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Effects of Pesticide Use in Rice Fields on Birds

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Abstract.—Waterbird use of agricultural wetlands has increased as natural wetlands have declined. Use of rice (*Oryza sativa*) habitats by some waterbird species is considered essential to sustaining populations. Although use of rice habitats by waterbirds has been documented throughout the world, little information is available on potential risks as a result of chemicals used in rice cultivation. The current review summarizes understanding of the use and consequences to birds of pesticide applications in rice habitats. Historically, organochlorine pesticides known to be applied for pest management in rice cultivation included dichlorodiphenyltrichloroethane (DDT), aldrin, dieldrin, endrin, heptachlor, technical hexachlorocyclohexane (HCH), toxaphene, endosulfan and sodium pentachlorophenate. Endosulfan and purified HCH (the gamma isomer lindane) are still in use. Cholinesterase-inhibiting insecticides currently used in rice include carbofuran, monocrotophos, phorate, diazinon, fenthion, phosphamidon, methyl parathion and azinphos-methyl—many products known to cause acute poisoning in birds. In addition, herbicides, fungicides, molluscicides and other pesticide types are used in rice cultivation. Some of the chemicals are highly toxic to birds and associated with mortality; several have the potential of causing adverse reproductive effects. Because of the large area under rice cultivation worldwide, the volume of pesticides applied to rice fields is significant. Innovations within the past few decades in rice production have increased pesticide use resulting in biodiversity losses in production areas and pollution of water resources. Management practices that address adverse effects of pesticide use in rice fields include increased adoption of Integrated Pest Management principles and less toxic products. *Received 5 March 2009, accepted 31 July 2009.*

Key words.—birds, conservation, effects, exposure, management, pesticides, review, rice, toxicity.

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Waterbird use of agricultural wetlands has increased as natural wetlands continue to decline worldwide. The use of rice (*Oryza sativa*) habitats by some waterbird groups (e.g. long-legged wading birds) is considered to be essential to sustaining populations (Czech and Parsons 2002). Although a large literature has begun to develop that documents the use of rice habitats by waterbirds throughout the world (see this volume), relatively little information is available concerning the potential risks faced by birds as a result of chemicals used in rice cultivation. Existing studies on exposure and effects of agrochemicals on birds have not been compiled and interpreted, although limited studies on other non-target organisms have been reviewed (Abdullah *et al.* 1997). Here, we review and summarize current understanding of the use and consequences to birds of pesticide applications in rice habitats. In addition, documented best management practices are reported, as well as identified information gaps.

METHODS

The objective of the literature review was to characterize acquired knowledge pertaining to exposure and effects of pesticides used in rice cultivation worldwide to birds (primarily aquatic species). We examined review papers and the primary literature pertaining to pesticide exposure and direct/indirect effects. Literature searches were conducted through ISI Web of Science® and web-based search engines. Studies were limited from the 1980s to the present, although for some outcomes, older studies were included for completeness. In addition, based on published data on application rates and compound toxicity, predictions were developed describing likely acute effects of insecticides to birds. Finally, information useful to the development and implementation of management practices protective of birds and prey populations as well as identified research needs were also compiled.

RESULTS

Approximately 65 peer-reviewed publications were found to address specifically the relationship between pesticide use in rice cultivation and impacts to aquatic birds. These were evenly divided between those that dealt primarily with exposure (52%) and effects (48%). A greater proportion (64%) provided information on management and cultivation practices likely to reduce adverse effects. Relatively few papers (20%) identified specific information needs that could be addressed through further research.

Exposure

Use of Pesticides in Rice Cultivation. Pesticide use in some rice-producing countries is significant. For example, 70% of herbicide used during the 1980s in Japan was applied to rice (Ishibashi *et al.* 1983). Heavy pesticide use countries in Asia include Korea, Malaysia, the Philippines (Cagauan 1995) and China (Wood *et al.* 2010). High yield rice varieties grown in major rice production regions of the world necessitate the heavy use of pesticides which, in turn, reduces the potential for crop rotation—specifically, rice-fish cultivation (Cagauan 1995).

Recently, herbicides registered for use on rice in Spain include bentazone, molinate, 2-methyl-4-chlorophenoxyacetic acid (MCPA), and propanil (Terrado *et al.* 2007). Clomazone is a relatively recent addition to the suite of herbicides used to control rice weeds, specifically targeting Barnyard Grass (*Echinochloa crus-galli*) (Lee *et al.* 2004). Clomazone has been registered for use on rice in Australia and South America historically, but only recently in the USA (Quayle *et al.* 2006). Another relatively new addition to the pesticide arsenal in Australia (also registered for use in Japan) is the herbicide benzofenap which replaces bensulfuron methyl (Quayle *et al.* 2007). Additional grass and broadleaf weeds are controlled in rice fields with sulfonylurea herbicides (thifensulfuron-methyl, rimsulfuron, thifensulfuron, prosulfuron, sulfometuron-methyl, cinosulfuron) (review in Ferrero *et al.* 2001). Herbicide use increased in recent decades in the Philippines; MCPA, 2,4-D amine, 2,4-D plus piperophos, butachlor, trifluralin, bifenox and EPTC (a selective thiocarbamate herbicide) are compounds used (Cagauan 1995). Rice herbicides used in rice cultivation in Brazil include clomazone, propanil and quinclorac (Marchesan *et al.* 2007).

Insecticide use is similarly varied throughout rice growing countries. Carbofuran was used in rice fields in West Africa (Mullie *et al.* 1991). Dimethoate and parathion are used in rice cultivation in southern Europe (Jorgensen *et al.* 1997). Imadocloprid is used in China to control the planthoppers *Sogatella furcifera* and *Nilaparvata lugens* (Yu *et al.* 2007). Rice farmers in Vietnam use pesticides from a list of 64 compounds (50% insecticides including fenobucarb, cartap and lambacyhalothrin; 25% fungicides including validamycin, propiconazole and hexaconazol; 25% herbicides including fenoxaprop-P-ethyl, 2,4-D, pretilachlor and fenclorim) (Berg 2001).

Based on accounts from research papers, the primary rice pesticides used in the Philippines were dominated by insecticides (92% of all pesticide sprays) including endosulfan, methyl parathion, cypermethrin, monocrotophos and chlorpyrifos (Heong *et al.* 1995) as well as endrin, azinphos methyl, carbosulfan, carbofuran, carbaryl, methomyl, isoprocarb (MIPC) and permethrin (Cagauan 1995). In addition, 4.1% of pesticides used were fungicides and 3.9% were herbicides (Heong *et al.* 1995). A more recent study established that the primary rice pesticides used in Mexico include carbofuran, chlorpyrifos, glyphosate and lesser quantities of parathion, malathion, methomyl, benomyl, propanil and 2,4-D amine (Osten *et al.* 2005). Based on interviews with farmers, monocrotophos was the most commonly used insecticide in commercial rice plantations in Bolivia, followed by methamidophos (R. Renfrew, unpublished data). Frequency of monocrotophos application ranged from twice per growing season to every two weeks, and application rates were typically reported as 600 g of active ingredient per ha (ai/ha; R. Renfrew, unpublished data).

A broad suite of compounds used in rice was identified in the peer-reviewed literature. Using that list as a starting point, we compiled a list of products thought to be registered for use in rice currently. Our sources (Table 1) included pesticide data compilations (e.g. Thomson 2001; Tomlin 2004) as well as industry catalogues of products (International Research Information Ltd. 2004), national guides of registered products (e.g. CASAFE 2003), compilations of maximum residue limits established through the United Nation's Food and Agriculture and World Health Organizations' (FAO/WHO) Joint Meetings on Pesticide Residues (JMPR), extension service publications (e.g. Punjab National Bank 2008) and websites of large pesticide manufacturing companies (e.g. Sanonda Zhengzhou Pesticide Co. Ltd. 2005). Names were standardized as per Tomlin (2004). A total of 274 active ingredients were identified as being currently registered for rice cultivation worldwide.

For insecticides (83 active ingredients identified), we further attempted to find application rates in rice in order to carry out a preliminary risk ranking exercise. Rates were obtained from sources such as those listed above, especially FAO/WHO JMPR evaluations which list application rates used in various countries following good agricultural practices (GAP data). Where it was not possible to obtain rice-specific application rates, or where we believed those rates not to be fully representative of rice-growing conditions worldwide, we used rates compiled for other field crops. Minimum and maximum rates for the various active ingredients are given in Table 2. Given the wide variation in reported application rates and the fact that several sources provide a spray concentration rather than an application rate per hectare, it is likely the actual variation in spray rates is higher than indicated. Some pesticides are applied as granular formulations; at least one (clothianidin) is a seed treatment only.

