Radial and vertical distributions of radiocesium in tree stems of *Pinus densiflora* and *Quercus serrata* 1.5 y after the Fukushima nuclear disaster

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Highlights

- Radiocesium concentrations in tree stems of pine and oak were determined.
- Vertical distributions of radiocesium were different between the species.
- Radial distributions of radiocesium in wood were similar in both species.
- Radiocesium distributions among stem parts differed between the species.
- Transportation and allocation of radiocesium would differ between the species.

Abstract

The radial and vertical distributions of radiocesium in tree stems were investigated to understand radiocesium transfer to trees at an early stage of massive contamination from the Fukushima nuclear disaster. A conifer species (Japanese red pine) and a broad-leaved species (Japanese *konara* oak) were selected to determine whether the radiocesium contamination pattern differs between species. Stem disks were collected at several heights and separated into outer bark, inner bark, and wood. The radiocesium concentration was the highest in the outer bark, followed by that in the inner bark and wood. The vertical distribution of the radiocesium concentration at each stem part differed between the species. The difference between species in radiocesium concentration of the outer bark could be explained by presence or absence of leaves at the time of the disaster. However, the reasons for the differences between species in the radiocesium concentration of the inner bark and wood are unclear. The radial distribution in the wood of the studied species showed a common pattern across stem disk heights and species. However, the radiocesium concentration ratio between sapwood and inner bark was significantly different between species. Although the radial contamination pattern in the wood was similar in the studied species during the early stage of contamination, the radiocesium transport pathway and allocation would be different between the species, and the contamination pattern will likely be different between the species at later stages. Continued investigations are important for understanding the radiocesium cycle and the accumulation of radiocesium in the tree stems of each species.

Keywords

Fukushima, radiocesium, bark, wood, radial distribution, vertical distribution

1. Introduction

A considerable amount of radiocesium (^{134}Cs and ^{137}Cs) was emitted into the atmosphere by the Fukushima Dai-ichi nuclear disaster in March 2011. The fallout was largely deposited in forests because they cover much of the land (71% of Fukushima Prefecture). Radiocesium would be well mixed with stable Cs within the biological cycle in forest ecosystems (Yoshida et al., 2004), and ^{137}Cs, which has a long physical half-life (30.2 y), will remain in forest ecosystems for many decades. Therefore, understanding radiocesium dynamics is critical to forest management in contaminated areas. In particular, ^{137}Cs accumulation in trees is one of the most important concerns for timber use and forest decontamination.

Radiocesium may enter a tree via root uptake, translocation from the foliar surface, or even from the bark surface (Ertel and Ziegler, 1991; Tagami et al., 2012). In a tree stem, radiocesium is mobile and passes through tree rings (Kohno et al., 1988; Kudo et al., 1993; Momoshima and Bondietti, 1994), resulting in whole-stem contamination. Some radiocesium may form ionic bonds with carboxylic groups in cell walls, the cytoplasm of living cells, and the cell debris of the heartwood, the inner part of wood (Brown, 1964). Because heartwood is composed of dead cells and hence does not function in water transport, radiocesium that transferred to heartwood is likely to remain there for a long time. Parallel distribution of ^{137}Cs to that of ^{40}K in a Japanese cedar (*Cryptomeria japonica*) stem (Kudo et al., 1993) and to those of alkaline metals in Scots pine (*Pinus sylvestris*) stems (Yoshida et al., 2011) suggests that radiocesium transferred to heartwood stayed there and reached to an equilibrium distribution to the elements with similar chemical properties.

Previous studies have indicated that the radiocesium distribution among sapwood (the outer part of wood, which transports water and has living cells) and heartwood differs between species. For example, Japanese cedar and cypress (*Chamaecyparis obtusa*) reportedly have higher ^{137}Cs concentration in the heartwood than in the sapwood (Kohno et al., 1988), whereas Scots pine is reported to have a higher ^{137}Cs concentration in the sapwood than in the heartwood (Thiry et al., 2002; Yoshida et al., 2011). Soukhova et al. (2003) reported different ^{137}Cs distributions in Scots pine and silver birch (*Betula pendula*) and attributed the difference to the different radial ray compositions of those species.

Although ^{137}Cs accumulation in tree stems is understood to a certain extent, further research is needed to improve our understanding and ability to predict ^{137}Cs accumulation in wood. Research into the early stages of contamination is particularly lacking. Moreover, for proper forest management in Fukushima, native species growing in the local environment must be studied. Kuroda et al. (2013) reported that ^{134}Cs and ^{137}Cs were detected in the heartwood of three species (*Pinus densiflora*, *Quercus serrata*, and *Cryptomeria japonica*) collected from Fukushima forests half a year after the Fukushima Dai-ichi nuclear accident. This fact indicates that there is rapid inflow of radiocesium to tree stems and rapid translocation to heartwood, highlighting the importance of research into the early stages of contamination.

In the present study, we investigated the radial and vertical distributions of radiocesium in tree stems of two dominant species, Japanese red pine (*P. densiflora* Sieb. & Zucc.) and Japanese *konara* oak (*Q. serrata* Thunb.), 1.5 y after the Fukushima Dai-ichi nuclear disaster, focusing on whether the radiocesium transfer pattern differs between species.

