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Alteration of the Cesium-137 soil profile by wild boar rooting after the Fukushima Daiichi Nuclear Power Plant accident

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A R T I C L E I N F O

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A B S T R A C T

Large amounts of Cesium-137 (^{137}Cs, physical half-life ca. 30 years) were released into the environment from the 2011 Fukushima Daiichi Nuclear Power Plant (FDNPP) accident in Japan. Initially contaminants from the accident were deposited on the surface soil layers in terrestrial ecosystems, but over time vertical migration of ^{137}Cs has occurred, with most ^{137}Cs now distributed a few centimeters below the soil surface. Wild boar (*Sus scrofa*) rooting likely alters the distribution of contaminants in soil profiles. Such impacts may be particularly pronounced in areas contaminated from the FDNPP accident, given the increase in abundance of wild boar in these areas. Our objective was to determine whether rooting by wild boar alters the distribution of ^{137}Cs contamination within soil profiles through collection of soil cores from control sites and two different types of rooting samples, “new depressions” and “old depressions” within the difficult-to-return zone surrounding the FDNPP. The ^{137}Cs ratio [activity concentration of ^{137}Cs at each depth / the maximum activity concentration of ^{137}Cs in each core sample] rapidly decreased after reaching a peak at the control sites. However, the ^{137}Cs ratio at all rooted sites decreased more slowly, and ^{137}Cs was distributed deeper into the soil layers than control sites, across both new and old rooting sites. Moreover, the ^{137}Cs soil profiles varied greatly among core samples and sampling sites. These results demonstrate that soil disturbance from wild boar rooting can directly influence the distribution of contaminants within soil profiles. However, the extent to which deeper layers are affected by contamination depends on the unique characteristics (i.e., depth of rooting) of each wild boar rooting disturbance. Further studies are needed to examine the effects of long-term rooting on the alteration of ^{137}Cs in soil profiles.

1. Introduction

On 11 March 2011, a 9-magnitude earthquake struck off the northeastern coast of Japan and resulted in one of the most significant releases of anthropogenic radiological contamination in history. Cesium-137 (hereafter ^{137}Cs, physical half-life of ca. 30 years) was one of the main anthropogenic radiological contaminants released from the Fukushima Daiichi Nuclear Power Plant (FDNPP) accident, and it is estimated that approximately 13–15 PBq of ^{137}Cs was discharged into the environment (Chino et al., 2011; Povinec et al., 2013). Cesium-137 has a long physical half-life and high bioavailability, generating concerns regarding the long-term impact of ^{137}Cs contamination to ecosystems and wildlife (Koarashi et al., 2019; Saito et al., 2020). Cesium-137 contamination is of particular concern in terrestrial ecosystems. The mineral soil found within forest ecosystems can serve as large ^{137}Cs reservoirs; having the ability to store substantial amounts of ^{137}Cs after the radionuclide's initial deposition on the forest canopy and top organic soil leaches through to the mineral layer (Fujii et al., 2014; Hashimoto et al., 2013; Manaka et al., 2019). Similarly, grassland ecosystems can act as long-term ^{137}Cs reservoirs, and ^{137}Cs contamination has been reported on the surface soil layers in grassland ecosystems after both the FDNPP accident (Terashima et al., 2014) and the Chernobyl nuclear accident (Chamard et al., 1993). In pastures and agricultural lands, the amount of ^{137}Cs and its physicochemical fractions in the soil affect ^{137}Cs activity concentrations through root uptake. The degree to which plants absorb radionuclides from the soil is also dependent on the surrounding soil characteristics such as pH and cation exchange capacity (IAEA, 2010). Thus, a combination of ^{137}Cs behavior and soil characteristics affects the dynamics and circulation of ^{137}Cs in various terrestrial environments. Evaluating ^{137}Cs behavior in soil can be used to better understand ^{137}Cs dynamics in an ecosystem and the long-term radiation risks in an environment after a nuclear power plant accident (Koarashi et al., 2019).

Research from both the Chernobyl and FDNPP nuclear accidents indicate that ^{137}Cs can be absorbed in the topsoil layer where it can re-

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main for decades (Terashima et al., 2014; Almgren and Isaksson, 2006; Filipović-Vinceković et al., 1991; Konopleva et al., 2009). The amount of ^{137}Cs in soil depends on the soil type since the composition and content of organic matter, clay minerals, and mineral content with high Cs retention capacities vary by soil type (Yamaguchi et al., 2012; Yasutaka and Tsuji, 2013). Generally, after ^{137}Cs is deposited on soil it is fixed to the frayed edge sites found within illite or micaceous clay minerals (i.e., residual fixation fraction of ^{137}Cs), making clay content an important factor regulating ^{137}Cs retention in soil (Fujii et al., 2014; Manaka et al., 2019; Cremers et al., 1998). Cesium-137 reaches the mineral soil and moves slowly within the soil profile, gradually moving from the upper to the lower soil layers over a long period of time due to the influence of water and gravity (Hashimoto and Komatsu, 2021). After ^{137}Cs is deposited in a mineral soil layer, its transportation rate is not stable and is time-dependent. It is thought that the transport rate of ^{137}Cs is greater during the short period immediately following the deposition, and decreases with time (Yamaguchi et al., 2012). In addition to these abiotic factors, biotic factors such as disturbance by animals (i.e., bioturbation) also affect ^{137}Cs profiles in soil (Hashimoto and Komatsu, 2021). In the case of farmland, it was reported that most ^{137}Cs accumulated in the surface layer (0–5 cm) in non-tilled fields; however, ^{137}Cs profiles were homogenized throughout soil mixed by tilling (Yamaguchi et al., 2012; Shiozawa et al., 2011).

