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Research Article

Keywords:

Posted Date: August 27th, 2025

DOI: <https://doi.org/10.21203/rs.3.rs-7406781/v1>

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Arsenic residues in pastures and horses after MSMA-based herbicide control of *Urochloa decumbens*

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Abstract

The diet of horses is primarily based on grasses such as Coast Cross (*Cynodon* spp.), with *Urochloa decumbens* (formerly *Brachiaria decumbens*) being the main invasive species. The control of *U. decumbens* is typically performed using the herbicide MSMA (monosodium methylarsonate), which degrades into four main arsenic species after application: arsenite, arsenate, monomethylarsonic acid, and dimethylarsinic acid. This study aimed to determine whether the spray of MSMA is a determining factor in increasing arsenic concentrations in the environment and in horses. Arsenic levels in soil, plants, and animals were compared following herbicide use. Analytical results revealed higher arsenic accumulation in *U. decumbens* (26.20 ± 0.08 ppm) compared to Coast Cross (12.66 ± 2.94 ppm) ($p < 0.0001$). *U. decumbens* accumulated 10.69 ± 1.69 ppm in the MSMA-treated area versus 6.06 ± 0.96 ppm in the control area ($p < 0.0001$). Coast Cross showed arsenic concentrations of 10.73 ± 1.22 ppm in the MSMA-treated area and 1.60 ± 0.41 ppm in the control area ($p < 0.0001$). Horses grazing on MSMA-treated pastures presented significantly higher arsenic levels in blood (1.67 ± 0.047 ppm) compared to the control group (1.35 ± 0.0531 ppm) ($p < 0.05$). In urine, arsenic levels were 1.62 ± 0.189 ppm in the MSMA group and 1.195 ± 0.111 ppm in the control group ($p < 0.05$). The results demonstrate that MSMA sprayed in equine pastures leads to significant arsenic accumulation in forage plants and in horses, representing a potential risk to animal health and food safety.

Horses (*Equus ferus caballus*) play fundamental roles in Brazil's economic, social, and public health sectors. They are used in sports, draft work, transportation, leisure, agricultural activities, and more recently, in assisted therapies (Hausberger et al. 2008; Couter 2014; Adelman and Thompson 2017). They also have strategic importance in public health, particularly in the production of antivenoms essential for treating snakebites (Queiroz 2005). Brazil holds the third-largest horse population in the world, with over 5.8 million animals (IBGE 2023), supporting approximately 3.2 million direct and indirect jobs (Lima and Cintra 2016).

Equine nutrition relies predominantly on pastures, offering a practical and cost-effective feeding strategy (Brandi and Furtado 2009). However, horses are selective grazers and tend to avoid grasses from the genus *Urochloa* spp. Grasses of the genus *Cynodon* exhibit characteristics favorable to equine nutrition, and Coastcross-1 (*Cynodon dactylon* (L.) Pers. cv. Coastcross-1) a sterile hybrid (Wallau et al. 2010) is an excellent pasture option for horses (Chambliss 2000). This selectivity can result in *Urochloa decumbens* (Stapf) R.D. Webster, formerly *Brachiaria decumbens* (UPOV, 2023) dominance in pastures (Zanine, 2006; Bott et al. 2013; Mulinari 2015), a species not only poorly palatable but also associated with photosensitization (Barbosa et al. 2006) and secondary nutritional hyperparathyroidism (Gênova and Paulino 2011). Invasive plants drastically reduce forage productivity, with losses reaching up to 80% (Rodrigues et al. 2005).

Among the control strategies for these species, herbicide use in extensive systems stands out due to its practicality and economic feasibility (Merotto et al. 2022). MSMA

(monosodium methanearsonate) is the most used herbicide in Brazil for selective control of *U. decumbens* in the Coastcross-1 fields, although it is legally authorized only for crops like sugarcane and cotton. Its application on pastures constitutes off-label use and violates Brazilian pesticide legislation (Brasil 1989).

MSMA is an organoarsenic herbicide with high potential risk to human and animal health due to the presence of arsenic (As), a toxic, mutagenic, carcinogenic, and bioaccumulative element (ATSDR 2007; Ray et al. 2013; Su et al. 2023; Ganie et al. 2024). Ingestion of contaminated forage may lead to arsenic bioaccumulation in animal tissues, potentially compromising equine health and the safety of animal-derived products such as milk and meat (Hughes 2002; Yadav et al. 2024). Improper MSMA use also contributes to soil and groundwater contamination, endangering ecosystems and natural resources (Smedley and Kinniburgh 2002; Eldan and Masue-Slowey 2024; Darma et al. 2022; Mohsin et al. 2023).

