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RESEARCH ARTICLE

Co-invasion of the invasive *Aporrectodea caliginosa* and the newly recorded *A. rosea* (Crassiclitellata, Lumbricidae) in soils of southern Western Siberia: changes in soil salinity and water-soluble ion availability in a field experiment

Kirill A. Babiy[1], Stanislav Yu. Kniazev[1], Anna S. Abramenko[1], Vladislav E. Shcherbakov[1], Elena V. Golovanova[1]

1 *Omsk State Pedagogical University, 14 Tukhachevskogo Emb., Omsk, 644099, Russia*

Corresponding author: Kirill A. Babiy (labinvert@omgpu.ru)

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Abstract

Secondary salinisation and solonetzicity of soils are among the most pressing environmental problems in Western Siberia, where saline soils occur in almost every administrative region. Bioremediation using earthworms is a promising alternative to costly conventional reclamation, but most evidence comes from laboratory studies. Two European invaders, *Aporrectodea caliginosa* and the recently discovered *Aporrectodea rosea*, inhabit the soils of southern Western Siberia. This study aimed to assess, through a field experiment at four sites, the effects of the two species on the ionic composition of the meadow chernozem solonetzic and solonchak soils and on the sodium adsorption ratio (SAR). We conducted the field experiment over 84 days on four plots in Omsk Oblast, using a complete randomised block design with four treatments: *A. caliginosa* monoculture, *A. rosea* monoculture, their association, and a control without earthworms. Cation and anion concentrations were measured by capillary electrophoresis. The effects of earthworms were layer-specific and species-specific. In monoculture, *A. rosea* significantly reduced SAR in the 20–30 cm layer, while *A. caliginosa* showed no such effect. The association of the two species resulted in a significant increase in potassium, magnesium, and calcium across all layers, an effect absent in either monoculture. These patterns are interpreted as

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a synergistic outcome of bioturbation by species that differ in their mode of action on the pore space. Our findings confirm the potential of *A. rosea* as a bioremediation agent for saline soils in Western Siberia and provide a basis for extrapolating the results.

Keywords
Earthworms, invasive species, Omsk Oblast, soil bioturbation, soil salinisation, sodium adsorption ratio

Introduction

In Western Siberia, the problem of secondary soil salinisation and solonetzisation is acute, caused by poor drainage, the shallow occurrence of mineralised groundwater, and the disruption of hydrological regimes due to economic activities (Elizarov et al. 2020). Saline soils are widespread in all administrative units of the region, except Tomsk Oblast (Chernousenko et al. 2025). The consequences of salinisation for agriculture include reduced productivity, an increased risk of less tolerant crops, and a loss of biodiversity (Tao et al. 2024).

Traditional technologies for the remediation of saline soils are effective, but are often limited by their complexity, high cost, inefficient use of water resources, and the low sustainability of the results obtained (Nosova et al. 2024; Duan et al. 2026). Under these conditions, bioremediation, which is distinguished by its low level of pollution, high efficiency, and long-lasting effect, is of particular relevance. Among bioremediation agents, earthworms are among the most widespread soil invertebrates, known as 'ecosystem engineers'. They play a key role in improving saline soils through bioturbation (Wu et al. 2025; Duan et al. 2026).

The ability of earthworms to positively influence saline soils has been predominantly confirmed in laboratory studies. The addition of vermicompost from *Pheretima guillemi* (Michaelsen, 1895) significantly reduced the soil sodium adsorption ratio (SAR) when the soil was irrigated with brackish water (Li et al. 2021). The invasive *Aporrectodea trapezoides* (Duges, 1828) is considered an effective agent for the bioremediation of saline soils in the Bohai Bay region of China (Tao et al. 2024), while *Eisenia fetida* (Savigny, 1826), in combination with arbuscular mycorrhizal fungi, improves maize growth and enhances its salt tolerance by regulating the microbial community and increasing the proportion of macroaggregates in the rhizosphere (Zhang et al. 2018, 2019). However, the direct transfer of laboratory results to field conditions requires critical verification, as the direction of the effects can be significantly altered under the influence of natural factors. For example, the multiple increase in phosphorus availability observed in the laboratory (Ros et al. 2017) was substantially attenuated in the field, and the key factor in the mobilisation of phosphorus under natural conditions was not the species diversity of earthworms, but the presence of specific bioturbating species (Vos et al. 2022). Therefore, reli-

able and practically significant data on the effect of earthworms on saline soils can primarily be obtained through field research.