Wild boar (*Sus scrofa*) is considered “ecosystem engineers” because of their ability to profoundly change, directly or indirectly, the characteristics of their surrounding environment (Barrios-Garcia and Ballari, 2012; O’Bryan et al., 2022; Sütő et al., 2020). In particular, soil disturbance created by their rooting behavior strongly impacts their habitat (Barrios-Garcia and Ballari, 2012; Sütő et al., 2020; Keiter and Beasley, 2017) . Wild boar rooting affects both the surface organic layer and mineral soil to a depth of 5–15 cm or more, rooting up to 50–100 cm when searching for food (e.g., roots etc.) (Sütő et al., 2020; Hirata, 2015; Horčičková et al., 2019; Howe and Bratton, 1976; Wirthner et al., 2012). Rooting by wild boar has the potential to alter the soil chemistry (Wirthner et al., 2012; Gray et al., 2020), arthropod communities (Matas et al., 2021), and vegetation cover and composition of an ecosystem (Horčičková et al., 2019; Wirthner et al., 2012; Gray et al., 2020; Arrington et al., 1999), and thus wild boar likely play an important role in the redistribution of contaminants within soil profiles. Recent research has shown that some wildlife species, including wild boar, are increasing in number within evacuated areas, despite the radiological contamination after the FDNPP (Lyons et al., 2020) and the Chernobyl nuclear accidents (Deryabina et al., 2015). Alteration of contaminants in the soil profiles by wild boar rooting likely influences the movements of contaminants through erosion and their availability to plants and other biota, and such impacts may be particularly pronounced in areas contaminated from the FDNPP accident, given the increase in abundance of wild boar in these areas. Despite their global distribution and potential role in the movement of contaminants, there are currently no studies assessing the impact of wild boar rooting on the redistribution of contaminants within ecosystems.

Therefore, our objective in this study was to determine whether rooting by wild boar alters the distribution of ^{137}Cs contamination within the soil profile. We targeted areas within the difficult-to-return zone surrounding the FDNPP (Fig.1), as these areas have the highest densities of wild boar and greatest levels of environmental contamination of ^{137}Cs (Lyons et al., 2020). We hypothesized rooting by wild boar will alter the distribution of ^{137}Cs within the soil profile. In addition, given that areas disturbed through wild boar rooting remain altered for several months or even years after rooting, we hypothesized processes influencing abiotic and biotic transport of ^{137}Cs within the soil would be similar between recent and previously rooted areas. We tested these hypotheses through collection of soil cores from two different types of rooting samples, "new depressions" and "old depressions" as well as control sites, and examined whether there were different ^{137}Cs soil profile trends between the three sample types.

2. Material and methods

2.1. Study site

Our study site was located in Ottozawa-Chuoudai district, Okuma town (37.4161°N, 140.9798°E, Sampling station (St.). 1, Fig. 1) and Minami-Tsushima district, Namie town (37.5541°N, 140.7871°E, St. 2, Fig. 1), Fukushima prefecture, Japan. The elevation was approximately 80 m in the coastal zone (St. 1) and 500 m in the mountainous zone (St. 2) a.s.l., respectively. The distance from the FDNPP was approximately 5 km SW (St. 1) and 25 km NW (St. 2) and these sites were approximately 23 km from each other. These sites were located within the evacuation area (i.e., difficult-to-return zone), and therefore there has been no anthropogenic disturbance at either site since the accident. According to the Tomioka and Tsushima weather station of Japan Meteorological Agency, the average precipitation for St. 1 and St. 2 was 1520 mm year^{-1} in 1976–2021 and 1370 mm year^{-1} in 1977–2021 (Japan Meteorological Agency, 2023). Geologically, the soil type was designated as immature soil at St. 1, whereas volcanic ash soil and underlying granite rock dominate at St. 2 (National Agricultural and Food Research Organization, 2023). The map of Fukushima Prefecture was obtained from the Ministry of Land, Infrastructure, Transport and Tourism (MLIT) of Japan (Ministry of Land, Infrastructure, Transport and Tourism (MLIT) of Japan, 2023), and ^{137}Cs air dose rate (μSv/h) on October 29 in 2020 was obtained from the Japan Atomic Energy Agency (JAEA) (Japan Atomic Energy Agency (JAEA), 2023). This map was created using ArcGIS Pro 3.1.6 (ArcGIS Pro 3.1.6, 2023) (Fig. 1).

