CANADIAN
WATER QUALITY GUIDELINES: IMIDACLOPRID

SCIENTIFIC SUPPORTING DOCUMENT

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NOTE TO READERS

The Canadian Council of Ministers of the Environment (CCME) is the major intergovernmental forum in Canada for discussion and joint action on environmental issues of national, international and global concern. The 14 member governments work as partners in developing nationally consistent environmental standards, practices and legislation.

This document provides the background information and rationale for the development of the Canadian Water Quality Guidelines for imidacloprid. For additional technical information regarding these guidelines, please contact:

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ABSTRACT

This scientific supporting document describes the development of Canadian Water Quality Guidelines for imidacloprid. It contains a review of technical background information on the chemical and physical properties of imidacloprid, a review of uses in Canada, the distribution and behaviour of imidacloprid in the environment, and the toxicological effects of imidacloprid in freshwater and marine aquatic life, terrestrial crops, mammals and birds. This information is used to derive ambient water quality guidelines for imidacloprid. The guidelines in this document are based on the best available toxicity data at the time of writing, June 2006.

Imidacloprid is an insecticide active ingredient used to control sucking insects, such as aphids, leafhoppers, psyllids and beetles in agricultural crops and turfgrass, as well as domestic pests such as fleas and cockroaches. It is most commonly applied as a soil and foliage treatment, and as a seed dressing. Crops to which this compound is applied include: various grains, maize, fruits, vegetables, potatoes, hops and turf. Imidacloprid is water soluble, persistent in soil and relatively non-volatile under field conditions. Imidacloprid is not expected to bioaccumulate. Imidacloprid residues may be measured in water and soil using High Performance Liquid Chromatography (HPLC), or gas chromatography-mass spectrometry (GC-MS).

Imidacloprid is a nicotinoid neurotoxin that acts by irreversibly blocking acetylcholine receptors. Although mammals and insects both have acetylcholine receptors that can be blocked by imidacloprid, insects are more sensitive than mammals. Symptoms of imidacloprid poisoning include staggering, trembling, immobility and lethargy. Sensitivites of non-target organisms to imidacloprid varies. Imidacloprid may be highly toxic to beneficial insects, such as the honeybee and it is recommended that imidacloprid application be limited in areas frequented by honeybees. Imidacloprid may also induce toxicity in aquatic invertebrates and juvenile fish at low levels and is considered acutely toxic to birds. Moderate concentrations of the insecticide may decrease growth of algae. Phytoxicity is not predicted for crops if the insecticide is applied post-emergence.

Canadian water quality guidelines for the protection of agricultural water uses were not derived for imidacloprid. Insufficient data were available at the time of writing to support the development of these guidelines.

Sufficient toxicity data were available to derive interim freshwater and marine water quality guidelines for imidacloprid for the protection of aquatic life. The interim freshwater guideline was based on a 28-day LOEC (EC_{15}) of 2.25 µg a.i./L for reduced emergence of the midge *Chironomus riparius*. As this is a low effects level chronic study, a safety factor of 0.1 was applied, giving an interim freshwater quality guideline of 0.23 µg a.i./L.

The interim marine water quality guideline was based on a 48-hour LC_{50} of 13 µg a.i./L for larvae of the saltmarsh mosquito *Aedes taeniorhynchus*. As the study was acute, and imidacloprid is non-persistent in water, a safety factor of 0.05 was applied to this value, giving a guideline value of 0.65 µg a.i./L.

These guidelines are intended to protect all forms of aquatic life and all aspects of aquatic life cycles during an indefinite period of exposure to the water column (CCME, 1991).

RÉSUMÉ

Le présent document scientifique complémentaire porte sur l'élaboration des recommandations canadiennes relatives à l'imidaclopride pour la qualité des eaux. Il présente un examen des données techniques de base concernant les propriétés chimiques et physiques de l'imidaclopride, une revue de ses utilisations au Canada, la répartition et le comportement de l'imidaclopride dans l'environnement et les effets toxicologiques de l'imidaclopride sur la vie aquatique dulcicole et marine, les cultures terrestres, les mammifères et les oiseaux. Cette information sert à élaborer les recommandations relatives à l'imidaclopride pour la qualité des eaux ambiantes. Les recommandations formulées dans le présent document sont fondées sur les meilleures données toxicologiques disponibles au moment de la rédaction, en juin 2006.

L'imidaclopride est une matière active insecticide utilisée pour lutter contre les insectes suceurs tels que les pucerons, les cicadelles, les psylles et les coléoptères dans les cultures agricoles et sur les pelouses ainsi que pour lutter contre les organismes nuisibles domestiques comme les puces et les coquerelles. Le plus souvent, il est utilisé sur le sol et sur le feuillage et dans l'enrobage des semences. Les cultures sur lesquelles le composé est appliqué comprennent diverses céréales, le maïs, des fruits, des légumes, la pomme de terre, le houblon et le gazon. L'imidaclopride est soluble dans l'eau, persistant dans le sol et relativement non volatil dans les conditions naturelles. Il ne semble pas que l'imidaclopride se bioaccumule. Il est possible de mesurer les résidus d'imidaclopride dans l'eau et dans le sol par chromatographie liquide haute performance (CLHP) ou par chromatographie en phase gazeuse couplée à la spectrométrie de masse (CG-SM).

L'imidaclopride est une neurotoxine nicotinoïde qui agit en bloquant de manière irréversible les récepteurs de l'acétylcholine. Tant les récepteurs d'acétylcholine des mammifères que ceux des insectes peuvent être bloqués par l'imidaclopride, mais les insectes sont plus sensibles que les mammifères. Les symptômes de l'empoisonnement par l'imidaclopride sont les suivants : chancellement, tremblement, immobilité et léthargie. La sensibilité à l'imidaclopride des organismes non visés varie. L'imidaclopride peut être fortement toxique pour les insectes bénéfiques, tels que les abeilles, et il est recommandé de réduire l'application d'imidaclopride dans les zones fréquentées par les abeilles. L'imidaclopride peut aussi être toxique pour les invertébrés aquatiques et les poissons juvéniles à de faibles concentrations, et il a des effets toxiques aigus sur les oiseaux. Des concentrations modérées de l'insecticide peuvent réduire la croissance des algues. Si l'insecticide est appliqué en postlevée, il ne devrait pas être toxique pour les cultures.

On n'a pas établi de recommandations canadiennes pour la qualité de l'eau visant la protection des utilisations de l'eau à des fins agricoles pour l'imidaclopride. Au moment de rédiger le présent document, les données dans ce domaine étaient insuffisantes.

Les données toxicologiques étaient suffisantes pour élaborer des recommandations provisoires relatives à l'imidaclopride pour la qualité des eaux visant la protection de la vie dulcicole et marine. La recommandation provisoire pour la protection de la vie dulcicole est fondée sur une CMEO après 28 jours (CE_{15}) de 2,25 µg m.a. L^{-1} pour la réduction de l'émergence chez le moucheron *Chironomus riparius*. Comme il s'agit d'un résultat d'une étude d'exposition chronique, un facteur de sécurité de 0,1 a été appliqué à la CMEO, ce qui donne la recommandation provisoire pour la qualité des eaux douces de 0,23 µg m.a. L^{-1}.

La recommandation provisoire pour la protection de la vie marine est fondée sur une CL_{50} après 48 heures de 13 µg m.a. L^{-1} pour les larves du moustique des marais salés *Aedes taeniorhynchus.* Comme il

s'agissait d'une étude d'exposition aiguë, et que l'imidaclopride n'est pas persistant dans l'eau, un facteur de sécurité de 0,05 a été appliqué à la valeur, ce qui donne la recommandation de 0,65 µg m.a. L^{-1}.

Ces recommandations visent à protéger tous les organismes aquatiques et tous les aspects de leurs cycles vitaux pour une période indéfinie d'exposition dans la colonne d'eau (CCME, 1991).

LIST OF ACRONYMS

a.i.	Active ingredient
CAS	Chemical Abstracts Service
CCME	Canadian Council of Ministers of the Environment
CWQG	Canadian Water Quality Guideline
DT	Dissipation time
EC_{50}	Median effects concentration
EEC	Estimated environmental concentration
GC-MS	Gas chromatography – mass spectrometry
HPLC	High performance liquid chromatography
K_d	Dissociation constant
K_{oc}	Soil (organic carbon) adsorption coefficient
K_{ow}	Octanol-water partition coefficient
LC_{50}	Median lethal concentration
LC-APCI-MS	Liquid chromatographic atmospheric pressure chemical ionization mass spectrometry
LC/MS	Liquid chromatography – mass spectrometry
LD_{50}	Median lethal dose
LOEC, LOEL	Lowest observed effect concentration, level
MATC	Maximum acceptable toxicant concentration
MDL	Method detection limit
MENVQ	Quebec Ministry of the Environment
MMPP	Manitoba's Management Plus Program
NOAEL	No observed adverse effects level
NOEC, NOEL	No observed effect concentration, level
PEI	Prince Edward Island
PMRA	Pest Management Regulatory Agency
Rfd	Reference dose
SPE	Solid phase extraction
SFPOH	Swiss Federal Office of Public Health
U.S. EPA	United States Environmental Protection Agency
U.F.	Uncertainty factor
UV	Ultra-violet

1. INTRODUCTION

This document describes the development of Canadian Water Quality Guidelines (CWQGs) for imidacloprid. CWQGs are numerical limits based on the most current, scientifically-defensible toxicological data. They are nationally-consistent benchmarks designed to protect, sustain and enhance the present and potential uses of a water body. CWQGs are developed under the auspices of the Canadian Council of Ministers of the Environment (CCME) and are used by provincial, territorial, and federal jurisdictions to evaluate water quality. Often, CWQGs form the scientific basis for site-specific guidelines or objectives used by managers in the various Canadian jurisdictions.

Included in this document are discussions of the chemical and physical properties of imidacloprid, production and uses, sources, and pathways for entry of imidacloprid into the Canadian environment. Available data on environmental fate and persistence are summarized. A comprehensive assessment of the toxicity of imidacloprid to aquatic life, as well as to crop plants and livestock, are presented to evaluate the environmental hazards posed by imidacloprid in water. Together, this information is used, in accordance with "A Protocol for the Derivation of Water Quality Guidelines for the Protection of Aquatic Life" (CCME 1991) to derive numerical water quality guidelines. Development of water quality guidelines for the protection of agricultural water uses (irrigation and livestock water) was also investigated, but there were insufficient data available to derive guidelines for these water uses according to the protocol (CCME 1993).

2. PHYSICAL AND CHEMICAL PROPERTIES

Imidacloprid has the molecular formula $C_9H_{10}ClN_5O_2$ (Figure 1), with a molecular weight of 255.7 g/mol (Table 1). In appearance, it consists of colorless crystals. The insecticide is quite water soluble even at the lowest solubility value reported (510 mg/L, see Table 1; Krohn 1989, reviewed in Mulye 1995) and could potentially leach to groundwater (Cohen et al. 1984, cited in Mulye 1995) or be transported in runoff (Mulye 1995). In the literature, some variation exists in reported vapour pressures for imidacloprid (Table 1), likely as a result of differences in the formulation of the imidacloprid-containing products. However, according to the comparatively low vapour pressure values (Kennedy and Talbert 1977), imidacloprid would be relatively non-volatile under field conditions (U.S. EPA 1975b, cited in Mulye 1995). Imidacloprid did not dissociate when titrated with either acid or base (Wohlers 1988, as reviewed in Mulye 1995). The octanol/water partition coefficient (log K_{ow}) of imidacloprid is 0.57 (Tomlin 2000), suggesting that it would not accumulate in aquatic biota (Krohn and Hellpointner 2002).
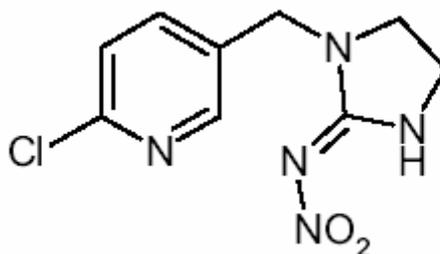
Figure 1. Chemical structure of imidacloprid (Krohn and Hellpointner 2002)

Table 1: Physical-Chemical Properties of Imidacloprid

Physical-Chemical Property	Imidacloprid	Reference(s)
Appearance	Colourless crystals	Tomlin 2000
Chemical name	**IUPAC:** 1-(6-chloro-3-pyridylmethyl)-*N*-nitroimidazolidin-2-ylideneamine; **CAS:** 1-[(6-chloro-3-pyridinyl)methyl]-*N*-nitro-2-imidazolidinimine	Tomlin 2000
Chemical formula	$C_9H_{10}ClN_5O_2$	Tomlin 2000
CAS number	138261-41-3	Tomlin 2000
Molecular Weight	255.7 g/mol	Tomlin 2000
Water Solubility	0.510 g/L @ 20°C 0.61 g/L @ 20°C	Krohn 1989, reviewed in Mulye 1995 Tomlin 2000
Melting Point	144°C 143.8°C	Tomlin 2000 Byrtus et al. 2002
Vapour Pressure	4×10^{-10} Pa @ 20°C 9×10^{-10} Pa @ 25°C 2×10^{-7} Pa @ 20°C	Tomlin 2000 EXTOXNET 1998
Henry's Law Constant (H)	1.0025×10^{-7} Pa m^3 mol^{-1} @ 20°C 2×10^{-10} Pa·m^3·mol^{-1} @ 20°C	Mulye 1995 Tomlin 2000
Partition Coefficient (K_{ow})	log P = 0.57 @ 21°C	Tomlin 2000
Soil Adsorption Coefficient (K_{oc})	262.0 210	Orme and Kegley 2003 Nemeth-Konda et al. 2002
Ultraviolet Absorption	Maxima at: 1) 211 nm (extinction coefficient = 1.378 X 10^4) 2) 269 nm (extinction coefficient = 2.0545 X 10^4)	Wilmes 1988, reviewed in Mulye 1995

3. ANALYTICAL METHODS

There are a number of different analytical methods that are currently being used for detection and measurement of imidacloprid. The methods differ in their applicability to different types of environmental media, and also in the detection levels that can be achieved.

Concentrations of imidacloprid in water and soil can be measured using a gas chromatography-mass spectrometry (GC-MS) technique (Vilchez et al. 1996). Samples of imidacloprid are transformed into a volatile compound through hydrolysis in a basic medium. Using a liquid-liquid extraction with chloroform will allow for sufficient extraction and pre-concentration of the hydrolysis product. The detection limits using this technique have been reported as 0.16 µg/L for water and 1 µg/kg for soil (Vilchez et al. 1996).