Once applied, MSMA in aqueous solution dissociates into monomethylarsonic acid (MMA) and sodium ions (Eldan and Masue-Slowey 2024). In the environment, MMA undergoes microbial and abiotic transformations, converting into inorganic species: arsenate (As^{V}), arsenite (As^{III}) and organic compounds: dimethylarsinic acid (DMA) (Feng et al. 2005; Hemmat-Jou et al. 2024) (Figure 1). Factors such as soil texture and composition, pH, organic matter content, water table depth, and redox potential influence arsenic speciation and mobility (Matterson et al. 2014; Gannon 2016). Microbial degradation of MMA has been widely reported, highlighting the role of biological activity in arsenic transformation and mobility (Diekens and Hiltold 1967; Von Endt et al. 1968; Akkari et al. 1986).

Arsenic species enter plants via different mechanisms: As^{V} , MMA^{V} , and DMA^{V} utilize phosphate transporters, whereas As^{III} , MMA^{III} , and DMA^{III} are absorbed through aquaporins (Cao and Ma 2004; Raab et al. 2008; Li et al. 2009; Mishra et al. 2021). DMA may also be methylated to form trimethylarsine, a volatile gas (Feng et al. 2005).

Arsenic is a semi-metallic element (atomic number 33, mass 74.92 u), widely distributed in the Earth's crust mainly in igneous and sedimentary rocks (Kabata-Pendias 2007; NRC 2007). Its presence in the atmosphere is associated with coal combustion and mining activities (Abreu et al. 2012). Historically, arsenic was used in pigments, medicines, and cosmetics by ancient Egyptian, Chinese, and Greek civilizations (Cullen 2008). During the Middle Ages and Renaissance, it was promoted as a universal remedy, with arsenic trioxide (As_2O_3) being widely used. From the 17th to 19th centuries, it gained notoriety as the “king of poisons,” frequently used in homicides (Nriagu 2002). With advances in chemistry, its applications expanded to agriculture (as fungicides, pesticides, and herbicides) and to the electronics and metallurgical industries (Gontijo and Bittencourt 2005).

The trivalent (As^{III}) and pentavalent (As^{V}) forms are predominant in biological and environmental systems, differing in toxicity, mobility, and solubility (Smith and Steinmaus 2011; Genchi et al. 2022). Inorganic arsenic compounds interfere with essential cellular processes such as oxidative phosphorylation and thiol metabolism, potentially causing multi-organ failure (Schrenk et al. 2024). Chronic ingestion, even at low doses, is associated with cancer (skin, liver, lung, bladder) and cardiovascular, hematological, and neurological disorders (ATSDR 2007).

After gastrointestinal absorption, arsenic is widely distributed in the body, accumulating mainly in the liver, kidneys, lungs, and spleen (Gupta 2007; Gupta and Gupta 2025). As^{III} strongly binds to mitochondrial enzyme sulfhydryl groups, inhibiting cellular respiration. As^{V} can replace phosphate groups in biomolecules, forming compounds like glucose-6-arsenate that block key metabolic pathways such as glycolysis and the pentose phosphate pathway (Delnomededieu et al. 1944; Crane and Lipmann 1953; Aposhian 1989; Winski and Carter 1995). As^{III} also depletes glutathione (GSH), impairing cellular antioxidant defenses (Shila et al. 2005).

In plants, arsenic disrupts oxidative phosphorylation and glycolysis, reducing ATP production and increasing reactive oxygen species generation (Finnegan and Chen 2012; Hu et al. 2020). Its effects include chlorosis, inhibition of root and shoot growth, oxidative stress, nutritional imbalances, and decreased total biomass (Shri et al. 2009; Mishra et al. 2017). Plant defense mechanisms include restricted uptake (Abedin et al. 2002), intracellular sequestration (Dixit et al. 2015; Zhang et al. 2018), biotransformation (Song et al. 2010), and active excretion (Li et al. 2016; Garbinski et al. 2019).

Given the widespread off-label use of MSMA in equine pastures, it is essential to investigate the environmental and toxicological impacts of this practice, particularly the accumulation of arsenic in soil, plants, and animals within this production system (Mahoney 2015b; Rajendran et al. 2024). The main objective of this study was to detect and quantify increases in arsenic concentrations in soil, forage, and horses following MSMA herbicide sprayed in pastures.