Transport and Fate of Pesticides Used in Rice. Significant research has been conducted to evaluate the transport of rice pesticides to groundwater and to adjacent surface water bodies. Lee *et al.* (1994) found the potential for groundwater contamination from labeled carbofuran to be very low. The majority of pesticide detections in groundwater below rice paddies in the Philippines were generally below protective levels (0.5 µg/l for total pesticides), although maxima exceeded these thresholds significantly (1.14-4.17 µg/l) (Bouman *et al.* 2002). Herbicides detected in rice irrigation water in southern Europe include propanil, molinate, MCPA, bentazone, 8-hydroxybentazone and thiobencarb. Water samples ranged from 1.9 to 55.9 µg/l; the propanil degradation product DCA was detected at 16.5 to 470 µg/l in water and 119 µg/kg in soil samples. Insecticides included temephos, fenitrothion, diazinon, carbendazim and carbofuran (review in Santos *et al.* 1998). Contamination of groundwater in southern Europe has been documented (>0.1 µg/l of single pesticides) and contamination is expected to be exacerbated under flooded conditions due to pesticide transport. Pesticides detected in soil water from rice fields in Greece include atrazine, lauramide DEA, alachlor, propanil, molinate, lindane, parathion, prometryne, metolachlor, carbofuran, terbufos, paraoxon-methyl (high concentration of 10 µg/l), lindane, beta-hexachlorocyclohexane (BHC) and deltamethrine. Lindane and carbofuran were detected most frequently but not generally at levels that exceeded safe groundwater thresholds; propanil was the most frequently detected herbicide (Papdopoulou-Mourkidou *et al.* 2004). The fate of benzofenap is largely in soil with little mobility in water (Quayle *et al.* 2007). Facile transport of rice pesticides through soil water and into the phreatic zone in soils of the Axios River basin of southern Greece point to potential groundwater contamination for the region although researchers found no evidence that deep groundwater was contaminated (Papadopoulou-Mourkidou *et al.* 2004).

In addition to evidence of groundwater contamination, rice cultivation is associated with surface water degradation. Pesticide profiles in nearby river systems were consistent with agrochemical use in rice fields; the Axios River served as a large drainage canal

Table 1. Pesticides (names as per Tomlin (2004)) thought to be registered currently for use in rice (excluding post-harvest uses). A few synonyms are given in the case of unusual or poorly known products.

Pesticide
Algicide
Nabam
Bactericide
oxolinic acid
Tecloftalam
validamycin A
Biological
Bacillus thurigiensis
Metarhysium anisopliae
Trichoderma viridae
Fumigant
sulfuryl fluoride
Fungicide
3-allyloxy-1,2-benzothiazole 1, 1-dioxide
Azoxystrobin
Benomyl
benzylaminobenzenesulfonate (BABS)
bismerthiazol (& copper salt)
blasticidin-S
bromuconazole
Captafol
Carbendazim
Carboxin
Carpropamid
chlorothalonil
copper sulfate
Cyazofamid
cyproconazole
diammonium ethylenebis (dithiocarbamate)
Diclocymet
Diclomezine
Difeconazole
Edifenfos
epoxiconazole
fenbuconazole
Fenoxanil
fentin hydroxide
Ferimzone
Fludioxonil
Flutolanil
Furametpyr
guazatine acetates
hexaconazole
Hymexazol
Imazalil
Ipconazole
Iprobenfos
Iprodione
isoprothiolane
kasugamycin hydrochloride hydrate
Mancozeb
Mepronil
methasulfocarb
Metiram
metominostrobin
Myclobutanil
Pefurazoate
Pencycuron
phenylmercury acetate
Phtalide
Polyoxorim
Prochloraz
propiconazole
Propineb
Pyroquilon
Quintozene
simeconazole
Sulphur
TCMTB
Tebuconazole
thiabendazole
Thifluzamide
thiophanate-methyl
Thiram
Tiadinil
Triadimenol
Tricyclazole
Trifloxystrobin
Herbicide
2,4-D
2,4-DB
acifluorfen-sodium
ACN
Anilofos
Azimsulfuron
Benfuresate
bensulfuron methyl
Bensulide
bentazon(e)
benzobicyclon
Benzofenap
Bifenox
bispyribac sodium
Bromobutide
Butachlor
Butamifos
Butralin
Cafenstrole
carfentrazone ethyl
Chlorpropham
Cinmethylin
Cinosulfuron
Clomazone
Clomeprop
Cumyluron
cyclosulfamuron
cyhalofop butyl
Cyromazine
Daimuron
Dicamba
dichlorprop-P
diclofop-methyl
Dimepiperate
dimethametryn
diquat dibromide
Endothal
Esprocarb
ethoxysulfuron
Etobenzanid
fenoxaprop-P
fenoxaprop-P-ethyl
Fentrazamide
flucetosulfuron
Fluchloralin
Flufenacet
Flumioxazin
fluoroglycofen-ethyl
flurenol-butyl
Glyphosate
halosulfuron-methyl
Haloxyfop
Imazapyr
imazosulfuron
Indanofan
Ioxinil
Linuron
MCPA
MCPA-thioethyl
MCPB
Mecoprop
Mefenacet
mefenpyr diethyl
Metamifop
Metosulam
Metsulfuron
metsulfuron methyl
Molinate
Naproanilide
Oryzalin
Oxadiargyl
Oxadiazon
oxaziclomefone
Oxyfluorfen
Paraquat
pendimethalin
Penoxsulam
Pentoxazone
Picloram
Piperophos
Pretilachlor
Profoxydim

Table 1. (Continued) Pesticides (names as per Tomlin (2004)) thought to be registered currently for use in rice (excluding post-harvest uses). A few synonyms are given in the case of unusual or poorly known products.

Pesticide
Propanil
Pyrazolinate
pyrazosulfuron-ethyl
Pyrazoxyfen
Pyribemzoxim
Pyributicarb
Pyridate
Pyriftalid
pyriminobac-methyl
Quinclorac
Quinoclamine
Sethoxydim
Simazine
Simetryn
sodium chlorate
TCA sodium
Thenylchlor
Thiazopyr
thiobencarb (bentiocarb)
Tiocarbazil
Triaziflam
Triclopyr
Insecticide
Acephate
Acetamiprid
alpha-cypermethrin
Azadirachtin
azinphos methyl
azinphos-ethyl
Benfuracarb
Bensultap
beta-cypermethrin
Buprofezin
Carbaryl
Carbofuran
Carbosulfan
cartap hydrochloride
chlorfenvinphos
Chlorpyrifos
chlorpyrifos-methyl
chromafenozide
Clothianidin
Cycloprothrin
Cyfluthrin
Cypermethrin
deltamethrin
diazinon
dichlorvos
dicrotophos
diflubenzuron
dimethoate
dimethylvinphos
dinotefuran
disulfoton
endosulfan
EPN
ethiprole
etofenprox
fenitrothion
fenobucarb (BPMC)
fenthion
fenvalerate
fipronil
furathiocarb
imidacloprid
isoprocarb
isopropyl O-(methoxyaminothiophosphoryl) salycilate (optunal, isocarbofos)
isoxathion
lambda-cyhalothrin
lindane (gamma BHC)
malathion
mecarbam
methamidophos
methomyl
methoprene
methoxyfenozide
metolcarb (MTMC)
monocrotophos
naled
nitenpyram
omethoate
parathion-methyl
permethrin
phenthoate
phorate
phosphamidon
pirimiphos methyl
propoxur
pymetrozine
pyridaphenthion (ofunack)
quinalphos
silafluofen (silatop)
tebufenozide
tetrachlorvinphos
thiacloprid
thiamethoxam
thiocyclam (hydrogen oxalate)
thiosultap
tralomethrin
triazophos
trichlorfon
vamidothion (Kilval)
XMC (Maqbal)
xylylcarb
zeta-cypermethrin
Molluscicide
metaldehyde
niclosamide
Pheromone
(Z)-hexadec-11-enal
Plant activator
acibenzolar-S-methyl (Bion)
probenazole
Plant growth regulator
6-benzylaminopurine
forchlorfenuron
giberellic acid
inabenfide
nitophenolate mixture
paclobutrazol
prohexadione-calcium
uniconazole
Rodenticide
flocoumafen
Safener
fenclorim

for rice cultivation with pesticides flowing back and forth between rice fields and river water. Uniform distribution of pesticides throughout the river basin was attributed to use in irrigated rice fields (Papdopoulou-Mourkidou *et al.* 2004, similar results in Miao *et al.* 2003a). Pesticide concentrations exceeding safe drinking water standards found in surface and groundwater associated with rice cultivation include molinate, dimepiperate, thiobencarb, thiocarbazyl, bentazone and oxadiazon; concentrations ranged from 0.1-30 μg/l (Miao *et al.* 2003b). Quayle *et al.* (2006) found the average maximum cloma-

Table 2. Estimated minimum and maximum application rates (g of active ingredient/ha) for insecticides in current use in rice fields. Multiple FAO/WHO JMPR reports were accessed (November 2008) through the web site: http://www.fao.org/agriculture/crops/core-themes/theme/pests/pm/lpe/en/ and are not individually listed in the Literature Cited (see the web site for full citation details).