A second technique for determining concentrations in water uses a photochemical-fluorimetric method (Vilchez et al. 1998). This methodology is based on the conversion of imidacloprid to the fluorophore 1-(6-chloro-3-pyridyl-methyl)-2-(hydroxyimino)-3,4-didehydroimidalozolidene through photodegradation. Using this methodology, the detection limit was reported to be 0.7 µg/L, with a linear concentration range of 2.5-100 µg/L (Vilchez et al. 1998).

The GC-MS technique has also been utilized to determine the photocatalytic degradation of imidacloprid in industrial water (Aguera et al. 1998). Coupling GC-MS with liquid chromatographic atmospheric pressure chemical ionization mass spectrometry (LC-APCI-MS) allowed Aguera et al. (1998) to detect five degradation products, three of which were identified (i.e., chloronicotinic acid, chloronicotinic aldehyde, and 1-(6-chloro-3-pyridilmethyl)-imidazolidin-2-ona)) (Aguera et al. 1998). Overall, the LC-APCI-MS technique was considered to be a good and complementary method for monitoring imidacloprid but did not provide enough information for structural elucidation (Aguera et al. 1998).

High-performance liquid chromatography (HPLC) has also been used to measure levels of imidacloprid residues in water and soil (Baskaran *et al*. 1997). Baskaran *et al*. (1997) suggest that levels of imidacloprid cannot be determined directly using gas chromatography as a result of its thermolabile and polar N-nitroguanidinyl moiety. Volatility may also be increased as a result of the substitution of the acidic hydrogen of the NH at the 3-position of the imidazolidine ring (Baskaran et al. 1997). HPLC allows for the separation and detection of analytes using conditions that are milder than those utilized by gas chromatography. Using a reversed-phase HPLC with UV detection with a mobile phase of acetonitrile-water and either a solid-phase extraction (SPE) or liquid-liquid extraction method, the extract can be collected and evaporated using a rotary evaporator (Baskaran et al. 1997). By re-dissolving the extract in acetonitrile-water, the extract can be concentrated and then analyzed. Using this methodology, detection limits of 0.5 µg/L and 5 µg/kg were reported for imidacloprid in water and soil respectively (Baskaran et al. 1997).

The National Water Research Institute (NWRI) of Environment Canada uses a HPLC-MS-MS analysis method that allows quantification of imidacloprid in water samples to a level of 0.01 µg/L (Stoughton 2006). For this method, samples are injected directly into the LC-MS-MS

system, and the mobile phase contains aqueous acetonitrile and formic acid. For samples where matrix interference could occur with direct injection, NWRI has used solid-phase extraction in combination with an added internal standard. With this method, the imidacloprid in a 50 mL sample of water would have a detection limit of 0.1 µg/L and a quantification limit of 1.0 µg/L (Culp 2006). If the sample volume is increased to 1L, the quantification limit is lowered to 0.05 µg/L. A similar method using LC-MS-MS with solid phase extraction is described by König (1997), with a reported limit of quantification of 0.05 µg/L. Others have also used LC-MS methods and achieved detection limits of 0.02 µg/L (Byrtus et al. 2002) and 0.001 µg/L (Giroux 2003). The method by Giroux (2003) involved groundwater samples, which likely had fewer "interferences". Groundwater samples of 250 mL were used, and a liquid-solid extraction was conducted with a C-18 cartridge. The extract was eluted with acidified methanol containing diethylamine, then dry evaporated. The extract was then reconstituted in the mobile phase of the injection standard. Quantification was conducted with liquid chromatography coupled with mass spectrometry (LC-MS-MS) in multiple reaction monitoring (MRM) mode. A type C-18 chromatographic column was used (Isabelle Giroux, Ministère du Développement durable, de l'Environnement et des Parcs, personal communication, March 2007).

Generally, for analysis of imidacloprid in water samples, solid-phase extraction with use of LC-MS-MS methods appears to allow for the lowest detection limits.

4. PRODUCTION, USES AND SOURCES TO THE ENVIRONMENT

Imidacloprid, produced by Bayer CropScience Inc., is an insecticide active ingredient used to control sucking insects, such as aphids, leafhoppers, psyllids, thrips, whiteflies and beetles in agricultural crops, to control white grubs in lawns and turfgrass, as well as to control domestic pests such as fleas and cockroaches. Trade names for imidacloprid include Admire, Advantage, Confidor, Gaucho, Genesis, Impower, Intercept, Maxforce IC, and Merit (PMRA EDDENet Labels search, 2005). It is most commonly applied as a soil and foliage treatment, and as a seed dressing (Tomlin 2000).

Imidacloprid (in the form of *Admire 240 Flowable*) was sold and used for the first time in Canada in 1995 for the control of the Colorado potato beetle in eastern Canada (PMRA 2001). It is also approved for use on the Colorado Potato Beetle in tomato crops, to control the Spotted Tentiform Leafminer in apple crops, aphids in field lettuce, and aphid and whitefly control in greenhouse-grown plants. In addition, it is used as a seed treatment in Canola, rape, mustard and corn (PMRA 2001). It was first registered for use in the United States in 1994 (Cox 2001).

Crops to which imidacloprid is applied include: grains, maize, fruits, vegetables, potatoes, hops and turf (EXTOXNET 1998; Tomlin 2000). Typical application rates range from approximately 50 to 320 g/ha (PMRA EDDENet Labels search 2005). Application rates vary with plant type. For instance, the recommended application rate for the formulation Admire 240F to potato crops is 1.3 L Admire/ha (i.e., 312 g imidacloprid/ha) to soil, or 26-39 mL Admire/100 kg seed pieces (i.e., 6.2-9.4 g imidacloprid/100 kg seed pieces) (PMRA EDDENet Labels search 2005). For tomatoes, the application rate is 200 mL Admire/ha (i.e., 48 g imidacloprid/ha) to foliage, or 7-10 mL Admire/100 m row (i.e., 1.7-2.4 g imidacloprid/100 m row) to soil. Admire 240F may be applied to field lettuce foliage at the rate of 200 mL/ha (i.e., 48 g imidacloprid/ha) and to soil at the rate of 650 mL/ha (i.e., 156 g imidacloprid/ha).

Imidacloprid is also used in urban areas to control turf pests in household lawns, parks, athletic fields, golf courses, etc., and this type of use appears to be increasing. For example, in Ontario, licensed pesticide applicators have started using imidacloprid on lawns and turf as a replacement for diazinon, which was taken off the market for lawncare use in 2004 (John Struger, Environment Canada, personal communication, October 2006; Struger et al. 2002). For treatment of turfgrass to control white grubs, the recommended application rate is approximately 280 g a.i./ha (PMRA EDDENet Labels search 2005).

Formulations of imidacloprid are available as: a slurry for seed treatments, flowable concentrate for seed treatment, granule, wettable powder, soluble concentrate, suspension concentrate (flowable concentrate), water dispersible granules, and dustable powder (Tomlin 2000). Formulations of imidacloprid include other chemicals such as crystalline quartz silica (e.g., Merit 0.5G) and naphthalene (e.g., Leverage 2.7) (Cox 2001).

Imidacloprid is also used for flea control on domestic pets (Tomlin 2000). It is typically available as a solution that can be applied topically once a month to dogs and cats (PMRA

EDDENet Labels search 2005).   Products contain varying percentages of active ingredient depending on the weight of the animal to which it is intended to be applied.

Recently, imidacloprid has been investigated for potential use in controlling emerald ash borer, an exotic insect pest on ash trees in North America, through either direct stem injections or soil injections around the tree (Kreutzweiser et al. 2007b). This use is not yet registered in Canada.

Based on data available for 7 provinces, the total annual quantity of imidacloprid sold or used in Canada has been estimated at approximately 19,600 kg a.i. (Brimble et al. 2005). A large proportion of imidacloprid use is occurring in Atlantic Canada. For example, in 2003 a total of 7821 kg a.i. of imidacloprid were sold in New Brunswick and a total of between 1,000 and 9,999 kg a.i. were sold in Prince Edward Island in 2002 (CEI 2003; Brimble et al. 2005; PMU 2005). Nova Scotia reported considerably lower use than the other Atlantic provinces with 248 kg sold/used in 2003 (Brimble et al. 2005). Two recent changes in imidacloprid use have been noted for the western provinces. In British Columbia (BC), a major increase in imidacloprid useage for flea control occurred during the past decade. While imidacloprid was not an active ingredient in flea control in 1991 or 1995, by 1999 it comprised 62% of externally applied products. This correlates to 96.27 kg imidacloprid sold by veterinarians in BC in 1999 (ENKON Environmental Limited 2001). In Alberta, imidacloprid is expected to replace lindane, which is being phased out (Byrtus et al. 2002). This could result in increased sales in future years. In 2003, imidacloprid sales/use in BC were 528 kg a.i., and Alberta sales reported for 1998 were 9.5 kg (Brimble et al. 2005; Richard Casey, Alberta Environment, Edmonton, Alberta, personal communication). Data from Saskatchewan show that imidacloprid is currently among the top 10 pesticides in use in that province, however, actual estimates of quantities used or sold were not available (Sam Ferris, Saskatchewan Environment, Regina, personal communication, 2004). Information on Manitoba's imidacloprid application gives an estimate of imidacloprid utilization over 25 531 ha agricultural land (MMPP 2003). Quantities used in Manitoba in 2003 are estimated at 5939 kg a.i. (Brimble et al. 2005). In Ontario, the amount of imidacloprid used in 2003 for agricultural uses was approximately 527 kg (McGee et al. 2004; Brimble et al. 2005). The total quantity of imidacloprid used in Ontario is likely considerably larger if you account for uses such as flea and tick control on pets and applications in greenhouses and on turfgrass.

5. CONCENTRATIONS IN CANADIAN WATERS

Shortly after imidacloprid was registered for use in Canada in 1995, groundwater studies in Ontario and Quebec in 1996 and 1997 suggested that imidacloprid was not leaching through the soil (PMRA 2001). In a study conducted between 1996 and 1998 that examined 29 groundwater samples collected on three separate occasions from sites in PEI adjacent to potato fields where imidacloprid had been used, detection of imidacloprid occurred in only one sample, at trace concentrations (Krohn and Hellpointner 2002; PMRA 2001). However, no detection limit is reported for this analysis. In 1999 and 2000, imidacloprid was below detection limit in Alberta surface water samples (0.02 – 0.05 µg/L) (Byrtus et al. 2002). However, imidacloprid was not being used as a seed treatment at the time of sampling, and thus, was not expected to be detected in Alberta surface waters.

Past studies in the Atlantic provinces have shown concentrations of imidacloprid in runoff (PMRA 2001). For example, runoff collected from potato farms in PEI following rainfall events in 2001 and 2002 had concentrations ranging from below the detection limit of 0.5 µg/L to 11.9 µg/L (Denning et al. 2004). However, a more recent monitoring effort in Prince Edward Island, New Brunswick and Nova Scotia, conducted throughout 2003 and 2004, did not detect imidacloprid in runoff. The more recent study focused on sampling during periods that were high risk for surface runoff events. Analysis included 45 samples from PEI, 42 samples from New Brunswick and 18 samples from Nova Scotia. With detection limits of 2.0 µg/L in Nova Scotia and New Brunswick and 1.0 µg/L in PEI, imidacloprid was not detected in any of the water samples from this study (Murphy and Mutch 2005). Sampling of surface waters in the Atlantic provinces was also conducted from 2003 to 2005 (Murphy et al. 2006). During this period, imidacloprid was not detected in any of the 82 samples analyzed from PEI (detection limit of 0.2 µg/L), nor any of the 48 samples analyzed from Nova Scotia. However, imidacloprid was detected in two out of 57 samples from New Brunswick surface waters, with a maximum concentration of 0.3 µg/L (Murphy et al. 2006; Environment Canada 2006). Similarly, in a study that looked at imidacloprid in both runoff from potato fields and in surface water of Black Brook, New Brunswick from 2003 to 2005, maximum spike concentrations during rain events were nearing 0.3 µg/L (Hewitt 2006).

Studies in Ontario on tile drains found low concentrations of imidacloprid in runoff water (PMRA 2001). In a 2000-2001 study conducted by Environment Canada, the Ontario Ministry of Environment, and the City of Toronto that looked at agricultural and lawn care pesticides in the Don and Humber River watershed, imidacloprid was not detected (at a detection limit of 1 µg/L), but traces were observed in two samples (John Struger, Environment Canada, personal communication, October 2006). Out of 167 samples collected from approximately 40 different sites in surface waters of Ontario in 2004, no concentrations of imidacloprid above the analytical limits of quantitation (13 µg/L) or detection (4 µg/L) were found (John Struger, Environment Canada, personal communication, October 2006; Environment Canada 2006). It should be noted that this detection limit was high. There were 15 of the 167 samples, primarily from an urban stream and a fruit belt stream, that showed trace peaks but could not be reliably identified as imidacloprid (John Struger, Environment Canada, personal communication, October 2006).

A report from the Ministère de l'Environnement states that imidacloprid and its metabolites were detected in 35% of groundwater samples collected near potato fields throughout Quebec (Giroux 2003). Samples were collected from shallow wells located close to the treated fields, and therefore represented a worst-case scenario. Detection limits for imidacloprid, and its metabolites imidacloprid-urea, imidacloprid-guanidine and imidacloprid-olefin were 0.001, 0.0009, 0.0008, and 0.0007 µg/L, respectively. According to this report, the maximum concentration of imidacloprid detected was 6.4 µg/L, and maximum concentrations of the metabolites imidacloprid-urea, imidacloprid-guanidine and imidacloprid-olefin were 0.018, 0.4, and 0.0023 µg/L, respectively (Giroux 2003).

Low levels of imidacloprid have been detected in groundwater in various locations through the United States. For example, in a groundwater study conducted by Bayer (1997-98) in California, concentrations of imidacloprid found in groundwater ranged from <0.1 to 1 µg/L (Bacey 2000). The specific locations sampled were not reported. PMRA (2001) also reports that imidacloprid was detected in groundwater in the states of New York and Michigan subsequent to 3 years of use in those areas.

In summary, it appears that imidacloprid is not being detected in the majority of Canadian waters that are close to areas of application. However, it should also be noted that some monitoring studies have used analytical methods with high detection limits that are likely above the threshold for potential adverse effects. At the limited number of sites where imidacloprid has been detected in surface and groundwaters, these typically represent worst-case scenarios (e.g., surface water concentrations during rainfall events, or groundwater concentrations in shallow wells). Some of the higher concentrations that have been detected are within the range of imidacloprid concentrations that have been observed to cause adverse effects on non-target aquatic organisms.

6. ENVIRONMENTAL FATE AND BEHAVIOUR

Much of the following information on environmental fate and behaviour is summarized in Table 2 (below).