2.2. Sample collection

Soil sample collections were performed at a control site on 8th December 2020 (St. 1), an abandoned farmland on 12th February 2021 (St. 1), grassland on 15th October 2021 (St. 2), and an abandoned farmland on 21st November 2021 (St. 2). Across these locations, we sampled four grassland sites and three farmland sites, including one control site for each habitat type that included both new and old rooted areas (Table 1). We defined "new depressions" as those where the soil had been exposed by rooting of wild boar recently (i.e., the soil has been disturbed and where grass had not grown over the soil after rooting), and "old depressions" as rooting where grass had grown over the soil after rooting of wild boar (Fig. 2). We sampled two control sites, three new rooting sites, and two old rooting sites total between both stations (Table 1). At each sampling site we measured ambient dose equivalent rate at 1 m height around the rooting site (TCS-172B, Hitachi Aloka Medical Ltd., Japan), maximum depth of rooting, and maximum width of rooting (Table 1). To account for variability within each site due to heterogeneity in soil mixing from rooting, two or three soil cores were collected at a depth of 30 cm within each disturbed area (or the control sites) using a liner core sampler DIK-110C (DAIKI, Japan) with a plastic cylinder insert 5 cm in diameter. In cases where rock prevented sampling to the full depth of 30 cm, soil samples were collected as deep as possible, with all samples collected at least 19 cm from the surface (Table 2).

2.3. Laboratory analyses

In the laboratory, soil cores were sectioned off at 2 cm intervals from the surface to a depth of 30 cm. Each sectioned soil sample was dried at 50 °C for at least 24 h. The dried samples were homogenized in a mortar, and then passed through a 2 mm sieve. The sieved samples were weighed and packed into 100 mL containers and then measured to determine ^{137}Cs activity concentration (Bq/kg). The measurements were performed with standard electrode coaxial Ge detectors (CANBERRA GC4020, Canberra, USA) at the Institute of Environmental Radioactivity, Fukushima University. Measurements were taken up to 18,000 s to ensure statistical error was <5% for samples. Measurement data were

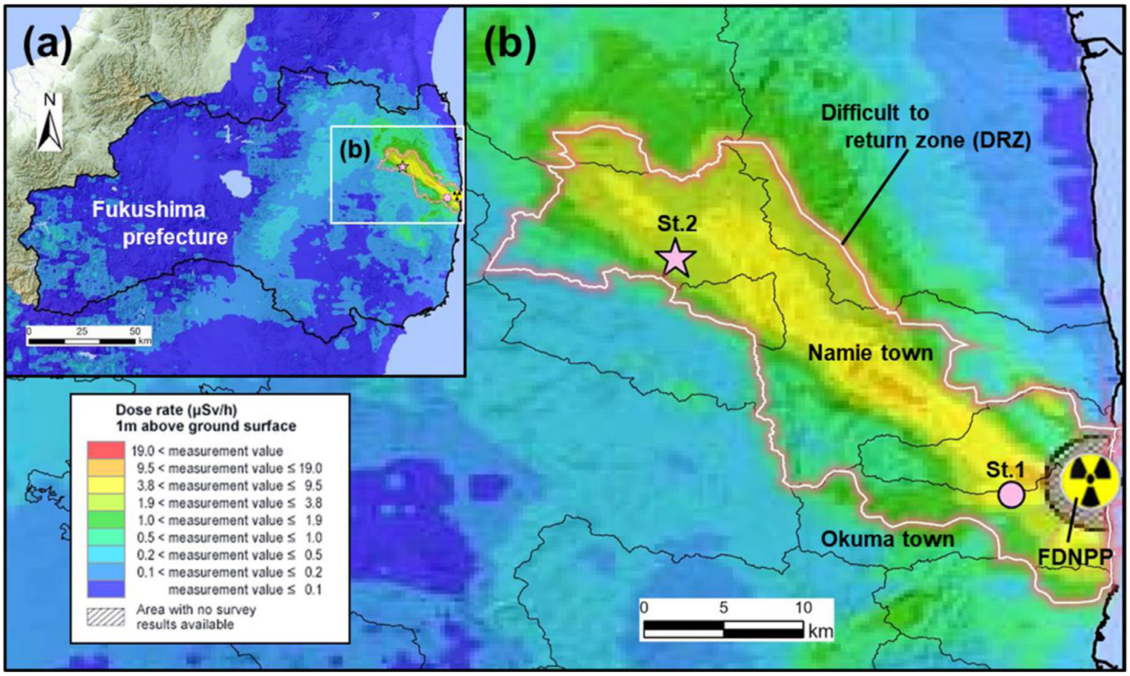

Fig. 1. Sampling sites relative to (**a**) Fukushima Prefecture, and (**b**) the difficult-to-return zone (DRZ) and surrounding area (including the Specified Reconstruction and Revitalization Base areas). Circle and star symbols on the map indicate the sampling sites, St. 1 and St. 2, respectively. Circular radiation symbol indicates the location of the Fukushima Daiichi Nuclear Power Plant (FDNPP).

Table 1
Site name, municipality, land and rooting types, ambient dose equivalent rate at 1 m height, maximum depth of rooting, and maximum width of rooting at sampling locations, Okuma town (St. 1) and Namie town (St. 2) in Fukushima (refer to Fig. 1).

Station	Site	Municipality	Land type	Rooting type	Ambient dose equivalent rate (μSv/h)	Maximum depth of rooting [m]	Maximum width of rooting [m]
St. 1	OF-1	Okuma	Farmland	New rooting	8.5	0.18	1.2
St. 1	OF-2	Okuma	Farmland	Control site	8.7	–	-
St. 2	NG-1	Namie	Grassland	New rooting	1.6	0.18	1.1
St. 2	NF-1	Namie	Farmland	New rooting	2.3	0.15	2.0
St. 2	NG-2	Namie	Grassland	Old rooting	1.6	0.18	1.0
St. 2	NF-2	Namie	Farmland	Old rooting	1.6	0.40	1.5
St. 2	NG-3	Namie	Grassland	Control site	1.9	–	–

Table 2
Sampling core depth, mean migration linear depth, and mean migration mass depth for each core and each site, Okuma town (St. 1) and Namie town (St. 2) in Fukushima (refer to Fig. 1 and Table 1).