Pesticide	Min. rate (g ai/ha)	Max. rate (g ai/ha)	Sources for application rates in field crops	Sources for rice-specific application rates
acephate	360	1020	Tomlin 2004	Sanonda Zhengzhou Pesticide Co. Ltd. 2005
acetamiprid	75	300	Tomlin 2004	
alpha-cypermethrin	10	20		APVMA 2008 (NRA 50651/1298)
azadirachtin	3.4	48		Lacy *et al.* 2008
azinphos methyl	400	1000		Cagauan 1995, FAO/WHO JMPR 1995
azinphos-ethyl	500	700	Thomson 2001	
benfuracarb	400	2500	Thomson 2001, Tomlin 2004	
bensultap	250	1500	Thomson 2001, Tomlin 2004	
beta-cypermethrin	125	250	Assume same rate as cypermethrin	
buprofezin	50	50	FAO/WHO JMPR 1999	IRRI 2004
carbaryl	528	2242	Tomlin 2004	Cagauan 1995, FAO/WHO JMPR 2002, IRRI 2004, Punjab National Bank 2008
carbofuran	600	2000		Mullie *et al.* 1991, Cagauan 1995, FAO/WHO JMPR 2000, CASAFA 2003, IRRI 2004, Punjab National Bank 2008
carbosulfan	400	1000		FAO/WHO JMPR 2003
cartap hydrochloride	250	1000	Tomlin 2004	Ishibashi *et al.* 1983, IRRI 2004
chlorfenvinphos	250	1000	Tomlin 2004	FAO/WHO JMPR 1972
chlorpyrifos	400	1000		FAO/WHO JMPR 2004, IRRI 2004, Sanonda Zhengzhou Pesticide Co. Ltd. 2005
chlorpyrifos-methyl	750	1500	FAO/WHO JMPR 1991	
chromafenozide	50	200	Thomson 2001, Tomlin 2004	
clothianidin				Used as seed treatment only
cycloprothrin	20	40	Tomlin 2004	
cyfluthrin	15	50	Tomlin 2004, FAO/WHO JMPR 2007	
cypermethrin	125	250		Cagauan 1995
deltamethrin	2.5	15	Tomlin 2004	FAO/WHO JMPR 2000, IRRI 2004
diazinon	300	1000	Tomlin 2004	FAO/WHO JMPR 1993, IRRI 2004
dichlorvos	300	750	Chemet Chemicals 2008	FAO/WHO JMPR 2003
dicrotophos	300	600	Thomson 2001	
diflubenzuron	62	280		FAO/WHO JMPR 2002
dimethoate	400	400	FAO/WHO JMPR 2003	IRRI 2004
dimethylvinphos	600	800		Tomlin 2004

Table 2. (Continued) Estimated minimum and maximum application rates (g of active ingredient/ha) for insecticides in current use in rice fields. Multiple FAO/WHO JMPR reports were accessed (November 2008) through the web site: http://www.fao.org/agriculture/crops/core-themes/theme/pests/pm/lpe/en/ and are not individually listed in the Literature Cited (see the web site for full citation details).

Pesticide	Min. rate (g ai/ha)	Max. rate (g ai/ha)	Sources for application rates in field crops	Sources for rice-specific application rates
dinotefuran	100	200	Thomson 2001	
disulfoton	280	3000	FAO/WHO JMPR 1998, Thomson 2001	
endosulfan	260	2000	Tomlin 2004	FAO/WHO JMPR 1993, Tomlin 2004, Punjab National Bank 2008
EPN	500	1000	Tomlin 2004	
ethiprole	670	670		Japan Food Safety Commission 2004
etofenprox	75	200	Thomson 2001	FAO/WHO JMPR 1993
fenitrothion	375	900	Picado and Ramirez 1998 (for crops including rice), Thomson 2001	FAO/WHO JMPR 2003
fenobucarb (BPMC)	500	1200	Tomlin 2004	Ishibashi *et al.* 1983, Cagauan 1995, IRRI 2004, Tomlin 2004
fenthion	300	1400	Tomlin 2004	FAO/WHO JMPR 1995
fenvalerate	56	224	Thomson 2001	
fipronil	12.5	150	Tomlin 2004	FAO/WHO JMPR 2001, IRRI 2004, Lacy *et al.* 2008, Punjab National Bank 2008
furathiocarb				
imidacloprid	30	200	Tomlin 2004	FAO/WHO JMPR 2002, Qiu *et al.* 2004
isocarbofos				
isoprocarb	250	750	Tomlin 2004	Ishibashi *et al.* 1983, Cagauan 1995, IRRI 2004
isoxathion	600	900	Tomlin 2004	Tomlin 2004
lambda-cyhalothrin	2	5	Tomlin 2004	
lindane (gamma BHC)	1000	2000		Punjab National Bank 2008
malathion	300	2000	Tomlin 2004	FAO/WHO JMPR 1998, IRRI 2004, Lacy *et al.* 2008
mecarbam	1120	2240	Thomson 2001	
methamidophos	150	2200		FAO/WHO JMPR 2003, Sanonda Zhengzhou Pesticide Co. Ltd. 2005
methomyl	200	600	Thomson 2001	Sanonda Zhengzhou Pesticide Co. Ltd. 2005
methoprene	56	112	Thomson 2001	
methoxyfenozide	200	200	Thomson 2001, FAO/WHO JMPR 2003, Tomlin 2004	FAO/WHO JMPR 2003
metolcarb (MTMC)	510	1200		Cagauan 1995, IRRI 2004
monocrotophos	250	2000		FAO/WHO JMPR 1994, Cagauan 1995, Settle *et al.* 1996
naled	225	840	Thomson 2001	
nitenpyram	15	400		Tomlin 2004

Table 2. (Continued) Estimated minimum and maximum application rates (g of active ingredient/ha) for insecticides in current use in rice fields. Multiple FAO/WHO JMPR reports were accessed (November 2008) through the web site: http://www.fao.org/agriculture/crops/core-themes/theme/pests/pm/lpe/en/ and are not individually listed in the Literature Cited (see the web site for full citation details).

Pesticide	Min. rate (g ai/ha)	Max. rate (g ai/ha)	Sources for application rates in field crops	Sources for rice-specific application rates
omethoate	900	1500	Thomson 2001, Tomlin 2004	Sanonda Zhengzhou Pesticide Co. Ltd. 2005
parathion-methyl	500	840		Cagauan 1995, FAO/WHO JMPR 2000
permethrin	250	500	Tomlin 2004	Cagauan 1995
phenthoate	510	1020	Tomlin 2004	Cagauan 1995
phorate	750	1000	Tomlin 2004	FAO/WHO JMPR 2005, Punjab National Bank 2008
phosphamidon	280	1680	Thomson 2001	
pirimiphos methyl	500	1000	Picado and Ramirez 1998	
propoxur	150	1120	FAO/WHO JMPR 1996, Thomson 2001	
pymetrozine	95	300	Thomson 2001, Tomlin 2004	
pyridaphenthion	60	1000	Thomson 2001	
quinalphos	190	1000	Thomson 2001, Tomlin 2004	
silafluofen	50	300	Tomlin 2004	Tomlin 2004
tebufenozide	67	336	Thomson 2001, Tomlin 2004	
tetrachlorvinphos	560	1680	Thomson 2001	
thiacloprid	100	360	Tomlin 2004	FAO/WHO JMPR 2000
thiamethoxam	10	200	Tomlin 2004	Punjab National Bank 2008
thiocyclam	375	375		Punjab National Bank 2008
thiosultap				
tralomethrin	7.5	20	Tomlin 2004	
triazophos	200	450	Tomlin 2004	Cagauan 1995, IRRI 2004, Tomlin 2004, Sanonda Zhengzhou Pesticide Co. Ltd. 2005
trichlorfon	300	3000	Tomlin 2004	Sanonda Zhengzhou Pesticide Co. Ltd. 2005, Lacy *et al.* 2008
vamidothion	150	500	FAO/WHO JMPR 1992	
XMC	600	600	Tomlin 2004	Thomson 2001 (also tea crops)
xylylcarb	40	40		Thomson 2001 (also tea and fruit crops)
zeta-cypermethrin	19	28	Tomlin 2004	Boyd 2005

zone concentration in water to be 202 µg/l (the half life of clomazone in rice water was 5 d); the maximum molinate concentration in water was 1009 µg/l. Surface water contamination from rice herbicide use was demonstrated in Brazil; rainy years had greater surface runoff and greater overall pollution (Marchesan *et al.* 2007). River waters were also contaminated with concentrations of clomazone > propanil > quinclorac. In addition to immediate transport, researchers have investigated the persistence of pesticides in associated water resources. Perera *et al.* (1999) detected propanil in rice field soil and water two weeks after application. Propanil had the greatest persistence of herbicides and insecticides tested in the Ebro Delta of southern Spain. Cinosulfuron was detected in rice paddy water up to two months after application (Ferrero *et al.* 2001). Clomazone persistence in rice paddies is four weeks whereas propanil dissipates within 24 h; quinclorac can persist in rice soil for up to a year and for three weeks in rice water (Quayle *et al.* 2006). According to studies in Brazil, clomazone is widely detected in water from rice-growing areas and has been found to persist for 130 d (Crestani *et al.* 2007).

A number of studies have similarly investigated the persistence of compounds in rice tissues and other harvestable plant resources. The residue of carbofuran remaining on brown rice grains was 0.17 ppm, which is somewhat less than the maximum residue limit of 0.2 ppm set by FAO/WHO (Lee *et al.* 1994). Additionally, relatively high residues of propanil were detected in edible plant (non-rice) tissues nearly two months after treatment and potentially pose a threat to humans and other life forms (Perera *et al.* 1999).

Exposure of Rice Fauna to Pesticides. Relatively few studies have investigated exposure of rice-associated fauna to pesticides. Most of these studies pertain to the timing of cultivation practices or the specific ecology of animals studied. For example, researchers found that highly mobile insects, including many species of beneficial predators, have a greater probability of coming into contact with pesticides (Hamilton 2008). Thus, due to their ecology, these species are at greater risk for exposure and adverse effects.

In addition, Flickinger and King (1972) found that seed-eating invertebrates (such as the crayfish *Procambarus clarki, Cambarus diogenes* and aquatic snails *Physa* spp., *Lymnaea* spp.) inhabiting rice fields where pesticide-treated seed was aerially dispersed into shallow water accumulated the largest concentrations of aldrin and dieldrin of all species in the food web. Consequently, birds (primarily waterfowl) feeding on these invertebrates were disproportionately killed through secondary poisoning. Also due to cultivation practices, Osten *et al.* (2005) found that exposure of ducks was correlated with the timing of insecticide applications to the rice crop. Attributes of avian ecology also contributed to exposure. For instance, late-staying spring migrants were at greater risk of exposure to dieldrin in rice fields in Texas, USA (Flickinger 1979).

