BACWA
BAY AREA
CLEAN WATER
AGENCIES

July 19, 2024

Areej Jahangir, Chemical Review Manager
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U.S. Environmental Protection Agency (EPA)
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Subject: 3-Iodo-2-propynyl butylcarbamate (IPBC) Registration Review Draft Risk Assessment (EPA–HQ–OPP–2011–0420)

Dear Areej Jahangir:

On behalf of the Bay Area Clean Water Agencies (BACWA), we thank you for the opportunity to provide comments on the 3-Iodo-2-propynyl butylcarbamate (IPBC) Draft Risk Assessment (Draft RA).

BACWA is a joint powers agency whose members own and operate publicly owned treatment works (POTWs) and sanitary sewer systems that collectively provide sanitary services to over seven million people in the nine-county San Francisco Bay Area (Bay Area). We take our responsibilities for safeguarding receiving waters seriously.

Every day, BACWA members provide wastewater treatment for millions of gallons of wastewater that is discharged to fresh or saltwater bodies, including local creeks and rivers, bays, and the Pacific Ocean. These waterways provide crucial habitat to a wide array of aquatic species and waterfowl, including several endangered species. In some cases, waters receiving POTW discharges ("receiving waters") may be effluent dominated in that there is little to no dilution, either because the receiving water is small or there is a lack of mixing at certain times due to thermal or saline stratification.

IPBC is an antimicrobial and fungicidal pesticide that is used ubiquitously in the indoor environment including in soaps, detergents, paints, wood preservative stains, caulks, sealants, adhesives, air fresheners, surface cleaners, ready-to-use wipes, floor care products, bathroom cleaners, window cleaners, fabric care products (including stain removers and fabric softeners), preserved floor cleaners, household items, clothing manufactured from treated textiles, pressure treated wood, PVC floor materials, carpet, industrial uses (including paper mills and metal working), plastics, textiles, and many other uses. [Draft RA, pp. 13-15]

Estimating aquatic exposures to IPBC in POTW effluents

In the Draft RA, EPA limited down-the-drain indoor modeling to industrial uses (pulp/papermill and metalworking), stating that these uses "*were modeled because they are expected to have the most direct route of aquatic environmental exposures and, thus, would be protective of other* *uses expected to go down-the-drain. Therefore, environmental uses modeled here are assumed to be protective of other registered uses.*" [Draft RA, p. 101]

EPA has been evaluating POTW discharges from indoor pesticide use and discharges to the sewer system in its pesticides risk assessments since the late 1990s. As described by an EPA scientific team (Shamim et al. 2014), EPA uses simplified models like its Exposure and Fate Assessment Screening Tool (E-FAST) in combination with monitoring data and benchtop studies to estimate POTW effluent concentrations.[1] As EPA noted in its pyrethroids ecological risk assessment,[2] this modeling approach is imperfect, but in combination with monitoring data it has been useful in understanding aquatic risks. Since the mid-2010s, BACWA has been in dialog with EPA scientists to improve the accuracy of EPA's POTW modeling approach, focusing on improvements that can be made with relatively small investments of EPA's scientific staff resources. We appreciate that some of our recommendations, (e.g., effluent dilution factors that recognize the prevalence of "zero dilution" discharges) have been implemented in the current version of the E-FAST model.

There are several indoor IPBC uses that have a direct down-the-drain pathway including (but not limited to) dish soap, laundry detergent, fabric care products (including stain removers, and fabric softeners), and household cleaners. Based on the very limited public data around concentrations of IPBC in these products [Draft RA, pp. 14-15], we are concerned that the IPBC discharges from cumulative residential and commercial use of these products alone could pass IPBC through municipal wastewater treatment plants at concentrations sufficient to be a toxicity concern to aquatic life. The RA does not contain sufficient information for us to estimate these discharges nor to determine which product categories generate the largest IPBC discharges. BACWA asks EPA to expand modeling to non-industrial indoor uses of IPBC, including these products.

POTW modeling is needed to inform POTW-specific mitigation measures

Local agencies in most states (including California) lack the statutory authority to regulate pesticide use in urban areas. Most IPBC uses are preservatives, many of which are in products that are distributed nationally, i.e., are manufactured in one state (where the pesticide is "applied") and sold in another state as an unlabeled component of an end-use product such as laundry detergent. These preserved products are exempted from regulation by state pesticide regulatory agencies because they lack pesticidal claims, leaving no practical tools for states to assist local agencies in compliance with the Clean Water Act and/or Endangered Species Act. Thus, it is essential that EPA employs the pesticide regulatory and Endangered Species Act consultation processes to assess and prevent urban water pollution as defined by the Clean Water Act and our NDPES permits.

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[1] Shamim, M. et al. 2014. Conducting Ecological Risk Assessments of Urban Pesticide Uses. In Jones et al. Describing the Behavior and Effects of Pesticides in Urban and Agricultural Settings; ACS Symposium Series 1168; American Chemical Society: Washington, DC, 2014; pp 207-274.

[2] EPA OPP EFED (2016). Preliminary Comparative Environmental Fate and Ecological Risk Assessment for the Registration Review of Eight Synthetic Pyrethroids and the Pyrethrins. Part I. Assessing Pyrethroid Releases to POTWs of Pyrethroids and Pyrethrins (DP Barcode D425791). Full document available online at https://downloads.regulations.gov/EPA-HQ-OPP-2010-0479-0022/content.pdf

If the pesticides reregistration process fails to identify and implement mitigation, an undue burden to address the problem is placed on local governments. Often, there are few ways for a POTW to mitigate a toxic pollutant problem other than extremely costly treatment plant upgrades. In addition, wastewater facilities may be subject to additional requirements established as part of Total Maximum Daily Loads (TMDLs) set for the water bodies by EPA and state water quality regulatory agencies. The cost to wastewater facilities and other dischargers to comply with TMDLs can be up to millions of dollars per water body per pollutant. It is therefore essential that pesticide registration and pesticide registration review processes adequately consider potential impacts to wastewater quality, so that such impacts to the beneficial uses of the receiving water are prevented (i.e., uses and/or discharges associated with endangered species impacts do not occur).

Since EPA controls pesticides labels, even our state pesticide regulatory agency cannot readily address pesticide water pollution and compliance with our NPDES permit if the pesticide discharges stem from consumer pesticide products. EPA action is imperative.

BACWA requests that EPA ensure IPBC uses affecting POTWs are addressed

Based on this example, we request that EPA ensure that it does not overlook the presence of antimicrobial pesticides from consumer and commercial products in POTW effluents. IPBC is used in a myriad of ways in the indoor environment. The comprehensive conceptual model shown below (from Sutton et al. 2019) can be used as the basis for this analysis. It identifies the pathways between pesticide uses and POTWs and illustrates how pesticides used indoors flow to the sewer system, to POTWs, and ultimately into the environment via effluent, air emissions, and biosolids.[3]

In BACWA's more than ten years of experience reviewing and commenting on EPA's pesticide registration/re-registration process, we have learned that if EPA does not model the specific down-the-drain pathway for a use-type, then it will not propose mitigations for that use-type. We are very concerned that the absence of adequate indoor use modeling and the lack of mitigations will result in direct harm to aquatic life from indoor uses of IPBC.

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[3] Sutton et al. (2019). Occurrence and Sources of Pesticides to Urban Wastewater and the Environment. In K. Goh (Ed.), Pesticides in Surface Water: Monitoring, Modeling, Risk Assessment, and Management (pp. 63-88). Washington, DC: American Chemical Society.

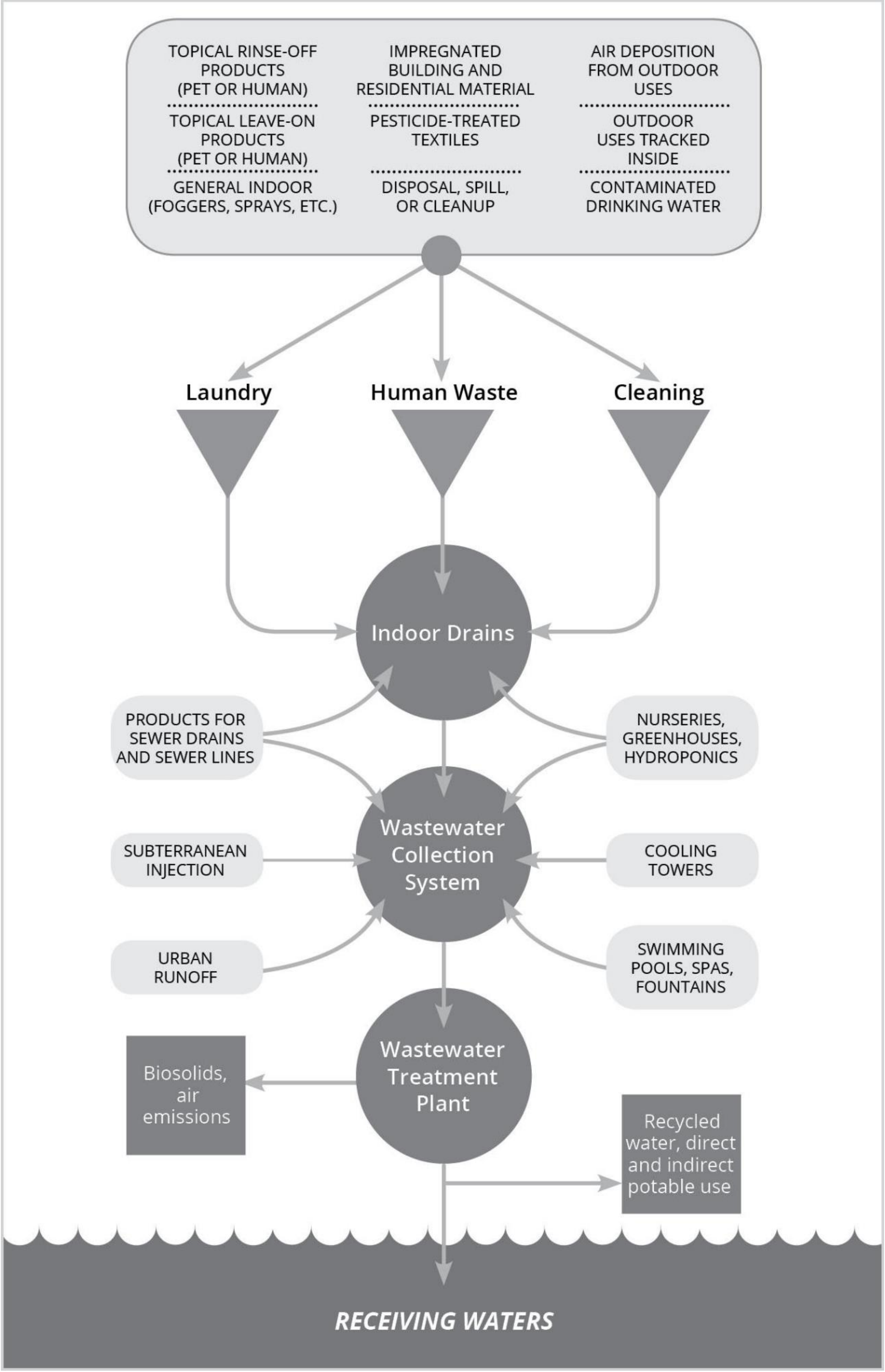

Figure source: Sutton et al., 2019

Significant gaps in IPBC ecotoxicity data are concerning
The March 2024 IPBC Draft RA documented the significant risks IPBC poses to aquatic life. IPBC is highly to very highly toxic to freshwater fish, highly toxic to freshwater invertebrates, highly toxic to estuarine/marine fish, and highly to very highly toxic to estuarine/marine invertebrates. [Draft RA, p. 9]

However, several data gaps in the IPBC ecotoxicity database were acknowledged in the Draft RA:

- Chronic ecotoxicity endpoints for freshwater invertebrates: The Draft RA noted that "there are no acceptable chronic ecotoxicity endpoints available for IPBC on freshwater invertebrates." This is particularly important because "sublethal effects (time to first brood) were observed at the lowest concentration tested (3.0 µg a.i./L)" [Draft RA, p.87]

- Ecotoxicity endpoints for aquatic vascular plants: In the Draft RA, EPA noted that "to quantitively assess the chronic risks to freshwater invertebrates, and risks to aquatic vascular plants, new freshwater invertebrate life cycle and aquatic vascular plant studies would be needed." [Draft RA, p.87]

- Ecotoxicity endpoints for benthic species: EPA noted that "(n)o exposure or ecotoxicity data are available for benthic species, however, IPBC is not expected to accumulate in the benthos or soil given its water solubility, mobility in soil, and its rapid biodegradation expected in sediment." [Draft RA, p.103]

  However, these factors—while arguing against the likelihood of chronic effects on benthic species—do not preclude the possibility of acute toxicity to benthic invertebrates, particularly when IPBC is known to be highly toxic to aquatic invertebrates [Draft RA, p. 86] and in EPA's risk assessment was shown to exceed Levels of Concern (LOCs) for aquatic invertebrates in modeling of outdoor uses. [Draft RA pp. 96-99]

  Therefore, acute (and ideally also chronic) ecotoxicity studies should be required from the registrants for freshwater and marine benthic invertebrates, and EPA should incorporate the results into exposure modeling for outdoor uses of IPBC on these non-target organisms.

Given the other impacts to aquatic life noted in the Draft RA, these toxicity endpoints are necessary for a more robust understanding of the potential impacts of IPBC in the aquatic environment. BACWA recommends that EPA require registrants to submit the missing ecotoxicity data as noted above for freshwater invertebrates, aquatic vascular plants, and benthic species, and reevaluate risk to aquatic life with this information included.

Additional scientific studies identified by BACWA may be of use to EPA
BACWA has identified other study data for IPBC that may be of interest to EPA:

1) Norström et al. 2008[4] found higher levels of IPBC in POTW effluent than influent at two

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[4] Karin Norström, Mikael Remberger, Lennart Kaj, Per Wiklund, Eva Brorström-Lundén, Results from the Swedish National Screening Programme 2008. Subreport 1. Biocides: 3-Iodo-2-propynyl butyl carbamate (IPBC) and 2,2-

out of three sampled urban wastewater treatment plants.
2) Jarrard et al. 2004[5] and Tierney et al. 2006[6] identified effects of IPBC on coho salmon olfactory function at concentrations lower than ~1 µg/L.
3) Several other governments have done RAs of IPBC, including Canada[7], Australia[8], and the EU[9]. Canada has established a water quality guideline of 1.9 µg/L.

We have attached these studies and reports for your use, with the exception of Jarrard et al. which could not be obtained without a license.

Thank you for your consideration of our comments. If you have any questions, please contact BACWA's Project Managers:

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Respectfully Submitted,

Lorien Fono

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dibromo-2-cyanoacetamide (DBNPA), https://www.ivl.se/english/ivl/publications/publications/results-from-the-swedish-national-screening-programme-2008.-subreport-1.-biocides-3-iodo-2-propynyl-butyl-carbamate-ipbc-and-22-dibromo-2-cyanoacetamide-dbnpa.html

[5] H.E Jarrard, K.R Delaney, C.J Kennedy, Impacts of carbamate pesticides on olfactory neurophysiology and cholinesterase activity in coho salmon (Oncorhynchus kisutch), Aquatic Toxicology, Volume 69, Issue 2, 2004, Pages 133-148, ISSN 0166-445X, https://doi.org/10.1016/j.aquatox.2004.05.001.

[6] Keith B. Tierney, Amber L. Taylor, Peter S. Ross, Christopher J. Kennedy, The alarm reaction of coho salmon parr is impaired by the carbamate fungicide IPBC, Aquatic Toxicology, Volume 79, Issue 2, 2006, Pages 149-157, ISSN 0166-445X, https://doi.org/10.1016/j.aquatox.2006.06.003.

[7] IPBC, Canadian Water Quality Guidelines for the Protection of Aquatic Life, Canadian Council of Ministers of the Environment, https://ccme.ca/en/res/ipbc-en-canadian-water-quality-guidelines-for-the-protection-of-aquatic-life.pdf

[8] Carbamic acid, butyl-, 3-iodo-2propynyl ester (IPBC) Evaluation statement 22 December 2022, Australian Government, https://www.industrialchemicals.gov.au/sites/default/files/2022-12/EVA00112%20-%20Evaluation%20statement%20-%2022%20December%202022.pdf

[9] Regulation (EU) n°528/2012 concerning the making available on the market and use of biocidal products Evaluation of active substances Assessment Report, IPBC, Product-type 6 (Preservatives for products during storage), September 2013.

Attachments:

1) Shamim, M. et al. 2014. Conducting Ecological Risk Assessments of Urban Pesticide Uses. In Jones et al. Describing the Behavior and Effects of Pesticides in Urban and Agricultural Settings; ACS Symposium Series 1168; American Chemical Society: Washington, DC, 2014; pp 207-274.
2) EPA OPP EFED (2016). Preliminary Comparative Environmental Fate and Ecological Risk Assessment for the Registration Review of Eight Synthetic Pyrethroids and the Pyrethrins. Part I. Assessing Pyrethroid Releases to POTWs of Pyrethroids and Pyrethrins (DP Barcode D425791). Full document available online at https://downloads.regulations.gov/EPA-HQ-OPP-2010-0479-0022/content.pdf
3) Sutton et al. (2019). Occurrence and Sources of Pesticides to Urban Wastewater and the Environment. In K. Goh (Ed.), Pesticides in Surface Water: Monitoring, Modeling, Risk Assessment, and Management (pp. 63-88). Washington, DC: American Chemical Society.
4) Karin Norström, Mikael Remberger, Lennart Kaj, Per Wiklund, Eva Brorström-Lundén, Results from the Swedish National Screening Programme 2008. Subreport 1. Biocides: 3-Iodo-2-propynyl butyl carbamate (IPBC) and 2,2-dibromo-2-cyanoacetamide (DBNPA).
5) Keith B. Tierney, Amber L. Taylor, Peter S. Ross, Christopher J. Kennedy, The alarm reaction of coho salmon parr is impaired by the carbamate fungicide IPBC, Aquatic Toxicology, Volume 79, Issue 2, 2006, Pages 149-157, ISSN 0166-445X.
6) IPBC, Canadian Water Quality Guidelines for the Protection of Aquatic Life, Canadian Council of Ministers of the Environment.
7) Carbamic acid, butyl-, 3-iodo-2propynyl ester (IPBC) Evaluation statement 22 December 2022, Australian Government.
8) Regulation (EU) n°528/2012 concerning the making available on the market and use of biocidal products Evaluation of active substances Assessment Report, IPBC, Product-type 6 (Preservatives for products during storage), September 2013.

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Chapter 10

Conducting Ecological Risk Assessments of Urban Pesticide Uses

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Recent studies have reported pesticides in toxicologically significant concentrations in surface water, sediments, stormwater, and publicly owned treatment works (POTW) influent/effluent wastewater from residential uses at locations across the United States. The USEPA faces many challenges in assessing the ecological risks from indoor and outdoor residential pesticide uses, many of which stem from limitations in quantifying exposure from the wide array of application scenarios available for residential pesticide use. Data on the timing, frequency and location of residential pesticide application at a national scale has been collected and submitted to the USEPA. These data will be useful for constructing representative residential exposure scenarios. In the absence of these data and tools, the USEPA has relied on urban monitoring data for conducting the ecological risk assessments. The use of certain chemicals as mosquito adulticides has resulted in exposure and risk to non-target aquatic organisms. Various methods and approaches to assess exposure are presented to conduct ecological risk assessments of these insecticides. Pesticides released to domestic wastewater from indoor residential uses are being assessed with the Exposure and Fate Assessment Screening Tool (E-FAST). Bench-scale treatability studies and POTW monitoring data will be used to refine exposure estimates of pesticides in wastewater, surface water and biosolids resulting from indoor uses.

In Describing the Behavior and Effects of Pesticides in Urban and Agricultural Settings; Jones, et al.;
ACS Symposium Series; American Chemical Society: Washington, DC, 2014.

Introduction

Urban uses of pesticides are widespread and their use patterns present many challenges in conducting a national scale ecological risk assessment (ERA). Pesticides are used outdoor and indoor in residential, public, commercial, industrial and military areas. In California alone, nearly ten million pounds of pesticides active ingredients were used in the year 2009 (California Department of Pesticide Regulations Pesticides Use Reporting or CDPR PUR database (*1*)). Use in urban areas includes nearly thirty PUR categories with the top five being structural pest control, rights-of-ways, public health, landscaping, and indoor homeowner use.

Ecological risks associated with urban uses of pesticides is a critical emerging issue. As highlighted by the 2007 USGS report "*The Quality of our Nation's Waters* (*2*)," urban streams have the highest frequency of U.S. stream sites with pesticide concentrations that exceed aquatic life benchmarks (83%). Agriculturally dominated streams had the next highest frequency of aquatic life benchmark exceedance (57%), followed by mixed use streams (42%) and undeveloped sites (13%). This chapter describes major risk assessment challenges and approaches being considered by USEPA for assessing ecological risks from urban/residential pesticide uses. Specifically, three residential/urban assessment scenarios are described: (1) stormwater discharges resulting from outdoor uses; (2) exposure from adulticide uses; and (3) releases to POTWs (waste water discharges) from indoor uses. Within each of these assessment scenarios, the available methods and data being considered for modeling pesticide exposure and risk are summarized. In addition, the results from selected model-based assessments are compared to available information from targeted pesticide monitoring studies.

Assessing Stormwater Discharges from Outdoor Urban Uses

Outdoor urban uses of pesticides can result in significant exposure to water bodies through drift and runoff. These uses include structural pest control, rights of ways, and landscaping. Many pesticides are labeled for outdoor uses to control insect pests such as ants, cockroaches, fleas, occasional invaders, spiders, and wasps, in addition to others used for lawn care. Control is accomplished by professional pest control operators (PCOs) and homeowners through different pesticide formulations, application methods, and timing.

Many types of documentation, information and data are used by USEPA in conducting the ecological risk assessment for all pesticides including those used outdoors in urban settings. In a regulatory setting, labels are considered first in determining pesticide exposure in various compartments of the environment, as the label is the legal document governing the permitted pesticide use patterns. Labels specify pesticide contents of active(s)/inert material(s), formulation type, target pests/areas, and detailed use instructions (application rate, number of applications permitted, frequency, timing and type of applications. In addition to label use information, pesticide usage data are also important as it indicates quantity, seasonality, historical and geographic usage extent of currently registered pesticides. Monitoring data are also important and could be the only reliable exposure data available for use in a risk assessment due to limitations associated with the current modeling approaches. Important aspects of exposure modeling uncertainties for outdoors uses include establishing a conceptual model for varied types of outdoor uses along with percent/type of areas treated, percent of pesticide available for washoff, and other possible sources of pesticide contamination (*i.e.*, drift, contaminated airborne particles and others). As discussed in more detail later, recent studies have concentrated in obtaining such important modeling parameters in addition to many other data such as frequency/seasonality of applications, and most frequently used application rate, frequency, equipment and formulations. This data could be used as inputs for the exposure models to characterize and refine the exposure estimates.

Use Characterization

Early CDPR Surveys (2001-2005)

The California Department of Pesticide Regulation (CDPR) funded a number of use and usage surveys between 2001 and 2005 to get a better understanding of the pesticide use pattern in urban environments. The 2001 survey (*3*), involved the San Diego Creek and East Costa Mesa/Newport Beach watershed areas of Orange County, CA. A majority of the surveyed people that apply pesticide products (58.3%) reported applications one to three times, or four to six times per year. Another survey was conducted in 2002, of residents of the Chollas Creek area of San Diego County and the Delhi Channel area in Orange County (*4*). Ants and other insects were the primary target pests. The most frequent use pattern of pesticide application was once every few months (43.1%). Of the responses, 47.2% indicated that they had purchased or used a weed control product, 77.1% indicated that they purchased or used an insecticide, and 32.5% indicated they had purchased or used a product to control plant diseases. The 2003 survey covered the areas of the Arcade Creek watershed in Sacramento, Five Mile Slough watershed in Stockton and San Francisco Bay (*5*). From 20-41% indicated they did not apply pesticides in their homes and 37-65% of respondents identified insects as their primary pest of concern. Other pests included snails/slugs (24.4-29.2%) and vertebrates (15-27%). The majority (58-64%) indicated they applied pesticides on hard surfaces such as perimeters of buildings, driveways, sidewalks, or walls; further 44-47% responded that they applied pesticides 1-3 times per year.

The previous surveys examined residential users of pesticides; in contrast, a 2005 survey (*6*) evaluated pesticide use by pesticide managers and applicators in three urban watersheds: Arcade Creek (Sacramento County), Chollas Creek (San Diego County), and Upper Newport Bay/San Diego Creek (Orange County), CA. The CDPR PUR Report database indicated that in 2003 structural PCO use comprised 40% of the total reported non-agricultural use, rights-of-ways (32%), landscape maintenance (15%), public health (12%), and regulatory pest control (1%) in Sacramento, Orange and San Diego Counties. Structural pest control comprised 93-98% of the total insecticide usage. An analysis of usage indicated that organophosphates had been declining and pyrethroids increasing. Rights-of-ways accounted for 47-60% of the total herbicide use. The top herbicides used were glyphosate and diuron. Landscape maintenance reported 38-53% of the total herbicide use. The most commonly applied herbicide was glyphosate. San Diego County was the major urban pesticide user (48%), followed by Orange County and Sacramento County.

Pyrethroid Working Group Use Surveys (2009-2013)

In response to concerns over increasing pyrethroid use and detections in California, a survey was conducted by Pyrethroid Working Group (PWG) for CDPR in 2009 (MRID 48762913 (*7*)), which assessed pesticide usage by professional pest management companies. Outdoor usage represented 83% of the total pounds of pesticides applied in urban environment, with indoor usage constituting the balance. Application frequency was monthly or every other month for residential customers (80% of responses) and monthly for commercial customers (83% of responses) (Table 1). For outdoor use, the dominant type of formulations used were liquid sprays (liquids 95% and wettable powder 2%); granules represented 3%, with very small amounts of baits. The most common equipment used in applying liquid sprays included power sprayers, followed by handheld or back pack sprayers. Granular products were most often used in broadcast application. Treatment types included home or fence perimeter treatments (1-2 feet up and 1-5 feet out with 1x1 ft being the most common) and/or spot treatment while treatment of the entire yard was less common. Hard surfaces such as patios, outdoor congregation areas and driveways were almost always treated. Less commonly treated areas include vertical walls and uncovered storage. Pest management professionals were asked to name the “Top 5” pesticide products they used, based on volume. The product most commonly named was Termidor (fipronil, named by 73% of respondents). The named products were related to their corresponding active ingredients, which included bifenthrin, fipronil, and deltamethrin (named among the “Top 5” by 60-75% of the pest management professionals surveyed); followed by indoxacarb, *beta*-cyfluthrin, permethrin, cyfluthrin, cypermethrin, *lambda*-cyhalothrin and chlorfenapyr (named among the “Top 5” used by 22-33% of the pest management professionals surveyed); and thiamethoxam, abamectin, and pyriproxyfen (named among the “Top 5” by 2-10% of the pest operators surveyed). Timing of application for most compounds was found to be throughout the year although few compounds were applied more often either in spring and winter or in the summer.