Station	Site	Core number	Sampling core depth (cm)	Mean migration linear depth (cm)	Mean migration mass depth (g cm^{-2})	Mean migration linear depth (cm) in each station			The mean migration mass depth (g cm^{-2}) in each station		
						Average	±	SD	Average	±	SD
St. 1	OF-1	1	19	5.40	12.1	6.13	±	1.26	20.9	±	7.6
		2	22	7.58	25.1						
		3	30	5.40	25.5						
St. 1	OF-2	1	30	5.88	21.2	4.50	±	–	17.0	±	–
		2	27	3.12	12.8						
St. 2	NG-1	1	30	7.29	16.7	6.11	±	1.60	15.7	±	3.3
		2	29	4.29	12.1						
		3	30	6.74	18.4						
St. 2	NF-1	1	30	5.44	21.8	6.16	±	2.10	22.0	±	9.0
		2	30	8.53	31.1						
		3	29	4.51	13.2						
St. 2	NG-2	1	30	8.80	23.2	8.72	±	1.35	21.2	±	2.1
		2	29	10.03	21.5						
		3	30	7.34	19.0						
St. 2	NF-2	1	30	10.67	39.7	8.16	±	3.95	27.1	±	13.6
		2	29	3.61	12.7						
		3	29	10.20	28.9						
St. 2	NG-3	1	30	4.25	6.5	3.66	±	0.59	6.5	±	0.8
		2	30	3.65	7.3						
		3	30	3.07	5.8						