6.1 Fate in Air

Imidacloprid's low vapour pressure (Table 1) results in a very low potential for volatilization. Therefore, imidacloprid is most likely to be found in air during and immediately after spraying on crops, where it will exist primarily as an aerosol, with very little occurring in a gaseous state. Imidacloprid is rapidly photodegraded and transformed by photochemical radicals and therefore is not likely to persist in air (Krohn and Hellpointner 2002).

6.2 Fate in Soil

6.2.1 Persistence

In a regulatory note on imidacloprid from the Pest Management Regulatory Agency (PMRA), it is noted that the dissipation time, DT_{50}, for imidacloprid (i.e., the time required for half of it to dissipate) is in the order of 1-2 years (PMRA 2001). There is some variation in the literature on reported dissipation times and half-lives, and the potential for accumulation. In a review of six aerobic soil metabolism studies that examined loss of imidacloprid from eight different soil types, Sabbagh et al. (2002) noted that calculated half-lives ranged from 83 days to greater than a year. Studies determining dissipation times indicate that imidacloprid exhibits strong persistence in soil under standardized laboratory conditions and more variable persistence under a range of field conditions (Krohn and Hellpointner 2002). For instance, the geometric mean of 5 studies, conducted under laboratory conditions at $20^{\circ}C$, produced a DT_{50} of 156 days (SD=40) (Krohn and Hellpointer 2002). The DT_{50} determined from 11 bare soil field trials in Northern and Southern Europe was 96 days (SD=38), after mathematical adjustment to equivalent temperature ($20^{\circ}C$) (Schad 2001 as reviewed in Krohn and Hellpointer 2002). Based on these studies, Krohn and Hellpointner (2002) suggest that there is little potential for accumulation as a result of repeated annual application. However, lengthier DTs have also been determined from field studies. Mulye (1996) reviewed a two-year field lysimeter investigation in Germany using imidacloprid applied to seed potatoes (Hellpointner 1994 a,b) and from the study results calculated a DT_{50} of approximately 2 years, indicating that the compound would persist in soil. Values for DT_{90} are high, in several cases exceeding 450 or even 1000 days, suggesting that the potential exists for carry-over of imidacloprid residues one year after application (Scholz 1991, as reviewed in Mulye 1997a; Philpot and Yen 1998a, 1998b, 1998c, and 1998d as reviewed in Mulye 1999). This residue persistence was demonstrated in a cornfield dissipation study in Minnesota (Rice et al. 1991 as reviewed by Mulye 1995).

There is evidence that dissipation times are reduced when imidacloprid is applied to cropped, rather than fallow, fields (Scholz and Spiteller 1992; Krohn and Hellpointer 2002). When

imidacloprid was applied to bare soil at several sites in northern Europe, the mean DT_{50} was 174 days, while cropped conditions reduced it to 83 and 124 days (Krohn and Hellpointer 2002). It is likely that persistence in vegetated areas is decreased through plant (Rouchaud et al. 1994) and microbial (Capri et al. 2001; Krohn and Hellpointner 2002) uptake and metabolism. Rouchaud et al. (1994) applied imidacloprid as a seed treatment to a sugar beet field and demonstrated that high imidacloprid concentrations were observed in the leaves of these plants. These concentrations were highest in the young plants and progressively decreased with time, as plant growth and metabolism diluted the concentrations. This study also showed that organic matter significantly affects the rate of biodegradation for imidacloprid. Fields in which cow manure had been newly applied showed much greater persistence of imidacloprid levels, while biodegradation was enhanced in fields with aged manure (Rouchaud et al. 1994). It has been proposed that newly organic fertilizer protects the chemical from microbial degradation, whereas with age, the organic fertilizer enhances the soil microbial activity that metabolizes imidacloprid (Rouchaud et al. 1996). Application of dissolved organic carbon appears to reduce imidacloprid sorption by competing with the pesticide for sorption sites on the soil surface (Flores-Cespedes et al. 2002).

Research has demonstrated that granule or liquid formulations do not affect imidacloprid's persistence or metabolism in soil (Sarkar et al. 2001).

In summary, persistence of imidacloprid in soil is affected by various factors, including temperature, organic matter of the soil, and whether the field is cropped or not. The time required for 50% of the field-applied imidacloprid to dissipate (DT_{50}) can range anywhere from approximately 80 days to 2 years. Assuming typical DT_{50}s of 1 to 2 years, PMRA has classified imidacloprid as persistent in soil based on the classification scheme of Goring et al. (1975).

6.2.2 Sorption/Desorption

Adsorption is the main fate process for imidacloprid in soil (Sabbagh et al. 2002). Imidacloprid has a medium (Tomlin 2000; Krohn and Hellpointer 2002) to high (Krohn and Hellpointer 2002) sorption tendency for soil. Sorption intensity for imidacloprid and its metabolites is influenced by soil type and depends largely on organic carbon content (Cox et al. 1998; Capri et al. 2001). Freundlich sorption coefficient values (K_f) increase with enrichment of soil organic carbon content and with residence time (Cox 1997; Cox et al. 1997; Capri et al. 2001). The K_f is also correlated, to a less extent with the cation exchange capacity of the soil and the % clay content (Sabbagh et al. 2002). Soil sorption is also influenced by the soil:solution ratio (Cox et al. 1998). For instance, imidacloprid exhibits lower sorption behavior when the soil contains a higher water content due to its high solubility. Sorption-desorption is also concentration-dependent, with higher sorption rates when there is a lower initial concentration of imidacloprid present (Cox et al. 1998). At higher initial concentrations of imidacloprid, sorption is low and desorption is high, therefore there is a greater potential for mobility with increasing concentration (Cox et al. 1997; McCall et al. 1981, as cited in Mulye 1997a). Cox et al. (1997) determined that the order of sorption tendency for imidacloprid and several of its main transformation products was as follows: imidacloprid-guanidine> imidacloprid-guanidine-olefin> imidacloprid> imidacloprid-urea. This order of sorption was true regardless of the soil type investigated.

Studies have demonstrated that the sorption-desorption process of imidacloprid and its metabolites is hysteretic (i.e., a lag time occurs before absorption-desorption response to variable environmental conditions that would be expected to affect equilibrium sorption concentrations), particularly at high soil:solution ratios (Cox et al. 1997; Cox et al. 1998; Celis and Koskinen 1999). This behavior can be attributed to a portion of the sorbed compound that is bound irreversibly to soil surfaces (Cox et al. 1997; Cox et al. 1998; Celis and Koskinen 1999). The desorption-resistant fraction increases with increased residence time, as well as under acidified conditions (Celis and Koskinen 1999). Trapping in the micropores of soil clay minerals and organic matter may result in a desorption resistant fraction (Krohn and Hellpointer 2002).

6.2.3 Runoff and Leaching

Based on the high water solubility of imidacloprid (see Table 1) and its persistence, PMRA (2001) considers imidacloprid to have 'high' leaching potential.

In a survey of test wells conducted in 1998, imidacloprid and several of its transformation products were detected in groundwater from one location in Michigan and one in New York State at 0.20 μg/L and 1.90 μg/L, respectively (Mulye 1999). Felsot et al. (1998) noted that imidacloprid can leach to depths of at least 105 cm when irrigation conditions are unmatched to water evapotranspiration rates so that the soils become saturated or near-saturated.

However, there is evidence to suggest that, if used correctly (e.g., at recommended rates, without irrigation, and when heavy rainfall is not predicted), imidacloprid does not characteristically leach into the deeper soil layers despite its high water solubility (Rouchaud et al. 1994; Tomlin 2000; Krohn and Hellpointner 2002). In a series of field trials conducted by Rouchaud et al. (1994, 1996), in which imidacloprid was applied to sugar beet plots, it was consistently demonstrated that no detectable leaching of imidacloprid to the 10-20 cm soil layer occurred. Imidacloprid was applied to a corn field in Minnesota, and no imidacloprid residues were found in sample column segments below the 0 - 15.2 cm depth segment (Rice et al. 1991, as reviewed in Mulye 1995).

Column leaching studies have also shown that leaching by imidacloprid is typically low (Tomlin 2000; EXTOXNET 1998; Krohn and Hellpointner 2002). Krohn and Hellpointner (2002) conducted a column leaching study simulating a worst case scenario flooding event. The concentrations of imidacloprid in the leachate from these columns were below the limit of quantification (<1.0 μg/L).

Sorption-desorption dynamics (see above) explain the immobility of imidacloprid despite its high water solubility (Oi 1999). Oi (1999) proposed that the elevated sorption potential of the chemical at lower concentrations will progressively inhibit any leaching with increasing soil depth. Leaching will also be inhibited by the desorption hysteresis process that is characteristic of imidacloprid. Krohn and Hellpointner (2002) attributed the fact that imidacloprid was kept in the upper portion of the soil to the presence of the desorption resistant fraction. Moreover, sorption of imidacloprid to soil increases with time, lessening leaching potential of imidacloprid

residue. A laboratory soil column investigation, using C^{14}- labelled imidacloprid aged 30 days prior to application, found only 0.14% of applied radioactivity in leachate, and nearly 50% in the top layer of soil (Fritz and Brauner 1988, reviewed in Mulye 1995). In a two-year field experiment in Germany, ^{14}C- labelled imidacloprid was applied as a seed potato treatment in two lysimeters (Hellpointner 1994a,b, reviewed in Mulye 1996). Because little water was added within the first month following application, the imidacloprid was able to age during this time. Total radioactivity recovered from leachate collected over the two years was less than 0.05% of that applied. An experiment that utilized batch and column leaching tests confirmed that a change occurs in the sorption process of imidacloprid with time, leading to stronger sorption to the soil and a lower potential for mobility (Oi 1999).

These factors should be taken into account when evaluating leaching and runoff potential. Imidacloprid is at higher risk for runoff and leaching immediately after application, particularly during rainfall events. Recent application would mean a higher concentration of the chemical, as well as less time for sorption.   Fresh application of dissolved organic carbon also increases the leaching potential of imidacloprid in soil by competing with the pesticide for sorption sites on the soil surface (Flores-Cespedes et al. 2002). According to Cox et al. (1998), based on the sorption-desorption dynamics of imidacloprid, low application rates that favor a high soil:solute ratio should decrease the risk of leaching (Cox et al. 1998).

6.3 Fate in Water

The persistence of imidacloprid in the aqueous environment depends on environmental factors including exposure to light, pH, temperature and microbial community, in addition to application rate and formulation.

Imidacloprid exhibits an aqueous photolysis half-life of approximately 4 hours, taking into consideration variable light frequencies over the course of a day (Tomlin 2000; Krohn and Hellpointner 2002). Imidacloprid's major photolysis breakdown products in water are 6-chloro-nicotinaldehyde, 6-chloro-N-methylnicotinacidamide, 6-chloro-3-pyridyl-methylethylendiamine, imidacloprid urea, 6-hydroxynicotinic acid, and a minor breakdown product is imidacloprid guanidine (Bacey 2000).   Wamhoff and Schneider (1999) state that 6-chloro-3-pyridyl-methylethylendiamine is the major photolysis breakdown product of imidacloprid in water. The photolysis rate fits first-order reaction kinetics (Moza et al. 1998).

The rate of hydrolysis of imidacloprid increases with temperature (Zheng and Liu 1999). This hydrolysis has also been shown to fit a first-order kinetic relationship (Zheng and Liu 1999). The main reaction product identified as a result of hydrolysis is 1-[(6-chloro-3-pyridinyl)methyl]-2-imidazolidone (Zheng and Liu 1999).

Degradation rates of imidacloprid, in darkness and without microbial action, vary with pH, but studies have yielded conflicting results. Sarkar et al. (1999) determined that the mean half-life for imidacloprid increased with alkalinity, increasing from 36.3 days at pH 4 to 41.6 days at pH 9. In contrast, several sources indicate that imidacloprid more readily degrades under alkaline conditions (Yoshida 1989, as cited by Mulye 1995; Zheng and Liu 1999; U.S. EPA 2005). An

experiment conducted by Yoshida (1989 as cited by Mulye 1995) determined that, while no hydrolysis products were detected at pH 5 and pH 7 at any sampling intervals, imidacloprid transformed slightly at pH 9, with a calculated half-life of 346.5 days. The U.S. EPA lists imidacloprid as stable at pH 5 and pH 7, with a half-life of 355 days at pH 9, supporting enhanced degradation (U.S. EPA 2005). Based on these results, the compound is stable to hydrolysis at environmentally relevant pH.

In the aqueous environment, imidacloprid is also metabolized by microorganisms. DT_{50} values of 30, 130 and 160 days have been calculated in the absence of light and with variable sediment (Krohn and Hellpointner 2002). Combining metabolic and photolytic processes reduces the DT_{50} values to the range of days (Heimbach and Hendel 2001, cited in Krohn and Hellpointner 2002). Spiteller (1993) examined the degradation of imidacloprid in a 30-day laboratory study using water and sediment collected from a pond. Radio-labeled imidacloprid was applied to the water at an initial rate of 680 µg/L. By the end of the exposure, 67.6% of the radioactivity remained in the water column, with 64.0% as parent imidacloprid and 3.6% as metabolites. In the sediment, 29.3% of the radioactivity was detected, with 20.4% as extractable parent imidacloprid, 0.7% as extractable metabolites, and 8.2% as bound residues. There was also 0.7% of the radioactivity detected as CO_2, and <0.1% as other volatile metabolites. Therefore, after 30 days, not much biodegradation had occurred, and a first half-life (DT_{50}) for imidacloprid of 129 days was estimated (Spiteller 1993). A similar study with pond water and sediment was conducted by Henneböle (1998) to determine the influence of exposure to either artificial light (xenon lamp) or sunlight on degradation of imidacloprid. Applied to the water at an initial rate of 620 µg/L, the half-life of the radio-labelled imidacloprid was estimated at less than 14 days. After 21 days, 5.8% of the imidacloprid had been mineralized in the exposure to sunlight, and 9.8% had been mineralized in the exposure to xenon light. Residues bound to the sediment at 21 days accounted for 67.6% of the applied radioactivity in the sunlight exposure, and 47.7% in the xenon light exposure (Henneböle 1998). Anaerobic metabolism in the absence of light was measured at a DT_{50} of 27 days (Krohn and Hellpointner 2002).

Mesocosm studies suggest that under natural conditions, dissipation times are even shorter. Moring et al. (1992) determined a half-life for imidacloprid in the water column of 1.4 days in an outdoor microcosm study with four surface applications of the active ingredient, each spaced two weeks apart. Imidacloprid did not appear to persist in the sediment either, with residues below detection limits one month after the last application (Moring et al. 1992). In another mesocosm study, Confidor SL 200 (containing 17.3% imidacloprid) was applied twice, three weeks apart, to artificial ponds at concentrations ranging from 0.6 to 23.5 µg a.i./L (Ratte and Memmert 2003). Three weeks after the initial application, and before the second application, imidacloprid was detected in the ponds at 12 to 20% of the nominal concentrations. The calculated mean DT_{50} for imidacloprid in water was 8.2 days (range from 5.8 to 13.0 days). In the pond sediments, the highest concentrations were measured 7 days after the second application. By 56 to 70 days after the second application, concentrations in the sediment were below the limit of quantitation of 7 µg/kg. The average DT_{50} for the whole pond system (water and sediment) was 14.8 days (Ratte and Memmert 2003).