Another survey of PCOs and LCOs was sponsored by PWG (Winchell and Cyr, MRID 49292101 (*8*)). The survey covered six national regions, excluding California and included both pest control operators (PCOs) and lawn care operators (LCOs). Pyrethroids were associated with 58% of the outdoor insecticide applications overall for all regions. Overall, for all regions the percentage uses were bifenthrin (40%), cyfluthrin/*beta*-cyfluthrin (17%), *lambda*-cyhalothrin (12%), deltamethrin (11%), permethrin (9%), cypermethrin (8%), and other pyrethroids (2%). The percent of LCOs and PCOs that applied each pyrethroid active ingredient, by use site, is depicted in Figure 1. Seven types of surfaces were investigated of which only a selection is presented in the figure.

Table 1. Service Interval for Residential and Commercial Pesticide Accounts

Service Interval	*Residential (%)*	*Commercial (%)*
Weekly	4	6
Monthly	39	83
Every other month	41	7
Quarterly	12	0
Other	0	4

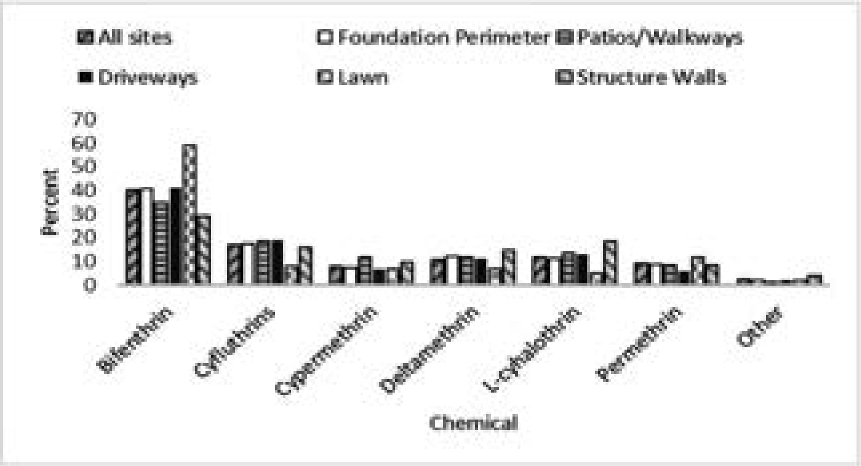

Figure 1. Percent of Respondents' Pyrethroid Active Ingredient Use in Outdoor Applications by Selected Use Sites, Excluding California.

The percent applying pyrethroids to different types of surfaces in an urban environment, including California, is depicted in Figure 2. By far, the foundation perimeter treatments are the most commonly applied by PCOs. Note that all regions but California receive approximately the same number of building foundation perimeter treatments. Meanwhile, lawn treatments are lower. The methodology to estimate California use was different since the questions asked to PCOs and LCOs were different. The foundation perimeters treatment represented an estimated value since this specific question was not asked in California.

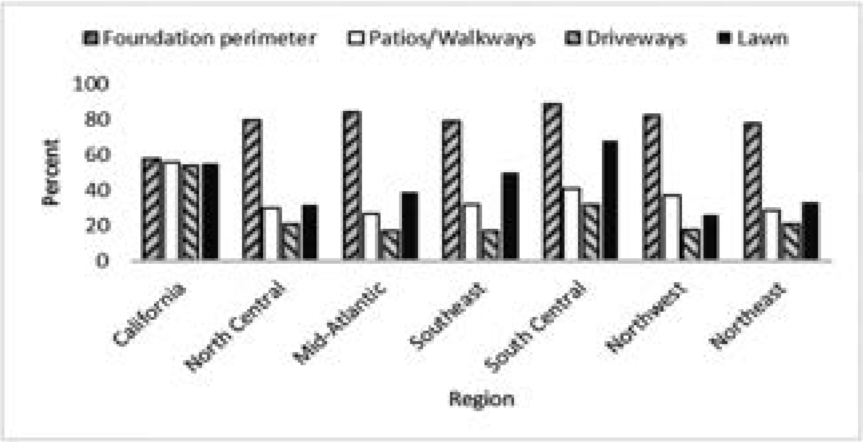

Figure 2. Percent of PCOs and LCOs Applying Pyrethroids to Selected Sites by Region.

The number of applications per year, average area treated, and the active ingredient most commonly used on each of the use sites for all regions, except CA, is summarized in Table 2. Each use site receives on average close to 4 applications per year although the foundation perimeters are treated more often than other use sites, and the fraction of the use site ranges from 36% (driveways away from the garage door or wall) to 77% for lawns. The active ingredient most commonly applied is bifenthrin, irrespective of the use site.

Figure 3 summarized for each active ingredient, the frequency by which PCOs and LCOs responded that they used each active ingredient for each region. This figure confirms that bifenthrin is the active ingredient most commonly used. Note the high use of cypermethrin in the south central region, compared to the other regions. Approximately a two-fold increase of cypermethrin applied as compared to other regions, is unexplained at this time.

These surveys were supplemented by work by Fugate and Hall (*9*), which includes frequency of consumer use of specific insecticides, in and around homes, outdoor non-plant, and lawn and garden in 2011. (This report was not provided to the USEPA. Rather, certain data were extracted and provided in MRIDs 49292101 (*8*) and 49292102 (*10*)). Nationally, the likelihood of consumer use of LCO services to apply fertilizer and chemicals is 14% and consumer use of PCO services is 26%. The likelihood of a consumer to purchase lawn and garden insecticides is 31% and outdoor non-plant insecticides is 15%. The likelihood of a consumer applying lawn and garden insecticide is 47% and outdoor non-plant insecticides is 28%. Bifenthrin is the insecticide most likely to be purchased, followed by *lambda*-cyhalothrin.

Table 2. Averages of Treatments per Year, Fraction of Use Site Surface Area Treated and Pyrethroid Active Ingredient Most Commonly Used by Use Site in Six National Regions, Excluding California

Use Site Type of Surface	*Average Number of Treatments Per Year*	*Fraction of Use Site Surface Area Treated*	*Most Commonly Used Active Ingredient*
Building foundation perimeters	4.25	2.4 ft up; 2.9 ft out	Bifenthrin
Patios and walkways away from building	3.73	44%	Bifenthrin
Driveways away from the garage door and wall	3.66	36%	Bifenthrin
Lawn	3.62	77%	Bifenthrin
Landscape and ornamental areas	3.82	63%	Bifenthrin
Structure walls	3.71	42%	Bifenthrin
Eaves	3.38	44%	Bifenthrin

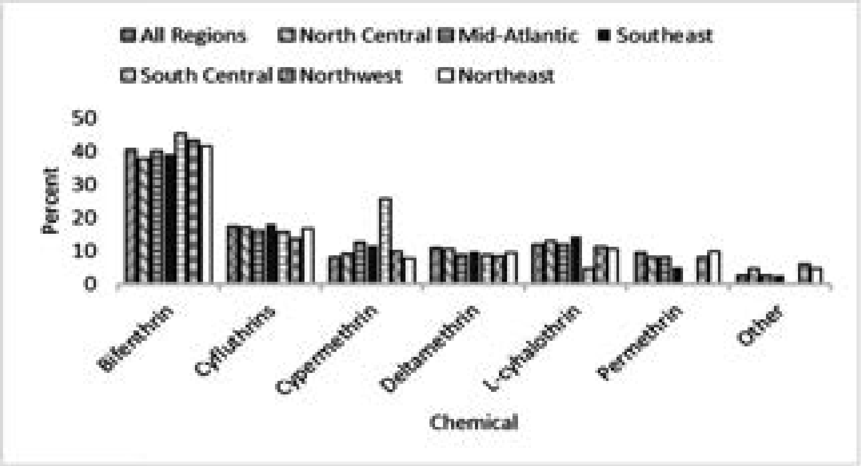

Figure 3. Percent of Respondents' Pyrethroid Active Ingredient Use in Outdoor Applications by Region, Excluding California.

Winchell (*10*) (MRID 49292102 (*10*)) provided an interpretation of the following studies: MRIDs 48762913 (*7*), 49292101 (*8*), Wilen (*3*), and the work by Fugate and Hall (*9*). Winchell used certain data manipulations to derive suitable variables, with the potential to be useful in modeling for aquatic exposure in an urban environment. These manipulations were different for CA and other regions of the U.S. due to differences in survey design. These variables for aquatic modeling include 1) the fraction of the use site treated with each active ingredient; 2) the seasonal application frequency made to each use site; and, 3) the percentage of the use site's surface area that is treated. The work by

Fugate and Hall (*9*) helped to establish the extent of pyrethroid use in different geographical regions of the U.S. (compared to other insecticides), and the fraction of the households receiving pyrethroid applications outdoors (including LCOs, PCOs, and resident's applications), and to compare against the 2010 and 2013 results. Regarding the frequency of applications, it was estimated that in CA, it ranged from 4-8 per year, while in other regions of the U.S., it ranged from 4-5 per year. The percentage of the use site surface area, treated with pyrethroids was not asked in the CA surveys and data for other regions of the U.S. would be used to cover CA.

Value of Surveys

These reports include data on the frequency of homeowners using lawn care or pest control services, the frequency of consumers using outdoor non-plant and lawn & garden insecticides, and data on the frequency of a consumer using specific insecticide active ingredients. The datasets provided the starting point to determine the overall likelihood of an individual homeowner using an applicator service, and then from the survey responses, determine the likelihood by region and use site of the top six pyrethroids being used by both professional applicators and/or homeowners themselves.

Of all the above surveys, it is apparent that the most recent ones, conducted in 2009 and 2013 (MRIDs 48762913 (*7*), 49292101 (*8*), and 49292102 (*10*)), with supplemental data from Wilen (*3*), and Fugate and Hall (*9*), may be used to estimate the needed usage and the amount of pesticide applied on each use site per region. The studies have the potential to establish the conceptual model for outdoor pesticide exposure for a variety of outdoor use sites, along with percent/type of areas treated, and, with the help of the washoff studies, the percent of pesticide available for wash-off, and other possible sources of pesticide contamination (*i.e.*, drift, contaminated airborne particles and others). But more than that, they could be used in characterizing and refining exposure and in finding mitigation measures to reduce exposure, such as frequency/seasonality of applications, and most frequently used application rate, frequency, equipment, and formulations (typical application pattern), percent area treated by each use site, *etc*. Winchell (*10*) (MRID 49292102 (*10*)) synthetizes previous useful studies in tables that are suitable to do the above tasks for the pyrethroid insecticides.

Modeling Approach for Stormwater Discharges

The Environmental Fate and Effects Division (EFED) currently obtains estimated exposure concentrations (EECs) by modeling the residential and impervious scenarios in the Pesticide Root Zone Model coupled with the Exposure Analysis Modeling System (PRZM/EXAMS). Two PRZM/EXAMS runs are executed for each application type/weather combination. The application types are dependent on the label and may include three types of applications: (1) application to pervious areas alone with drift to adjacent impervious surfaces such as application to a lawn and/or garden adjacent to impervious driveway and/or porch; (2) application to impervious surfaces alone with drift to adjacent pervious surfaces such as application to driveway and/or porch adjacent to a lawn and/or garden; or (3) a combined application to pervious and impervious surfaces such as application to both impervious driveway and/or porch and to the lawn and/or garden).

At the present time, the CA impervious PRZM scenario is considered as the most suitable available modeling approach for impervious runoff. The PRZM CA impervious scenario may be used in the Tier 2 coupled aquatic models PRZM/EXAMS along with the CA residential or other appropriate scenario such as CA rights-of-ways (ROW) to obtain EECs. The "residential" and various other "urban" use patterns require the PRZM CA residential and CA impervious scenarios for modeling. Both scenarios are run separately. This approach assumes that no watershed is completely covered by either the ¼ acre lot (the basis for the residential scenario) or undeveloped land (the basis for the ROW scenario), for residential and ROW use patterns, respectively. By modeling a separate scenario for impervious surfaces, it is also possible to estimate the amount of exposure that could occur when the pesticide is over-sprayed onto this surface. Using two scenarios in tandem requires post-processing of the modeled output in order to derive a weighted EEC that represents the contribution of both the pervious (*i.e.*, residential and ROW scenarios) and the impervious surfaces. Exposure from both scenarios can also be weighted and aggregated. The second critical assumption is that 50% of a ¼ acre lot will be pervious and 50% impervious. In addition to the footprint of the typical house, it is assumed that a typical house would have a driveway of approximately 25 by 30 feet or 750 square feet and roughly 250 square feet of sidewalk. A typical suburban home could also be assumed to have roughly 300 square feet of deck space and 900 square feet of garage. Finally, it is assumed that a substantial portion of the typical home would be planted in landscaping (*e.g.*, residential lawn and/or ornamentals) with an estimate of 2,000 square feet. The sum of all these areas is 5,200 square feet. Taking a total ¼-acre lot size of 10,890 square feet and subtracting the house square footage yields a total remaining area of 5,690, or roughly 50% of the total lot untreated area. The residential and impervious scenarios are parameterized to represent a California urban site. For modeling uses in other metropolitan regions (not located in California), the residential and impervious scenarios can be run with meteorological files from other locations of the U.S.

Pathway Identification Study

The main objective of this study (Davidson *et al.*, MRID 49137401 (*11*); and Davidson *et al.* (*12*)) was to identify the major transport mechanisms of pyrethroids from a range of outdoor residential applications and determine the effects of mitigation measures put in place by the USEPA to control off-site transport. The study was conducted at a test facility which represented typical California residential developments. It consisted of six replicate house lots which included front lawns, stucco walls, garage doors, driveways and residential lawns. The off-site movement of different pyrethroids applied to these surfaces (representing pervious and impervious surfaces) was assessed using irrigation and simulated artificial rainfall to complement the natural rainfall events. The results showed that natural and simulated rainfall events contributed to the majority of mass loss compared to the mass loss due to lawn irrigation. Runoff losses expressed as a percentage of chemical applied were highest for the driveway and garage door surfaces compared to grass lawn, grass perimeter and house wall surfaces. Also, a comparison of historic applications with revised application due to label changes showed that the amount of losses from garage and driveway were dramatically reduced (40 times lower) using the revised application practices.

Washoff/Runoff Study from Impervious Surfaces

The main objective of the study was to examine the potential for simulated rain to washoff of a pyrethroid (cypermethrin) that had been applied to different external building materials using two different representative formulations (Trask *et al.* (*13*); MRID 48072902 (*14*)). The building materials selected were those typically used for construction of residential/urban structures in California that may receive applications of pyrethroids. These included: clean painted/unpainted concrete, clean painted/unpainted stucco, clean painted/unpainted wood with a dusty surface, clean vinyl/aluminum siding and clean asphalt. Washoff quantified as percent of applied mass of cypermethrin ranged from <0.01/0.07 to 16.8/11.3% for the two representative formulations. Clean vinyl siding had the highest percent of applied cypermethrin in runoff whereas clean unpainted stucco had the least amount of cypermethrin in washoff. All building materials had similar runoff volumes except for the clean asphalt which was lower in comparison.

Runoff Losses from Treated Turfgrass

In a study conducted in 2008, the authors examined the potential of pyrethroid insecticides uses on turf to contribute to residue detections in Sacramento, CA urban sediments, particularly due to over irrigation (*i.e.*, irrigation producing excess runoff) (Hanzas *et al.* (*15*); and MRID 47647801 (*16*)). Model pyrethroids included bifenthrin and *beta*-cyfluthrin in both granular and liquid formulations. Four treated turf plots were prepared, using normal irrigation or three over irrigation events. Runoff flow was measured during the irrigation events and runoff samples taken and analyzed for bifenthrin and *beta*-cyfluthrin. For the bifenthrin over irrigated plots, during the first irrigation event, 0.052-0.081% of the applied chemical was found in runoff, while no reported bifenthrin was found in the non-over irrigated plots. Meanwhile, for *beta*-cyfluthrin, 0.23-0.58% of the applied was found in runoff of the first over irrigation, with no runoff in the non-over irrigated plots. During the normal simulated rainfall event, simulating a winter storm, the amount of chemical present in runoff was much smaller (≤0.011% of the applied for all chemicals and formulations). It was noted that for *beta*-cyfluthrin, the majority of the chemical loss occurred during the first over irrigation event while for bifenthrin the loss was more evenly distributed across three over irrigation events, particularly for the granular formulation.

Monitoring of Urban Waters

Two recent extensive reviews are available on monitoring of urban pesticides in receiving water bodies in the United States, especially in California. The first review was submitted to US EPA by the PWG covering available data for synthetic pyrethroid in surface water and sediment in the United States (Giddings, *et al.*, MRID 49314703 (*17*)). The second review was conducted for the California Stormwater Quality Association (CASQA) and the County of Sacramento covering monitoring data from California urban watersheds on pyrethroids and fipronil toxicity (Ruby, MRID 49354001 (*18*)). This section deals with only few examples of targeted surface water/sediment monitoring data for pesticides used outdoors. Therefore, selected chemistry data are included herein with emphasis on pesticides used in urban areas and reaching surface waters mainly by urban runoff into surface waters (urban creeks and lakes and rivers passing through urban areas). Urban runoff water, contaminated with urban pesticides, is usually pumped, drained and/or naturally flow into these water bodies. Many factors will affect detected concentrations in these water bodies such as the pesticide physical/chemical and fate properties; labeled use patterns; pattern of timing of the application; application procedure; usage intensity (depends mainly on pest pressure which is associated with many factors such as climate); hydrological setting, urban drainage (sources/quantities); and characteristics of urban areas/receiving waters, climatic conditions. Effects of these factors, will be included when reported.

Monitoring of Stormwater Discharges and Affected Water Bodies

Urban areas stormwater discharges and affected water bodies were extensively monitored in California. Targeted monitoring data in these studies were for stormwater discharges and affected water bodies (water and underlying sediment). In the first study, monitoring data were for eight pyrethroids and the organophosphate insecticide chlorpyrifos (*19*). In the second study, monitoring data were for 63 insecticides/herbicides/degradates in the water column plus nine pyrethroids and chlorpyrifos in water and underlying sediment (*20*). For northern California, the first study included the city of Vacaville and urban areas along the American River, Sacramento River and San Joaquin River (the cities of Folsom, Cordova, Sacramento and Stockton) while the second study included the cities of Roseville, Martinez/Pleasant Hill, Stockton and Dublin. For southern California, the second study included urban areas of Laguna Nigel, AlisoViejo, San Diego, and Lakeside (Figure 4). Sampling events took place during or shortly after rain events (Rain) and during the dry season (Dry). Sources of pesticides contamination were verified to be stormwater run-off from treated residential areas during the rainy season and landscape water run-off from treated landscaped areas during the dry season.

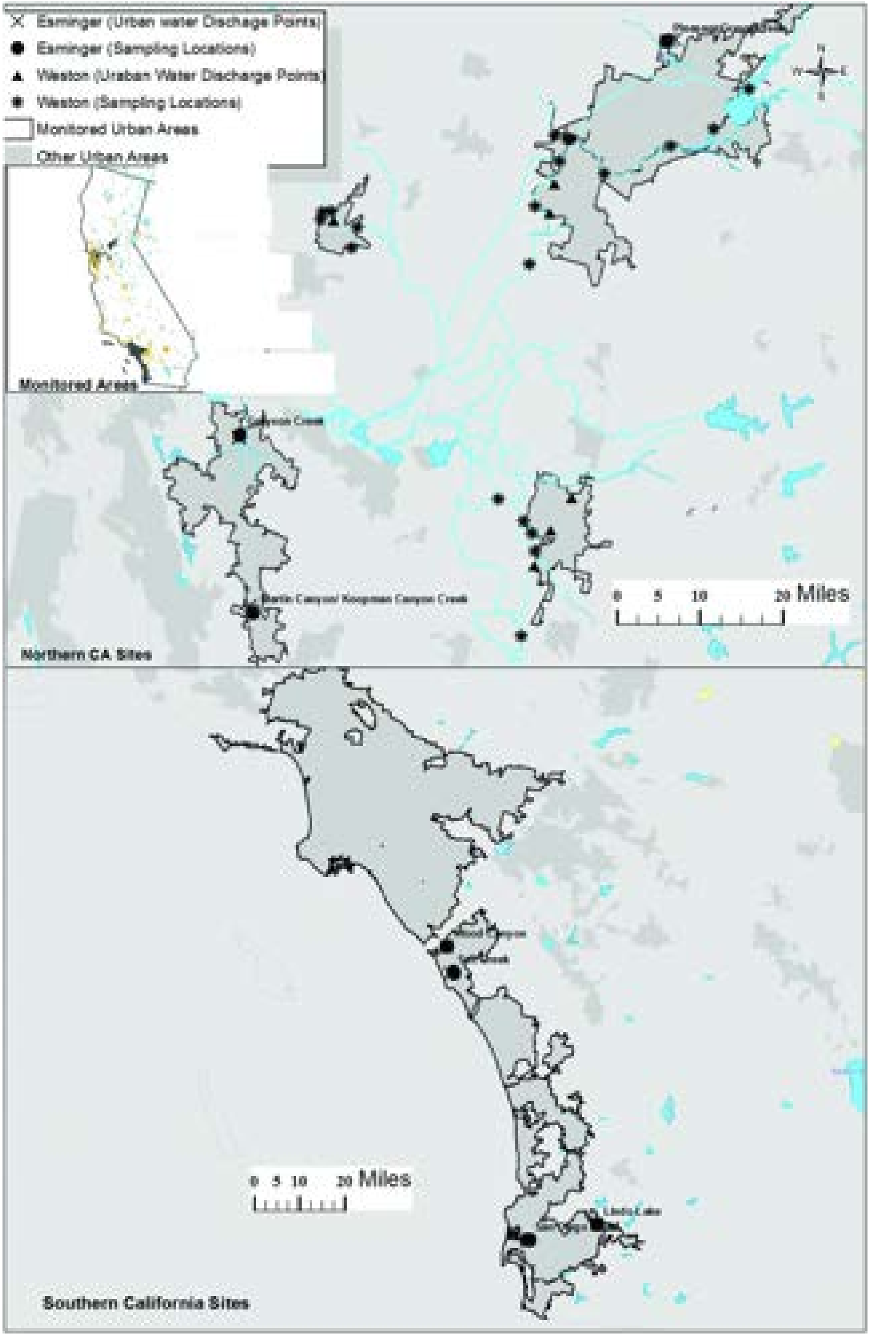

Figure 4. Monitored urban areas in Northern and Southern California (Weston and Lydy (19); and Ensminger and Kelley (20)).

In the Weston and Lydy study (*19*), concentrations detected in sump waters were high enough to be of toxic concern and were found to be related to either the pyrethroids or chlorpyrifos based on the toxicity identification evaluation (TIE) data. This was also confirmed by chemical analyses and comparison to known toxicity thresholds. Chemical analyses of 33 sump water samples show that the overall percentage of samples containing concentrations exceeding 1 ng/L ranged from 3 to 79% for eight pyrethroids and chlorpyrifos. The majority of the samples contained bifenthrin (79%) and chlorpyrifos (77%) with lesser percentages containing: permethrin (61%), cyfluthrin (55%), λ-cyhalothrin (*lambda*-cyhalothrin, 45%), cypermethrin (33%), deltamethrin (12%), esfenvalerate (6%) and fenpropathrin (3%). As expected, pesticides in sump waters that were discharged (by pumping) into receiving creeks and rivers were diluted to lower levels. Varied levels of pesticides and toxicity were found in receiving creeks and rivers as it passed through the urban areas of Sacramento (the Sacramento River), Stockton (the American River) and Vacaville (two urban creeks). Water column toxicity, related to the pyrethroid bifenthrin, was not observed in the Sacramento River but was evident along the urban creeks, the American river, and at only one site in the San Joaquin River. For example, no evidence of contamination with pyrethroids and toxicity was observed upstream in the water as the creeks enter the city of Vacaville while a high level of toxicity was observed in waters leaving the city downstream. In these water samples, pyrethroid concentrations were 4-10 times the toxicity with, bifenthrin and cyfluthrin providing most of the toxic units (TU). The level of pesticide contamination in receiving waters appeared to be related to the intensity of rain events. For example, no toxicity was observed in water samples taken from the San Joaquin River near Stockton just after the first rain event, but toxicity was evident, in one location at the edge of the city, following a more intense second rain event. Again, water toxicity was established to be related to pyrethroids as it contained 0.7 TU of bifenthrin and 0.3 TU of permethrin.

Monitoring data from the Ensminger and Kelley (*20*) study may be considered as an example of concentrations and detection frequencies (DFs) for registered and extensively used pesticides in urban areas. Therefore, data from this study are summarized herein for reported DFs and concentrations of insecticides, herbicides and pyrethroids detected in urban drain waters (DRNs) and receiving water bodies (RWBs) during dry (Dry) and rainy (Rain) seasons. Table 3 contains reported sampling information and abbreviations used in this summary and associated graphs. In the summary, statewide data reported for all locations in the study are used to obtain maximum and minimum concentrations and DFs for each pesticide. DFs are calculated for each pesticide as percent (%) from the number of samples containing that pesticide over the detection limit (number of detects) divided by the total number of samples (number of detects plus non-detects; trace detections were considered, in this summary, as non-detects). The summary includes data on pesticides that were most frequently detected in the samples of DRNs and RWBs during Dry and Rain events. The number of samples for each urban area are included in the summary table as it is an important indicator for intensity of sampling (Table 3).