Effects

Direct Toxicological Effects to Birds. Researchers have documented field mortality of many avian species as a result of rice pesticide applications. Flickinger and King (1972) showed that aldrin caused mortality of 32 bird species. Species found dead in or near aldrin-treated rice fields included: Fulvous Whistling Duck (*Dendrocygna bicolor*), Greater White-fronted Goose (*Anser albifrons*), Snow Goose (*Chen caerulescens*), Canada Goose (*Branta canadensis*), Blue-winged Teal (*Anas discors*), Mottled Duck (*Anas fulvigula*), Mallard (*Anas platyrhynchos*), Pied-billed Grebe (*Podilymbus podiceps*), White-faced Ibis (*Plegadis chihi*), Great Blue Heron (*Ardea herodias*), Reddish Egret (*Egretta rufescens*), King Rail (*Rallus elegans*), Purple Gallinule (*Porphyrio martinica*), Common Moorhen (*Gallinula chloropus*), Black-necked Stilt (*Himantopus mexicanus*), Killdeer (*Charadrius vociferous*), Lesser Yellowlegs (*Tringa flavipes*), Western Sandpiper (*Calidris mauri*), Semipalmated Sandpiper (*Calidris pusilla*), Pectoral Sandpiper (*Calidris melanotos*), Short-billed Dowitcher (*Limnodromus griseus*), Laughing Gull (*Leucophaeus atricilla*), Mourning Dove (*Zenaida macroura*), Great Horned Owl (*Bubo virginianus*), Eastern Meadowlark (*Sturnella magna*), Red-winged Blackbird (*Agelaius phoeniceus*), Boat-tailed Grackle (*Quiscalus major*), Common Grackle (*Quiscalus quiscula*), Dickcissel (*Spiza americana*) and House Sparrow (*Passer domesticus*). Dieldrin was shown to cause mortality of geese in rice habitats in Texas (Flickinger 1979) and a large die-off of nesting White-faced Ibis (Flickinger and Meeker 1972). Granular carbofuran resulted in bird mortality in Senegal when granules were spilled outside of cropped areas and ingested by birds. Affected species included weavers and wagtails (Mullie *et al.* 1991).

Aldrin-treated rice seed use in the Gulf Coast states, USA, was associated with a precipitous decline in numbers of Fulvous Whistling Duck (Flickinger and King 1972). Pesticide residues in eggs, nestlings and adults were sufficiently high to account for this dramatic population loss. Wading bird population reductions (Purple Heron *Ardea purpurea*, Little Egret *Egretta garzetta*) have been associated with pesticide use in southern Europe (review in Czech and Parsons 2002).

Carbofuran, which replaced aldrin, was also highly toxic to birds (reviewed in Flickinger *et al.* 1980) but was five times less persistent. Species found dead or intoxicated from carbofuran exposure included Western Sandpiper, Pectoral Sandpiper and Red-winged Blackbird. Flickinger *et al.* (1986) reported that carbofuran granules in rice fields killed ducks, shorebirds, blackbirds, fish, crayfish and other aquatic invertebrates. Nine species of passerines and two shorebird species were found dead in recently planted, unflooded rice fields in Texas. In all, 106 birds were killed in a single incident, including Buff-breasted Sandpiper (*Tryngites subruficollis*), Least Sandpiper (*Calidris minutilla*), Mourning Dove, Northern Mockingbird (*Mimus polyglottos*), Dickcissel, Savannah Sparrow (*Passerculus sandwichensis*), Great-tailed Grackle (*Quiscalus mexicanus*), Red-winged Blackbird, Common Grackle, Brown-headed Cowbird (*Molothrus ater*) and Eastern Meadowlark.

Inhibition of critical neural enzymes by pesticide exposure has been documented from rice applications. Depression of brain cholinesterase in birds killed by carbofuran exposure ranged from 20-50%. Mortality continued for two weeks after treatment (Flickinger *et al.* 1986). Osten *et al.* (2005) showed depressed neural enzyme levels in ducks due to carbofuran exposure in rice fields in Mexico. Bobolinks (*Dolichonyx oryzivorus*) feeding in Bolivian rice fields were exposed to monocrotophos that was applied for insect control. Approximately 40% of birds captured in nets at roosts away from the fields exhibited lethal and sublethal levels of cholinesterase activity in their blood (R. Renfrew, A. M. Saavedra, P. Mineau and M. Hooper, unpublished data). Netted birds likely provide a conservative estimate of the actual proportion of birds with depressed cholinesterase, because nets do not sample birds with moderately to severely-impaired motor skills.

Pesticides have been used to target avian pest species in rice, and in some areas this practice continues, usually illegally. Organophosphates such as monocrotophos and parathion were used to control Dickcissels in Venezuelan rice and sorghum (Basili and Temple 1999a). Documentation of die-offs included one case with 1,000 lethally-poisoned individuals and evidence of five other control efforts at roosts. Most notably, one farmer claimed to be "knee-deep" in carcasses after a pesticide application at a Dickcissel roost (Basili and Temple 1999b). Farmers in Bolivia used pesticides, including carbofuran and monocrotophos, to reduce rice consumption by Bobolinks. Most rice growers in Bolivia claim that pesticides are no longer used to control Bobolinks because populations are considerably smaller compared to 30-50 years ago, and pose less of a threat to rice (R. Renfrew, unpublished data). In fact, actual damage to rice crops from birds can be lower than perceived by farmers (Basili and Temple 1999a). However, during a 2005-2007 study, farmers consistently reported that it was common practice to place undiluted monocrotophos (600 g/l) directly into active nests of Wattled Jacanas (*Jacana jaca-*

na) and Purple Gallinules located within rice fields (R. Renfrew, unpublished data). Zaccagnini and Matthern (cited in Zaccagnini 2002 and Bernardos and Zaccagnini 2008) reported on the use of parathion (then legally registered as an insecticide in rice) to control pest bird species in Santa Fe and Entre Rios Provinces in Argentina in 1991. Following one such application to a 50 ha field, the authors found over 500 blackbirds (mixed species), 20 waterfowl and another 50 birds of ten other species, but this is only a partial accounting of the total mortality.

In a comprehensive analysis based on insecticide use in the USA, Mineau and Whiteside (2006) found that the risk of death to birds from field application rates of pesticides in rice showed annual variation, but was generally stable over decades and also relatively high (10% of the area in rice cultivation receives pesticide treatments likely to cause avian death). Rice was found to be one of the top 15 crops with the highest cumulative risk to birds; the cumulative cropping area in the USA where avian mortality was expected totaled approximately 115,000 ha in the early 2000s.

In the present analysis, the relative toxicity and kill potential was ranked for all currently registered products (Table 3). This measure of toxicity (HD5 or Hazardous Dose 5%) is based on a species sensitivity distribution of median lethal doses (LD50s) and is an estimate of the LD50 for a hypothetical species in the 5% lower tail of the distribution of all species-specific LD50 values after scaling for body mass in order to account for the fact that the toxicity of most pesticides does not scale linearly to body mass (Mineau *et al.* 1996). Mineau *et al.* (2001) argued that this is the best unbiased measure of toxicity for birds at large. The next step was to convert application rates given in grams of active ingredient per hectare to the number of HD5 doses per square meter of field (or the number of LD50 doses delivered per square meter of field for 1 kg of bird of a species at the 5% tail of the species sensitivity distribution). The number of HD5 doses per square meter varied from 0 (eleven compounds) to nearly 1,000 (carbofuran).

The risk that birds would be killed by a given pesticide application was calculated from an empirically-based model initially introduced in Mineau (2002). However, this model was modified to take into account the addition of more field studies and a recent re-evaluation of all the component agricultural studies by a panel of four evaluators mandated by the European Food Safety Authority (EFSA 2008). Mineau (2002) argued for the importance of dermal exposure when assessing field data; however, because of the uncertainty surrounding dermal exposure to non-cholinesterase-inhibiting pesticides (EFSA 2008), the algorithm used here considered only the oral toxicity of the various pesticides to birds in arriving at a probability of kill. For example, a probability of 0.20 indicates that, given the existing corpus of avian field studies (n > 100), we would expect to find avian mortality in approximately one in five applications. It was recently argued (Mineau *et al.* 2009), based on comparison of these risk ratings with poisoning incidents that a probability of kill greater than 10% is worthy of concern; probabilities of mortality of 50% or more indicate a critical risk that cannot be mitigated. No differentiation was made here between liquid and granular applications. As argued by Mineau and Whiteside (2005), the risk from clay-based granules is likely lower than that of a foliar application (especially if applied to standing water), but other granule bases (e.g. corn-cob or silica) are thought to carry a risk as high or higher than corresponding spray applications. Carbofuran granules, for example, are silica-based and their risk to birds is likely underestimated here. Seed treatment chemicals were not considered in the rankings. Because of the attractiveness of rice seed to birds, this necessitates a different assessment.

