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Temporal and regional variability of cumulative ecological risks of pesticides in Japanese river waters for 1990–2010

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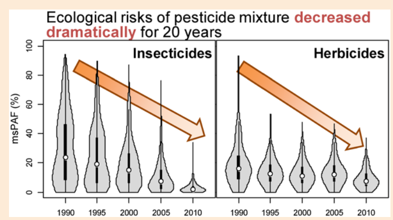
Supplementary material

We quantitatively evaluated the cumulative ecological risks from multiple pesticides used in paddy fields in Japan. Moreover, we visualized the temporal and regional variability of those risks for 1990–2010. Considering the region-specific parameters of environmental conditions, region-specific predicted environmental concentrations were estimated at 350 river-flow monitoring sites in Japan. Then the multi-substance potentially affected fraction (msPAF) was calculated as a risk index of multiple pesticides by using the computation tool NIAES-CERAP. The median msPAF values for insecticides and herbicides decreased by 92.4% and 53.1%, respectively, from 1990 to 2010. This substantial reduction in ecological risk was attributed to the development of low-risk pesticides by manufacturers, the efforts of farmers in risk reduction, and tighter regulation by the Japanese government. In particular, the substantial reduction of the ecological risk from insecticides was largely due to the decrease in the use of organophosphorus insecticides.



Keywords: ecological risk assessment, mixture toxicity, cumulative risk, species sensitivity distribution, aquatic organisms, river.

Introduction

In Japan, pesticide registration criteria based on ecological risk assessments are set by Japan's Ministry of Environment under the Agricultural Chemicals Regulation Law.[1)] Under the risk assessment scheme, acute toxicity tests are conducted for aquatic organisms, and then the acute effect concentration (AEC) is determined as the minimum value of the 50% effective concentration (EC_{50}) or the 50% lethal concentration (LC_{50}) divided by an uncertainty factor that considers species sensitivity differences. Subsequently, the predicted environmental concentration (PEC), which is the peak concentration in river water at the time of pesticide application, is calculated using an environmental model based on a standard scenario in a model basin.[2)] Finally, if the PEC is less than the AEC, the short-term aquatic risk is deemed to be insignificant, and the pesticide is considered to fulfill the registration criteria.

However, there are many different pesticides with low concentrations in the actual aquatic environment. For example, according to a pesticide-monitoring survey conducted on the Sakura River in Ibaraki Prefecture,[3)] more than 30 pesticides were detected simultaneously after rice transplanting. Even if each individual pesticide is evaluated as safe, there is still concern about the mixture toxicity of multiple pesticides. Although in recent years the total amount of pesticide use has tended to decrease, the number of active ingredients has increased. This has led to a "small amount, large variety" of pesticides. When there was less variety of pesticides, it was sufficient to assess the risk of each individual pesticide. However, a change in the method of assessing ecological risk is needed to consider the mixture toxicity of multiple pesticides in this new era.

The objective of the present study was to assess whether the cumulative ecological risk of pesticides used in paddy fields has changed over time due to the shift to the "small amounts, large variety" of pesticides. Therefore, we quantitatively evaluated the cumulative ecological risks of pesticides in Japanese river waters and illustrate the temporal and regional variability of those risks for 1990–2010.

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Materials and methods

1. General description

The species sensitivity distribution (SSD)[4] was used as the key concept for the quantitative ecological risk assessment. Sensitivity to environmental contaminants varies markedly among species, and this variation can be described by the statistical distribution (often a log-normal distribution) estimated from sampled toxicity data (EC_{50} or LC_{50} values). The potentially affected fraction (PAF) can be calculated using the SSD as an index of the magnitude of ecological risk.[5–7] Moreover, the SSD can be applied to assessing the cumulative ecological risk of multiple pesticides as the multi-substance PAF (msPAF) by combining it with mixture toxicity models.[8] Recently, a cumulative ecological risk assessment tool (NIAES-CERAP) that considers the mixture toxicity of multiple pesticides has been developed.[9] The SSD parameters for 68 pesticides[10] are already input in this tool, and the msPAF can be automatically calculated by selecting the pesticide name and inputting the environmental concentration (μg/L) in the Microsoft Excel worksheet.

The region-specific PECs at the Tier 2 level (PEC_{Tier2}) for paddy fields were estimated using the method developed by Yachi *et al.*[11] In their method, important region-specific parameters of environmental conditions—in this case river flow, paddy rice cropped area, and pesticide usage ratio[5]—were used to estimate PECs at 350 river flow–monitoring sites in Japan. This method was validated by comparing the estimated and measured concentrations of 27 pesticides.[11] Moreover, we investigated the actual past use of pesticides based on various studies[12–17] to estimate historic PECs.