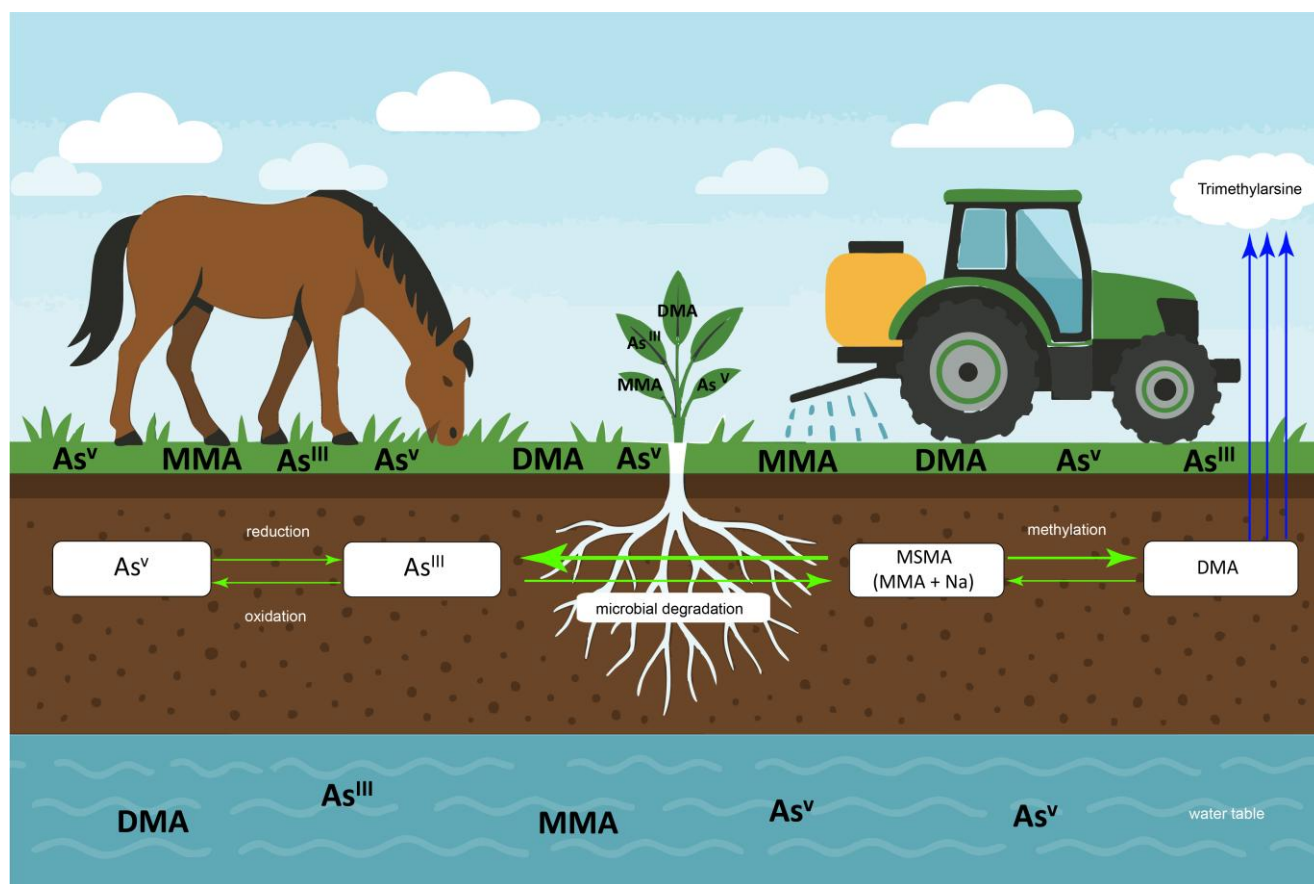


Figure 1. Arsenic speciation following MSMA application. As^{III} (arsenite); As^{v} (arsenate); MMA (monomethylarsonate); DMA (dimethylarsinic acid). Adapted from Feng et al. (2005).

Materials and Methods

Study area and experimental design

Two pasture areas, each measuring 25,000 m², located on a horse farm with over 40 years of breeding history in the state of São Paulo, Brazil, were used in this study. Both areas had flat terrain, were established with Coastcross 1, and exhibited medium infestation of *Urochloa decumbens*. The sites were located at latitude 23°32'23" S and longitude 48°03'29" W, with a mean altitude of 737 m. The location is 11.53 km south of the Tropic of Capricorn. The climate is classified as Cfa according to Köppen–Geiger (Köppen 1901; Alvares et al. 2013). During the experimental period, the mean maximum temperature was 28.18 °C, the mean minimum temperature was 16.7 °C, and total rainfall

was 112 mm (Clima-Data 2023). The soil is classified as dystrophic Red Latosol (Rossi 2017). Adjacent areas consisted of other pastures.



