

# **Metal Risk Tradeoff Assessment Document**

## **Summary**

Development of Methodologies for Risk Trade-off Analysis toward Optimum Chemical Substance Management,

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National Institute of Advanced Industrial Science and Technology (AIST)  
Research Institute of Science for Safety and Sustainability (RISS)

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#### Authors

Kikuo Yoshida (Project leader)	AIST	
Kiyotaka Tsunemi (Assessment leader)	AIST	
Yuriko Ishikawa	AIST	
Sosuke Ohno	AIST	
Kyoko Ono	AIST	
Masashi Gamo	AIST	
Jun-ichi Takeshita		AIST
Wataru Naito	AIST	
Masashi Kamo	AIST	
Ryoji Makino	AIST	

## **1. Introduction**

There has been a possibility in recent years that new risks may arise from substance substitution. In Europe, six substances—lead, cadmium, mercury, hexavalent chromium, and fire retardants polybrominated biphenyl (PBB) and polybrominated diphenyl ether (PBDE)—have had their use in products restricted by the European Parliament and European Council mandate (Restriction on Hazardous Substances; RoHS) as specified toxic substances used in electric and electronic equipment which went into effect on July 1, 2006. Private companies are expected to comply and respond by substituting for substances restricted by this order which has been moving forward since 2000 on products such as solders.

However, there is also a possibility that risks from substituted substances are increasing. Therefore, to verify whether there is a risk tradeoff between the substituted and substituting substances, certain problems must be overcome, including the difficulty of risk evaluation due to lack of information on exposure and hazardousness of the substituting substances and differences in the end points of the substituted and substituting substances which prevents risk comparisons. Thus there are no means to verify that total risk are being mitigated by substitution of metals, and thus the development of methodology and implementation of risk tradeoff evaluation for analysis is desired.

This evaluation document aims to implement tradeoff evaluation for human health and ecological risks that arise from substance substitution for solder alloys and discuss economic impact to evaluate the desirability of measures for total risk mitigation. In order to do this, there is a need to supplement missing information on exposure and hazardousness by model inference and to compare the risks of the substituted and substituting substances scientifically and quantitatively using the same criteria. Metals such as lead are being substituted under the European RoHS mandate, but whether risk tradeoffs exist needs to be verified. Therefore, we conducted a risk tradeoff evaluation on lead and lead-free solder alloys used for electric equipment boards.

## **2. Estimation of Emission**

### **2.1 Estimation of Mounted Solders**

This study analyzed 4 metals used in solder, including Pb, Cu, Ag, and Sn. Years of analysis were 2000, 2010, and 2020, and time-dependent changes were examined. Two scenarios were set, one in which there is no substitution for lead solder (Sn-Pb) and another in which lead solders are replaced with lead-free solders (Sn-Ag-Cu). Four stages of life cycle were used for the evaluation, including production (metal refining and lead solder manufacturing), processing (solder mounting), final product use, and disposal (recycle, incineration, landfill). Based on a material flow analysis for lead and lead-free solders (Fuse et al., 2011), the amount of metal emission to the environment at each stage of the cycle was estimated. Figure 1 shows the estimation result for the mounted quantity based on material flow analysis. The left side is the scenario without substitution while the right side is with substitution. The mounted quantity decreases with time from 2000 to 2020, but

the real quantity of lead solders reaches zero by 2013 because linear regression was applied here for substitution beyond 2010.

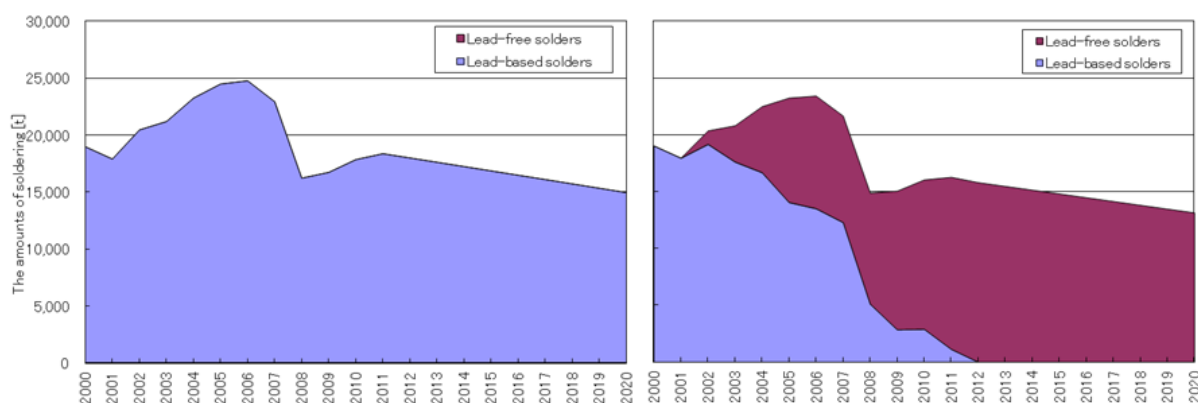


Fig. 1 Estimation for solder mounting by material flow analysis  
(L) Scenario without substitution (R) Scenario with substitution

## 2.2 Setting the Emission Factor and the Amount of Emission

An emission factor was set for each metal at each life cycle stage. An emission factor was set for metal refining and solder manufacturing at the production stage. For metal refining, the emission factor was set based on the amount of production for each metal and PRTR emission data. For emission from solder manufacturing and solder mounting stages, an atmospheric emission factor was calculated by estimating the volatilization rate of 4 metals at operating temperature using the relationship between the vapor pressure of the elements at the refining stage and the emission factor while also taking into account the dust collection rate. The water emission factor was set by accommodation for the operation situation and assumed that there is no emission from the final product use stage. Atmospheric emission from the disposal stage and the water emission factor from landfill repository sites were set for each metal based on the method used in the metal emission scenario document.

Tables 1 and 2 show the atmospheric emission factor and water emission factor, respectively. As the results for the estimation of emission per scenario, Figs. 2 and 3 shows the emission for each metal and the emission for each life cycle, respectively.