2. Analyzed pesticides, sites, and years

The SSDs of the 67 pesticides analyzed were already available in NIAES-CERAP: 25 insecticides (fenobucarb, carbosulfan, benfuracarb, fenitrothion, fenthion, phenthoate, diazinon, fipronil, ethiprole, etofenprox, silafluofen, imidacloprid, clothianidin, dinotefuran, thiacloprid, thiamethoxam, nitenpyram, spinosad, pymetrozine, cartap, thiocyclam, diflubenzuron, buprofezin, tebufenozide, and chlorantraniliprole), 9 fungicides (hydroxyisoxazole, orysastrobin, IBP, isoprothiolane, tricyclazole, phthalide, pyroquilon, chlorothalonil, and probenazole), and 33 herbicides (bensulfuron-methyl, imazosulfuron, pyrazosulfuron-ethyl, cyclosulfamuron, propyrisulfuron, pyrimisulfan, pyriminobac-methyl, simetryn, bentazon, oxadiazon, pentoxazone, oxadiargyl, pyraclonil, carfentrazone-ethyl, pyrazolynate, benzofenap, tefuryltrione, pyrazoxyfen, pretilachlor, mefenacet, cafenstrole, butachlor, fentrazamide, indanofan, thiobencarb, esprocarb, molinate, benfuresate, clomeprop, bromobutide, daimuron, cumyluron, and quinoclamine).[9,10] All are, or were previously, used in Japanese paddy fields. Moreover, carbofuran, as a metabolite of carbosulfan, and nereistoxin, as a metabolite of cartap and thiocyclam, were considered in the risk assessment. Cartap is rapidly degraded to nereistoxin; therefore, the toxicity of cartap is actually nereistoxin toxicity.

The analyzed sites were 350 river flow–monitoring sites in Japan where a method for estimating region-specific PEC has already been developed by Yachi *et al.*[11] The river flow and paddy-field area in each basin area at the river flow–monitoring sites are available in their results. The analyzed five fiscal years were 1990, 1995, 2000, 2005, and 2010. Information on the amount of pesticide used and the application method at the time was used for each of these years.

3. Analysis of temporal and regional variability of the PEC

PEC_{Tier2} values were calculated based on the environmental model that is used in Japan's pesticide registration system.[2] The PEC_{Tier2} takes into account the behavior of the pesticide in the environment and a more realistic estimate than a simple calculation of the PEC_{Tier1}.[2] The flowchart of the calculation method is shown in Fig. 1. Among the scenario parameters used in the PEC calculation (Table 1), the river flow, paddy rice cropped area, and pesticide usage ratio were determined based on temporal and regional characteristics; other parameters were assigned constant values based on a standard scenario because of the low sensitivity of PEC_{Tier2} to them.[5] The static period, when paddy water is regulated to prevent outflow, was set as 7 days for 2010 and 3 days for 2005, 2000, 1995, and 1990. Because the paddy water static period after application was changed from 3–4 days to 7 days in 2006,[18] the toxicity study period (PEC calculation period) was fixed as 3 days for all PEC calculations.

To calculate the PEC_{Tier2} of each pesticide, we determined the typical application method and typical single dose (g a.i./10 a) in a paddy field based on the following information: the PEC_{Tier2} calculation scenario noted in each pesticide registration criteria report, the application method used for each paddy-field lysimeter test, and the application method of formulations shipped in large volumes (Supplemental Data Table S1).

River-flow data were organized as the average of median values (50th percentile value of the daily flow rate per year) for the 5 years from 2004 to 2008 at 350 river flow–monitoring sites, and the data were converted into the relative flow rate per 100 km^2 of basin area.[11]

The ratio of the paddy-field area in each basin at the river flow–monitoring sites was estimated by Iwasaki *et al.*[19] To estimate the actual paddy-rice cropped area during each of the five fiscal years, we calculated the calibration factors based on the ratio of the area of rice cropped to the paddy-field area in each fiscal year at the prefectural level.[11] The calculated calibration factor for each river flow–monitoring site was assigned the value of the prefecture in which the site was located. Finally, the area of rice paddy cropped per 100 km^2 of basin was calculated at each river flow–monitoring site using the estimated percentage of the paddy-field area and the calibration factor.

To estimate the pesticide usage ratio in the paddy-rice cropped area, we first organized the amount of the shipment of each formulation product as active ingredients in each prefecture based on the statistics of Noyaku Youran.[12] Next, because a single dose of a pesticide in a paddy field and the environmen-