On that basis, 31 insecticides or 42% of assessed insecticide products carry a significant risk of mortality to birds at rates applied. Twelve products, namely bensultap, carbofuran, diazinon, dicrotophos, disulfoton, ethyl-(p-nitrophenyl) ester (EPN), fenthion, methamidophos, monocrotophos, phorate, phosphamidon and quinalphos

Table 3. Avian acute toxicity risk from currently-registered rice insecticides (estimated for both the high and low end of application rates for each pesticide). The measure of toxicity (HD5 or Hazardous Dose 5%) is based on a species sensitivity distribution of median lethal doses (LD50s) and is an estimate of the LD50 for a hypothetical species at the 5% lower tail of the distribution of all species-specific LD50 values scaled to body mass in order to account for the fact that the toxicity of most pesticides does not scale linearly to body mass (Mineau *et al.* 1996). Mineau *et al.* (2001) have argued that this is the best unbiased measure of toxicity for birds at large.

Pesticide	End of range of possible application rates based on Table 2	Application rate (g ai/ha)	Avian HD5 (Mineau *et al.* 2001)	Number of HD5 equivalent doses per kg of bird per m^2	Predicted risk of mortality
carbofuran	high	2000	0.21	952	0.89
monocrotophos	high	2000	0.42	476	0.82
disulfoton	high	3000	0.81	370	0.78
bensultap	high	1500	0.41	366	0.78
phorate	high	1000	0.34	294	0.74
carbofuran	low	600	0.21	286	0.73
quinalphos	high	1000	0.42	238	0.70
phorate	low	750	0.34	221	0.68
EPN	high	1000	0.53	189	0.65
diazinon	high	1000	0.59	169	0.63
fenthion	high	1400	0.87	161	0.62
phosphamidon	high	1680	1.08	156	0.61
dicrotophos	high	600	0.42	143	0.59
methamidophos	high	2200	1.7	129	0.57
EPN	low	500	0.53	94.3	0.50
isoxathion	high	900	0.96	93.8	0.49
propoxur	high	1120	1.31	85.5	0.47
dicrotophos	low	300	0.42	71.4	0.43
isoxathion	low	600	0.96	62.5	0.40
bensultap	low	250	0.41	61.0	0.40
monocrotophos	low	250	0.42	59.5	0.39
benfuracarb	high	2500	4.23	59.1	0.39
diazinon	low	300	0.59	50.8	0.36
naled	high	840	1.72	48.8	0.35
azinphos-ethyl	high	700	1.53	45.8	0.33
quinalphos	low	190	0.42	45.2	0.33
azinphos methyl	high	1000	2.28	43.9	0.33
parathion-methyl	high	840	2.13	39.4	0.30
chlorfenvinphos	high	1000	2.73	36.6	0.29
omethoate	high	1500	4.14	36.2	0.29
disulfoton	low	280	0.81	34.6	0.28

Table 3. (Continued) Avian acute toxicity risk from currently-registered rice insecticides (estimated for both the high and low end of application rates for each pesticide). The measure of toxicity (HD5 or Hazardous Dose 5%) is based on a species sensitivity distribution of median lethal doses (LD50s) and is an estimate of the LD50 for a hypothetical species at the 5% lower tail of the distribution of all species-specific LD50 values scaled to body mass in order to account for the fact that the toxicity of most pesticides does not scale linearly to body mass (Mineau *et al.* 1996). Mineau *et al.* (2001) have argued that this is the best unbiased measure of toxicity for birds at large.

Pesticide	End of range of possible application rates based on Table 2	Application rate (g ai/ha)	Avian HD5 (Mineau *et al.* 2001)	Number of HD5 equivalent doses per kg of bird per m^2	Predicted risk of mortality
fenthion	low	300	0.87	34.5	0.28
azinphos-ethyl	low	500	1.53	32.7	0.27
triazophos	high	450	1.68	26.8	0.23
fenitrothion	high	900	3.37	26.7	0.23
chlorpyrifos	high	1000	3.76	26.6	0.23
phosphamidon	low	280	1.08	25.9	0.23
parathion-methyl	low	500	2.13	23.5	0.21
trichlorfon	high	3000	13.4	22.39	0.21
omethoate	low	900	4.14	21.74	0.20
endosulfan	high	2000	9.53	21.0	0.20
azinphos methyl	low	400	2.28	17.5	0.17
lindane (gamma BHC)	high	2000	12.54	15.9	0.16
pyridaphenthion	high	1000	6.56	15.2	0.15
dichlorvos	high	750	5.18	14.5	0.15
vamidothion	high	500	3.72	13.4	0.14
naled	low	225	1.72	13.1	0.14
triazophos	low	200	1.68	11.9	0.13
propoxur	low	150	1.31	11.5	0.12
fenitrothion	low	375	3.37	11.1	0.12
chlorpyrifos	low	400	3.76	10.6	0.11
carbosulfan	high	1000	9.52	10.5	0.11
benfuracarb	low	400	4.23	9.46	0.10
chlorfenvinphos	low	250	2.73	9.16	0.10
methamidophos	low	150	1.7	8.82	0.10
lindane (gamma BHC)	low	1000	12.54	7.97	0.09
thiacloprid	high	360	4.73	7.61	0.09
carbaryl	high	2242	30	7.47	0.09
pirimiphos methyl	high	1000	13.5	7.41	0.08
methomyl	high	600	8.46	7.09	0.08
fipronil	high	150	2.15	6.98	0.08
dimethoate	unique	400	5.78	6.92	0.08
tetrachlorvinphos	high	1680	25.32	6.64	0.08

Table 3. (Continued) Avian acute toxicity risk from currently-registered rice insecticides (estimated for both the high and low end of application rates for each pesticide). The measure of toxicity (HD5 or Hazardous Dose 5%) is based on a species sensitivity distribution of median lethal doses (LD50s) and is an estimate of the LD50 for a hypothetical species at the 5% lower tail of the distribution of all species-specific LD50 values scaled to body mass in order to account for the fact that the toxicity of most pesticides does not scale linearly to body mass (Mineau *et al.* 1996). Mineau *et al.* (2001) have argued that this is the best unbiased measure of toxicity for birds at large.

Pesticide	End of range of possible application rates based on Table 2	Application rate (g ai/ha)	Avian HD5 (Mineau *et al.* 2001)	Number of HD5 equivalent doses per kg of bird per m^2	Predicted risk of mortality
chlorpyrifos-methyl	high	1500	25.32	5.92	0.07
dichlorvos	low	300	5.18	5.79	0.07
acephate	high	1020	18.52	5.51	0.07
isoprocarb	high	750	14.2	5.28	0.06
phenthoate	high	1020	23.2	4.40	0.05
carbosulfan	low	400	9.52	4.20	0.05
vamidothion	low	150	3.72	4.03	0.05
fenobucarb (BPMC)	high	1200	31.1	3.86	0.05
pirimiphos methyl	low	500	13.5	3.70	0.05
XMC	unique	600	18.1	3.31	0.04
chlorpyrifos-methyl	low	750	25.32	2.96	0.04
endosulfan	low	260	9.53	2.73	0.04
imidacloprid	high	200	8.43	2.37	0.03
methomyl	low	200	8.46	2.36	0.03
trichlorfon	low	300	13.4	2.24	0.03
tetrachlorvinphos	low	560	25.32	2.21	0.03
phenthoate	low	510	23.2	2.20	0.03
thiacloprid	low	100	4.73	2.11	0.03
acephate	low	360	18.52	1.94	0.03
isoprocarb	low	250	14.2	1.76	0.02
carbaryl	low	528	30	1.76	0.02
fenobucarb (BPMC)	low	500	31.1	1.61	0.02
malathion	high	2000	139	1.44	0.02
acetamiprid	high	300	20.91	1.43	0.02
pyridaphenthion	low	60	6.56	0.91	0.01
xylylcarb	low	40	6.2	0.65	0.01
xylylcarb	high	40	6.2	0.65	0.01
fipronil	low	12.5	2.15	0.58	0.01
acetamiprid	low	75	20.91	0.36	0.01
imidacloprid	low	30	8.43	0.36	0.01
malathion	low	300	139	0.22	0.00
silafluofen	high	300	193	0.16	0.00

Table 3. (Continued) Avian acute toxicity risk from currently-registered rice insecticides (estimated for both the high and low end of application rates for each pesticide). The measure of toxicity (HD5 or Hazardous Dose 5%) is based on a species sensitivity distribution of median lethal doses (LD50s) and is an estimate of the LD50 for a hypothetical species at the 5% lower tail of the distribution of all species-specific LD50 values scaled to body mass in order to account for the fact that the toxicity of most pesticides does not scale linearly to body mass (Mineau *et al.* 1996). Mineau *et al.* (2001) have argued that this is the best unbiased measure of toxicity for birds at large.