2. Material and methods

2.1. Study sites and sampling

A Japanese red pine forest (pine forest) and a deciduous broad-leaved forest (oak forest) in Kawauchi Village, about 20 km southwest of the Fukushima Dai-ichi Nuclear Power Plant, were selected for the study (Fig. 1). Pine and oak forests were representative forest types in the village. Samples were collected from trees that were adjacent to a 40 m × 40-m census plot in the pine forest (520 m above sea level) and a 50 m × 30-m census plot in the oak forest (530 m above sea level). In the pine plot, tree density (diameter at breast height > 5 cm) was 1,513 ha^{-1} and Japanese red pine (*P. densiflora*) accounted for 73% of the trees. The forest canopy was completely dominated by the red pines, but was not fully closed; the remaining 27% of the trees in the plot vegetated in the understory (e.g., *Toxicodendron trichocarpum* 5%, *Q. serrata* 4%, and *Swida controversa* 4%). In the oak plot, the tree density was 1,413 ha^{-1} and Japanese *konara* oak (*Q. serrata*) accounted for 38% of the trees, followed by Japanese clethra (*Clethra barbinervis*; 12%), Japanese wild cherry (*Cerasus jamasakura*; 6%), sawtooth oak (*Q. acutissima*; 5%), and Japanese *mizunara* oak (*Q. crispula*; 5%). The forest canopy was dominated by *Q. serrata*, *Q. acutissima*, and *Q. crispula*, but was not closed, forming a multistory vertical structure. The air dose rate was 0.2 μSv h^{-1} at the pine forest and 1.8 μSv h^{-1} at the oak forest, as measured in late July 2012 at 1 m above the ground using an ionization chamber-type survey meter (ICS-331B; Hitachi Aloka Medical Ltd., Tokyo, Japan). The ^{137}Cs contamination in soil (the sum of the contamination found in the litter layer, fermentation layer, humus layer, and mineral soil) was 1.1×10^5 Bq m^{-2} (a standard deviation (σ) = 3.0×10^4 Bq m^{-2}) at the pine plot and 1.5×10^5 Bq m^{-2} ($\sigma = 4.9 \times 10^4$ Bq m^{-2}) at the oak plot in September 2012. These values were the means of three sampling points collected diagonally at each plot and were used to calculate the aggregated transfer factor (T_{ag}; m^2 kg^{-1}) from soil to tree. The sampling points were at least 2 m away from tree stems in order to avoid thick roots. The litter layer, fermentation layer and humus layer were collected from a 50 cm × 50-cm area at each point. The mineral soil (brown forest soil) to a depth of 20 cm was collected using cylindrical soil samplers.

Three pines (*P. densiflora*) and three oaks (*Q. serrata*) that were adjacent to the respective census plots were logged in early September 2012. The diameter at breast height (DBH), tree height, and age of each tree are shown in Table 1. The trees were selected from individuals of different diameter classes growing in the dominant tree layer in order to represent the diameter distribution of each species. The diameters of both species in their respective census plots showed normal distributions: mean DBH of *P. densiflora* in the pine plot was 24.5 cm (σ = 5.8 cm, n = 175) and that of *Q. serrata* in the oak plot was 19.8 cm (σ = 5.8 cm, n = 82). All pines and oaks selected were considered to be mature as the youngest was 36 y of age. Disk samples that were approximately 5 cm thick were removed from each logged stem at 0.3, 1.3, 5, and 10 m above ground. Additional disk samples were removed from pines at 15 m above ground, from a short oak tree at 7.5 m, and from tall oaks at 12.5 m.

2.2. Sample preparation and analysis

The disks were separated into three parts: outer bark (cork), inner bark (phloem), and wood (xylem). The outer bark was removed from the disks using a chisel, after which the inner bark, including the cambium, was removed. The disks collected at the following stem heights were used for xylem analysis: pine1 (1.3 and 15 m), pine2 (1.3, 5, 10, and 15 m), pine3 (1.3 and 15 m), oak1 (1.3 and 7.5 m), oak2 (1.3, 5, and 10 m), and oak3 (1.3 and 10 m). Each disk was further separated along tree-ring boundaries into sub-samples of several rings each. Each sub-sample weighed ca. 50 g (dry mass at 80°C). This sample separation resulted in sufficient material for γ-ray spectrometry and provided enough resolution for investigation of the radial migration of radiocesium in the tree stems. Distances from the pith to each separated ring boundary were measured along four radii and their average was used as the distance between pith and ring boundary. The distance of the sapwood–heartwood boundary from the pith was measured in the same way.

All samples, except mineral soil samples, were ground using a Wiley mill before packing into plastic containers. Mineral soil samples were packed after drying for at least 7 d, and sieving with 2-mm mesh. About 1 g of each sample was dried at 80°C for 48 h to calculate the dry mass.

The radioactivity of ^{134}Cs (605 keV) and ^{137}Cs (662 keV) in tree and soil samples were measured using a high-purity Ge semiconductor detector (Tennelec, Tennessee, USA) at the Radioisotope Research Center of Kyoto University. The γ-ray detection efficiency was calibrated with the standard, which was prepared by using a reference standard QCY.44 supplied by Radiochemical Center Ltd., Amersham (Veronica et al., 1992), and provided by the Radioisotope Research Center, Kyoto University. The measuring time was 7,200–50,000 s, depending on the radioactivity of each sample. The detection limit of each radionuclide was calculated using Cooper's equation (Cooper, 1970; eq. 8, A_m = 3). The radioactivity was decay-corrected to September 1, 2012. All radiocesium concentrations in the present study are shown in dry mass (at 80°C) base.

