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Comparative ecotoxicity of imidacloprid and dinotefuran to aquatic insects in rice mesocosms

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ABSTRACT

There are growing concerns about the impacts of neonicotinoid insecticides on ecosystems worldwide, and yet ecotoxicity of many of these chemicals at community or ecosystem levels have not been evaluated under realistic conditions. In this study, effects of two neonicotinoid insecticides, imidacloprid and dinotefuran, on aquatic insect assemblages were evaluated in experimental rice mesocosms. During the 5-month period of the rice-growing season, residual concentrations of imidacloprid were 5–10 times higher than those of dinotefuran in both soil and water. Imidacloprid treatment (10 kg/ha) reduced significantly the populations of*, Crocothemis servilia mariannae* and *Lyriothemis pachygastra* nymphs, whereas those of *Orthetrum albistylum speciosum* increased slightly throughout the experimental period. However, *Notonecta triguttata*, which numbers were high from the start, later declined, indicating possible delayed chronic toxicity, while *Guignotus japonicus* disappeared. In contrast, dinotefuran (10 kg/ha) did not decrease the populations of any species, but rather increased the abundance of some insects, particularly Chironominae spp. larvae and *C. servilia mariannae* nymphs, with the latter being 1.7x higher than those of controls. This was an indirect effect resulting from increased prey (e.g., chironomid larvae) and lack of competition with other dragonfly species. The susceptibilities of dragonfly nymphs to neonicotinoids, particularly imidacloprid, were consistent with those reported elsewhere. In general, imidacloprid had higher impacts on aquatic insects compared to dinotefuran.

1. Introduction

Paddy ecosystems are composed not only of rice cultivation fields but also other landscape features such as irrigation ponds and canals, which have played an important role in maintaining the biodiversity and providing ecosystem services (Schoenly et al., 1998; Natuhara, 2013). However, the biodiversity of paddies has been threatened by a variety of anthropogenic disturbances including farmland consolidation for higher agricultural productivity (Dugan, 1993; Watanabe et al., 2013) and pesticides (Katayama et al., 2015). Currently, many researchers in Japan and other countries are concerned about the ecological risks posed by pesticides. Pesticides can spread rapidly into the landscapes surrounding agricultural fields through aerial drift and water runoff, and thus cause adverse effects on biotic communities either directly or indirectly (Relyea and Hoverman, 2006; Phong et al., 2009; Hayasaka et al., 2012c; Hayasaka, 2014). On the other hand, the use of pesticides is often necessary for controlling pests and reduce labor to the farmers.

Neonicotinoids are a novel class of insecticides, which had been developed since the early 1990 s to replace older chemicals (Anderson et al., 2014) such as organochlorine, organophosphate, and carbamate insecticides, which are more harmful to humans (Gray and Hammitt, 2000; Anderson et al., 2014; Damalas and Koutroubas, 2016). Neonicotinoids are systemic chemicals, meaning that they are highly soluble and thus can be absorbed by the plant – the entire plant becomes toxic to the target pest and other organisms (Sánchez-Bayo, 2014). These chemicals mimic the acetylcholine neurotransmitter and are highly neurotoxic to insects and other arthropods. Neonicotinoids have become one of the most widely used classes of insecticides with a global market share of more than 25% (Jeschke et al., 2011; Simon-Delso et al., 2015), mainly because of their safety to vertebrates and efficacy in controlling sucking pests.