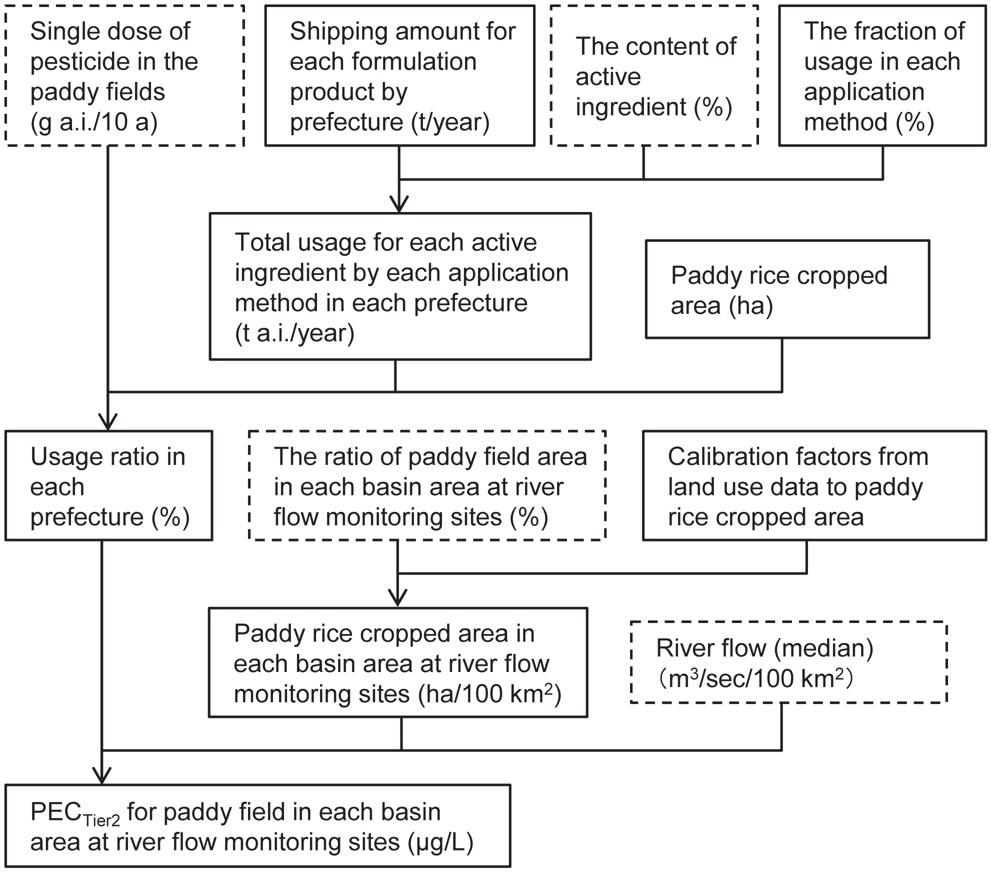

Fig. 1. Flowchart for estimating the PECs for paddy fields in each basin at river flow–monitoring sites. Boxes outlined with solid lines indicate values that have changed across years, and boxes outlined with dashed lines indicate values that remained constant from year to year.

tal dynamics after its use differed among application methods, it was necessary to separate the amount of pesticide used by application method as follows: nursery box application, ground application, unmanned helicopter application for paddy rice, and other methods in upland field and orchard. Thus, we used the method of Yachi *et al.*,[20] who calculated the usage for each active gradient by each application method in each prefecture.

Table 1. Input parameters for PEC_{Tier2} calculation in a standard scenario

Parameter	Standard scenario	Unit
Paddy rice cropped area	500	ha/100 km^2
River flow	3	m^3/sec/100 km^2
Pesticide usage ratio	10	%
Daily surface runoff	30	m^3/ha/day
Daily lateral seepage	20	m^3/ha/day
Ridge soil density	1	g/cm^3
Ratio of contiguous water and soil volume	2.4	—
Ridge soil organic carbon content	2.9	%
Tributary water volume	86,400	m^3/day
Tributary sediment volume	2000	m^3
Tributary sediment density	1	g/cm^3
Tributary sediment organic carbon content	1.2	%

This method required information of the application list for each of the five fiscal years[13–17] for each formulation product and assumed that the pesticide usage was evenly distributed among each target pest for the applicable crops. Next, the pesticide usage ratio for each application method (%) was calculated as the (total usage in each prefecture/single dose of pesticide in the paddy fields)/(paddy-rice cropped area in each prefecture). Here, the pesticide usage ratio was based on the amount of each formulation product shipped by prefecture; therefore, it could be calculated only on a prefectural basis, not on a basin basis. Thus, the pesticide usage ratio for each river flow–monitoring site was assigned the value of the prefecture in which the site was located.

To calculate the PEC_{Tier2}, the following data regarding environmental fate were needed: organic carbon–normalized soil partition coefficient (K_{oc}), half-life in water (DT_{50}), and the result of a paddy-field lysimeter test (Supplemental Data Table S1). These data were collected from the assessment report of the pesticide registration criteria in Japan,[1] Noyaku Syouroku,[21] a risk assessment report of the Food Safety Commission of Japan,[22] the Noyaku Handbook,[23] and the Pesticide Manual.[24] The methods used to organize multiple data and estimate missing data have been described in our previous paper.[11] The organized data are shown in Supplemental Data Table S1. Note that the concentration of spinosad is the sum of the concentrations of spinosyn A and spinosyn D. Likewise, the concentration of pyriminobac-methyl is the sum of the concentrations of (*E*)- and (*Z*)-isomers.

4. Cumulative ecological risk assessment using the SSD

The PEC values calculated for each pesticide in 1990, 1995, 2000, 2005, and 2010 at 350 river flow–monitoring sites were used to quantify the cumulative ecological risk of multiple pesticides using the NIAES-CERAP tool. Because differences in the timing of pesticide application were not considered, the assessment of the cumulative ecological risk from exposure to multiple pesticides assumed simultaneous exposure. In the case of multiple application methods for the same pesticide, such as ground application in paddy fields and nursery box application, the PECs for all application methods were summed and used for the cumulative ecological risk assessment.

There are two main reference models for describing the mixture toxicity: the concentration addition (CA) and independent action (IA) models.[25] The CA model is commonly applied as a conservative model for predicting the mixture toxicity of chemicals with the same mode of action (MoA) or chemicals for which the MoA is unknown. The IA model is commonly applied to chemicals with different MoAs. The application of these mixture toxicity models to the SSD to calculate the msPAF has been described in our previous study,[26] in which the method used to calculate the msPAF was experimentally validated.