3. Results and Discussion

3.1. Radiocesium distribution among stem parts

In both species, radiocesium concentration (Bq kg^{-1}) was the highest in the outer bark, followed by that in the inner bark and whole wood (Table 2). The ratio of ^{134}Cs to ^{137}Cs was about 0.6 in most samples. The burden of radiocesium (Bq) in each disk was also the largest in the outer bark, followed by that in the whole wood and inner bark. The burdens followed the same order at all analyzed heights. The concentration was reportedly the highest in the inner bark or cambium about 10 y after the Chernobyl accident (Thiry et al., 2002; Yoshida et al., 2011). Thiry et al. (2002) showed that 7% of the total ^{137}Cs in the stem was distributed in the outer bark, 18% in the inner bark, and 75% in the whole wood of 58-year-old Scots pine trees 12 y after the Chernobyl accident. In the present study, 74% was distributed in the outer bark, 6% in the inner bark, and 20% in the whole wood of Japanese red pine trees (collected at 1.3 m above the ground). Thus, 1.5 y after the disaster, the remaining surface contamination was still serious and further transfer to the interior of the stem might occur.

3.2. Vertical distribution of radiocesium in each stem part

The vertical distribution of the radiocesium concentration in the outer bark was different between the species (Fig. 2). In oaks, the radiocesium concentration in the outer bark was higher in the upper stem than in the lower stem and had a significant correlation with sampling height (n = 14, p < 0.001). In pines, however, the radiocesium concentration in the outer bark did not vary significantly with sampling height. The remarkably high deposition on the upper part of the outer bark of oaks might have occurred because the leafless canopy at the time of the initial massive deposition promoted direct capture of radiocesium by the bark surface.

The vertical distribution of radiocesium concentration in the inner bark was almost uniform in both species. However, the distribution pattern differed among individual oaks, whereas the distribution pattern was similar among individual pines: although oak1 and oak2 did not show a significant correlation between radiocesium concentration and sampling height, oak3 did (n = 5, p < 0.01). This difference between species in individual variation may be due to branching. The pines had living branches only at the top part of the main stem, whereas the oaks had living branches at multiple heights. Accordingly, in the oaks, radiocesium in the foliage would be supplied to the main stem at multiple heights via the inner bark (phloem), and that might result in individual differences in radiocesium vertical distribution patterns.

In the wood of both species, there was a significant correlation between radiocesium concentration and sampling height (Fig. 3). The radiocesium concentration in the whole wood of the pines was slightly higher in the upper stem than in the lower stem (regression coefficient = 1.1). On the other hand, that in the oaks was significantly higher in the upper stem than in the lower stem (regression coefficient = 7.4). The slight increase of radiocesium concentration toward the upper stem of the pines can be explained well by the vertical variation in heartwood ratio. The upper stem contained less heartwood than the lower stem, and the sapwood contained more radiocesium than the heartwood. In addition, the radiocesium concentration in the sapwood of the pines was almost constant across sampling heights. Therefore, the radiocesium contamination to the stem wood of the pines likely progresses uniformly regardless of stem height. This agrees with the report by Thiry et al. (2002), ^{137}Cs concentrations in the stem wood of Scots pines 12 y after the Chernobyl accident were largely unaffected by stem height. On the other hand, the vertical distribution of radiocesium in the oaks cannot be explained by the vertical variation in heartwood ratio alone because in this species, the sapwood in the upper stem had a significantly higher radiocesium concentration than that in the lower stem. One possible reason for the relationship between radiocesium concentration and height in oaks is direct radiocesium transfer from the outer bark. Several studies have suggested the possibility of radiocesium absorption by bark (Ertel and Ziegler, 1991; Tagami et al., 2012). The remarkably high radiocesium concentrations in the outer bark and whole wood in the upper part of oaks and their significant correlation ($n = 7$, $p < 0.01$) imply the possibility of bark absorption; however, this must be demonstrated in a future study. Although the reason remains unclear, the vertical contamination pattern differs between the species examined.

3.3. Radial distribution of radiocesium in wood

The radial distribution of the radiocesium concentration had a similar pattern among the analyzed heights and species (Figs. 4 and 5). The concentration was (1) the highest at the outermost part; (2) almost uniform throughout the sapwood, except for the outermost part; and (3) reduced toward the center in the heartwood.

One reason for the relatively high concentration in the outermost part of the stems is that the contamination occurred recently. In addition, radiocesium may be preferentially translocated to the young growing part of stem wood, as suggested by the analysis of Scots pine 12 y after the Chernobyl accident (Yoshida et al., 2011).