Their high toxicity to insects is a cause for concern. For example, honeybees in France were weakening or declining in numbers soon after the neonicotinoid imidacloprid was introduced in 1994 (van der Sluijs et al., 2013). Other pollinators such as wild bees, butterflies,

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moths, and hoverflies are equally exposed; where neonicotinoids are used, 11–24% of pollen and 17–65% of nectar is contaminated with these insecticides (van der Sluijs et al., 2013; Sánchez-Bayo and Goka, 2014). In addition to the risks on terrestrial ecosystems, adverse effects of neonicotinoids on aquatic agro-ecosystems have become apparent with increasing use of these insecticides (Morrissey et al., 2015; Yokoyama et al., 2015). One of the reasons is that surface runoff and leaching potential of neonicotinoids into aquatic environments are quite high (Huseth and Groves, 2014; Bonmatin et al., 2015). Thus, the potential impacts of these insecticides on non-target aquatic organisms need to be re-evaluated by pesticide regulating agencies in developed and developing countries alike (Smit et al., 2015).

Also, it is considered that neonicotinoids have, in general, relatively low soil adsorption compared to other classes of systemic insecticides such as fipronil and chlorantraniliprole (JPPA, 2011), but recent studies suggest that most neonicotinoids applied to crops as seed-dressings remain in the soil for more than a year (Goulson, 2013; Jones et al., 2014), while those applied to nursery-boxes persisted in soil at μg/kg (ppb) levels for a year or more (Hayasaka et al., 2012c). Furthermore, realistic predictions of ecological impacts of neonicotinoids to the biodiversity of the agricultural landscapes should consider not only aquatic organisms but also benthic species, which feed on detritus and provide an essential ecosystem service (Kreutzweiser et al., 2009; Peijnenburg et al., 2012). And yet, our knowledge about the ecotoxicological impacts of most neonicotinoids on aquatic communities, including benthic organisms, is still very limited (Stoughton et al., 2008; Pestana et al., 2009; Sánchez-Bayo et al., 2016).

As a first step in evaluating the ecological risks of pesticides, laboratory acute and chronic toxicity tests based on the OECD test guidelines are indispensable (Hayasaka et al., 2012b, 2013c); however, these tests do not take into consideration environmental uncertainties derived from the ecosystem complexity and other variables, e.g., water flow, weather and degradation conditions, so they are insufficient for evaluating the ecological risks and/or safety of pesticides (Smetanová et al., 2014). A better understanding of the impacts of pesticides is obtained by testing the chemicals in mesocosms (Szöcs et al., 2015). Nonetheless, ecotoxicological studies of neonicotinoids on biocenosis of paddies are few (Sánchez-Bayo and Goka, 2006; Hayasaka et al., 2012c; Daam et al., 2013) compared to those conducted with other pesticides in lotic and lentic systems (e.g. Relyea and Hoverman, 2006; Beketov et al., 2008; Rico and Van den Brink, 2015).

Here, we monitored the effects of the two most commonly applied neonicotinoids in Japan, imidacloprid and dinotefuran, on aquatic insect communities of an experimental rice mesocosms. The community effects of the former have been studied previously (Sánchez-Bayo and Goka, 2006; Hayasaka et al., 2012a), whereas the ecological impacts of the latter compound are currently unknown. A comparison of the impacts of these two compounds are thus warranted.

2. Materials and methods

2.1. Target insecticides

The systemic neonicotinoids imidacloprid and dinotefuran were registered in Japan in 1992 and 2002, respectively. Both imidacloprid and dinotefuran target nicotinic acetylcholine receptor (nAChR) located in central nervous system in both invertebrate and vertebrates, but have higher toxicity to the former taxa due to the specific receptor subunits found in arthropods (Tomizawa and Casida, 2003). Dinotefuran is a furanicotinyl insecticide, third generation neonicotinoid, which have a characteristic non-aromatic (±)-tetrahydro-3-furylmethyl moiety instead of the aromatic chloronicotinyl or chlorothiazole moiety of other neonicotinoids (Wakita et al., 2003). Their physicochemical and some acute and chronic toxicity data are given in Table 1. Note that dinotefuran is 65 times more water soluble than imidacloprid, and both neonicotinoid insecticides decompose quickly in aquatic environments by photolysis. Soil adsorption of dinotefuran is 5–10 times lower than that of imidacloprid. Consequently, dinotefuran is relatively more prone to leaching and more likely to be found in aquatic ecosystems than imidacloprid. Based on the acute and chronic toxicity data, neither imidacloprid nor dinotefuran are toxic to the zooplankton crustacean *Daphnia magna*, but imidacloprid is quite toxic to most other crustaceans (e.g., *Hyalella azteca*) and very toxic to chironomids and aquatic insects (Stoughton et al., 2008; Morrissey et al., 2015; Cavallaro et al., 2016). Also, acute and chronic toxicity values of imidacloprid to many aquatic invertebrates were similar (Table 1). Data on toxicity of dinotefuran to other invertebrate taxa are lacking, and like other neonicotinoids, these insecticides are not toxic to fish. In Japan, these two insecticides, particularly dinotefuran, have widespread use compared to other neonicotinoids, according to the information of WebKis-Plus of National Institute for Environmental Studies (NIES), Japan (http://w-chemdb.nies.go.jp/).