The CA–IA mixing model was used for multiple pesticides, some with the same MoA and some with different MoAs. Figure 2 shows a conceptual diagram for calculating the cumulative ecological risk for assumed six pesticides, where pesticides A, B, and C are MoA-1; pesticides D and E are MoA-2; and pesticide F is MoA-3. First, the cumulative ecological risk ($PAF_{MoA\text{-}1}$) of pesticides A, B, and C is calculated using the CA model based on each SSD and PEC. Next, the $PAF_{MoA\text{-}2}$ of pesticides D and E is calculated using the CA model based on each SSD and PEC. Finally, the msPAF is calculated using the IA model from the $PAF_{MoA\text{-}1}$, the $PAF_{MoA\text{-}2}$, and the individual ecological risk of pesticide F ($PAF_{MoA\text{-}3}$). Thus, the CA–IA mixing model can be used to calculate the msPAF no matter how many pesticides there

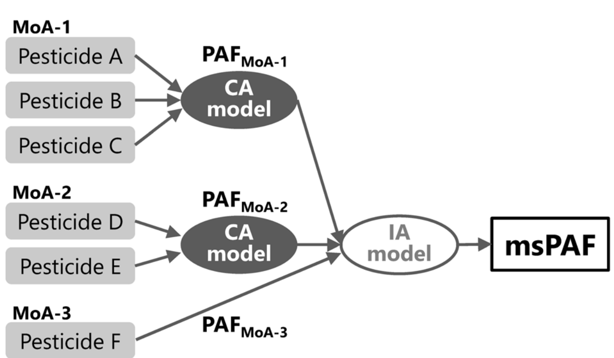

Fig. 2. Conceptual diagram of the calculation of cumulative ecological risk using the CA–IA mixing model.

are. Regarding the information of the MoA for each pesticide, the NIAES-CERAP tool is based on the Insecticide Resistance Action Committee (IRAC) classification of insecticides, the Fungicide Resistance Action Committee (FRAC) classification of fungicides, and the Herbicide Resistance Action Committee (HRAC) classification of herbicides. However, the classification of herbicides was based on the legacy HRAC code rather than the current one.

The NIAES-CERAP tool finally outputs four provisional categories of risk based on the msPAF values: >50%, high risk; 5–50%, middle risk; 0.1–5%, low risk; and <0.1%, not detectable. These categorizations are based on comparisons of PAF values and the actual effects on the aquatic community in mesocosm studies.[7,27] The regional variabilities of msPAFs were illustrated as a risk map using the above four categories and a violin plot, which is a box plot with the addition of the probability density of the data at different sites, smoothed by a kernel density estimator.

Results

Figure 3 shows risk maps indicating the regional variability of the cumulative ecological risk of insecticides (A) and herbicides (B) in 2010 at 350 river flow–monitoring sites in Japan. Ecological risk from fungicides was undetectable (<0.1%) at all sites, and therefore is not shown. Risk maps for all 5 fiscal years are shown in Supplemental Data Fig. S1. Sites with high risk (msPAF >50%), which were scattered across Japan in 1990, were not observed in 2010 for either insecticides or herbicides (Fig. 3). In 2010, 67 sites were classified as medium risk and 258 sites as low risk for insecticides, whereas these respective risk levels were observed at 243 and 90 sites for herbicides. Thus, the ecological risks of herbicides were greater than those of insecticides in 2010.

Figure 4 shows violin plots indicating the regional distribution of the msPAF and PAF of each MoA (Fig. 2) for insecticides in the 5 fiscal years. The msPAF values for insecticides tended to decrease with time and decreased dramatically from 2005 to 2010. The median values of the 350 sites were 23.6%, 19.2%, 15.3%, 7.3%, and 1.8% in 1990, 1995, 2000, 2005, and 2010, respectively. Although the msPAF was dominated by MoA-1A and -1B (organophosphorus and carbamate insecticides) from 1990 to 2005, it was dominated by MoA-4A (neonicotinoid insecticides) in 2010. The contribution of insecticides with other MoAs was small. The PAF values of MoA-1A, -1B, and -14 decreased with time; those of MoA-3A and -4A peaked in 2000 and then declined; and those of MoA-2B increased slightly from 2000 onward.

Figure 5 shows violin plots indicating the regional distribution of the msPAF and PAF of each MoA (Fig. 2) for herbicides in the 5 fiscal years. As with insecticides, the msPAF values for herbicides tended to decrease over time. The median values of the 350 sites were 16.2%, 12.5%, 11.3%, 12.3%, and 7.6% in 1990, 1995, 2000, 2005, and 2010, respectively. The msPAFs were dominated by MoA-C (mainly triazine herbicides), -B (mainly