The uniform radiocesium concentration observed throughout the sapwood would be due to both diffusion and active transport. Thiry et al. (2002) indicated that the distribution pattern of ^{137}Cs in sapwood is in good agreement with the distribution of free water in wood, which increases from the inner sapwood to the outer sapwood. In addition, active radial transport through rays must be taken into account. The radial solute exchange between xylem and phloem occurs via the rays (van Bel, 1990). In Japanese cedar, alkali metals are transported actively from the sapwood to the outer heartwood via rays (Okada et al., 2012). Moreover, Soukhova et al. (2003) explained that the radial ^{137}Cs distribution in pine differs from that in birch because of the different ray composition between these species. The proportion of tracheid and parenchyma cells in the rays affects radial transport characteristics. In the present study, the radiocesium concentration ratio of sapwood to inner bark was significantly higher in oaks than in pines (Welch's t-test; $p < 0.001$), at 0.23 ($\sigma = 0.041$, $n = 7$) in oaks and 0.088 ($\sigma = 0.026$, $n = 7$) in pines. This difference seems to suggest that oaks transport more radiocesium from phloem to sapwood via rays than do pines. The active radial transport through rays is an important point that must be considered in a future study.

In the heartwood, the movement of radiocesium toward the center must be caused by diffusion alone because there are no living cells in heartwood. The decrease in the radiocesium concentration in proportion to the distance from the sapwood–heartwood boundary in both species confirms movement by diffusion. However, species-specific radial distribution patterns of ^{137}Cs concentration might appear after several years. Uniform distribution patterns have also been observed in Scots pines (Thiry et al., 2002; Soukhova et al., 2003; Yoshida et al., 2011), whereas increasing radiocesium concentration toward the center of the heartwood has been reported for Japanese cedar, cypress (Kohno et al., 1988), and birch (Soukhova et al., 2003). These different patterns of ^{137}Cs accumulation would result from different radial distributions of water, different heartwood compositions, and different processes of heartwood formation between species or individuals. These factors must be observed carefully and reviewed periodically after radioactive fallout in order to understand the mechanism of radiocesium accumulation in heartwood.

3.4. Radiocesium transfer to pine and oak

Radiocesium transfer from the outside to the inside of the tree may occur via three routes: the roots, foliage, and bark surface (Ertel and Ziegler, 1991; Tagami et al., 2012). The contribution of bark absorption seems to be low in the pines of the present study because there was no correlation between the radiocesium concentration of the outer bark and the inner parts along the stem. On the other hand, the bark absorption might have occurred in the oaks as there was a significant correlation between the radiocesium concentration in the outer bark and that in the inner parts of these trees.

Absorption from the foliar surface, in the case of the Fukushima disaster, might have occurred in evergreens (the pines), but not in deciduous species (the oaks). This is because the disaster occurred during the leafless period for deciduous species. Tagami et al. (2012) reported that woody plants with old leaves at the time of the accident showed higher ^{137}Cs concentrations in newly emerged leaves than did plants without old leaves; however, plants with waxy leaf surfaces had lower concentrations than plants with old leaves without a waxy surface. This suggests that, because pines have waxy leaf surfaces, the contribution of foliar absorption to radiocesium concentration in the pines of the present study was low. The vertically uniform distribution of radiocesium concentration in the inner bark and sapwood of the pines also implies that contamination from the upper part of the tree via foliar absorption was not significant. However, Thiry et al. (2002) estimated that a significant portion of radiocesium incorporation in stem wood was likely due to foliar absorption in the old Scots pines affected by the Chernobyl accident. Furthermore, Tagami and Uchida (2010) reported that trees do not take up large amounts of stable Cs from the soil. Therefore, it is difficult to determine the contribution of foliar absorption to radiocesium concentration in the stems of pines from our results alone.

To understand the respective contributions of root, foliar, and bark absorption, periodic monitoring of T_{ag} (aggregated transfer factor) is necessary. The T_{ag} of inner bark and wood (Table 2) were on the same order (10^{-4}) as those reported for the Chernobyl accident (Calmon et al., 2009). However, the present study was conducted during the early stages of radiocesium dynamics and T_{ag} values are changeable. If the T_{ag} values increase with time, the contribution of root absorption can be estimated. On the other hand, if the T_{ag} values do not change significantly, it can be concluded that the dominant route of radiocesium contamination in these trees was foliar or bark absorption.

4. Conclusion

In the early stages of contamination, there was a common pattern in the radial distribution of radiocesium in whole wood in Japanese red pines and Japanese *konara* oaks at different heights. The radiocesium concentration ratio of sapwood to inner bark was significantly different between the species, indicating differential radiocesium allocation and radial transport via rays between the species. The outer bark of oaks had significantly higher radiocesium concentration in the upper stem than in the lower stem, which is likely due to their leafless canopy at the time of the disaster. The radiocesium concentration in the sapwood was vertically constant in the pines, but it was higher in the upper stem than in the lower stem in the oaks. Although the reason is unclear, the vertical contamination pattern in the wood differs between these two species. Further periodic investigations are necessary to reveal the species-specific patterns and mechanisms of radiocesium accumulation in tree stems.

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References

Brown, G.N., 1964. Cesium in Liriodendron and other woody species: Organic bonding sites. Science 143, 368–369.

Calmon, P., Thiry, Y., Zibold, G., Rantavaara, A., Fesenko, S., 2009. Transfer parameter values in temperate forest ecosystems: a review. J. Environ. Radioact. 100, 757–766.

Cooper, J.A., 1970. Factors determining the ultimate detection sensitivity of Ge(Li) gamma-ray spectrometers. Nucl. Instrum. Methods 82, 273–277.

