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Assessment of Potential Risk to Aquatic Organisms by Zinc Originating from Swine Farm Effluent in a Rural Area of Japan

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Abstract

Commercial swine production can cause local water pollution if effluent water containing high levels of nitrogen (N), phosphorus (P), pathogen and heavy metals is discharged, but the actual status remains poorly documented. We thus selected zinc metal (Zn) as a tracer and conducted water quality observations in a rural river of Japan to reveal the significance of mid-sized swine farms on Zn pollution there. Results showed that one farm was the major contributor of Zn load in watershed, corresponding to 82% of total Zn load at a normal stage of water, which was apparently attributable to insufficient wastewater treatment at the farm. Consequently, the Zn concentration at the outlet was estimated at 37 μg L^{-1}, revealing a significant ecological risk at 9.1% to aquatic organisms. We then analyzed the effectiveness of improved wastewater treatment of the farm and found that it had a plausible capacity to keep Zn concentration at the outlet below the environmental quality standard level (30 μg-Zn L^{-1}) and the ecological risk at a safety level (<5%) as well. Based on these results, we concluded that since a swine farm could be a significant source of pollutants in rural areas, appropriate treatment of wastewater would be essential to sustain the potential pollution by Zn.

Discipline: Agricultural environment
Additional key words: ecological risk, water quality

Introduction

Effluent water from a commercial swine farm can be a significant source of pollutants, including nitrogen (N), phosphorus (P), pathogens, and heavy metals, in public waters (Steinfeld et al. 2006). Such pollution has been an environmental concern in Japan, where there were 5670 swine farms in 2012. Eighty percent of the latter are small- to medium-sized farms, each of which raises less than 2000 hogs (MAFF). However, the extent of effluent water pollution from swine farms remains poorly documented, since individual farms tend to be distributed in rural areas, where the water quality has not been monitored as part of any nationwide water quality monitoring program such as that of the Ministry of the Environment (MOE) of Japan.

Kasuya et al. (2010) monitored the water quality of a typical rural river with medium-sized swine farms in its watershed for two years. They reported serious pollution of the water by N and P, much of which was due to swine farm effluent. However, they did not attempt to quantify the contribution of swine farm effluent to the pollution because there were other sources of N and P in watershed, including upland fields, domestic wastewater, and natural sources.

Metallic zinc (Zn) can be an effective tracer for quantifying the contribution of swine farming to water pollution because Zn, as well as copper (Cu), is typically present at high concentrations in effluent from swine farms (Abe et al. 2012, Suzuki et al. 2010). These elements are added to commercial swine feeds to promote gain in body weight (Nicholson et al. 1999, Tanaka 2005), but the swine absorb only 10–20% of the Zn and Cu and excrete the rest in their

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feces (Tanaka 2005). Consequently, swine feces tend to be heavily contaminated with Zn and Cu, with reported concentrations of 431–471 mg kg^{-1} and 135–374 mg kg^{-1} for Zn and Cu, respectively (Isobe & Sekimoto 1999, Nicholson et al. 1999). In Japan, much of the fecal waste is removed from the barn for manure production, while the rest is diverted into wastewater alongside urine and maintenance water used to clean the swine enclosures. Consequently, the wastewater from a swine farm is highly contaminated with Zn and Cu; typical concentration ranges are 1.3–40 (average 11) mg L^{-1} for Zn and 0.44–21 (average 3.7) mg L^{-1} for Cu (Abe et al. 2012).

Zinc is a crucial element for organisms, but its toxicity to aquatic organisms at high concentrations has remained a concern (EU 2010). In 2003, the Japanese government therefore established environmental quality standards (EQSs) for total Zn in public water to protect aquatic ecosystems; the standard (upper limit) is 0.03 mg-Zn L^{-1} in freshwater. A holistic assessment of Zn concentrations in public waters of Japan (Tsushima et al. 2010), using public water monitoring data from 1991 to 2002, showed that about 20% of public waters in Japan exceeded the EQSs. Major sources of pollution were determined to be effluents from mines, industrial facilities, and publically owned treatment works (POTWs). In that analysis, however, the contribution of swine farms to Zn pollution was not studied.

The goal of this research was to determine the extent to which a typical rural river in Japan was polluted by Zn originating from swine farms. We observed water quality in the river, estimated the extent of wastewater treatment by the farms, and then assessed the ecological risks to riverine habitats based on the annual average concentration of Zn estimated at the watershed outlet. We also formulated a plausible countermeasure for the pollution.

Materials and methods

1. Study area

This study was executed in Aichi prefecture, central Japan (Fig. 1), in the same watershed studied by Kasuya et al. (2010). The total drainage area of this watershed is about 3.2 km^2 and the land mainly comprises upland fields (54%) and rice paddies (16.5%).