In Fig. 2, the atmospheric emission for Pb and Sn was estimated to decrease gradually even without substitution. Because changes in stocks for lead-solder products shift a few years later than the amount of production and processing, Pb and Sn emission was estimated at 1–2 t in 2020, even for a substitution scenario. However, emission of Ag in addition to Pb and Sn was estimated for water. Figure 3 reveals that the refining, manufacturing, and disposal stages contribute to emission, where incineration contributes particularly to atmospheric emission, and refining and landfilling contributes to water emission.

Table 1 Atmospheric emission factor at each life cycle stage

Process		Pb	Sn	Cu	Ag
Production	Refining	$2.3 \times 10^{-5}$	0	$3.3 \times 10^{-6}$	$9.6 \times 10^{-6}$
	Lead solder manufacturing	$1.7 \times 10^{-5}$	$1.2 \times 10^{-5}$	0	0
	Lead-free solder manufacturing	0	$1.3 \times 10^{-5}$	$1.2 \times 10^{-5}$	$1.5 \times 10^{-5}$
Processing	Lead solder mounting	$3.0 \times 10^{-4}$	$2.1 \times 10^{-4}$	0	0
	Lead-free solder mounting	0	$2.3 \times 10^{-4}$	$2.5 \times 10^{-4}$	$2.1 \times 10^{-4}$
Use		0	0	0	0
Desposal	Recycle	0	0	0	0
	General incineration	$4.6 \times 10^{-4}$	$5.5 \times 10^{-4}$	$3.6 \times 10^{-4}$	$1.5 \times 10^{-4}$
	Industrial incineration	$2.3 \times 10^{-3}$	$2.8 \times 10^{-3}$	$1.8 \times 10^{-3}$	$7.5 \times 10^{-4}$
	Landfillin	0	0	0	0

Table 2 Water emission factor at each life cycle stage

Process		Pb	Sn	Ag	Cu
Production	Refining	$1.2 \times 10^{-7}$	0	$7.4 \times 10^{-3}$	$1.4 \times 10^{-5}$
	Lead solder manufacturing	$2.2 \times 10^{-6}$	$1.5 \times 10^{-6}$	0	0
	Lead-free solder manufacturing	0	$1.7 \times 10^{-6}$	$1.8 \times 10^{-6}$	$1.5 \times 10^{-6}$
Processing	Mounting	0	0	0	0
Use		0	0	0	0
Disposal	Recycle	0	0	0	0
	General incineration	0	0	0	0
	Industrial incineration	0	0	0	0
	Landfilling	$1.0 \times 10^{-3}$	$1.0 \times 10^{-3}$	$1.0 \times 10^{-3}$	$1.0 \times 10^{-3}$

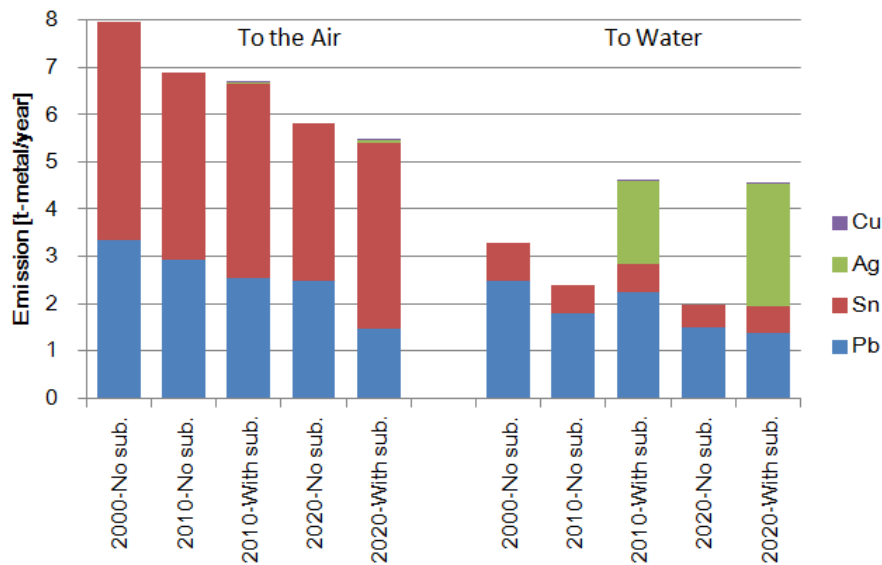


Fig. 2 Estimation of emission into the environment for each metal per scenario

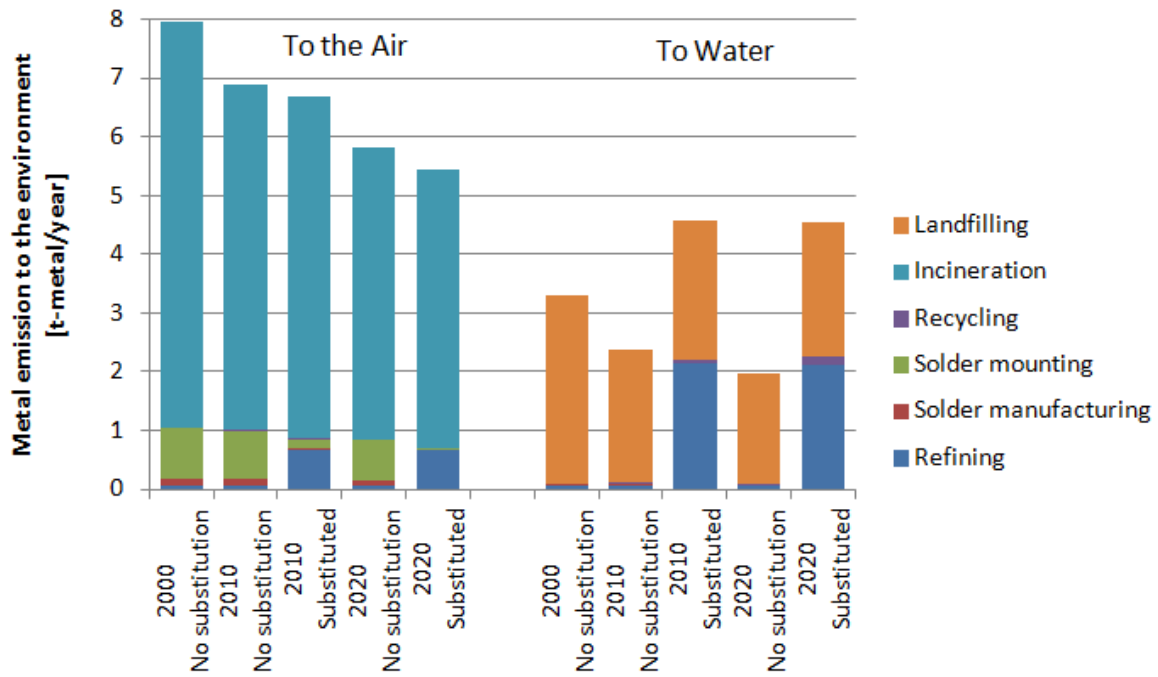


Fig. 3 Estimation of emission into the environment for each life cycle per scenario