In the south of Western Siberia, the invasive European earthworm *Aporrectodea caliginosa* (Savigny, 1826) has become widespread (Kniazev et al. 2022). This salt-tolerant species (Castleberry et al. 2025) has potential as a bioremediation agent: it increases the availability of Ca2+ and Mg2+ (Babiy et al. 2021, 2022, 2025), which promotes cation exchange with Na^+ and mitigates the alkalinity of saline soils (Li et al. 2024; Liu et al. 2026), while an increase in the proportion of large aggregates (>2 mm) enhances salt leaching (Hedde et al. 2013). The recent discovery of a new European invader, *Aporrectodea rosea* (Savigny, 1826), in the north of Omsk Oblast (Golovanova et al. 2024), a species that almost always cooccurs with *A. caliginosa* in Europe (Jupke et al. 2024), suggests the potential for synergistic effects when they inhabit saline soils together in the region.

The aim of our study is to evaluate changes in soil ion availability, as well as to assess the alteration of soil salinity (sodium adsorption ratio), under the influence of the invasive species *A. rosea* and *A. caliginosa* under field conditions. Conducting field experiments in four spatially separated sites provides a statistically robust and internally consistent characterisation of the impact of *A. caliginosa* and *A. rosea* on solonetzic and solonchak soils. This experimental design minimises the risk that the observed patterns are merely a chance consequence of local conditions, providing a reliable basis for extrapolating the results to this soil type.

Materials and methods

Experimental sites and mesocosm design

The experimental sites were located within the Omsk Oblast in four plots with meadow chernozem solonetzic and solonchak chernozem soils (Gleyic Chernozem (Sodic) according to WRB, 2022): (1) Omsky District (2.1 km northeast of the village of Pokrovka, 55°21'15"N 73°25'14"E); (2) Kormilovsky District (2.4 km northeast of the village of 1-ya Fominovka, 54°49'11"N 74°13'30"E); (3) Azovsky District (2.4 km northeast of the village of Koshkarevo, 54°43'12"N 73°18'41"E); (4) Lyubinsky District (1.7 km northwest of the settlement of Vostochny, 55°08'16"N 72°45'43"E). The soil characteristics are presented in Table 1. The specimens used in the study were mature earthworms collected from two locations in the Omsk Oblast: *A. rosea* – from the vicinity of the village of Yekaterininskoye, Tarsky District (56°53' N, 74°35' E); and *A. caliginosa* – from the vicinity of the village of Chernoluchye, Omsky District (55°16' N, 73°02' E).

The experiment was carried out during the growing season from 22 July to 13 October 2025 (84 days) at the four sites. The experimental design at each site consisted of a uniform scheme, a randomised complete block design with three replications. The experiment included the following treatments: monocultures of *A. caligi-*

nosa, monocultures of *A. rosea*, an association of the two species (*A. caliginosa* + *A. rosea*) and a control treatment without earthworms. In total, 48 mesocosms were established, with 12 at each site.

Table 1. Characteristics of the gleyic chernozem (sodic) of experimental sites

Experimental sites (district)	Humus (%)	pH	Water-soluble ions (mg/kg)					
			K^+	Na^+	Mg^{2+}	Ca^{2+}	Cl^-	SO_4^{2-}
Omsky	11.1	6.95	18.57	4.60	4.27	119.4	9.04	27.03
Kormilovsky	11.2	6.75	36.37	6.56	17.57	6.95	5.12	5.94
Azovsky	9.49	7.3	54.83	16.97	55.83	6.47	9.77	12.47
Lyubinsky	8.7	8.2	107.3	49.03	18.30	80.20	10.16	14.77

Mesocosms were constructed from PVC pipes (length 300 ± 50 mm, diameter 110 mm), the bottom and top of which were sealed with mesh fabric (pore size 82 μm). The mesocosms were filled with soil from the upper humus horizon (the first 30 cm), which was premixed and sieved through a 4 mm mesh to remove macrofauna. The mesocosms were buried in the soil so that the soil surface within the mesocosms was level with the surrounding ground surface.