Pesticide	End of range of possible application rates based on Table 2	Application rate (g ai/ha)	Avian HD5 (Mineau *et al.* 2001)	Number of HD5 equivalent doses per kg of bird per m^2	Predicted risk of mortality
nitenpyram	high	400	261	0.15	0.00
pymetrozine	high	300	208	0.14	0.00
tebufenozide	high	336	250	0.13	0.00
thiocyclam	unique	375	333	0.11	0.00
thiamethoxam	high	200	180	0.11	0.00
dinotefuran	high	200	193	0.10	0.00
etofenprox	high	200	193	0.10	0.00
methoxyfenozide	unique	200	261	0.08	0.00
fenvalerate	high	224	322	0.07	0.00
methoprene	high	112	193	0.06	0.00
dinotefuran	low	100	193	0.05	0.00
pymetrozine	low	95	208	0.05	0.00
cypermethrin	high	250	579	0.04	0.00
chromafenozide	high	200	483	0.04	0.00
etofenprox	low	75	193	0.04	0.00
beta-cypermethrin	high	250	775	0.03	0.00
diflubenzuron	high	280	953	0.03	0.00
methoprene	low	56	193	0.03	0.00
tebufenozide	low	67	250	0.03	0.00
silafluofen	low	50	193	0.03	0.00
cypermethrin	low	125	579	0.02	0.00
azadirachtin	high	48	261	0.02	0.00
fenvalerate	low	56	322	0.02	0.00
beta-cypermethrin	low	125	775	0.02	0.00
permethrin	high	500	3128	0.02	0.00
deltamethrin	high	15	97	0.02	0.00
chromafenozide	low	50	483	0.01	0.00
cyfluthrin	high	50	485	0.01	0.00
cycloprothrin	high	40	483	0.01	0.00
permethrin	low	250	3128	0.01	0.00
buprofezin	unique	50	680	0.01	0.00
tralomethrin	high	20	292	0.01	0.00

Table 3. (Continued) Avian acute toxicity risk from currently-registered rice insecticides (estimated for both the high and low end of application rates for each pesticide). The measure of toxicity (HD5 or Hazardous Dose 5%) is based on a species sensitivity distribution of median lethal doses (LD50s) and is an estimate of the LD50 for a hypothetical species at the 5% lower tail of the distribution of all species-specific LD50 values scaled to body mass in order to account for the fact that the toxicity of most pesticides does not scale linearly to body mass (Mineau *et al.* 1996). Mineau *et al.* (2001) have argued that this is the best unbiased measure of toxicity for birds at large.

Pesticide	End of range of possible application rates based on Table 2	Application rate (g ai/ha)	Avian HD5 (Mineau *et al.* 2001)	Number of HD5 equivalent doses per kg of bird per m^2	Predicted risk of mortality
diflubenzuron	low	62	953	0.01	0.00
nitenpyram	low	15	261	0.01	0.00
thiamethoxam	low	10	180	0.01	0.00
cycloprothrin	low	20	483	0.00	0.00
cyfluthrin	low	15	485	0.00	0.00
zeta-cypermethrin	high	28	987	0.00	0.00
deltamethrin	low	2.5	97	0.00	0.00
tralomethrin	low	7.5	292	0.00	0.00
zeta-cypermethrin	low	19	987	0.00	0.00
azadirachtin	low	3.4	261	0.00	0.00
lambda-cyhalothrin	high	5	428	0.00	0.00
lambda-cyhalothrin	low	2	428	0.00	0.00
alpha-cypermethrin	high	20	9634	0.00	0.00
alpha-cypermethrin	low	10	9634	0.00	0.00

carry a critically high (>50% mortality) risk to birds. Kills of exposed birds are predicted to be frequent and largely unavoidable. The insecticides carbofuran, phorate and EPN carry a higher than 50% risk of mortality at the lowest use rate in rice. Twenty-one of the assessed insecticides carry a greater than 10% risk of mortality at their lowest estimated use rate (Table 3).

Emergency use exemption requests for carbofuran use in USA rice crops were recently considered (but rejected) by the United States Environmental Protection Agency (USEPA). Despite a long history of bird kills and recent restrictions on some carbofuran products in North America and Europe, carbofuran and its "clone" furathiocard are marketed for rice by several manufacturers. Several other organophoshorous and carbamate insecticides of high toxicity to birds are still marketed for rice (Table 2).

Toxicity or application information was missing for 18 insecticides preventing the inclusion of those products in the above ranking (Table 4). Based on toxicity alone, several of these products could be expected to give rise to frequent bird mortality. It should be noted that there is a complete lack of toxicity information for several pesticides developed primarily for the Asian market. Japan is one of the few developed countries that do not require information on avian toxicity as a condition of registration. Even North American or European pesticide manufacturers appear not to have generated (or publicly released) avian toxicity information for products intended primarily for Asia.

Indirect Effects on Birds. Indirect effects on birds as a result of pesticide applications in rice fields include reduced prey populations and habitat changes. Prey species include macroinvertebrates as well as vertebrates such as amphibians and fish. Tourenq *et al.* (2003) found that intensive soil management of rice fields and pesticide applications reduced prey resources for aquatic birds. Mesléard *et al.* (2005) showed that insecticide treatments (primarily fipronil) of rice fields in France reduced abundance of six macroinvertebrate families (Baetidae, Corixidae, Dysticidae, Hydrophilidae, Coenagrionidae, Libellulidae). Reduced abundance of predatory insects was pronounced. The use of fungicides was negatively correlated with the abundance of two mollusk families (Lymnaeidae, Physidae), but herbicide use correlated positively with abundance of mollusks and Tubificidae. Although overall macroinvertebrate biomass was higher in conventional compared to organic plots, this was largely due to the influence of gastropods. Biomass of prey organisms that constituted critical components of local waterbird populations (Coleoptera, Odonata, amphibian larvae) was four times greater in organic compared to conventional rice fields (Mesléard *et al.* 2005).

Researchers have documented synergistic effects on prey populations resulting from cultivation practices. When paraquat

Table 4. Insecticides used in rice but with missing toxicity or rate information preventing their inclusion for avian mortality assessment. Note that the cited toxicity of furathiocarb is lower than indicated here. However, tests were performed with 'stabilized' material which would prevent conversion to carbofuran, the insecticidally active molecule. The carbofuran value is therefore assumed here.

Pesticide	Avian HD5 (Mineau *et al.* 2001; see text or Table 3)	Low rate suitable for risk assessment in rice (g a.i./ha)	High rate suitable for risk assessment in rice (g a.i./ha)
cartap hydrochloride	No data	250	1000
clothianidin	41.5	No data	No data
dimethylvinphos	No data	600	800
Ethiprole	No data	670	670
furathiocarb	0.21	No data	No data
isocarbofos (optunal)	0.26	No data	No data
mecarbam	No data	1120	2240
metolcarb (MTMC)	No data	510	1200
thiosultap	No data	No data	No data

was applied to unflooded paddies, minimal impact on aquatic invertebrates was observed, however when applied in flooded conditions, paraquat negatively impacted dragonfly nymphs, leeches, snails and crustaceans (review in Ishibashi *et al.* 1983). Other herbicides applied to rice (including thiobencarb, simetryne, chlormethoxynil and oxadiazon) were acutely toxic to mosquito and chironomid larvae, and to tadpoles. Bioaccumulation and sedimentation of herbicides are thought to be responsible for rapid death in midges, tadpoles, dragonfly nymphs, water beetle larvae and pond snails (Ishibashi *et al.* 1983). Molluscicides (such as organotin compounds fentin chloride and fentin acetate) used in the Philippines to combat an introduced herbivorous snail resulted in adverse human health effects (e.g. skin pathologies) which led to their ban in 1990. In southern Europe, a significant ecological effect of pesticide application is the bioaccumulation of toxic chemicals in crayfish (Jorgensen *et al.* 1997). Odonata, Ephemeroptera, Nematocera (Diptera) and Hydrocorisae (Hemiptera) were reduced by carbofuran applications; larval Ephemeroptera appeared most sensitive (Mullie *et al.* 1991). Flickinger *et al.* (1980) found many species adversely affected by carbofuran applications including cricket frogs (paralysis, abnormal behavior), earthworms, back swimmers, whirligig beetles, dragonfly naiads and predaceous diving beetles (all killed). Impacts to aquatic invertebrates were predicted to translate to adverse indirect effects to insectivorous birds (Mullie *et al.* 1991).

In a limited study, authors found little evidence that Bt (*Bacillus thuringiensis*)-rice negatively affected arthropods (Li *et al.* 2007). Risk assessment using the water quality monitoring model RICEWQ in Greece suggested that recommended applications of molinate are hazardous to *Daphnia* spp., algae, fish and *Gammarus* spp. (Karpouzas *et al.* 2006). Thiobencarb has been shown to cause up to 65% reduction in chironomid larvae and planktonic invertebrates (reviewed in Burdett *et al.* 2001). Chironomids are an important food source for waterfowl. Molinate exposure increased development time for *Chironomus tepperi;* ontological changes were correlated with a reduction in abundance (Burdett *et al.* 2001). Emergence success was reduced as a result of exposure to clomazone. Although other impacts were equivocal, clomazone reduced macrophyte biomass. *C. tepperi* was most sensitive to thiobencarb which caused significant mortality, wing shortening and delayed emergence—all of which can adversely affect population levels.

Doses of 500-1000 g ai/ha of carbofuran were found to be very detrimental to many forms of animal life (Mullie *et al.* 1991). Disruption of entire food webs associated with rice cultivation has been shown to result from pesticide applications. Loss of early season pest predator populations was associated with resurgence of later season rice pests (Settle *et al.* 1996). Carbofuran and monocrotophos had the largest adverse impact on pest predators associated with paddy soil surface. The loss of predatory insects in rice fields in France treated with insecticide caused cascading effects to the macroinvertebrate community including increased pest invertebrate abundance and increased populations of invertebrates of little food value to birds (Mesléard *et al.* 2005).

Rice farmers in West Africa have reported on the acute effects of carbofuran on frogs as well as a reduction in macroinvertebrate biomass (Mullie *et al.* 1991). Frogs showed abnormal behavior within 4 h of application; death followed within a 24 h period. Toads in rice fields in Argentina had lower butyryl-cholinesterase than toads sampled in control locations. More than 80% of sampled toads in rice fields showed a positive reactivation when samples were exposed to an anti-cholinesterase antidote; in contrast, no toad samples from reference locations reactivated (Maximiliano Attademo *et al.* 2007). In addition to cholinesterase inhibition, toads from agricultural habitats showed increased levels of glutathione S-transferase (GST) activity indicating detoxification.