The formulation of the imidacloprid product further influences persistence in the aquatic environment. Higher half-life values were found for powder formulations than for liquid (Sarkar et al. 1999). Persistence also increased with increase of application rate (Sarkar et al. 1999).

In the marine environment, imidacloprid that was used in the Willapa Bay estuary in western Washington State to control the burrowing shrimp was widely dispersed to extremely low levels soon after application (Felsot and Ruppert 2002). Residues were measured in water at a distance of 152 m from the application point within a matter of minutes, due to dispersion from tidal flow. Imidacloprid dissipated from the water column very quickly, with very low levels (0.6 µg/L) detected in only one of three blocks sampled the day after application (detection limit was 0.6 µg/L). Imidacloprid was much more persistent in sediment, with concentrations of 3.86 to 6.33 µg/kg still detected in sediment residues at 28 days after application (detection limit was 2.5 µg/kg) (Felsot and Ruppert 2002).

6.4 Transformation Products

The transformation products of imidacloprid in aerobic soil typically include1-[(6-chloro-3-pyridinyl)methyl]-2-imidazolidinone (imidacloprid urea), 6-chloronicotinic acid and 6-hydroxynicotinic acid (Rouchaud et al. 1996) (see Figure 2), which will lead to the formation of carbon dioxide. For instance, depending on the soil type, imidacloprid labelled with imidazolidin-^{14}C had a maximum mineralization to CO_2 of 8.8% or 14% after incubation for 12 weeks (Anderson 1995, as reviewed in Mulye 1997a). With anaerobic sediment/water conditions and no light exposure, the only major transformation product is des-nitro imidacloprid (Heim et al. 1996 as reviewed in Mulye 1997a).

Phototransformation and biotransformation in the aqueous environment appear to be significant transformation routes (Stevens and Halarnkar 1996, as reviewed by Mulye 1997a). In water, phototransformation accelerates degradation, producing major metabolites imidacloprid urea (Wamhoff and Schneider 1999) and 1-[(6-chloro-3-pyridinyl)methyl]-1H-imidazol-2-amine (imidacloprid-guanidine olefin) (Anderson 1991, as reviewed in Mulye 1995). A study comparing imidacloprid degradation under non-sterile, aerobic conditions with and without light exposure found that the presence of light decreased the half-life from 331 days to 4.19 days (Stevens and Halarnkar 1996, reviewed by Mulye 1997a). Under sterile, aerobic conditions, the half-lives were 499 and 28.4 days with and without light exposure respectively (Stevens and Halarnkar 1996, reviewed by Mulye 1997a). The major transformation products resulting from incubation under the environmentally relevant non-sterile conditions and light exposure were des-nitro imidacloprid, imidacloprid urea, 6-chloronictinic acid and an unknown compound. Under dark, non-sterile conditions, des-nitro and another unidentified compound were produced. Due to the relatively low concentrations of the unidentified metabolites, these products were not expected to be environmentally significant and the PMRA waived the requirement for aquatic toxicity investigations (Mulye 1997b). Hydrolysis is not expected to be a major route of imidacloprid transformation as it appears to be stable at ecologically relevant pH (Yoshida 1989, as reviewed by Mulye 1997a).

The des-nitro imidacloprid produced under dark, anaerobic conditions has been found to be more persistent than its parent compound (Fritz and Hellpointner 1991, as reviewed in Mulye 1995). Both des-nitro imidacloprid and imidacloprid urea are highly water soluble, with solubilities of 180 – 230 g/L and 9.3 g/L at 20°C, respectively (Krohn 1996a, 1996b, as reviewed in Mulye 1997a).

6-chloronicotinic acid   6-hydroxynicotinic acid

Desnitro-imidacloprid   Imidacloprid urea
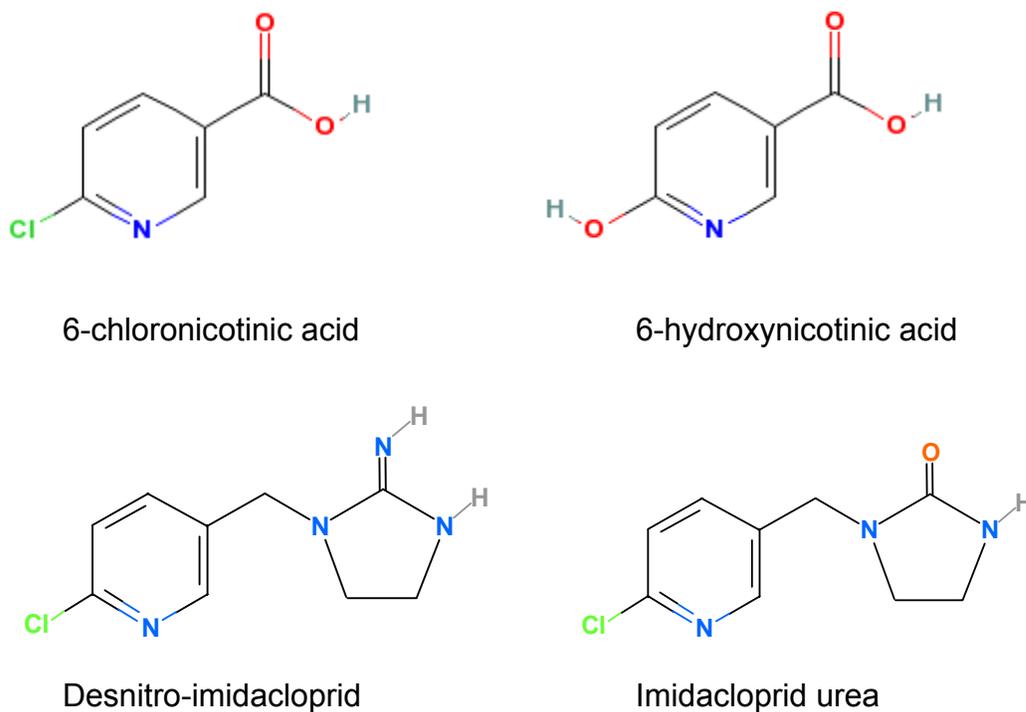
Figure 2. Typical degradates of imidacloprid
(Structures adapted from: PubChem Substance 2005; Bacey 2000; Shimomura et al. 2002)

6.5 Biotransformation and Bioaccumulation

Imidacloprid has a log K_{ow} of 0.57 (Krohn and Hellpointner 2002; Tomlin 2000) indicating a low potential for accumulation in aquatic species according to U.S. EPA (1975a). Imidacloprid does not appear to bioaccumulate in biota (Krohn and Hellpointner 2002; PMRA 2001). The transformation products des-nitro imidacloprid (NTN 33823; log K_{ow} < -2 for pH between 4 and 7 and log K_{ow} = -1.7 at pH = 9; Krohn 1996a, as reviewed in Mulye 1997a) and imidacloprid urea (NTN33519; log K_{ow} = 0.46; Krohn 1996b) should also have low bioaccumulation potential.

In animals, imidacloprid undergoes rapid passage through the body. It is rapidly absorbed in almost its entirety by the gastrointestinal tract and quickly eliminated via urine and feces (Tomlin 2000). In a study involving oral administration of imidacloprid to rats, 96% of the compound was eliminated after 48 hours, with only approximately 15% eliminated as the unchanged parent product (Tomlin 2000). Metabolism proceeds through hydroxylation to the imidazolidine ring, hydrolysis to 6-chloronicotinic acid, loss of the nitro group with formation of the guanidine and conjugation of the 6-chloronicotinic acid with glycine (Tomlin 2000). All the metabolites that were detected in the organs of farm animals contained the 6-chloronicotinic acid moiety.

In plants, imidacloprid is metabolized via loss of the nitro group, hydroxylation at the imidazolidine ring, hydrolysis to 6-chloronicotinic acid, and formation of conjugates (Tomlin 2000). The main metabolites detected in plants were imidacloprid-guanidine olefin and imidacloprid-guanidine (Tomlin 2000).

Table 2: Laboratory Studies on Fate Processes of Imidacloprid and its Transformation Products

Characteristic	Value	Comments/Interpretation	Reference
Hydrolysis of Imidacloprid	Half-life = ~1 year (pH=9; T=25°C)	Hydrolysis-resistant at pH 5 and 7	U.S.EPA 2005
Phototransformation of Imidacloprid on soil	Half-life = 38.9 days (T=25±2°C)		Tomlin 2000
Phototransformation of Imidacloprid in water	Half-life = ~4 hours	Rapid phototransformation in water	Tomlin 2000; Krohn and Hellpointner 2002
Aerobic biotransformation of Imidacloprid in soil	Half-life = 188 to 997 days (T=20°C)	Persistent in various soils under aerobic conditions	Mulye 1995, based on classification scheme of Goring et al. 1975
Anaerobic biotransformation of Imidacloprid in sediment/water system	Half-life = 27 days (T=25°C)	Slightly persistent under anaerobic conditions. The major transformation product (des-nitro imidacloprid) would be persistent	Reviewed in Mulye 1995, based on classification scheme in McEwen and Stephenson 1979
Adsorption/ Desorption coefficients of Imidacloprid	K_{f-oc} = 412	Hysteretic process, particularly at high soil:solution ratios.	Cox et al. 1997
Adsorption/Desorption coefficients of some major imidacloprid metabolites	Imidacloprid-urea: K_{f-oc} = 203 Imidacloprid-guanidine: K_{f-oc} = 2740 Imidacloprid-guanidine-olefin: K_{f-oc} = 3200		Cox et al. 1997
Leaching	Low leaching	Low mobility at low application	Oi 1999; Krohn and

	potential when applied correctly.	rate and with aging.	Hellpointer 2002
Soil Dissipation Times	DT_{50} = 426 days Studies: DT_{50} in bare soil = 210 - 456 days DT_{90} in bare soil $\geq$ ~1095 days	Estimated ~54 – ~78% carry-over of imidacloprid residues one year after application; imidacloprid "very persistent" in bare soil.	PMRA 2001 Reviewed in Mulye 1999.
	DT_{50} in soil planted with potatoes = 63 - 178 days DT_{90} in soil planted with potatoes >1099 days	Estimated ~ 43-60% carry-over of imidacloprid residues one year after application; "very persistent" in soil planted with potatoes	
	DT_{50} in soil with turf = 16 - 21 days DT_{90} <456 days	Estimated ~ 18% carry-over of imidacloprid residues one year after application; slight persistence in turf using DT_{50} values, "higher persistence" using DT_{90} values and "potential for carry-over"	

7. GUIDELINES FROM OTHER JURISDICTIONS

Quebec, in 1998, derived interim guidelines for imidacloprid, in accordance with the Ministère de l'Environnement du Quebec protocol for water quality criteria (MENVIQ 1990), which largely follows the U.S. EPA protocol (Stephan et al. 1985). This method utilizes the Final Acute Value (FAV) as an estimate of the concentration corresponding to a cumulative probability of 0.05 of the LC_{50} values for genera with acceptable acute data (Stephan et al. 1985). Where the geometric mean of the acute toxicity values for a sensitive species of concern results in a lower value than this estimate, the Species Mean Acute Value is used instead. The FAV for imidacloprid was determined as 85,090 µg/L, based on the geometric mean of two *Daphnia magna* LC_{50} values. The acute guideline was calculated by dividing the FAV by an interspecies sensitivity factor of 5 (because salmonids are represented), and by a factor of 2 to move from a 50% mortality to a low percentage of mortality. The acute guideline is 8509 µg/L. Studies that reported chronic values were not readily available, so the Chronic Aquatic Life Criteria was determined through application of an interspecies sensitivity factor of 5 and a standard acute/chronic ratio of 45 to the FAV, resulting in an interim chronic criteria value of 378 µg/L (Guay 1998). It should be noted that data for more sensitive species such as insects (e.g., chironomids, mayflies) or ostracods were not available in the dataset used to derive the Quebec criteria.

The U.S. EPA (1995) has established a chronic reference dose (RfD) of 0.057 mg/kg/day, for use in human health risk assessment. This RfD is generated from a study that determined the no adverse effects level (NOAEL) for male rats exposed to dietary intake of imidacloprid. The RfD was calculated by dividing the NOAEL by an uncertainty factor (UF) of 100.

In Switzerland, imidacloprid was registered by the Federal Office of Public Health (FOPH) as a Toxic Substance Class 3 (SFOPH, 2005) within the Information System for Dangerous and Ecologically Relevant Substances (IGS). Switzerland's Toxic Substances classification is based on acute oral threshold levels, usually determined on rats. The acute oral threshold for a Toxic Substance Class 3 is 50-500 mg/kg (SFOPH 2005).

8. TOXICITY TO AQUATIC ORGANISMS

8.1 Mode of Action

Imidacloprid is a systemic insecticide (Tomlin 2000) meaning that it is taken up by plants, primarily through the roots, and transported within the vascular system of the plant where it can affect plant-feeding pests. Imidacloprid acts as a nicotinic acetylcholine (Ach) agonist (Song and Brown 1998). It binds irreversibly to the nicotinergic receptors in postsynaptic nerves, preventing acetylcholine from binding. Imidacloprid is not degraded by acetylcholinesterase (Hovda and Hooser 2002). This blockage leads to the accumulation of acetylcholine, which ultimately results in paralysis and death. It has a higher binding affinity for insect nerve receptors when compared to mammalian receptors (Matsuda et al. 2000).

Formulations of imidacloprid include other chemicals such as crystalline quartz silica (e.g., Merit 0.5G) and naphthalene (e.g., Leverage 2.7) (Cox 2001). These chemicals have associated toxicological characteristics. Unless noted, the following summary of toxicity tests considers technical-grade imidacloprid only.

8.2 Toxicity to Freshwater Life

8.2.1 Fish

While acute toxicity to adult fish occurs at relatively higher concentrations (over 80,000 μg/L), the early life stages of fish exhibit much higher sensitivity (Cox 2001).