For the experiments, 108 mature individuals of each earthworm species were collected (36 individuals per site). Each mesocosm (except for control treatments) received four mature individuals for monoculture treatments and two individuals of each species for the two-species association treatments. The mean biomass of individual earthworms in the mesocosms was 0.42 ± 0.04 g for *A. caliginosa* and 0.13 ± 0.01 g for *A. rosea*.

Sample collection and analytical methods

Upon completion of each experiment, the mesocosms were extracted from the soil and dismantled layer by layer at 10 cm intervals. Soil samples for analysis were collected from each layer and the earthworms were preserved. Each site was considered a replicate; samples from within a site were pooled into composite samples. The mass of the earthworms was determined using an Adam HCB-1002 balance (Adam, UK).

The concentrations of water-soluble cations (Na^+, K^+, Mg^{2+}, Ca^{2+}) and anions (Cl^-, SO_4^{2-}) in the soil samples were determined by capillary electrophoresis using a Capel-104T system (Lumex, Russia), following the certified methodologies Lumex Method M 03-08-2011 (for cations) and Lumex Method M 03-06-2010 (for anions). Extracts were prepared from the soil samples in deionised water in a soil-to-water ratio of 1:5. The filtered extracts were then analysed using the capillary electrophoresis system. The analysis parameters were as follows: capillary length 60 cm, internal diameter 75 μm; sample injection at 30 mbar×s for 5 s; separation voltage

25 kV; temperature 20°C; and detection at a wavelength of 254 nm. The leading electrolyte for the cations contained benzimidazole, tartaric acid, and 18-crown-6; for the anions, it contained chromium trioxide, diethanolamine, and cetyltrimethylammonium hydroxide. The experimental data were processed using Multichrom and Elforun software for Windows.

The pH of soil water extracts (1:5 ratio) was measured using a Hanna Edge HI 2002-02 pH meter (Hanna Instruments, USA) according to GOST 26423-85.

The sodium adsorption ratio (SAR) was calculated using the following equation (Liu et al. 2026):

$$SAR = \frac{Na^{+}}{\sqrt{\frac{Ca^{2+} + Mg^{2+}}{2}}},$$

where Na^{+}, Ca^{2+}, and Mg^{2+} are the concentrations of Na^{+}, Ca^{2+}, and Mg^{2+} in the soil water extract.

Statistical analysis

The normality of the data distribution was assessed using the Shapiro-Wilk test. All data were transformed using the natural logarithm of the response ratio (LnRR), where LnRR (variable) = ln (soil property under the influence of earthworm or species association / background soil property at the site). A LnRR < 0 indicates a negative effect of earthworms on ion availability and SAR, whereas a LnRR > 0 indicates a positive effect. The statistical significance of the effects of treatment on the composition of soil cations was evaluated using one-way analysis of variance (ANOVA), followed by pairwise comparisons using Tukey's HSD test at a significance level of $p = 0.05$. To assess changes in SAR, the nonparametric Kruskal-Wallis test was used. All statistical analyses were performed with Statistica 13.0.

Results

The results presented below demonstrate that, under the influence of earthworms, the direction and magnitude of changes in soil ionic composition relative to initial concentrations (background levels) at the study sites depend substantially on soil horizon depth and species composition.

In the 0–10 cm layer, ANOVA revealed statistically significant changes relative to the background levels only for chlorides (Table 2). A trend towards changes in magnesium ($p = 0.088$) and sulfate ($p = 0.080$) ion concentrations was also detected (Fig. 1). The combined presence of the two earthworm species (*A. rosea* + *A. caliginosa*) resulted in a statistically significant change in chloride levels compared to the

A. caliginosa monoculture (p = 0.036) and the control (p = 0.029) (Fig. 1), but did not differ significantly from the monoculture of *A. rosea* (p = 0.195). No significant differences were recorded between the monocultures or between either monoculture and the control.