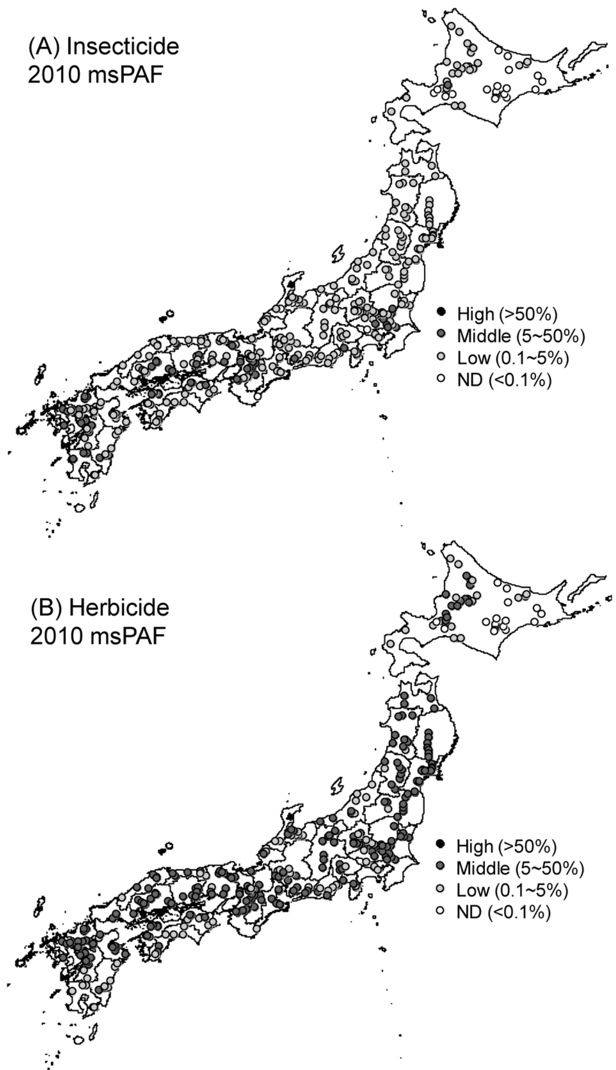

Fig. 3. Geographical distribution of msPAFs for mixtures of insecticides and herbicides at 350 river flow–monitoring sites in 2010.

sulfonylurea herbicides), -E (such as oxadiazole, triazolinone, and oxazolidinedione chemical families), and -K3 (mainly chloroacetamide herbicides) in 1990; by MoA-B and -K3 in 1995–2000; and by MoA-B, -E, and -K3 in 2005–2010. The contribution of herbicides with other MoAs was small. The PAF values of MoA-B showed small temporal variation, those of MoA-C decreased dramatically from 1990 to 1995, those of MoA-E disappeared in 1995 but then increased from 2000 onward, and those of MoA-K3 peaked in 1995 and declined thereafter.

Discussion

The cumulative ecological risks of insecticides decreased significantly between 1990 and 2010. The decrease was largely attributable to the decrease in the risks of insecticides with MoA-1A and -1B. This trend was due to the following three reasons: (1) decrease in the usage of organophosphorus insecticides in paddy-rice cultivation; (2) the alteration of insecticide application in rice paddy fields from surface-water application to nursery-box application; and (3) the fact that the use of several insecticides (*e.g.*, diazinon) was no longer allowed in paddy-rice cultivation after pesticide registration criteria based on ecological risk assessment were introduced in 2005. These results are consistent with our previous analysis,[7] which indicated that substituting nursery-box for surface-water application reduces the ecological risk of pesticides.

The risk of herbicides also decreased during the 20-year period, but not as much as the risk of insecticides. The ecological risk of herbicides decreased due to the following three reasons: (1) the decreased use of triazine herbicides (MoA-C); (2) the temporary expiration of the registration for oxadiazon; and (3) the change in the paddy water static period after application from 3–4 days to 7 days in 2006.[18] It has been shown that the pesticide runoff of the three herbicides decreased by 1/2–1/10 by extending the water static period from 4 to 7 days.[18] In 1990, the herbicide chlornitrofen (MoA-E) was still widely used in paddy-rice cultivation (1200 tons shipped), but it was not considered in this study due to a lack of data. Chlornitrofen is less likely to run off into rivers because of its low water solubility. However, the risk of MoA-E in 1990 may be underestimated, considering the large volume of chlornitrofen shipped that year.

The results of the present study showed that fungicides contributed little to the ecological risk. However, fungicides are generally toxic to aquatic fungi and fungus-like organisms, and toxicity to such organisms has been neglected when assessing the ecological effects of fungicides. The guidance document for assessing the risk of plant-protection products toward aquatic organisms in the European Union[28] suggests that further research into potential effects on fungi is needed and that the selection of relevant species for which standardized ecotoxicity tests may be developed should be identified as a research need. Recently, we developed an efficient and ecologically relevant bioassay method using 5 species of aquatic fungi and fungus-like organisms.[29,30] This method would be useful for assessing the ecological effects of fungicides in the future.

In addition, uncertainty in our ecological risk assessment includes uncertainty in the PEC estimation at the 350 sites, which was discussed previously.[11] A comparison of the estimated PECs with the pesticide concentrations measured by monitoring studies showed good consistency between the values. However, 4 factors would have contributed to the estimation error: (1) regional characteristics were taken into account only for the river-flow rate, paddy rice cropped area, and pesticide usage ratio in each application method, but not for the runoff from rice-paddy fields and the soil quality (density and organic carbon content) in ridge soil and tributary sediment; (2) for the pesticide usage ratio in each application method, fixed values were used within each prefecture, but uneven usage is actually expected even within a prefecture; (3) in the calculation of the pesticide usage ratio in each application method, all shipped pesticide products were assumed to be used at one time, but in some cases, pesticides are applied on multiple occasions in the paddy field; and (4) the physical-chemical properties (soil adsorption and degra-