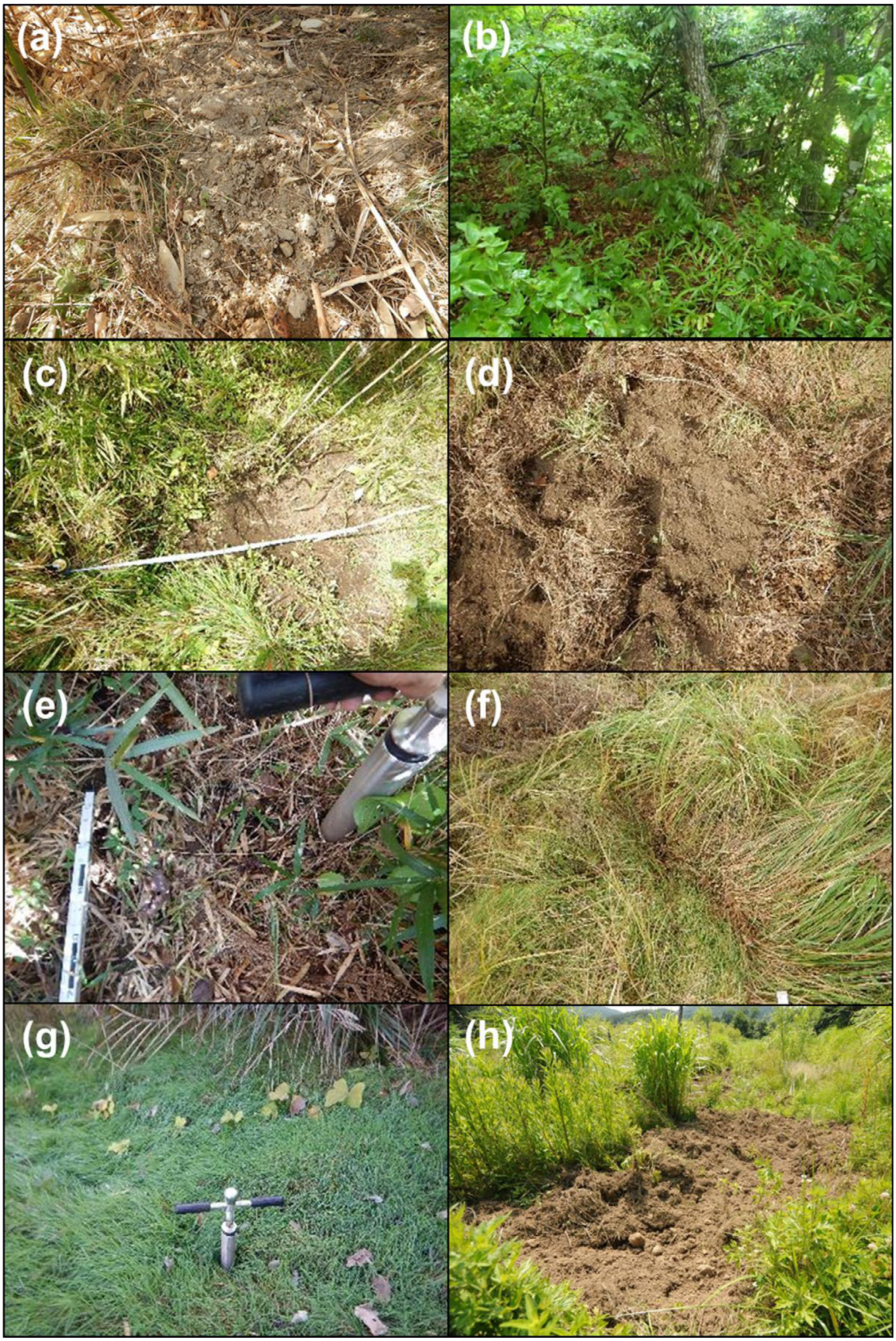

Fig. 2. Sampling sites representing: (a) new rooting in farmland [OF-1], (b) control site in farmland [OF-2] in Okuma town, (c) new rooting in grassland [NG-1], (d) new rooting in farmland [NF-1], (e) old rooting in grassland [NG-2], (f) old rooting in farmland [NF-2], (g) control site in grassland [NG-3] in Namie town, and (h) general rooting activity by wild boar (*Sus scrofa*). All photos were taken at the time of sampling except for photo (b) (taken June 2016).

Table 3
Soil deposition and coefficients result of the fitted Gaussian distribution for each site, Okuma town (St. 1) and Namie town (St. 2) in Fukushima (refer to Fig. 1 and Table 1).

Station	Site	Inventory (kBq/m^2)			Gaussian distribution		
		Average	±	SD	a (peak value)	b (critical point)	c (growth rate)
St. 1	OF-1	6620	±	3510	0.967	5.677	4.517
St. 1	OF-2	5930	–	–	0.847	3.661	1.977
St. 2	NG-1	1490	±	757	0.780	3.167	6.947
St. 2	NF-1	1801	±	472	0.999	2.853	5.389
St. 2	NG-2	2857	±	600	0.869	6.538	7.369
St. 2	NF-2	1039	±	577	1.725	−17.471	15.728
St. 2	NG-3	3122	±	3136	0.963	3.751	1.663

corrected for the decay of ^{137}Cs to the date of sampling based on the physical half-life of ^{137}Cs. Weight of each sectioned soil sample was divided by the sampling area of the core sampler to calculate mass depth (kg/m^2, dry). The ^{137}Cs inventory in each sectioned soil layer (Bq/m^2) was calculated by multiplying the ^{137}Cs activity concentration by the sectioned mass depth. The mass depth and ^{137}Cs inventory of each sectioned soil layer were integrated to obtain cumulative mass depth (kg/m^2) and cumulative ^{137}Cs inventory (Bq/m^2) for each sample.

2.4. Statistical analysis

The "^{137}Cs ratio" was calculated by dividing the activity concentration of ^{137}Cs at each depth by the maximum activity concentration of ^{137}Cs in each core sample. We used two or three sample replicates from each site and performed a nonlinear regression analysis by the least squares estimation using all replication data for the soil profile in each sampling site to confirm the profile property of ^{137}Cs at each depth. Regression analysis of ^{137}Cs ratio (y) assumed the scatter of data around the ideal curve followed a Gaussian distribution according to Eq. (1):

$$y = a\ exp\left\{-\frac{(x-b)^2}{2c^2}\right\} \tag{1}$$

where, x = depth, a = peak value, b = critical point, c = growth rate. The analyses were performed using the JMP 13.2.1 software package (SAS, Cary, NC, USA).

The "mean migration depth (median depth or migration center, Z) for ^{137}Cs in soil" was calculated refer to Ramzaev and Barkovsky (Ramzaev and Barkovsky, 2018). First, the relative radionuclide activity in each soil layer (q_i) was calculated according to Eq. (2):

$$q_i = A_i / A_{total} \tag{2}$$

where, A_i = inventory of ^{137}Cs in ith layer (kBq m^{-2}), A_{total} = total inventory of ^{137}Cs (kBq m^{-2}).

Next, the mean migration depth (Z) for ^{137}Cs in soil was calculated according to Eq. (3):

$$Z = \sum_{i=1}^{n} Z_i \times q_i \tag{3}$$

where, Z_i = the center of each layer in the soil profile, q_i = the relative ^{137}Cs activity.

We calculated the mean migration depth, Z, in terms of linear depth (cm) and mass depth (g cm^{-2}).

3. Results and discussion

Our results demonstrate that soil disturbance from wild boar rooting can substantially alter the distribution of ^{137}Cs through the soil profile, creating a more homogeneous distribution of ^{137}Cs within rooted areas that is maintained for at least several months after rooting occurs. However, there was considerable variability in soil ^{137}Cs inventory and activity among cores collected at the same site (Table 3, Fig. 3). For example, one core sample was an order of magnitude higher in ^{137}Cs than other core samples collected from site NG-3 (Fig. 3), potentially indicating the presence of a hotspot of ^{137}Cs contamination. The average soil ^{137}Cs inventory also varied considerably among sites, ranging from 1039 to 6620 kBq/m^2 (Table 3). The average and standard deviation (SD) of mean migration linear depth (cm) and the mean migration mass depth (g cm^{-2}) were 7.1 ± 2.3 cm, 21.4 ± 7.9 g cm^{-2} in rooting sites and 4.0 ± 1.2 cm, 10.7 ± 6.5 g cm^{-2} in control sites, respectively (Table 2). The results of the ^{137}Cs ratio based on fitting nonlinear regression models for each sampling site are shown in Fig. 4 and Table 3. In both control sites of Okuma and Namie towns [Fig. 4 (b) and (g)], the ^{137}Cs ratio was highest in the 2–4 cm soil profile and immediately decreased after 4 cm depth. After 8 cm depth, the ^{137}Cs ratio was approximately zero. These results from our control sites are consistent with previous research in Fukushima, as most radioactive ^{137}Cs in the soil surrounding the FDNPP accident remains within 5 cm of the soil surface (Terashima et al., 2014).

