risk in several countries in Central and Southwestern Europe, with surface water concentrations greater than 100 ng L^{-1} (Zhao et al., 2025a,b).

Phytoremediation is a cost-effective, nature-based, *in situ* remediation technique that can be used alongside chemical-physical approaches to mitigate PFAS contamination (Cornelissen et al., 2025). Having already been widely applied to remediate sites contaminated with heavy metals, phytoremediation relies on the ability of plants to accumulate, and potentially degrade or insolubilize, various types of contaminants in their tissues (Sharma et al., 2023). A variety of woody, herbaceous, and aquatic plant species have been tested for PFAS phytoremediation under a range of experimental conditions, from laboratory trials to pilot-scale and field applications (Marzi et al., 2025). In practical terms, the main parameter currently used to classify a plant's ability to effectively remove PFAS is the removal efficiency (R), which is the ratio of the mass of PFAS in the plant to the mass of PFAS in the matrix from which it is extracted (Marzi et al., 2025). For woody and herbaceous species, this parameter generally falls within the range of 10%, whereas much higher values (e.g., 40-60%) have been reported for aquatic species (Awad

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et al., 2022; Marzi et al., 2025; Phung et al., 2024). Two additional parameters commonly used in the context of phytoremediation are (1) the bioconcentration factor (BCF), defined as the ratio between the concentration of PFAS within plant tissue and (2) the concentration in the matrix, and the translocation factor (TF), defined as the ratio between PFAS concentration in aboveground and belowground tissues (Marzi et al., 2025).

Taken together, R, BCF, and TF provide complementary information on the PFAS mass balance within the plant-matrix system, describing both the overall mass of PFAS removed and how it is distributed among plant compartments. In fact, different plant species exhibit distinct tendencies to sequester PFAS in specific tissues (Xu et al., 2022a,b), and these differences can be relevant for designing effective phytoremediation strategies. Phytoextraction is a phytoremediation strategy that exploits the ability of plants to take up contaminants from soil and translocate them into above-ground tissues (Garbisu and Alkorta, 2001). It is generally assumed that the potential for removing contaminants from polluted sites increases as plant biomass increases (Schück and Greger, 2020a), and, following the accumulation of PFAS, the harvested plant biomass can subsequently be subjected to thermal degradation, such as pyrolysis and hydrothermal liquefaction (Nason et al., 2024; Sørmo et al., 2023).

Plants can also immobilize contaminants in roots and rhizosphere (phytostabilization), thereby reducing their mobility and bioavailability and limiting off-site dispersion (Bolan et al., 2011). Aquatic plants are particularly effective for phytostabilization-based strategies in aquatic ecosystems (e.g., rivers, lakes, wetlands, and ponds) (Huang et al., 2021), and this is especially relevant when considering that surface water and groundwater are major pathways for PFAS transport and dispersion in the environment (Ackerman Grunfeld et al., 2024). Their roots and rhizomes help stabilize contaminated sediments and reduce the remobilization of pollutants caused by water movement. Moreover, aquatic vegetation can immobilize contaminants within sediments or associated biofilms, decreasing their availability in the water column (Kochi et al., 2020). As for aboveground tissues in phytoextraction, increased root biomass is associated with greater phytostabilization efficiency (Schück and Greger, 2020b).

In addition to being a crucial parameter in the context of phytoremediation, biomass variation is a primary and straightforward indicator of the negative effects of contaminants on plants, and it can effectively differentiate between plant species with varying sensitivities to PFAS. A large body of literature has investigated plant-PFAS interactions to better understand the molecular mechanisms driving PFAS uptake, translocation, and accumulation across various plant species. Due to their specific chemical properties, PFAS classes exhibit distinct retention/adsorption tendencies in both water and soil media, where interactions with organic matter may reduce their bioavailability for plant uptake (Fu et al., 2025; Mei et al., 2021). PFAS root uptake can be mediated by aquaporins, ion channels, or carrier proteins, and they can be translocated to aerial tissue both by apoplastic or symplastic pathways (Goswami et al., 2025; Mei et al., 2021). The efficiency of this process is strongly influenced by PFAS chemical properties: short-chain PFAS and PFCAs generally display higher TF, whereas long-chain PFSAs tend to be retained in belowground tissues due to their greater hydrophobicity and stronger interactions with cell walls and lipid membranes (Felizeter et al., 2012; Xu et al., 2022a,b).

Importantly, PFAS accumulation in plant tissue can perturb gene transcription, protein synthesis, and cellular metabolism, and lead to phytotoxicity mainly through oxidative stress and potential genotoxic damage, but the evidence is highly fragmented and species-specific (Du et al., 2020; Fan et al., 2020; Li et al., 2021b; Luche et al., 2026; Wang et al., 2024). In addition to the direct growth inhibition associated with phytotoxicity, plants can modify the ratio of roots to aerial parts, as well as the relative distribution of resources in leaves, stem, and root system, as a plastic response to abiotic stress (Delerue et al., 2022; Ding et al., 2025; G. Li et al., 2024; Zhang et al., 2020). Thus, variation of plant biomass under abiotic stress conditions can be considered a universal and informative indicator, as it represent the result of the integration of physiological disturbances that are potentially induced by the accumulation of pollutants (Delerue et al., 2022; Zhang et al., 2020).

This meta-analysis aims to synthesize the available data on the biomass responses of plants exposed to PFAAs, with the goal to (i) discriminate between effects at the whole-plant level and across belowground and aboveground compartments, (ii) assess the contribution of key experimental variables (e.g., PFAA compound and experimental design), and (iii) compare responses among different plant species. Beyond identifying highly sensitive species potentially relevant to ecotoxicology and environmental monitoring, this work also highlights plant species showing promise for phytoremediation. Overall, the evidence gathered in this study provides practical insights to inform the design and interpretation of PFAS phytoremediation studies.

2. Material and method

2.1. Literature screening and data extraction

Potentially eligible studies were searched for in Embase, PubMed, and Scopus. The keywords used in each set, combined with the Boolean operator 'OR' within each set and 'AND' between the two sets, are reported in Table S1 in **Supporting Information-1** (**SI-1**). Studies of any year of publication were included. Only peer-reviewed research articles with full texts available in English were considered. The eligibility criteria (i.e., the inclusion and exclusion criteria) for the type of plant, exposure, comparators, outcome measures, and studies are reported in Table 1. The records (titles and abstracts) were screened with ASReview (van de Schoot et al., 2021) with the default parameters. The details for the systematic review procedure are presented in the PRISMA 2020 flow diagram (Fig. 1). The data from included studies ($n = 68$) were extracted from both the full text and supplementary information, and the extracted elements are reported in Table 2. The full dataset containing the coded data from the included articles is reported in Table S2 (**SI-1**).

All the PFAS concentrations were converted to mg kg^{-1}. The biomass means and standard deviations presented as graphs were extracted using WebPlotDigitizer (Rohatgi, 2025). If the type of dispersion was not specified, it was deemed to be standard deviation (SD). The standard errors (SEs) were converted to *SD* using $SD = SE^* \sqrt{n}$. The interquartile ranges (IQRs) were converted to *SD* using *SD*s = IQR/1.35 (Higgins

Table 1
Eligibility criteria.