Ertel, J., Ziegler, H., 1991. Cs-134/137 contamination and root uptake of different forest trees before and after the Chernobyl accident. Radiat. Environ. Bioph. 30, 147–157.

Kohno, M., Koizumi, Y., Okumura, K., Mito, I., 1988. Distribution of environmental Cesium-137 in tree rings. J. Environ. Radioact. 8, 15–19.

Kudo, A., Suzuki, T., Santry, D.C., Mahara, Y., Miyahara, S., Garrec, J.P., 1993. Effectiveness of tree rings for recording Pu history at Nagasaki, Japan. J. Environ. Radioact. 21, 55–63.

Kuroda, K., Kagawa, A., Tonosaki, M., 2013. Radiocesium concentrations in the bark, sapwood and heartwood of three tree species collected at Fukushima forests half a year after the Fukushima Dai-ichi nuclear accident. J. Environ. Radioact. 122, 37–42.

Ministry of Education, Culture, Sports, Science and Technology (MEXT), 2012. MEXT, Japan. http://radioactivity.nsr.go.jp/ja/contents/7000/6289/24/203_0928.pdf (accessed on July 6, 2013).

Momoshima, N., Bondietti E.A., 1994. The radial distribution of ^{90}Sr and ^{137}Cs in trees. J. Environ. Radioact. 22, 93–109.

Okada, N., Hirakawa, Y., Katayama, Y., 2012. Radial movement of sapwood-injected rubidium into heartwood of Japanese cedar (*Cryptomeria japonica*) in the growing period. J. Wood Sci. 58, 1–8.

Soukhova, N.V., Fesenko, S.V., Klein, D., Spiridonov, S.I., Sanzharova, N.I., Badot, P.M., 2003. ^{137}Cs distribution among annual rings of different tree species contaminated after the Chernobyl accident. J. Environ. Radioact. 65, 19–28.

Tagami, K., Uchida, S., 2010. Can elemental composition data of crop leaves be used to estimate radionuclide transfer to tree leaves? Radiat. Environ. Biophys. 49, 583–590.

Tagami, K., Uchida, S., Ishii, N., Kagiya, S., 2012. Translocation of radiocesium from stems and leaves of plants and the effect on radiocesium concentrations in newly emerged plant tissues. J. Environ. Radioact. 111, 65–69.

Thiry, Y., Goor, F., Riesen, T., 2002. The true distribution and accumulation of radiocaesium in stem of Scots pine (*Pinus sylvestris* L.). J. Environ. Radioact. 58, 243–259.

van Bel, A. J. E., 1990. Xylem-phloem exchange via the rays: the undervalued route of transport. J. Exp. Bot. 41, 631–644.

Veronica, T., Isozumi, Y., Aoki, T., 1992. Determination of photopeak efficiencies of voluminal samples for the measurement of environmental radioactivities. Bull. Inst. Chem. Res. Kyoto Univ. 70, 399–407.

Yoshida, S., Muramatsu, Y., Dvornik, A. M., Zhuchenko, T. A., Linkov, I., 2004. Equilibrium of radiocesium with stable cesium within the biological cycle of contaminated forest ecosystems. J. Environ. Radioact. 75, 301–313.

Yoshida, S., Watanabe, M., Suzuki, A., 2011. Distribution of radiocesium and stable elements within a pine tree. Radiat. Prot. Dosim. 146, 326–329.

Figure 1

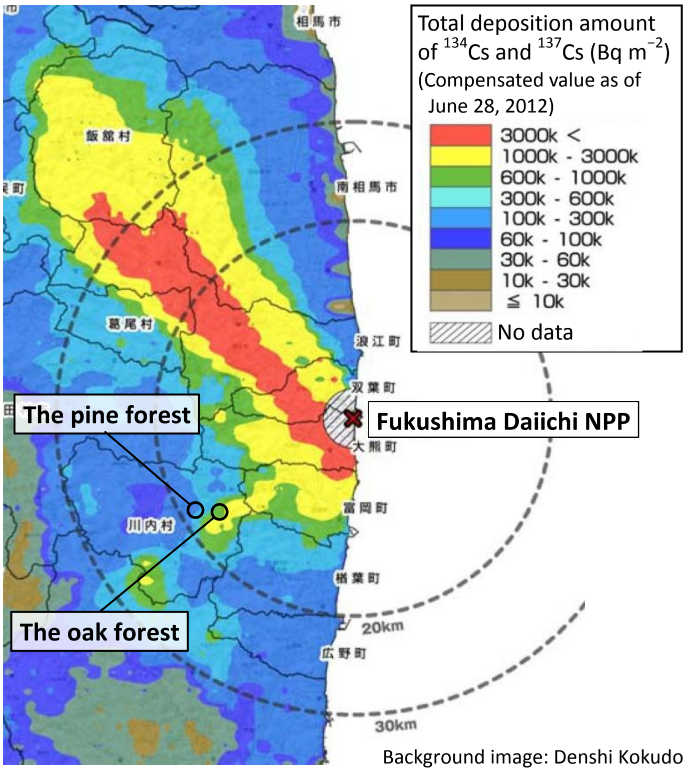

Figure 1. Radiocesium deposition on the ground surface (MEXT, 2012) and locations of study sites. The map has been modified from the original version.