Table 3. Reported Sampling Information and Abbreviations Used in the Summary of Ensminger and Kelley (*20*) Statewide Monitoring Data

				Number Of Samples (N) [1]					
				Insect[2]		*Herb*[3]		*Pyreth*[4]	
Urban Area	*City (Source Of Urban Runoff Water "Drn")*	*Receiving Water Body "Rwb"*	*Sampling Season*	*Drn*	*Rwb*	*Drn*	*Rwb*	*Drn*	*Rwb*
Sacramento **"Sac"**	Roseville	Pleasant Grove Creek	Dry	14	5	12	4	8	3
			Rain	12	4	12	4	9	3
San Francisco Bay **"Sfb"**	Martinez/Pleasant Hill; And Dublin	Grayson Creek; And Martin Canyon/Koopman Canyon Creek	Dry	20	8	20	8	10	4
			Rain	17	7	17	7	12	5
Greater Los Angeles (Orange County) **"Orn"**	Laguna Nigel; And Aliso Viejo	Salt Creek; And Wood Canyon Creek	Dry	23	9	24	10	9	3
			Rain	4	2	4	2	None	None
San Diego **"Snd"**	San Diego; And Lakeside	San Diego River; And Lindo Lake	Dry	14	10	14	10	None	None
			Rain	5	2	5	2	None	None
Statewide			Dry/Rain	8	8	8	8	5	5

[1] ***Number of Samples (n)*** **=** The total number of samples for each sampling event. For example, in the Sacramento area (SAC), insecticides were monitored in 14 drain water samples (DRN) during the dry season (Dry) and in 12 drain water samples (DRN) during the rainy season (Rain). Additionally, insecticides were also monitored in 5 receiving water samples (RWB) during the dry season (Dry) and in 4 receiving water samples (RWB) during the rainy season (Rain). [2] ***Insect***= Monitored Insecticides: carbaryl (carb), chlorpyrifos (Chl) diazinon (Diaz), fipronil (Fip), fipronil degradates (FipD= desulfinyl fipronil, desulfinyfepronil amide, fipronil amide, fipronil sulfide, and fipronil sulfone), malathion (Mal), and oxamyl (oxa). [3] ***Herb***= Monitored Herbicides: 2,4-D (2,4-D), ACET, bromacil (Brom), dicamba (Dicam), diuron (Diur), MCPA, oryzalin (Oryzal), oxyfluorfen (Oxyfl), pendimethalin (Pendi), prodiamine (Prodi), prometon (Promet), simazine (Simaz), triclopyr (triclo) and trifluralin (Trifl). [4] ***Pyreth***= Monitored Pyrethroids: bifenthrin (bif), cyfluthrin (Cyf), λ-cyhalothrin (λ-cyh), cypermethrin (cyp), fenvalerateesfenvalerate (FenEsV) and permethrin (Per= *cis* and *trans*).

In this summary, concentrations and DFs for each pesticide are examined in DRNs vs. RWBs, Dry vs. Rain, and in varied geographical locations. The objectives are to summarize reported data as DFs and concentrations for pesticides detected in urban surface waters and examine the effects of dry flow vs. rainstorm flow and geographic location on these parameters. Summaries are established for the top five insecticides, five herbicides and all of the monitored pyrethroids.

Insecticides Detection Frequencies/Concentrations

The top five insecticides that were frequently detected in source and receiving waters include carbaryl, fipronil, fipronil degradates (total of desulfinyl fipronil, desulfinyfipronil amide, fipronil amide, fipronil sulfide, and fipronil sulfone), malathion, and diazinon (Figure 5). The insecticides chlorpyrifos and oxamyl were not in the top 5 because they were both less frequently detected in DRNs (DF = 8-24% “N = 2” and 4% “N = 1”, respectively) and oxamyl was not detected in RWBs although chlorpyrifos was at a DF of 29% (N=1).

A summary is calculated from reported monitoring data for each insecticide as follows:

(1) For each geographic location (SAC, SFB, ORN and SND), the Max/Min DFs and concentrations are calculated separately for DRN waters (Dry and Rain) and RWBs (Dry and Rain) from the Dry and Rain data;

(2) A statewide Max/Min DFs and concentrations are calculated for DRN waters and RWBs separately from the combined SAC, SFB, ORN and SND values arrived at from step 1;

(3) Each set of statewide values, such as Carb-DRN or Carb-RWB, was calculated from eight data entries (N = 8 = 2 x 4; two each for SAC, SFB, ORN and SND) and in case of no detection the value of N will be <8 and no detection at all the value of N will be zero.

Data in Figure 5 show that the most frequently detected insecticides in source and receiving waters were carbaryl, fipronil and fipronil degradates (75-100%; N= 4-6). The organophosphate insecticides malathion, and especially diazinon, were detected at lower range of frequencies (24-100%; N= 2-5). Data show no apparent differences in DFs between source waters (DRN) and receiving waters (RWB) possibly due to proximity of sampling in location and timing. Except for fipronil, the maximum detected concentrations for the top 5 insecticides ranged from 0.1 to 0.8 μg/L. For fipronil, the maximum was 2.1 μg/L observed in DRN waters from Orange County. As expected, maximum chemical concentrations in drain waters were higher than those detected in receiving water bodies (2 to 5x) reflecting the effect of dilution. It is also noted that both chlorpyrifos and diazinon are still being detected despite drastic reduction in urban use resulting from EPA’s regulatory actions. As pointed out by the most recent USGS report on trends in pesticides in the US rivers and streams, concentrations of diazinon declined, by nearly two orders of magnitude, in urban streams across the country from the year 2003 to 2008 due to phasing out of its use (*21*). However, the report pointed out that use of new or alternative pesticides, such as fipronil, caused a widespread increase in fipronil concentrations in urban streams. An observed, trend in fipronil concentrations in 12 locations throughout the U.S. shows concentration increase in 10 locations with a decrease in only one location in NC and no change in one other location in TX.

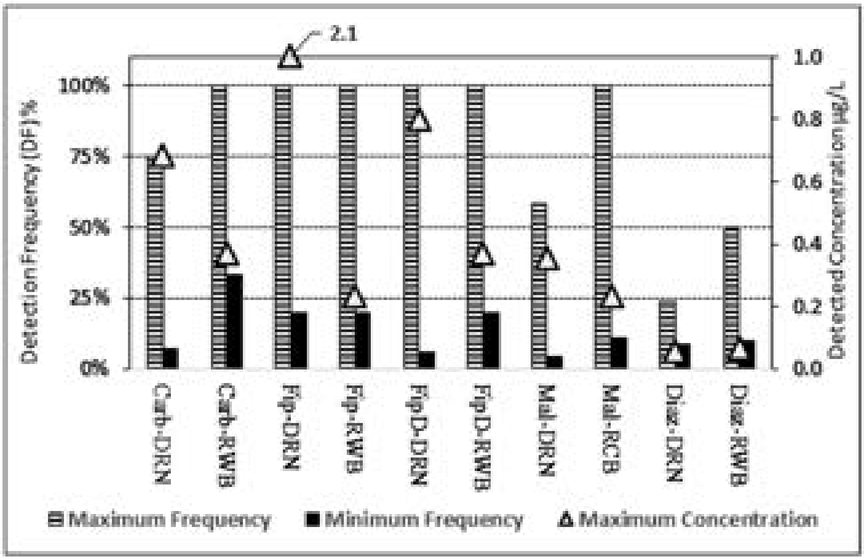

Figure 5. A summary graph for the top five insecticides frequently detected in source drain water (DRN) and receiving water bodies (RWB) in four of the major urban areas of California.

Herbicides Occurrence Frequencies/Concentrations

The top five herbicides that were frequently detected in source and receiving waters were 2,4-D, triclopyr, dicamba, diuron and MCPA (Figure 6). Other herbicides were detected at lower DFs and concentrations.

Data in Figure 6 show that the most frequently detected herbicides in receiving waters were 2,4-D, triclopyr, and diuron (75-100%; N= 6-8). Slightly lower DF were observed for dicamba and MCPA (67-100%; N=3-6). Except of dicamba, the maximum detected concentrations for the top 5 herbicides ranged from 6.7 to 27.6 µg/L. For dicamba, the maximum was 3.1 µg/L observed in DRN waters from the Sacramento area. In drain waters, the maximum concentrations of four of the top five herbicides (MCPA, dicamba, diuron and 2,4-D), in drain waters, were higher than those detected in receiving water bodies (1.1 to 51x) reflecting variable effect of dilution. In contrast, the maximum concentrations of triclopyr in DRN waters were much lower (0.2x). Results obtained for triclopyr may be explained by the possibility that receiving waters at these locations may have been contaminated with this herbicide before the point of DRN discharge. Although the DFs for herbicides are higher than insecticides, both data show no apparent differences in DFs between source waters (DRN) and receiving waters (RWB).

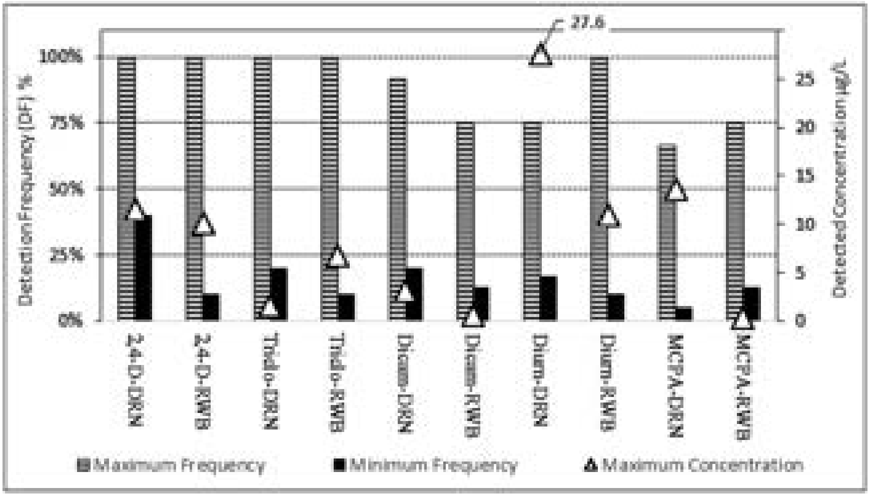

Figure 6. A summary graph for the top five insecticides frequently detected in source drain water (DRN) and receiving water bodies (RWB) in four of the major urban areas of California.

Pyrethroids Occurrence Frequencies/Concentrations

Pyrethroid insecticides that were frequently detected in source and receiving waters included bifenthrin, cyfluthrin, λ-cyhalothrin, cypermethrin, fenvalerate/ esfenvalerate and permethrin (Figure 7). The pyrethroid bifenthrin was detected in all source and receiving water samples with DFs ranging from 56-100% (N= 3-4) followed by permethrin with a range of 20-33% (N= 1-3). DFs for the other pyrethroids were much lower than bifenthrin and permethrin as they were in the range of 0-22% (N= 0-1).

Detected concentrations of pyrethroids in source and receiving waters ranged from 0-203 ng/L. In all of the monitoring events, higher pyrethroid concentrations were observed in source waters (DRNs) as compared to receiving water bodies (RWBs). Source water concentrations were 1.6-7.3 times higher than receiving waters in 4 out of 6 monitoring events and no pyrethroid was detected in the receiving waters of two out of the six events. This is probably a result of partitioning of the pyrethroids to the organic carbon in suspended/underlying sediments of receiving water bodies.

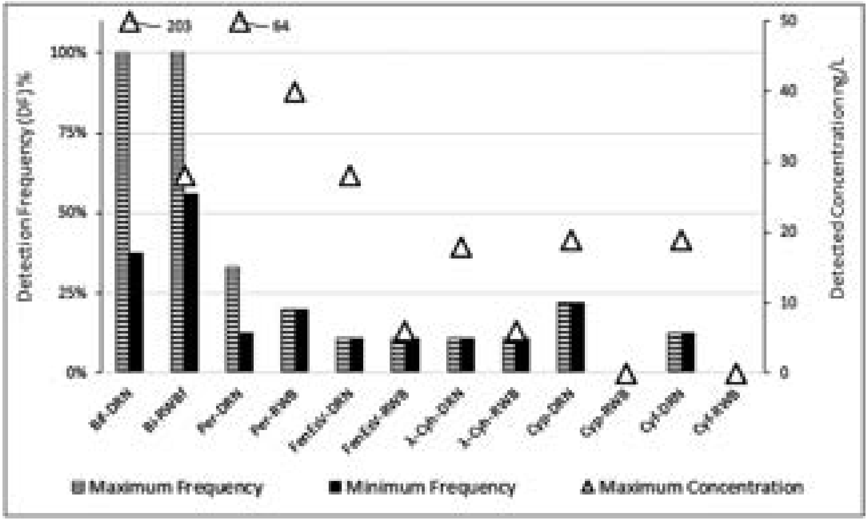

Figure 7. A summary graph for pyrethroid insecticides frequently detected in source drain water (DRN) and receiving water bodies (RWB) in three of the major urban areas in California.

Variations Associated with Geographical Locations

Variations in both concentrations and DFs are summarized in two ways based on the reported monitoring data for the four major urban areas of California. The first is by comparing maximum DFs of the insecticide in all monitoring events (DRN/Dry, DRN/Rain, RCB/Dry and RWB/Rain; referred to as the DFs comparison). The second is by comparing maximum/minimum DFs and maximum concentrations detected in the major source of contamination; that is the storm water drains in the two monitoring events (DRN/Dry and DRN/Rain; referred to as the DRN DF/Concentration comparison). The two types of comparisons are conducted herein for insecticides, herbicides and pyrethroids.

For insecticides, Figure 8 shows differences in the maximum DFs of monitored insecticides between various urban locations in the state of California.

The DFs comparison show that all of the top five insecticides were detected, in varied maximum DFs, in three of the major urban areas of California (SFB, ORN and SND). Diazinon was the only insecticide that was not detected in SAC area. It is also apparent that urban areas of southern California (ORN and SND) show higher maximum DFs, for these five insecticides, compared to the northern urban areas of the State (SAC and SFB). Observed differences could be a reflection of expected higher insecticides usage in the hot climate of the south as compared to the northern part of the State.

Figure 9 shows differences in DFs and concentrations detected in storm waters reflecting the contribution of this important source of insecticides reaching surface waters.

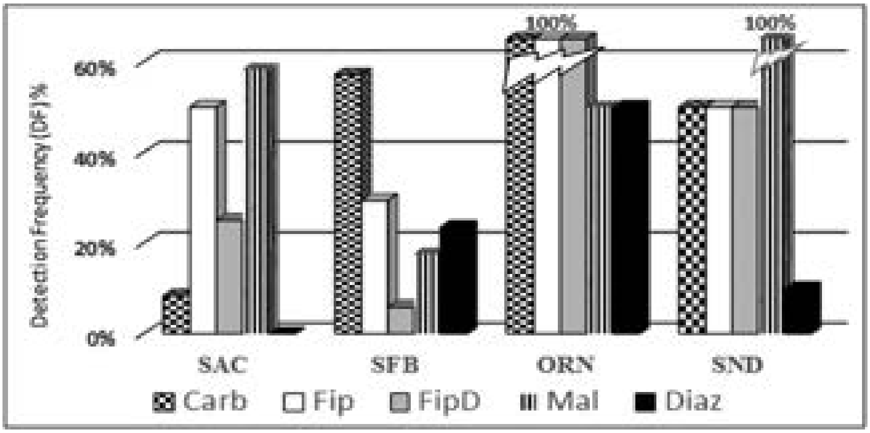

Figure 8. Maximum detection frequencies (DFs) for the top five insecticides detected in source/receiving waters of four of the major urban areas of California (SAC= Sacramento, SFB= San Francisco Bay, ORN= Orange County and SND= San Diego).

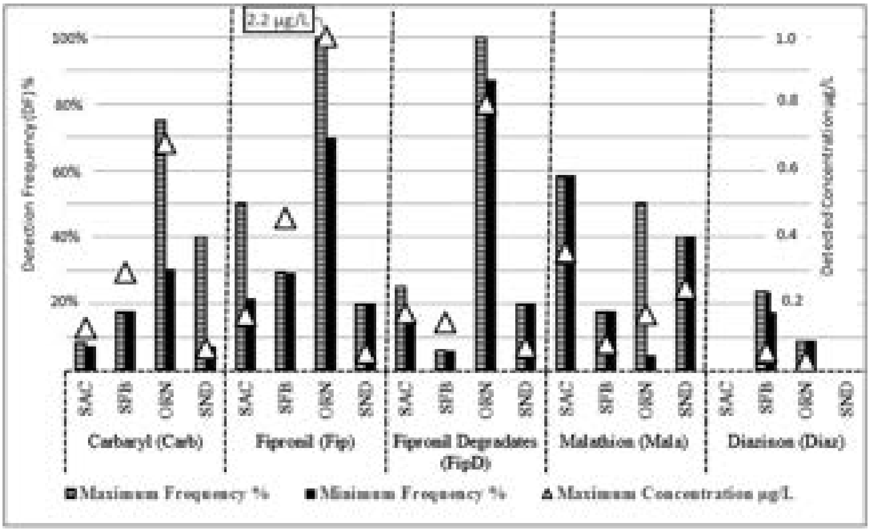

Figure 9. Max/min Detection Frequencies (DFs) and maximum concentrations of major insecticides detected in source drain waters (DRNs) in Sacramento (SAC), San Francisco Bay (SFB), Orange County (ORN) and San Diego (SND) urban areas of California.

The DRN DF/concentration comparison show variations in the insecticide load of the storm water. ORN County appears to have the highest detections for three out of the four insecticides (Carb, Fip, and FipD) and the second highest for the other two insecticides (Mal and Diaz). Data also show that maximum DFs appear to be associated with higher concentrations detected in the storm water in all of the four urban areas.

For herbicides, similar analyses was conducted (not shown) and results show that all of the top five herbicides were detected, in varied maximum DFs, in all of the major urban areas of California. Herbicides were detected at higher DFs than 40%, except for MCPA which was detected at a DF of 13% in ORN, 20% in SND. ORN showed the highest DFs of three herbicides (2,4-D, Triclo, and Diurn) followed by SAC with the lowest being the SND area. The herbicide 2,4-D was the most frequently detected in all of the four area followed by triclopyr in SFB and ORN. MCPA had the least DFs ranging from 13 to 75% with the least DFs in ORN followed by SND, SFB and SAC (highest).

DRN DF/concentration comparison show that ORN county with the highest detections for three out of the four herbicides (2,4,-D, Triclo and Diur) and the 3rd and 4th highest for the other herbicides (Dicam and MCPA). Data also show that maximum DFs do not always coincide with higher concentrations detected in the storm water. For example, SAC had the lowest DF of diuron compared to the other three areas of California but had the second highest observed concentrations and SFB had the 3rd DFs associated with the highest concentrations. Additionally, data on the maximum concentrations observed in source and receiving waters are summarized in Figure 10.

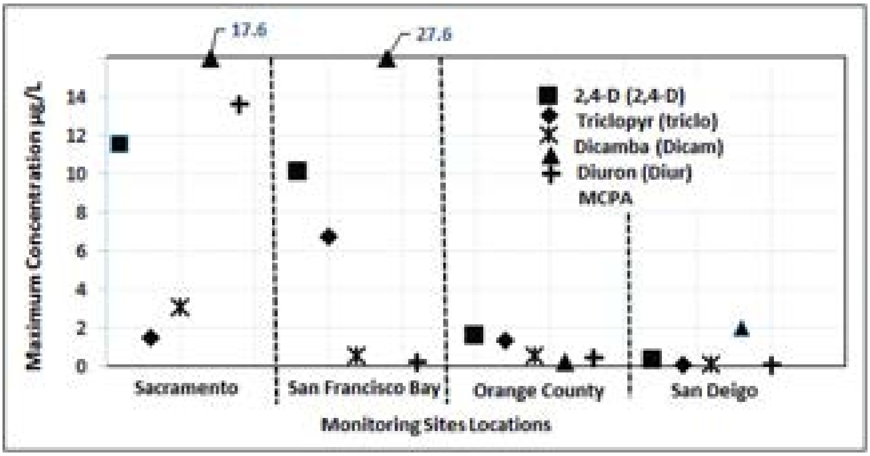

Figure 10. Maximum concentrations of herbicides detected in urban monitoring data from Northern and Southern California.

The summary shows that higher concentrations of the top 5 herbicides (>6.8 to 27.6 µg/L) were observed in source and receiving waters of Northern California urban areas (SAC and SFB) compared to ORN and SND of southern California (<2 µg/L). In Northern California, observed maximum diuron concentrations were the highest (17.6 to 27.6 µg/L) followed by 2, 4-D with maximum concentrations ranging from 10.1 to 11.5 µg/L. The MCPA maximum concentration was highest in SAC area (13.6 µg/L) while triclopyr was highest in SFB area (6.75 µg/L).

Similar analyses was performed on the pyrethroids data which includes only three urban areas SAC, SFB and ORN; SND was not monitored. Results of the DRN DF/concentration comparison show that bifenthrin was detected in DRN waters in the three monitored areas with maximum DFs/concentrations of 56%/26 ng/L), 93%/33 ng/L) and 100%/203 ng/L). The other four pyrethroids were only detected in SAC (cyfluthrin with DF/concentration of 13%/18.9 ng/L and cypermethrin with DF/concentration of 22%/18.9 ng/L), ORN (λ-cyhalothrin with DF/concentration of 11%/18.0 ng/L and fenvalerate/esfenvalerate with DF/concentration of 11%/28.0 ng/L). Additionally, data on the maximum DFs/concentrations observed in source and receiving water are summarized in Figure 11.

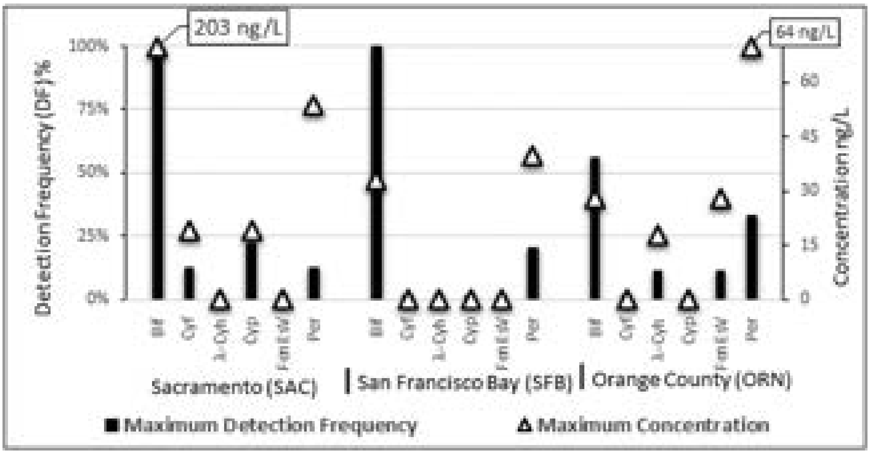

Figure 11. Observed maximum detection frequencies (DFs)/concentrations for pyrethroids detected in source/receiving waters of three major urban areas of California (San Diego (SND) not monitored).

Summary data indicate that the maximum DFs for bifenthrin were 100% in SAC/SFB areas and 56% in ORN County area. In the SAC area, the maximum observed concentrations of bifenthrin, permethrin, cypermethrin and cyfluthrin were 203, 53.9, 18.9 and 18.9 ng/L, respectively. In SFB area, only permethrin and bifenthrin were detected at maximum concentrations of 40 and 33 ng/L. Finally, in ORN County the maximum observed concentrations were 64 ng/L for permethrin, 28 ng/L for bifenthrin, 28 ng/L for fenvalerate/esfenvalerate and 18 ng/L for λ-cyhalothrin.

Variations Associated with Wet/Dry Conditions

Urban pesticides are mainly transported from application sites into surface waters by urban runoff waters resulting from rain storms and/or irrigation. It is thus expected that DFs and concentrations in drain and receiving waters to be related to pesticide properties (persistence and solubility), water availability (rain and irrigation), and timing of application. Additionally, application rate and frequency of application are expected to play a role in determining expected pesticide concentrations in surface waters as these factors are important in determining the pesticide load in quantity and timing. The latter factors can be deduced from usage data.

The results of the monitoring study indicated that most pesticides were detected during wet than dry conditions. One exception was fipronil and its degradates which were detected at higher frequency during dry flow in ORN County. Other reported results included the following: (1) First rainstorm gave the highest DFs in all of monitored site except in ORN county; (2) Detection of fipronil and its degradates with the first storm was similar to dry flow conditions and correlated with usage in Northern California; (3) Pesticides used in urban areas may show continuous load, similar to fipronil, independent of rain; (4) Bifenthrin had high detections associated with rain events although it is mostly applied during the dry season; and (5) Herbicides had more frequency of detections during the rainy season which coincides with timing of its application. Furthermore, the authors used the difference between DFs during wet flow and DFs during dry flow as an indicator for the influence of rain on pesticide detections. The results indicate that most of the pesticides are influenced by rain giving higher detection with the exception of fipronil degradates. Rain appeared to cause the highest detections for bifenthrin followed by diuron, MCPA, 2,4-D, malathion, dicamba, triclopyr, pendimethalin, carbaryl and fipronil (lowest).

The influence of dry and rain conditions on DFs and concentrations was examined based on monitoring data from stormwater ouflows (DRN) using bubble graphs and an example of these graphs is shown in Figure 12 for the top five frequently detected insecticides. DRN data were used because it reflect pesticide load carried out by run-off. In general, Figure 12 shows, that larger number/size and higher positions are for detections following rain compared to small number/size and lower positions for detections associated with dry flow. This is true for almost all of the examined insecticides, except of carbaryl, fipronil and fipronil degradates observed in ORN County.

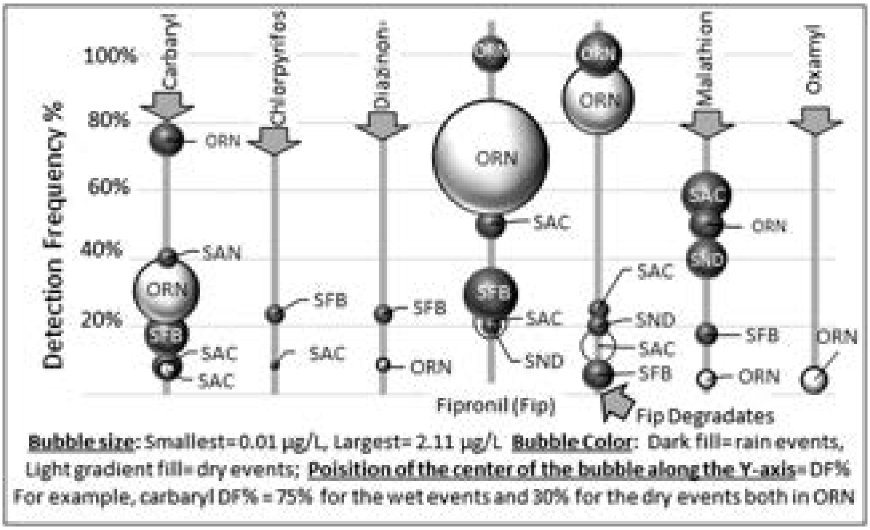

Figure 12. Influence of dry and rain flow conditions on DFs and concentrations of insecticides in the urban areas of Sacramento (SAC), San Francisco Bay (SFB), Orange County (ORN) and San Diego (SND).