Table 2. Results of a one-way analysis of variance (ANOVA) of the effect of earthworms on changes in the ion content (LnRR = ln(treatment/background)) in soil layers

Soil depth (cm)	Ion					
	K^+	Na^+	Mg^{2+}	Ca^{2+}	Cl^-	SO_4^{2-}
0–10	2.03	0.27	2.76	1.56	4.62*	2.89
10–20	4.56*	1.18	1.08	1.05	4.62*	2.15
20–30	1.14	0.1	3.55*	4.65*	0.57	1.07

Note: * indicates a statistically significant difference (p < 0.05).

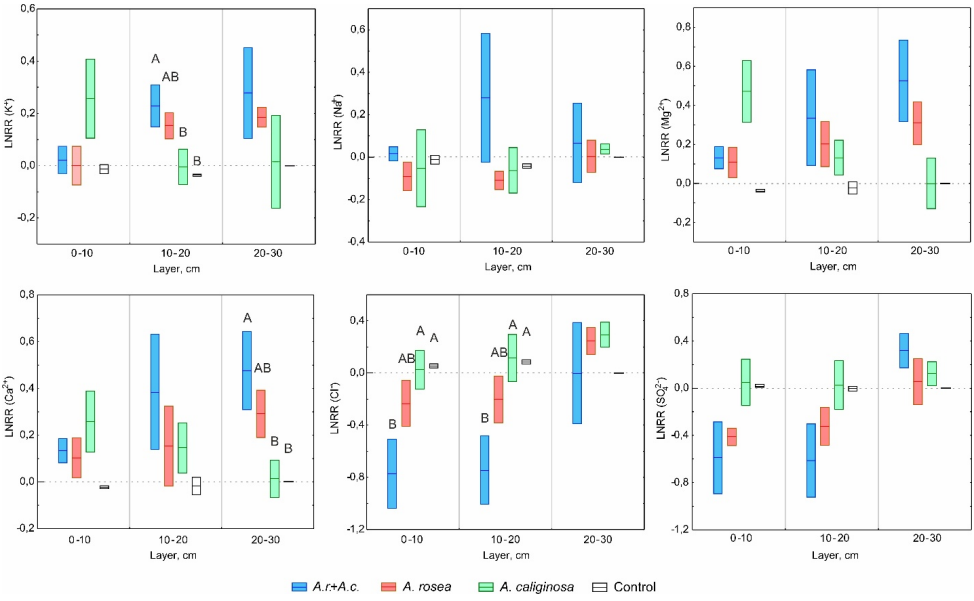

Figure 1. Intensity of the effect of earthworm species (LnRR = ln(treatment/background)) on the ionic composition of the soil. Comparisons within earthworm species are indicated in capital letters. Treatments with different letters are significantly different (p = 0.05). The dashed line represents the zero effect of the earthworm (LnRR = 0). The horizontal line in each box marks the mean LnRR, and the box shows the mean ± standard error (SE).

To assess the potential for soil desalinisation, the sodium adsorption ratio (SAR) was calculated, as well as the response ratio relative to baseline values at the sites (LnRR). In the upper (0–10 cm) and middle (10–20 cm) soil layers, no statistically significant differences in SAR were found between the experimental groups

(Table 3). For the lower layer (20–30 cm), significant differences were established (p = 0.019). The pairwise comparisons were then made using a post-hoc test for more detailed analysis (Fig. 2). In the 20–30 cm layer, a statistically significant difference in SAR was found between the *A. rosea* monoculture and the control group (p = 0.045; see Table 3). The differences between *A. rosea* and *A. caliginosa* were close to the significance level (p = 0.086). The combined presence of the two species (*A. rosea* + *A. caliginosa*) did not cause significant changes in SAR compared to the control or either of the monocultures (p > 0.05).