### 3. Exposure Analysis

#### 3.1 Estimation of Atmospheric Concentration

Atmospheric concentration and total deposition amount for each metal in 11 regions in Japan (Hokkaido, Tohoku, Hokuriku, Kanto, Chubu, Tokai, Kinki, Chugoku, Shikoku, Kyushu, and Okinawa) were estimated using AIST-ADMER. The 2005 AMeDAS data were used for weather, and input parameters for each metal were set as follows. Domestic observed values (Sakata and Marumoto, 2004) or estimated values by Stokes equation were used for dry deposition velocity. For the washout ratio of Cu and Pb, the relationship between bond strength (ionicity) of the metal elements with water and the washout ratio was calculated based on measured values from overseas (Poster and Baker, 1994). The washout ratio for Sn and Ag were estimated from a linear relationship since there were no data for these metals. Table 3 shows the physical parameters used in the analysis.

Table 3 Physical parameters used in ADMER

paramaters	unit	Pb	Sn	Cu	Ag
decomposition rate constant	[1/sec]	0	0	0	0
washout ratio	[-]	$6.4 \times 10^4$	$1.5 \times 10^5$	$1.3 \times 10^5$	$9.5 \times 10^4$
dry deposition velocity	[m/sec]	$7.3 \times 10^{-3}$	$4.3 \times 10^{-3}$	$1.9 \times 10^{-2}$	$6.2 \times 10^{-3}$
background concentration level	[ng/m <sup>3</sup> ]	0	0	0	0

For AIST-ADMER analysis, an estimated emission of each metal at each life cycle stage was allocated based on the allocation index shown in Table 4. However, sources for emission from refining, solder manufacturing (Kanto), and disposal were designated based on point-source information for each office. Emission from solder manufacturing (excluding Kanto) and solder mounting were allocated on a 5 × 5 km grid based on 2003 industrial statistics for shipment value.

Analysis was conducted per metal and scenario for years 2000, 2010, and 2020. However, both substitution and no-substitution scenarios had the same setting in 2000 since there was no substitution yet.

Table 4 Allocation index for each metal emission

<b>Process</b>	<b>Kanto region</b>	<b>Other regions</b>
<b>Refining</b>	point-source information for each office	
<b>Solder manufacturing</b>	point-source information for each office	<b>2003 industrial statistics for shipment value</b> • Manufacture of fabricated metal products
<b>Solder mounting</b>	<b>2003 industrial statistics for shipment value</b> • Manufacture of fabricated metal products • Manufacture of general machinery • Manufacture of electrical machinery, equipment and supplies • Manufacture of information and communication electronics equipment • Manufacture of electronic parts and electronic circuits • Manufacture of transportation equipment • Manufacture of precision instruments and machinery	
<b>General incineration</b>	point source information for incineration facility	
<b>Industrial incineration</b>	point source information for incineration facility	

Figure 4 shows the change over time in the national average of atmospheric concentrations of each metal per scenario in 2000, 2010, and 2020. While the atmospheric concentration of Pb in the no-substitution scenario decreased by 25%, a decrease of more than

80% can be expected from substitution. However, the atmospheric concentration of Sn was increased by substitution due to the high Sn content of lead-free solder.

Figure 5 shows the estimation results for atmospheric concentration for each region. Comparing Pb before and after substitution, it was revealed that the concentration of each metal tended to be high in the Kanto, Tokai, and Kinki regions in no-substitution scenario. The atmospheric concentration of Pb was estimated to decrease significantly by substitution. The atmospheric concentration of Sn was also estimated to be high in the same regions, but was estimated to increase by substitution. This is due to contributions by the metal products manufacturing industry, transportation machinery, and the equipment industry at the processing (mounting) stage. There was an increase in the atmospheric concentration of Cu and Ag, but the absolute quantity was extremely small.

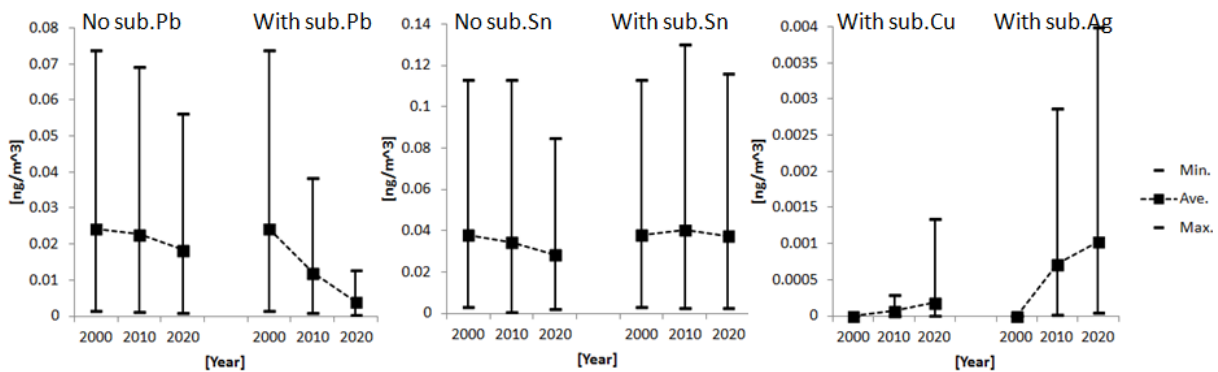


Fig. 4 Results of ADMER analysis. Time change in atmospheric concentration for each metal per scenario (national average).