Sites disturbed by wild boar rooting displayed three patterns: (i) similar to the control sites, the ^{137}Cs ratio was highest in the surface soil layer [i.e., 0–4 cm profile, Fig. 4 (c) and (d)]; but the ^{137}Cs ratio was not a sharp peak and ^{137}Cs was distributed deeper into the soil layers compared with the control sites. (ii) the ^{137}Cs ratio in deeper soil layers was higher than in control sites [i.e., 6–10 cm, Fig. 4 (a) and (e)]; and iii) there was no clear trend in the peak of ^{137}Cs ratio in relation to soil depth across sites [Fig. 4 (F)]. In addition, while the ^{137}Cs ratio immediately decreased after reaching a peak at the control sites, the ^{137}Cs ratio at all rooted sites decreased slowly and ^{137}Cs was distributed deeper into the soil layers (i.e., 10 cm or deeper) (Fig. 4). Moreover, the ^{137}Cs soil profiles varied greatly among core samples and sampling sites (Figs. 3 and 4). Therefore, it seems that the extent to which the deeper layers are affected by contamination depends on the unique characteristics (i.e., extent and depth of rooting) of each wild boar rooting disturbance.

Within the soil profile, ^{137}Cs is absorbed in plant and tree tissues through root uptake, and both contaminant distribution and root system depth play an important role in determining contaminant absorption (e.g., Nimis 1994). It follows then, that by altering the vertical distribution of contaminants in the soil, wild boar rooting has the possibility to influence bioavailability (e.g., root uptake) of ^{137}Cs to plant species differently in impacted areas, depending on the extent of boar rooting.

Data collected after the Chernobyl accident indicate that most ^{137}Cs in the undisturbed soil remained in the upper 10 cm, especially the upper 5 cm, 6 to 8 years after the accident (Rosen et al., 1999). In addition, more than 50 % of the ^{137}Cs contamination originating from the Chernobyl nuclear accident still remained in the upper 10 cm soil layer in German forest ecosystems 19 years later (Konopleva et al., 2009). Since wild boar rooting affects both the surface organic layer and mineral soil (e.g., Figs. 3 and 4), our data suggest the alteration of ^{137}Cs soil profiles by rooting activity of wild boar can be more influential than the long-term movement of ^{137}Cs soils in the ecosystem.

One biological factor affecting the ^{137}Cs soil profile is the transport of solid ^{137}Cs through earthworm feeding and egestion (Jarvis et al., 2010). Jarvis et al. (2010) developed and verified a model that took into account the effects of ingestion, transport, and excretion of ^{137}Cs