Figure 2. Sampling locations, where AM represents the MSMA area and AC represents the Control area.

In one paddock, 3,600 g ha⁻¹ of MSMA (Volcane®, Du Pont do Brasil S.A., Paulínia, São Paulo, Brazil), equivalent to 1,458.36 g ha⁻¹ of arsenic, was applied, diluted in 350 L ha⁻¹ of spray solution. The maximum temperature on the day of application was 29.92 °C. There was no wind or precipitation during application. Rainfall of 7.62 mm occurred the day after application. *U. decumbens* showed intense chlorosis during the first week, progressing to complete desiccation by the second week, whereas Coastcross-1 exhibited slight yellowing in the first week (Figure 3), with full recovery of normal coloration from the second week onwards.



Figure 3. *U. decumbens* showed intense chlorosis, whereas Coastcross-1 exhibited slight yellowing in the first week.

Sixteen horses were used in this study. Eight animals grazed in the herbicide-treated paddock (MSMA group), consisting of six mares and two geldings, aged 3–22 years and weighing 370–550 kg. The control paddock (Control group) housed eight horses (five mares and three geldings), aged 2–18 years and weighing 320–580 kg.

Animal use complied with Brazilian legislation (Law No. 11,794 of October 8, 2008, and Decree No. 6,899 of July 15, 2009), as well as the guidelines of the National Council for the Control of Animal Experimentation (CONCEA). The protocol was approved by the Ethics Committee on the Use of Animals of the Federal University of São Carlos (CEUA/UFSCar) on June 5, 2024 (protocol CEUA No. 5503040324, ID 002047), under the responsibility of Mario Cesar Garcia Duarte, covering the production, maintenance, and/or use of animals belonging to the phylum Chordata, subphylum Vertebrata (except humans) for research or teaching purposes.

Samples of 500 g of soil (0–20 cm depth) and samples of Coastcross and *U. decumbens* (including roots) were collected from nine points in each experimental area. In the laboratory, the aerial portions of each plant species (stems and leaves) were separated from roots and oven-dried at 65 °C. Blood samples were collected by venipuncture into BD Vacutainer® tubes containing anticoagulant. Urine samples from mares were collected using a flexible urethral catheter (Jorvet® SKU J0090SM). In geldings, urine was collected in a plastic beaker attached to a wooden handle during spontaneous urination. All samples were stored at –18 °C.

Table 1. Timeline of experimental events between day zero (D0) and day 55 (D55).

Events	
D0	Soil and Plant Collection
D1	Herbicide sprayed
D30	Collection of environmental and animal samples + animal entry into experimental areas
D35	Environmental and animal collections
D40	Environmental and animal collections
D45	Environmental and animal collections
D50	Environmental and animal collections
D55	Environmental and animal collections

Sample processing

Solid samples were ground to 710 μm particle size and all samples were digested with HNO_3 and HCl according to Krug (2019). Arsenic concentrations were determined by microwave plasma–atomic emission spectrometry (MP-AES) using an Agilent 4200 MP-AES (Agilent Technologies Inc., Santa Clara, CA, USA) at the Soil Laboratory of UFSCar, Sorocaba campus. Calibration curves were prepared from dilutions of a mono-element arsenic standard solution (Arsenic Standard 1 g L^{-1} , Sigma-Aldrich®, Cajamar, São Paulo, Brazil). Calibration checks were performed after every 30 samples.

Table 2. Average digestion time for each sample. HNO_3 (Nitric Acid) and HCl (Hydrochloric Acid).

Amostra	Tempo médio em HNO_3 (horas)	Tempo médio em HCl (horas)	Tempo total (horas)
Leaves and stems	24	12	36
Roots	24	12	36
Blood	24	12	36
Urine	24	8	32

Data analysis

Differences between the two groups (MSMA and Control) were evaluated using Student's t-test for independent samples. Data distribution was assessed with the Shapiro–Wilk test. Homogeneity of variances was verified with Levene's test, which indicated that variances could be considered homogeneous (Triola 2006; Montgomery and Runger 2014; Field 2017).

Results

Environmental samples

A total of 63 soil samples were collected over seven different sampling periods, and no arsenic (As) was detected in samples from the MSMA area. In the Control area, 18 samples were collected, and no As was detected using the methodology applied in this study.

For root samples, in the MSMA area, 63 samples of Coastcross-1 (nine samples across seven different sampling times) were analyzed and only two samples showed detectable As concentrations, one with 6.40 ppm at D50 and another with 3.60 ppm at D55. For *U. decumbens*, among 63 collected samples, only one showed detectable As concentration (2.29 ppm), observed at D0. In the Control area, where 18 Coastcross-1 root samples and 18 *U. decumbens* root samples were collected (totaling 36 samples), none exhibited detectable As concentrations.