With newly fertilized rainbow trout eggs (*Oncorhynchus mykiss),* Cohle and Bucksath (1991) conducted a series of 60-day trials that explored hatching, survival and growth from egg to juvenile in the rainbow trout. From these experiments, NOECs were determined at >19,000 μg a.i./L for hatching and survival (Cohle and Bucksath 1991; Gagliano 1992). The NOEC for growth and survival was determined as low as 1200 μg a.i./L, with a LOEC of 2300 μg a.i./L (Cohle and Bucksath 1991; Gagliano 1992). For juvenile rainbow trout, a 96-h survival NOEC of 50,000 μg a.i./L has been demonstrated, with an LC_{50} of 211,000 μg a.i./L (Grau 1988b) A behavioural effects study produced a NOEC of 42,000 μg a.i./L in juvenile rainbow trout, and a LOEC of 64,000 μg a.i./L (Bowman and Bucksath 1990b).

Grau (1987b) investigated survival in the Golden Orfe (*Leuciscus idus melanotus*), reporting a NOEC of 178,000 μg a.i./L and an LC_{50} of 237,000 μg a.i./L, which suggests lower sensitivity than that observed for the rainbow trout.

Based on mortality, adult bluegill sunfish (*Lepomis macrochirus*) had a NOEC of 25,000 μg a.i./L and an LC_{50} of 105,000 μg a.i./L (Bowman and Bucksath 1990a). No early life stage studies were found for the golden orfe or the bluegill sunfish.

In summary, freshwater fish do not appear to be particularly sensitive to imidacloprid, with toxic effects occurring at concentrations that are at least two orders of magnitude higher than imidacloprid concentrations that have been measured in Canadian waters.

8.2.2 Aquatic Invertebrates

Imidacloprid induces toxic effects at very low levels in a number of aquatic organisms.

Toxicity values for the midge, *Chironomus tentans*, suggest high sensitivity and chronic toxicity to imidacloprid (Mulye 1995). A 96-h NOEC has been determined at 1.24 µg a.i./L, based on survival (Gagliano 1991). Using growth as a measure of effect, a 10-day study reported a NOEC of 0.67 µg a.i./L and a LOEC of 1.24 µg a.i./L (Gagliano 1991). Effects at similar concentrations were reported by Stoughton (2006) for *Chironomus tentans*, with a 96-h LC_{50} for technical grade imidacloprid of 5.75 µg a.i./L. For another midge species, *Chironomus riparius*, the 28-day LOEC (EC_{15}) and EC_{50} for reduced adult emergence were 2.25 and 3.11 µg a.i./L, respectively (Dorgerloh and Sommer 2001). Reduced development rates of *C. riparius* leading to delays in emergence were observed at concentrations as low as 3.7 µg a.i./L (Dorgerloh and Sommer 2001). Larvae of another insect species, the black fly *Simulium vittatum*, showed similar acute sensitivity, with 48-h LC_{50} values ranging from 6.75 to 9.54 µg a.i./L (Overmyer et al. 2005).

The sensitivity of crustaceans to imidacloprid appears to vary. Sánchez-Bayo and Goka (2006b) noted that ostracod species appear to be much more sensitive than cladocerans. For example, 48-h EC_{50}s for immobilization in the ostracod species *Ilyocypris dentifera*, *Cypridopsis vidua*, *Cypretta seurati*, and *Chydorus sphaericus* were 3, 3, 16, and 2209 µg a.i./L, respectively. In contrast, the 48-h EC_{50} for immobilization in the cladoceran water flea, *Daphnia magna*, was 6029 µg a.i./L (Sánchez-Bayo and Goka 2006b). Other studies also indicate that *D. magna* is more tolerant of elevated concentrations of imidacloprid. Song et al. (1997) conducted 48-h toxicity tests on newly hatched *Daphnia magna*. The LC_{50}s for *Daphnia magna* ranged from 17,360 µg a.i./L at 20°C to 10,440 µg a.i./L at 27°C. Based on immobilization, a 21-day EC_{50} of greater than 7300 µg a.i./L was reported for *Daphnia magna* (Young and Blakemore 1990). As well, a 48-h LC_{50} of 85,000 µg a.i./L and a 48-h mortality NOEC of 42,000 µg a.i./L have been determined (Young and Hicks 1990). Young and Blakemore (1990) also reported a 21-d NOEC of 1800 µg a.i./L and a 21-d LOEC of 3600 µg a.i./L, suggesting chronic toxicity to *Daphnia magna* (Mulye 1995).

In comparison, toxicity testing for the yellow fever mosquito, *Aedes aegypti,* conducted in the same study as that with newly hatched *Daphnia magna*, had a much lower LC_{50}, at 44 µg a.i./L for both 20°C and 27°C (Song et al. 1997).

Amphipods appear to have intermediate sensitivity in comparison with other crustaceans. England and Bucksath (1991) reported a 96-h mortality NOEC of 0.35 µg a.i./L and a 96-h LC$_{50}$ of 520 µg a.i./L for the amphipod *Hyalella azteca*. Even more sensitive results were reported by Stoughton (2006), in which exposure to technical grade imidacloprid resulted in a 96-h LC$_{50}$ of 65.4 µg a.i./L. For the amphipod *Gammarus pulex*, a 28-day LOEC for mortality was reported at 256 µg a.i./L (Hendel 2001).

Only a few studies were identified that investigated the toxicity of common transformation products of imidacloprid to invertebrates. A NOEC of 1000 µg/L 6-chloronicotinic acid for *Chironomus tentans* larvae was reported, however, this was the only concentration used in the study (Bowers and Lam 1998, as reviewed in Mulye 1999). For a study involving *C. tentans* larvae exposed to the transformation product des-nitro imidacloprid, the NOEC was determined as 8190 µg/L and the LC$_{50}$ was >82,800 µg/L (Bowers 1996b, as reviewed in Mulye 1997a), suggesting less potential toxicity from the transformation product than from the parent compound. Based on mortality and sublethal effects, a 96-hour EC$_{50}$ was reported at 17,000 µg/L, with a LOEC of 82,800 µg/L (Bowers 1996b, as reviewed in Mulye 1997a). The amphipod *Hyalella azteca* also exhibited less sensitivity to this transformation product than to the parent compound, with a 96-h LC$_{50}$ reported at 51,800 µg/L and a 96-h EC$_{50}$ of 29,000 µg/L, based on sublethal effects (Roney and Bowers 1996, as reviewed in Mulye 1997a). A NOEC and LOEC were also determined based on a combination of mortality and sublethal effects, at 22,100 µg/L and 43,800 µg/L, respectively (Roney and Bowers 1996, as reviewed in Mulye 1997a). Acute exposure to the transformation product imidacloprid urea resulted in a 96-h LC$_{50}$ of > 94,830 µg/L, a 96-h EC$_{50}$ of > 94,830 µg/L and a 96-h NOEC of 94,830 µg/L for *Hyallela azteca* (Dobbs and Frank 1996, reviewed in Mulye 1997a). In summary, the transformation products of imidacloprid appear to be considerably less toxic to invertebrates than the parent compound.

8.2.3 Algae and Plants

Although imidacloprid is an insecticide, there is evidence that at higher concentrations it may be harmful to plants and algae.

Imidacloprid exposure resulted in decreased growth by the blue-green alga (*Anabaena flos-aquae)* at moderate concentrations. Specifically, from a 4-d exposure to imidacloprid, a NOEC of 24,900 µg a.i./L, LOEC of 40,500 µg a.i./L, EC$_{25}$ of 26,700 µg a.i./L and EC$_{50}$ of 32,800 µg a.i/L were reported for the blue-green algae (Bowers 1996a, as reviewed in Mulye 1997a).

In comparison, a 4-d growth inhibition NOEC of 6690 µg a.i./L, an EC$_{25}$ of 9340 µg a.i./L and an EC$_{50}$ of 12,370 µg a.i./L was shown for the freshwater diatom *(Navicula pelliculosa)*, exposed to imidacloprid (Hall 1996, as reviewed in Mulye 1997a).

The green alga *Pseudokirchneriella subcapitata* exhibited less sensitivity to imidacloprid than *A. flos-aquae* and *N. pelliculosa*. The 5-d EC$_{50}$ and NOEC values determined for growth inhibition in *P. subcapitata* were reported as more than 119,000 µg a.i./L, with the chemical nearing its solubility limit in the test media (Gagliano and Bowers 1991).

In a toxicity test with the green alga *Scenedesmus subspicatus*, no concentrations higher than 10,000 µg a.i./L (nominal) were used because of solubility challenges (Heimbach 1986). From this test, a 96-h EC_{50} for growth inhibition was reported as greater than 10,000 µg a.i./L, and the NOEC based on both types of effects was given as 10,000 µg a.i./L (Heimbach 1986).

In general, it appears that algae are at least three orders of magnitude less sensitive to imidacloprid than many insect and ostracod species.

8.2.4 Mesocosm and Field Studies

A few mesocosm studies have been conducted with imidacloprid. These types of studies, by more closely simulating a natural system, can provide insight into the toxicity of imidacloprid when processes such as dissipation and partitioning to other media (e.g., sediment) are taken into consideration. They can also demonstrate potential community-level effects that may occur indirectly as a result of imidacloprid application through effects on interactions between different species.

An outdoor mesocosm study was conducted with tanks containing diverse communities of phytoplankton, zooplankton, and macroinvertebrates to investigate effects of imidacloprid (Moring et al. 1992). Technical grade imidacloprid (95.8% purity) was applied to the surface of the tanks in 4 applications made at two week intervals. An overall mesocosm "no significant adverse effect concentration" of 6 µg a.i./L was reported for the study. At this concentration, minor adverse effects were only observed in a few invertebrate species, and these recovered quickly. At the next highest concentration of 20 µg a.i./L there were decreases in overall phytoplankton density and densities of Copepoda, mayflies, caddisflies, and the amphipod *Hyalella azteca* (Moring et al. 1992).

Community effects of imidacloprid were also examined in an outdoor mesocosm study conducted by Ratte and Memmert (2003). This study reported an overall mesocosm NOEC of 0.6 µg a.i./L and a LOEC of 1.5 µg a.i./L. A no observable ecologically adverse effect concentration (NOEAEC), i.e., the concentration where there was recovery within 8 weeks of the exposure from any adverse effects observed, was reported at 9.4 µg a.i./L. The results of this study must be treated with caution, however, because they are based on nominal concentrations, despite the fact that measurements showed significant decreases in the concentrations between applications. The study also used a formulated product, Confidor SL 200, which contained only 17.3% active ingredient. Therefore, it is unknown what effect the other substances in the formulation had.

Sánchez-Bayo and Goka (2005; 2006a) studied community effects of imidacloprid in a field study where rice seedlings treated with Admire (containing 1% imidacloprid) were planted in flooded paddies. Mean water concentrations of imidacloprid at the initiation of the study were 240 µg a.i./L, but declined steadily to a concentration of 0.1 µg a.i./L by 90 days. During this exposure, no mortality or malformation effects were observed in Japanese medaka fry (*Oryzias latipes*). However, at the first monthly sampling, 100% of all medaka fry in the fields treated with imidacloprid were infested with a protozoan ectoparasite, compared with only 40% of fry in the control fields. The high occurrence of parasitism in the imidacloprid-treated fields suggests that these fish were under physiological stress, making them more susceptible to the parasites (Sánchez-Bayo and Goka 2005). Effects on the zooplankton and benthic invertebrate communities were also studied. The major effect of imidacloprid that was observed was an absence of many zooplankton crustaceans (e.g., ostracods) and benthic species (e.g., *Chironomus yoshimatsui*) when imidacloprid concentrations in the water were greater than 1 µg a.i./L. As a result of these changes to the invertebrate populations, green algae blooms also developed in the fields treated with imidacloprid (Sánchez-Bayo and Goka 2006a). It should be noted that this study simulates a case where imidacloprid would be intentionally applied to an aquatic system, whereas, in Canada imidacloprid is only registered for terrestrial use, and therefore any imidacloprid entering aquatic systems would be through indirect routes such as spray drift or runoff.

8.3 Toxicity to Marine Life

8.3.1 Fish

A 7-day investigation of the toxicity of imidacloprid to the larvae of the marine inland silverside (*Menidia beryllina*) reported a NOEC of 62,000 µg a.i./L and a LOEC of 96,900 µg a.i./L, based on survival, as well as an LC_{50} of 77,500 µg a.i./L (Environment Canada 2005). For the effect of growth inhibition, the same study reported a NOEC of 17,400 µg a.i./L, a LOEC of 34,000 µg a.i./L, an IC25 of 62,200 µg a.i./L, and an IC_{50} of 72,300 µg a.i./L.

For the adult sheepshead minnow, *Cyprinodon variegatus*, exposed to imidacloprid in a 96-hour acute toxicity test, the reported LC_{50} was 161,000 µg a.i./L, with a NOEC of 58,200 µg a.i./L (Ward 1990a).

8.3.2 Aquatic Invertebrates

The mysid shrimp (*Mysidopsis bahia*) appears to be very sensitive, with a reported 96-h LC_{50} of 34.1 µg a.i./L for technical grade (96.2% purity) imidacloprid (Ward 1990b). For imidacloprid formulated as the product Admire (240 g a.i./L), the LC_{50} (96-hour) was very close to that for the technical product, at 36 µg a.i./L (Lintott 1992). With formulated products, there is the potential that some of the observed toxicity could be due to other ingredients in the formulation, and not just the active ingredient. However, the close agreement of the *M. bahia* LC_{50}s from Lintott (1992) and Ward (1990b) suggest that in this case most of the toxicity from Admire can likely be attributed to imidacloprid. A 96-h NOEC of 13.3 µg a.i./L was determined for mysid shrimp

exposed to technical imidacloprid, based on mortality (Ward 1990b).   Exposure to Admire yielded a 96-h NOEC of 21 µg a.i./L based on mortality (Lintott 1992). Reported MATCs from chronic toxicity testing with mysids include 0.23 µg a.i./L for growth effects and 0.643 µg a.i./L for reproductive effects (Ward, 1991). These results indicate that imidacloprid is acutely and chronically toxic to mysid shrimp (Mulye 1995).

Song et al. (1997) reported a 48-h LC_{50} of 361,000 µg a.i./L for adult saltwater brine shrimp (*Artemia* sp.). Juvenile brine shrimp exhibited lower toxicity with approximately 40% of the juveniles dying at 800,000 µg a.i./L, the highest dose administered, after 72 hours (Song and Brown 1998). For the juvenile salt marsh mosquito (*Aedes taeniorhynchus*) the 72-h LC_{50} was 21 µg a.i./L (Song and Brown 1998), while the 48-h LC_{50} for the first instar stage was 13 µg a.i./L (Song et al. 1997).

8.3.3 Algae and Plants

No studies have been found on the toxicity of imidacloprid to marine algae or plants.

8.4 Effect of Exposure Regime

Many of the toxicity studies available for imidacloprid have involved continuous exposures over the duration of the study. In the case of long-term studies, however, this may not represent a realistic scenario. Imidacloprid used for agricultural purposes is typically only applied to a field once or twice in a season, so any transport to surface waters is likely to occur in short-duration pulses, followed by dissipation and biodegradation.