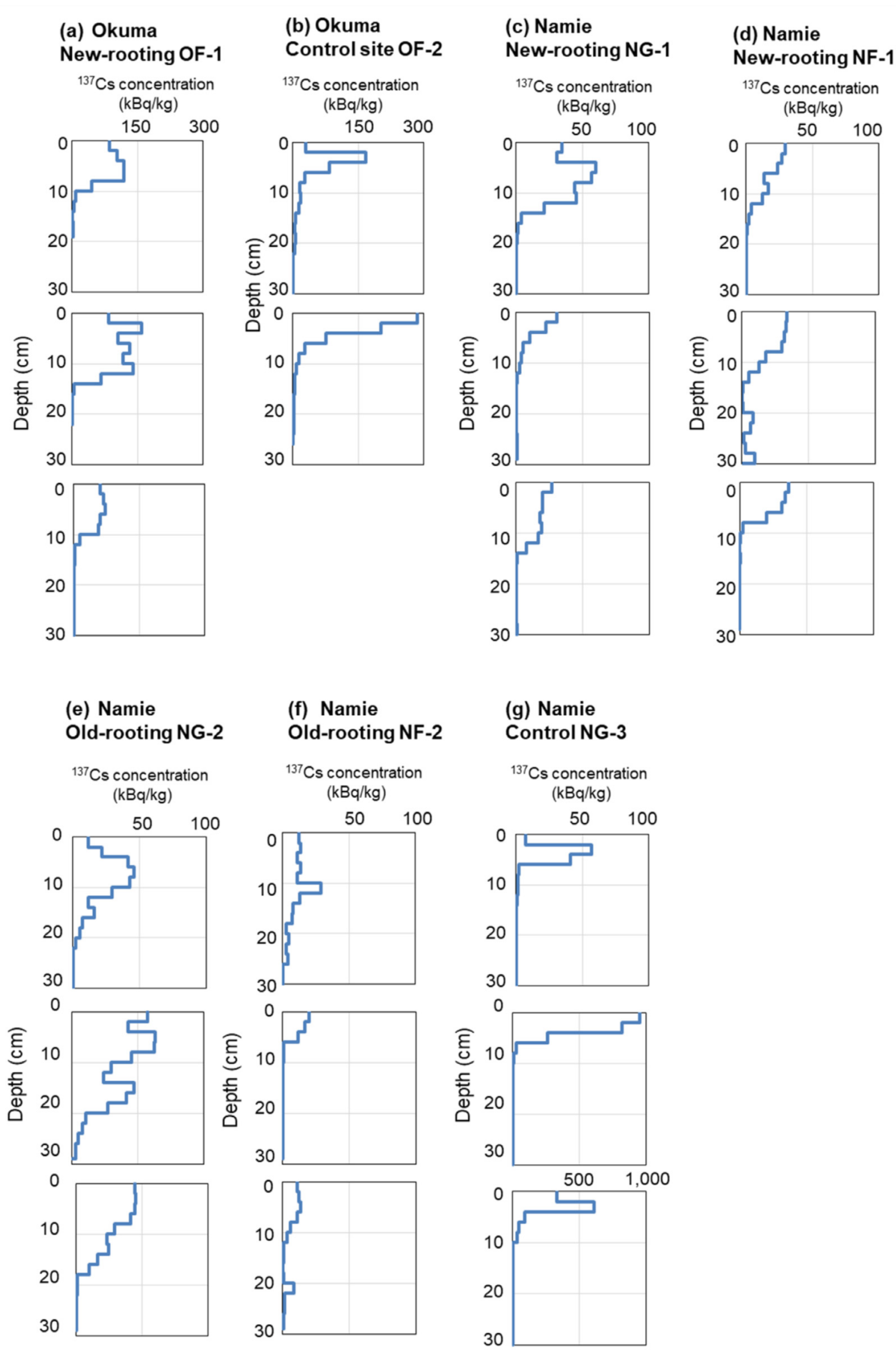

Fig. 3. Vertical distribution of ^{137}Cs activity concentration in each sampling site (i.e., soil profiles by wild boar rooting and control), Okuma town (St. 1) and Namie town (St. 2) in Fukushima (refer to Fig. 1 and Table 1). (a) New rooting in farmland [OF-1], and (b) Control site in farmland [OF-2] in Okuma town, (c) New rooting in grassland [NG-1], (d) New rooting in farmland [NF-1], (e) Old rooting in grassland [NG-2], (f) Old rooting in farmland [NF-2], and (g) Control site in grassland [NG-3] in Namie town.