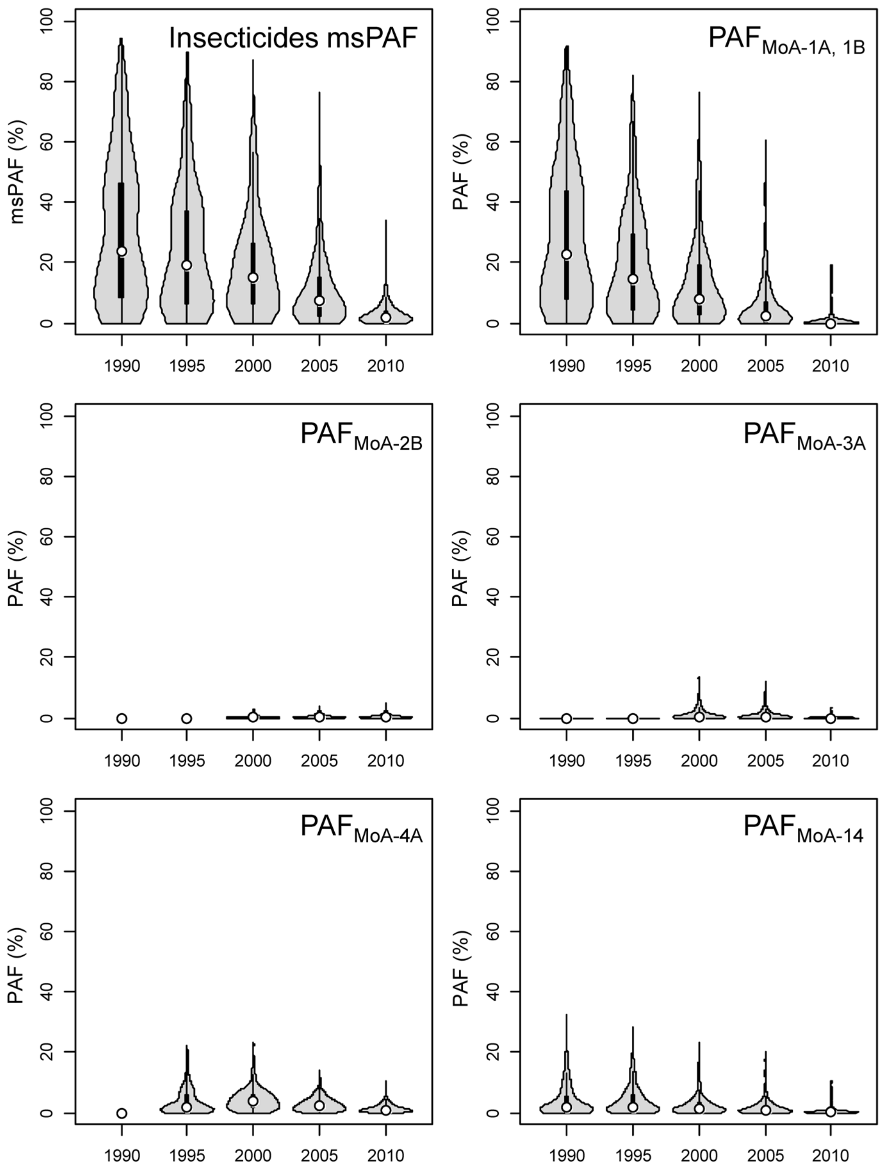

Fig. 4. Cumulative ecological risk of insecticides from 1990 to 2010 with all MoAs (msPAFs) and each MoA (PAF_{MoA}). The distributions of the 350 study sites are shown as violin plots (boxes, open circles, and whiskers mean the same as in a boxplot). The classification of modes of action is as follows: 1A, 1B: acetylcholinesterase inhibitors; 2B: GABA-gated chloride channel blockers; 3A: sodium channel modulators; 4A: nicotinic acetylcholine receptor competitive modulators; 14: nicotinic acetylcholine receptor channel blockers.

dation) used in the PEC estimation may differ from the behavior of pesticides in the actual environment. Further research is needed to improve the accuracy of the PEC estimation, as was discussed in the previous study.[11)]

The objective of the present study was to show the temporal variability of the cumulative ecological risk of multiple pesticide during the shift to "small amounts, large variety" of pesticides. Consequently, from 1990 to 2010, the median values of the ecological risks of pesticides decreased by 92.4% and 53.1% for insecticides and herbicides, respectively. The substantial reduction in ecological risk was the result of the development of low-risk pesticides by manufacturers, the efforts of farm producers to reduce pesticide usage and prevent pesticide runoff into rivers, and tighter regulation (introducing new registration criteria) by the government. Organic farming accounted for only 0.4% of Japan's total arable land in 2010.[31)] Therefore, organic farming contributed little to the reduction of the ecological risk of pesticides. In 2021, Japan's Ministry of Agriculture, Forestry and Fisheries announced "Measures for achievement of Decarbonization and Resilience with Innovation (MeaDRI)" and set a goal of 50% reduction in the risk-weighted use of chemical pesticides by 2050.[32)] To monitor the progress in achieving this goal, it is necessary to conduct a cumulative ecological risk assessment, as in the present study. A quantitative ecological risk assessment is a useful tool for evaluating the efficiency of the various measures of risk management.