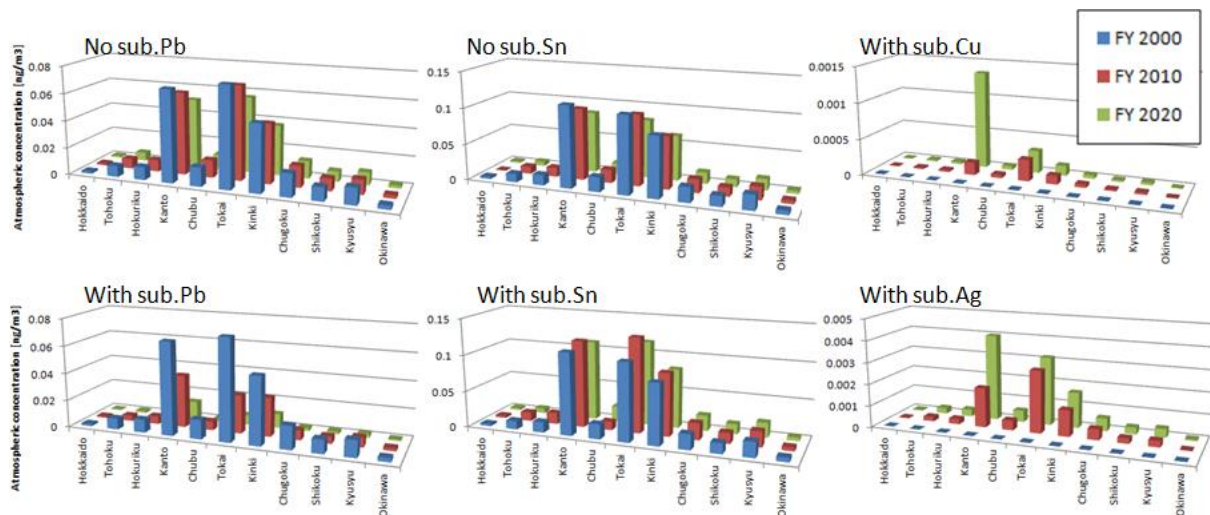


Fig. 5 Results of ADMER analysis. Time change in atmospheric concentration for each metal per scenario (regional).



### 3.2 Estimation of Concentration in Water

Metallic concentration in rivers was estimated for 7 water systems in the Kanto region using the results of water emission from the previous section and the estimated atmospheric deposition by AIST-ADMER into the AIST-SHANEL. Metallic concentration in seawater in Tokyo Bay was also estimated using the coast ecological risk assessment model (AIST-RAMTB). Figure 5 shows the calculation results for AIST-SHANEL.

Table 5 Estimation of concentration in water for each water system (unit:  $\mu$  g/L)

	Kuji River	Naka River	Tone River	Ara River	Tama River	Tsurumi River	Sagami River
2000-No sub. Pb	0.00074	0.00221	0.00410	0.01004	0.01139	0.01649	0.00519
2000-No sub. Sn	0.00094	0.00282	0.00536	0.01377	0.01561	0.02370	0.00709
2010-No sub. Pb	0.00132	0.00290	0.00591	0.01543	0.01676	0.02810	0.00863
2010-No sub. Sn	0.00187	0.00423	0.00906	0.02509	0.02698	0.04814	0.01363
2010-With sub. Pb	0.00059	0.00149	0.00363	0.00907	0.00889	0.01780	0.00441
2010-With sub. Sn	0.00135	0.00228	0.00589	0.01667	0.01671	0.03574	0.00813
2010-With sub. Ag	0.00006	0.00013	0.00023	0.00056	0.00064	0.00086	0.00035
2010-With sub. Cu	0.00002	0.00003	0.00006	0.00014	0.00017	0.00019	0.00009

Metallic concentration in sea water, which is important for estimating the accumulated amount in organisms, was estimated using AIST-RAMTB. The studied area was Tokyo Bay, and the origin of the load to the sea water was set as the atmosphere and the rivers. Atmospheric load was input by interpolating a 5-km data mesh from AIST-ADMER to a 1-km data mesh, while the load from rivers was calculated from dissolved concentration at the mouth of the Naka River, Edo River, Edo River drainage canal, Sumida River, Ara River, Tama River, and Tsurumi River provided by AIST-SHANEL. Figure 6 shows the results.