	Inclusion criteria	Exclusion criteria
Types of plant	Any plant species exposed to PFAS	Transgenic plants, microalgae
Types of exposure	Exposure to known concentrations of PFCAs and/or PFSAs	Co-exposure with other pollutant; intentional inoculation with microorganisms; exposure to PFAS-contaminated media with no intentional supplementation
Types of comparators	Explicit biomass value of the untreated control (same growth stage, no exposure); if expressed as a percentage of the control, the control is assumed to be 100%.	No control condition
Type of outcome measures	Biomass reported as fresh or dry weight. Data with units (e.g., mg, g) included even if type of weight is unspecified.	Biomass values reported as increment/decrement percentages with negative values
Types of studies	Any experimental study	Review articles, case reports, book chapters, press reports, statements, comments, systematic reviews and meta-analyses, editorial, opinion papers, qualitative studies

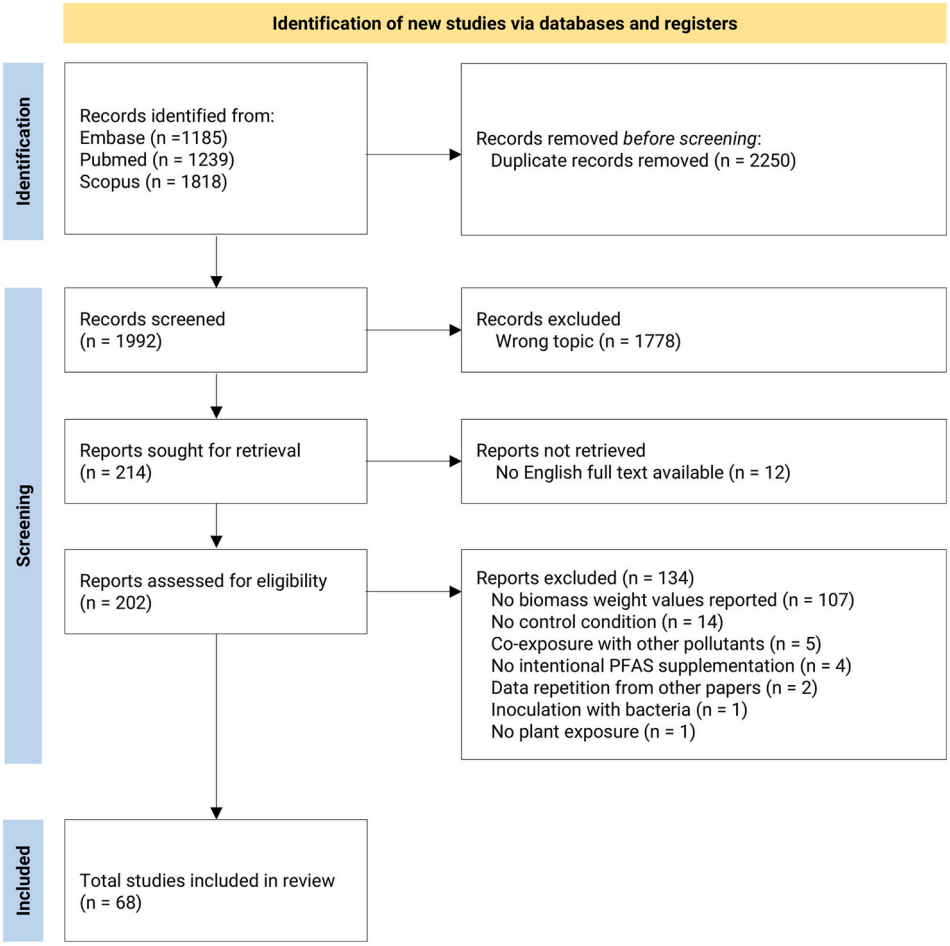


Fig. 1. The PRISMA flow diagram with details on the search strategy.

Table 2
Data extraction elements.

Topic	Items
Publication	First author last name, publication year, title, journal, DOI
Plant and growth conditions	Plant genus and species, type of plant, exposure media, growth condition
PFAS exposure	List of supplemented PFAAs, PFAA subclass, single or co-exposure, carbon chain length, number of carbon atoms, concentration, concentration unit of measurement, time of exposure (days)
Outcome	Mean and standard deviation for control and treated sample biomass, sample size (n), type of dispersion calculated, weight unit of measurement, weight type, type of biomass

et al., 2024). If no sample size was specified, but the *SD* was reported, $n = 3$ was assumed. If no *SD* was reported, $n = 1$ was assumed unless otherwise specified. If the *SD* value was equal to 0, it was replaced with 0.1. When raw replicate values were provided for both the control and PFAS-exposed groups, but standard deviations were not, they were calculated. Missing values for *SD* were imputed in R using the *mice* package, with the weighted predictive mean matching (PPM) imputation method (Tutz and Ramzan, 2015).

2.2. Effect size calculation and meta-analytic models

All the analyses were performed in R (version 2025.05.0, build 496) (R Core Team, 2023). For each observation, the effect size was calculated as the log-transformed ratio of the means (i.e., log

ratio of means) using the "ROM" measure within the "escalc()" function in the R *metafor* package (Viechtbauer, 2010), from control group mean (*m2i*) and PFAS-exposed group mean (*m1i*) using the following equation:

$$ln(RoM) = ln\left(\frac{m1i}{m2i}\right)$$

Variance was calculated from standard deviations (*sd2i* and *sd1i*) and sample size (*n*) using the following equation:

$$Var(ln(RoM)) = \frac{sd2i^2}{nm2i^2} + \frac{sd1i^2}{nm1i^2}$$

Three analyses were conducted, each considering a different type of biomass: whole-plant biomass (WB), belowground biomass (BB), and aboveground biomass (AB). For each of these, the general model was tested using the multilevel "rma.mv()" function as in *metafor*. To account for the dependencies amongst the effect sizes within studies, observation IDs were nested within each study as a random effect. For the meta-analytical effects, we calculated confidence intervals and prediction intervals. The latter reflected the variation of the true effect sizes.

Heterogeneity was analysed using the *Q* statistic and the I^2. For each of the three analyses, orchard plots for the general model were generated using the *metafor, ggplot2* (Wickham, 2016), and *orchard* (Nakagawa et al., 2023) R packages. Additionally, the contribution of various moderators was assessed: (i) categorical moderators (carbon chain length, PFAA class, weight type, exposure media, growth condition, plant species, type of plant) for which estimates and confidence intervals (CIs) were plotted in orchard plots alongside with *Q* statistics, and (ii) continuous moderators (concentration and time of exposure), for which a meta-regression was performed using the "regplot()"

function in *metafor*.

For each plant type or species, the effect sizes, number of observations, and statistical significance were represented using bubble plots. The R scripts and data frame used for data analysis and figure generation are provided in Supporting Information-2.

2.3. Publication bias and sensitivity analysis

Publication bias, including selective reporting, was assessed using funnel plots (Fig. S1, SI-2), implemented in *metafor*. Moreover, right-truncated meta-analysis (RTMA) using the *phacking* R package was applied to obtain effect estimates adjusted for *p*-hacking and publication bias by modelling the distribution of published non-affirmative results (Mathur, 2024). In addition, diagnostic RTMA plots included the distributions of *z*-scores (Fig. S2, SI-2), and cumulative distribution function (CDF) plots (Fig. S3, SI-2) were generated. Robust random-effects models with a *t*-distributed random effect were fitted using the *metaplus* R package (Beath, 2016) (Table 3; Fig. S4, SI-2).

3. Results

3.1. Overview of included articles

Following the literature screening, a total of $k = 776$ observations (Table S2, SI-1) from $n = 68$ articles published between 2005 and 2025 were included in the meta-analysis (Fig. 2A). The effects of 12 PFAAs on plant biomass were evaluated, including 9 PFCAs and 3 PFSAs (Table S1, SI-2). PFOA and PFOS were the most frequently tested PFAS, appearing in 49 and 39 papers, respectively (Fig. 2B). The change in biomass following exposure to PFAS was evaluated in 45 plant species, including aquatic plants ($n = 21$), herbaceous plants ($n = 21$), and woody plants ($n = 3$) (Fig. 2C).