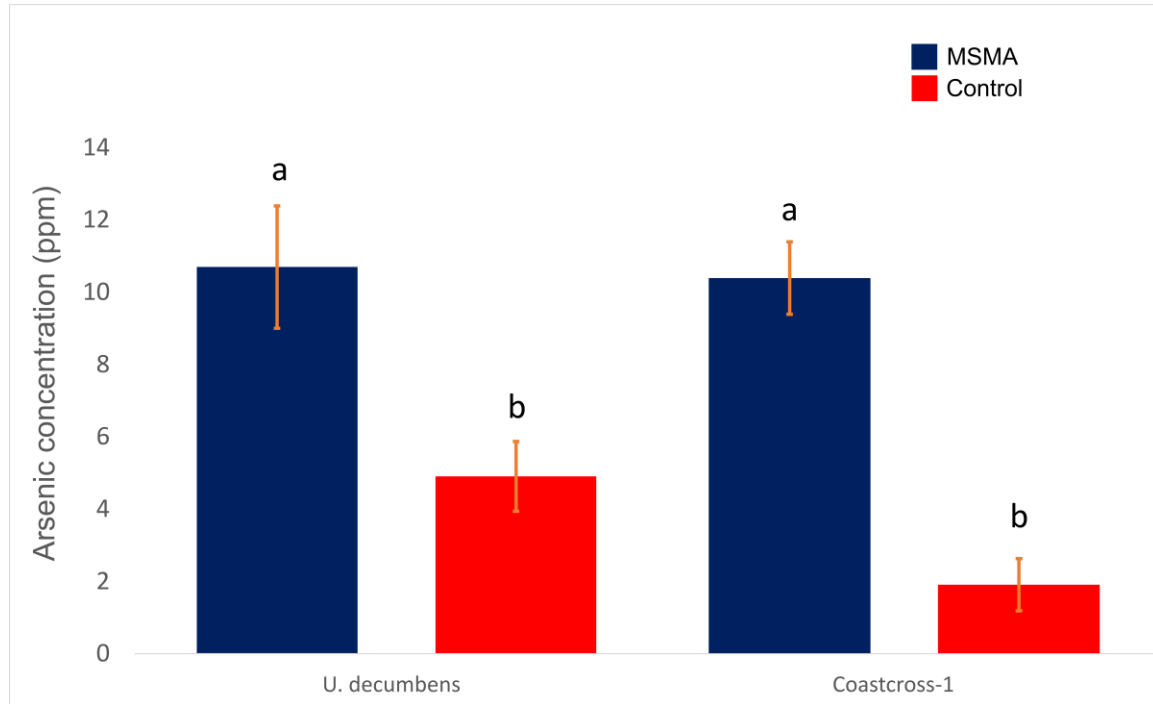
In the aerial part of plants (stems and leaves), arsenic was detected in the MSMA and Control areas for both Coastcross-1 and *U. decumbens*. The aerial parts of *U. decumbens* plants collected from the MSMA area had a mean As concentration of 10.69 ± 1.69 ppm, whereas the mean for plants from the Control area was 4.9 ± 0.96 ppm ($p < 0.05$) (Figure 3). Furthermore, the aerial parts of Coastcross plants from the MSMA area accumulated higher As concentrations, with a mean of 10.73 ± 1.22 ppm, compared to 1.96 ± 0.41 ppm in plants from the Control area ($p < 0.05$) (Figure 4). The As concentrations detected at different sampling periods are shown in Table 3.

Table 3. Comparison of arsenic concentrations in the aerial parts of Coastcross-1 with those in the aerial parts of *U. decumbens* plants collected from the MSMA area.

	D0	D30	D35	D40	D45	D50	D55
Coastcross-1	9.00 ± 1.46^a	5.40 ± 1.37^a	6.87 ± 1.73^a	5.97 ± 1.99^a	25.33 ± 2.74^a	$12.66 \pm 2.18^{a*}$	9.55 ± 1.49^a
<i>U. decumbens</i>	8.26 ± 2.07^b	8.66 ± 0.47^b	3.80 ± 0.95^b	9.53 ± 1.82^b	02.06 ± 1.10^b	$26.20 \pm 6.98^{a*}$	14.4 ± 2.77^b

Means are followed by standard deviation and different lowercase letters in the columns indicate significant differences ($p < 0.05$) between aerial parts of Coastcross-1 and *U. decumbens*. Means followed by “*” differ under $p < 0.01$.