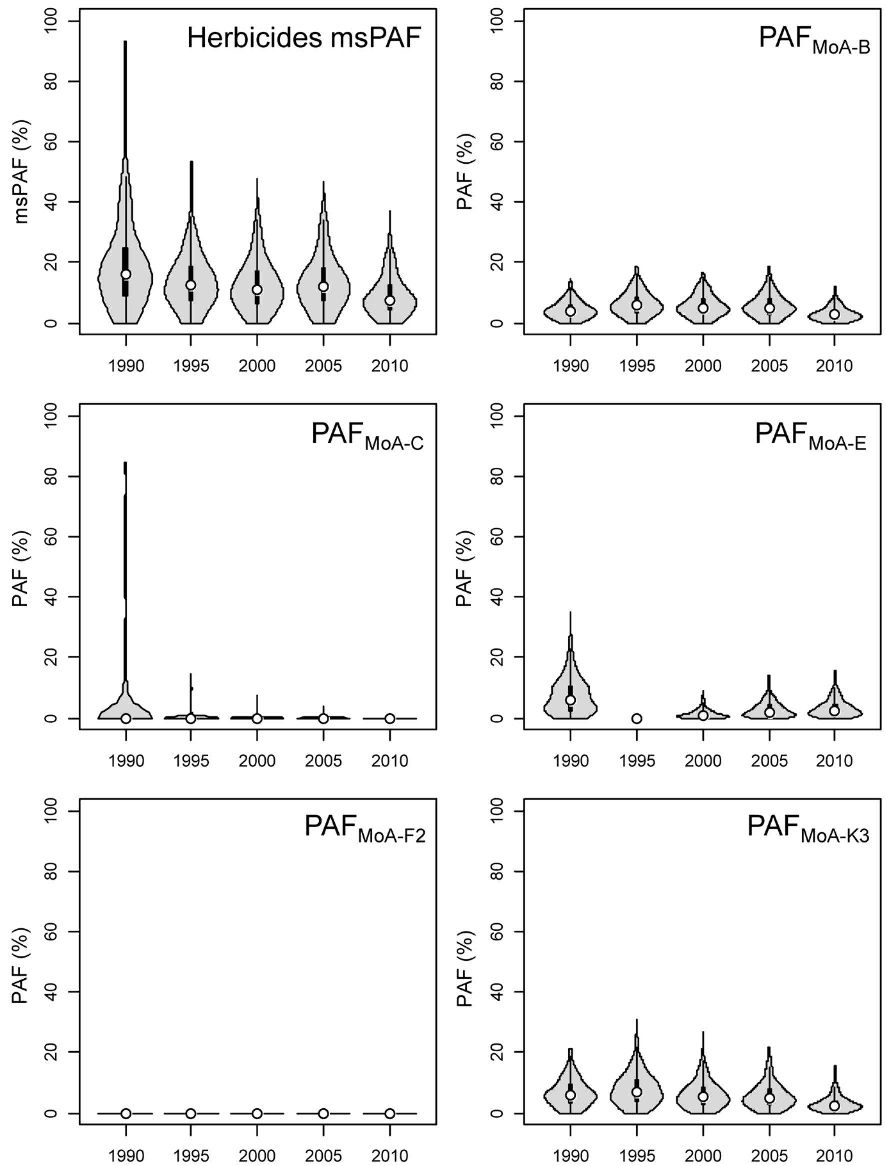

Fig. 5. Cumulative ecological risk of herbicides from 1990 to 2010 with all MoAs (msPAFs) and each MoA (PAF_{MoA}), as in Fig. 4. The classification of modes of action is as follows: B: inhibitors of acetolactate synthase; C: inhibitors of photosynthesis by photosystem II; E: inhibitors of protoporphyrinogen oxidase; F2: inhibitors of 4-hydroxyphenyl-pyruvate-dioxygenase; K3: inhibitors of very-long-chain fatty acid synthesis.

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Electronic supplementary materials

The online version of this article contains supplementary materials (Supplemental Tables S1 and Fig. S1), which is available at http://www.jstage.jst.go.jp/browse/jpestics/.

References

1) MOE: "Assessment report of the pesticide registration criteria concerning toxicity to aquatic organisms," Japan's Ministry of Environment, Tokyo (in Japanese). http://www.env.go.jp/water/sui-kaitei/kijun.html (Accessed December, 2021)
2) MAFF: Other data and information to be attached to the application for registration of an agricultural chemical. Japan's Ministry of Agriculture, Forestry and Fisheries, Tokyo. "http://www.acis.famic.go.jp/eng/shinsei/3987_e.pdf" (Accessed December, 2021)
3) T. Iwafune, K. Inao, T. Horio, N. Iwasaki, A. Yokoyama and T. Nagai: Behaviour of paddy pesticides and major metabolites in the Sakura River, Ibaraki, Japan. *J. Pestic. Sci.* **35**, 114–123 (2010).
4) L. Posthuma, G. W. Suter and T. P. Traas (eds.): "Species Sensitivity Distributions in Ecotoxicology (Environmental and Ecological Risk