Figure 2

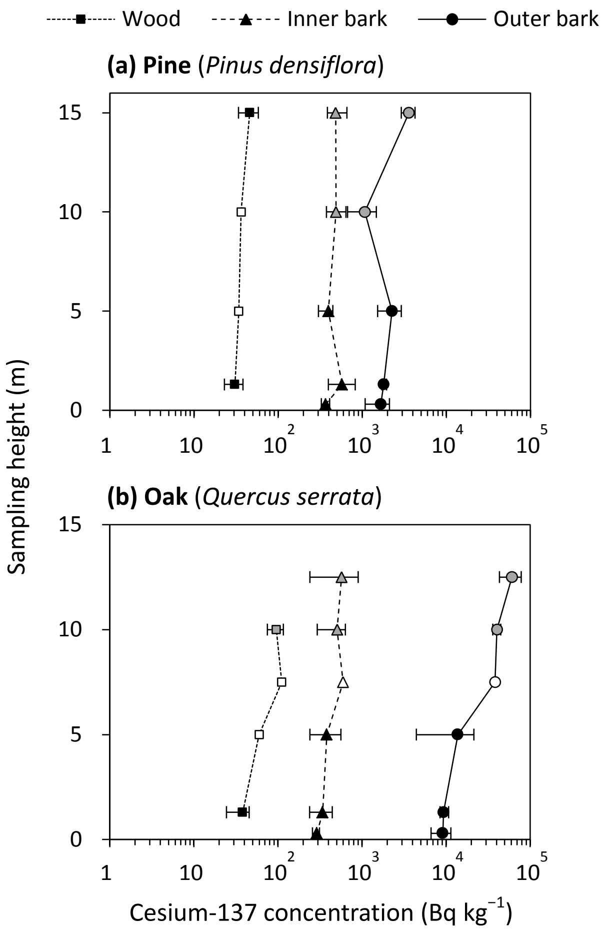

Figure 2. Vertical distribution of ^{137}Cs concentration in each stem part of pines (*Pinus densiflora*) and oaks (*Quercus serrata*). Black symbols are the mean value from three individuals, gray symbols are the mean values from two individuals, and error bars indicate the maximum and minimum values. White symbols show the values from one individual.

Figure 3

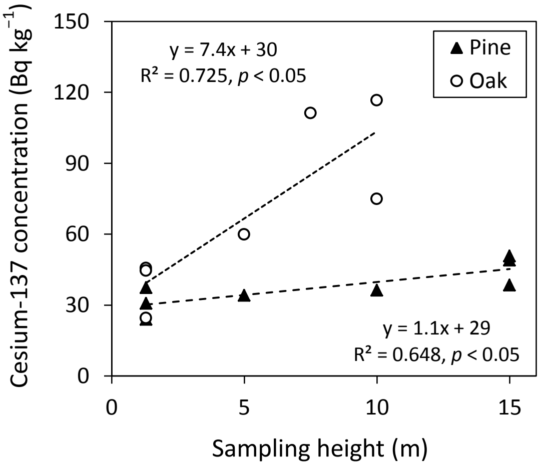

Figure 3. Relationships between the sampling height of wood disk and ^{137}Cs concentration in the wood of pines (*Pinus densiflora*) and oaks (*Quercus serrata*).