Figure 4. Comparison of mean arsenic concentrations in the aerial parts of *U. decumbens* from the MSMA area and the Control area.



Different letters above bars indicate significant differences between groups ($p < 0.05$) for each plant, separately.

Animals samples

The mean As concentration in the blood of animals grazing in the MSMA area was 1.67 ± 0.04 ppm, significantly higher ($p < 0.05$) than the mean concentration in animals from the Control area 1.35 ± 0.05 ppm). Comparisons across different experimental periods are presented in Table 4. In agreement, animals grazing in the MSMA area presented a mean urinary As concentration of 1.62 ± 0.18 ppm, compared to 1.19 ± 0.11 ppm in animals from the Control area ($p < 0.05$). Comparisons across different samples and experimental periods are presented in Table 4.

Table 4. Comparison of arsenic concentrations (ppm) in the blood and urine of animals from the MSMA group and the Control group across different experimental periods.

		D30	D35	D40	D45	D50	D55
Blood	MSMA	$2.04 \pm 0.08^{a*}$	$1.74 \pm 0.14^{a*}$	1.84 ± 0.09^a	1.56 ± 0.11^a	1.76 ± 0.07^a	$1.70 \pm 0.07^{a*}$
	Control	$1.44 \pm 0.03^{b*}$	$1.23 \pm 0.14^{b*}$	1.36 ± 0.22^b	1.26 ± 0.03^b	1.75 ± 0.18^a	$1.16 \pm 0.18^{b*}$
Urine	MSMA	$2.82 \pm 0.91^{a*}$	1.72 ± 0.04^a	1.28 ± 0.14^a	1.69 ± 0.08^a	0.83 ± 0.01^a	1.31 ± 0.08^a
	Control	$0.66 \pm 0.18^{b*}$	2.46 ± 0.18^b	1.28 ± 0.01^a	1.26 ± 0.03^b	0.59 ± 0.11^b	1.21 ± 0.03^a

Means are followed by standard deviation and different lowercase letters in the columns indicate significant differences ($p < 0.05$) for each samples, blood and urine, separately. Means followed by “*” differ under $p < 0.01$.

Discussion

Soil

Analysis of arsenic (As) concentrations in equine pasture soils following the application of MSMA herbicide revealed levels below the detection limit ($<LOD$) at all sampling periods, indicating no detectable accumulation of As in the soil. These results suggest that the bioavailable As from MSMA applied at a rate of $3,600 \text{ g ha}^{-1}$ (equivalent to $1,644.93 \text{ g ha}^{-1}$ As) was insufficient to be detected by the analytical method employed or that immobilization processes in the soil occurred.

The absence of detectable As may be associated with several factors. One consideration is the application rate. Older studies report the use of much higher MSMA rates, which resulted in detectable soil As levels (Eldan and Masue-Slowey 2024), sometimes dozens of times greater than in this experiment.

Sorption of arsenic compounds in soils is highly dependent on mineralogical composition, pH, and organic matter (Bowell 1994; Sadiq 1997; et al. 2002; Shimizu et al. 2010; Campbell and Nordstrom 2014; Han et al. 2025). Chemical precipitation processes can occur, especially in the presence of phosphates and carbonates, which compete for adsorption sites and promote the formation of less soluble species (Dixit and Hering 2003; Nadagouda et al. 2024). These factors may lead to downward migration of As beyond the 0–20 cm sampling depth. In agricultural areas where MSMA is applied, As leaching has been observed in light-textured, well-drained soils, particularly during periods of high rainfall, when water percolation facilitates transport to groundwater (Welch et al. 2000; Reddy et al. 2010).

As has a high affinity for Fe oxides and hydroxides, which may be responsible for the co-precipitation of inorganic As in soils (Smedley and Kinniburgh 2002). Under typical MSMA spray conditions, the movement of its metabolite MMA into deeper soil layers is restricted (Willians et al. 2024). Within hours to days after herbicide application, and before sorption in the rhizosphere occurs, certain soil microorganisms may metabolize a fraction of the dissolved MMA, methylating it into DMA or demethylating it into arsenate. These processes, along with “aging” reactions, can immobilize MMA

metabolites (Eldan and Masue-Slowey 2024). Aging is a chemical phenomenon in which molecules transition to a more stable, lower-energy state (Barret et al. 1951).

In aerated soils, As^{5+} predominates, showing low solubility and a tendency to complex with soil components, limiting root availability (Dixit and Hering 2003; Fendorf et al. 2010). In poorly oxygenated or anoxic soils, As^{5+} can be reduced to As^{3+} , which is more soluble and more easily absorbed by plants via aquaporin channels (Meharg and Hartley-Whitaker 2002). In the studied areas, management practices favor high soil oxygenation, potentially limiting As^{3+} formation (Alam 2024).