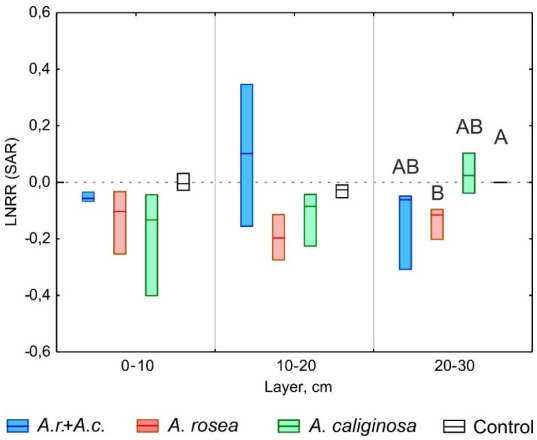

Figure 2. Change in the sodium adsorption ratio (SAR) under the influence of earthworms (LnRR = ln(treatment/background)). Comparisons within earthworm species are indicated in capital letters. Treatments with different letters are significantly different (p = 0.05). The dashed line represents the zero effect of the earthworm (LnRR = 0). The horizontal line in each box marks the median LnRR, and the box extends from the 25th to the 75th percentile.

Table 3. Results of the Kruskal–Wallis test for the SAR index in soil layers

Soil depth (cm)	H	df	p
0–10	2.493	3	0.477
10–20	5.360	3	0.147
20–30	**9.993**	3	**0.019**

Note: Significant differences at p < 0.05.

Regardless of treatment, earthworms increased their biomass. In monocultures, *A. caliginosa* gained on average 177%, while *A. rosea* gained 357%; in the two-species treatment, *A. caliginosa* gained 18% and *A. rosea* gained 99%. The distribution of mean biomass and the density of earthworms across the three layers within the mesocosms at the end of the experiment was also analysed for the four study sites.

Table 4. Average biomass and density of *Aporrectodea caliginosa* and *A. rosea* per mesocosm and per square meter in different soil layers under monospecific and co-occurring conditions at the end of the experiment

Treatment	Species	Soil depth (cm)	Biomass (g)	Biomass (g/m^2)	Abundance (ind.)	Density (ind./m^2)
monocultures	*A. caliginosa*	0–10	0.25	26	0.42	44
		10–20	0.78	82	1.00	105
		20–30	1.32	139	2.08	219
	A. rosea	0–10	0.12	13	0.42	44
		10–20	0.32	34	1.17	123
		20–30	0.46	48	1.25	132
two-species associations	*A. caliginosa*	0–10	0.33	34	0.63	66
		10–20	0.68	72	1.17	123
		20–30	0.63	66	1.00	105
	A. rosea	0–10	0.08	8	0.33	35
		10–20	0.15	16	0.67	71
		20–30	0.02	2	0.08	8

It should be noted that the values of both the biomass and the density of *A. rosea* were lower than those of *A. caliginosa* in all layers (Table 4). In monoculture, both *A. rosea* and *A. caliginosa* were distributed throughout the mesocosm; however, in the association of two species, the presence of *A. caliginosa* resulted in the displacement of *A. rosea* toward the upper horizons, along with an overall reduction in its abundance relative to monoculture.

Discussion

According to current understanding, the bioturbation activity of earthworms is a key factor that determines their influence on the physical properties and chemical composition of the soil (Gutierrez and Jones 2006; Capowiez et al. 2024). However, the direction of these changes is not universal and depends primarily on the earthworm species and soil type (Babiy et al. 2021, 2022; Clause et al. 2014) – this may explain the consistent difference between our results for meadow chernozem solonetzic and solonchak soils (Fig. 1 and Table 2) and data obtained for other soil types (Babiy et al. 2021, 2022, 2023, 2025). The ions most sensitive to earthworm activity, both in our study and according to published data (Resner et al. 2015; Richardson et al. 2022), were calcium, magnesium, and potassium. At least two mechanisms underlie this sensitivity. The first is the species-specific ability of *A. caliginosa* and *A. rosea* to produce CaCO3 granules (Canti & Piearce 2003; Versteegh et al. 2014),

which directly and differentially affects the availability of calcium and magnesium (Wu et al. 2020). The second relates to the gut microflora, which functions as a bioreactor and determines the species-specific composition of the casts and the availability of macronutrients (Vos et al. 2022); bacterial communities have been shown to differ even between related species *A. caliginosa* and *Aporrectodea tuberculata* (Eisen, 1874) (Sapkota et al. 2020).