Table 3
Summary results of general models for whole-plant, aboveground biomass and belowground biomass meta-analyses.

	SE	*Z*	*p*	95% CI	95% PI
Whole-plant biomass					
−0.13	0.03	−3.76	1.7E−04	−0.19, −0.06	−0.63, 0.38
Aboveground biomass					
−0.21	0.05	−3.99	6.7E−05	−0.31, −0.11	−1.03, 0.61
Belowground biomass					
−0.11	0.07	−1.63	1.0E−01	−0.23, 0.02	−1.07, 0.86
Heterogeneity of overall effect size					
Q	Total I^2 (%)	$I^2_{Between}$ (%)		I^2_{Within} (%)	
Whole-plant biomass					
2760 (<0.001) ***	98.45	30.80		67.65	
Aboveground biomass					
13,974 (<0.001) ***	98.71	40.15		58.56	
Belowground biomass					
23,589 (<0.001) ***	99.43	39.21		60.23	
Correction for outliers (t-distribution)					
Effect size *(p)*	95% CI	Tau^2	Loglik	AIC	BIC
Whole-plant biomass					
−0.08 (<1e−04)	−0.11, −0.05	0.012	−28	62	72
Aboveground biomass					
−0.08 (<1e−04)	−0.12, −0.05	0.041	−134	275	286
Belowground biomass					
−0.04 (0.14)	−0.09, 0.01	0.098	−204	415	426

Note. PI = Prediction interval. Loglik = Logarithm of likelihood.

3.2. PFAS effects on whole-plant, aboveground, and belowground biomass

To assess how PFAS exposure induced changes in plant biomass, the meta-analysis was stratified into whole-plant biomass (WB), aboveground biomass (AG), and belowground biomass (BG), based on the type of biomass measured in each included study. The model coefficients, test for heterogeneity (*Q* statistic and I^2), corrected models, and orchard plot for each meta-analysis are shown in Table 3 and Fig. 3. A total of 186, 305, and 277 observations were collected for WB, AB, and BB, respectively, coded from 34, 38, and 35 articles (Fig. 3A).

The meta-analytical effect size was negative in all three meta-analyses, but was statistically significant only for WB and AB. For all three meta-analyses, significant heterogeneity was observed ($p < 0.001$). In the case of WB, the total I^2 (98.45%) indicated that most of the variability was not due to random factors, with 67.65% of the heterogeneity attributed to differences within individual studies and 30.80% to variability between studies, and the same trend was observed for AB and BB (see the top and middle parts of Table 3). After accounting for outliers, the direction and statistical significance of the effect sizes remained unchanged across all the stratifications of the meta-analysis (see the bottom part of Table 3). Indeed, a limited number of outliers and no clear evidence for *p*-hacking were detected (Fig. S1–S5; **SI-2**).

The relevance of the categorical and continuous moderators was analysed (Fig. 3B), which included PFAS concentration, carbon chain length and class, exposure media, time of exposure, type of weight measured (i.e., fresh or dry weight), plant species, plant type, and growth conditions. PFAS concentration was significantly related to biomass outcomes across all biomasses, followed by plant species, WB, and AB. The introduction of PFAA class (i.e., exposure to PFCAs, PFSAs, or co-exposure) into the models yielded a statistically significant effect only for WB; similarly, exposure media and plant type were significantly related only to AB and BB.

3.3. Contribution of PFAS type and exposure conditions on biomass outcomes

The relationships between the PFAS characteristics (i.e., concentration, carbon chain length, PFAA classes) and the variation in plant biomass were analysed, and the results are shown in Fig. 4. The meta-regression analysis (Fig. 4A) revealed a significant ($p < 0.05$) inverse correlation between PFAS concentration and variation in WB ($\beta = -0.09$), AB ($\beta = -0.12$), and BB ($\beta = -0.14$). Introducing the carbon chain length (nC) of the PFAS used into the meta-analytical models, divided into short-chain ($nC \leq 7$), long-chain ($nC > 7$), or mixed short- and long-chain PFAS, was not significant for any of the analysed outcomes (Fig. 4B). Nevertheless, exposure to short-chain PFAS significantly reduced WB, while exposure to long-chain PFAS decreased both WB and AB. Exposure to a specific class of PFAAs (i.e., PFCAs or PFSAs) or co-exposure to PFAS from both classes was a significant variable with regard to WB (Fig. 4C). Furthermore, exposure to PFSAs or PFCAs individually produced a significant reduction in WB and AB.

The various studies included in the meta-analysis used diverse experimental conditions, such as time of exposure, growth media, growth conditions, and the type of weight measured, when exposing plants to PFAS. These conditions were introduced as moderators to potentially explain heterogeneity in the meta-analytic model (Fig. 5). Exposure duration (Fig. 5A) was a statistically non-significant factor in biomass variation, whereas growth media (Fig. 5B) were significantly related to AB variation, with liquid and agar media associated with a significant reduction in biomass. Growth conditions (Fig. 5C) were a statistically non-significant factor in determining biomass variation; however, studies conducted in a growth chamber showed the most significant reductions in WB and AB. The specific type of weight measured (Fig. 5D) – fresh weight (FW) or dry weight (DW) – significantly affected WB variation. In fact, when biomass variation was

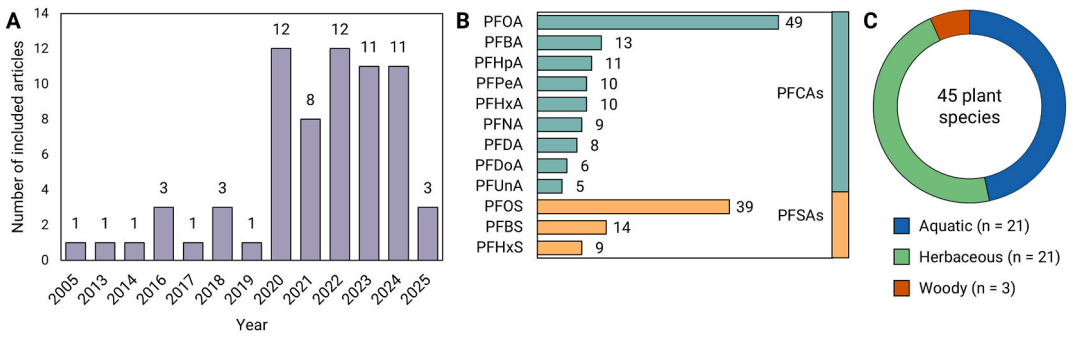


Fig. 2. Overview of included articles. (**A**) Publication year of the research papers included in the meta-analysis. (**B**) The number of articles testing specific PFAS, divided into perfluoroalkyl carboxylic acids (PFCAs) and perfluoroalkyl sulfonic acids (PFSAs). (**C**) The number of plant species in which changes in biomass were evaluated following exposure to PFAAs, with the number of aquatic, herbaceous and woody species reported.