Sediment Monitoring

Stream bed sediment samples were collected, during dry flow conditions, in creeks, a river, and a lake receiving waters from identified storm drains of five urban areas in Northern and Southern California (CA-N and CA-S) (*20*). The CA-N site was from Grayson creek receiving stormwater from the mixed residential/commercial urban area of Martinez/Pleasant Hill in the San Francisco Bay area. The CA-S sites were from Salt Creek, Wood Canyon Creek, San Diego River, and Lindo Lake receiving storm waters from the mostly residential or mixed residential/commercial urban areas of Laguna Nigel (Orange Co.), Aliso Viejo (Orange Co.), San Diego and Lakeside cities, respectively (Figure 13). In this California study, sediment samples were analyzed for 9 pyrethroids and chlorpyrifos and only 8 pyrethroids were identified. The pyrethroid fenpropathrin and the insecticide chlorpyrifos were not detected.

In another study, occurances and potential sources of pyrethroids in stream bed sediments from seven U.S. metropolitan areas were assessed. Sediment samples were collected in 2007 from 98 urban streams within the metropolitan areas of Atlanta, GA (ATL); Boston, MA, NH (BOS); Dallas−Fort Worth, TX (DAL); Denver, CO (CO); Milwaukee−Green Bay, WI (MGB); Seattle−Tacoma, WA (SEA); and Salt Lake City, UT (SLC) (*22*) (Figure 13). In this national scale study, sediment samples were analyzed for 14 pyrethroids and reported data were for five pyrethroids.

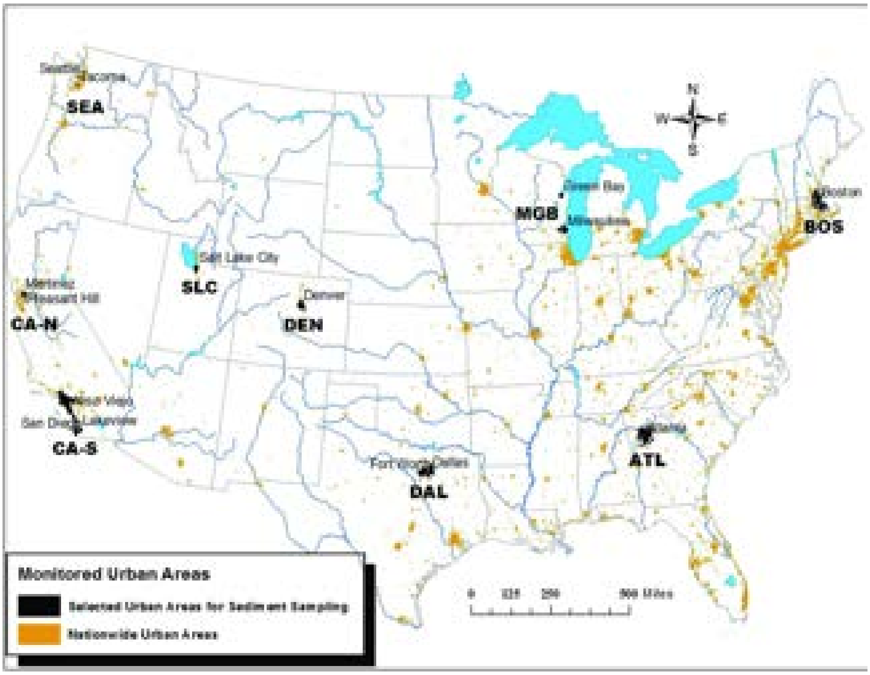

Figure 13. Stream bed sediment sampling sites for the statewide California study by Ensminger and Kelley (20) and the nationwide study by Kuivila et al. (22).

The pyrethroids bifenthrin, λ-cyhalothrin, cypermethrin, permethrin and resmethrin were monitored in both studies. Data from Ensminger and Kelley (*20*) included monitoring data for sediments underlying storm drains in addition to receiving water bodies. The data show relatively high DFs for bifenthrin, cyfluthrin, permethrin deltamethrin, λ-cyhalothrin and cypermethrin (41-97%) with maximum concentrations ranging from 32 to 680 μg/kg dry sediment. Fenvalerate/esfenvalerate maximum DF/concentration was reported to be relatively lower (14% and 24 μg/kg). However, of interest in this section is the pyrethroid chemicals data for sediments underlying receiving water bodies as it can be compared with data obtained for the nationwide bed stream sediments study conducted by Kuivila, *et al.* (*22*) This will permit comparison between bed sediments obtained nationwide from urban areas varied in hydrology, weather, pesticide use, timing of application and land characteristics/use. Figure 14 summarizes the concentration and DF data obtained from both studies for bifenthrin, λ-cyhalothrin, cypermethrin, permethrin.

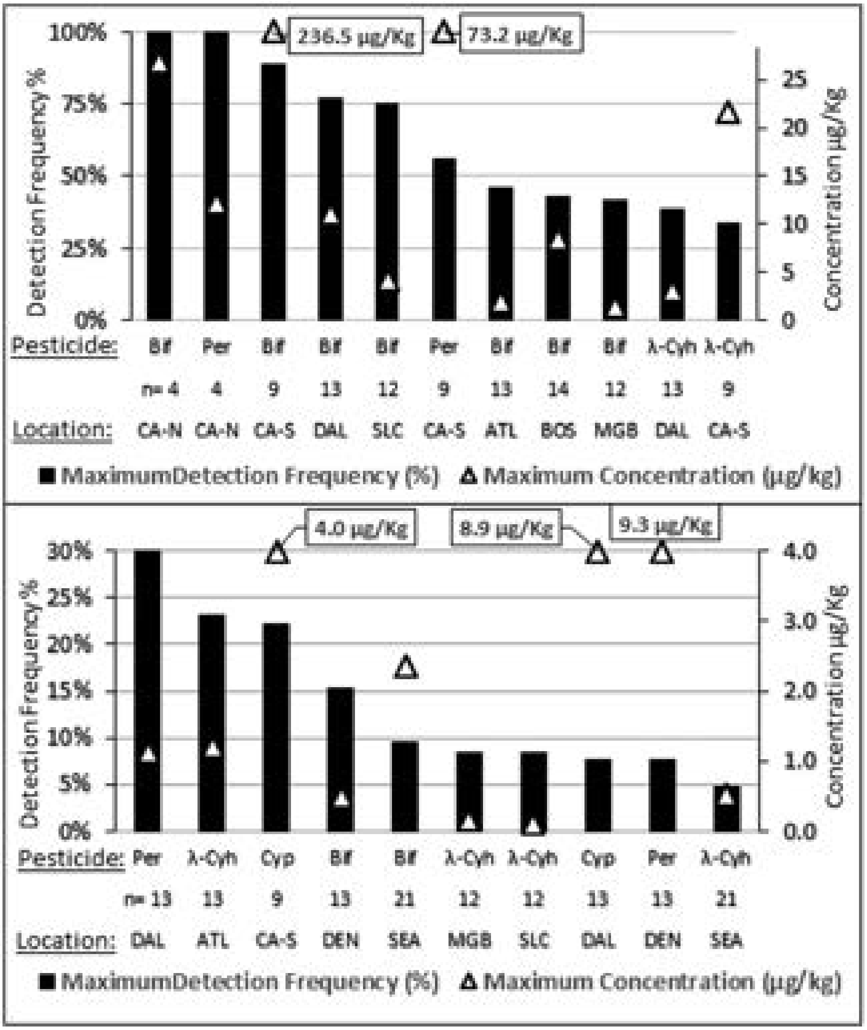

Figure 14. A summary of the sediment concentration/DF data obtained for bifenthrin (bif), λ-cyhalothrin (λ-Cyh), cypermethrin (Cyp) and permethrin (Per) (n = number of samples; for sampling location abbreviation refer to map in Figure 13).

Data show variable occurrence frequencies and concentrations of pyrethroids detected in bed sediment streams across the country. Reported data may be categorized by the frequencies of occurrence into three categories as shown in Table 4.

Table 4. Categories for the Frequencies of Pyrethroid Occurances in Bed Sediments

Detection Frequencies	***Maximum Concentrations***	***Bed Sediment Location***	***Pyrethroid Detected***	***Exception***	***Reference***
56 - 100%	**11.2 – 237** µg/kg	**CA-S, CA-N, DAL**	Bif	SLC= **4.2** µg/kg	Figure 14 Top Graph
		CA-N	Per	None	
33 - 46%	**1.4 - 8.4** µg/kg	**BOS, ATL, MGB**	Bif		
		DAL	λ-Cyh	CA-S= **22** µg/kg	
5 - 31%	**2.4 - 9.2** µg/kg	**DEN**	Per	None	Figure 14 Bottom Graph
		DAL, CA-S	Cyp		
		SEA	Bif		
	0.1 - 1.2 µg/kg	**ATL, SEA, MGB, SLC**	λ-Cyh		
		DEN	Bif		

Monitoring vs. Modeling

Targeted monitoring data, similar to those discussed earlier, are important resource for regulators of urban pesticides. These type of data are available for pesticides that have been in use for many years. Quality monitoring data can be used as a ground truth for verifying modeled estimated environmental concentrations (EECs) that determine aquatic exposure. EECs are used in ecological and drinking water assessments. In the case of pesticides used in urban setting, monitoring data are much more important due to the usually high uncertainties associated with modeling surface water exposure in the urban environment. In some cases, it was necessary to use EECs from monitoring data instead of modeling due to lack of scenarios that would represent application of a given pesticide and associated processes. For example, application rate/acre, number of applications and timing are required to perform PRZM/EXAMS modeling. Label information is not enough and assumptions had to be made to estimate these key parameters. For example, in the case of house perimeter treatment the label usually gives application rate in lbs a.i/1000 sq. ft and possibly a recommended treatment of 2 ft around the house. A residential area factor is usually estimated in order to arrive at a reasonable application rate for modeling which needs estimates of housing density/acre and area that would be treated (need to assume house dimensions). The assumptions should be reasonably conservative and represent the area where the pesticide is to be applied. The task of arriving at reasonable estimates becomes much more difficult when the pesticide is to be used on a national scale. Many scenarios would be needed to represent housing densities across the U.S. Additionally, other needed parameters, such as timing of application, is assumed conservatively to happen at one time for all houses within a 10-hectare area. PRZM calculates daily load of pesticide transported by water run-off and erosion into 20,000 m^3 pond 2 m deep with no outlet to further simulate degradation. In contrast, urban runoff waters transports pesticides, through urban drainage/pumping systems (in some cases through POTWs), into surface water bodies such as urban creeks, lakes, and rivers. Pesticides arriving to these water bodies may then be transported via running water rather being held into a pond with no outlet. Although EECs estimated in pond, by EXAMS, are expected to be conservative but much higher conservatism may result in unrealistic estimates especially with highly persistent pesticides that accumulates in the pond yielding high EECs.

Monitoring data and examples of previous modeling for surface water EECs are compared for two of the most frequently detected pesticides; Fipronil and bifenthrin (Table 5).

Table 5. Modeled vs. Monitored EECs for CA

Chemical	***Treatment Type***	***Label Rate*** *(lbs ai/A)*	***Modeled rate (No. of Applications)*[1]**	***Modeled EECs (ng/L)***			***Observed EECs in CA Monitoring (ng/L)*[2]**
				Peak	***21-day***	***60-day***	
Fipronil	House perimeter treatment	2 ft. @ 0.357	0.012 lbs ai/A (1)	41.2	26.7	15.2	Maximum= 232 and 90th%= 83
	Broadcast fire ant treatment	Not Stated	0.014 lbs ai/A (1)	6.3	4.1	2.3	
Fipronil total degradates	House perimeter treatment	Three degradates modeled individually similar to parent using the fate properties of the degradates and the max daily conversion observed in environmental fate studies corrected for differences in molecular weights		19.5	15.5	14.1	Maximum= 372 and 90th%= 125
	Broadcast fire ant treatment			3.2	2.6	2.4	
Bifenthrin	Many residential uses	Varied label rates with calculate range of application rate of 0.001 to 2.2 lbs ai/A and from one application to twelve applications @ 7-180 days intervals		Capped by the solubility limit of 14 ng/L			Maximum= 27.2 and 90th%= 24.2

[1] Modeled rate = Could be different from label rate because it is an adjusted rate based on treated area of the acre. [2] Maximum and 90th percentile values detected in Receiving water bodies (RWB).

Monitored maximum and 90th percentile EECs for both fipronil and bifenthrin are higher than modeled EECs suggesting possible improper parameterization of the model. In the case of fipronil, lower EECs might be related to the application rate calculated for modeling and possibly a reflection of the scenario used. In case of bifenthrin, modeled EECs were capped to the limit of solubility of the chemical which is 14 ng/L. Modeled EECs are higher than the 14 ng/L concentration expected for this insoluble chemical. In fact, bifenthrin was detected to occur at concentration as high as 27.2 ng/L which is almost two times its laboratory determined solubility. The observed relatively high occurrence for bifenthrin in run-off and surface waters may be attributed to factors such as water chemistry, such as presence of dissolved organic carbon or colloids, and possible effects of the formulation that makes bifenthrin more soluble in surface waters than in pure laboratory water.

The above example of comparison between monitored and modeled data is at best an approximation due to many factors such as (1) Modeled EECs were not a result of proper parameterization of the model to represent monitored areas, (2) The summary concentrated on the maximum observed concentrations in order to identify sites having the highest EECs indicating their vulnerability noting that these values may have been influenced by contamination from other sources such as spills and transported pesticides from areas upstream or with airborne particles and/or drift (3) Ideally, only targeted monitoring data, for an identified vulnerable site, may be compared to modeling data using parameters representing the same site. This is not the case for the comparison above, monitoring data were for four different urban areas of the State of California, consisted of 47 values for fipronil and degradates and only 14 values for bifenthrin and the maximum number of values for each site ranged from 1 to 8 (dry + wet events) and only from 1 to 5 for dry event and 1 to 4 for wet events, Monitoring data needed for comparison should represent only one area and should be extensive (daily or weekly). The monitoring data used were with intervals ranging from 24 to 167 days 7 to 118 days. for bifenthrin, (4) It is important to point out that the ultimate maximum exposure EECs in receiving water bodies is dependent on the mass of pesticide transferred into the water body. Winter stormwater high flow with low concentration is expected to contribute more pesticide mass to receiving waters than summer low flow with high concentration, (5) In flowing waters, such as rivers and streams, observed concentrations are expected to be influenced by the flow status of the rivers and streams because higher dilution will occur at high flow compared to low flow, and (3) EECs are also influenced by the pesticide fate and transport properties as well as the receiving water characteristic such as type of suspended matter (content of dissolved/suspended organic carbon and other colloidal materials). Such contents may additionally influence the bioavailability of the pesticide and its toxic effects.

Potential Refinement for Modelling EECs

Currently, the USEPA considers the modeling approaches described above and resultant EECs as crude estimates and provide only a screening-level information. This is due to uncertainties regarding: variability in site characteristics that govern runoff, effect of different formulations, types of impervious surfaces, application methods and timing and national representation of regions with varied landscape, housing density and hydrological features. Therefore, refinement of residential and impervious surface exposure scenarios is needed. This can be done by incorporating recent findings that could be used to accurately parameterize the residential and impervious scenarios used in modeling. This information helps in refinement by providing data necessary to establish national representative scenarios for vulnerable sites. The conceptual model for establishing these standard scenarios involves the following steps: (1) identify vulnerable urban watersheds based on available monitoring data and different climatic conditions, pesticide pressure, and hydrology; (2) understand the hydrology of the chosen watersheds especially the drainage system inputs and outputs; (3) classify each of the chosen watersheds according to land use (commercial, industrial, mixed and others), determine fractions of pervious, impervious surfaces and drainage systems for urban runoff waters; (4) choose 10-hectare vulnerable areas of the watershed that represent typical residential, commercial, industrial and mixed developments (that is the catchment area for PRZM); (5) specify the types of surface areas, in the chosen catchments, that would be treated for various label use patterns (i.e., home perimeters patios, driveways, etc.) and the fraction of that area that would be treated (i.e., fraction treated for home perimeters patios, driveways, etc.); (6) determine the application rates for the residential and impervious surface exposure scenarios; (7) adjust the rate for varied impervious surfaces based on available washoff studies (this adjustment would be dependent on the modeled chemicals); (8) establish a pattern for timing of application within the chosen watershed; and (9) run PRZM simulations with outputs processed through mixing cells into varied receiving water bodies (urban streams, lakes and rivers) to arrive at exposure EEC averages needed for risk assessment.

Assessing Adulticide Uses

Mosquito control remains as an important issue in urban environments in the United States due to the need to limit mosquito-borne diseases, such as West Nile virus (affecting human health) (*23*), or dog heartworm (affecting pets) (*24*). Wide area adult mosquito control is accomplished through a different pesticide method of application. In lieu of conventional ground or aerial applications using fine, medium or course droplets (according to American Society of Agricultural and Biological Engineers (ASABE) Standard S-572.1), mosquito adulticides are applied as ground or aerial mists, using extremely fine droplets, known as Ultra-Low Volume (ULV) droplets. Adulticide application rates are usually a very small fraction of the rate of coarser droplet applications used to control other insects (*e.g.*, in the ounces of product per acre range). Conventional pesticide applications are typically intended to hit the crop (*i.e.*, for foliar applications), while the ULV droplets are intended to remain airborne to hit mosquitoes in flight. A critical review of ULV technology, including efficacy, variables that affect space ULV applications, and some information on non-target impact, has been published (*25*).

Examples of pesticides applied through ULV spray products are permethrin, prallethrin, *d*-phenothrin (commonly known as Sumithrin®), pyrethrins, etofenprox, malathion and naled. These chemicals are often times co-formulated with the synergist piperonyl butoxide (PBO) to enhance their activity. Given that many of these pesticides are considered very toxic to aquatic organisms, an approach to calculate aquatic estimated environmental exposure concentrations (EECs) is required. In this section of the chapter, a brief description of how the USEPA assesses ecological exposure from adulticides is presented, using modeling and open literature data for aerial and ground applications, respectively. The section will provide a synopsis of the use information and modeling, which includes discussions of aquatic and terrestrial exposure. For aerial applications, the AGricultural DISPpersal drift model (AGDISP v.8.26) is used for this purpose. For ground applications, a review of literature information and other lines-of-evidence provided an upper bound deposition level. A short example of an adulticide ecological risk assessment's results will be provided, and compared against monitoring data.

Use of Adulticides

In 2005, a Pesticide Registration (PR) Notice, titled "Labeling Statements on Products Used for Adult Mosquito Control", was issued (PR Notice 2005-1 (*26*)). The PR Notice 2005-1 (*26*) provided recommendations for label language for pesticides products for wide area ground or aerial adult mosquito control products, applied only through ULV spray or fog.

The PR Notice 2005-1 (*26*) included seven major recommendations (*27*). Among the recommendations, adult mosquito control applications should be limited to trained personnel and users should consult their State and Tribal agency to determine if permits or regulatory requirements exist. Additionally, adulticide applications should be clearly distinguished from conventional applications of insecticides in the label directions. The "Environmental Hazards" section of the labels should be clear and direct applications over bodies of water should be allowed under certain circumstances. Bee precautionary language should allow adulticide applications in order to respond to threats to public health that might be identified. As of October 15, 2005, registrants were expected to submit label amendments reflecting recommended label language; however, some labels were changed after this date. This language also provided more consistent instructions across different products relative to the quality of spray droplet, application rate, seasonal or annual rate, *etc.* Adulticide application parameters are also highly dependent on actual weather conditions, such as wind speed and direction. PR Notice 2005-1 (*26*) addresses such issues as well. The labels for mosquito adulticides now include restrictions surrounding the size of the droplets from the applications. According to the recommendation, two droplet dimensions should be specified: one is the Dv0.5 (the volume median diameter: half of the volume of spray contains droplets which are smaller than the stated value), and the other is the Dv0.9 (90% of the spray is contained in droplets smaller than this value), both expressed in microns (*e.g.*, Dv0.5 <60 μm and Dv0.9 <115 μm, for aerial applications). Furthermore, labels now indicate the frequency and timing of applications, and the maximum annual application rate. This information is very useful and allows the assessor to determine which conditions should be assessed for risk of aquatic (and terrestrial) exposure. Moreover, the altitude of aerial applications is oftentimes also specified (*e.g.*, ≥75 ft).

Modeling Approach for Adulticide Assessment

As indicated earlier, the modeling approach for the aerial adulticide use includes calculations of spray drift using the exposure model AGDISP. This computer program estimates the deposition of the pesticide to the treated area, which is the application efficiency. Further, by means of its toolbox “Deposition Assessment,” the deposition to adjacent bodies of water (*i.e.* the standard pond), the value of spray drift can be obtained. AGDISP provides a prediction of spray drift under circumstances where a mosquito adulticide is used. Besides the Dv0.5, Dv0.9, and boom height, other parameters of importance in modeling in AGDISP include the spray volume (usually expressed in gallons per acre), wind speed range (miles per hour), wind direction, spray material (*e.g.*, oil or water based), and specific gravity. The spray volume, material, and specific gravity, are specified or can be estimated from the label or the material safety data sheet (MSDS) for the product, or from product chemistry submissions. To model aerial applications, the lowest boom height allowed in the label is selected, which is expected to result in the highest deposition and drift. The model output of AGDISP includes the spray drift fraction (obtained from the “Deposition Assessment” tool of the model’s Toolbox), and application efficiency (fraction of the material that deposits in the target area under the aircraft, which is expected to be much lower than the default values for typical agricultural applications). In order to obtain aquatic EECs, these values are utilized as input parameters in the aquatic models PRZM/EXAMS. To obtain terrestrial EECs, the application efficiency can be used to correct the application rate in T-REX (Terrestrial Residue Exposure, v.1.5.2). The “adjusted application rate”, based on application efficiency estimated by AGDISP, is the rate that is entered into T-REX for estimating exposure and risk to non-target terrestrial animals. The model can also be used to estimate exposure to wildlife off the field, by means of the terrestrial point estimate of the “Deposition Assessment” tool.

The AGDISP model has been used for aerial applications; however, it has not been approved for wide use in EFED for ground applications. Recently, in response to a request to amend certain labels, and a petition by the Health Effects Division (HED), EFED evaluated aerial ULV applications using the model AGDISP (*28*). The model chemical was etofenprox. Given the same application parameters (*i.e.*, drop size distribution, application material, application height), model results indicated that the deposition value is sensitive to wind speed as an input parameter. For etofenprox the wind speed range allowed by the label is 1-10 mph. Based on AGDISP modeling of aerial applications, at wind speeds of 1 mph the application efficiency (percent of the chemical that deposits on the crop) was estimated to be ~33%. Additionally, the application efficiency decreases with increasing wind speeds.

For ground applications, eight open literature studies (Table 6) and a dissertation focused on the mechanistic aspects of drift from ground-based adulticide applications (Schleier III (*29*), Schleier III *et al.* (*30*)). EFED evaluated these articles and detected that peak deposition rates, measured in a variety of dosimeters, and at different wind speeds and distances from the application sites were similar to aerially based ULV applications. Consensus of the studies indicated that ground ULV pesticide deposition is similar to that from aerial ULV pesticides. For ground applications, the deposition is expected to range from 0 to 33% of the applied (Table 6).

Table 6. Summary of Peak Deposition Rates Reported in Literature Studies [1]

Reference	***Material***	***Peak deposition (ng/cm^2)***	***Peak deposition (% applied)***[2]	***Distance from application source to peak deposition (m)***	***Wind speed (mph)***[2]
Tucker *et al.* (*31*)	Fenthion	2.92	2	8	NR
	Malathion	85.8	15	8	NR
	Naled	57.3	20	8	NR
Moore *et al.* (*32*)	Malathion	84.1	14	30.4	0.9–3.4
Tietze *et al.* (*33*)	Malathion	50	9	5	2.1–4.0
Knepper *et al.* (*34*)	Malathion	9,222	NA	7.6	1
	Permethrin	14,389	NA	7.6	1
Tietze *et al.* (*35*)	Malathion	473	NA	Unknown	0–2.5
Schleier & Peterson (*36*)	Naled	74	33	50	1.5
	Permethrin	4.6	5.9	25	4.3
Pierce *et al.* (*37*)	Permethrin	5.1	10	Unknown	6−12
Preftakes *et al.* (*38*)	Permethrin	8	10	25−50 m	4.8

[1] Source: USEPA (*28*). [2] NA – insufficient information to assess; NR – not reported.

The review concluded, based on EFED's analysis and guidance provided in the label, that a deposition rate of 33% for sprays reaching agricultural crops is a conservative estimate for both ground (based on submitted literature data) and aerial (based on AGDISP modeling) ULV applications for etofenprox.

Adulticide Insecticides Monitoring Data

Monitoring data for adulticides is scarce. Given that they are applied at extremely low rates, and some of these pesticides have other uses, monitoring results can be confounded with other uses. For example, permethrin can be used as an adulticide; however, it can be used on agricultural crops, in residential settings, and industrial sites, and it has uses that may lead to residues in wastewater discharges, and consequently in treatment plant effluents.

In 2000, Milam *et al.* (*39*) published a report of monitoring for toxicity of ground and aerial permethrin adulticidal applications (product Biomist®) in Arkansas. Toxicity was performed *in situ* in 10 replicate test chambers plus controls. Test organisms included *Daphnia pulex*, *Ceriodaphnia dubia*, and *Pimephales promelas*. Five test organisms were placed in each chamber. Once permethrin was allowed to settle, the chambers were transferred to the laboratory for the remainder of the exposure period (24 or 48 hours). *P. promelas* did not appear to be susceptible to aerial or ground ULV permethrin applications, showing 100% survival in all instances. Both *D. pulex* and *C. dubia* appeared to be more susceptible to aerial than to ground applications and showed variable survival rates from ground applications of permethrin.