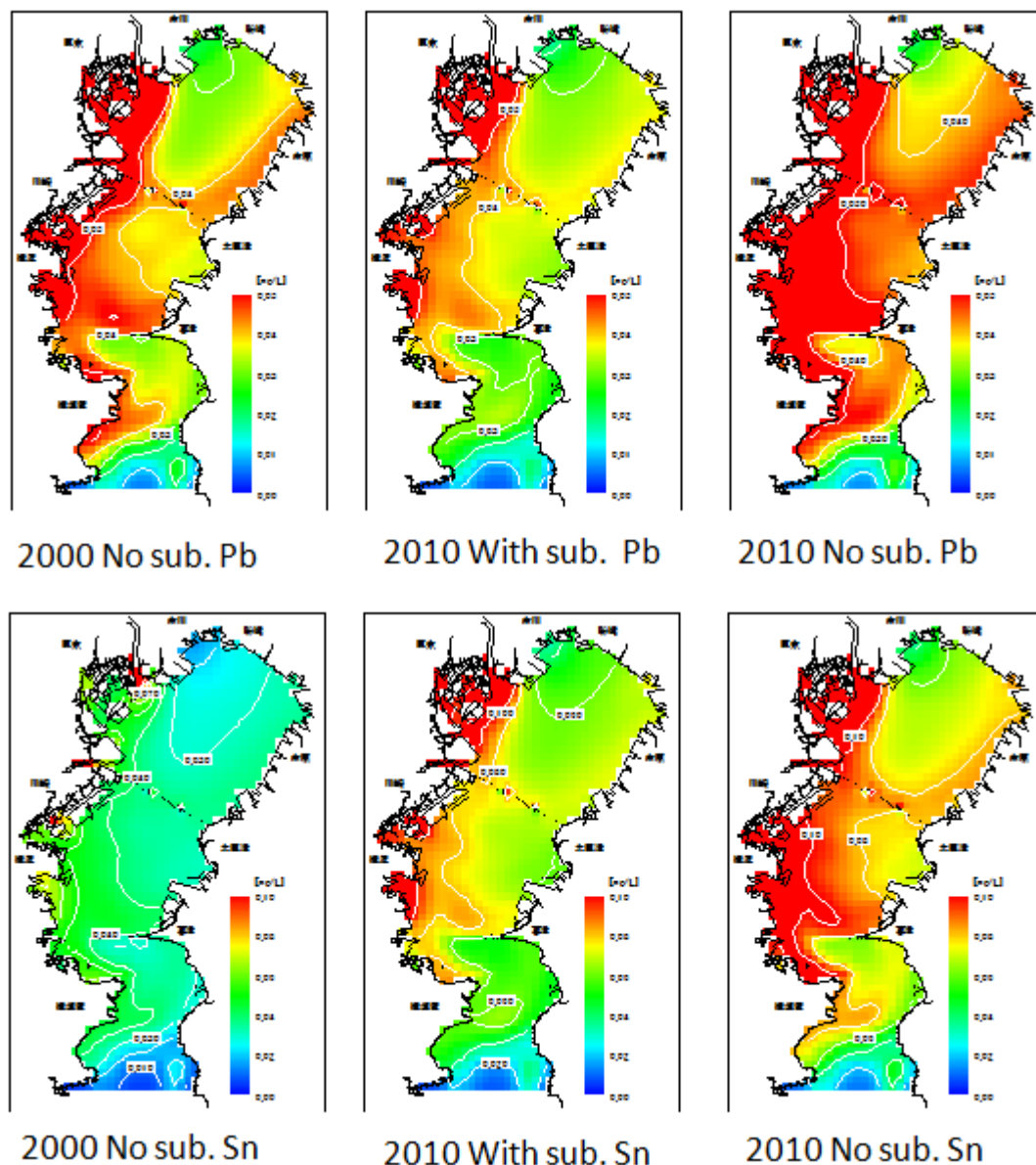


Fig. 6 Estimation of environmental concentration in seawater (first-layer, dissolved concentration, ng/L)

### 3.3 Estimation of Human Intake

Atmospheric deposition calculated by AIST-ADMER (sum of wet and dry deposition) was assumed to be mixed evenly into the soil to depth of 20 cm and converted to soil concentration increase by assuming soil porosity as 0.5 and soil bulk density as 2.5 [g/cm<sup>3</sup>]. Furthermore, the contribution from solders to the concentration in food products was derived for rice, crops other than rice, animal feed, beef, and milk by multiplying a transfer factor set against the increase in soil concentration.

Contribution to concentrations was estimated for every 5-km mesh. By adding the amount of production for each food product, the amount of human exposure via the food products was

estimated. Human exposure was also estimated from concentration data in fish derived from AIST-RAMTB, and human intake by atmospheric inhalation was estimated based on atmospheric concentration from AIST-ADMER. Figures 7 and 8 show the results. The level became 100 ppm for lead to 1 ppm for tin, silver, and copper compared to usual daily intake of Japanese people. Leafy vegetables had a great contribution of lead, while beef greatly contributed tin.

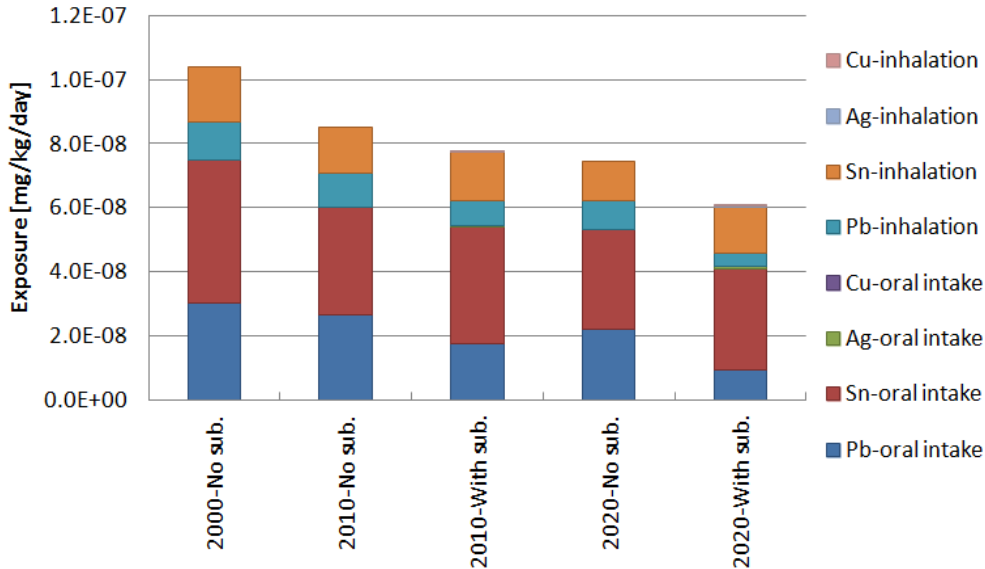


Fig. 7 Human exposure per scenario (average value of Kanto • male)