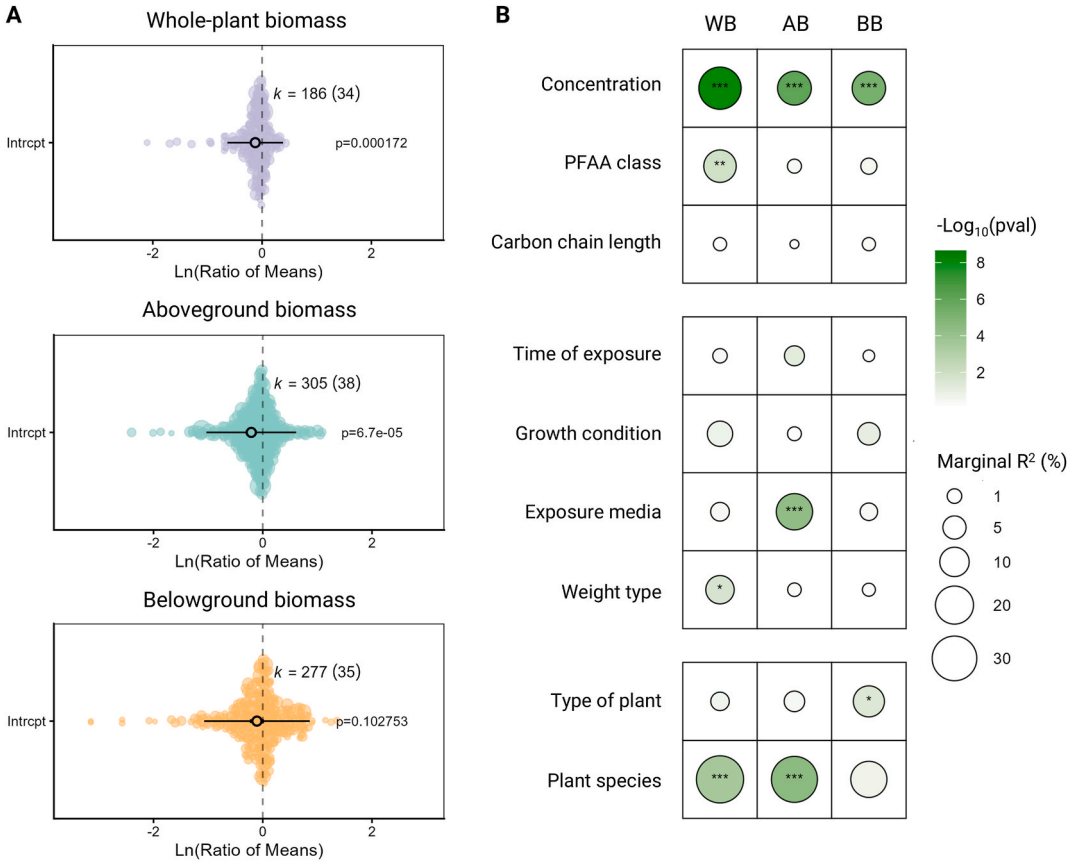


Fig. 3. General models and overview of moderators' analysis. (**A**) Orchard plots depicting the overall mean effect sizes for whole-plant biomass, aboveground biomass and belowground biomass meta-analyses. The size of the points is proportional to the precision of each observation. The numbers with and without parentheses indicate the number of studies and observations, respectively. (**B**) Bubble plots showing marginal R^2 (%) and statistical significance for categorical and continuous moderators in the meta-analysis for whole-plant biomass (WB), aboveground biomass (AB) and belowground biomass (BB). Bubble size is proportional to marginal R^2, and bubble fill colour ranges from white to green according to *p*-value. (*$p < 0.05$; **$p < 0.01$; ***$p < 0.001$).

evaluated in terms of FW, a significant reduction in WB (or AB) was observed.

3.4. Plant-specific response to changes in biomass following exposure to PFAS

The studies included in the meta-analysis involved 45 different plant species, categorised as woody, aquatic or herbaceous for the purposes of the analysis (Figs. 3C and 6). In the latter, exposure to PFAS resulted in a significant reduction in WB, AB, and BB (Fig. 6A). A substantial decrease in WB was observed in species such as *Abutilon theophrasti, Avena sativa, Cucumis sativus*, and *Nicotiana tabacum*. Similarly, a reduction in AB was observed in *A. theophrasti*, *Arabidopsis thaliana, Brassica napus*, and *Nicotiana benthamiana*, and a reduction in BB was observed in *B. campestris* and *B. napus* (Fig. 6B). However, an opposite trend, though statistically non-significant, was observed for both AB and BB in some herbaceous species, including *B. juncea, B. pekinensis*, and *Ocimum basilicum*. In contrast, exposure to PFAS led to a statistically significant increase in BB in the aquatic species (including helophyte and hydrophyte) analysed, markedly in the case of *Eichornia crassipes*, while a general decrease in WB and AB was observed (Fig. 6A and B). In the case of *Baumea articulata*, PFAS exposure resulted in an increase in both AB

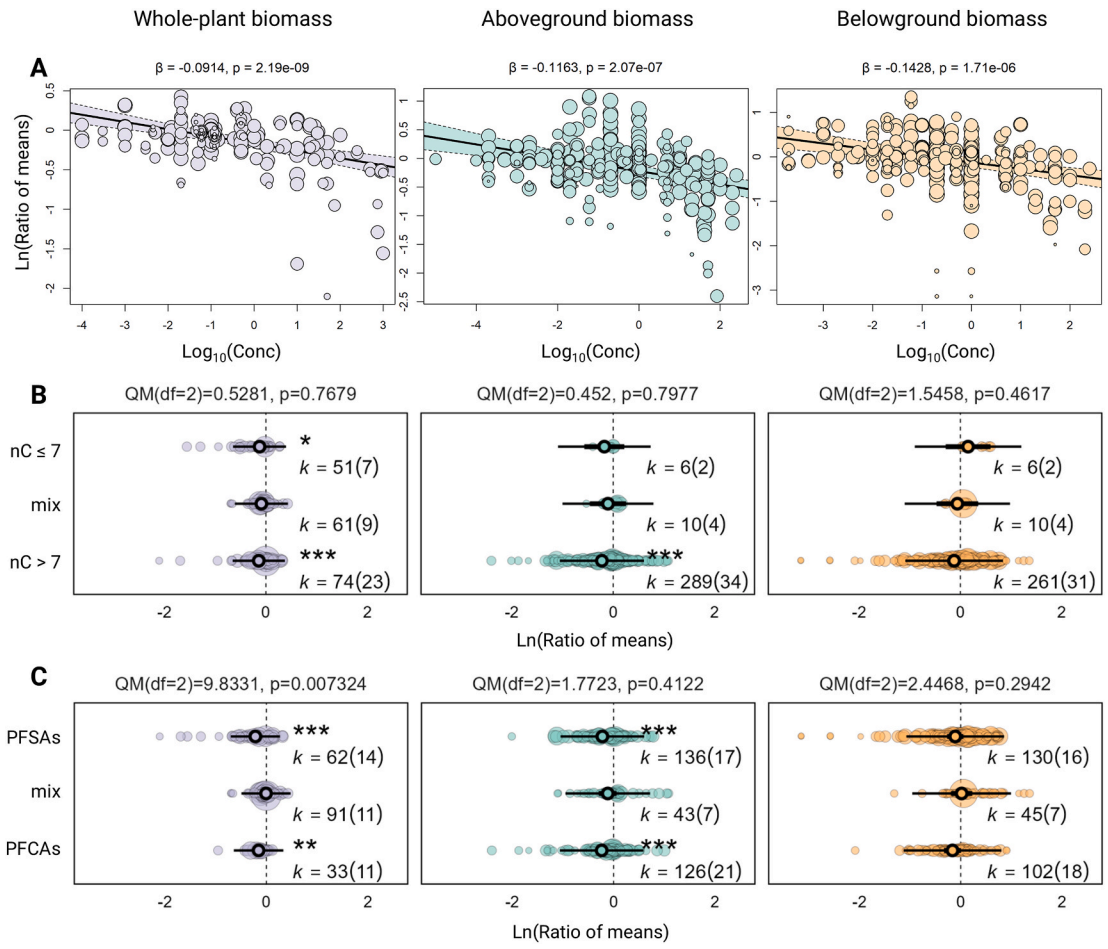


Fig. 4. Characteristics of tested PFAS. (**A**) Meta regression relating log-transformed PFAS concentration (Conc) to effect size for whole-plant biomass (violet, left column), aboveground biomass (green, centre column), and belowground biomass (orange, right column). The solid lines represent the slope estimate of the model, with 95% confidence intervals delimited by dashed lines. (**B-C**) Orchard plots for the PFAS carbon chain length (nC) and type of PFAAs (PFSAs or PFCAs, or mix) tested. The numbers with and without parentheses are the number of studies and observations, respectively. The size of the points is proportional to the precision of each observation.