Weston *et al.* (*40*) reported their results from monitoring of aerial applications of pyrethrins and PBO in August 2005, using the product Evergreen Crop Protection EC 60-6 (containing 6% pyrethrins and 60% PBO), on ~50,000 hectares over the densely populated area of Sacramento, CA. (across the American River). Treated areas were primarily commercial and residential. Water and sediments from six creeks draining the treatment area were sampled and tested for toxicity (water *C. dubia* test (~6-8-day tests); sediment *Hyalella azteca* (10-day test) and chemistry (pyrethrins, chlorpyrifos, diazinon and PBO in water; pyrethroids, pyrethrins, PBO and chlorpyrifos in sediment). Additionally, two separate experiments were performed to determine the effect of PBO on sediment sorbed pyrethroids: one was conducted with a sediment that showed near total lethality to *H. azteca*, and another with a sediment spiked with bifenthrin. The sediment's LC_{50}s were determined, with PBO present in the overlaying water at 0, 4, and 25 µg/L. Water analysis indicated that the sum of pyrethrins I and II, were not detected above the reporting limit of 0.01 µg/L, which was attributed to degradation via photolysis and adsorption by bed sediments; however, PBO was undetected prior to application and reached a maximum level of 3.92 µg/L after application. Sediment sample analyses revealed that pyrethrins I were present at a concentration of up to 403 µg/kg dry weight after application and PBO concentrations were up to 61.4 µg/kg dry weight. There was no evidence of aquatic toxicity due to the application of pyrethrins and PBO alone. The additional testing indicated that PBO concentrations of 2-4 µg/L in the overlaying water were sufficient to enhance previously present sediment pyrethroid toxicity to *H. azteca* by a factor of up to two. Even though there is uncertainty about the PBO actual exposure duration in the environment, at the treatment site PBO was applied on three consecutive nights. This could cause prolonged PBO concentrations in the environment. Water sampling occurred at 10 and 34 hours after application and the difference in PBO concentration between samplings was not appreciable. This article was the first to show that the synergist PBO could pose additional risk to aquatic animals, compared to risk posed by individual insecticide active ingredients, at an environmentally realistic PBO concentration.

The Sacramento-Yolo Mosquito and Vector Control District provided a water quality monitoring effort for the same applications by Weston *et al.* (*40*) (Ziegler (*41*)). No sediment samples were taken for analysis. Water samples were analyzed for pyrethrins and PBO, with respective reporting limits of 0.2 and 1.0 μg/L. Since applications were made in the evening (usually after 8:00 pm), for each application event water samples were taken three times, which represented before application (baseline), in the morning on the day after application (representing immediate post-application), and in the afternoon on the day after application (next day post-application, taken approximately 15 hours after the immediate post-application samples). For the first application event, immediate post-application samples were not taken. Results indicated that the pre-application (baseline) samples were non-detects at the reporting limit for both pyrethrins and PBO. For the immediate post-application samples, 35% and 56% of the water samples were reported as detects for pyrethrins and PBO, respectively. For the next day post-application samples, pyrethrins were not detected in any samples and PBO was detected in 35% of the samples. The maximum pyrethrins concentration reported was 3.77 μg/L and PBO was at a maximum concentration of 20 μg/L.

Schleier and Peterson (*42*) derived LC_{50}s for permethrin, permethrin synergized with PBO, permethrin in the product Permenone®, Permanone® plus PBO, technical naled, and naled in the product Trumpet®, towards the representative medium–to-large ground-dwelling non-target insect, the house cricket (*Acheta domesticus* (L.)). Using ground ULV applications, there were no significant differences in mortalities of caged house crickets exposed to Permanone® or naled, compared to controls. The authors calculated EECs using the Industrial Source Complex Short-Term (ISCST3) dispersion model, which resulted in exceedance of the levels of concern (LOCs) for the house cricket in all cases, except for technical grade permethrin. However, using actual environmental concentrations, only the risk quotient (RQ) for technical grade naled exceeded the LOC. RQs were 10- to 100-fold lower using the measured environmental concentrations than using modeling.

In another monitoring effort, Kuivila *et al.* (*22*) reported sampling for several synthetic pyrethroids in 7 metropolitan areas across the U.S., which excluded California. Among the pyrethroids analyzed, resmethrin was included, which has been used primarily for mosquito abatement. The study reported a frequency of detection of resmethrin in sediment samples of 4% and a highest concentration of 38.3 μg/kg dry weight, a median 5.3 μg/kg, with a method detection limit of 0.5 μg/kg dry weight. According to the article, given that resmethrin is used primarily as an adulticide, the source of the chemical for the site that showed the maximum resmethrin concentration at a site within Estes Park, Colorado (an undeveloped watershed), is aerial applications of resmethrin for mosquito control. According to this article, a previous study had reported a maximum resmethrin concentration in suspended sediment of a San Joaquin Valley, California watershed of 19 µg/kg (*43*).

Phillips *et al.* (*44*) incorporated toxicity testing to monitoring relative to mosquito adulticide applications. As a requirement of a National Pollutant Discharge Elimination System (NPDES) General Permit to comprise discharges to waters from mosquito control applications in California in 2011, the California State Water Resources Control Board and the Mosquito Vector Control Association of California conducted chemical and toxicity analyses in the water column and sediment pre- and post-applications of malathion, naled (and its degradate dichlorvos), permethrin, *d*-phenothrin, pyrethrins, etofenprox, and PBO (plus a suite of other pyrethroids), during 15 mosquitocide applications in 2011 and 2012. Settings included and were labeled as urban, agricultural and wetland environments. Pre-application water and sediment samples were collected in the evening of each application day. Post-application water samples were collected in the early morning hours (12-hr post-application) and evenings of the day after each application (24-hr post-application). The post-application sediment samples were taken 4-7 days post-applications, to allow time for partitioning with the sediments. The toxicity of malathion and naled was assessed using *Ceriodaphnia dubia*, while the toxicity of pyrethroids and pyrethrins was assessed using *Hyalella azteca*.

Only four post-application sediment samples were more toxic than their corresponding pre-application samples; however, the toxicity could not be attributed to the spray events and there was a limited number of chemicals tested (Table 7).

Toxicity of nine out of 16 toxic water samples was related to applications of naled and attributed to it's degradate dichlorvos. Given the limited number of adulticide chemicals available in the market, and that naled is only one of two organophosphate pesticides used for this purpose, the authors recommended best management practices to prevent toxicity due to naled applications. They indicated that some practices are already being implemented (Table 7).

Table 7. Summary of Sampling Results from Monitoring Mosquitocide Applications. (Source: Phillips *et al.* (*44*))

Chemical	*Toxicity*	*Concentrations and Other Notes*
	Sediment	
Pre-App Samples	Out of 17 samples, only one exhibited significant toxicity (*H. azteca*), taken before a *d*-phenothrin application.	The corresponding post-application sample exhibited the same mean survival (74%), and it was not found to be significant.
Permethrin	Only one urban site was sampled for permethrin, which exhibited significant toxicity pre-application (*H. azteca*). See above.	The permethrin concentration was below the toxicity threshold.
Pyrethrins	Two of the urban sites exhibited toxicity post-application (*H. azteca*).	There were no detections of pyrethrins and no sample exceeded the PBO toxicity threshold.
d-phenothrin	Five wetland and five agricultural sites were sampled for *d*-phenothrin, all of which did not exhibit significant toxicity pre- and post-application (*H. azteca*).	No sample had concentrations of *d*-phenothrin or PBO exceeding their toxicity thresholds.
	Water Column	
Pre-App Samples	Out of 53 samples, only one exhibited significant toxicity (*H. azteca*), taken before a *d*-phenothrin application.	The corresponding post-application sample was not significantly toxic at the same site.
Malathion	Two sites were tested, which did not exhibit significant toxicity (*C. dubia*).	The concentrations of malathion were below the organism threshold.
Naled	Six urban, two wetland and one agricultural sites were tested. Significant toxicity was observed in both wetland and all six urban sites (*C. dubia*).	Naled was not detected in any of the sites but its degradate, dichlorvos, was observed at concentrations exceeding the organism threshold in both wetland and four of the urban sites. Trichlorfon, another precursor of dichlorvos, was noted at levels exceeding thresholds in one of the wetland sites.

Continued on next page.

Table 7. (Continued). Summary of Sampling Results from Monitoring Mosquitocide Applications

Chemical	*Toxicity*	*Concentrations and Other Notes*
Etofenprox	The 24-hr post-application sample exhibited significant toxicity (*H. azteca*).	The chemical's concentration was below the reporting limit.
Permethrin	Six agricultural, one wetland and five urban sites were sampled. Three permethrin post-application sites exhibited significant toxicity (one agricultural and two urban) (*H. azteca*).	Permethrin, bifenthrin and PBO concentrations were all below toxicity thresholds in these samples (*H. azteca*) with the exception of one bifenthrin concentration exceeding the threshold in one of the urban sites that exhibited toxicity (12- and 24-hours post-application).
Pyrethrins	Six urban and six wetland sites were monitored, of which only one urban site exhibited toxicity (*H. azteca*).	Even though the concentration of pyrethrins and PBO were below their toxicity thresholds, it turned out that the concentrations of PBO were the highest reported for the samples that exhibited toxicity. The authors speculated that the PBO may have synergized the toxicity of other pyrethroids present in the samples.
d-phenothrin	None of the six agricultural, six wetland and six urban sites monitored exhibited significant toxicity post-application (*H. azteca*).	The concentrations of *d*-phenothrin and PBO were below their toxicity thresholds.

In Describing the Behavior and Effects of Pesticides in Urban and Agricultural Settings; Jones, et al.;
ACS Symposium Series; American Chemical Society: Washington, DC, 2014.

Example Ecological Risk Assessment

In 2008, EFED issued an analysis of the ecological risk assessment for permethrin for the following endangered or threatened species in California (*45*): California red-legged frog *(Rana aurora draytonii*), California clapper rail (*Rallus longirostris obsoletus*), Salt marsh harvest mouse (*Reithrodontomys raviventris*), San Francisco garter snake (*Thamnophis sirtalis tetrataenia*), and Bay checkerspot butterfly (*Euphydryas editha bayensis*). One of the assessed uses of permethrin was for vector control through ULV applications. In the assessment, aquatic and terrestrial species were evaluated. At the time of the review, some labels did not comply with PR Notice 2005-1 (*26*), and therefore, analyses were performed using both pre- and post-PR Notice 2005-1 labels. Table 8 provides the urban aquatic EECs. Compared to a peak water EEC of 0.221 µg/L (post-CFR 2005-1), the single monitoring study that provided water column concentrations of permethrin (*44*) presented a maximum concentration of 0.03 µg/L. The ecological risk assessment did not provide sediment concentrations for comparison; however, they can be estimated based upon the value of organic/carbon partition coefficient (K_{OC} = 76800 L/kg). An estimated conversion factor of 3073 from pore water concentration to sediment concentration is calculated using a spreadsheet and the constants that define the EXAMS ecological pond. The peak pore water concentration was 0.0515 µg/L. The estimated peak sediment concentration is 158 µg/kg, which is above 2-fold higher than the monitored concentration of 65.9 µg/kg.

Table 8. Water Column EECs (µg/L) for Permethrin Uses in California

Scenario	*App Rate (lb a.i./A)*	*Peak (µg/L)*
Recreational areas (Pre-CFR 2005-1)	0.007x7	0.496
Recreational areas (Post-CFR 2005-1)	0.007x7	0.221

In the past, EFED has based its adulticide evaluations on existing turf Pesticide Root Zone Model (PRZM) scenarios for modeling aquatic exposure (*e.g.*, FL, PA or CA turf). These scenarios are used as surrogates for areas such as, but not limited to parks, campsites, woodlands, athletic fields, golf courses, garden playgrounds, and recreational areas; however, for uses in other urban sites, such as residential, the combination of the residential and impervious scenario, run in tandem may be utilized in upcoming assessments. It is expected that the development of new scenarios depicting residential sites and/or impervious surfaces may be further used in the future.

Assessing Pesticide Releases to POTWs

In the context of ecological risk assessment of conventional pesticides at USEPA, the issue of household wastewater releases of pesticides was first raised by public stakeholders from California during the Re-registration Eligibility

Decision (RED) process of the pyrethroid insecticide permethrin (*46*). Concerns were raised that clothes pretreated with permethrin may cause adverse water quality impacts due to releases to POTWs when washed and result in subsequent discharges to receiving waters by POTWs. It is noteworthy that potential releases of antimicrobial pesticides to POTWs have routinely been considered in OPP environmental risk assessments due to their widespread use in consumer care products that result in substantive 'down the drain' releases (*e.g.,* antibacterial ingredients in hand soaps). In contrast, this issue is relatively new for conventional pesticides where exposure from outdoor uses has traditionally been assumed to dominate environmental risk concerns. Monitoring data described later in this section indicates that for some pesticides, releases to (and from) POTWs may be significant to the extent that this exposure pathway requires consideration in USEPA environmental risk assessments. More recently as part of OPP's pesticide Registration Review process, the aforementioned concerns were echoed and additional concerns were identified regarding the potential for environmental exposure to pesticides resulting from their sorption onto biosolids and subsequent biosolid application to land (*47*).

In this section, we summarize the currently available information regarding conventional pesticide releases to POTWs in the U.S. and approaches being considered for evaluating these exposure pathways in OPP's forthcoming ecological risk assessments. We first discuss sources and pesticide uses associated with releases to POTWs. Following this, we describe approaches and data that are being used to model the fate of these releases in POTWs. Finally, we summarize available monitoring data which have been generated specifically to characterize potential pesticide exposure to and from POTWs.

Pesticide Sources to POTWs

In response to the concerns raised by the public regarding the potential release of conventional pesticides to POTWs, OPP reviewed indoor uses of conventional pesticides and identified those that present a high potential for "down the drain" (DtD) releases (Table 9). Generally, these include pesticidal treatments of fabric, clothing and carpets, pet shampoos, and drains with hydrologic connections to sewer systems. Selected uses in greenhouses have been evaluated previously in the context of pesticide releases to both POTWs (assuming connectivity with sewer systems) and surface waters (assuming direct discharge to bodies of water). These uses are therefore being considered as exposure pathways of potential concern in current and forthcoming environmental risk assessments by OPP.

A number of indoor pesticide uses are considered to have lower potential for substantive releases to POTWs based on labeled uses. These include labeled applications of indoor foggers, baits, crack and crevice treatment, and bed and mattress treatments where a hydrological connection to sewer systems is considered highly unlikely or at most, rare. Considerable discussion arose around the use of 'spot on' treatments for pets (*e.g.,* flea and tick control) as well as insecticide-impregnated collars. With spot on treatments, it is expected (and advised on some pesticide labels) that shampooing soon after application of spot on treatments would reduce the efficacy of such treatments, and those would not be cost effective and discouraged. Regarding pet collars, the potential substantive releases to POTWs are considered low based on their expected slow release rate of pesticides from the collars.

Table 9. Indoor Uses of Conventional Pesticides and Their Potential for 'Down the Drain' Release to POTWs

Uses With High Potential for Substantive Release to POTWs
• Pet lotions or shampoos (*e.g.*, treatment for fleas and ticks)
• Products for the treatment of shoes/clothing/textiles (*e.g.*, miticides, sanitizers, deodorizers)
• Pre-treated clothes/textiles, bed sheets, linens, etc.
• Drain treatments that convey water to sanitary sewer systems (root herbicides)
• Storm drain/storm system treatments connected to sewer systems (*e.g.*, root herbicides and filtration media for storm water filtration systems)
• Sewage system treatments (*e.g.*, filtration media for municipal wastewater filtration)
• Carpet treatments (except materials preservatives) removed from carpets during shampooing then subsequently disposed with wash water down-the-drain
• Lice shampoos, skin lotion treatments (*e.g.*, for mites)[1]
• Selected greenhouse uses with drains connected to sewer systems
• Pool treatment[2]
Uses With Lower Potential for Substantive Release to POTWs
• Pesticide-containing pet collars and spot-on treatments
• Bed and mattress treatments (except products to treat bed sheets)
• Storm water system treatments not connected to sewer systems
• Crack and crevice treatment
• Indoor foggers
• Indoor baits

[1] Although this is considered a pharmaceutical use, EPA in agreement with FDA is assessing exposure from down the drain releases. [2] Even though pools are typically considered outdoor use patterns, generally localities require discharging their water to sanitary sewers.

It is important to note that the pesticide uses identified in Table 9 do not represent *all* potential sources of pesticide input to POTWs. Rather, they represent those uses that are currently being assessed as part of DtD modeling in OPP environmental risk assessments. For example, pesticides may potentially be released by industrial discharges to POTWs from pesticide manufacturers. However, such releases are subject to regulation under other environmental statutes and regulatory programs (*e.g.,* state and federal pretreatment programs under the authority of the Clean Water Act), and not under FIFRA. It is recognized that certain outdoor residential uses of pesticides may contribute to pesticide loadings to storm water systems which are connected to POTWs. Modeling of outdoor residential use of pesticides in OPP environmental assessments is presently focused on direct loadings to surface water. Information from the open literature suggests that some POTWs may experience greater flow during wet weather events even when direct connections to storm water inputs are not apparent (*19*). Presumably, such inputs represent groundwater intrusion and/or fugitive inputs from storm water runoff. For these and other sources of pesticides to POTWs unaccounted for in Table 9, OPP is relying on targeted monitoring data to ascertain inputs to and discharges from POTWs.

Modeling Approach for POTW Assessment

In order to address the issue of releases to domestic wastewater, OPP has relied on the consumer exposure model, *Exposure and Fate Assessment Screening Tool* (E-FAST, v.2.0) that was developed for assessing industrial chemicals in EPA's Office of Pollution Prevention and Toxics (*48*). The 'Down-the-Drain' module (DtD) of E-FAST v.2.0 is specifically designed to address sources of a chemical that could potentially be disposed into domestic wastewater from a DtD application. The DtD module can be used to represent residential, domestic and certain commercial facilities (*e.g.*, supermarkets, storage facilities and warehouse uses likely to end up in drains). This model provides screening-level estimates of chemical residues in surface water that may result from household uses and the disposal of consumer products into wastewater.

Conceptually, the E-FAST's DtD module assumes that in a given year the entire production volume of a chemical (*i.e.*, the amount of pesticide) is parceled out on a daily basis to the entire U.S. population and converted to a mass release per capita, and subsequently, a daily per capita release to a wastewater treatment facility (*i.e.,* g/person/day). This mass is then diluted into the average daily volume of wastewater released per person to arrive at an estimated concentration of the chemical in wastewater prior to entering a treatment facility. The underlying equations used by the DtD module are shown below. The daily per capita release is defined as follows.

$$H_R = \frac{PV}{Pop} \times \frac{1000\ g}{1\ kg} \times \frac{1\ year}{365\ days}$$

where,

i. H_R is the daily per capita release of the chemical (g/person/day);
ii. P_V is the production volume of the chemical being evaluated that is produced annually in the USA that is discharged into domestic wastewaters (kg/year); and
iii. *Pop* is the 2003 U.S. resident population (2.908×10^8 persons) (U.S. Bureau of the Census, 2004-2005).

The chemical's concentration in untreated wastewater is then reduced by the fraction removed during wastewater treatment processes. The remaining chemical is discharged into surface water (*e.g.,* a river or stream), where it is assumed that it is instantaneously diluted, with no further removal. The surface water concentration is calculated using the following general equation.

$$SWC = \frac{H_R \times \frac{1}{Q_H} \times \left(1 - \frac{WWT}{100}\right) \times \frac{10^6\ \mu g}{g}}{SDF}$$

where,

i. *SWC* is the surface water concentration (µg/L);
ii. Q_H is the household wastewater volume released daily (it is estimated to be 388 L per person per day), it includes only domestic and residential POTWs;
iii. *WWT* is the wastewater treatment removal (percent removed prior to discharging into a body of water, %); and
iv. *SDF* is the stream dilution factor.

The Stream Dilution Factor (SDF) is the volume of the receiving stream flow divided by the volume of the wastewater released from the POTW or effluent flow (SDF = SF/EF). There are four types of stream flows that the developers of the model have deemed adequate for the protection of aquatic life and human health (acute and chronic). Additionally, flows have been characterized to represent mid-sized receiving bodies of water and smaller streams. It should be noted that the DtD module of E-FAST is a screening-level model and the results should be treated as such. It does not take into account processes such as degradation prior to treatment at the facility, or partitioning (*i.e.,* sorption by sediment or particulate matter).

Model Inputs

There are two main input values in the E-FAST's DtD module: the production volume (PV), and the percent removal from wastewater treatment (WWT). (The BCF is an input parameter, which the model uses for calculations that are not relevant to EFED's purpose to calculate aquatic EECs.) The PV can be obtained from the registrant(s) sources or can be supplied by the Biological and Economic Analysis Division (BEAD). Model results are sensitive to the WWT, which in turn is dependent on the physicochemical properties of the active ingredient of concern and the extent of wastewater treatment (*e.g.,* primary, secondary, tertiary, or ultrafiltration). An estimate of WWT is available from the Sewage Treatment Plant Fugacity Model (STPWIN™) of EPI Suite v.4.11 (*49*). This model provides estimates of the fate of organic chemicals in conventional wastewater treatment plant that uses activated sludge secondary treatment. According to the STPWIN™ Help manual, EPI Suite's STP program was conservative predicting removal percent (WWT) 88% of the time using its default half-lives of 10,000 hours for 29 of 33 chemicals evaluated, for primary clarifier, aeration vessel and settling tank; however, the evaluation was based on a set chemicals which are not pesticides. A more suitable and reliable alternative, is data derived from a bench scale study (described further below) that may be required either during the registration process of the chemical or during registration review, to further refine this input parameter. Finally, for a few chemicals, WWT can be obtained from actual monitoring studies of influent and effluent from POTWs. This has been used in the past to refine estimates of permethrin.

Table 10 provides a summary of removals by various mechanisms for eight pyrethroid insecticides predicted by STPWIN™. As shown in the table, the module predicts that for these chemicals, the main removal mechanism is sludge adsorption. The total biodegradation is low while the release to air is minimal.

Model Outputs

In the past, EFED has conducted preliminary DtD screens of a pesticide to determine the need for a bench scale POTW treatability study. In some cases the modeling results indicated that the study is not needed (*e.g.*, pyrethrins, spinosad). The modeling is possible if the production volume or its estimate, is available to the assessor. The assessor models the chemical with the aid of the EPI Suite v.4.11 model and gets an estimate of the level of removal (*i.e.*, WWT) from the module 'Sewage Treatment Plant Fugacity Model (STP)' using the default half-lives of 10,000 hours (~417 d) in the primary tank, the aeration tank, and the settling tank. This may be considered a conservative value (alternatively EPI Suite provides the second option to enter half-lives derived from monitoring experiments, or the third option to use model-estimated half-lives for the above mentioned processes). Suitable flows and the 10th percentile concentrations are used to derive RQs. The RQs derived from this process are compared against the LOCs. If they are well below the LOCs, it may be determined whether a treatability study is required using best professional judgment and considering the conservativeness of the preliminary risk assessment.

The most recent assessment for which the E-FAST's DtD module was used was an ecological risk assessment for deltamethrin. Uses assessed included sewage systems treatments (*50*). It was assumed that the upper bound value of the production volume is 50 kg a.i./year. After running the chemical using the DtD module, the acute concentration was found to be 0.000425 ppb and the chronic concentration was 0.000425 ppb (the same value). For freshwater and estuarine/marine fish and vascular and non-vascular plants there were no exceedances of LOCs. A summary of the findings on invertebrates is shown in Table 11.

Table 10. Removal Percent of Eight Pyrethroids in Wastewater Treatment Plants Obtained from EPISUITE v.4.11 and Its STPWIN Module[1]

Process	*Bifent.*	*Fenprop.*	*Cyhalot.*	*Permet.*	*Cyflut.*	*Cypermet*	*Esfenval.*	*Deltamet.*
Sludge Adsorption	93.2	91.4	93.1	92.7	91.2	93.0	92.1	92.1
Total Biodegradation	0.78	0.77	0.78	0.78	0.77	0.78	0.77	0.77
Total to Air	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00
Total Removal	94.0	92.2	93.8	93.4	91.9	93.8	92.9	92.8

[1] All results were rounded to two decimal places or three significant figures.

Table 11. Summary of Acute and Chronic RQs for Aquatic Freshwater and Estuarine/Marine Invertebrates Exposed to Deltamethrin

Use	*App Rate*	*Peak EEC (µg/L)*	*21-day EEC (µg/L)*	*Acute RQ*	*Chronic RQ*
Freshwater Invertebrates[2]	50 kg/yr	0.000425	0.000425	0.11[1]	>16.3[1]
Estuarine/Marine Invertebrates[3]	50 kg/yr	0.000425	0.000425	0.11[1]	0.58

[1] RQs that exceed the EPA's levels of concern. [2] Acute RQ = use-specific peak EEC/0.004 ppb [for the amphipod *G. fossarum*]. Chronic RQ = use-specific 21-day EEC/10-d NOAEC of <$2.6x10^{-5}$ ug a.i./L [for the amphipod, *H. azteca*]. Chronic RQ values are expressed as ">" values because the NOAEC is non-definitive ("<") [3] Acute RQ = use-specific peak EEC/0.0037 ppb [for mysid shrimp, *Americamysis bahia*]. Chronic RQ = use-specific 21-day EEC/0.00073 ppb [from data for *A. bahia*].

Refinement of E-FAST (Bench Scale Study)

Based on experience with DtD modeling with pyrethroids, OPP requested additional data from registrants to improve modeling of the fate and removal efficiency of pyrethroids in POTWs. In response, registrants conducted a bench scale study simulating four processes that occur in POTWs: primary settling, aerobic biological treatment, anaerobic digestion, and ultra-filtration (Cleary and McGrath, MRID 48762906 (*51*)). Pyrethroids studied in these processes included: permethrin, deltamethrin, bifenthrin, cyfluthrin, *lambda*-cyhalothrin, cypermethrin, esfenvalerate, and fenpropathrin. Although in treatment plants, they occur simultaneously, these processes were evaluated separately from each other (*i.e.*, they were treated as modules). First, primary settling, anaerobic digestion, and ultrafiltration were measured in batch mode (*i.e.*, a set amount of sample was submitted to the process and evaluated after a period of time, *e.g.*, two hours for primary settling, up to 35 days for anaerobic digestion). Meanwhile, the aerobic biological treatment was evaluated in a continuous process (*i.e.*, process was continuously circulated and sample was evaluated throughout the procedure for 50 days, with a target solids retention time (SRT) of 10 days). Table 12 provides a results synopsis of the study. Note that the estimated removals are for specific modules and not overall removals. The percentages are not additive.

Wastewater from a treatment plant in Ridgewood, New Jersey, was spiked with known levels of the eight pyrethroids, well above background levels (5 µg/L of each pyrethroid, with the exception of permethrin at 50 µg/L), in order to characterize each process. This study was not conducted in compliance with Good Laboratory Practice Standards set forth in Title 40, Part 160 of the Code of Federal Regulations.