It should be noted that, in our experiment, the effects of *A. rosea* and *A. caliginosa* were manifested in a layer-specific manner (Fig. 1). Such stratified dynamics are in good agreement with the literature and are usually attributed specifically to the bioturbation activity of earthworms: the meta-analysis by Ferlian et al. (2018) demonstrated that invasive earthworms deplete the organic layer of carbon and nitrogen, enriching the mineral horizons with these elements; Similar redistribution of Ca, Mg, and K was described by Resner et al. (2015); and *Lumbricus terrestris* Linnaeus, 1758 doubled the inorganic nitrogen content in the 13–39 cm layers compared with the upper 0–13 cm layer (Amador and Gorres 2005).

The vertical distribution of bioturbation activity also depends on climatic factors: Ter Borg et al. (2026) showed a positive correlation between burrowing intensity and soil moisture: during periods of moisture deficit at the beginning of the season, burrows were formed predominantly at greater depths, while as moisture increased, the zone of activity shifted toward the surface. The precipitation dynamics during our field experiment, according to (Pogoda… 2025) – with a maximum in July (134 mm), decreasing to 75 mm in August, 18 mm in September, and 34 mm in October – could have contributed to the displacement of earthworms towards deeper, better moisture-retaining layers. It was in these deeper layers that the spectrum of significantly altered ions shifted qualitatively, from chlorides and potassium to magnesium and calcium (Fig. 1), which is consistent with the stratified effects described above. Similar species-specific and layer-specific differences have also been observed in our earlier research: for Na^+, Mg^{2+}, and Ca^{2+} content in *Eisenia nana* Perel, 1985 and *Eisenia nordenskioldi* (Eisen, 1878) (Babiy et al. 2023), and for the pair of native *Eisenia nordenskioldi pallida* (Malevic, 1956) and invasive *A. caliginosa*, in terms of the uneven distribution of K^+, Mg^{2+}, and Ca^{2+} (except for NH_4^+) across layers (Babiy et al. 2025).

A key finding of the present study is that statistically significant effects on chlorides, as well as increases in potassium, magnesium, and calcium content, were manifested precisely in the presence of both *A. rosea* and *A. caliginosa* together (Fig. 1). This differs fundamentally from the results obtained previously for another pair of species – the native *E. n. pallida* and the invasive *A. caliginosa* – in meadow (Babiy et al. 2025): their cooccurrence did not yield an additive sum of effects, but rather led to a qualitatively different result - a significant reduction in the content of water-soluble potassium, calcium, and magnesium ions.

This difference in the nature of the effects appears to be related to the presence or absence of competitive relationships between species, which reduce their bioturbation activity. In the experiment with *E. n. pallida* and *A. caliginosa*, competition

was expressed as a decrease in survival and biomass of both species: although at the beginning of the experiment the biomass of *E. n. pallida* exceeded that of *A. caliginosa*, by its conclusion, in both mono- and two-species treatments, it was lower by 17 ± 2% and 89 ± 9%, respectively (Babiy et al. 2025); the decrease in biomass of both species across all treatments indicated competition between them. In the present experiment, in contrast, the earthworms increased in abundance and biomass across all treatments, regardless of species composition (Table 4). The absence of signs of suppression allows the observed effect to be interpreted as synergistic: two species with different patterns of bioturbation activity (Capowiez et al. 2015; Capowiez et al. 2024), when co-occurring, form a denser network of biopores and mix the humus horizon more intensively. This, combined with the simultaneous increase in biomass of both species, leads to a more intensive soil process overall. This is consistent with the view that competition between *A. caliginosa* and *A. rosea* is weak (Uvarov 2009) when they co-occur in nature (Jupke et al. 2024; Ter Borg et al. 2026), which is explained by their separation according to habitat depth and periods of peak abundance (Falco et al. 2015). The species studied also differ in their use of bacterial and fungal resources and in their response to root exudates (Zhong et al. 2024).

Alongside the effects on individual ions, notable species-specific differences were also evident in the integral salinity index, SAR (Table 3). Since *A. caliginosa* is considered a salt-tolerant species (Casteberry et al. 2025), and closely related *A. trapezoides* is an effective agent for the bioremediation of saline soils (Tao et al. 2024), we expected that *A. caliginosa* would be the species responsible for a reduction in SAR. However, in the upper horizons (0–10 and 10–20 cm), no significant changes in SAR were detected, and a significant reduction in the 20–30 cm layer was recorded only in *A. rosea* treatment (Fig. 2). This result was unexpected: we found no data in the literature on a positive effect of *A. rosea* on soil salinity; in contrast, salinisation of coastal agricultural soils has been shown to inhibit this species (Ivask et al. 2012).