Assessment)," Lewis Publisher, Boca Raton, USA, 2001.
5) T. Nagai, K. Inao and T. Horio: Probabilistic ecological risk assessment of paddy herbicide in Japanese river waters using uncertainty analysis: A case study for simetryn. *J. Pestic. Sci.* **33**, 393–402 (2008) (in Japanese).
6) T. Nagai, K. Inao, A. Yokoyama, T. Iwafune and T. Horio: Probabilistic ecological risk assessment of eleven paddy herbicides. *Jpn. J. Risk Anal.* **20**, 279–291 (2010) (in Japanese).
7) T. Nagai and A. Yokoyama: Comparison of ecological risks of insecticides for nursery-box application using species sensitivity distribution. *J. Pestic. Sci.* **37**, 233–239 (2012).
8) D. de Zwart: Ecological effects of pesticide use in the Netherlands: modeled and observed effects in the field ditch. *Integr. Environ. Assess. Manag.* **1**, 123–134 (2005).
9) NIAES: "An assessment tool of cumulative ecological risk of multiple pesticides NIAES-CERAP," Institute for Agro-Environmental Sciences, NARO, 2018 (in Japanese). https://www.naro.go.jp/publicity_report/publication/laboratory/niaes/manual/079666.html (Accessed April, 2021)
10) T. Nagai: Ecological effect assessment by species sensitivity distribution for 68 pesticides used in Japanese paddy fields. *J. Pestic. Sci.* **41**, 6–14 (2016).
11) S. Yachi, K. Inao and T. Nagai: Analysis of region-specific predicted environmental concentration of paddy pesticides at 350 river flow monitoring sites. *J. Pestic. Sci.* **42**, 1–9 (2017) (in Japanese).
12) Japan Plant Protection Association (ed.): "Noyaku Youran," Japan Plant Protection Association, Tokyo (in Japanese).
13) JA Zen-nou (ed.): "Kumiai Noyaku Souran 2011," Japan Rural Education Association, Tokyo, 2011 (in Japanese).
14) S. Yoneyama, K. Ando and T. Tsuzuki (eds.): "Noyaku Binran ver 10," Rural and Mountainous Village Culture Association, Tokyo, 2004 (in Japanese).
15) S. Yoneyama (ed.): "Noyaku Binran ver 9," Rural and Mountainous Village Culture Association, Tokyo, 2002 (in Japanese).
16) S. Katsuki, Y. Sugayama and M. Goto (eds.): "Noyaku Binran ver 8," Rural and Mountainous Village Culture Association, Tokyo, 1995 (in Japanese).
17) S. Katsuki (ed.): "Noyaku Binran ver 7," Rural and Mountainous Village Culture Association, Tokyo, 1989 (in Japanese).
18) JAPR: "Report on the evaluation project for the pesticide runoff prevention technology," Japan Association for Advancement of Phyto-Regulators, Tokyo, 2009 (in Japanese).
19) N. Iwasaki, K. Inao and T. Nagai: Estimation of paddy rice planted area in an upstream basin—Analysis based on National Land Numerical Information and Statistics of Agriculture—. *Theory Application GIS* **24**, 31–38 (2016) (in Japanese).
20) S. Yachi, K. Inao and T. Nagai: Development of a simple method for estimating the usage of paddy insecticides with each application method. *J. Pestic. Sci.* **41**, 1–10 (2016) (in Japanese).
21) FAMIC: "Noyaku Syouroku," Food and Agricultural Materials Inspection Center, Tokyo (in Japanese). http://www.acis.famic.go.jp/syouroku/index.htm (Accessed April, 2021)
22) FSC: "Risk assessment report of pesticide," Food Safety Commission of Japan, Tokyo (in Japanese). https://www.fsc.go.jp/fsciis/evaluationDocument/list?itemCategory=001 (Accessed April, 2021)
23) Japan Plant Protection Association (ed.): "Noyaku-handbook 2016," Japan Plant Protection Association, Tokyo, 2016 (in Japanese).
24) C. Tomlin (ed.): "Pesticide Manual 17th edition," BCPC Publications, 2015.
25) Z. De Zwart and L. Posthuma: Complex mixture toxicity for single species and multiple species: Proposed methodologies. *Environ. Toxicol. Chem.* **24**, 2665–2676 (2005).
26) T. Nagai: Predicting herbicide mixture effects on multiple algal species using mixture toxicity models. *Environ. Toxicol. Chem.* **36**, 2624–2630 (2017).
27) NIAES: "Technical guidance document on species sensitivity distribution analysis for pesticide ecological risk assessment," National Institute for Agro-Environmental Sciences, 2016 (in Japanese). https://www.naro.affrc.go.jp/archive/niaes/techdoc/ssd/ (Accessed April, 2021)
28) EFSA Panel on Plant Protection Products and their Residues: Guidance on tiered risk assessment for plant protection products for aquatic organisms in edge-of-field surface waters. *EFSA J.* **11**, 3290 (2013).
29) T. Nagai: A novel, efficient, and ecologically relevant bioassay method using aquatic fungi and fungus-like organisms for fungicide ecological effect assessment. *Environ. Toxicol. Chem.* **37**, 1980–1989 (2018).
30) T. Nagai: Sensitivity differences among five species of aquatic fungi and fungus-like organisms for seven fungicides with various modes of action. *J. Pestic. Sci.* **45**, 223–229 (2020).
31) MAFF: Circumstances surrounding organic farming. Japan's Ministry of Agriculture, Forestry and Fisheries, Tokyo, 2020 (in Japanese). "https://www.maff.go.jp/j/seisan/kankyo/yuuki/attach/pdf/meguji-full.pdf" (Accessed July, 2021)
32) MAFF: Strategy for sustainable food systems, MeaDRI. Japan's Ministry of Agriculture, Forestry and Fisheries, Tokyo, 2021. "https://www.maff.go.jp/e/policies/env/env_policy/meadri.html" (Accessed December, 2021)