Table 12. Results Synopsis: Removal Percent of Eight Pyrethroids in Certain Treatment Processes Simulated in a Bench Scale Wastewater Treatability Study[1]

Process	*Bifent.*	*Fenprop.*	*l-Cyhal.*	*Permet.*	*Cyflut.*	*Cyperme.*	*Esfenval.*	*Deltamet.*
Primary Settling	LR[2]	LR[2]	LR[2]	LR[2]	LR[2]	LR[2]	LR[2]	LR[2]
Aerobic Chamber	51.9	80.1	48.6	86.6	73.2	76.3	56.1	59.1
Anaerobic Digestion	32.1	45.5	57.0	43.5	81.2	78.1	79.2	77.1
Ultrafiltration	91.7	95.7	93.1	96.9	95.7	95.4	93.6	92.6

[1] The percent shown is for each of the individual modules (refer to text).

[2] LR means limited removal was achieved in this process.

Primary Settling

The primary settling experiment was conducted in batch mode. In primary settling, incoming wastewater (primary influent), was kept in a quiescent state for a specific period of time (in this study it was 2 hours), to allow heavy particles to settle. The result of the process was primary effluent (the supernatant) and primary sludge. Pyrethroids were measured in the primary influent wastewater and in the primary effluent and sludge. The primary effluent had concentrations of pyrethroids that were very similar to the concentrations in the primary influent. Primary settling did not appear to be effective to remove substantial amounts of pyrethroids from the primary influent.

Aerobic Biological Treatment

The primary effluent was added to the aerobic biological treatment system to reduce its organic content. The aerobic system was kept at *ca.* 20°C and it consisted of two submodules: the aeration system in which dissolved oxygen promotes aerobic biological degradation, and secondary settling. This part of the experiment was run for 50 days, where secondary sludge and primary effluent were fed to the aerobic chamber, in a continuous flow system. The target SRT was 10 days, which was reported to represent a likely best case scenario. Pyrethroids were removed moderately from the secondary influent (or primary effluent from the primary settling module), in the aerobic chamber. Removals ranged from 52 to 87 percent, for bifenthrin and permethrin, respectively.

Anaerobic Digestion

A specific amount of primary sludge (*i.e.*, sludge from primary settling) was submitted to digestion and run in batch mode for 35 days at *ca.* 35°C under anaerobic conditions. Pyrethroids were also removed moderately from primary sludge under these testing conditions in the anaerobic chamber. Among eight pyrethroids tested, removals ranged from 32 to 81 percent, for bifenthrin and cyfluthrin, respectively, attributed to anaerobic biological digestion.

Ultrafiltration

In ultrafiltration, the supernatants from the secondary settling were filtered and remaining solids were removed, reducing further the suspended particles, and thus the organic matter associated with those particles, and its associated pyrethroids. This process was run in batch mode, using an apparatus and method similar to the one used to measure total suspended solids. Removal represents the amount remaining in the effluent minus the amount applied of each pyrethroid in the influent. Ultrafiltration appeared to be the process that removed the highest percentage of pyrethroids from the secondary effluent, with over 90 percent of pyrethroid removed from the final effluent. It is noted, however, that ultrafiltration is not a process employed by all WWTPs nationwide. Results presented in Table 12 are the means of two values, using a 0.1 µm filter.

Utility of the Bench Scale Study

The bench scale treatability study is useful in understanding the relative contributions of the different processes that occur at a treatment facility. Removal processes include primary settling, which shows very limited removal, and aerobic and anaerobic digestion, which show moderate levels of removal. Only ultrafiltration appeared to remove over 90% of the material in the bench scale study. Results from modeling (using EPISUITE giving total removal) and monitoring data (as discussed below) indicate levels of removal of above 90%. However, direct comparison of the bench scale study results to modeling and monitoring data is confounded by the fact that the bench scale study design does not enable determination of an overall removal efficiency based on the sum of the simulated treatment processes. Therefore, the utility of the bench scale study mostly relates to how separate processes affect pyrethroid removal and not for an estimate of the overall removal efficiency of pyrethroids from POTWs.

POTW Monitoring Data

The available information on the occurrence of pesticides in U.S. POTW influent, effluent and biosolids was reviewed and is summarized here with a focus on the following questions:

1. Which pesticides are most commonly detected in POTWs and how does this relate to their intended uses?
2. What is the removal efficiency of pesticides by wastewater treatment processes and how does this compare to estimates based on modeling and bench scale treatability studies?

Although a number of country-wide surveys of pesticides and other micropollutants in POTW wastewater have been conducted in Europe (*e.g.,* Loos *et al.* (*52*); Luo *et al.* (*53*)), an analogous U.S. wide survey was not identified in this review. Instead, several state-wide and POTW-specific surveys were identified and are summarized below.

Pesticides in POTW Influent and Effluent

Oregon POTWs

In one of the most comprehensive surveys of chemical contaminants in POTW effluent in the U.S., Hope *et al.* (*54*) analyzed effluents from 52 POTWs throughout Oregon once during the summer and during the fall of

2010. Of the 406 chemicals included in the survey, 149 were categorized as pesticides or pesticide-related chemicals (pesticide precursors, degradation products). The most frequently detected pesticide-related compounds include: 2,6-dichlorophenol (93%), arsenic (86%), DEET (78%), 2,4,6-trichlorophenol (72%), 2,4-dichlorophenol (62%), diuron (46%) 2,4,5-trichlorophenol (16%) and 2,3,4,5-tetrachlorophenol (13%) (Table 13). However, the presence of many of these compounds cannot be unambiguously traced to pesticide use. Specifically, the chlorinated phenols may be used as chemical intermediaries, are no longer registered as pesticides, and/or may be produced as a byproduct of effluent chlorination. Arsenic has some remaining commercial and industrial uses (*e.g.,* as a component of the chromated copper arsenate wood preservative) but also occurs naturally in the environment and may be released to POTWs via other commercial or industrial processes.

On the contrary, the presence of the insect repellant DEET most likely results from its widespread application to skin and subsequent washoff into household drains. DEET, the third most frequently detected pesticide, has the second greatest median concentration (232 ng/L and the greatest maximum concentration detected (13,600 ng/L). Diuron, the 6th most frequently detected pesticide, is a pre- and post-emergent herbicide with numerous agricultural and residential use sites, including application to water bodies for aquatic weed control. Particularly relevant to its occurrence in POTW effluent is its use as a mildewcide in certain paints and stains. This use could conceivably lead to down-the-drain releases to POTWs through washing of brushes and other painting equipment. Diuron and DEET were also among the most commonly detected pesticides in POTW effluent across Europe (*52*).

Triclopyr (detected in 11% of the samples), is used for broadleaf control in a variety of agricultural and residential settings. With no registered indoor uses in the U.S., direct release of triclopyr to POTWs via household drains is not expected. However, its use for weed control in residential settings could result in releases into stormwater runoff and subsequently to POTWs with hydrologically-connected stormwater conveyances. Interestingly, the herbicide 2,4-DB has no registered indoor or residential uses. Potential reasons for occurrence in 10% of the Oregon POTW effluent samples are not clear. Imidacloprid (10%) and imazapyr (9%), both have widespread residential uses for insect and weed control, respectively. Imidacloprid is also commonly used for flea control on pets via pet collars and spot-on treatments. It seems possible that its presence in POTWs could relate to pet washing or inappropriate disposal down the drain.

Hope *et al.* (*54*) report that detection of the fungicide propiconazole, used to prevent mold on wood, may have been related to discharge from a wood processing facility that discharged to a POTW. Propiconazole is also an ingredient in paints and stains which may also lead to releases to POTWs, possibly through washing of painting equipment and/or runoff into storm water connected to POTWs. The authors also note that fluridone, imazapyr and terbutylazine are applied directly to surface water for algae and macrophyte control and speculate that surface water intrusion into POTW conveyance systems may be occurring.

Table 13. Pesticides and Related Compounds Detected in a Survey of 52 Oregon POTWs. Source: Hope *et al.* (*54*)

Chemical	*CAS*	*LOQ (ng/L)*	*% Detect. (n=102)*	*Min. (ng/L)*	*Median (ng/L)*	*Max. (ng/L)*	*Category*
2,6-Dichlorophenol	87-65-0	7.7	93	10.3	82.4	864	other [1]
Arsenic (TR)	7440-38-2	250	86	260	620	4320	other [2]
DEET (*N,N*-diethyl-*m*-toluamide)	134-62-3	5	78	53	232	13600	other (insect repellant)
2,4,6-Trichlorophenol	88-06-2	19	72	25	55.6	339	wood preservative [3]
2,4-Dichlorophenol	120-83-2	19	62	19.8	68.5	470	other [1]
Diuron	330-54-1	4	46	38	89	775	herbicide
2,4,5-Trichlorophenol	95-95-4	19	16	21.4	42.4	300	other [3]
2,3,4,5-Tetrachlorophenol	4901-51-3	19	13	43.6	48.3	200	other [1]
Triclopyr	55335-06-3	300	11	310	620	3900	herbicide
2,4-DB	94-82-6	610	10	660	127	7440	herbicide
Imidacloprid	138261-41-3	20	10	202	237	387	insecticide
Imazapyr	81334-34-1	40	9	534	1670	17200	herbicide
Azobenzene	103-33-3	19	7	55	108	178	other [3]
Carbaryl	63-25-2	5	7	66	137	663	insecticide
2,4-D	94-75-7	100	3	1600	1630	1890	herbicide
Chlorpropham (CIPC)	101-21-3	7.7	3	17	46.1	72.4	herbicide

Continued on next page.

Table 13. (Continued). Pesticides and Related Compounds Detected in a Survey of 52 Oregon POTWs

Chemical	*CAS*	*LOQ (ng/L)*	*% Detect. (n=102)*	*Min. (ng/L)*	*Median (ng/L)*	*Max. (ng/L)*	*Category*
Dicamba	1918-00-9	300	3	380	700	760	herbicide
Prometon	1610-18-0	4	3	55	63	64	herbicide
Propiconazole	60207-90-1	20	3	387	7210	9020	fungicide
Pentachlorophenol	87-86-5	100	2	220	260	300	fungicide
Baygon	127779-20-8	4	1	42	42	42	insecticide
Dichloroprop (2,4-DP)	120-36-5	300	1	370	370	370	herbicide
Fluridone	59756-60-4	7.7	1	27	27	27	herbicide
Pentachlorobenzene	1825-21-4	380	1	416	416	416	other (PCP degradate)
Pentachlorophenol	87-86-5	380	1	700	700	700	fungicide
Simazine	122-34-9	4	1	56	56	56	herbicide
Terbutylazine	5915-49-3	4	1	61	61	61	herbicide

[1] Pesticide precursor or other chemical intermediary. [2] Organo arsenate and residential CCA uses no longer registered in the U.S. [3] Pesticide is no longer registered in the U.S.

California POTWs

In another comprehensive state-wide survey, Markle *et al.* (*55*) sampled 31 POTWs in California for the presence of eight pyrethroids in influent, effluent, and/or biosolids. This effort was conducted by the Pyrethroid Working Group (PWG), a consortium of registrants representing eight pyrethroids, in response to pyrethroid re-evaluation activities by both the California Department of Pesticide Regulation and the USEPA. The POTWs the surveyed represent approximately 40% of the treated municipal wastewater in California and include primary, secondary and tertiary treatment as terminal wastewater treatment processes. Samples were taken from January through March, 2013 during dry weather period. Consecutive grab samples were taken from influent, effluent and biosolids (when available) and did not account for hydrologic retention time between entry to the POTW and discharge. Extensive quality control measures were instituted including separate analytical measurement by two laboratories.

Results indicate high detection frequencies (*e.g.*, 43% to 100%) for 7 of the 8 pyrethroids sampled in POTW influent (Table 14). Frequencies of detection exceeded 80% for bifenthrin, cyfluthrin, *lambda*-cyhalothrin, cypermethrin and permethrin. Fenpropathrin was the least detected pyrethroid in effluent at 4.5% and is the only pyrethroid sampled that is not registered for residential uses in California. This suggests residential uses of these products are contributing to their loadings to California POTWs. By far the highest maximum and median influent concentrations reported are for permethrin (3,800 and 230 ng/L, respectively), which may be related to its topical use to treat lice infestations.

In POTW effluent, the greatest detection frequencies are observed for bifenthrin (82%), followed by cypermethrin (81%), permethrin (65%), cyfluthrin (60%), *lambda*-cyhalothrin (48%) and esfenvalerate (32%; Table 15). Comparatively, the rates of detection for deltamethrin and fenpropathrin are much lower (16% and 3% respectively) in effluent than influent. Consistent with the influent sampling results, the greatest maximum and median concentrations in POTW effluent are observed for permethrin (170 and 9.4 ng/L, respectively). Cypermethrin showed the next highest effluent concentrations with maximum and median values of 13 and 1.3 ng/L, respectively. Maximum and median concentrations for the other six pyrethroid are 2 orders of magnitude below that for permethrin.

It is instructive to compare the results of POTW monitoring to that predicted by down-the-drain modeling (DtD) using E-FAST described earlier, as a way of evaluating model predictions. Previous DtD assessments were conducted with permethrin and deltamethrin (USEPA (*45*) and USEPA (*50*), respectively) and are shown in Table 16 along with the monitored concentrations in effluent summarized in Table 15. With permethrin, the predicted concentrations in POTW effluent is 0.09 ppb, which is an order of magnitude above the median concentration measured in California POTWs by Markle *et al.* (*55*) However, it is about 2X below the maximum concentration detected in California POTW effluent (0.17 ppb). With deltamethrin, the predicted concentration (0.0004 ppb) is comparable to the median and maximum measured concentrations (0.0003 and 0.001 ppb, respectively).

Table 14. Summary of Pyrethroid Measurements in Influent from 31 California POTWs. (Source: Markle *et al.* (*55*))

Chemical	*# of Detects*	*% Detected*	*LOQ (ng/L)*	*Max. (ng/L)*	*Min. (ng/L)*	*Average[1] (ng/L)*	*Median[1] (ng/L)*
Bifenthrin	64	96%	5	74	ND	15	9.7
Cyfluthrin	59	88%	5	55	ND	11	7.4
Lambda-Cyhalothrin	54	81%	5	72	ND	5.6	2.8
Cypermethrin	54	81%	5	200	ND	35	21
Deltamethrin	29	43%	10	210	ND	8	3.3
Esfenvalerate	31	46%	5	360	ND	8.1	1.7
Fenpropathrin	3	4.5%	5	130	ND	4.6	1.7
Permethrin	67	100%	50	3800	30	330	230

ND = Not detected. A total of 67 influent samples were collected (62 samples + 5 repeats). [1] Median and average values were calculated assuming the limit of quantitation for non-detects.

Table 15. Summary of Pyrethroid Measurements in Effluent from 31 California POTWs. (Source: Markle *et al.* (*55*))

Chemical	*# of Detects*	*% Detected*	*LOQ (ng/L)*	*Max. (ng/L)*	*Min. (ng/L)*	*Average[1] (ng/L)*	*Median[1] (ng/L)*
Bifenthrin	51	82%	0.5	3.9	ND	0.89	0.6
Cyfluthrin	37	60%	0.5	4	ND	0.6	0.3
Lambda-Cyhalothrin	30	48%	0.5	1.6	ND	0.3	0.2
Cypermethrin	50	81%	0.5	13	ND	2.11	1.3
Deltamethrin	10	16%	1.0	1.2	ND	0.31	0.3
Esfenvalerate	20	32%	0.5	0.6	ND	0.25	0.2
Fenpropathrin	2	3.2%	0.5	0.8	ND	0.22	0.2
Permethrin	40	65%	5.0	170	ND	20	9.4

ND = Not detected. A total of 67 effluent samples were collected. [1] Median and average values were calculated assuming the limit of quantitation for non-detects.

Table 16. Estimated Environmental Exposure Concentrations of Permethrin and Deltamethrin, from POTW Discharges

Chemical	*Production Volume (kg)*	*WWT (%)*[1]	*Predicted Conc. (ppb)*	*Measured Conc. (ppb) (min, med, max)*
Permethrin	60,900	93.4	0.09	ND, 0.009, 0.17
Deltamethrin	50	65	0.0004	ND, 0.0003, 0.001

[1] WWT = percent removal from wastewater treatment.

It is also of interest to evaluate the removal of pyrethroids by POTW treatment, since this information can help inform modeling approaches for estimating pyrethroid loadings from POTWs. Influent and effluent data from Markle *et al.* (*55*) were used to calculate percent removal of pyrethroids using the following equation:

$$\%\ \text{Removal} = \left(1 - \frac{\text{Effluent Concentration}}{\text{Influent Concentration}}\right) \times (100\%)$$

When the effluent concentration was reported below limits of quantitation (LOQ), the concentration was equated to the LOQ. When the influent was reported to be below the LOQ, no calculation was made. On average, pyrethroid concentrations measured in POTW effluent are approximately 10% those measured in influent, representing a reduction of approximately 90% (Figure 15). The higher mean % removal indicated for esfenvalerate (97%) and fenpropathrin (99%) are based on very few samples and are therefore considered highly uncertain.

In terms of POTW-specific factors affecting pyrethroid concentrations, there was typically a large reduction in pyrethroid concentrations in effluent from primary to secondary treatment, although only one plant sampled had primary treatment as its terminal treatment process. The relationship between secondary and tertiary treatment was less clear, whereby some POTWs containing secondary treatment had higher concentrations in effluent compared to those with tertiary treatment and vice versa.

It is noted that the study by Markle *et al.* (*55*) was not specifically designed to estimate % removal efficiency of pyrethroids because samples were taken concurrently from influent and effluent without regard to the retention time of treated water in the POTW. Therefore, differences between concentrations of pyrethroid in influent and effluent may reflect, not only partitioning and degradation processes associated with wastewater treatment, but also variation in pesticide loadings over time. Nonetheless, average % removal efficiencies based on the monitoring data (90-99%) are quite similar to those calculated using the STPWIN™ model summarized in Table 10 (91-93%).

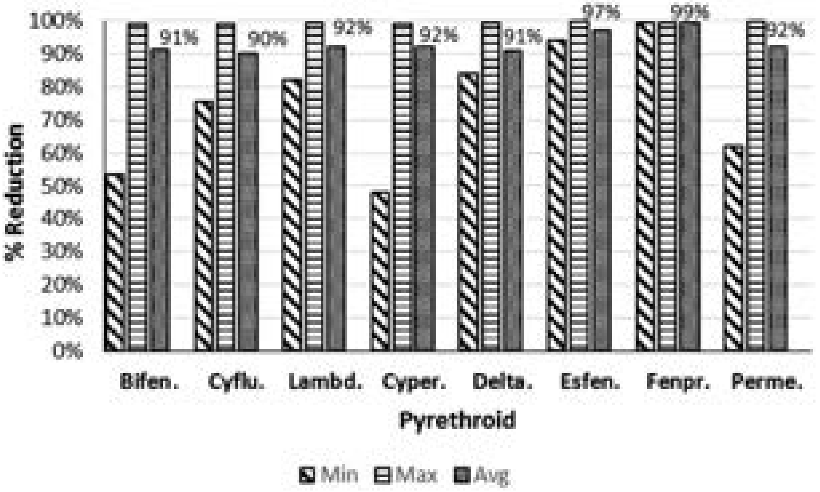

Figure 15. Percent reduction in pyrethroid concentrations in POTW effluent relative to influent. (Source: Markle et al. (55))

Sacramento POTW

In contrast to the previous two studies which conducted limited sampling of POTW wastewater across many facilities, Weston *et al.* (*56*) focused their efforts on a single facility, the Sacramento Regional County Sanitation District Treatment Plant. Concentrations of eight pyrethroids in influent and effluent were sampled over multiple time periods from November 2010 to January 2012. Twelve, 24-h composite samples were taken monthly from influent and seven 24-h, flow-weighted composite samples were taken from effluent (4 during rain events and 3 during dry events). Importantly, the timing of effluent samples was adjusted to account for the retention time of the wastewater in the plant. This facilitates more accurate estimation of % removal efficiency compared to the previous study by Markle *et al.* (*55*) Weston *et al.* (*56*) also sampled three POTW wastewater interceptors during the course of this study, one of which (City interceptor) received both municipal sewage and storm water runoff while the other two (Folsom and Laguna interceptors) received only municipal sewage.

Results from this study indicate that four pyrethroids were detected in all (100%) of the 12 monthly POTW influent samples (permethrin, bifenthrin cypermethrin, and *lambda*-cyhalothrin). Among these, permethrin was the dominant pyrethroid detected in terms of overall concentration and typically ranged between 200 and 400 ng/L. Cypermethrin and bifenthrin were generally found between 20 and 40 ng/L in influent while cyhalothrins were found up to 30 ng/L. Cyfluthrin was detected once in influent during the study, while deltamethrin, fenpropathrin and esfenvalerate were not detected in any of the 12 influent samples. Attempts to correlate temporal peaks in influent concentrations with known use pattern or sales data were not successful. Analysis of pyrethroids in the wastewater interceptor upstream of the treatment plant suggest that storm water runoff was not the dominant source of pyrethroids to the plant. Concentrations of permethrin in the City interceptor (receiving stormwater) were slightly lower than those which did not receive stormwater. Furthermore, all the interceptors sampled contained substantially lower concentrations of permethrin than what was found in the POTW influent, suggesting that other sources of permethrin to the plant are likely. The other pyrethroids were found in similar concentrations in the three interceptors compared to POTW influent. The authors speculate that indoor uses of pyrethroids, container washing and possibly improper disposal of unwanted pesticide may be leading to the loadings to the Sacramento POTW.

In terms of effluent quality, permethrin was again the dominant pyrethroid detected in all but one of the seven effluent samples, ranging from 12-45 ng/L. Bifenthrin and cyhalothrins ranged from 1-5 ng/L in effluent and were detected 43% and 29% of the time, respectively. Concentrations of permethrin, bifenthrin and cyhalothrin were up to 2 times the respective 96-h EC_{50} values reported for the freshwater amphipod, *Hyalella azteca*. However, attempts to correlate observed toxicity to *H. azteca* in effluent samples with toxic units or TIE procedure were not definitive in terms of the cause of toxicity. Removal efficiencies of the pyrethroids from the POTW influent generally ranged from 90-95%, which is similar to the findings reported by Markle *et al.* (*55*) in their California-wide POTW survey.

Weston and Lydy (19)

In this study, Weston and Lydy sampled three California POTWs (Sacramento, Stockton, and Vacaville) for the presence of 8 pyrethroids and chlorpyrifos during three dry and three wet seasons in 2008 and 2009. The authors indicate that except for a small portion of the Sacramento POTW influent, all plants contained sanitary sewer systems that were separate from stormwater systems. They further note that the Stockton POTW included tertiary treatment via routing secondary treated wastewater through 240 ha of treatment ponds which yielded a retention time of about 30 days. Results above 1 ng/L are considered by the authors to be reliable. A total of 18 POTW samples were taken. Other samples of agricultural drains and urban runoff were also analyzed but are not discussed here.

Weston and Lydy (*19*) report that of all the samples, quantifiable concentrations of one or more pyrethroids were found in 67% of the samples taken. Across all three facilities, chlorpyrifos (40%), bifenthrin (39%) and permethrin (33%) were most commonly detected (Table 17). Generally, the highest concentrations of pyrethroids and chlorpyrifos are seen with the Sacramento POTW. In terms of toxicological relevance, 22% of the effluent samples containing bifenthrin, 17% containing *lambda*-cyhalothrin, and 6% of the samples containing cypermethrin exceeded the respective EC_{50} or LC_{50} values for *H. azteca*. The authors note that the presence of pyrethroids is surprising especially given the low levels of suspended solids in the effluent (< 8 mg/L). They suggest that sewer disposal of household pesticides, use of pet and lice control shampoos and laundering of permethrin-treated clothing may be potential sources of pyrethroids to the POTWs. Despite 25-50% greater flows in wet weather, Weston and Lydy (*19*) report similar concentrations in effluent during dry and wet weather flows, which indicates that pesticide loadings from urban/residential runoff may be contributing to loadings to POTWs.

Pesticides in POTW Biosolids

Section 405(d) of the Clean Water Act (CWA) requires the U.S. Environmental Protection Agency (EPA) to identify and regulate toxic pollutants that may be present in biosolids (sewage sludge) at levels of concern for public health and the environment. Historically, the focus of identification and regulatory efforts has been on industrial chemicals, pharmaceuticals, metals, and selected antimicrobial chemicals. (*57*). However, recent studies have raised attention on the occurrence of conventional pesticides in biosolids, which often are treated and applied to land. Potential consequences of land-applied biosolids that contain appreciable amounts of pesticides include alteration of soil and terrestrial biota, runoff to surface waters and contamination of ground water.

In addition to quantifying pyrethroid concentrations in POTW influent and effluent, the previously summarized study conducted Markle *et al.* (*55*) also measured pyrethroids in biosolids from 24 of the POTWs included in the survey (Table 18). In terms of overall detection frequency, results mirror those described previously for influent and effluent, with the highest detection frequencies reported for bifenthrin (96%), permethrin (92%), cypermethrin (90%), and cyfluthrin (87%). The maximum concentration of permethrin (11,000 ng/g d.w.) is 10X that of the pyrethroids with the next highest maxima concentrations (bifenthrin, cypermethrin). Median concentrations are greatest for permethrin (1,200 ng/g d.w.), bifenthrin (120 ng/g d.w.) and cypermethrin (79 ng/g d.w.). Permethrin was also reported in sewage sludge from the U.K. (*58*) and Switzerland (*59*).

As a consequence of these and other reports of conventional pesticides in POTW biosolids, OPP has undertaken efforts along with counterparts in the Office of Water to develop approaches to screen uses of conventional pesticides for their potential to end up and persist in biosolids. The initial efforts focused on identifying pesticide uses with the greatest potential for releases down the drain (Table 9). Subsequently, efforts have focused on developing screening level models for evaluating the potential risks associated with pesticides in land-applied biosolids. One approach being evaluated is adapting the current Office of Water Biosolids Core Risk Assessment Model (BCRAM) for a screening level assessment. Other approaches being investigated include adapting existing OPP models (*e.g.*, PRZM) and exposure scenarios for evaluation of land applied biosolids.