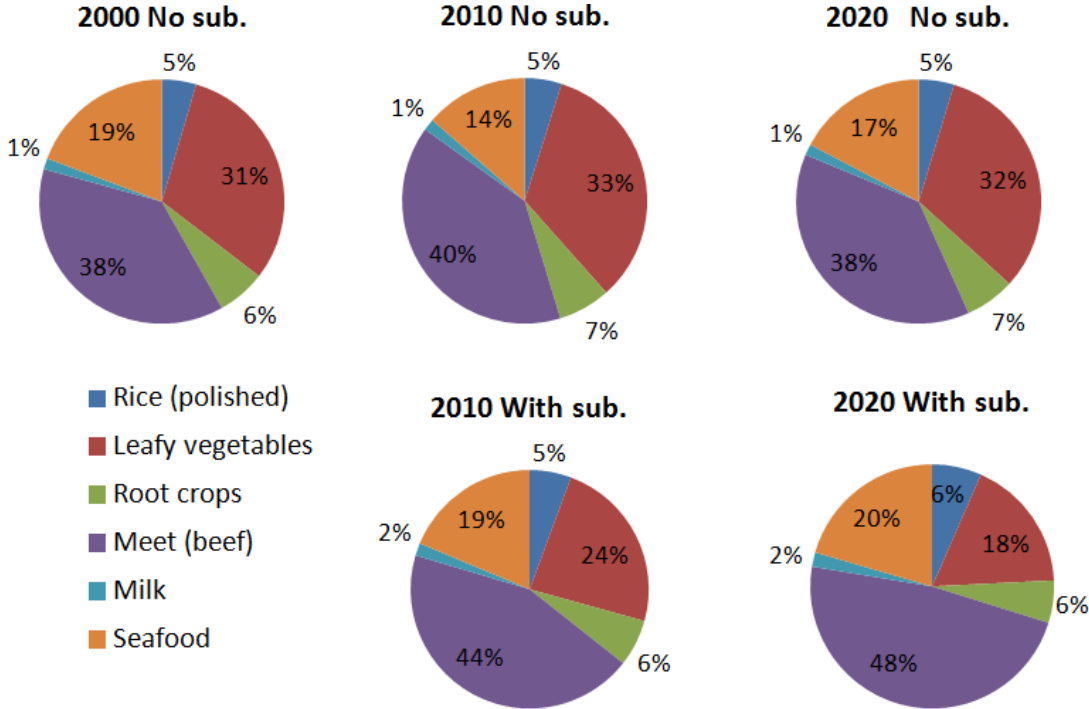


Fig. 8 Breakdown of food products for oral exposure (sum of 4 metals)

#### 4. Effects on Human Health and Tradeoff Evaluation

Symptoms observed for human epidemiological studies and poisoning cases for studied metals (lead, silver, tin, copper) were summarized, while the NOAEL and LOAEL in repeat-dose studies using rats and mice were organized. Except for lead, epidemiological data is limited. Therefore, an evaluation was conducted using an inference algorithm based on relative values of NOAEL/LOAEL per end point organs based on animal experiment data. Figure 9 shows the results for the dose-response relationship for reference substances (effect to kidney by cadmium, effect to liver by vinyl chloride monomer) in humans and dose-response relationship for the above substances derived from the relative NOAEL value (central value) at each end point.

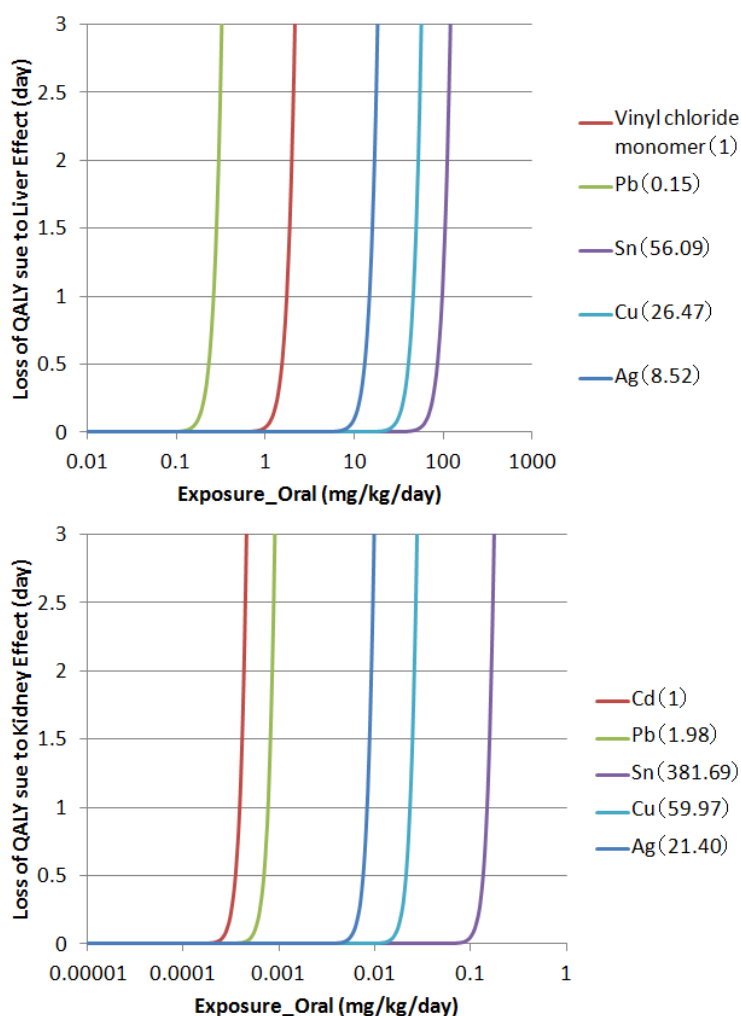


Fig. 9 Estimation of dose-response relationship for 4 metals.

(Upper bar: effect to liver, lower bar: effect on kidney, values inside the brackets are the relative values of NOAEL of each metal where the NOAEL of the reference substance is normalized to 1).

An increase in quality-adjusted life year (QALY) loss was surveyed after increasing the exposure amount from the previous section by 1% from baseline intake for each metal and converting the additional portion to 1 µg/kg/day. Table 6 shows the results from calculating the factors for converting an increase or decrease in the exposure to QALY loss. Table 7 shows the results for QALY loss by solder substitution or lack thereof calculated for both the oral and inhalation routes.

Table 6 Estimation of QALY loss per 1 µg/kg/day exposure change for each metal

Metal	Baseline exposure (mg/kg/day)	Slope factor			
		Loss QALY per 1µg/kg/day (day)			
		Oral intake		Inhalation	
		Liver	Kidney	Liver	Kidney
Cu	0.02	$2.86 \times 10^{-54}$	$1.30 \times 10^{-01}$	$1.04 \times 10^{-13}$	$6.00 \times 10^{-05}$
Sn	0.013	$1.47 \times 10^{-68}$	$1.38 \times 10^{-14}$	$7.55 \times 10^{-26}$	$1.62 \times 10^{-16}$
Ag	0.00012	$1.37 \times 10^{-107}$	$1.92 \times 10^{-28}$	$2.27 \times 10^{-60}$	$1.08 \times 10^{-54}$
Pb	0.00072	$7.33 \times 10^{-34}$	5.28	$1.48 \times 10^{-08}$	$3.06 \times 10^{-01}$