Another hypothesis is that the herbicide did not reach the soil directly at the time of application. Mature plants can create a “canopy effect,” intercepting herbicide droplets in their aerial parts (Hewitt et al. 2024). This is more common in areas with mature broadleaf plants (EMBRAPA 1999). Although no broadleaf species were present in the treated area, a dense layer of plant matter covering the soil may have reduced MSMA penetration to the surface.

CONAMA Resolution No. 420/2009 establishes a reference value of 13 mg kg^{-1} for agricultural soils in Brazil and 35 mg kg^{-1} for industrial soils. The United States Environmental Protection Agency (EPA 2014) considers soils with $>24 \text{ mg kg}^{-1}$ As as potentially contaminated, while in the European Union, thresholds vary between 10 and 25 mg kg^{-1} . Arsenic concentrations in uncontaminated soils generally range from 0.1 to 40 mg kg^{-1} , with a global average estimated at $5\text{--}10 \text{ mg kg}^{-1}$ (Smedley and Kinniburgh 2002; Kabata-Pendias and Pendias 2011). The maximum permissible level in drinking water is 10 ppb according to the WHO (2019), US EPA (EPA 2014), and the European Union.

An additional relevant aspect is the volatilization of organic As. MSMA, being an organoarsenic compound, may undergo microbial methylation and subsequent volatilization, reducing its detectable residual concentration (Cullen and Reimer 1989; Fulk and Ratanpal 2025). Under aerobic conditions and in soils with high microbial activity, As can be transformed into gaseous forms such as trimethylarsine (TMAs), which dissipate rapidly into the atmosphere (Bentley and Chasteen 2002).

Overall, the absence of detectable As may indicate environmentally favorable conditions, suggesting that, under the specific conditions of this study, MSMA sprayed did not result in significant accumulation of bioavailable As in equine pastures. However, complementary assessments of groundwater and biota are recommended to fully exclude possible environmental impacts (Kabata-Pendias and Pendias 2001; Matterson et al. 2014; Mahoney et al. 2015; Miner and Cohen 2023).

Plants

The detection of As in aerial parts of Coastcross-1 (*Cynodon dactylon* cv. Coastcross) and *Urochloa decumbens*, along with its absence in roots, raises questions about the mobility and bioaccumulation of this element in plant tissues following MSMA application. This phenomenon can be explained by soil physicochemical processes, mechanisms of As uptake and translocation within plants, and interactions with the rhizosphere (Garcia-Rodriguez et al. 2024; Thongnok 2025). In soils, As occurs mainly as inorganic arsenate (As^{5+}) and arsenite (As^{3+}) (Kamal and Miah 2021; Rikta 2025). Arsenate is structurally similar to phosphate (PO_4^{3-}) and can be absorbed via phosphate transporters (Zhao et al. 2009), explaining why plants may accumulate As even when soil levels are low and undetectable by the analytical method used here. However, its mobility is reduced due to strong adsorption to Fe and Al oxides (Smith et al. 1998), which may explain its absence in roots.

The detection of As in aerial parts indicates that, once absorbed, it was translocated to shoots and leaves via the xylem (Zhu et al. 2025). Compared to arsenate, arsenite has greater mobility within plants and can be actively transported via the phloem (Abedin et al. 2002; Shri et al. 2024; Rajendran et al. 2024). The conversion of As^{5+} to As^{3+} within roots facilitates transport, as As^{3+} is less retained in root tissues. Certain biofilm-forming rhizospheric bacteria produce reductive compounds that enhance this conversion, increasing phytoextraction (Tournay et al. 2023).

U. decumbens accumulated more As in aerial parts (26.2 mg kg^{-1}) than Coastcross-1 (12.66 mg kg^{-1}) and did not accumulate As in roots in treated areas. This may relate to morphological and physiological differences between the species. These findings contrast with Rahmanret et al. (2007), who reported that *Cynodon* spp. species have denser and deeper root systems, favoring As retention in roots and limiting aerial translocation. Conversely, Lombi et al. (2002) showed that *Urochloa* species have higher tolerance to heavy metals and toxic elements, possibly enabling greater As translocation to shoots.