Cabbage is the most widespread crop in the upland fields. The cropping season extends from August to the following May, depending on the variety of cabbage.

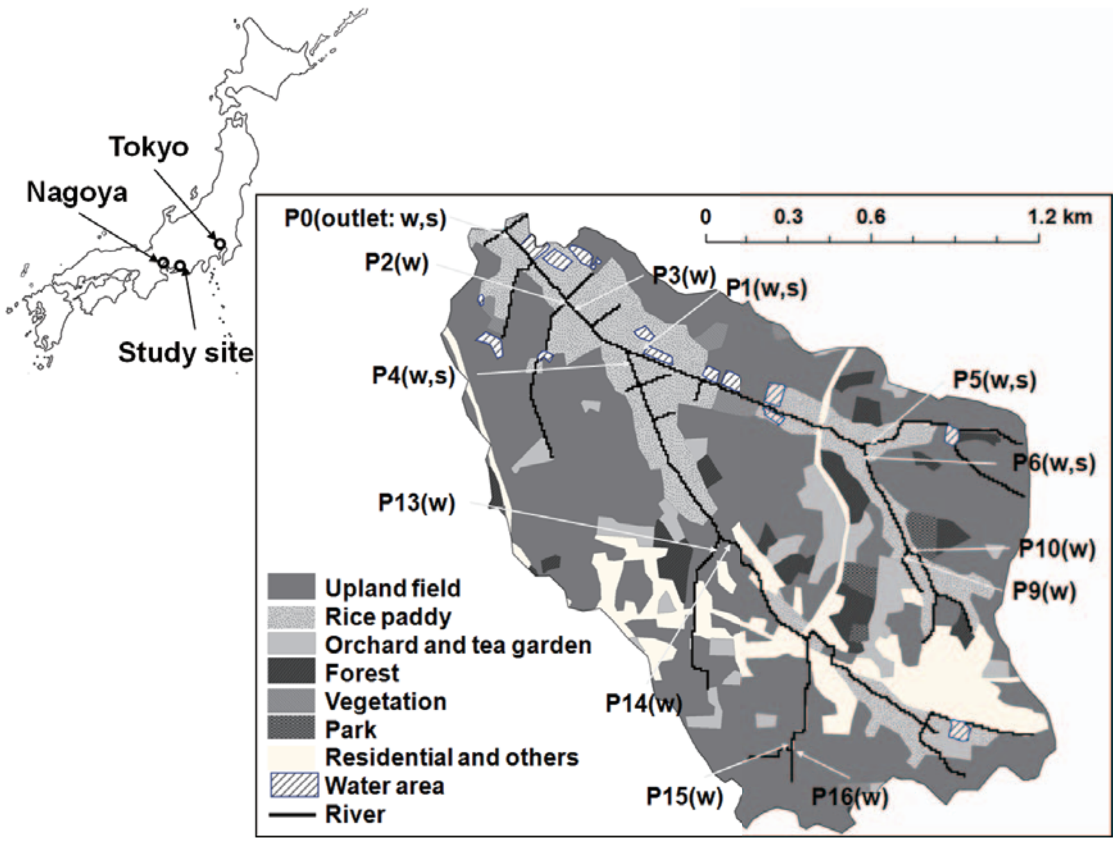

Fig. 1. Location of the study site (left) and observation sites in the watershed (right)
w, sampling sites in winter; s, sampling sites in summer.
The P1 site and its upper reaches (P5, P6, P9 and P10) are in the northern tributary, while P4 and its upper reaches (P13-P16) are in the southern tributary.

Chemical fertilizer is applied at rates of 280–300 kg-N ha^{-1}, 150 kg-P_2O_5 ha^{-1}, and 300 kg-K_2O ha^{-1}. Organic manure (at a rate of 20 t ha^{-1} for hog manure) is applied between July and December before planting (Aichi prefecture 2011).

Paddy rice is grown between late April and late September, with the cultivation period depending on the variety. Recommended application rates of chemical fertilizer are 80–110 kg-N ha^{-1}, 50 kg-P_2O_5 ha^{-1}, and 40 kg-K_2O ha^{-1}. Organic manure (at a rate of 10 t ha^{-1} for hog manure) is applied before planting (Aichi prefecture 2011).

Two swine farms are located near streams (second-order tributaries) in the northeastern part of the watershed. One farm raises 2765 hogs and discharges its effluent just above site P5; the other raises 1760 hogs and discharges its effluent just above site P6 (Fig. 1). The proportions of fattening and breeding hogs and the types of wastewater treatment at the two farms are unknown. There is no significant point source of Zn in watershed other than these farms.