Stoughton (2006) looked at the effect of pulse exposures to imidacloprid (as the Admire[®] formulation) on the freshwater invertebrate *Chironomus tentans* under various different exposure regimes, with organisms being transferred to clean water during recovery periods between pulses. In all three exposure regimes, the cumulative duration of exposure was the same, but with shorter, more frequent pulses, less mortality was observed; on the other hand, the short, frequent pulses had a greater effect on dry weight. Stoughton (2006) suggests that with shorter duration recovery periods the organisms may not have been able to depurate or metabolize all of the accumulated imidacloprid before the next pulse, thereby resulting in a more continuous duration of sublethal stress that affected growth. The study suggests that exposure to a contaminant as pulses can be equally toxic or less toxic than continuous exposures depending on the endpoints evaluated. Similarly, Alexander (2006) found that adverse effects were observed in mayflies at the same concentration with either a 12-hour pulse exposure (followed by 19 days exposure to clean control water), or a 20-day continuous exposure to imidacloprid, Alexander (2006) suggests that the effects of imidacloprid may be both immediate and prolonged, and that even short-term exposures may have long-term impacts.

8.5 Toxicity of Technical Grade Imidacloprid Versus Formulations

Exposure of aquatic organisms to formulated products can yield different levels of toxicity than exposure to just the active ingredient. Formulants present in a pesticide product may themselves exert some toxicity, or may increase the toxicity of the active ingredient by affecting its uptake, metabolism or excretion. Stoughton (2006) compared the toxicities of technical grade imidacloprid and the formulated product Admire® to two freshwater invertebrates, the midge *Chironomus tentans*, and the amphipod *Hyalella azteca*. Interestingly, different results were observed with the two species. In the case of *H. azteca*, Admire® was considerably more toxic than the technical grade imidacloprid, with 96-h LC_{50}s of 17.44 µg a.i./L for Admire® and 65.43 µg a.i./L for imidacloprid. On the other hand, *C. tentans* showed similar responses to the two substances, with 96-h LC_{50}s of 5.40 µg a.i./L for Admire® and 5.75 µg a.i./L for the technical grade imidacloprid. Therefore, it may not be possible to make any general statements on the relative toxicity of imidacloprid and its formulated products, as this could vary depending on the species.

9. TERRESTRIAL TOXICITY

9.1 Toxicity to Mammals

Canadian livestock may be exposed to pesticide residues through consumption of contaminated feed/crops, or through ingestion of contaminated water. Imidacloprid is rapidly absorbed from the gastrointestinal tract and within 48 hours of administration is effectively eliminated through urine (70 to 80%) and feces (20 to 30%) (Tomlin 1994; PMRA 2001). In mammalian systems, imidacloprid is hydroxylated and hydrolyzed to the critical metabolic product, 6-chloronicotinic acid. This metabolite is further conjugated and eliminated or reduced to guanidine (Tomlin 1994).

Imidacloprid is moderately toxic to mammals via the oral route of exposure (PMRA 2001). Acute toxicity of imidacloprid to rats varies, depending on the route of exposure, with oral dosing posing the greatest toxic threat (Mulye 1996). The oral LD_{50}s for rats exposed to one dose of technical-grade imidacloprid was 424 mg a.i./kg bw for males and 450 – 475 mg a.i./kg bw for females (Mulye 1996). The symptoms of acute toxicity to domestic animals resemble a general nicotine-like response: fatigue, twitching, cramps, and muscle weakness (Hovda and Hooser 2002). In rats, oral exposure induced apathy, respiratory disturbances, decreased motility, staggering gait, narrowed palpebral fissures, transient trembling and spasms, which subsided in under one week (Mulye 1996). The thyroid appears to be a target organ in imidacloprid dosing, as thyroid lesions are associated with high doses in short-term and chronic feeding studies of imidacloprid (Hovda and Hooser 2002). In a two-year rat study, the oral NOAEL for thyroid toxicity was determined to be 100 mg a.i./kg diet (i.e., 5.7 mg a.i./kg bw/day for males and 7.6 mg a.i./kg bw/day for females), with a LOAEL of 300 mg a.i./kg diet (i.e., 16.9 mg a.i./kg bw/day for males and 24.9 mg a.i./kg bw/day for females) (U.S. EPA 1995).

The U.S. EPA (1994) considers imidacloprid to be a non-genotoxic chemical to mammals. Imidacloprid has been classified as a 'Group E' chemical, one for which no evidence of

carcinogenicity exists (U.S. EPA 1995). Tomlin (2000) states that it is neither mutagenic nor teratogenic. A 2-year mouse study observed no carcinogenic effects and determined a NOAEL of 1,000 mg a.i./kg diet for non-carcinogenic effects such as decreased body weight, food consumption, water intake, and liver and spleen weight (PMRA 1995). The U.S. EPA (1995) reports that no carcinogenic effects were observed in a 2-year study, in which rats were fed imidacloprid at doses as high as 1,800 mg a.i./kg. With a NOEL reported for oncogenicity in rodents at 208 mg a.i./kg bw/day, PMRA suggests there is no evidence for oncogenicity in rodents (reviewed in Mulye 1996).

EXTOXNET (1998) lists imidacloprid as weakly mutagenic, based on 2 positives in a series of 23 mutagenic assays submitted to the U.S. EPA for registration. The first positive assay tested for chromosome aberrations in an *in vitro* cytogenetic study with human lymphocytes as a result of imidacloprid exposure (U.S. EPA 1995). The second assay measured positive cytogenotoxicity in Chinese hamster ovary cells (U.S. EPA 1995). A subsequent study by Shah et al. (1997) also provides a contradictory assessment of the genotoxic potential of imidacloprid. This study used a ^{32}P-postlabelling assay to detect the presence of DNA adducts in calf thymus DNA that were exposed to the insecticide. The presence of DNA adducts, the covalent binding of DNA to a particular chemical, is the well-established cause of genotoxicity and the initial event in the process of carcinogenesis. Therefore, the assay is a direct measure of the genotoxic potential of a chemical. This study used Admire, the end use product of imidacloprid and reported significant adduct formation for this chemical, relative to controls. Although this study did not use the pure chemical imidacloprid, this result merits further investigation.

In a reproductive study on both rats and chinchilla rabbits, imidacloprid was administered to pregnant females through gavage on gestation days 6 to 15 for the rat and 6 to 18 for the rabbit (PMRA 1995). The study reported no developmental effects on the fetuses at 30 mg a.i./kg bw/day and 24 mg a.i./kg bw/day in the rat and rabbit, respectively. Skeletal abnormalities were observed at higher doses, with LOAELs of 100 and 72 mg a.i./kg bw/day for the rat and rabbit, respectively (PMRA 1995). Effects on maternal body weight gain (and maternal food consumption, in the case of rabbits) were observed at lower concentrations, with LOAELs of 30 and 24 mg a.i./kg bw/day for the rat and rabbit, respectively. The compound is not considered teratogenic for either species (Mulye 1996).

In a rat reproductive study with dietary administration of imidacloprid using two generations, each with two litters, the parental NOEC was 250 mg a.i./kg diet (PMRA 1995). At a dietary concentration of 700 mg a.i./kg (i.e., 47.3-56.7 mg a.i./kg bw/day for males and 52.3-62.8 mg a.i./kg bw/day for females), decreased weight gain was observed in the parents. Similarly, effects on pup body weight were observed at 700 mg a.i./kg diet; however, no reproductive effects were observed at this concentration (PMRA 1995).

Imidacloprid exhibits low toxicity via the dermal route of exposure (PMRA 2001). The LD$_{50}$ for dermal application of imidacloprid to rats was reported as >5000 mg a.i./kg bw (Mulye 1996). A dermal rat LD$_{50}$ of >2000 mg a.i./kg bw has been reported for an end-use product containing imidacloprid, Admire 240F (Mulye 1996). Female rats suffered reduced gain in body weight with dermal exposure, which was the only sign of toxicity (reviewed in Mulye 1996).

Imidacloprid is not considered an irritant to rabbit skin and eye, and is not a skin sensitizer in guinea pigs (Mulye 1996).

Imidacloprid exhibits low toxicity via the inhalation route of exposure (PMRA 2001). The inhalation 4-h LC_{50} for rats was reported as >0.069 mg a.i./L air (as aerosol) and >5.3 mg a.i./L air (as dust) (Mulye 1996). Signs in rats exposed through inhalation to imidacloprid included slightly laboured breathing, decreased motility, piloerection, slight tremor and decreased body-weight gain (Mulye 1996). Rats exposed to imidacloprid through inhalation for 6 hours/day, 5 days/week for 4 weeks showed increased liver weight, increased coagulation time, and clinical chemistry changes at an exposure rate of 0.191 mg a.i./L air/day (i.e., 51.9 mg a.i./kg bw/day) (PMRA 1995).

Systemic toxicity associated with imidacloprid exposure was not observed in rabbits repeatedly exposed via skin over 21 days, or in dogs repeatedly exposed to oral doses of technical compound over 52 weeks (PMRA 1995; Mulye 1996). For these studies, toxicity endpoints were a NOEL of >1000 mg a.i./kg bw/day for the rabbit and >72 mg a.i./kg bw/day for the dog (Mulye 1996). However, in another study, dogs exposed to imidacloprid in their diet for 13 weeks at a concentration of 600 mg a.i./kg diet (i.e., 22.1 mg a.i./kg bw/day for males and 24.8 mg a.i./kg bw/day for females) exhibited signs of trembling and emaciation (PMRA 1995).

9.2 Toxicity to Birds

The PMRA (2001) considers imidacloprid to be acutely toxic to birds and to cause avian reproductive toxicity.

Numerous studies have looked at effects in bobwhite quail (*Colinus virginianus*). The LC_{50} based on a study in which bobwhite quail were exposed to imidacloprid through dietary intake for 5 days was 1420 mg a.i./kg diet and the NOEC was < 69 mg a.i./kg diet (Toll 1990a, reviewed in Mulye 1995). The exposed birds exhibited hyporeactivity, ataxia, wing drop, diarrhea, opisthotonos (i.e., rigidity and severe arching of the back), immobility and intoxication (Toll 1990a, reviewed by Mulye 1995). Autopsies found abnormalities such as fluid-filled crops, fluid- and gas-filled intestines, and mottled/discoloured livers (Toll 1990a, reviewed in Mulye 1995).

A further investigation showed that adult bobwhite quail exposed to imidacloprid in their diet for 20 weeks also produced eggs with thinner shells, reduced strength and lower hatchability, and young with significantly lower body weights compared to control birds (Toll 1990c, reviewed in Mulye 1995). In this study, treatments and controls showed comparable feed consumption, but adult male bobwhite quails exposed to the highest concentration of imidacloprid (240 mg a.i./kg diet) had significantly decreased body weight (Toll 1990c, reviewed in Mulye 1995). The NOEC from this study was 120 mg a.i./kg diet, and the LOEC was 240 mg a.i./kg diet (Toll 1990c, reviewed in Mulye 1995). Autopsies did not reveal signs of physical effects in the birds (Toll 1990c, reviewed in Mulye 1995). In another study with 10-day old bobwhite quail exposed to imidacloprid through their feed for 5 days, reduced feed consumption was observed at 312 mg

a.i./kg diet and reduced body weight was observed at 625 mg a.i./kg diet (Toll 1990a, reviewed in Mulye 1995).

Bobwhite quail exposed to an oral dose of imidacloprid by gelatin capsule showed clinical signs of intoxication similar to those of birds exposed through diet, such as hyporeactivity, ataxia, immobility, fluffed feather coat and wing drop (Toll 1990d, reviewed in Mulye 1995). Toxicity endpoints derived from this study included a 14-d LD_{50} of 152 mg a.i./kg body weight, a NOEL (for mortality) of 25 mg a.i./kg, and a LOEL (for mortality) of 50 mg a.i./kg (Toll 1990d, reviewed in Mulye 1995). At a higher dose of 800 mg a.i./kg bw there was a significant reduction in feed consumed by the quails (Toll 1990d, reviewed in Mulye 1995). Autopsies revealed emaciation, fluid-filled crop, fluid- and gas-filled intestines and enlarged and fluid-filled colon in exposed birds (Toll 1990d, reviewed in Mulye 1995).

Stafford (1991, reviewed in Mulye 1996) reported indications of physical distress in house sparrows *(Passer domesticus)* that received an oral dose of imidacloprid (granular) by gelatin capsule that included ataxia, hyporeactivity, loss of flight, diarrhea, immobility and moribundity. The LD_{50} determined for the house sparrow was 41 mg a.i./kg bw (Stafford 1991, reviewed in Mulye 1996). Similarly, the LD_{50}s determined for the pigeon (*Columba livia)* and canary (*Serinus canaries)* were 25 - 50 mg a.i./kg bw (Grau 1986, 1987a, reviewed in Mulye 1996). The Japanese quail (*Coturnix coturnix)* had an LD_{50} of 31 mg a.i./kg bw (Grau 1988a, reviewed in Mulye 1996). NOEL values, based on mortality for the house sparrow and the Japanese quail were 3 mg a.i./kg bw and 3.1 mg a.i./kg, respectively (Stafford 1991, reviewed in Mulye 1996; Grau 1988a, reviewed in Mulye 1996). The LOEL for the house sparrow, based on mortality, was 6 mg a.i./kg (Stafford 1991, reviewed in Mulye 1996). These studies indicate that imidacloprid is highly toxic to these species (Mulye 1996).

Mallard ducks (*Anas platyrhynchos*) that received imidacloprid in their diet appeared to be more resistant to the pesticide's effects than bobwhite quail. A 5-day study found that no mortality occurred in mallards that had dietary exposure to imidacloprid up to the highest dose group of 5000 mg a.i./kg diet (Toll 1990b reviewed in Mulye 1995). However, the study determined that the NOEC for body weight and feed consumption of the mallard was 69 mg a.i./kg diet (Toll 1990b, reviewed in Mulye 1995). Ataxia was reported for some imidacloprid-exposed birds, but autopsies did not reveal pronounced physical abnormalities (Toll 1990b, reviewed in Mulye 1995). Another dietary intake study with mallards, conducted over 20 weeks, reported a LOEC of 240 mg a.i./kg diet for reduction of adult female body weight and reduced eggshell thickness and strength (Stafford 1992, reviewed in Mulye 1995). Mallard ducks that were orally dosed with capsules of imidacloprid had an LD_{50} of 283 mg a.i./kg bw, suggesting moderate toxicity for this species (Hancock 1996, reviewed in Mulye 1997a). Symptoms of intoxication, such as ataxia, hyporeactivity, diarrhea and immobility were observed at concentrations as low as 25 mg a.i./kg bw (Hancock 1996, reviewed in Mulye 1997a).