This species-specific effect can, it seems, only be explained by taking into account the differences in the bioturbation activity of *A. rosea* and *A. caliginosa*, since the formation of macroaggregates and the alteration of soil porosity are considered among the main mechanisms by which earthworms affect salinity (Zhang et al. 2019; Li et al. 2021). *A. rosea* and *A. caliginosa* belong to different groups of bioturbators (Capowiez et al. 2024), which in itself accounts for differences in the nature of their impact on soil: they differ in burrow length and diameter (Capowiez et al. 2015), and the increase in the proportion of large macroaggregates (>2 mm) occurs to different degrees: 3.3 times for *A. caliginosa* and 6.6-fold for *A. rosea* relative to the control (Hedde et al. 2013). These differences in the transformation and, consequently, in the efficiency of salt leaching may explain the significant effect of *A. rosea* on SAR in the 20–30 cm layer, in contrast to the absence of a significant effect of *A. caliginosa*.

Importantly, the cooccurrence of *A. rosea* and *A. caliginosa* did not affect SAR on the lower horizon, despite a significant increase in calcium and magnesium content – this occurred because sodium content also increased simultaneously; a significant reduction in SAR was recorded only in the *A. rosea* monoculture (Fig. 2). Two explanations can be proposed for this. First, co-occurrence alters the vertical distribution of the species (Felten and Emmerling 2009): in the treatment of two species, *A. rosea* was not found in the layer 20-30 cm, while in the monoculture it was consistently present. Second, biomass dynamics may also play a role: The increase in *A. rosea* biomass in monoculture (357%) significantly exceeded that of *A. caliginosa* (177%), and larger earthworms produce more casts and have a greater effect on soil structure (Hedde et al. 2013). However, since the absolute values of *A. rosea* biomass and density were lower than those of *A. caliginosa* (Table 4), the decisive factor in the reduction of SAR may instead have been the specific nature of its bioturbation activity. Furthermore, *A. caliginosa* is capable of feeding on the castings of other species (Zangerlé et al. 2011) and, by reworking the aggregates created by *A. rosea*, can offset its contribution to the transformation of pore space, such that the two-species association approximates the *A. caliginosa* monoculture.

The weak expression of earthworm effects at the end of the experiment (Fig. 1) is probably related to its duration, 84 days. According to Zorn et al. (2005), statistically significant differences in the degree of soil mixing became apparent only on day 175, whereas at day 80 only initial signs of bioturbation were evident. This suggests that, with a longer experimental duration, the observed effects might become more pronounced.

Conclusion

The results of the present study expand our understanding of the ability of the invasive earthworms *A. rosea* and *A. caliginosa* in southern Western Siberia to alter the ionic composition of the meadow chernozem solonetzic and solonchak soils and to mitigate the negative impact of soil salinisation. It has been demonstrated that the effects of earthworms on soil ionic composition are layer-specific and depend on species composition.

It was established that *A. rosea* in monoculture is capable of significantly reducing the sodium adsorption ratio (SAR) of soil in the lower 20–30 cm layer in 84 days, whereas *A. caliginosa* does not exhibit such an effect. However, the cooccurrence of both species leads to a significant reduction in chloride content and an increase in potassium, magnesium, and calcium content – an effect not observed in either monoculture individually. The impact of the two species on soil ionic composition is not additive but rather qualitatively different, depending on the species composition of the community. The observed differences can be explained by the dissimilar capacities of *A. rosea* and *A. caliginosa* to transform soil pore space, as

well as by changes in the vertical distribution of earthworms under cohabitation compared with conditions in which they occur separately.

The results confirm the potential of earthworms, primarily *A. rosea*, as agents for the bioremediation of saline soils and suggest that they could be used for biomelioration under the conditions of Western Siberia. Further research should be directed towards assessing the long-term effects of separate and joint habitation by these species, as well as toward testing the identified patterns in other types of saline soils.

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