and BB, though this latter was statistically non-significant (Fig. 6B). The woody plants considered, *Salix eleagnos, S. triandra*, and *S. purpurea*, for which a limited number of observations were available, did not show significant changes in biomass, although a general trend of decrease in AB and BB was observed (Fig. 6A and B).

4. Discussion

4.1. Exposure of plants to PFAAs is associated with WB and AB reduction, but not BB

After collecting data on plant exposure to PFAAs from 68 studies published over the last ten years (Fig. 2), the meta-analytic models were generated to assess the impact of PFAS exposure - specifically PFAAs - on changes in WB, AB, and BB. The core findings of the general models, before and after correction for outliers, and the orchard plots for WB, AB, and BB are reported in Table 3 and Fig. 3, respectively. The general models revealed that exposure to PFAAs was associated with a significant decrease in WB and AB, but not BB (Fig. 3A). Analysing biomass variation as a trait that is easily measurable and comparable is a highly informative approach in ecotoxicology (Delerue et al., 2022).

The presence of contaminants (e.g., PFAS) in plant growing medium falls under the category of abiotic stress, which, by definition, creates suboptimal conditions for plant growth (Zhang et al., 2020). However, it has been suggested that plants may actively suppress growth in response to abiotic stress by activating stress signalling pathways, though the exact underlying mechanism remains unclear. It has been hypothesised that crosstalk between growth regulation and stress response occurs at multiple levels (Zhang et al., 2020). Several studies on plants subjected to abiotic stress have highlighted the role of reactive oxygen species (ROS) as messengers capable of modulating gene expression and activating specific hormonal pathways, thus suggesting their potential role in actively inhibit growth, rather than growth inhibition being merely a consequence of the toxicity induced by oxidative damage (Mase and Tsukagoshi, 2021; Xia et al., 2015).

Exposure to PFAS, especially PFOA and PFOS, has been shown to increase the production of various ROS in plants, including H_2O_2, •OH and $O_2^{\bullet-}$ (Chen et al., 2020; Li et al., 2020c; 2025; Li et al., 2024b; Xiao et al., 2024; Yang et al., 2015), and to alter the levels of phytohormones such as auxin, cytokinin, jasmonic acid and abscisic acid (Guo et al., 2020; Pennacchio et al., 2005; P Li et al., 2020a; Sun et al., 2022; Xu et al., 2025). In this scenario, it is possible that the overproduction of ROS, in addition to causing changes in antioxidant enzyme activity and oxidative damage, may mediate an adaptive response with repercussions on plant growth, modulating specific phytohormones. The findings of this meta-analysis indicated that PFAS presence may reduce WB and AB, but not BB, likely due to several factors. It is generally understood that BB is typically less abundant than AB, especially in species associated with forest and cropland ecosystems, where the root: shoot ratio (R:S) tends to favour aboveground biomass (Jackson et al.,