2. Water sample preparation and load calculation

We measured water discharge rates and collected water samples at several sites in the river system when the water stage was typical of winter and summer conditions (Fig. 1). The water discharge was calculated by multiplying the cross-sectional area by the flow velocity, both of which were measured manually at each site.

The winter (dry season) measurements were conducted on 10 February, 2011. Rainfall during the two days preceding the measurements totaled only 1.5 mm, and was the first after a one-month drought, which meant no significant effect of rainfall on river flow was apparent on the observation date. The summer (rainy season) measurements were conducted on 31 August, 2011, which is also the irrigation season for rice paddies. Rainfall during the week preceding the measurements totaled 41 mm. We collected two extra samples on 31 August, 2011 at sites just below the outlets where effluent from swine farms near sites P5 and P6 was discharged.

Total Zn and Cu concentrations of all samples were measured by nitrate-perchloric acid digestion followed by inductively coupled plasma mass spectrometry. We also measured concentrations of dissolved Zn and Cu in summer samples after filtration through a 0.2 μm-pore cartridge.

The loading of Zn (Cu), $L_{\mathrm{Zn\,(Cu)}}$ (μg s^{-1}), in the discharge was calculated as follows:

$$L_{\mathrm{Zn\,(Cu)}} = DR \times C_{\mathrm{Zn\,(Cu)}} \quad (1)$$

where DR (L s^{-1}) is the water discharge and $C_{\mathrm{Zn\,(Cu)}}$ (μg L^{-1}) is the concentration of either Zn or Cu in water.

3. Estimating Zn and Cu loads from the swine farms

We calculated the water discharge and Zn and Cu loads at sites P5 and P6 derived from the swine farms (point source), upper tributaries to site P6 (i.e. sites P9 and P10) and irrigation water from a pond or canal as follows:

$$P5_{DR(Zn,\,Cu)} = Farm_{DR(Zn,\,Cu)} + Irr_{DR(Zn,\,Cu)} \quad (2)$$

$$P6_{DR(Zn,\,Cu)} = Farm_{DR(Zn,\,Cu)} + P9_{DR(Zn,\,Cu)} + P10_{DR(Zn,\,Cu)} + Irr_{DR(Zn,\,Cu)} \quad (3)$$

where the subscript $DR(Zn, Cu)$ indicates the water discharge and Zn or Cu load parameters. *P5*, *P6*, *P9,* and *P10* indicate the parameter values at sites P5, P6, P9, and P10, respectively, while *Farm* and *Irr* indicate swine farm effluent and irrigation water, respectively.

In Eqs. 2 and 3, the observed water discharge and Zn and Cu loads in winter and summer were substituted for *P5*, *P6*, *P9*, and *P10* at the corresponding sites. We calculated the discharge from each swine farm by multiplying the number of hogs by the daily water discharge associated with one hog (10 L hog^{-1} day^{-1}; LEIO 1997). The discharge of irrigation water was unknown, so we estimated the value at each site by balancing the water discharge in each equation. The Zn and Cu concentrations in the irrigation water were assumed to be the average background concentrations in watershed (5.9 μg-Zn L^{-1} and 1.7 μg-Cu L^{-1} at site P13, where effluent from an irrigation pond was being discharged).

The Zn (Cu) load from the swine farms was calculated by inserting values for the remaining unknowns into Eqs. 2 and 3, whereupon concentrations of Zn (Cu) in the effluent were calculated by dividing the Zn (Cu) load by the water discharge from each farm. The water discharge and Zn (Cu) concentrations at sites P9 and P10 measured during the winter were used for the corresponding summertime values.

4. Estimating the annual output of Zn, the developed load, and the output ratio for a swine farm

The annual Zn load in the effluent was calculated by multiplying the average concentration of Zn in the effluent during the winter and summer by the annual effluent discharge. The output ratio for a farm was defined as the ratio of the total annual Zn load from the farm to the total Zn excreted by all hogs on the farm each year. The latter, which was defined as the annual developed load, was estimated as follows.

The ratio of the mass of Zn to that of phosphorus (P) in swine excreta in Japan was estimated at 0.0154 based on annual amounts of P and Zn in excreta (Mishima et al. 2005). The amount of P excreted by one hog was reportedly 8.7 g-P day^{-1} for a fattening hog and 15.6 g-P day^{-1} for a breeding hog (LEIO 1997). From these numbers, we obtained annual Zn excretion rates per hog of 48.9 g y^{-1} for a fattening hog and 87.7 g y^{-1} for a breeding hog, which were then multiplied by the number of hogs in each farm to determine the obtained potential annual developed load at the farm.