Table 7 Estimation of QALY loss per scenario (Upper: oral, Lower: inhalation)

Loss QALY (day)	Year of 2000	Year of 2010		Year of 2020	
		No-sub	With-sub	No-sub	With-sub
Pb (liver)	$2.2 \times 10^{-41}$	$1.9 \times 10^{-41}$	$1.3 \times 10^{-41}$	$1.6 \times 10^{-41}$	$6.8 \times 10^{-42}$
Pb (kidney)	$1.6 \times 10^{-07}$	$1.4 \times 10^{-07}$	$4.8 \times 10^{-08}$	$8.2 \times 10^{-08}$	$2.5 \times 10^{-08}$
Sn (liver)	$6.2 \times 10^{-76}$	$4.6 \times 10^{-76}$	$5.1 \times 10^{-76}$	$4.3 \times 10^{-76}$	$5.0 \times 10^{-76}$
Sn (kidney)	$5.8 \times 10^{-22}$	$4.6 \times 10^{-22}$	$2.2 \times 10^{-22}$	$4.3 \times 10^{-22}$	$2.2 \times 10^{-22}$
Ag (liver)	-	-	$2.7 \times 10^{-117}$	-	$7.3 \times 10^{-117}$
Ag (kidney)	-	-	$1.8 \times 10^{-38}$	-	$5.0 \times 10^{-38}$
Cu (liver)	-	-	$1.0 \times 10^{-64}$	-	$3.9 \times 10^{-64}$
Cu (kidney)	-	-	$2.3 \times 10^{-12}$	-	$8.5 \times 10^{-12}$
total	$1.6 \times 10^{-07}$ <<0.001	$1.4 \times 10^{-07}$ <<0.001	$4.8 \times 10^{-08}$ <<0.001	$8.2 \times 10^{-08}$ <<0.001	$2.5 \times 10^{-08}$ <<0.001

Loss QALY (day)	Year of 2000	Year of 2010		Year of 2020	
		No-sub	With-sub	No-sub	With-sub
Pb (liver)	$2.0 \times 10^{-16}$	$1.7 \times 10^{-16}$	$1.3 \times 10^{-16}$	$1.5 \times 10^{-16}$	$6.7 \times 10^{-17}$
Pb (kidney)	$4.1 \times 10^{-09}$	$3.5 \times 10^{-09}$	$2.7 \times 10^{-09}$	$3.0 \times 10^{-09}$	$1.4 \times 10^{-09}$
Sn (liver)	$1.4 \times 10^{-33}$	$1.2 \times 10^{-33}$	$1.2 \times 10^{-33}$	$1.0 \times 10^{-33}$	$1.2 \times 10^{-33}$
Sn (kidney)	$3.1 \times 10^{-24}$	$2.6 \times 10^{-24}$	$2.7 \times 10^{-24}$	$2.2 \times 10^{-24}$	$2.5 \times 10^{-24}$
Ag (liver)	-	-	$1.5 \times 10^{-70}$	-	$4.5 \times 10^{-70}$
Ag (kidney)	-	-	$7.1 \times 10^{-65}$	-	$2.2 \times 10^{-64}$
Cu (liver)	-	-	$2.4 \times 10^{-25}$	-	$8.9 \times 10^{-25}$
Cu (kidney)	-	-	$1.4 \times 10^{-16}$	-	$5.2 \times 10^{-16}$
total	$4.1 \times 10^{-09}$ <<0.001	$3.5 \times 10^{-09}$ <<0.001	$2.7 \times 10^{-09}$ <<0.001	$3.0 \times 10^{-09}$ <<0.001	$1.4 \times 10^{-09}$ <<0.001

## 5. Ecotoxicity and Risk Tradeoff Evaluation

In the ecological risk tradeoff evaluation for metals, risk was quantified by using the potentially affected fraction (PAF) as an index for the species sensitivity distribution (SSD).

Much data exists on the biological toxicity of metals. However, the toxicity values of metals are known to change according to water quality. Therefore to compare risks, toxicity data from various conditions need to be corrected for water quality to evaluate the target under the same condition. Methodology for comparing and evaluating the biological risks of metals under the same conditions was developed (risk evaluation methodology development for metals with respect to biological utility). For this tradeoff evaluation, a screening evaluation was conducted using an SSD that does not take biological utility into account for lead, tin, copper, and silver. Risk quantification was attempted using SSD with respect to biological utility for lead and copper.

### 5.1 Species Sensitivity Distributions of Metals

Figure 10 shows the SSD that does not take biological utility into account used in the screening evaluation for each metal. The HC<sub>5</sub> values [ $\mu\text{g/L}$ ] calculated from SSD were 0.19 (silver), 3.35 (copper), 9.63 (lead), and 4.28 (tin). Figure 12 shows the SSD which took biological utility into account for lead and copper. Hardness, pH, and DOC were assumed as 50 mg/L, pH 7, and 1 mg/L, respectively, for water quality. HC<sub>5</sub> [ $\mu\text{g/L}$ ] calculated from this SSD was 1.91 (copper) and 4.18 (lead).