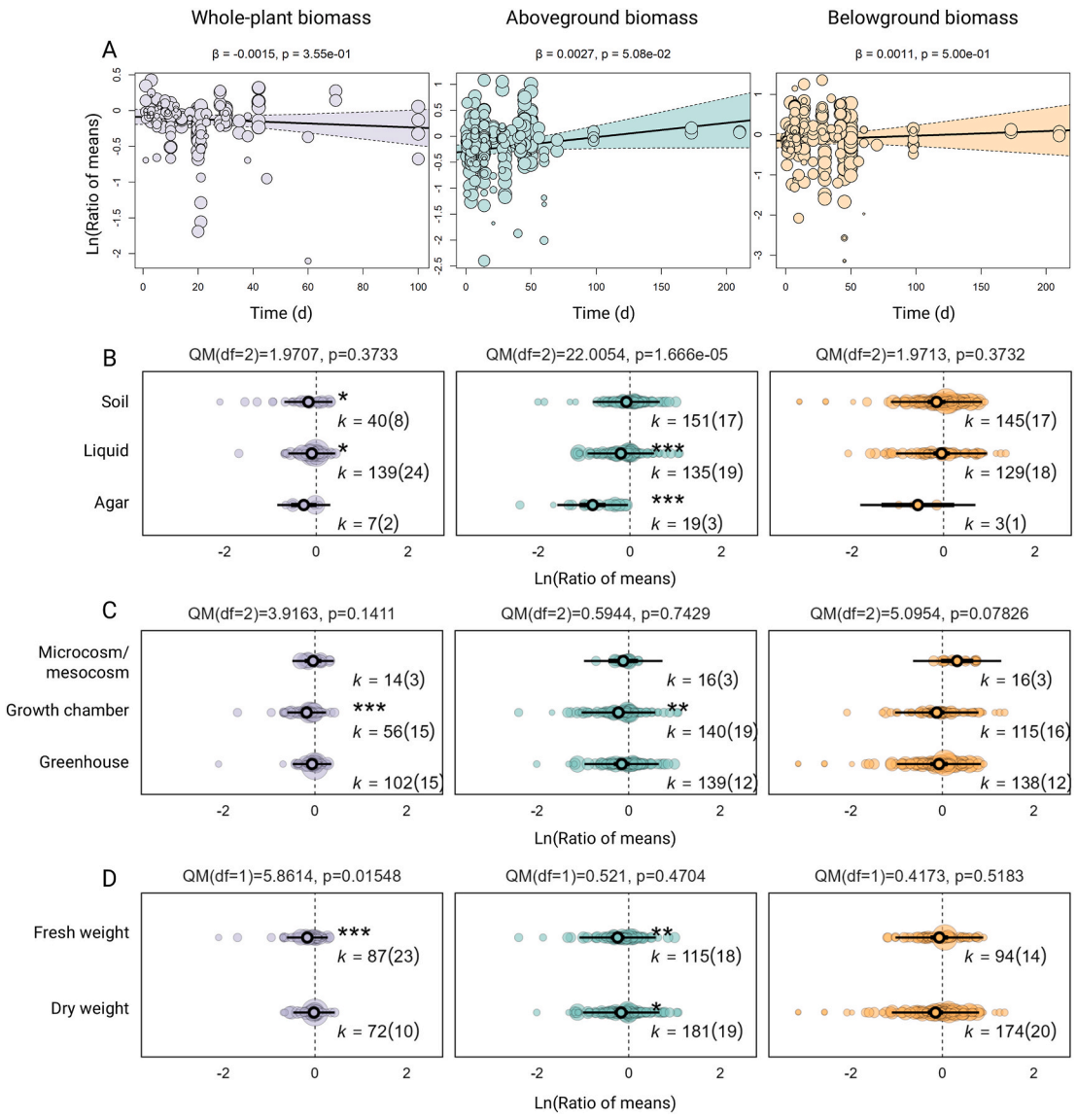


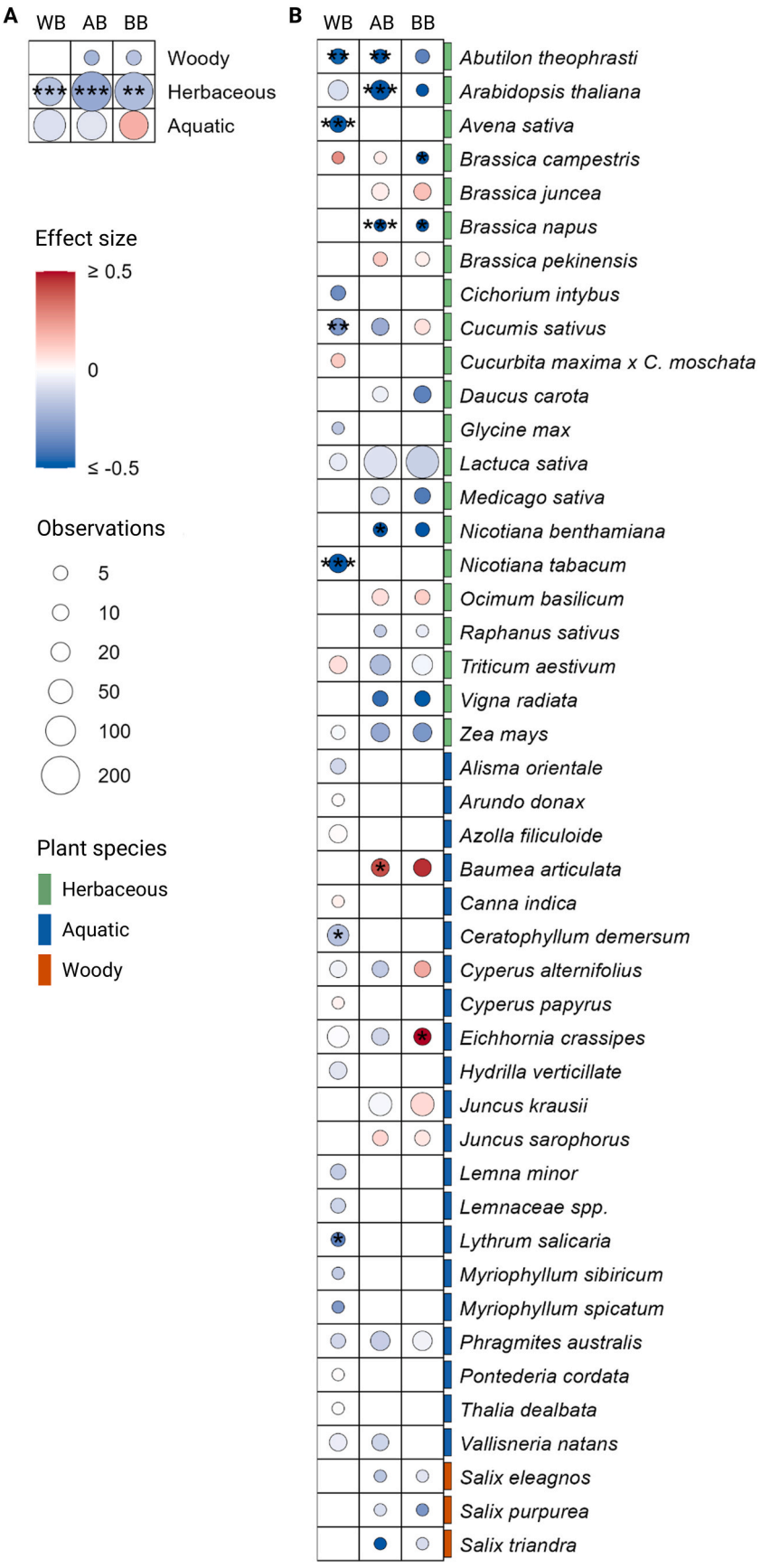
Fig. 5. Experimental conditions as variables potentially influencing changes in plant biomass following PFAS exposure. (**A**) Meta regression relating time of PFAS exposure (days) to effect size for whole-plant biomass (violet, left column), aboveground biomass (green, centre column), and belowground biomass (orange, right column). Solid lines represent the slope estimate of the model, with 95% confidence intervals delimited by dashed lines. (**B-D**) Orchard plots for exposure media, growth condition, and type of weight measured. The numbers with and without parentheses are the number of studies and observations, respectively. The size of the points is proportional to the precision of each observation.

1996). This may result in AB contributing more than BB to the overall phenotype when assessed at the whole-plant scale. However, it has been observed that under abiotic stress conditions, such as drought stress, the plant may allocate more carbon to the roots, leading to an increase in R:S at the expense of stem and leaf mass (Eziz et al., 2017; Zhou et al., 2022). Furthermore, it should be noted that the general meta-analytic model of BB is affected by the specific behaviour of aquatic plants (Fig. 6), which will be discussed in a later section.

4.2. *The reduction of WB and AB is influenced by the experimental design*

The heterogeneity across all three models ($I^2 > 98\%$) indicated the need to explore potential moderators that could explain the variability of the effect sizes. The core findings for the moderator analysis are summarized in Fig. 3B. The moderator analysis considered both continuous and categorical moderators, including the characteristics of PFAS tested (Fig. 4), the experimental conditions (Fig. 5), and the type of plant analysed (Fig. 6, see Section 4.3).

As expected, concentration was a crucial parameter in determining biomass variation (Fig. 4A), as increased exposure to PFAS is likely to result in greater accumulation in the plant, which can affect plant growth in a dose-response manner. However, toxicology and ecotoxicology are increasingly recognising the existence of hormetic phenomena, whereby a bell-shaped curve is observed within a specific concentration range (Erofeeva, 2022). The curve obtained in the meta-regression of AB appears to resemble this phenomenon. Although the molecular mechanism underlying hormesis is unclear, it appears to represent an adaptive response to stress and/or a compensatory mechanism resulting from a disruption to homeostasis (Agathokleous et al., 2020; Calabrese, 1999). With regard to the specific PFAS tested, exposure to long-chain PFAS (nC > 7) was linked to a reduction in WB and AB (Fig. 4B). This is consistent with studies that have directly compared the toxicity of long-chain and short-chain PFCAs and PFSAs in tobacco, coontail and *Acorus calamus*, which have shown that PFOA and PFOS are more toxic than their four-carbon counterparts (*i.e.* PFBA and PFBS), mainly due to their ability to induce greater oxidative stress (Li et al.,



(caption on next page)

Fig. 6. Overview of plant-specific trend in biomass variation following PFAS exposure. Bubble plots showing the number of observations and effect size for different plant type (**A**) or plant species (**B**) in the meta-analysis for whole-plant biomass (WB), aboveground biomass (AB) and belowground biomass (BB). Bubble size is proportional to the number of observations included, and a divergent palette for positive (*red*) or negative (*blue*) effect size has been applied. Plant species classified as herbaceous, aquatic or woody are flanked by green, blue, or orange bars, respectively. (*$p < 0.05$; **$p > 0.01$; ***$p < 0.001$).