5. Estimating annual average Zn concentration at the watershed outlet

We calculated the annual average concentration of Zn at the watershed outlet (site P0) by dividing the total Zn load by the total water discharge at the normal stage. We assumed the total Zn load to be derived from the two swine farms above sites P5 and P6 and the background water, which was not influenced by the swine farms. We assumed the annual delivery ratio for Zn load from the farms at 1.0 and used the reported annual water discharge at the watershed outlet, site P0, for the period 2004–2006 (Kasuya et al. 2010).

6. Ecological risk assessment for aquatic organisms exposed to Zn

We compared the annual average concentration of Zn with a threshold value to assess the ecological risk, in accordance with the preceding assessment of Naito et al. (2010), who used the total Zn concentration (both dissolved and particulate forms of Zn) in Japanese public waters. For the threshold value, we used the HC_5 value, which is 26.7 μg-Zn L^{-1}; corresponding to the 95% protection level with respect to the risk at organism level and calculated from the species sensitivity distribution (SSD; Tsushima et al. 2010). The SSD parameters were 154 and 2.90 μg-Zn L^{-1} for the geometric mean and standard deviation of the geometric mean, respectively (Tsushima et al. 2010).

7. Scenario analysis for risk management

A two-criterion scenario was tested to determine whether Zn concentrations and the ecological risk at the outlet would be lowered by improving wastewater treatment from the farm at P5. The test scenario assumed the Zn concentration in effluent water to be controlled by the criteria for the maximum and average concentrations of Zn in effluent waters: 4400 and 470 μg-Zn L^{-1}, respectively (Fig. 3; Abe et al. 2012). We assumed that the effluent discharge remained the same.