Table 17. Pyrethroids and Chlorpyrifos in Effluent from Three California POTWs. (Source: Weston and Lydy (*19*))

POTW[1]	***Bifen.***	***Cyfl.***	***Cyp.***	***Delt.***	***Esfen.***	***Fenp.***	***L. Cyh***	***Perm.***	***Chlor.***
Maximum Concentration Detected (ng/L)									
Sacr.	2.7	1.7	17.0	0	3.7	0	5.5	17.2	24.1
Stock.	4.8	0	0	1.3	0	0	0	7.9	5.5
Vaca.	6.3	0	0	2.7	0	0	2.8	7.6	0
Overall Detection Frequency (n=18) [2]									
	39%	6%	6%	11%	6%	0%	17%	33%	40%
Frequency exceeding EC_{50} or LC_{50} [3]									
	22%	0	6%	NA	NA	NA	17%	0	0

[1] Sacr. = Sacramento; Stock. = Stockton; Vaca. = Vacaville. [2] Detection frequency = # samples > 1 ng/L/total samples from all 3 plants (n=18). [3] Frequency of exceeding EC_{50} or LC_{50} for *H. azteca* (Bif =3.3 ng/L; Cyf = 1.9 ng/L; Cyp = 1.7 ng/L; L. Cyh = 2.3 ng/L; Per =21.1 ng/L and chlor = 96 ng/L).

Table 18. Summary of Pyrethroid Measurements in Biosolids from 24 California POTWs. (Source: Markle *et al.* (*55*))

Chemical	*# of Detects*	*% Detected*	*LOQ (ng/g)*	*Max. (ng/g)*	*Min. (ng/g)*	*Average[1] (ng/g)*	*Median[1] (ng/g)*
Bifenthrin	50	96%	2.5	1100	ND	150	120
Cyfluthrin	45	87%	2.5	190	ND	34	29
Lambda-Cyhalothrin	27	52%	2.5	200	ND	29	28
Cypermethrin	47	90%	2.5	1000	ND	110	79
Deltamethrin	16	31%	5.0	78	ND	28	24
Esfenvalerate	16	31%	2.5	42	ND	15	14
Fenpropathrin	3	5.8%	2.5	71	ND	12	6.8
Permethrin	48	92%	25	11000	30	1500	1200

ND = Not detected. A total of 52 influent samples were collected. [1] Median and average values were calculated assuming the limit of quantitation for non-detects.

Conclusions

As part of the Registration Review Program in USEPA, the first pyrethroid ecological risk assessments are less than two years away. Their widespread and diverse urban use patterns present many challenges in conducting a national scale ecological risk assessment. The problem formulations and public comment process has been extremely valuable in focusing on issues that need to be addressed. The Pyrethroid Working Group (PWG) has conducted a number of studies in response to the Data-Call-In (DCI) from USEPA and California Department of Pesticide Regulation (CADPR). Analysis of data from some of these studies is presented in this chapter while other studies are currently being reviewed. These data along with a wealth of information from public literature would be used in conducting ecological risk assessments for urban use pesticides.

To assess the exposure estimates from outdoor urban uses, EFED is currently using the residential and impervious scenarios in PRZM/EXAMS which only provide screening level information. To further refine these urban scenarios, results obtained from studies submitted for pathway identification, impervious surfaces washoff/runoff, turfgrass runoff and others could be used. Additionally, quality monitoring data may be used in verifying modeled EECs. Other factors that should be considered in improving these urban scenarios include characteristics of the pesticide to be modeled such as expected solubility in natural/urban drainage waters and washability from varied types of impervious surfaces. Any other significant pesticide load from sources such as ground water, drift and airborne dust contaminated with pesticides should also be considered.

The available evidence indicates that uses of conventional pesticides are resulting in relevant loadings to and from POTWs in the U.S. Information on use patterns can be used to identify those uses which are more likely to result in releases down the drain. However, POTW monitoring studies have also identified the presence of some pesticides for which the occurrence in POTW effluents is not easily explained by their labeled use patterns. Less obvious practices such as container washing, pet washing and possibly improper disposal of unwanted pesticide may be leading to pesticide loadings to POTWs. Efforts to date to model pesticide loadings to POTWs have relied on coarse, screening level models (*e.g.,* E-FAST). Information to refine critical model input parameters (*e.g.,* % removal efficiency) has been collected for some pesticides and suggest reasonable agreement between predicted and measured model parameters. The need for more comprehensive surveys of pesticides in U.S. POTW effluent is clear, as no national level survey information was identified to date. Information from such surveys in Europe (*e.g.*, Loos *et al.* (*52*)) and pesticide use pattern can provide useful information for identifying candidate pesticides for additional monitoring.

DISCLAIMER: The content of this chapter does not necessarily represent the official views of the U.S. EPA.

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UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
WASHINGTON, D.C. 20460

OFFICE OF CHEMICAL SAFETY
AND POLLUTION PREVENTION

September 30, 2016

MEMORANDUM

SUBJECT: Preliminary Comparative Environmental Fate and Ecological Risk Assessment for the Registration Review of Eight Synthetic Pyrethroids and the Pyrethrins

DP Barcode: D425791, D429461, D433338, D433339 and D435888

PC Codes: 069001, 097805, 109303, 109701, 109702, 127901, 128825, 128831, 128897, *and* 118831, 128807, 129064, 209600

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The U.S. Environmental Protection Agency has developed a pyrethroid registration review risk assessment strategy that would assess the pyrethroids as a class with regard to ecological risks, rather than conducting assessments by individual chemical. The high toxicity of pyrethroids to aquatic animals is well established. As such, risks to aquatic animals are expected to be a dominant

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[1] ERB1 to ERB6 refers to Environmental Risk Branches 1 to 6 in EFED.

concern with currently registered uses of pyrethroids; therefore, this comparative risk assessment focuses on the risks to aquatic animals (although aquatic plants were also included). This assessment also focuses on the pyrethroids for which the Pyrethroid Working Group (PWG), a consortium of registrants representing a number of pyrethroids, has conducted multiple environmental fate and ecological effects studies, and for which ample monitoring data is available. These include the synthetic pyrethroids bifenthrin, cypermethrin (cypermethrin, *zeta*-cypermethrin, *alpha*-cypermethrin), cyfluthrin (cyfluthrin, *beta*-cyfluthrin), deltamethrin, esfenvalerate, fenpropathrin, cyhalothrin (*lambda*-cyhalothrin, *gamma*-cyhalothrin), and permethrin, with the addition of the pyrethrins.

This preliminary risk assessment (PRA) is organized in five parts, including the Tier I risk assessment for bees, *and* seven attachments as follows:

Parts of the PRA

Part I. Assessing Pyrethroid Releases to POTWs of Pyrethroids and Pyrethrins (DP Barcode D425791);
Part II. Assessing Outdoor Residential, Commercial, Turf, and Nursery Uses of Pyrethroids and Pyrethrins (DP Barcode D429461);
Part III. Assessing Agricultural Uses of Pyrethroids and Pyrethrins (DP Barcode D433338);
Part IV. Assessing the Mosquito Adulticide Uses of Pyrethroids and Pyrethrins (DP Barcode D433339);
Part V. Tier 1 Risk Assessment of Agricultural Uses of Pyrethroids and Pyrethrins for Bees (DP Barcode D435888).

Attachments of the PRA

Attachment I. Structures, CAS Numbers and Nomenclature of Synthetic Pyrethroids and Pyrethrins;
Attachment II. Ecological Effects Summary Tables of Most Sensitive Endpoints for Synthetic Pyrethroids and Pyrethrins;
Attachment III. Environmental Fate Assessments for Eight Synthetic Pyrethroids and Pyrethrins;
Attachment IV. Comprehensive Aquatic Toxicity Profiles for Pyrethroids and Pyrethrins;
Attachment V. Pyrethroid and Pyrethrin Usage Data to Support the Down the Drain Assessment (Memorandum by the Biological and Economic Analysis Division);
Attachment VI. Pyrethroid and Pyrethrins Aquatic Incident Database Review;
Attachment VII. Pyrethroid and Pyrethrins Pollinator Incident Database Review.

These seven attachments are associated with DP Barcode D425791; however, they may apply to any part of the PRA.

This assessment concludes that the down-the-drain, non-agricultural outdoor, agricultural, and adulticide use patterns of synthetic pyrethroids and pyrethrins may result in multiple exceedances of acute and chronic LOCs for freshwater and estuarine/marine fish, and for freshwater and estuarine/marine invertebrates, resulting in a potential reduction in survival, growth and reproduction to non-target aquatic animals. In general, the RQs for aquatic plants were below the LOCs and are considered at less risk. For further details on the conclusions of each part of the PRA, refer to the Executive Summary of each part attached.

In addition to aquatic taxa, EFED also assessed Tier 1 risks of the pyrethroids and pyrethrins to honey bees resulting from selected agricultural uses. The Tier 1 risk assessment considers default (high end) exposures through direct contact and consumption of contaminated pollen and nectar. It is also based on effects to individual bees determined from laboratory studies. Results of the Tier 1 assessment indicate that the assessed uses of all pyrethroids and pyrethrins pose a potential risk to bees at the individual level. Additional data, such as measurement of residues in pollen and nectar and effects studies at the colony level, would be needed in order to refine the Tier 1 estimates of risk.

As an additional line of evidence, a review of the OPP Incident Data System (IDS) which now incorporates the Ecological Incident Information System (EIIS) database was conducted. The IDS and EIIS databases contained numerous aquatic and pollinator incidents for the pyrethroids and pyrethrins. The number of reports listed in the EIIS database is believed to be only a small fraction of the total incidents involving non-target organism mortality and damage caused by pesticides. These incidents appear to support the findings of the PRA.

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
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OFFICE OF CHEMICAL SAFETY
AND POLLUTION PREVENTION

Preliminary Comparative Environmental Fate and Ecological Risk Assessment for the Registration Review of Eight Synthetic Pyrethroids and Pyrethrins

Part I. Assessing Pyrethroid Releases to POTWs

Chemical Names	PC Codes
Pyrethrins	069001
Permethrin	109701
Bifenthrin	128825
Cyfluthrin	128831
Beta-cyfluthrin	118831
Lambda-cyhalothrin	128897
Gamma-cyhalothrin	128807
Cypermethrin	109702
Alpha-cypermethrin	209600
Zeta-cypermethrin	129064
Deltamethrin	097805
Esfenvalerate	109303
Fenpropathrin	127901

Date: September 30, 2016

DP Barcode: D425791

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Table of Contents

Part I. Assessing Pyrethroid Releases to POTWs

1 Executive Summary

1.1 Purpose

The preliminary risk assessment (PRA) examines the potential ecological risks associated with labeled uses of a pesticide, based on the best available scientific and commercial information on the use, environmental fate and transport, and effects of the chemical on non-target organisms. The U.S. Environmental Protection Agency (U.S. EPA, USEPA, EPA, or the Agency) has developed a pyrethroid registration review risk assessment strategy that will assess the pyrethroids as a class with regard to ecological risks, rather than conducting assessments by individual chemical. The toxicity of pyrethroids to aquatic animals is well established, and expected to drive the risk conclusions; therefore the single, streamlined assessment will focus on the risks to aquatic organisms. The assessment focuses on the pyrethroids for which the Pyrethroid Working Group (PWG), a consortium of registrants representing a number of pyrethroids, has conducted multiple studies and for which there is ample monitoring data. These include bifenthrin, cypermethrin, cyfluthrins, deltamethrin, esfenvalerate, fenpropathrin, cyhalothrins, and permethrin, with the addition of the pyrethrins. With refinements, this approach is intended to cover all of the pyrethroids currently undergoing registration review. The assessment will be organized according to the use categories of indoor down-the-drain, outdoor residential, agricultural, and wide area mosquito adulticide uses. **Part I** of the comparative assessment deals with assessing pyrethroids and pyrethrins indoor down-the-drain releases to/from POTWs.[1]

1.2 Risk Conclusions

This comparative PRA examines the potential ecological risks associated with labeled uses of the insecticides bifenthrin, cypermethrin, cyfluthrins, deltamethrin, esfenvalerate, fenpropathrin, cyhalothrins, permethrin, and the pyrethrins, based on the best available scientific and commercial information on the use, environmental fate and transport, and ecological effects of these chemicals on non-target aquatic animals. This section of the PRA deals solely on the potential exposure from indoor down-the-drain releases to, and subsequently from, wastewater treatment plants (WWTPs) and/or publicly owned treatment works (POTWs) to freshwater and estuarine/marine bodies of water.[2]

The pyrethrins and synthetic pyrethroids are highly hydrophobic compounds, showing low solubility in water. Their octanol/water partition coefficients (K_{OW}'s) for the pyrethroids are high. For example, the range of *Log* K_{OW} values across chemicals, from

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[1] Portions of this PRA are based on, or are citations from Shamim, *et al.*, 2014.

[2] Fenpropathrin, *beta*-cyfluthrin, *alpha*-cypermethrin, and tefluthrin, four of the chemicals supported by the PWG, do not have any down-the-drain uses.

Laskowski (2002) is 4.53 – 7.00; however, in a recently submitted study the range is even higher (6.40 – 7.48; MRID 49314702). Based on these properties, there would be a potential to bioconcentrate in fish tissue; however, their fish bioconcentration factors (BCFs) for the majority of these compounds are lower than would be predicted based on K_{OW} alone. It appears that these compounds undergo metabolism in fish tissue and, with certain exceptions, undergo relatively rapid depuration. The pyrethroids and pyrethrins, show high organic carbon partition coefficients K_{OC}s in laboratory experiments, suggesting a high tendency to sorb with organic carbon in soil, water and sediments and dissolved organic carbon, or particulate matter in the water environment. A comprehensive review about the physicochemical and environmental fate properties of several pyrethroids was published in 2002 (Laskowski, 2002). Since then, during the Registration Review process, new environmental fate studies were requested and submitted to the Agency. Given that the results of the environmental fate studies are not used in modeling POTW releases of pesticides (other than the optional fish bioconcentration factor (BCF)), no further fate discussion will be provided in this first part of the comparative assessment.

In order to address the issue of releases to domestic wastewater, the Office of Pesticide Programs' (OPP) Environmental Fate and Effects Division (EFED) relied on the consumer exposure model, *Exposure and Fate Assessment Screening Tool* (E-FAST 2014) that was developed for assessing industrial chemicals in EPA's Office of Pollution Prevention and Toxics (USEPA 2015b). For further information about the model, refer to **Section 5.3**.

Synthetic pyrethroids and pyrethrins can be characterized as very highly toxic to freshwater and estuarine/marine fish and invertebrates on an acute basis (*i.e.*, LC_{50} or EC_{50} are <0.1 mg/L). Freshwater fish appear to be generally more sensitive to pyrethroids and pyrethrins than estuarine/marine fish on an acute and chronic basis. In general, freshwater invertebrates appeared to be more sensitive to pyrethroids than estuarine/marine invertebrates on an acute basis (cyfluthrin is an example of an exception). Meanwhile, on a chronic basis, there is no clear trend in the comparison of freshwater and estuarine/marine invertebrates. Aquatic invertebrates (freshwater and estuarine/marine) are usually orders of magnitude more sensitive to pyrethroids and pyrethrins than fish (freshwater and estuarine/marine).

Risk determinations for the indoor down-the-drain uses of pyrethroid and pyrethrins are summarized in **Table 1**. The assessment concludes that the use of bifenthrin, cypermethrin, cyfluthrins, deltamethrin, esfenvalerate, cyhalothrins, and permethrin, plus the pyrethrins, in accordance with registered labels, results for acute and/or chronic risk Levels of Concern (LOCs) exceedances for freshwater and estuarine/marine invertebrates, from the indoor down-the-drain exposure to POTWs which in turn result in releases to bodies of water. Additionally, it results in acute risk LOC exceedances for freshwater fish for bifenthrin, *lambda*-cyhalothrin, cypermethrin and esfenvalerate, plus chronic risk LOC exceedances to freshwater fish for bifenthrin. There are no potential acute or chronic risk LOC exceedances for estuarine/marine fish from the down-the-drain uses for bifenthrin, cypermethrin, cyfluthrin, deltamethrin, esfenvalerate, *lambda*-cyhalothrin, permethrin, and the pyrethrins. Since it has no residential indoor down-the-drain uses, for fenpropathrin there is no potential risk to freshwater or estuarine/marine aquatic animals. The reader is directed to the risk characterization section for further details about risk estimation (*i.e.*, risk quotients) and risk description (**Sections 7.1** and **7.2**, respectively).

Table 1. Summary of Risk Determinations for Down-the-Drain Uses of Pyrethroids and Pyrethrins for Freshwater and Estuarine/Marine Fish and Invertebrates

Chemical	**LOC Exceedances[1]**							
	FW Fish		**E/M Fish**		**FW Inverts**		**E/M Inverts**	
	Acute	**Chronic**	**Acute**	**Chronic**	**Acute**	**Chronic**	**Acute**	**Chronic**
Bifenthrin	*X*	**X**			**X**	**X**	**X**	**X**
Cyfluthrin					*X*	**X**	**X**	**X**
Lambda-Cyhalothrin	*X*				**X**	**X**	*X*	**X**
Cypermethrin	*X*				**X**	**X**	**X**	**X**
Deltamethrin					**X**	**X**	**X**	**X**
Esfenvalerate	*X*				**X**	**X**	**X**	**X**
Fenpropathrin	n/a[2]	n/a[2]	n/a[2]	n/a[2]	n/a[2]	n/a[2]	n/a[2]	n/a[2]
Permethrin					**X**	**X**	**X**	**X**
Pyrethrins					*X*	**X**	*X*	

[1] A light shaded and italics "*X*" means it exceeds the listed species LOC. A dark shaded and bolded "**X**" represents LOC exceedances of the listed species and non-listed species LOCs.

[2] n/a = There are no applicable down-the-drain uses for fenpropathrin.

1.3 Data Gaps and Uncertainties

For details about the uncertainties in this assessment, refer to **Section 7.2.6** of the Risk Description. In short, the uncertainties lie in the following four major categories:

i. Uncertainties in the usage information, which provided high end usage based on adjustment factors. The usage information was provided by the Biological and Economic Analysis Division (BEAD). For esfenvalerate production volume was available from the registrant, and these estimates were also high end values.
ii. Uncertainties in the wastewater treatment plant removal efficiencies, which are based on EPI Suite modelling.
iii. Uncertainties related to the E-FAST 2014's Down-the-Drain (DtD) module, which assumes no degradation or partitioning on its way to the treatment plant or in the body of water. Comparison of modelled and maximum monitored concentrations from California POTWs shows similar concentrations, within an order of magnitude (with the exception of permethrin).
iv. Ecological effects data gaps and uncertainties.

2 Introduction

In the context of ecological risk assessment of conventional pesticides at USEPA, the issue of household wastewater releases of pesticides was first raised by public stakeholders from California during the Re-registration Eligibility Decision (RED) process of the pyrethroid insecticide permethrin (USEPA 2006). Concerns were raised that clothes pretreated with permethrin may cause adverse water quality impacts due to releases to POTWs when washed and result in subsequent discharges to receiving waters by POTWs. It is noteworthy that potential releases of antimicrobial pesticides to POTWs have routinely been considered in OPP environmental risk assessments due to their widespread use in consumer care products that result in substantive 'down the drain' (DtD) releases (*e.g.*, antibacterial ingredients in hand soaps). In contrast, this issue is relatively new for conventional pesticides where exposure from outdoor uses has traditionally been assumed to dominate environmental risk concerns. Monitoring data described later in this section indicates that for some pesticides, releases to (and from) POTWs may be significant to the extent that this exposure pathway requires consideration in USEPA environmental risk assessments. More recently as part of OPP's pesticide Registration Review process, the aforementioned concerns were echoed and additional concerns were identified regarding the potential for environmental exposure to pesticides resulting from their sorption onto biosolids and subsequent biosolid application to land (USEPA 2011a).

Part I of the combined ecological risk assessment summarizes the currently available information regarding pyrethroid and pyrethrins pesticide releases to POTWs in the U.S. and approaches being considered for evaluating these exposure pathways in OPP's "Preliminary Comparative Environmental Fate and Ecological Risk Assessment for the Registration Review of Eight Synthetic Pyrethroids and Pyrethrins." It also discusses sources and pesticide uses associated with releases to POTWs. Following this, a description of approaches and data that are being used to model the fate of these releases in POTWs is provided. Furthermore, a summary of available monitoring data which have been generated specifically to characterize potential pyrethroid exposure to and from POTWs is provided. Additionally, aquatic toxicological endpoints for the pyrethroids and pyrethrins assessed are discussed. In the risk estimation section of the risk characterization, the risk quotient method is used to characterize the risk of pyrethroids reaching POTW effluents. This is followed by the risk description, where data available about monitoring, and other lines-of-evidence are integrated.

It should be noted that this part of the combined Pyrethroid and Pyrethrins Ecological Risk Assessment is focused solely on so called "down-the-drain" (DtD) releases leading to POTW exposure, and only on risk to aquatic organisms, which have been found to be the most sensitive receptors in previous pyrethroid ecological risk assessments. Even though concerns related to DtD releases from POTW facilities have been issued by stakeholders mainly from the state of California, this assessment is national in scope.

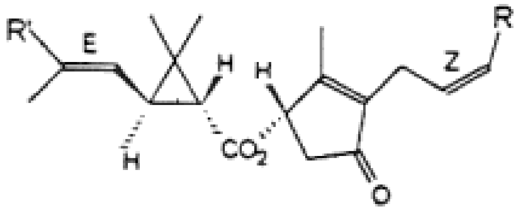
3 Problem Formulation

The Problem Formulation provides a strategic framework for the risk assessment. It sets the objectives for the risk assessment and provides a plan for analyzing the data and characterizing the risk (USEPA 1998). By identifying the important components of the risk assessment process, it focuses the assessment on the most relevant ecological receptors (species), chemical properties, exposure routes, and endpoints. The structure of this risk assessment is based on guidance contained in U.S. EPA's *Guidance for Ecological Risk Assessment* (USEPA 1998) and is consistent with procedures and methodology outlined in the Overview Document (USEPA 2004).

3.1 Pesticide Class, Type and Mode of Action

"Pyrethrum is a preparation of dried *Chrysanthemum cinerariaefolium* and/or *Chrysanthemum cineum* flower heads that contains the six insecticidally active chemicals known as pyrethrins. Each of the six naturally-occurring pyrethrins is comprised of a cyclopropane-carboxylic acid group and a cyclopentenolone (alcohol) group joined by an ester linkage… The various synthetic pyrethroid analogues are generally similar in structure to the pyrethrins, although there are some deviations from the basic chrysanthemic acid ester structure…" (Source: Spurlock and Lee, 2008).

It should be noted that the pyrethrins are natural insecticides, while other pyrethroids are synthetic compounds that mimic the pyrethrins backbone structure that is shown in **Figure 1**. The natural pyrethrins consist of two groups of chemicals: cinerin 1, jasmolin 1 and pyrethrin 1 comprise the Pyrethrins I; further, cinerin 2, jasmolin 2 and pyrethrin 2 comprise the Pyrethrins II. Consistent with past assessments, the structure of pyrethrin 1, which is one of the pyrethrins, will be used in this assessment to represent all the pyrethrins.



R = -CH_4 (for cinerin), -CH_3CH_4 (for jasmolin) or -CH=CH_2 (for pyrethrin)
R'= -CH_3 (for pyrethrins I) -CO_2CH_4 (for the pyrethins II)

Figure 1. The general structure and stereochemistry of the pyrethrins

"The first commercial photostable synthetic pyrethroid based on this approach was permethrin, synthesized in the early 1970s. Permethrin is still the most widely used synthetic pyrethroid in California today. While various photostable synthetic pyrethroids have since been developed based on different structural modifications to the basic chrysanthemate ester moiety, the halogenated vinylcyclopropylcarboxylates are among the most important in agriculture today, and include the various cypermethrins, cyfluthrins, and cyhalothrins…" (Source: Spurlock and Lee (2008))

Synthetic pyrethroids are classified as Type I and Type II, based on the *alpha*-carbon substitution. Type II pyrethroids are substituted with a cyano moiety (-C≡N) in the *alpha*-carbon (or alcohol carbon). Generally, Type II pyrethroids demonstrate increased activity, compared to Type I pyrethroids. Type I synthetic pyrethroids include for example bifenthrin and permethrin. Type II pyrethroids include cypermethrin, cyfluthrin, deltamethrin, esfenvalerate, fenpropathrin and *lambda*-cyhalothrin. Many of the pyrethroids have stereoisomers, based on the presence of asymmetric carbons, or of *cis*/*trans* isomers, that may be enriched to enhance their insecticidal activity (*e.g.*, *alpha*-cypermethrin, *zeta*-cypermethrin, *gamma*-cyhalothrin and *beta*-cyfluthrin).

Pyrethrins and synthetic pyrethroids are neurotoxic insecticides acting through direct contact and ingestion. The insecticidal effect of pyrethroids is characterized by a rapid "knock down," or paralysis, of insects. All pyrethroids act as axonic poisons, affecting both the peripheral and central nervous systems, and share similar modes of action. The primary biological effects of pyrethroids on insects and vertebrates reflect an inhibition of the correct firing of neurotransmitter delivery signals from one cell to another via nerve membrane inhibition of the voltage-gated Ca^{2+} (calcium ion) channels coupled with a stimulatory effect on the voltage-gated Na^+ (sodium ion) channels.

It is well established that severe neurological symptoms of poisoning with pyrethroids in mammals and insects are the result of modification of the Na^+ channels activity (cellular pores through which Na^+ ions are permitted to enter the axon to cause excitation) (Matsumura, 1985). Advanced electrophysiological experiments using voltage clamp and patch clamp, together with ligand-binding and ionic flux experiments, have unveiled unique actions of pyrethroids of keeping the Na^+ channel in the open state for an extremely long period, sometimes as long as several seconds. This modification of Na^+ channel properties leads to hyperactivity of the nervous system. This action leads to spontaneous depolarizations, augmented neurotransmitter secretion rate and neuromuscular block, which ultimately results in paralysis of the insect.

Pyrethroids have also been shown to suppress GABA and glutamate receptor-channel complexes and voltage-activated Ca^{2+} channels, but the toxicological significance of these actions is uncertain. As indicated above, relative to physiological responses, researchers have designated two types of pyrethroids, Type I and Type II. Usually, Type II pyrethroids demonstrate increased activity, compared to Type I pyrethroids. Type I pyrethroids action is mainly associated with compounds that cause nerve excitation symptoms typified by the appearance of repetitive firing of axons in the peripheral nervous system and a negatively correlated temperature reversible knockdown property (Clark & Matsumura, 1987).