Figure 4

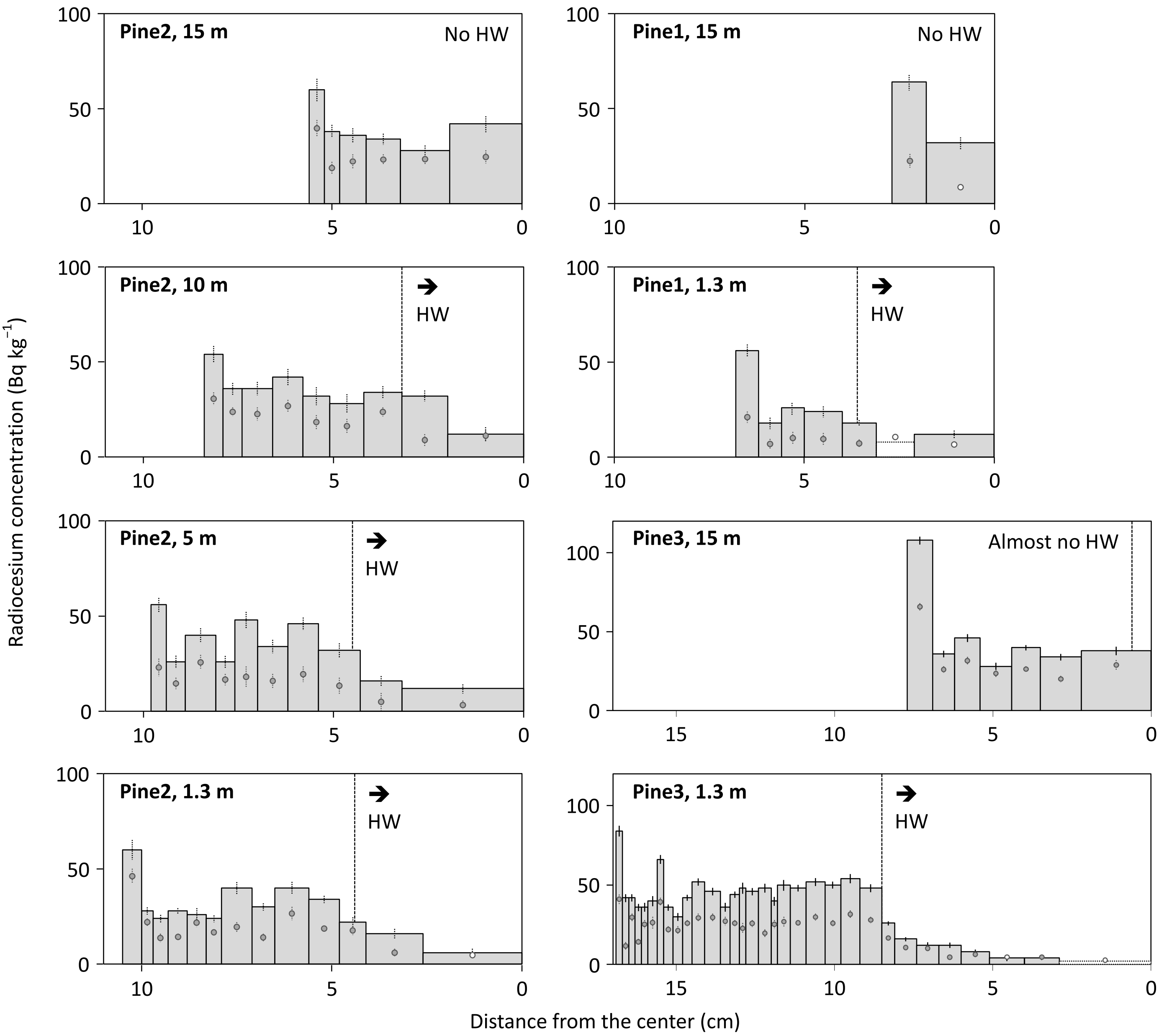

Figure 4. Radial distribution of ^{137}Cs and ^{134}Cs concentrations in wood disks collected at different vertical positions from three pines (*Pinus densiflora*). Gray bars indicate ^{137}Cs concentration and gray circles indicate ^{134}Cs concentration. Error bars indicate standard deviations from counting statistics (σ). White bars and white circles indicate that ^{137}Cs and ^{134}Cs were not detected and show the detection limit. Broken lines indicate the position of the sapwood–heartwood boundary. HW: heart wood.