Results and discussion

1. Distribution of Zn loads in the river system

The horizontal and vertical dimensions of the rectan-

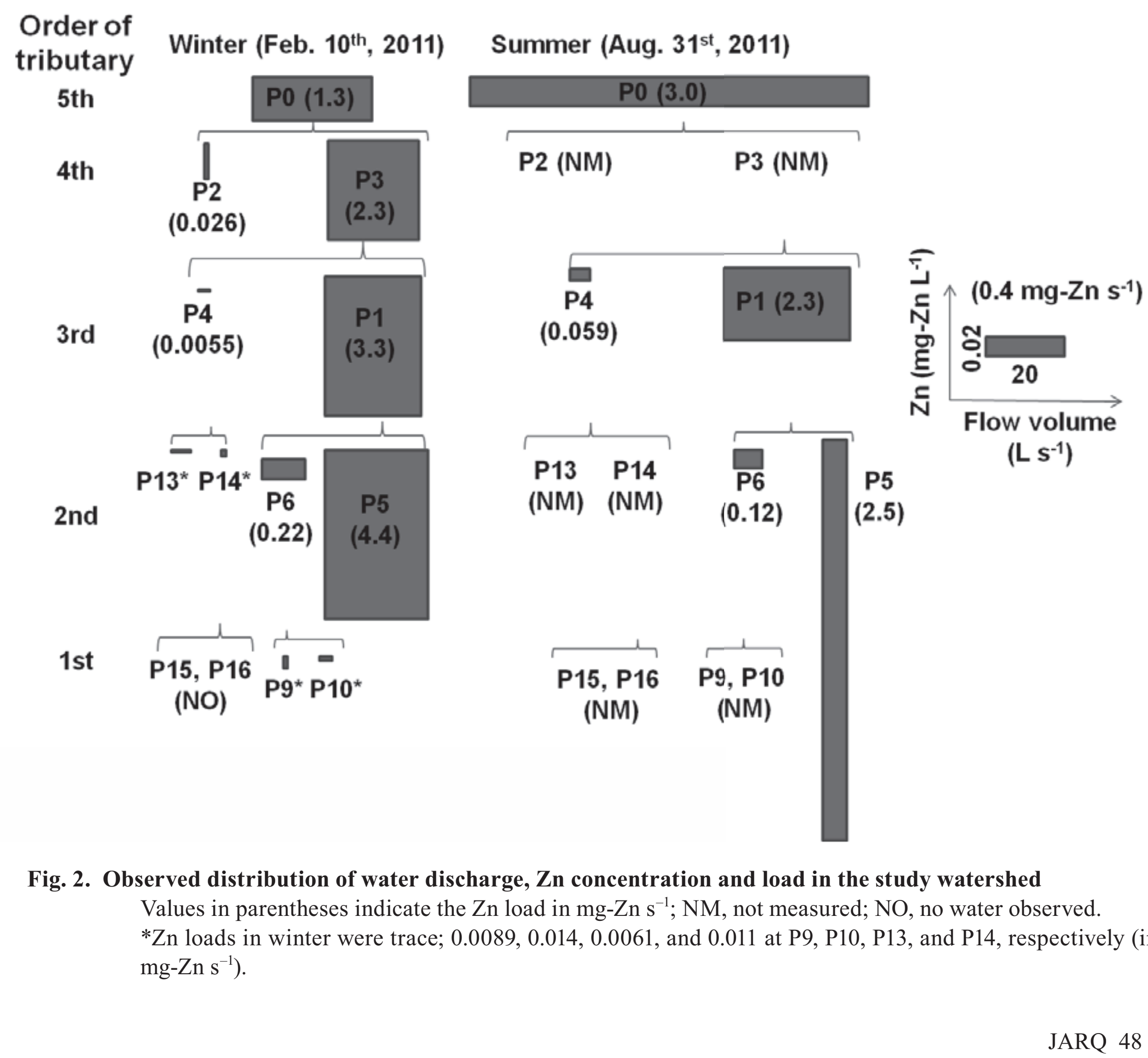

Fig. 2. Observed distribution of water discharge, Zn concentration and load in the study watershed
Values in parentheses indicate the Zn load in mg-Zn s^{-1}; NM, not measured; NO, no water observed.
*Zn loads in winter were trace; 0.0089, 0.014, 0.0061, and 0.011 at P9, P10, P13, and P14, respectively (in mg-Zn s^{-1}).

gles in Fig. 2 represent the water discharge (L s^{-1}) and Zn concentration (mg L^{-1}), respectively. The area of each rectangle therefore represents the Zn load in units of mg s^{-1} (Eq. 1). The water discharge was calculated from the measured cross-sectional area and flow velocity of the river.

In winter (Fig. 2, left panel), the observed water discharge peaked at site P5 on the northern tributary (26.0 L s^{-1}) and gradually decreased downstream to 23.1 L s^{-1} at site P3. The higher discharge at P5 was due to the input of maintenance water from an irrigation pond into the stream above site P5 (Fig. 1). The measured Zn concentration also peaked at site P5 (170 μg-Zn L^{-1}), and gradually decreased to 44.0 μg-Zn L^{-1} at site P0. Consequently, the Zn load peaked at site P5, 4.4 mg s^{-1}, which was about 3.3 times the Zn load at site P0 (1.3 mg s^{-1}). The Zn load at site P6, was only 0.22 mg s^{-1}, or 5% of the load at P5. At sites downstream of P5 and P6, the measured Zn loads gradually decreased, but all of the loads along the northern tributary exceeded those at site P4 and its upstream sites P13 and P14 on the southern tributary.

In summer, the water discharge distribution changed. The water discharge at P5 (6.1 L s^{-1}) was much smaller in summer than in winter, while discharges at P1 and P0 increased to 31.8 and 108 L s^{-1}, respectively (Fig. 2). The reason for this change was the drainage of irrigation waters from rice paddies, which are more numerous along the lower reaches of the river than in upstream areas such as around site P5. Consequently, the Zn load distribution was also quite different in summer than in winter. The Zn load was smaller at site P5 (2.5 mg s^{-1}) in summer than in winter (Fig. 2), while the load in summer at site P5 was even smaller than that at site P0 (3.0 mg s^{-1}), where the load was supplemented by the Zn load from site P6 (0.12 mg s^{-1}). At site P1, on the northern tributary downstream of both P5 and P6, the Zn load (2.3 mg s^{-1}) was almost the same as that at site P5, whereas the load at site P4, on the southern tributary, was consistently small (0.059 mg s^{-1}). These results suggest that P5 was the major source of Zn load at site P0 in both summer and winter.

The rate at which Zn is transported from source to outlet depends on the physical form of Zn compound and the flow velocity. Results from analysis of Zn speciation revealed that the percentage of soluble Zn peaked at site P0 (72.4%) followed by sites P6 (64.7%), P4 (32.7%), P1 (24.7%), and P5 (10.5%). The flow velocity from sites P5 to P0, a distance of about 1.2 km, ranged from 10 to 50 cm s^{-1} (average 26.5 cm s^{-1}) in summer. Therefore, we estimated that the average travel time of soluble Zn from site P5 to the watershed outlet was only 1.26 hours.

Most Zn at site P5 was in particulate form. The transport of particulate Zn to the outlet in summer may have been facilitated by the increasing trend of water discharge from sites P5 to P0 (Fig. 2). In winter, however, the water discharge was similar between sites P5 and P1, and increased only slightly between sites P1 and P0 (Fig. 2). Thus, the lower Zn load at site P0 in winter was probably attributable to the sedimentation of Zn particulate on the river bed in winter. This Zn was probably resuspended and transported downstream in summer when the water discharge increased along the lower reaches of the river (Fig. 2).