Oral dosing and dietary toxicity studies are particularly relevant because birds may be exposed to imidacloprid-treated seeds in their diet. The effects of seeds treated with imidacloprid as bird-repellent were examined on red-winged blackbirds, *Agelaius phoeniceus* and brown-headed cowbirds, *Molothrus ater*, (Avery et al. 1993; Avery et al. 1994). In these experiments, the birds quickly learned to avoid imidacloprid-treated seeds. The 5-day LOECs for deterrence from

feeding were 620 and 165 mg a.i./kg diet for treated rice and wheat seeds, respectively in red-winded blackbirds, and 620 mg a.i./kg diet for treated rice seeds in brown-headed cowbirds (Avery et al. 1993). Effects observed in those birds that did ingest the treated seeds were gastrointestinal distress and ataxia (Avery et al. 1994). Birds recovered fully once exposure was blocked (Avery et al. 1993). The authors suggested that avoidance of the treated seed was due to post-ingestional distress, and not to sensory repellency (Avery et al. 1994). Another study examined the effect of seed-eating behaviour and found that different bird species will be exposed to varying amounts of seed residues (Avery et al. 1997).

9.3 Toxicity to Crops and Other Plants

Imidacloprid is absorbed and distributed throughout plants acropetally (i.e., moves from base to new growth) (Tomlin 2000). Imidacloprid has been found to be translocated in a variety of crops and plants (Mukherjee and Gopal 2000; Dikshit et al. 2003). After administration to soil or to seed, imidacloprid has excellent root-systemic properties. Plant metabolism of imidacloprid was consistent over a wide variety of crop species and methods of administration. As with mammalian systems, the primary metabolite in plants is 6-chloronicotinic acid (Tomlin 2000).

There is a paucity of data available on imidacloprid's toxicity to crops. In experiments using wheat and barley, Pike et al. (1993) found that imidacloprid applied as a seed treatment, alone and in combination with various fungicides, was not deleterious to plant growth based on plant stand, tillers produced, or plant height at a concentration of 2.5 g/seed. When imidacloprid was sprayed on tomato plants at two to four times the recommended application rate (4 x 80 g a.i./ha), phytotoxic symptoms were not observed (Dikshit et al. 2003).

Given that imidacloprid is intended for application to crops to protect them from insect pests, it is not expected that terrestrial plants would show sensitivity to imidacloprid.

10. CANADIAN ENVIRONMENTAL QUALITY GUIDELINES

The Canadian Water Quality Guidelines for imidacloprid were derived according to "A protocol for the Derivation of Water Quality Guidelines for the Protection of Aquatic Life" (CCME 1991) and "Protocols for Deriving Water Quality Guidelines for the Protection of Agricultural Water Uses (Irrigation and Livestock Water)" (CCME 1993). The following sections describe the development of each of these guidelines.

Toxicity studies that were conducted with formulated products, rather than technical grade imidacloprid, were not considered for use in guideline derivation. Formulants used in pesticide products may augment the toxicity of the active ingredient by making it more bioavailable, or there may be toxicity associated with formulants themselves. Therefore, by not considering toxicity tests with formulated products, it is possible that the guidelines could be underprotective. However, the formulants used will not be the same across all pesticides with the same active ingredient, and potential effects of the formulants themselves may differ among species. For these reasons, it then becomes difficult to make comparisons of toxicity across studies. Therefore, the guidelines below are based only on studies with technical grade imidacloprid.

10.1 Protection of Freshwater Aquatic Life

The imidacloprid toxicity data available for freshwater life are compiled in Appendix A. Imidacloprid did not satisfy the minimum data requirements for a full guideline, which are detailed in the derivation protocol (CCME 1991). Only one primary chronic fish study was available. The available data met the requirements for an interim guideline, in which both primary and secondary studies may be included. Therefore, an interim freshwater quality guideline was developed for imidacloprid and is described below.

10.1.1 Derivation of Interim Freshwater Guideline

Insects and ostracods exhibited very high sensitivity to imidacloprid. Particularly sensitive species included the midges, *Chironomus tentans* and *Chironomus riparius*, and the ostracods *Cypridopsis vidua* and *Ilyocypris dentifera* The most sensitive studies for each of these species are described below.

Gagliano (1991) conducted a chronic toxicity test with *Chironomus tentans*. Second instar *C. tentans* larvae were exposed to a technical formulation of imidacloprid (95% purity) for the duration of 10 days. The culture media used for rearing and testing was blended spring water, while dimethylformamide was used as a solvent carrier for the imidacloprid. Test chambers consisted of 1L glass beakers containing 0.5 to 1 mm of silica sand as an artificial substrate. Renewal was conducted on every second day, throughout the length of the study. Two replicate test chambers were used for each treatment and the controls, with 10 chironomid larvae tested in each chamber. Samples of the fresh test solution were taken on day 0 and day 5 and analyzed by liquid chromatography. In addition, samples of older test water were taken on day 3 and at the

end of the test to determine the stability of the test water. All calculations were based on measured concentrations. During the test, the pH ranged from 8.1 to 8.2, hardness was 118 mg $CaCO_3$/L and the temperature was maintained between 20.8 to 22.3°C. Dissolved oxygen ranged from 5.8 to 7.9 mg/L (79 to 108% saturation) for most of the study, but on the final day was measured down at 2.0 to 4.0 mg/L, possibly due to increased oxygen demand from excess food in the test chambers. Chironomids were monitored daily for lethal and sublethal (i.e., growth) effects. This study determined a NOEC for growth of 0.67 μg a.i./L, a NOEC for survival of 1.24 μg a.i./L, a LOEC for growth of 1.24 μg a.i./L, and an LC_{50} of 3.17 μg a.i./L. Trace quantities of imidacloprid were detected in samples of the control water (0.20 μg a.i./L) on sampling days 5, 7 and 10. However, the researchers suggested that the contamination occurred during sample extraction and stated that no biological effects were associated with the detection, with significant differences occurring in the treatments relative to the controls. This study was classified as secondary due to the low replication, contamination detected in controls, and low dissolved oxygen measured at the end of the exposure period. A 48-h LC_{50} of 68.9 μg a.i./L was also reported in the study. This endpoint would be classified as primary because during this period of the exposure there was no contamination of the controls and dissolved oxygen levels remained acceptable.

Dorgerloh and Sommer (2001) conducted a 28-day test with the midge _Chironomus riparius_. First instar larvae (<2-3 days old) were exposed to technical grade imidacloprid (98.4% purity), without any solvents, in "M-7" culture medium, which consists of deionized water amended with mineral salts and vitamins. Test containers were 0.6L glass beakers containing a 1.5 cm layer of artificial sediment and a 6 cm (0.38L) layer of water. The test was static, with three replicate beakers per treatment, and 20 animals per beaker. Three times during the study (at 1 hour, 7 days, and 28 days), samples of the overlying water and porewater of the sediment were analyzed from the control and three of the seven test concentrations, using LC-MS/MS with solid phase extraction. During the test, the pH ranged from 7.9 to 8.6, dissolved oxygen was 7.4 to 8.8 mg/L, water hardness was 267 to 303 mg $CaCO_3$/L, and temperature was 20.1 to 20.4°C. The EC_{10} (comparable to a NOEC) for adult emergence was 2.09 μg a.i./L, the EC_{15} (comparable to a LOEC) was 2.25 μg a.i./L, and the EC_{50} for emergence was 3.11 μg a.i./L. The authors also looked at the effects of imidacloprid on the time to emergence, and reported a LOEC for reduced development rate of 3.7 μg a.i./L. This study was classified as secondary because it was static and used the initial nominal concentrations, even though measurements showed that imidacloprid concentrations in the treatments decreased to approximately 23% of nominal by the end of the exposure.

Sánchez-Bayo and Goka (2006b) found that the ostracods _Ilyocypris dentifera_ and _Cypridopsis vidua_ are highly sensitive to imidacloprid. Field-collected specimens of each species were exposed to >99.5% purity technical grade imidacloprid (without any solvents) for a duration of 48 hours. Tests were conducted both in the dark, and with a light/dark cycle of 16:8h. Only the results of the light/dark assays are reported here. The static tests were conducted in 10 mL glass vials, with five organisms per vial and 12 replicates per test concentration, with control organisms in water only. Water pH and dissolved oxygen were measured at the beginning and end of the test. pH ranged from 7.54 ± 0.51 at the start of the tests to 7.83 ± 0.44 at the end. Dissolved oxygen concentrations were 7.06 ± 1.47% at initiation, with an average decrease of 24% of the initial concentration over the course of the exposure. Temperature was maintained at

22°C. All reported test concentrations were nominal. For both *Ilyocypris dentifera* and *Cypridopsis vidua* 48-h EC_{50}s for immobilization were 3 µg a.i./L. Control mortalities were within acceptable limits, at 3 to 5% for *I. dentifera* and 5 to 10% for *C. vidua*. This study was classified as secondary because it used static conditions and results were based on nominal, calculated concentrations.

The protocol (CCME 1991) states that a guideline is preferentially derived using the most sensitive chronic study available. The most sensitive chronic datapoint is the 10-day LOEC of 1.24 µg a.i./L for reduced growth in *C. tentans* (Gagliano 1991). However, due to a number of uncertainties associated with the Gagliano (1991) study, as described above, the second lowest chronic LOEC, from Dorgerloh and Sommer (2001) was selected as the critical study. Therefore, the critical datapoint is the 28-day LOEC (EC_{15}) for reduced adult emergence of *Chironomus riparius* at 2.25 µg a.i./L (Dorgerloh and Sommer 2001). This is a low effects level chronic study, so a safety factor of 0.1 is applied, in accordance with the CCME protocol (1991). Therefore, the interim freshwater aquatic life guideline for imidacloprid was calculated as follows:

$IWQG_{FAL}$ = LOEC x SF
        = 2.25 x 0.1
        = 0.23 µg a.i./L

where,

$IWQG_{FAL}$= Interim Water Quality Guideline for Freshwater Aquatic Life
SF   = Safety Factor

Therefore, the interim water quality guideline for the protection of freshwater aquatic life is 0.23 µg a.i./L.

Figure 3 presents a selection of the most sensitive freshwater toxicity data that were available and illustrates where the critical study and resulting guideline values fall with respect to the distribution of other sensitive endpoints. It is worth noting that among the more sensitive species there was not a large difference in effect concentrations between acute and chronic studies. Had the most sensitive acute study been used to derive the guideline, i.e., a 48-h EC_{50} of 3 µg a.i./L, when divided by the safety factor of 20 for non-persistent substances (CCME 1991), a slightly lower guideline value of 0.15 µg a.i./L would have resulted. The 96-h LC_{50} for *Chironomus tentans* of 5.75 µg a.i./L reported by Stoughton (2006) is less than three times higher than the 28-d LOEC for emergence for the *Chironomus* species in Dorgerloh and Sommer (2001). At least eight other effect concentrations for technical grade imidacloprid (and an additional 10 effect concentrations with formulated products) fall within a factor of 10 of 2.25 µg a.i./L, lending support to the critical study (see Appendix A and Figure 3).

Toxicity Information		Species	Toxicity endpoint	Concentration (µg·L⁻¹)
Acute	Vertebrates	L. idus melanotus	96-h LC$_{50}$	
		O. mykiss	96-h LC$_{50}$	
		L. macrochirus	96-h LOEC	
		O. mykiss	96-h LOEC	
	Invertebrates	D. magna	48-h LC$_{50}$	
		D. magna	48-h LC$_{50}$	
		D. magna	48-h EC$_{50}$	
		C. sphaericus	48-h EC$_{50}$	
		C. tentans	48-h LC$_{50}$	
		H. azteca	96-h LC$_{50}$	
		H. azteca	96-h EC$_{50}$	
		A. aegypti	48-h LC$_{50}$	
		C. seurati	48-h EC$_{50}$	
		S. vittatum	48-h LC$_{50}$	
		C. tentans	96-h LC$_{50}$	
		C. vidua	48-h EC$_{50}$	
		I. dentifera	48-h EC$_{50}$	
Chronic	Vertebrates	O. mykiss	60-d LOEC	
	Invertebrates	D. magna	21-d LOEC	
		G. pulex	28-d LOEC	
		C. tentans	10-d LOEC	
		C. riparius	28-d LOEC	
		C. tentans	10-d LOEC	
	Mesocosms	various inverts	6-wk LOEC	
Canadian Water Quality Guideline 0.23 µg/L				

Toxicity endpoints:
■ primary
□ secondary
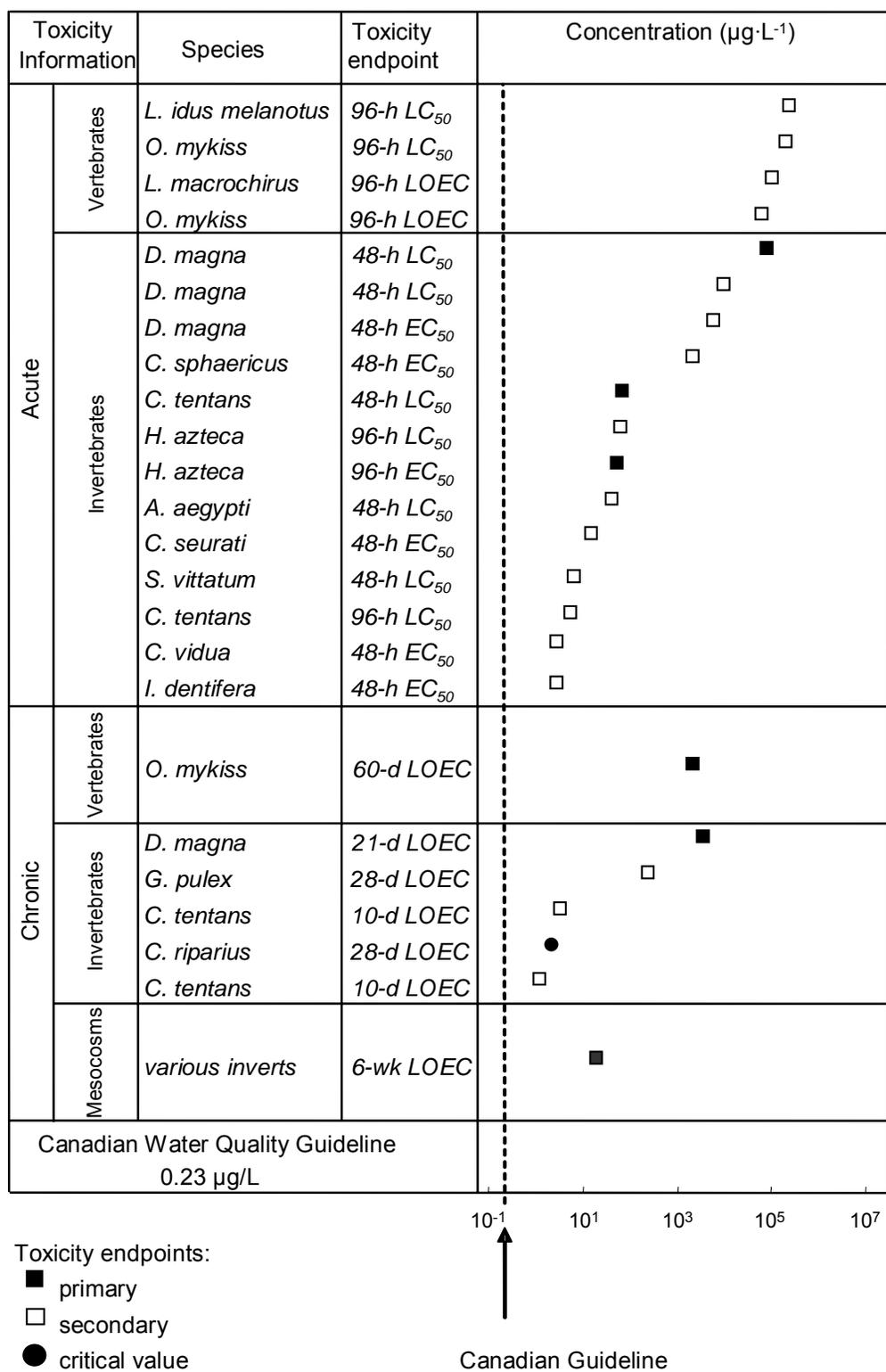● critical value

Canadian Guideline

Figure 3. Select freshwater toxicity data for imidacloprid.