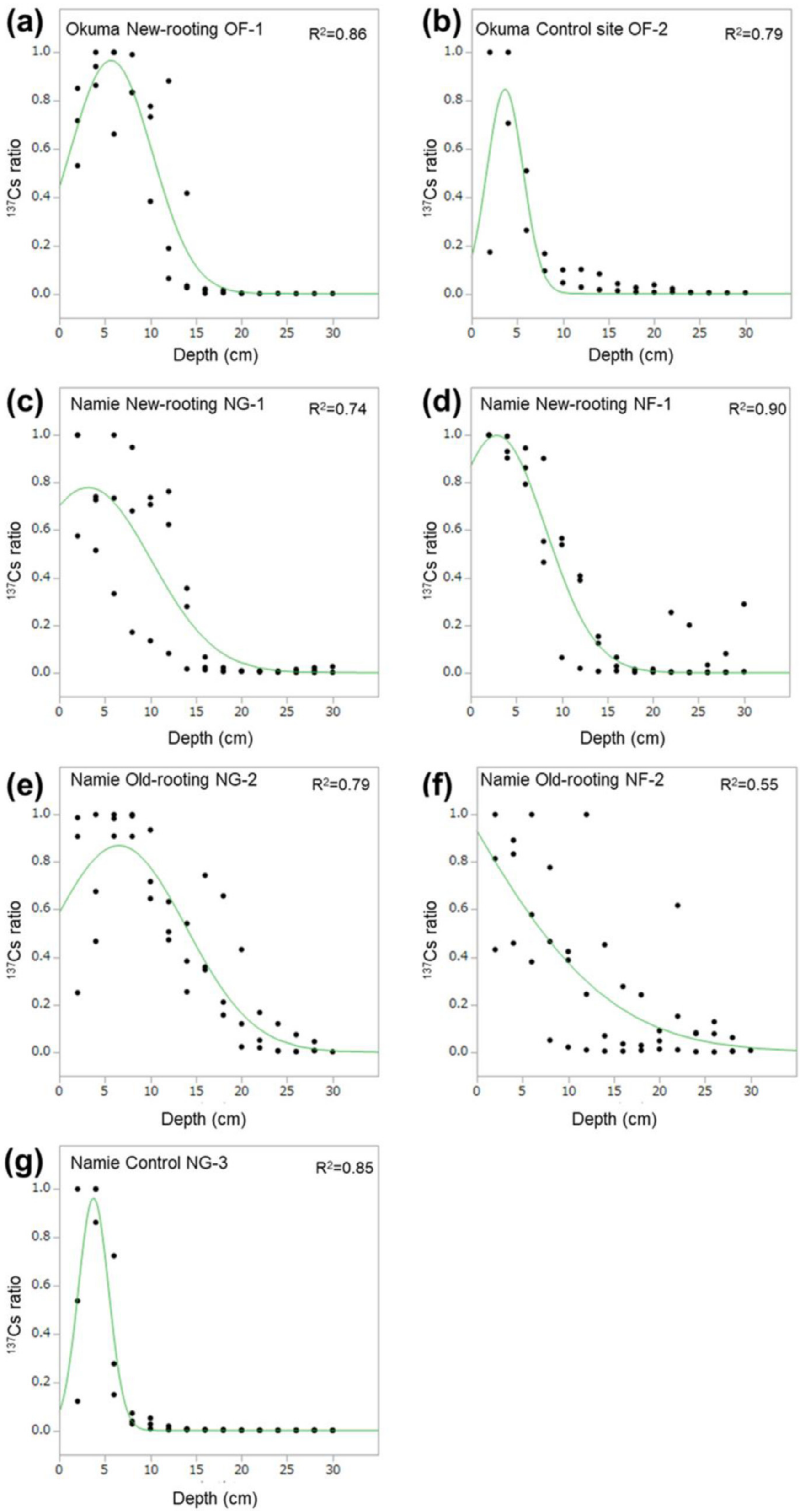

Fig. 4. Relationship between soil depth and ^{137}Cs ratio for each site (i.e., soil profiles by wild boar rooting and control). Curved lines indicate the correlation of the fitted Gaussian distribution for site (see Table 3). (a) New rooting in farmland [OF-1], and (b) Control site in farmland [OF-2] in Okuma town, (c) New rooting in grassland [NG-1], (d) New rooting in farmland [NF-1], (e) Old rooting in grassland [NG-2], (f) Old rooting in farmland [NF-2], and (g) Control site in grassland [NG-3] in Namie town.

by earthworms and concluded that transfer of ^{137}Cs in soil is strongly facilitated not only by physical advective-dispersive transport, but by earthworm activity, as well. So far, the influence of the biological transport and disturbance of ^{137}Cs soil profiles has focused on the activity of soil organisms (Hashimoto and Komatsu, 2021; Jarvis et al., 2010). However, in addition to the influence from the activity of soil organisms, the disturbance of ^{137}Cs soil profiles by wild boar rooting activities is also an important factor contributing to ^{137}Cs soil profiles. In particular, increased rooting activity associated with large wild boar populations, such as those found within the evacuated areas (Lyons et al., 2020), may lead to large, disturbed areas and have great impacts on soil disturbance and thus the mixing ^{137}Cs within the soil profile. In particular, the high ^{137}Cs contamination level in the evacuation zone and the presence of ^{137}Cs "hot spots" identified in this study (i.e., site NG-3), make the transport of contaminants through wild boar rooting more concerning as it has the potential to distribute contaminants more broadly and deeply within the soil environment.

There was no difference in the soil profiles between new and old rooting sites (Fig. 4). However, it is possible that once rooting has occurred, the ^{137}Cs profile of the soil may change over time due to abiotic and biotic transport and disturbance of ^{137}Cs by soil organisms, such as earthworms. Therefore, further studies are needed to examine the effects of long-term rooting on the alteration of ^{137}Cs in soil profiles. In addition, it is important to confirm whether the physicochemical fraction of ^{137}Cs in soil is changed after disturbance by rooting of wild boar in the future. Cesium-137 exists as different physicochemical fractions in the environment and physicochemical fraction is strongly related to the uptake and transfer of ^{137}Cs in plants and animals (Saito et al., 2020; Saito and Tsukada, 2022). Therefore, the physicochemical fractions of ^{137}Cs in soil are important factors in evaluating the bioavailability of ^{137}Cs to wildlife (Saito et al., 2020; Saito and Tsukada, 2022). Several studies have examined the influence of wild boar rooting on soil properties, but whether wild boar rooting had an effect varied depending on the soil parameters (pH, ions, etc.) being investigated (Barrios-Garcia and Ballari, 2012). Therefore, it is important to consider whether the physicochemical fraction of ^{137}Cs, and not only the soil profile, is altered by the rooting of wild boar, and further research is needed to better understand the dynamics of ^{137}Cs bioavailability to wildlife and ecosystems.

4. Conclusion

This study is the first of its kind to reveal that wild boar rooting disrupts the distribution of ^{137}Cs contaminants in soil within the difficult-to-return zone in Fukushima after the FDNPP accident. Most ^{137}Cs in the soil surrounding the FDNPP accident remained within 5 cm of the soil surface, however ^{137}Cs in the soil at wild boar-rooted sites reached deeper into the soil when compared to control sites. In addition, while the ^{137}Cs ratio immediately decreased after reaching a peak at the control sites, the ^{137}Cs ratio at all rooted sites decreased slowly. In addition, it seems that the extent to which the deeper layers are affected by contamination depends on the unique characteristics (i.e., extent and depth of rooting) of each wild boar rooting disturbance. There was no difference in the soil profiles between new and old rooting sites. However, it is possible that once rooting has occurred, the ^{137}Cs profile of the soil may change over time due to abiotic and biotic transport and disturbance of ^{137}Cs by soil organisms. Therefore, further studies are needed to examine the effects of long-term rooting on the alteration of ^{137}Cs in soil profiles. This research contributed to a clearer understanding of the terrestrial environmental impact of wild boar rooting which can better inform appropriate management actions given the growing population of wild boar within the difficult-to-return zone.

Declaration of Competing Interest

No conflict of Interest of this paper.

Data availability

Data will be made available on request.

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