Table 1
Physicochemical properties and acute and chronic toxicities of imidacloprid and dinotefuran to relevant species.

Properties	Imidacloprid	Dinotefuran
Water solubility at 20 °C (mg/L)	610[e]	39,830[e]
Octanol: water partition coefficient at 20 °C (logP)	0.57[e]	−0.549[e]
Hydrolysis half-life at 25 °C (days) (pH9)	355[e]	> 365[e]
Aqueous photolysis half-life at 25 °C (days)	30[e]	0.2[e]
Half-life in soil (days)	174[e]	75[e]
Sorption to soil (Koc)	132–310[c]	26[e]
Acute and chronic toxicities		
Crustaceans		
Cypridopsis vidua (48 h EC_{50}: μg/L)	3.0[b]	-
Daphnia magna (48 h EC_{50}: μg/L)	6,029–96,650[b]	110,600–968,300[b]
(21 d LOEC: μg/L)	3,600[b]	95,300[b]
Hyalella azteca (96 h LC_{50}: μg/L)	65.3[g]	-
(28 d LOEC: μg/L)	11.46[g]	-
Aquatic insects		
Aedes aegypti (48 h LC_{50}: μg/L)	44–360[b]	131[b]
Chironomus dilutes (14 d LC_{50}: μg/L)	1.52[a]	-
(40 d EC_{50}: μg/L)	0.39[a]	-
Chironomus tentans (96 h LC_{50}: μg/L)	2.65–5.75[b,g]	-
(28 d LOEC: μg/L)	1.14–3.46[b,g]	-
Cheumatopsyche brevilineata (48 h EC_{50}: μg/L)	4.22–5.24[b]	10.4[b]
Cloeon dipterum (96 h LC_{50}: μg/L)	1.02–37[b,f,h]	-
(28 d EC_{50}: μg/L)	0.13–0.85[b,f,h]	-
Fish		
Lepomis macrochirus (bluegill) (96 h LC_{50}: μg/L)	> 105,000[d]	99,300–100,500[b,d]
(47 d NOEC: μg/L)	9,000[b]	-
Oncorhynchus mykiss (rainbow trout) (96 h LC_{50}: μg/L)	83,000–229,100[b]	> 99,500[b,e]
(98 d LOEC: μg/L)	1,200[b]	-

[a] Cavallaro et al. (2016).
[b] ECOTOX database. (http://cfpub.epa.gov/ecotox/)
[c] Hayasaka et al. (2012c).
[d] Japan Plant Protection Association (JPPA) (2011).
[e] Pesticide Properties DataBase. (http://sitem.herts.ac.uk/aeru/iupac/index.htm)
[f] Roessink et al. (2013).
[g] Stoughton et al. (2008).
[h] Van den Brink et al. (2016).

2.2. Experimental design

The main aim of this study was to compare the ecological effects of dinotefuran with those of imidacloprid on aquatic insects of rice mesocosms. We followed a similar experimental design and monitoring