Figure 5

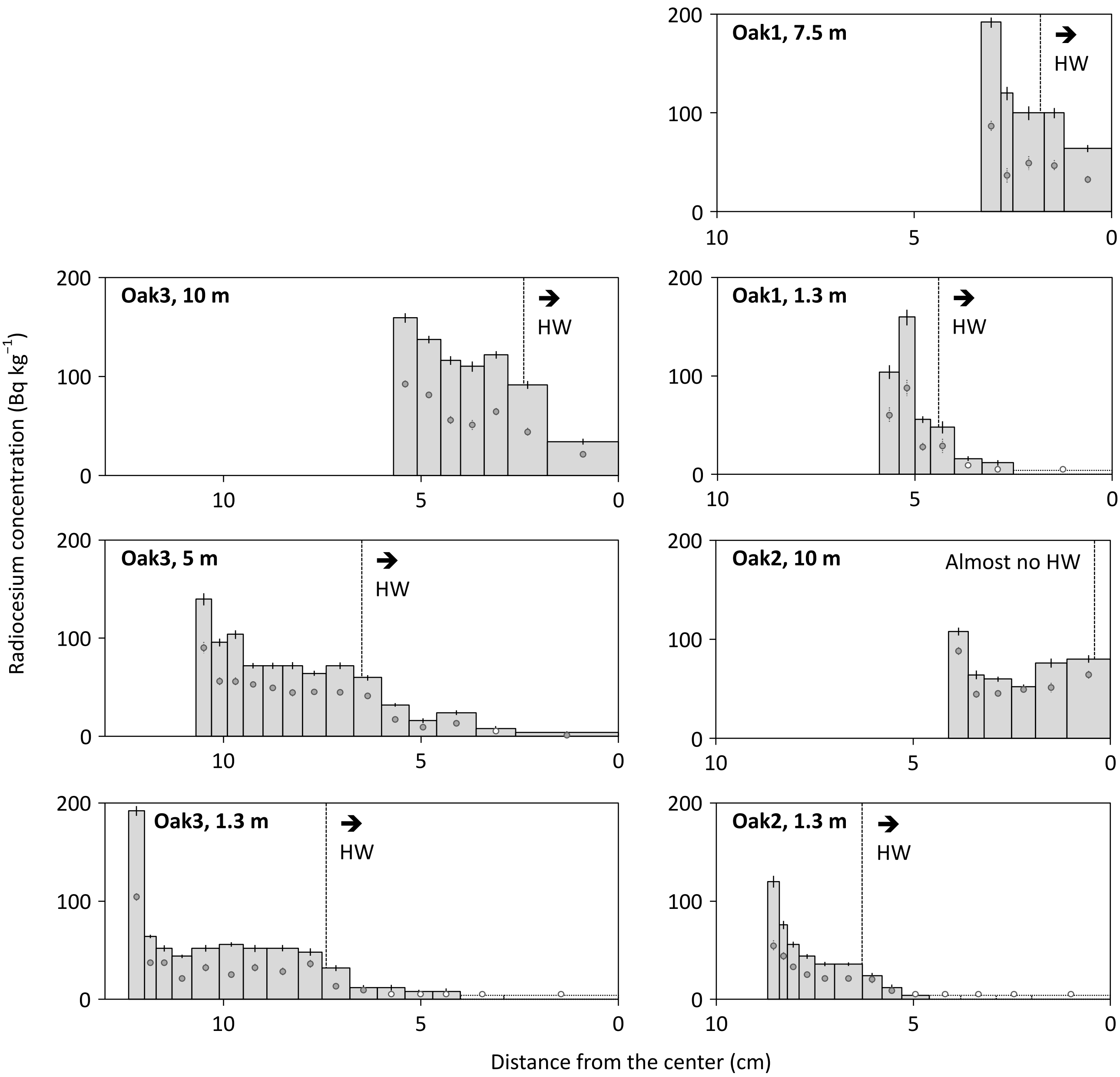

Figure 5. Radial distribution of ^{137}Cs and ^{134}Cs concentrations in wood disks collected at different vertical positions from three oaks (*Quercus serrata*). Gray bars indicate ^{137}Cs concentration and gray circles indicate ^{134}Cs concentration. Error bars indicate standard deviations from counting statistics (σ). White bars and white circles indicate that ^{137}Cs and ^{134}Cs were not detected and show detection limits. Broken lines indicate the position of the sapwood–heartwood boundary. HW: heartwood.

Table 1

Table 1. Description of sample trees.

Species	No.	DBH (cm)	Height (m)	Age (y)
Pine (*Pinus densiflora*)	1	15.1	17.9	36
	2	25.3	21.7	44
	3	36.9	20.9	54
Oak (*Quercus serrata*)	1	13.9	11.8	43
	2	20.3	16.7	43
	3	29.2	17.6	43

DBH: diameter at breast height

Table 2

Table 2. Cesium-137 concentration and radioactivity distribution in stem disks collected at 1.3 m above the ground and aggregated transfer factor (T_{ag}) from soil to each stem part

Species	No.	Part[a]	Concentration[b] (Bq kg^{-1})		Radioactivity distribution (% of whole disk)	T_{ag}[c] (m^2 kg^{-1})
Pine (*Pinus densiflora*)	1	Outer bark	2.0×10^3	(1%)	78	1.7×10^{-2}
		Inner bark	5.0×10^2	(4%)	7	4.2×10^{-3}
		Sapwood	2.9×10	(5%)	14	2.4×10^{-4}
		Heartwood	5.5	(19%)	1	4.7×10^{-5}
	2	Outer bark	1.8×10^3	(1%)	76	1.5×10^{-2}
		Inner bark	4.0×10^2	(5%)	4	3.4×10^{-3}
		Sapwood	3.4×10	(3%)	19	2.9×10^{-4}
		Heartwood	1.2×10	(14%)	1	1.0×10^{-4}
	3	Outer bark	1.7×10^3	(1%)	73	1.4×10^{-2}
		Inner bark	8.3×10^2	(2%)	6	7.0×10^{-3}
		Sapwood	4.6×10	(1%)	19	3.9×10^{-4}
		Heartwood	1.1×10	(5%)	1	8.9×10^{-5}
	Mean	Outer bark	1.8×10^3	(1%)	74	1.5×10^{-2}
		Inner bark	5.8×10^2	(2%)	6	4.9×10^{-3}
		Sapwood	3.6×10	(1%)	19	3.1×10^{-4}
		Heartwood	9.4	(5%)	1	7.9×10^{-5}
Oak (*Quercus serrata*)	1	Outer bark	1.1×10^4	(1%)	90	5.6×10^{-2}
		Inner bark	4.5×10^2	(2%)	4	2.3×10^{-3}
		Sapwood	9.9×10	(4%)	4	5.2×10^{-4}
		Heartwood	1.8×10	(10%)	2	9.5×10^{-5}
	2	Outer bark	9.0×10^3	(< 1%)	93	4.7×10^{-2}
		Inner bark	3.4×10^2	(3%)	3	1.8×10^{-3}
		Sapwood	5.0×10	(2%)	3	2.7×10^{-4}
		Heartwood	5.7	(12%)	1	3.0×10^{-5}
	3	Outer bark	8.5×10^3	(< 1%)	88	4.4×10^{-2}
		Inner bark	2.4×10^2	(3%)	3	1.3×10^{-3}
		Sapwood	6.6×10	(2%)	8	3.5×10^{-4}
		Heartwood	1.1×10	(9%)	1	5.7×10^{-5}
	Mean	Outer bark	9.4×10^3	(< 1%)	90	4.9×10^{-2}
		Inner bark	3.4×10^2	(2%)	3	1.8×10^{-3}
		Sapwood	7.2×10	(1%)	6	3.8×10^{-4}
		Heartwood	1.2×10	(6%)	1	6.1×10^{-5}

[a] Transition part from sapwood to heartwood was included in sapwood.

[b] Percentage figures in parentheses are relative standard deviations from counting statistics.

[c] Although ^{137}Cs in outer bark is not transferred from the soil, T_{ag} was calculated as a reference of deposition.