10.2 Protection of Marine Aquatic Life

The minimum data requirements for the derivation of a full marine water quality guideline have not been met according to the CCME protocol (1991). Only one chronic study of primary value was found, which concerned a fish species. The available studies satisfy the minimum requirements for an interim marine water quality guideline. An interim guideline allows for the use of either acute or chronic data and does not require a study on algae or vascular plants, which is lacking for imidacloprid.

10.2.1 Derivation of Interim Marine Guideline

Of the studies identified, the marine species most sensitive to imidacloprid was the larval salt marsh mosquito, *Aedes taeniorhynchus* (Appendix B).

It is not surprising that *Aedes taeniorhynchus* exhibits comparatively high sensitivity to imidacloprid, as imidacloprid is targeted to selectively control sucking insects. In an acute 48-hour toxicity test, the salt marsh mosquito was reported to have an LC_{50} of 13 µg a.i./L (Song et al. 1997; Song and Brown 1998). The test was conducted under static conditions, at a temperature of 27°C, with artificial seawater (containing 38.1 g/L Instant Ocean Salt) as a test media. Technical-grade imidacloprid, of >95% purity, was applied to treatments and acetone was utilized as a carrier solvent. This study was classified as a secondary study based on partial reporting of the physical and chemical characteristics of the test water, and because it was not indicated whether concentrations were nominal or measured. No further studies regarding toxicity of imidacloprid to the salt marsh mosquito were available.

The species tested in the second most sensitive study was the mysid shrimp, *Mysidopsis bahia* (Appendix B). Ward (1990b) conducted a 96-hour acute toxicity test, in which juvenile mysid shrimp were exposed to imidacloprid in a flow-through system. The technical formulation NTN 33893 was used, with purity 96.2% imidacloprid. The formulation was introduced in the carrier solvent dimethylformamide and no precipitate was observed throughout the experiment. Water salinity was maintained in the range of 20-23 ‰. Temperature ranged from 20.3 to 24.7°C and pH was maintained from 8.4-8.6. Dissolved oxygen was equal to or above 4.4 mg/L throughout the experiment. Shrimp were observed for mortality and sublethal effects. From this experiment, Ward (1990b) reported a 96-h LC_{50} of 34.1 µg a.i./L, a 96-h NOEC of 13.3 µg a.i./L, and a LOEC of 22.9 µg a.i./L based on survivorship and sublethal effects. Sublethal effects included lethargy and partial loss of equilibrium. All calculations were based on measured concentrations. This study was classified as primary. These toxicity values were corroborated by a study by Lintott (1992) that reported similar sensitivities for the mysid shrimp. In this study a 96-h NOEC of 21 µg a.i./L and a 96-h LC_{50} of 36 µg a.i./L were determined for the mysid shrimp (Lintott 1992).

CCME protocol (1991) states that a low effect level from a chronic study is preferred in guideline derivation. When this type of study is unavailable, short-term median lethal or median effective concentrations from acute studies can be converted to long-term no-effect

concentrations using available acute-chronic ratios (ACRs). If ACRs are not available, short-term low effect levels from acute studies can be used with a safety factor of 0.05 or 0.01 applied. Neither chronic low effect level marine studies nor ACRs were available for imidacloprid. Therefore a safety factor was applied to the short-term LC_{50} of the most sensitive species to derive an interim guideline. The recommended safety factor for imidacloprid is 0.05 because imidacloprid is considered relatively unpersistent in water.

Therefore, in accordance with the protocol (CCME 1991), the interim marine aquatic life guideline is calculated as follows:

$IWQG_{MAL} = LC_{50}$ x AF
= 13 x 0.05
= 0.65 μg a.i./L

where,

$IWQG_{MAL}$ = Interim Water quality Guideline for the Protection of Aquatic Life
AF   = Application factor

Therefore, the interim marine water quality guideline for the protection of aquatic life is 0.65 μg a.i./L

Figure 4 presents a selection of the most sensitive marine toxicity data that were available and illustrates where the critical study and resulting guideline values fall with respect to the distribution of other sensitive endpoints.

Toxicity Information		Species	Toxicity endpoint	Concentration (µg·L⁻¹)
Acute	Vertebrates	*C. variegatus*	*96-h LC$_{50}$*	
Acute	Invertebrates	*Artemia* sp. *M. bahia* *A. taeniorhynchus*	*48-h LC$_{50}$* *96-h LC$_{50}$* *48-h LC$_{50}$*	
Chronic	Fish	*M. beryllina* *M. beryllina*	*7-d LOEC* *7-d LOEC*	
Canadian Water Quality Guideline 0.65 µg·L⁻¹				

Toxicity endpoints:
- ■ primary
- □ secondary
- ● critical value

Canadian Guideline

10^{-1} 10^1 10^3 10^5 10^7
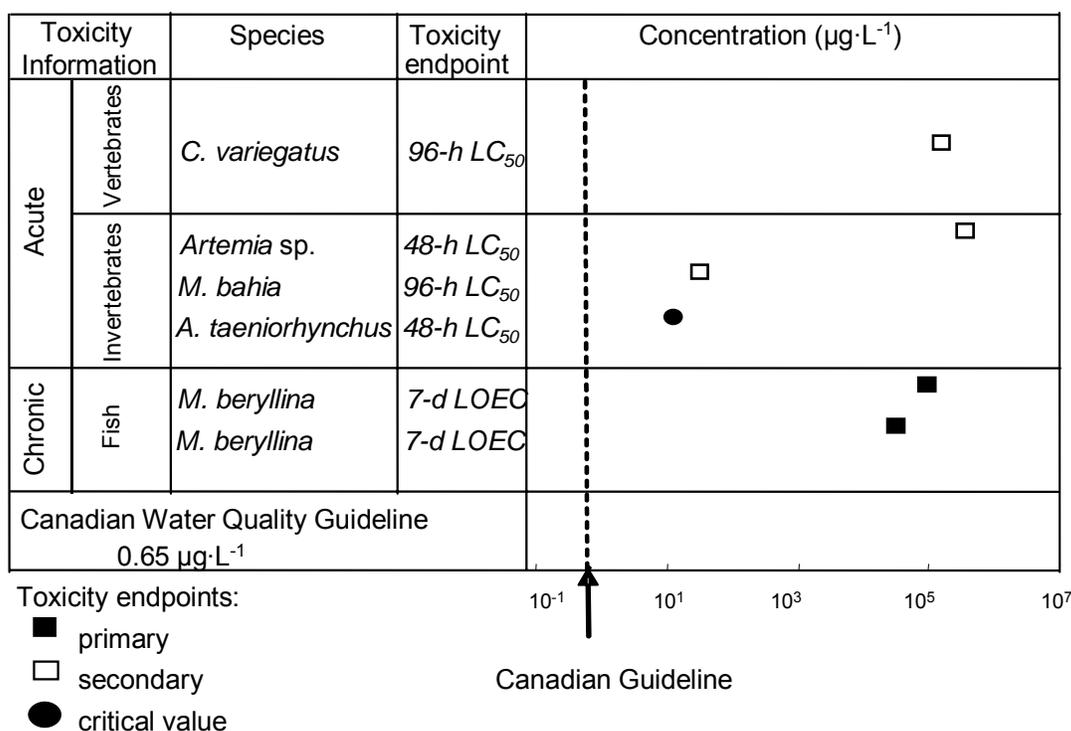
Figure 4. Select marine toxicity data for imidacloprid.

10.3 Protection of Irrigation Water

The imidacloprid toxicity data available for terrestrial plants are compiled in Appendix C. Insufficient data exist on the toxicity of imidacloprid to cereal, tame hay and pasture crops to derive either a full or interim guideline for the protection of irrigation water at this time. According to the protocol, the data needs include three or more species of cereals, tame hays, or pastures grown in Canada, using an appropriate method of administration relevant to irrigated water. At least two of these must be chronic tests that consider sensitive and biologically relevant endpoints. For this requirement, long-term irrigation studies are recommended. In addition, at least 3 studies are required on five or more crop species grown in Canada, including at least two of *Leguminosae, Compositae, Cruciferae, Cucurbitaceae, Liliaceae, Solanaceae, Umbelliferae, Chenopodiaceae*, with at least two of these being chronic tests.

To derive an interim guideline, two studies on two or more species of cereals, tame hays, or pastures grown in Canada are needed. In addition, at least 2 studies on two or more crop species grown in Canada, including at least two of *Leguminosae, Compositae, Cruciferae, Cucurbitaceae, Liliaceae, Solanaceae, Umbelliferae, Chenopodiaceae* are required.

10.4 Protection of Livestock Water

The imidacloprid toxicity data available for mammals and birds are compiled in Appendix D. There are insufficient data to develop either a full or interim water quality guideline for the protection of livestock water as outlined in the protocol (CCME 1993).

The data needs to develop a full guideline include three studies on mammalian species, two of which are livestock species raised in Canada, including one ruminant livestock species, and two of these studies must be chronic. A bioaccumulation study is also required for imidacloprid and its metabolites in livestock species. In addition, a minimum of two studies on avian species are required, at least one of which is a domestic poultry species raised in Canada, with at least one of those studies being chronic. The data requirements for an interim guideline for livestock water consist of two acute or chronic studies on two mammalian species raised in Canada, with one study on a livestock species and one acute or chronic study on an avian livestock species raised in Canada.

In mammalian systems, imidacloprid was found to be quickly eliminated within 48 hours of administration (Tomlin 1994; PMRA 2001). However, metabolites have been reported in the tissue and organs of farm animals (Tomlin 1994). Moreover, mobility data and the ultimate environmental compartment should be further investigated.

The derivation of the livestock water quality criterion should follow the procedure for a non-carcinogenic substance. Imidacloprid is currently considered a non-carcinogenic substance (U.S. EPA 1995).

10.5 Data Gaps and Research Recommendations

Additional toxicity data are needed for imidacloprid in order to meet the requirements for a full freshwater life guideline. These include two primary fish studies, in which at least one is chronic and one is for a warmwater species.

For a full marine guideline, a primary study on an algae or vascular plant species, two primary chronic invertebrate studies (for species of separate classes), and two primary studies on fish, at least one of which is chronic, are required.

The data requirements for a full livestock guideline consist of information from a chronic study on domestic poultry and three mammalian studies, including a ruminant and two livestock species raised in Canada. Therefore, the assessment of the chronic toxicity of the parent compound, with its metabolites, to one domestic poultry species raised in Canada, is recommended.

To develop a full irrigation water guideline, the data gaps are as follows: at least two studies on cereal crops, pastures, or tame hay and at least two long-term studies on two other crops.

11. GUIDANCE ON APPLICATION OF THE GUIDELINES

11.1 General Guidance on the Use of Guidelines

Canadian Water Quality Guidelines (CWQGs) are numerical concentrations or narrative statements that are recommended as levels that should result in negligible risk of adverse effects to aquatic biota. As recommendations, the CWQGs are not legally enforceable limits, though they may form the scientific basis for legislation or regulation at the provincial, territorial, or municipal level. CWQGs may also be used as benchmarks or targets in the assessment and remediation of contaminated sites, as tools to evaluate the effectiveness of point-source controls, or as "alert levels" to identify potential risks.

CWQG values are calculated conservatively, such that they protect the most sensitive life stage of the most sensitive aquatic species over the long-term. Hence, concentrations of a parameter that are less than the applicable CWQGs are not expected to cause any adverse effect on aquatic life. Concentrations that exceed the CWQGs, however, do not necessarily imply that aquatic biota will be adversely affected, or that the water body is impaired; the concentration at which such effects occur may differ depending on site-specific conditions. Where the CWQGs are exceeded, professional advice should be sought in interpreting such results.   As with other CWQGs, the guidelines for imidacloprid are intended to be applied towards concentrations in ambient surface waters, rather than immediately adjacent to point sources such as municipal or industrial effluent outfalls.

11.2 Detection Levels

The recommended guidelines for imidacloprid may be lower than the detection limits of some analytical methods. Therefore, in order to determine whether concentrations of imidacloprid in water samples exceed the guidelines or not, it is recommended that a method with a detection limit of 0.1 µg/L or lower be used. For examples of analytical methods with sufficiently low detection levels, refer to König (1997), Byrtus et al. (2002), Giroux (2003), Stoughton (2006), or Culp (2006).

11.3 Developing Site-Specific Objectives

In comparing analytical measurements of water samples with the CWQGs, it is generally recommended that total concentrations from unfiltered water samples be determined. It should be noted, however, that because imidacloprid has a tendency to bind to sediment, the presence of high levels of organic matter may decrease the bioavailability of the chemical. Therefore a site-specific guideline could be considered for waters with high levels of suspended sediments in the water column.

11.4 Best Management Practices

Direct application of imidacloprid to surface waters is not permitted in Canada. However, detectable concentrations of imidacloprid in aquatic systems can result from runoff and/or spray drift. Application instructions and mitigation measures specified on product labels, such as spray buffer zones, must always be followed. In addition, the use of best management practices can further reduce the potential contamination of aquatic systems by pesticides.

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