As detoxification mechanisms in plants may include As^{3+} methylation and complexation with thiol-containing compounds such as phytochelatins (Raab et al. 2007; Khan et al. 2024). Vacuolar sequestration and excretion via trichomes are strategies some grasses employ to reduce As toxicity in metabolically active tissues (Zhao et al. 2010; Ancheta et al. 2020; Nabi et al. 2021). Despite advances, significant gaps remain in understanding the molecular basis of As tolerance in different plant species (Guilherme et al. 2005; Sharma et al. 2021).

Animals

Blood and urine As levels are indicators of recent exposure, reflecting circulatory transport and immediate excretion (Demissie et al. 2024). MSMA, as an organoarsenic herbicide, is rapidly absorbed and metabolized, resulting in transient increases in these compartments. After exposure to organoarsenic compounds, As is converted into inorganic and methylated species and excreted mainly via urine (Vahter 2002; Drobna et al. 2009).

Blood As concentrations in MSMA group ranged from 1.56 to 2.04 mg L⁻¹, while Control group ranged from 1.23 to 1.75 mg L⁻¹. No clinical signs of intoxication or behavioral changes were observed in either group. Thus, the levels observed in this study were below toxic threshold.

As toxicokinetics in horses are similar to other species: after absorption, As is distributed via the bloodstream and eliminated primarily in urine, with minor deposition in keratinized tissues such as hooves and hair (Vahter 2002; Gupta and Gupta 2025). The absence of clinical signs suggests efficient elimination preventing prolonged accumulation in target organs.

Arsenic level in urine is an excellent marker of arsenic intoxication and has been used to identify exposure in animal populations (Escudero, 2016). Moreover, urine analysis has proven to be a practical and efficient method for detecting arsenic exposure

Ingestion of As via feed or water causes severe poisoning and milder liver lesions (Khaleda et al. 2025). In humans, consumption of As-rich rice increased serum AST and ALT but did not produce severe poisoning (Hosen et al. 2016). Foodborne As exposure can also cause gut dysbiosis and metabolic imbalance (Shao et al. 2024).

The higher blood As in the MSMA group likely resulted from greater ingestion via pasture, as Coastcross-1 and *U. decumbens* from MSMA area contained 9.63 mg kg⁻¹ and 12.66 mg kg⁻¹ As, respectively. Despite this, concentrations remained below toxic thresholds, aligning with studies suggesting that chronic low-dose As exposure may be tolerated in some species with efficient metabolism and excretion (Meharg and Hartley-Whitaker 2002).

Clinical signs of As intoxication in horses include hypersalivation, muscle tremors, ataxia, and watery diarrhea (Casteel 2021). Although absent in this study, prolonged exposure to elevated As levels could lead to subclinical or chronic effects (Balali-Mood et al. 2025). As can accumulate in keratin-rich tissues, potentially affecting structural integrity over time (Mandal and Suzuki 2002). Even exposures considered safe may induce oxidative stress, alter cellular functions and increasing the risk of inflammatory or degenerative changes (Flora et al. 2008). Additionally, As can act as an endocrine

disruptor, affecting hormonal and immune regulation in chronic exposures (Hughes 2002).

In the present study, the lack of clinical signs suggests that homeostatic mechanisms maintained As within safe limits during the evaluation period.

Conclusion

Arsenic levels in experimental soils were below the detection limit of the methods and equipment used. *U. decumbens* accumulated As in amounts sufficient to cause plant death, whereas Coastcross-1 accumulated amounts that caused mild, reversible phytotoxicity. Plants also accumulated As in untreated areas, likely from previous MSMA applications. Root As levels were below detection. The discrepancy between low soil As and high plant As suggests specific dynamics of As uptake and translocation. Blood and urine As concentrations were significantly higher in animals grazing treated pastures, indicating accumulation during the experiment, although no physical or behavioral changes were observed. MSMA effectively controlled *U. decumbens* without detrimental effects on Coastcross-1, suggesting possible physiological or biochemical resistance mechanisms. While no signs of toxicity were observed in the horses, chronic exposure warrants monitoring to detect possible subclinical effects. These findings highlight the need for further research into MSMA impacts on *Cynodon*-based pastures invaded by *Urochloa decumbens*.

Acknowledgements

The authors would like to acknowledge the Federal University of São Carlos and the Center for Science and Technology for Sustainability for their support and for providing access to their laboratories.

Author Contributions

MCGD. and JBC were involved in the conceptualization; MCGD and MCB contributed to the methodology; MCGD. performed writing—original draft; all contributed to writing—review and editing.

Funding

The authors declare that no funds, grants, or other support were received during the preparation of this manuscript.

Data Availability

Data are available on reasonable request from the authors

Declarations

This manuscript was originally written in Portuguese. The author acknowledges that internet-based resources, including artificial intelligence tools, were used to assist in its translation into English

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