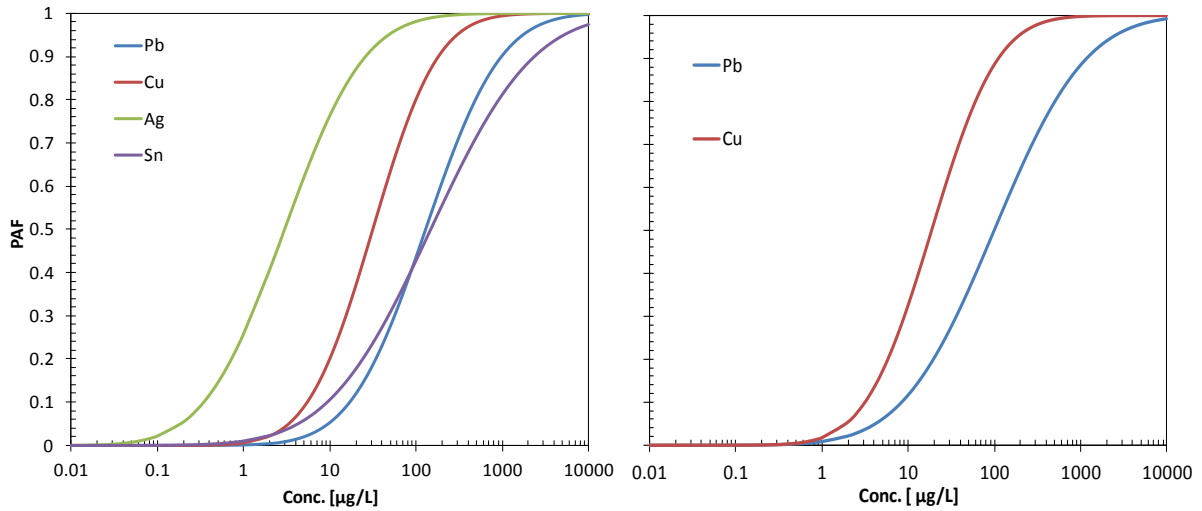


Fig. 10 SSD of each metal

(L: no water quality correction, R: with water quality correction, assuming hardness of 50 mg/L, pH of 7, and DOC of 1 mg/L)

### 5.2 Ecological Risk of Metals in Each Scenario

Using the constructed metal SSD, the ecological risks of metals were calculated under the assumed scenarios. Changes in the risk were derived by assuming each metal concentration at the point of highest concentration in each river in 2000 are equivalent of  $HC_5$ . Figure 11 shows the result for the Tone River.

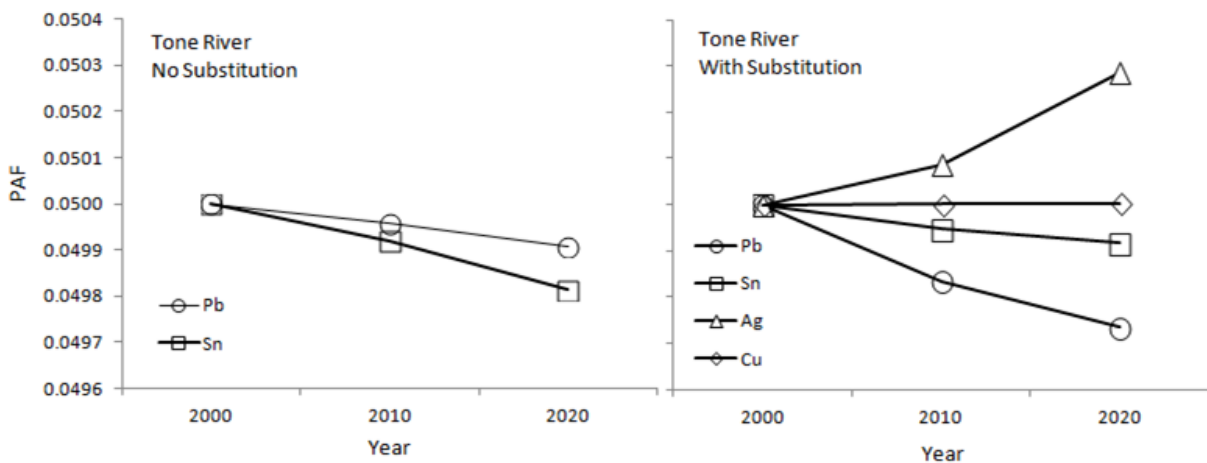


Fig. 11 Time change of risks in the Tone River. Each metal concentrations in 2000 was assumed as  $HC_5$ .

Assuming each metal concentration at the highest concentration point of Tonegawa as  $HC_5$ , the lead and silver risk decreased by 0.01% under the no-substitution scenario.

Assuming there was substitution, the risk for lead and tin had a decreasing tendency, while the silver risk had an increasing tendency. There was very little change for copper. Assuming each metal concentration at the highest concentration point as HC<sub>5</sub>, changes in biological risk by each metal were on the order of 0.01% in all rivers at maximum. Much of the lead concentration detected in each river had values less than HC<sub>5</sub>, while the largest value was essentially equivalent to HC<sub>5</sub>. Biological risks from lead in studied rivers are predicted not to be at a level of concern. In rivers evaluated in this analysis, changes in ecological risk were deemed quite small even when lead solders were replaced by lead-free solders.

## 6. Economic Analysis

Cost-effectiveness analysis for substituting lead solders with lead-free solders was conducted for the substitution scenario for 2020. The equation below was used to estimate the per-year increment of the purchase cost due to the increase in solder price, and the facility renewal cost for replacing solder mounting equipment with lead-free solder-appropriate equipment with high operation temperatures.

$$C_A = C_I \times \frac{r}{1 - (1 + r)^{-n}} + C_Y ,$$

where  $C_I$  is the facility investment cost,  $r$  is the discount rate (0.03),  $n$  is the expected lifetime (15 years),  $C_Y$  is the operating cost, and  $C_A$  is the cost per year, and cost of operation was assumed as constant. Table 8 shows the estimation results at the national level.

Table 8 Estimation of cost increase by solder substitution

Scenario	Solder	Use [t/yr]	Unit Price [yen/kg]	Cost [100 million yen /yr]
No sub. 2020	Lead solder	18,483	1,800	332.7
	Lead solder recycle	3,466	-900	-31.2
With sub. 2020	Lead-free solder	16,173	5,500	889.5
	Lead-free solder recycle	3,032	-2,750	-83.4
Scenario	Additional Facilities	Quantity [units]	Unit Price [1000 yen /unit]	Cost [100 million yen /yr]
No sub. 2020	—			0
With sub. 2020	Lead-free solder tank added	8,985	15,000	112.9
$\Delta C$				1220.5

However, the risk mitigation effect by solder substitution was calculated by QALY. Table 9 shows the results.



Table 9 Estimation of risk mitigation effect by solder substitution

Scenario	Risk [day/person/life]
No sub.	$3.4 \times 10^{-4}$
With sub.	$1.4 \times 10^{-4}$
$\Delta E$ (yr/pop./yr)	1.1

The above results show that the cost-effectiveness of substance substitution is quite poor, where QALY annual acquisition cost is  $\Delta C/\Delta E = 1.1 \times 10^2$  billion yen/year. This indicates that the only merit of hazard-based substitution like the European RoHS mandate is some degree of risk mitigation, and the cost may become quite large at a national level.

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