Table 1. Concentrations of total and soluble Zn and Cu in river water just below the outlets where swine farm effluents near sites P5 and P6 were discharged

	Zinc		Copper	
	Total μg-Zn L^{-1}	Dissolved μg-Zn L^{-1}	Total μg-Cu L^{-1}	Dissolved μg-Cu L^{-1}
P5 site	580	45 (7.8)	240	39 (16)
P6 site	42	23(55)	11	11 (100)

Parentheses indicate the ratio of concentration in soluble form to total concentration.

2. Actual status of effluent from swine farms

Analysis of the sample of swine farm effluent at site P5 revealed that Zn and Cu concentrations were extremely high, at 580 and 240 μg L^{-1}, respectively (Table 1). Moreover, the fact that Zn and Cu were present in the effluent primarily in particulate form (Table 1) probably accounts for the low proportion of dissolved Zn at site P5. In contrast, concentrations of Zn and Cu were much lower at site P6, while most of the Zn and Cu at that site was dissolved in water (Table 1).

We estimated the Zn and Cu concentrations in the effluent from swine farms at sites P5 and P6 in winter and summer and compared those estimates with the ranges of their concentrations in the effluent from a wastewater treatment facility (WWTF; Abe et al. 2012) (Fig. 3). At site P5 the Zn concentrations were estimated at 13 and 7.7 mg L^{-1} in winter and summer, respectively. Both concentrations exceeded the effluent standard (ES) for Zn (2.0 mg L^{-1}), and the maximum concentration of Zn in treated wastewater (4.4 mg L^{-1}) from the WWTF (Fig. 3, left panel). In contrast, Zn concentrations at site P6 were only 760 and 380 μg L^{-1} in winter and summer, respectively, concentrations that resembled the average concentration of Zn in the treated water (470 μg L^{-1}; Fig. 3, left panel).

Similarly, Cu concentrations in the effluent at site P5 were estimated at 6.4 and 1.3 mg L^{-1} in winter and summer, respectively. The concentration in winter exceeded the ES for Cu (3.0 mg L^{-1}) as well as the maximum concentration of Cu in treated wastewater (2.4 mg L^{-1}; Fig. 3, right panel). At site P6, however, the Cu concentrations were estimated at 1.1 and 0.15 mg L^{-1} in winter and summer, respectively. Although very different from each other, these concentrations were lower than those at site P5 and were close to the

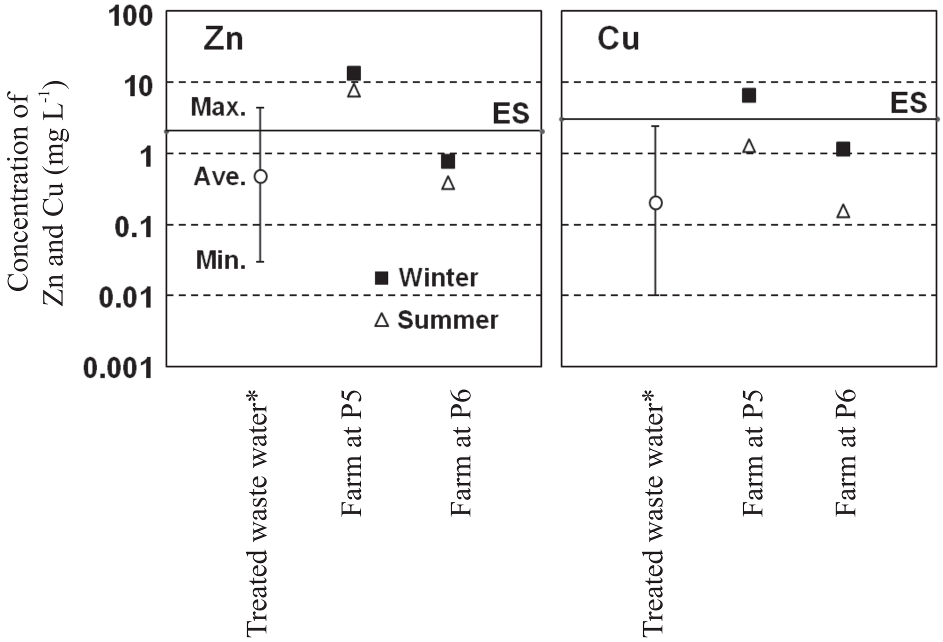

Fig. 3. Estimated Zn and Cu concentrations in the effluents of swine farms near sites P5 and P6
* Ranges for treated wastewater were according to Abe et al. (2012).
ES stands for the effluent standard in Japan; 2 mg-Zn L^{-1} and 3 mg-Cu L^{-1}.

average concentration of Cu in treated wastewater (200 μg L^{-1}).