2025; Li et al., 2024b; Xiao et al., 2022). While PFSAs appeared to have a greater tendency to reduce WB, the main finding was that reductions in WB and AB were not significant when exposure occurred to a mixture of different PFAAs (Fig. 4B and C). This aspect is probably linked to the fact that, in an attempt to mimic environmental conditions, studies testing PFAS mixtures tend to use doses that reflect environmentally relevant concentrations (ERCs) - in the order of ppb - which rarely produce observable phenotypes (Marzi et al., 2025). Furthermore, using PFAS mixtures at ERCs reduces the concentration of individual components, with different phytotoxicity potentials. As is often the case in ecotoxicology (Wolf and Segner, 2023), studies investigating and/or comparing the phytotoxicity of specific PFAS tend to use higher doses than ERCs, particularly in laboratory-scale studies where exposure times are generally limited (Luche et al., 2026).

Surprisingly, exposure time did not appear to correlate with biomass variations, with the meta-regression coefficients (β) for WB, AB and BB all close to zero and not significant (Fig. 5A). In ecotoxicology, exposure time is still a relatively neglected parameter, with other exposure criteria, such as concentration, being favoured when designing experiments (Connell et al., 2016; Jager, 2025). In our dataset, the exposure time considered was the period during which plants were exposed to PFAS via the growing medium. In some cases, exposure occurred during sowing or at the sprout stage (Fan et al., 2020; Kim et al., 2024; Sun et al., 2022). In others, exposure started at more advanced stages, using plants that had already been grown (Lintern et al., 2024; Lu et al., 2025; Pietrini et al., 2024). In the latter case, the exposure time did not cover the early stages of development, during which the plant is more sensitive to the presence of pollutants and has fewer defences with which to cope with their potential toxicity (Song et al., 2020). This heterogeneity in the onset of exposure may partly account for the non-significant meta-regression coefficients for exposure duration. By contrast, different exposure media were linked to significant biomass reduction (Fig. 5B).

When the exposure medium was agar, the growth medium of choice for *in vitro* studies, a drastic and significant reduction in AB was observed. Similarly, a liquid growth medium was linked to a significant reduction of AB compared to growth in soil. This may be due to PFAS sorption kinetics with soil organic matter (Moavenzadeh Ghaznavi et al., 2025), which results in greater bioavailability of the pollutant when exposure occurs via liquid or synthetic media.

Another significant factor linked to a reduction in WB and AB was growth under controlled conditions, such as in growth chambers (Fig. 5C). Among the studies included in the meta-analysis (see Table S2, SI-1), those with the highest exposure concentrations (ppm) were generally studies conducted in growth chambers with limited exposure times of 5–10 days (Lin et al., 2020; Pan et al., 2021; Sun et al., 2022; Wen et al., 2013). Interestingly, reductions in WB and AB were significant when changes in biomass were measured as fresh weight (FW) (Fig. 5D). Although recording FW is less laborious, this parameter is strongly influenced by the plant's water content, unlike dry weight (DW). The ability of PFOA to alter water uptake has already been demonstrated in a study on *Alisma orientale*, which also reported changes in the structure and function of different classes of biomolecules (Wang et al., 2022). Similarly, a study on hydroponically grown *S. triandra* showed that exposure to a mixture of 11 different PFAS reduced stomatal conductance and increased xylem vulnerability to air embolism, thereby altering xylem hydraulics (Battisti et al., 2023). This suggests that DW is a more reliable parameter for determining changes in biomass, as it is not affected by the bias associated with differences in water content that may be caused by PFAS exposure.

4.3. The change in biomass following exposure to PFAAs is highly dependent on the type of plant

The reports included in the meta-analysis cover studies conducted on 45 different plant species, including aquatic (Awad et al., 2022; Chen et al., 2024; Gang et al., 2025; Gonzales et al., 2023; Hanson et al., 2005; Hua et al., 2022; Q. Li et al., 2023b, 2023a;Li et al., 2022b; Li et al., 2024a; Li et al., 2020d; Li et al., 2024b; Xiao et al., 2024; 2022; Lintern et al., 2024; Lu et al., 2025, 2023; Qiao et al., 2021; Shen et al., 2024; Wang et al., 2022, 2020; Yang et al., 2023; Zhang et al., 2023c; Zhu et al., 2022), herbaceous (Bizkarguenaga et al., 2016; Chen et al., 2020; Du et al., 2020; Ebinezer et al., 2022; Fan et al., 2020; Gredelj et al., 2020; Hearon et al., 2022; Kim et al., 2024; Krippner et al., 2014; Lan et al., 2018; Li et al., 2021a, 2021b, 2025; Pennacchio et al., 2005; Li et al., 2022a; Li et al., 2020b, 2020a; Li and Li, 2021; Xiao et al., 2022; X. Li et al., 2023; Lin et al., 2020; Liu et al., 2023, 2022; Lupton et al., 2022; Müller et al., 2016; Pan et al., 2021; Pietrini et al., 2024; Rico et al., 2024; Song et al., 2021; Sun et al., 2022; Wang et al., 2024, 2023; Wattier et al., 2023; Wen et al., 2013; Xiang et al., 2020, 2018; Xu et al., 2022a,b, 2019; Yu et al., 2021, 2018; Zhang et al., 2023a; Zhang et al., 2022; Zhao et al., 2016, 2017; Zhu et al., 2024a, 2024b) and woody species (Sharma et al., 2020). The core findings of the plant-type and species-specific analyses are provided in Fig. 6.

In the case of woody species, only *Salix* spp. were included, for which a limited number of observations from a single study were available (Sharma et al., 2020), suggesting a general trend of reduce AB and BB (Fig. 6A). Nevertheless, in this study Sharma et al. (2020) highlight these species' ability to take up different PFAAs and efficiently translocate short-chain PFAAs to leaves, supporting their potential use in phytoextraction. However, as a hydroponic-based study, potential PFAS-soil interactions were excluded, limiting the translatability of these findings to field conditions (Sharma et al., 2020).

Conversely, a large number of observations have been collected for herbaceous species, for which a significant reduction in all considered biomasses has been observed (Fig. 6A). Many plant-specific exceptions have been observed (Fig. 6B), which remain difficult to interpret even when taxonomic classification subgroups are considered. For instance, considering the various *Brassicaceae* species included in the analysis, contrasting trends emerged. While *A. thaliana*, *B. napus* and *Raphanus sativus* showed a general reduction in WB, AB and BB, *B. campestris*, *B. juncea* and *B. pekinensis* showed an increasing trend, particularly in WB and AB. These three species have already been investigated for phytoremediation of cadmium-contaminated soils, with *B. juncea* in particular reported as the most tolerant and exhibiting the highest biomass production and Cd accumulation rates (Dhaliwal et al., 2022). In the study by Wattier et al. (2023), an increase in AB and BB, as well as efficient shoot translocation (TF > 1), was reported in hydroponically grown *B. juncea* exposed to high concentrations of PFOA (1–15 mg L^{-1}). Given that this experimental setup often results in marked biomass reduction in our meta-analysis (see Figs. 4A and 5B), these findings encourage further investigation into the potential use of this species for phytoremediation. However, within the studies included in this meta-analysis, evidence on growth responses of *B. campestris, B. juncea* and *B. pekinensis* to PFAS is limited to PFOA exposure (Wattier et al., 2023; Zhang et al., 2023a; Zhu et al., 2024a, 2024b), and co-exposure with PFOS has been considered only for *B. campestris* (Zhao et al., 2017). Testing these species under exposure to mixtures of multiple PFAS would therefore be advisable to better characterize their phytoremediation potential.