Carbofuran has been shown to greatly reduce the wildlife value of rice habitats. Many fish species including mosquitofish (*Gambu-*

sia spp.), Gulf Menhaden (*Brevootia patronus*), Atlantic Croaker (*Micropogonias undulates*) and European Carp (*Cyprinus carpio*) were found dead within a day of insecticide treatment (Flickinger *et al.* 1980). Insecticide use in rice-fish farming has caused economic losses in Asia (Cagauan 1995) including a 67% decline in fish yields in Malaysia where companion cropping of fish boosts overall production of rice paddies (see also Wood *et al.* 2010). Morphological effects in fish included darkening of skin, swelling of eyes, erosion of fin margins and others. Behavioral effects included erratic swimming, loss of appetite and depressed courtship behavior. A number of biochemical changes were also observed including changes to DNA and RNA. Notwithstanding the effects of carbofuran documented above, in general, pesticide classes ranked as follows in their toxicity to fish: pyrethroids > organochlorines > organophosphates > carbamates (Cagauan 1995).

Herbicides may reduce phytoplankton and thus indirectly affect the fish crop in rice-fish culture. Another impact is in the decomposition of targeted weed species which reduces oxygen levels in the paddy thus affecting fish populations. Acetyl-cholinesterase levels in brain and muscle of Silver Catfish (*Schilbe intermedius*) were significantly depressed (45-47%) following experimental exposure to clomazone. When impacted fish were removed from contaminated water, researchers documented a significant recovery of enzyme levels (Crestani *et al.* 2007). In addition, histological examination revealed liver damage in fish exposed to clomazone.

Best Practices

Mitigating Impacts through Substitution and Reduction. Toxic use reduction strategies promote the substitution or reduction of the most toxic biocidal compounds. The literature contains strong recommendations that the most severely toxic pesticides be substituted with less toxic compounds. One study showed that insecticides used to eliminate rice-damaging crayfish during the early growing season could be substituted with surfactant which has lower toxicity and tendency to accumulate. An additional benefit is that the substitution of surfactant for pesticides would merely inactivate the crayfish during the time when rice is vulnerable to burrowing. Substituting surfactant for insecticides would allow rice farmers to harvest crayfish as a secondary crop (Jorgensen *et al.* 1997) and leave them available as waterbird prey (Huner *et al.* 2002).

A compensating strategy supporting pesticide use reduction is to promote natural pest predator populations, which can be accomplished by increasing habitat diversity through reduced insecticide use. Tropical rice fields have the capacity to support a diverse community of pest predators and parasitoids that could be exploited to reduce chemical control of pests (Bambaradeniya *et al.* 2004). Paddy-level pest and predator communities can be disrupted or managed by larger-scale practices such as planting schedules, water management and chemical use (Settle *et al.* 1996).

The rice industry in Asia and elsewhere underwent dramatic changes during the past decades under the guise of a program termed the Green Revolution. In Indonesia, rice was packaged for mass consumption and included high-yield variety seed, fertilizer and pesticides (the application of which was calendar-based rather than based on field-observed need). Subsidies for pesticides were made available to Indonesian farmers and led to mass adoption (Settle *et al.* 1996). A devastating pest, the Rice Brown Planthopper (*Nilaparvata lugens*), is a direct outcome of Green Revolution cultivation practices—a "self-inflicted wound whose degree of damage is directly correlated to insecticide use" (Settle *et al.* 1996, p. 1977). Intensive insecticide use accelerated resistance in pests. Ten years into the Green Revolution, the Indonesian government banned the use of 57 insecticides, eliminated the pesticide subsidy, and promoted Integrated Pest Management (IPM). Without intensive pesticide inputs, pest predator populations developed in response to abundant detritivores and plankton-feeding insects present in rice paddies early in the season. Pesticides that eliminated this prey base delayed and stunted natural predator populations that effectively control herbivorous insects later in the season. Settle *et al.* (1996) demonstrated that pest problems in tropical rice systems were most likely caused by insecticide use. According to the authors, "the best strategy for biological control in tropical rice is for farmers to conserve the diversity of existing species through major reductions in pesticide use, to keep dry fallow periods short, and to maintain the heterogeneity of small-scale rice landscapes" (p. 1986). The success with which the Green Revolution was adopted and subsequently transformed rice production in Asia could be reproduced with "greener" methods (such as selective control strategies that minimize non-target impacts) now available to rice farmers.

Countries other than Indonesia have resorted to rice pesticide bans in order to protect human and environmental health. The USEPA suspended the use of aldrin-treated rice seed in 1974 (Flickinger *et al.* 1980). Philippine farmers were found to switch to high application rates of insecticides (e.g. endosulfan) due to the low efficacy of molluscicides in controlling golden snails (*Pomacea* spp.). The Philippine government considered banning the importation of endosulfan as a result (Cagauan 1995). In Argentina, die-offs of Swainson's Hawk (*Buteo swainsoni*) (Goldstein *et al.* 1999) led to a cancellation of monocrotophos in the country after restrictions on its use were ineffective (Hooper 1999; Hooper *et al.* 1999). The die-offs also prompted an eventual withdrawal of the product worldwide by a major agrochemical producer (Hooper *et al.* 1999, 2003). A subsequent mortality event of approximately 100,000 passerines due to intentional exposure to monocrotophos in two 35-ha fields in Argentina (Rivera-Milán *et al.* 2004) likely helped to spur regulatory action (Newcomer 2003). Although these incidents did not occur in rice fields, monocrotophos had been the primary insecticide used in rice prior to withdrawal. The result of extensive water contamination in Greece has been the ban of several pesticides used in rice cropping including molinate and bentazone (Miao *et al.* 2003b). In contrast to outright bans, uses and formulations of registered compounds can be restricted to protect non-target wildlife (Flickinger *et al.* 1980).

Reductions and substitutions of compounds have been proposed when adverse effects were considered too great. Research in Philippine rice paddies showed that fish are at risk during insecticide applications at recommended rates; some rates were proposed to be reduced by a factor of ten to be safe for fish (Cagauan 1995). Biological control (e.g. botanical insecticides) appears promising for rice-fish culture although these compounds also exhibit some toxicity to fish. A pond refuge has been recommended to help fish survive pesticide application in rice-fish culture. The herbicide butachlor was found to be highly toxic to fish and resulted in fish mortality at recommended rates of application (Cagauan 1995). Diquat may be an appropriate substitute for more toxic herbicides. Cagauan (1995) makes the case that promoting rice-fish culture could facilitate IPM by substituting "do not spray" with "grow fish." In addition, fish have been shown to help control insect pests and weeds in rice (Wood *et al.* 2010). Reduction of rice pesticides by 65% in Indonesia was associated with a 12% increase in rice yield (Pimentel 2002). The predicted risk of avian mortality from the five most toxic compounds used in rice could be reduced from 8-64% if lower recommended rates of application were employed (Table 3). In Argentina a tool was recently developed to help farmers predict bird mortality associated with user-defined application rates of specific pesticides, based on Mineau (2002). This type of outreach tool may serve as a model for outreach technicians in other countries that wish to explore with landowners the effects of product substitution or reducing site-specific application rates (Bernardos and Zaccagnini 2008).

Alternatives to pesticides for reducing crop damage by birds have been widely explored and employed, but a panacea to the problem of avian pests in rice has not been found. Years of intensive research has been dedicated to the search for non-lethal control methods, especially repellent applications to seeds (Avery 2002) such as caffeine (Werner *et al.* 2005). In developing countries, economic constraints limit the availability of repellents, and farmers most often use audio deterrents such as repetitious fireworks and gunfire, but also include visual tactics such as smoke, mirrors and reflective tape (Basili and Temple 1999a; Renfrew and Saavedra 2007).

Water and Soil Management. Cultivation practices, especially the manipulation of rice paddy soil and water, can be used to reduce pesticide toxicity to birds and other non-target fauna. Herbicide applications are predicted to facilitate outbreaks of diseases and pests due to the disruption of soil and aquatic fauna (Ishibashi *et al.* 1983). The accumulation of persistent organic pollutants such as dichlorodiphenyldichloroethane (DDD) and dichlorodiphenyldichloroethylene (DDE) in rice plants is influenced by soil type and presence/amount of rice straw (Yao *et al.* 2007). Amendments to paddy soil such as zero-valent iron, effectively dechlorinated DDT residues in soil (Yao *et al.* 2006). Degradation of carbofuran in soil was proportional to soil temperature (Lee *et al.* 1994). Sandy soil may allow carbofuran granules to persist for longer at the soil's surface thereby increasing the risk of exposure to non-target wildlife. In addition, carbofuran should be distributed evenly in rice fields to avoid concentration areas (Flickinger *et al.* 1980). Carbofuran was found to be largely sequestered in soil rather than in plant tissue. Soil pH is extremely important in regulating the persistence of carbofuran. Mineau (1993) reviewed the evidence for wildlife kills in flooded acidic soils occurring several months after application and suggested that chemical behavior rather than misuse could be the reason for off-season bird kills in rice fields in California, USA. Heavy clay soils, which shrink and crack under varying conditions, may facilitate the transport of pesticides and contaminated soil water to groundwater supplies (Papadopoulou-Mourkidou *et al.* 2004).