3. Annual Zn loads from swine farms

Annual Zn loads from the swine farms were calculated from the average concentration of Zn in the effluent (Fig. 3) and the annual total volume of effluent discharged by the farms. These calculations indicated that the Zn loads from the farms at P5 and P6 were 106 and 3.7 kg y^{-1}, respectively. We compared these estimates with the developed load from each farm.

The developed load of Zn from the farm at site P5 was estimated within the range 135 to 242 kg y^{-1} (fattening hog and breeding hog bases, respectively), depending on the proportions of fattening and breeding hogs at the farm. These values sufficed to explain the estimated Zn load of 106 kg y^{-1}. Similarly, in the case of the farm at site P6, the developed load of Zn was within the range 86.1 to 154.3 kg y^{-1}, values that also sufficed to explain the estimated Zn load of 3.7 kg y^{-1}. The results therefore indicated that the estimated Zn loads from both farms were reasonable with respect to material balance.

The output ratios for the farms at P5 and P6 were calculated as 44–78% and 2.4–4.3%, respectively. We discuss the reason for this significant difference in output ratios between the farms at P5 and P6 in the next section.

4. Identifying the dominant source of Zn in the watershed

We found that the concentration of Zn in the swine farm effluent at P5 exceeded both the ES (2.0 mg L^{-1}) and the maximum effluent concentration from the WWTF (4.4 mg L^{-1}; Abe et al. 2012). We consider this high concentration to be undesirable but nevertheless probable based on the following supportive information.

First, Zn concentrations in the effluents from both farms were similar during winter and summer, and there was a consistent difference in concentrations between the two farms (Fig. 3, left panel). Second, Cu concentrations in the farm effluents varied simultaneously with Zn concentrations (Fig. 3). Third, we found that the proportions of dissolved Zn (d-Zn) and dissolved Cu (d-Cu) were significantly lower at site P5 than P6 (Table 1). The Zn and Cu in the raw wastewater from the swine farms were primarily in particulate form (p-Zn and p-Cu, respectively; Suzuki et al. 2010). A WWTF removes particles from the wastewater, the result being that in treated effluent p-Zn and p-Cu account for only 19 and 9% of total Zn and Cu, respectively (Suzuki et al. 2010). Therefore, we concluded that wastewater treatment by the farms at sites P5 and P6 was inadequate and desirable, respectively. Fourth, the Zn output ratio clearly differed between the two farms, as shown in the previous section. We attribute this to differences between the two farms in the efficiency of Zn removal during the wastewater treatment process. Suzuki et al. (2010) reported that Zn removal efficiency increased as the wastewater treatment process proceeded. They found that the Zn effluent concentration after thorough treatment was only 5.3% of the

Table 2. Annual water discharge, Zn load by sources, average Zn concentration and ecological risk at the watershed outlet

Scenario	Water discharge 10^6 m^3 y^{-1}	Background kg-Zn y^{-1}	Zn load P5 site kg-Zn y^{-1}	P6 site kg-Zn y^{-1}	Zn conc. at the outlet μg-Zn L^{-1}	Ecological risk %
Actual status	3.5	20	106	3.7	37	9.1
Max conc.	3.5	20	44	3.7	19	2.6
Average conc.	3.5	20	4.7	3.7	8.1	0.3

* The water discharge is the average value for 2004 and 2006 based on Kasuya et al. (2010).

concentration in the raw wastewater, and that this reduction was mainly due to the removal of p-Zn from the wastewater (Suzuki et al. 2010). We conclude that the lower Zn output ratio of the farm at site P6 compared with that at the WWTF was due to the higher removal efficiency of p-Zn at the WWTF. In contrast, we attribute the higher Zn output ratio of the farm at site P5 to insufficient removal of p-Zn during wastewater treatment.

In summary, we have pointed out that the dominant source of Zn in the river was swine farm effluent at site P5. We have also argued that the cause of the high Zn load from that farm was insufficient treatment of wastewater from the farm, although the actual status of wastewater treatment there is uncertain.

5. Annual average concentration and ecological risk at the outlet

Table 2 shows the annual water discharge at normal stage (Kasuya et al. 2010), the Zn loads from the two swine farms and in background water, and the annual average Zn concentration at the outlet, site P0. The Zn load derived from the farm at site P5 was the highest (106 kg y^{-1}) and accounted for 82% of the total load at the watershed outlet (Table 2). The second highest Zn load (20 kg y^{-1}) was the background load, which accounted for 15% of the total, while the smallest load was attributed to the farm at P6 and accounted for only 2.8% of the total (Table 2).