Species-specific differences also emerged within the *Cucurbitaceae*

family, where an increasing trend occurred in the cross between *Cucurbita maxima* and *C. moschata,* whereas a significant decrease in WB was observed in *Cucumis sativus*. Du et al. (2020) reported a marked accumulation in leaves, reduced biomass and photosynthetic performance, and metabolic alterations in soil-grown *C. sativus* exposed to increasing PFOA concentrations (0-5 mg kg^{-1}). Indeed, *C. sativus* is widely used in phytotoxicity bioassays, such as tests for seed germination inhibition and root elongation, due to its high sensitivity to a wide variety of pollutants (Wang et al., 2001; Zhang et al., 2023b). A marked reduction in biomass was also evident when model species such as *A. thaliana, N. benthamiana*, and *N. tabacum* were tested. On the one hand, this may be because these included *in vitro* studies (Fan et al., 2020; Kim et al., 2024; Lin et al., 2020; Sun et al., 2022; Zhang et al., 2022), in which the experimental design tends to use concentrations higher than ERCs in order to produce observable phenotypes. On the other hand, it may also reflect their intrinsic sensitivity to abiotic and biotic stress, making them ideal models for studying the biological mechanisms underlying plant responses to environmental stress (Goodin et al., 2008; Pennacchio et al., 2005). In most cases, responses across organs (aboveground or belowground) in herbaceous species were consistent, with the same trend of reduction or increase for both AB and BB.

By contrast, a clear reduction in AB and an increase in BB appeared to occur in aquatic species (Fig. 6A), as seen with *C. alternifolius*, *J. krausii* and *E. crassipes*. For the latter species, the BB increase was statistically significant and was supported by studies that tested both single-compound exposure (i.e. PFOA or PFOS) or exposure to mixture of PFAAs under different growth conditions (Chen et al., 2024; Li et al., 2020d; Xiao et al., 2024; Qiao et al., 2021). Two studies simulating vertical-flow constructed wetlands (VFCWs) with *C. alternifolius* and *E. crassipes* reported a significant increase in belowground biomass (BB) of *E. crassipes* even at high PFOS concentrations (10 mg L^{-1}) (Li et al., 2020d; Qiao et al., 2021). In the latter study, Qiao et al. (2021) also showed that *E. crassipes* accumulates more PFOS than *C. alternifolius*, likely due to its larger root surface area. Overall, several studies have highlighted the ability of water hyacinth to bioaccumulate both short- and long-chain PFAAs (Chen et al., 2024; Xiao et al., 2024; Phung et al., 2024; Qiao et al., 2021), with PFAA removal rates in some cases ranging from 40.3 to 63.5% (Phung et al., 2024), thereby positioning this species as a promising candidate for larger-scale studies. Interestingly, a significant increase in both AB and BB was also recorded for *B. articulata*. However, these observations derive from a single study that tested exposure to PFOA and PFOS, in which Awad et al. (2022) reported that *B. articulata* was able to remove more than 20% of both PFAAs. This evidence, along with its documented tolerance to other classes of pollutants, including cadmium and polycyclic aromatic hydrocarbons (Zhang et al., 2010a, 2010b), makes it a promising species for further investigation in the phytoremediation of emerging contaminants.

4.4. Implications, limitations, and future directions

The meta-analysis showed that plant exposure to PFAAs was generally associated with a reduction in WB, particularly in AB. This trend was statistically significant across several herbaceous species, including *A. theophrasti*, *A. thaliana*, *A. sativa*, *B. napus*, *C. sativus*, *N. benthamiana*, and *N. tabacum*. In terrestrial systems, such AB reduction suggests that the actual phytoextraction potential could be limited, especially in the scenario of high PFAS contamination. On the other hand, this marked phenotype supports the use of these species in ecotoxicological assays and, where feasible, in environmental monitoring. Although based on a few studies and limited to exposure to single PFAS compounds, the observed biomass increases trends for *B. campestris*, *B. juncea*, and *B. pekinensis* highlight these species as potentially useful candidates for phytoremediation.

A general trend for BB increase was observed in aquatic plants such as *B. articulata*, *C. alternifolius*, *E. crassipes*, *J. krausii*, and *J. sarophorus*. This pattern was detected in both emergent helophytes and hydrophytes, and it was statistically significant for AB in *B. articulata* and for BB in *E. crassipes*. These results suggest that exposure to PFAAs in aqueous matrices (surface water, treated wastewater and constructed wetland systems) can be associated with an increased root surface area available for contaminant absorption and retention. From this perspective, our findings suggest that phytoremediation of PFAS is potentially more effective and sustainable in aquatic systems than in contaminated soils.

Moreover, in aquatic plants used for PFAS phytoremediation, an increase in root biomass may provide a larger surface area for colonization by endophytic and rhizosphere bacteria forming biofilms (Liu et al., 2025; Zhao et al., 2025a). Bacterial communities associated with aquatic macrophytes have already been reported to contribute to PFAS retention (Chen et al., 2025; Hua et al., 2022; Yang et al., 2025). Given the increasing number of studies on PFAS-degrading bacteria (Skinner et al., 2025), future applications in surface-water and wastewater treatment systems could make use of both aquatic plants and their associated microbiomes (Arslan and Gamal El-Din, 2021; Zhao et al., 2025a,b) to improve the overall efficiency of PFAS bioremediation.

A critical factor for the practical implementation of PFAS phytoremediation, in both terrestrial and aquatic systems, is the management of contaminated biomass. Our results suggest that in some species, especially aquatic plants, PFAS exposure can stimulate an increase in BB, which may in turn enhance PFAS accumulation or retention in root tissues. Consequently, harvested biomass may carry high contaminant loads and potentially re-release PFAS if managed improperly (Marzi et al., 2025). Therefore, any phytoremediation program must explicitly address post-harvest biomass handling, prioritizing treatment or disposal options that prevent PFAS dispersion into the environment. Integrating secure biomass management into phytoremediation design is essential for safely scaling up PFAS removal using plants.

In addition to highlighting differences among plant species, the analysis of moderators identified the key experimental factors that are associated with reduced or increased biomass production under PFAS exposure. Parameters that strongly influence plant development include selecting an appropriate PFAS concentration based on the specific objective of the investigation, exposure to individual or mixed PFAS, and selecting an appropriate growth medium. Together, these factors govern PFAS bioavailability and therefore should be carefully defined and justified when designing phytoremediation pilot studies in both contaminated soils and aqueous matrices.

These results should, however, be interpreted with some limitations in mind. This meta-analysis relies on biomass as an endpoint, selected because it is widely reported and comparable across studies, and therefore does not directly capture physiological, biochemical, or molecular alterations that may be relevant to PFAS tolerance. In addition, the analysis focuses on PFAAs, providing a first comparative overview for this major PFAS subclass, while other PFAS groups remain less explored in this framework. Nevertheless, the dataset assembled here provides a foundation for future syntheses that extend the chemical coverage to additional PFAS classes and integrate biomass with physiological, biochemical, and molecular readouts, as well as species- and tissue-specific accumulation patterns, as evidence on PFAS-plant interactions continues to expand.

CRediT authorship contribution statement

Sophia Luche: Writing – review & editing, Writing – original draft, Visualization, Validation, Software, Methodology, Formal analysis, Data curation, Conceptualization. **Jacek Buczny:** Writing – review & editing, Supervision, Software, Methodology, Formal analysis, Data curation. **Giovanna Visioli:** Writing – review & editing, Writing – original draft, Supervision, Project administration, Funding acquisition, Conceptualization.

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Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2026.129217.

Data availability

All data are included in the supplementary information and available from the authors.

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