The RICEWQ run-off model developed by the USEPA has been found to be useful in predicting pesticide contamination levels of surface waters within basins and in estimating the vulnerability of stream segments (Miao *et al.* 2003a). RICEWQ incorporates transport via overflow and drainage but does not address chemical leaching. A model linking RICEWQ and another model VADOFT was tested to provide a better (but still not comprehensive) picture of pesticide transport from rice fields (Miao *et al.* 2003b). Site-specific water management and seepage rate greatly influenced transport of pesticides from rice fields both through run-off and leaching. The herbicide paraquat, when applied to unflooded rice fields was found to be less toxic to aquatic invertebrates than thiobencarb. Cinosulfuron degradation was found to be accelerated under acid conditions (Ferrero *et al.* 2001). Degradation was affected by exposure to sunlight and other environmental factors. Pyrethroid toxicity is inversely correlated with water temperature (Cagauan 1995).

Holding drainage waters has been advanced as a strategy to allow dissipation of pesticides (Quayle *et al.* 2006). In the 1990s, the California Regional Water Quality Control Board mandated retention of irrigation water in rice fields to allow pesticides to degrade and/or dissipate. Water quality improvements did not result due to the routine practice of emergency releases (Crepeau and Kuivila 2000). A four-week holding time for water was recommended when clomazone is used in rice (Marchesan *et al.* 2007). Surface water contamination was affected by soil characteristics, topography, application schedule, growing practices, chemical properties of pesticides, rate of application and environmental conditions. To hold water for the appropriate time, managers need to improve water management infrastructure including levees (Marchesan *et al.* 2007).

Cultivation Strategies. A number of rice cultivation strategies have proved advantageous in mitigating non-target pesticide impacts. For example, if propanil is applied when rice foliage is present, resulting compound concentrations in water are reduced (Santos *et al.* 1998). Similarly, it has been a traditional cultivation practice in tropical countries such as Indonesia, to plant fields synchronously and to insert a long, dry season to break the pest cycle, but this has been associated with more prevalent and serious Rice Brown Planthopper outbreaks. Alternatively, a staggered planting without a long, dry season was found to be more conducive to outbreak control through the promotion of natural predator populations (Settle *et al.* 1996). Growing rice in crop rotation with legumes and pasture as is done in Australia has been shown to reduce the build-up of pests (Quayle *et al.* 2006). Mullie *et al.* (1991) found that the application of pesticides before flooding and after peak avian migration is the optimal strategy for protecting non-target wildlife. Similarly, in the USA, Flickinger *et al.* (1980) showed that pesticide applications should be timed to avoid peak bird use and made only when control of Rice Waterweevil (*Lissorhoptrus oryzophilus*) larvae is absolutely necessary. Routine methods of pesticide application (including broadcast of granules, spraying emulsifiable concentrates and wettable powders) may put fish at greater risk (Cagauan 1995). Drill-seeding may reduce impacts to the food web in fields with pesticide-treated rice seed (Flickinger and King 1972).

Root zone application is a possible solution but contamination of fish may be an additional challenge (Cagauan 1995). Multiple pesticide application regimes have been found to be incompatible with fish farming in rice fields (Cagauan 1995); rice-fish systems in Vietnam used less pesticide than farmers growing rice only (Berg 2001). Integrated rice-fish and duck-rice farming systems have been documented to reduce pesticide use by 50-65% (Lu and Li 2006; Zhang *et al.* 2008). Pesticides that degrade rapidly may be compatible and could help determine fish stocking schedules (Cagauan 1995).

Monitoring. A variety of toxicity tests are used to characterize pesticide risk to fish. Monitoring residue levels in tissues of dead fish has been found to be a useful diagnostic tool in assessing the pesticide degradation timeline (Cagauan 1995). Degradation times have been found to be generally as follows: organophosphates (3-13 d), pyrethroids (3-7 d), carbamates (1-6 d). Enzymes (butyrylcholinesterase and glutathione *S*-transferase) can be useful monitoring tools for evaluating exposure in rice habitats (Maximiliano Attademo *et al.* 2007). The Silver Catfish has been used as an indicator for water quality in Brazil and is especially useful in rice-fish culture (Crestani *et al.* 2007). Monitoring models such as RICEWG and VADOFT have been found to be useful risk assessment tools for predicting the fate of rice pesticides and identifying appropriate paddy closure durations (Karpouzas *et al.* 2006). The Avian Incidental Monitoring Program (AIMS) was developed to facilitate, coordinate and share current and historical documentation of avian exposure events in the USA (Mineau and Tucker 2002; American Bird Conservancy 2006), and could serve as a model for other countries or for a multinational program provided the basic incident-reporting infrastructure is present.

Farmer Training. Farmer training has been shown to reduce pesticide use by 60%. IPM farmers have slightly higher yields, higher returns and lower economic risk (Settle *et al.* 1996). Over a three-year period in Vietnam, IPM farmers reduced pesticide use by 65% and non-IPM farmers increased pesticide use by 40%, primarily because of increased numbers of pesticide-resistant insects (Berg 2001). Farmer misconceptions include the belief that 1) all insects are harmful to rice cropping, 2) any damage to rice plants will result in significant damage to yield and 3) pesticides provide a preventative benefit to plants with regard to insect pests. Decentralized farmer training programs have been successful in dispelling these false notions (Settle *et al.* 1996). Farmer negligence has been documented in cases where carbofuran applications were made while paddy water was draining from fields—possibly a common practice (Flickinger *et al.* 1980). In addition, rice farmers have illegally treated rice seed with carbofuran to kill bird pests (Flickinger *et al.* 1986). In Bangladesh, Dasgupta *et al.* (2006) have found that overuse and misuse of pesticides is caused by farmer ignorance and poor product labeling. The adoption of IPM is influenced by ownership and farmer education, training, experience, health status, and age. According to farmers, adoption of IPM resulted in improvements in soil, water and air quality, as well as increased abundance of birds, fish and soil invertebrates. In small-scale farming, adoption of IPM must involve multiple farms to reap benefits (Dasgupta *et al.* 2006).

In addition to training, incentives should be developed to curb the overuse/misuse of pesticides. For example, 47% of Bangladesh farmers are estimated to overuse despite the cost of chemicals (Dasgupta *et al.* 2007) and in China, overuse of pesticides is estimated at 40% (Peng *et al.* 2009). Overuse is influenced by misperception, pesticide toxicity, crop composition and location. In the Philippines, a survey of pesticide sprays showed that only 23% were applied at the right time for targeted pests; of these sprays, only 19% used the appropriate chemical (Heong *et al.* 1995). The most successful communication approaches to correct the misuse of pesticides include: strategic extension campaigns, radio-based campaigns, farmer field schools and farmer participatory research. In particular, farmer participatory approaches are most effective in transferring new knowledge to farmers and dispelling misconceptions (Heong *et al.* 1995). The message of paramount importance is convincing farmers that reducing pesticide use will not adversely affect profit (Hamilton 2008).

Information Gaps

The literature characterizing non-target exposure and effects from rice pesticide applications is undeveloped and few explicit recommendations for further investigation have been identified. Miao *et al.* (2003a) suggested that work with hydrological models predicting water quality of surface and groundwater resources associated with rice cultivation can be used for risk mitigation analysis. Scientists have been urged to develop models that simulate simultaneous irrigation and drainage with water management strategies (Miao *et al.* 2003b). Further research on the dechlorination of persistent pesticides has been identified as a need (Yao *et al.* 2006). Further work on the impacts of agrochemicals on early season prey populations of pest predators is also lacking (Settle *et al.* 1996).

Little information exists on pesticide residues found in fish in rice-fish culture (Cagauan 1995). Additionally, the long-term and population-level effects of sublethal exposures (especially in field situations and incorporating *in situ* environmental conditions) on vertebrates are largely unknown (Cagauan 1995). New-to-market pesticides have been largely unstudied and studies examining the impact of IPM on farmers' health are lacking (Dasgupta *et al.* 2006). Better research is needed on impacts of transgenic rice on non-target wildlife (Li *et al.* 2007). Additional modeling with the predictive water quality model RICEWQ could be applied to protect resources in other river basins (Karpouzas *et al.* 2006). Finally, research needs include understanding of effects of flooding on pesticide distribution (review in Czech and Parsons 2002).

CONCLUSIONS

- Pesticide use in some rice cultivation countries is significant and a broad suite of compounds has been applied to rice within recent decades worldwide.
- Several insecticides used extensively in rice in numerous countries are extremely toxic to birds and are expected to cause frequent and largely unavoidable mortality. This issue may be a serious problem in light of the importance of rice fields as habitat for many aquatic bird species.
- Much research has been conducted to evaluate the transport of rice pesticides to groundwater and to adjacent surface water bodies; significant contamination of associated water resources is widespread.
- Exposure of rice fauna to pesticides has been documented in a relatively small number of studies.
- Researchers have documented sub-lethal effects and field mortality of many avian species as a result of rice pesticide applications.
- Many studies have documented indirect effects on birds as a result of pesticide ap-

plications; effects include reduced prey populations and habitat changes.

- Substantial information is available to rice farmers to mitigate pesticide impacts to rice fauna through compound reduction or substitution, water and soil management, modified cultivation strategies, monitoring and farmer training.
- The literature documenting exposure and effects of rice pesticides on birds is relatively undeveloped; an evaluation of research needs is lacking.

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