The annual average concentration of Zn at the watershed outlet was estimated at 37 μg L^{-1} (Table 2), which exceeds the EQSs of 30 μg L^{-1}. Although our estimate is comparable to the observed Zn concentrations of 44 and 29 μg L^{-1} in winter and summer respectively (Fig. 2), water quality varies naturally due to factors such as flow rate, rainfall, temperature, and so forth. We must therefore collect more data to verify our conclusions.

The foregoing results indicate a significant ecological risk from exposure to Zn at the watershed outlet. The estimated Zn concentration at site P0, 37 μg L^{-1}, exceeds the HC_5 criterion (26.7 μg L^{-1}; Tsushima et al. 2010), while the estimated ecological risk is 9.1% based on this Zn concentration and the SSD parameters noted in the materials and methods section (Table 2).

The estimated Zn concentration at the outlet of 37 μg L^{-1} is among the top 13% of Zn concentrations at all public water monitoring sites in Japan, based on the probabilistic distribution model developed by Tsushima et al. (2010), who analyzed Zn concentrations measured during 1991–2002 in public waters throughout Japan. This concentration is equivalent to the mean concentration of Zn at sites influenced by industrial point sources (38 μg L^{-1}) other than POTWs (Tsushima et al. 2010). This result implies that poorly treated effluent from a swine farm can be as significant a source of Zn pollution in rural areas as industrial sources are in urban areas.

6. Analysis of scenarios to mitigate ecological risk

As discussed above, the major source of Zn loading in watershed was the swine farm effluent at site P5. This conclusion suggests that a reduction in the Zn load from that farm might lead to a decrease in both the Zn concentration and the ecological risk at the watershed outlet. To test this hypothesis, we considered two scenarios. In the first, the Zn effluent concentration was maintained at the maximum concentration for Zn in WWTF effluent, whereupon the Zn load from the farm was predicted to decrease by 58% to 44 kg y^{-1} (Table 2), and the Zn concentration at the outlet was predicted to be 19 μg L^{-1}. This value, which is lower than the EQSs and HC_5, suggests that the likely ecological risk would be negligible. In the second scenario, the Zn effluent concentration was maintained at the average concentration of Zn in treated wastewater, which meant that the Zn concentration at the outlet would further decrease to 8.1 μg L^{-1}. From these results, we conclude that Zn in the effluent from swine farms can seriously pollute public waters in rural areas, but that the degree of pollution can be greatly reduced by managing wastewater at the farm appropriately.

7. Implications and conclusions

Water quality has been periodically monitored in the lower reaches of rivers by the MOE as part of a public water quality monitoring program, but no samples were collected in the upper part of the watershed, in the area that was the focus of this research. Data at the nearest monitoring site along the main stream, which were collected 4.2 km below site P0, show that annual average Zn concentrations varied from 18 to 39 μg L^{-1} from 2004 to 2011 (EPD), a range that includes the value of 37 μg L^{-1} at station P0 estimated in this study (Table 2).

However, we found much higher Zn concentrations along the upper northern tributary of the river (sites P3, P1, and P5; Fig. 2 and Table 1). In the observations in this study, the Zn concentration peaked at 580 μg L^{-1} at a site near P5 in summer (Table 1), which far exceeds the average concentration at sites contaminated by mine effluents (89 μg L^{-1}; Tsushima et al. 2010). This observation suggests that local sources of pollution not included in the public water monitoring network may exist.

The European Union (2010) has investigated examples of localized water pollution and ecological risk at sites receiving effluent from industries and POTWs with high Zn concentrations. Similar localized pollution by Zn from point sources in urban areas has also been reported in Japan (Naito & Kamo 2008, Naito et al. 2010). This study has added another example of local pollution caused by a point source, but in this case the source of pollution was a swine farm rather than a POTW or industrial facility.

Abe et al. (2012) have pointed out that even a single swine farm can be a major source of pollution, because the Zn load from a farm raising thousands or tens of thousands of hogs can be equivalent to that from a POTW serving a town or small city. In addition, swine production is associated with other environmental burdens, including, inter alia, N, P, pathogens, and heavy metals (Steinfeld et al. 2006). Because effluent from a swine farm is usually discharged into a nearby public waterway, the farm can be a major source of Zn and other pollutants in a rural area. Appropriate treatment of wastewater from swine farms is therefore recommended to prevent potential water pollution by Zn and possibly other pollutants.

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