A brief discussion about relevant environmental fate properties of the pyrethroids is provided in the Risk Conclusions section (**Section 1.2**). Further detail about the fate of these pyrethroids and pyrethrins will be provided in a subsequent part of the comparative preliminary risk assessment. The table in the **Attachment I** provides the nomenclature (common names, chemical names), CAS numbers, as well as structures of the pyrethroids and pyrethrins involved in this assessment. Please, refer to the **Attachment III** for detailed environmental fate assessments for eight individual synthetic pyrethroids which are supported by PWG plus pyrethrins.

3.2 Conclusions from Previous Down the Drain Assessments

EFED has conducted previous down the drain risk assessments for the natural pyrethrins and for the pyrethroids permethrin and deltamethrin. For permethrin and pyrethrins, the risks to estuarine/marine species was not previously assessed, since the model used for these assessments is based on freshwater receiving bodies of water of the POTW effluents (*e.g.*, rivers or ponds). Later, EFED started assessing estuarine/marine organisms, to get RQs assuming that the receiving bodies of water behave similar to freshwater bodies of water.

In 2008 EFED conducted an endangered species assessment for permethrin, for the California red legged frog and a number of other threatened or endangered species (USEPA 2008). The EECs were based on three levels of removal and three sets of RQs were calculated. There were exceedances of acute and/or chronic LOCs for freshwater fish and invertebrates. There were no LOC exceedances for non-vascular plants. Estuarine/marine species were not included in this assessment.

Additionally, in the Problem Formulation for Registration Review of Pyrethrins (December 1, 2011, DP Barcode D391619), a preliminary down-the-drain assessment was conducted, in order to assess the need for a POTW treatability study. Based on the available data and results, there were no exceedances of LOCs for freshwater fish and invertebrates. Estuarine/marine species were not included in the assessment. Based on these results, no POTW treatability study was required.

More recently in 2013, an endangered species assessment was conducted for a number of threatened and endangered species in California for deltamethrin uses in sewage systems (USEPA 2013). There were no exceedances of LOCs for both freshwater and estuarine/marine fish and for vascular and non-vascular plants. However, there were exceedances of LOCs for freshwater and estuarine/marine invertebrates.

3.3 Overview of Pesticide Uses that Lead to POTW Exposure

In response to the concerns raised by the public regarding the potential release of conventional pesticides to POTWs, OPP has reviewed indoor uses of conventional and antimicrobial pesticides and identified those that present a high potential for “down the drain” (referred to as DtD) releases. For a table or chart describing these uses, comparing EFED and AD uses is provided in the **Appendix B**. Generally, these include pesticidal treatments of fabric, clothing and carpets, pet shampoos, and drains with hydrologic connections to sewer systems. In the past, selected uses in greenhouses have been evaluated in the context of pesticide releases to both POTWs (assuming connectivity with sewer systems) and surface waters through direct discharge to bodies of water. More recently, analysis of these uses shows that they are less likely to lead to exposure to POTWs in most instances. Therefore, they are not being considered as exposure pathways of potential concern in current and forthcoming environmental risk assessments by OPP.

For pyrethroids supported by the PWG, and pyrethrins, there are a number of sources that were identified by the Pesticide Re-evaluation Division (PRD) to have a potential to end up "down-the-drain" and consequently in wastewater treatment plant influent and subsequently effluent. These are detailed in **Table 4** (**Section 5**), and they include uses such as carpet treatments, certain pet uses, and treated clothing/impregnated fabrics among others. For additional information and details about use and usage of pyrethroids, refer to **Section 5**.

3.4 Conceptual Model

For a pesticide to pose an ecological risk, it must reach ecological receptors in biologically significant concentrations. An exposure pathway is the means by which a pesticide moves in the environment from a source to an ecological receptor. For an ecological pathway to be complete, it must have a source, a release mechanism, an environmental transport medium, a point of exposure for ecological receptors, and a plausible route of exposure. The conceptual model is intended to provide a written description and visual representation of predicted relationships between pyrethroids or pyrethrins, potential routes of exposure, and the effects related to the Agency assessment endpoints. The conceptual model consists of two major components: the risk hypotheses and a conceptual diagram (USEPA 1998).

3.4.1 Risk Hypothesis

Risk hypotheses are specific assumptions about potential adverse effects (*i.e.*, changes in assessment endpoints) and may be based on theory and logic, empirical data, mathematical models, or probability models (USEPA 1998). For this assessment, the risk is stressor-initiated, where the stressor is the release of pyrethrins and pyrethroids into the environment. The following risk hypothesis is presumed for this assessment:

> *Pyrethroids and pyrethrins, when used indoors, in accordance with current labels, may result in off-site movement of the compound via wash-off into surface waters via drains and municipal wastewater treatment plants, leading to exposure of nontarget aquatic plants and animals. This potential exposure pathway may result in adverse effects upon the survival, growth, and/or reproduction of non-target aquatic animals, but not to non-target aquatic plants*.

This risk assessment will evaluate the aforementioned risk hypothesis and determine if it is supported by the risk conclusions.

3.4.2 Conceptual Diagram

The conceptual model for potential risks of pyrethroids and pyrethrins to aquatic organisms for various indoor uses of the chemical that could potentially end up in a domestic wastewater "drain" is depicted in **Figure 2**. The stressor is the chemical of concern (pyrethroids and pyrethrins). It is noted that the transport pathway is wastewater flow, the exposure media is the treatment facility that discharges into a surface water body, where it may undergo dilution. The exposure route, receptor and possible attribute changes for aquatic organisms are similar to the ones in the conceptual model for conventional agricultural applications. The receptors are aquatic plants and aquatic animals (vertebrates and invertebrates). However, based on their toxicity profile, it is believed that aquatic plants are at lower risk than aquatic animals.

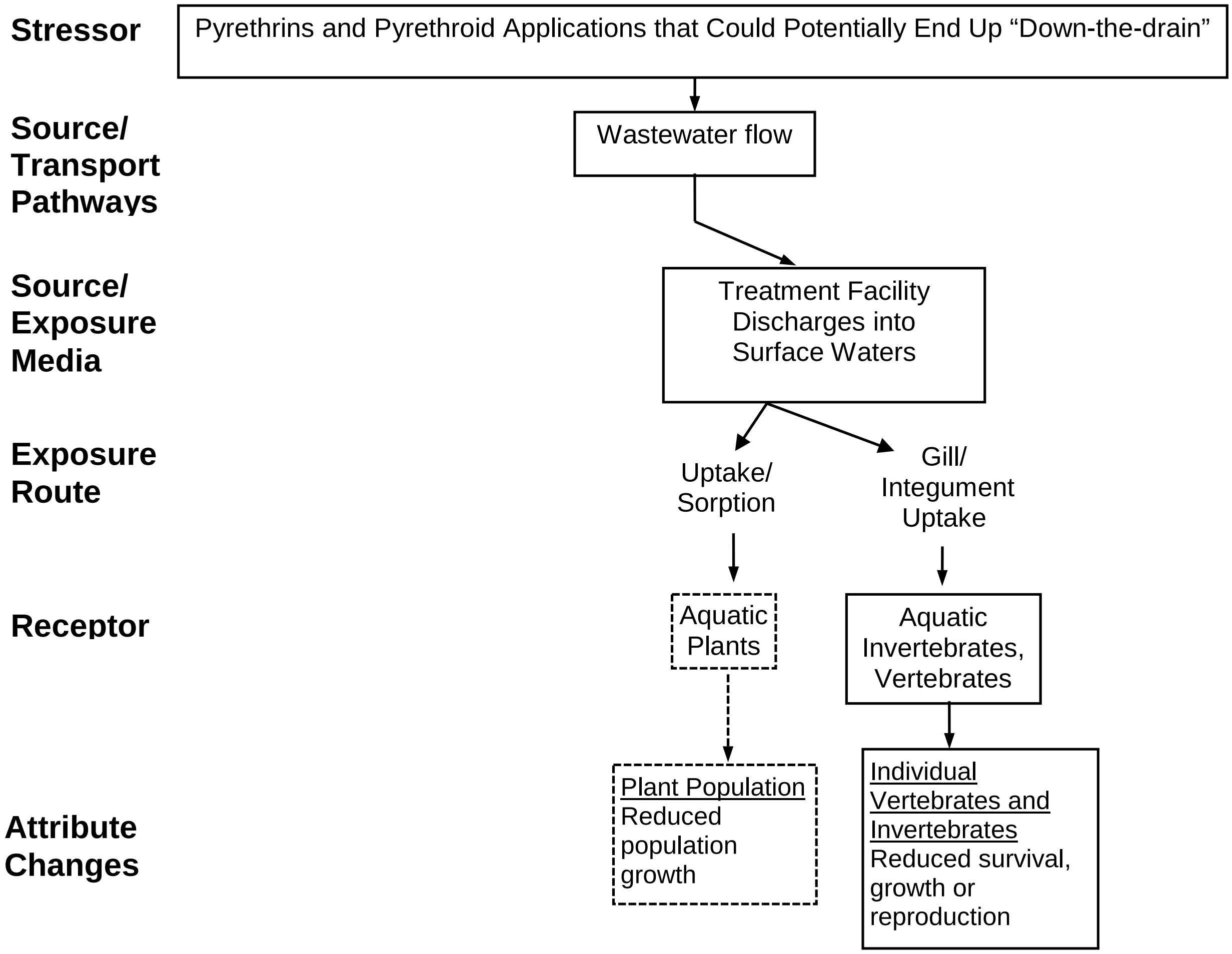


Figure 2. Conceptual diagram for potential risks of pyrethrins and pyrethroids to aquatic organisms for various indoor uses of the chemical that could potentially end up in the "drain".

4 Analysis Plan

The primary method used to assess risk in this assessment is the risk quotient (RQ) and follows closely methods outlined in the EPA Overview Document (USEPA, 2004). The RQ is the risk value for the assessment and is the result of comparing measures of exposure to measures of effect. A commonly used measure of exposure is the estimated environmental exposure concentration (EEC), and commonly used measures of effect include toxicity values such as the median lethal dose to 50% of the organisms tested (LD_{50}), median lethal concentration to 50% of tested organisms (LC_{50}), the no observed adverse effect level (NOAEL), and the no observed adverse effect concentration (NOAEC). The resulting ratio of the point estimate of exposure (EEC) and the point estimate of toxicity, *i.e.*, the RQ, is then compared to a specified level of concern (LOC), which represents a threshold for concern; if the RQ exceeds the LOC, risks concerns are triggered. Risk presumptions, along with the corresponding RQs, equations, and LOCs are summarized in **Section 7** and **Appendix B** of this assessment.

4.1 Measures of Exposure

In order to address the issue of releases to domestic wastewater, OPP has relied on the consumer exposure model, *Exposure and Fate Assessment Screening Tool* (E-FAST 2014) that was developed for assessing industrial chemicals in EPA's Office of Pollution Prevention and Toxics (USEPA 2015a).[3] The 'Down-the-Drain' module of E-FAST 2014 is specifically designed to address sources of a chemical that could potentially be disposed into domestic wastewater from a DtD application. For more information about calculations by the model, see **Section 5.3** (*Modeling Approach*). The E-FAST 2014 model estimates a pesticide concentration in surface water. A pesticide concentration in pore-water and sediment is not calculated and this assessment focuses on risk to organisms in the water-column. Besides modeling, available monitoring data on certain pyrethroids will also be used, for comparison purposes.

4.2 Measures of Effects

Measures of ecological effects are obtained from a suite of registrant-submitted guideline studies conducted with a limited number of surrogate species. The test species are not intended to be representative of the most sensitive species but rather were selected based on their ability to thrive under laboratory conditions. Measures of effect are based on deleterious changes in an organism as a result of chemical exposure. Functionally, measures of effect typically used in risk assessments include changes in survival, reproduction, or growth as determined from standard laboratory toxicity tests. The focus

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[3] The current version of E-FAST is 2014, which has the same graphical user interface than version 2.0. The user's manual is the same for both versions and it is dated 2007. It is available at the following site accessed 08/05/2016: https://www.epa.gov/tsca-screening-tools/e-fast-exposure-and-fate-assessment-screening-tool-version-2014.

on these effects for quantitative risk assessment is due to their clear relationship to higher-order ecological systems such as populations, communities, and ecosystems. Monitoring data such as adverse effect incident reports may also be used to provide supporting lines of evidence for the risk characterization. Due to the nature of ecological incident reporting, absence of incidents cannot be construed with absence of incidents. In addition, effects other than survival, reproduction, and growth may be considered, rarely are they used quantitatively to estimate risks since, in many cases, the relationship between these effects and higher-order processes is uncertain.

Laboratory-derived toxicity values include estimates of acute mortality (*e.g.*, LC50) and estimates of effects due to longer term, chronic exposures (*e.g.*, NOAEC, NOAEL). The latter can reflect changes seen in mortality, reproduction, or growth. In general, for a given assessment endpoint, the lowest (*i.e.*, most sensitive) relevant measure of effect is used in estimating the RQ. Examples of assessment endpoints and their respective measures of effect are listed in **Table 2** below.

Table 2. Summary of measures of exposure and effect for assessing the environmental risk of the proposed down-the-drain uses of pyrethroids and pyrethrins

Assessment Endpoint	Measure of Exposure	Measure of Effect
3. Survival and reproduction of individuals and communities of freshwater fish[1] and invertebrates.	Acute and chronic surface water EEC[2]	3a. Most sensitive freshwater vertebrate acute LC_{50} 3b. Most sensitive freshwater vertebrate chronic NOAEC and LOAEC 3c. Most sensitive freshwater invertebrate acute LC_{50} or EC_{50} 3d. Most sensitive invertebrate chronic NOAEC and LOAEC
4. Survival and reproduction of individuals and communities of estuarine/marine fish and invertebrates.	Acute and chronic surface water EEC[2]	4a. Most sensitive estuarine/marine vertebrate acute LC_{50} 4b. Most sensitive estuarine/marine vertebrate chronic NOAEC and LOAEC 4c. Most sensitive estuarine/marine invertebrate acute LC_{50} or EC_{50} 4d. Most sensitive estuarine/marine invertebrate chronic reproduction NOAEC and LOAEC
7. Maintenance and growth of individuals and populations of aquatic vascular and non-vascular plants	Acute surface water EEC[2]	7a. Vascular plant EC_{50} values based on yield and growth rate 7b. Nonvascular plant EC_{50} values based on cell density, growth rate, and biomass

LD_{50} = Lethal dose to 50% of the test population; NOAEC = no-observed adverse effect concentration; LOAEC = lowest-observed adverse effect concentration; LC_{50} = lethal concentration to 50% of the test population; $EC_{50/25}$ = effect concentration to 50/25% of the test population.

[1] According to EFED risk assessment guidance, freshwater fish may be used as surrogates to aquatic phase amphibians

[2] Based on stream flow SF_{1Q10} (acute) and SF_{7Q10} (chronic), 10th percentile, and stream dilution factor of 1.00 (refer to **Section 5.*C*** for further details about these stream flows and stream dilution factor).

5 Exposure Assessment

5.1 Pesticide Sources to POTWs

As outlined in **Section 3.3** (*Overview of Pesticide Uses that Lead to POTW Exposure*), OPP reviewed all indoor uses of conventional pesticides and identified those that present a high potential for “down the drain” releases. These are tabulated in the **Appendix B.** (*Identification of Pesticides of Concern Likely to End Up in POTWs*). These uses are therefore being considered as exposure pathways of potential concern in current and forthcoming environmental risk assessments by OPP.

A number of indoor pesticide uses are considered to have lower potential for substantive releases to POTWs based on labeled uses. These include labeled applications of indoor foggers, baits, crack and crevice treatment, and bed and mattress treatments where a hydrological connection to sewer systems is considered highly unlikely or at most, rare. Considerable attention has placed on the use of ‘spot on’ treatments for pets (*e.g.,* flea and tick control) as well as insecticide-impregnated collars. With spot on treatments, it is expected (and advised on some pesticide labels) that shampooing soon after application of spot on treatments would reduce the efficacy of such treatments, and those would not be cost effective and are discouraged. Regarding pet collars, the potential substantive releases to POTWs are considered low based on their expected slow release rate of pesticides from the collars.

It is important to note that the pesticide uses identified in **Appendix B** do not represent *all* potential sources of pesticide input to POTWs. Rather, they represent those uses that are currently being assessed as part of DtD modeling in OPP environmental risk assessments. For example, pesticides may potentially be released by industrial discharges to POTWs from pesticide manufacturers. However, such releases are subject to regulation under other environmental statutes and regulatory programs (*e.g.,* state and federal pretreatment programs under the authority of the Clean Water Act), and not under FIFRA. It is recognized that certain outdoor residential uses of pesticides may contribute to pesticide loadings to storm water systems which are connected to POTWs. Modeling of outdoor residential use of pesticides in OPP environmental assessments is presently focused on direct loadings to surface water. Information from the open literature suggests that some POTWs may experience greater flow during wet weather events even when direct connections to storm water inputs are not apparent (Weston and Lydy, 2010). Presumably, such inputs represent groundwater intrusion and/or fugitive inputs from storm water runoff. For these and other sources of pesticides to POTWs unaccounted for in **Appendix B**, OPP is relying on targeted monitoring data to ascertain inputs to and discharges from POTWs.

As a next step, all the indoor uses of the synthetic pyrethoids and pyrethrins were investigated, in order to distinguish which ones would lead to “down-the-drain” exposure (**Table 3**). This would allow for the identification of specific uses likely to lead to exposure through POTWs. This identification and evaluation was performed using OPPIN (an internal database of submitted data), as the first source of information, followed by inspection and confirmation with product labels.

Table 3. Universe of Indoor Uses for the Pyrethroids Supported by PWG and Pyrethrins (Includes Both Down-the-Drain Uses and Less Likely to End Up in Drains[1])

Active Ingredient	***All Indoor Uses***
Bifenthrin	Crack and crevice, bed frames, box springs, walls, floors, spot treatments, fogger, aerosol, spray (surface and space), ***pet bedding***, ***carpets***, ***drains***, greenhouse, ***bedding***
Cyfluthrin	Crack and crevice, walls, floors, spot treatments, fogger, aerosol, spray (surface and space), ***drainage systems***, ***carpet edges***
Beta-cyfluthrin	Crack and crevice, walls, floors, spot treatments, spray (surface and space)
Lambda-cyhalothrin	***Pet bedding***, ***edges of carpets***, ***upholstery***, storm drains, ***sewers***, greenhouse, ***drains***
Gamma-cyhalothrin	Crack and crevice, bed frames, box springs, mattress, walls, floors, spot treatments, aerosol, spray (surface and space), ***pet bedding***, ***edges of carpets***, ***upholstery***, storm drains, ***sewers***, greenhouse, drains
Cypermethrin	Crack and crevice , walls, floors, fogger, aerosol, spray (surface and space), ***draperies***, ***drains***, ***upholstered furniture***, ***pet bedding***, ***rugs/ carpets***, swimming pool areas
Alpha-cypermethrin	None
Zeta-cypermethrin	Crack and crevice, bed frames, box springs, mattress, walls, floors, spot treatments, fogger, aerosol, spray (surface and space), ***drain pipes/drain/plumbing installations***, ***rugs/carpets***, greenhouse
Deltamethrin	Crack and crevice, bed frames, box springs, mattress, walls, floors, pet collar, aerosol, spray (surface and space), furniture, ***drains***, ***rugs/ carpets***
Esfenvalerate	Crack and crevice, walls, floors, bed frames, box springs, mattress, fogger, spray (surface and space), ***pet bedding***, ***drain system treatments***, ***carpet treatments***, furniture, ***drains***
Fenpropathrin	None
Permethrin	Crack and crevice, bed frames, box springs, walls, floors, spot treatments, pet collar, fogger, aerosol, spray (surface and space), ***dog spray or dip***, ***treated clothing/impregnated fabric***, ***rugs/carpets***, greenhouses, ***upholstery***, ***clothing***, ***bedding pet/human***, furniture, drapes, ***sewage and sewer or drain pipe***, swimming pool areas
Pyrethrins	Crack and crevice, bed frames, box springs, walls, floors, fogger, aerosol, spray (surface and space), ***pet and human bedding***, ***upholstery/sofas/chairs***, ***clothing***, ***surface application to rugs/carpet and drapes***, ***carpet edges (bedbugs)***, ***pet shampoos/dusts/sprays***, ***sewer lines***, swimming pool areas

[1] **Bolded** and *italics* uses are believed to have a higher potential to lead to down-the-drain exposure.

For pyrethroids supported by the Pyrethroid Working Group and pyrethrins, there are a number of sources that were specifically identified by the EFED and PRD to have a potential to end up "down-the-drain" and consequently in wastewater treatment plant influent and subsequently effluent. These are detailed in **Table 4**. Note that for *beta*-cyfluthrin, *alpha*-cypermethrin and fenpropathrin no DtD uses were identified.

Table 4. Down-the-Drain Uses for the Pyrethroids Supported by PWG and Pyrethrins

Active Ingredient	***Down-the-Drain Uses***
Bifenthrin	Pet bedding, carpets, drains, bedding

Active Ingredient	*Down-the-Drain Uses*
Cyfluthrin	Drainage systems, carpet edges
Beta-cyfluthrin	None
Lambda & *gamma* cyhalothrin	Pet bedding, edges of carpets, upholstery, sewers, drains
Cypermethrin	Draperies, drains, upholstered furniture, pet bedding, rugs/carpets
Alpha-cypermethrin	None
Zeta-cypermethrin	Drain pipes/drain/plumbing installations, rugs/carpets
Deltamethrin	Drains, rugs/carpet
Esfenvalerate	Pet bedding, drain system treatments, carpet treatments, drains
Fenpropathrin	None
Permethrin	Dog spray or dip, treated clothing/impregnated fabric, rugs/carpets, upholstery, clothing, bedding pet/human, drapes, sewage and sewer or drain pipe
Pyrethrins	Pet and human bedding, upholstery/sofas/chairs, clothing, surface application to rugs/carpet and drapes, carpet edges (bedbugs), pet shampoos/dusts/sprays, sewer lines

5.2 Production Volume/Usage Information for Pyrethroids and Pyrethrins

In lieu of a production volume, which is usually provided by the registrant(s), the Biological and Economic Analysis Division (BEAD) provided usage information for several synthetic pyrethroids (**Table 5**; USEPA 2015b). In short, BEAD concluded, based on private market research data to determine the estimated amounts of pyrethroids used indoor in the U.S. by Pest Control Operators (PCOs) and consumers, that overall approximately 225,000 pounds (102,000 kilograms) active ingredient of pyrethroids and pyrethrins (combined) are estimated to be used indoors annually in the U.S., in ways that have some likelihood of resulting in DtD exposure. BEAD considered this a very high-end estimate, “since it assumes that all indoor treatments targeting flies, bedbugs, and cockroaches are equally likely to result in DtD exposure every time an application is made.” High-end values for individual active ingredients are listed in **Table 5**. Information is provided for six pyrethroids assessed in this document and pyrethrins, plus a number of other pyrethroids. For a characterization of the reported usage and adjustment factors, refer to USEPA 2015b (**Attachment V**).

Table 5. Down-the-Drain Pyrethroid/Pyrethrin Usage: Indoor Residential Use by Market Sector (Source of table: USEPA 2015b, Attachment V)

Active Ingredient (a.i.)	Amount (1,000 lb a.i.)		
	PCO	Consumer Household/ Houseplant/Pets	Total (high-end estimate only)
Bifenthrin	14-32	0.06-0.1	32
Cypermethrin	13-29	2-5	34
Permethrin	4-10	3-7	17
Deltamethrin	1-3	0.2-0.5	3.5
d-*Trans*-Allethrin	nr	0.06-0.1	0.1
Imiprothrin	nr	1-3	3
Lambda-Cyhalothrin	1-3	0.06-0.1	3
Cyfluthrin	1-2	nr	2

Active Ingredient (a.i.)	**Amount (1,000 lb a.i.)**		
	PCO	**Consumer Household/ Houseplant/Pets**	**Total (high-end estimate only)**
Prallethrin	nr	0.05-0.1	0.1
Pyrethrin	nr	89*	89
S-Bioallethrin	nr	1-2	2
Sumithrin	nr	0.01-0.03	0.03
Tetramethrin	nr	0.5-1	1
Tralomethrin	nr	0.02-0.05	0.05
Other *a.i.*s (#)	4-8	13-28	36
Total (high-end estimate only)	87	137	225

Source: Proprietary Data, 2011

PCO = pest control operator; nr = not reported

*Consumer use on pets

(#) Includes prallethrin, pyrethrin, esfenvalerate, and d-*trans* allethrin, among other unspecified pyrethroids.

Notes:

(i) The range of figures in each row represents use estimated based on sales by PCOs for bedbugs (approximately 6 %) or cockroaches (approximately 12 %). Therefore, the "high end estimates" are based on the sales for cockroach control. For additional discussion, please see the discussion of "adjustment factors" in USEPA 2015b.

(ii) Totals are rounded up to the nearest whole figure.

Individual information about esfenvalerate was not available from BEAD. In the absence of specific usage information for esfenvalerate, the registrant was consulted, and Production Volume was provided. Based on the available information, it is concluded that on average, production volume should not exceed ~6,000 kg A.I./year. Therefore, the generated EECs are considered upperbound estimates of concentrations of esfenvalerate in effluent-receiving bodies of water.

5.3 Modeling Approach

In order to address the issue of releases to domestic wastewater, OPP has relied on the consumer exposure model, *Exposure and Fate Assessment Screening Tool* (E-FAST 2014) that was developed for assessing industrial chemicals in EPA's Office of Pollution Prevention and Toxics (USEPA 2015b). The 'Down-the-Drain' module of E-FAST 2014 is specifically designed to address sources of a chemical that could potentially be disposed into domestic wastewater from a DtD application. The DtD module can be used to represent residential, domestic and certain commercial facilities (*e.g.*, supermarkets, storage facilities, and warehouses uses likely to end up in drains). This model provides first tier estimates of chemical residues in surface water that may result from household uses and the disposal of consumer products into wastewater.

Conceptually, the E-FAST 2014's DtD module assumes that in a given year the entire production volume of a chemical (*i.e.*, the amount of pesticide) is parceled out on a daily basis to the entire U.S. population and converted to a mass release per capita, and subsequently, a daily per capita release to a wastewater treatment facility (*i.e.*, g/person/day). This mass is then diluted into the average daily volume of wastewater released per person to arrive at an estimated concentration of the chemical in wastewater prior to entering a treatment facility. The underlying equations used by the DtD module are shown below. According to the E-FAST manual, the daily per capita release is defined as follows.

$$H_R = \frac{PV}{Pop} \times \frac{1000\ g}{1\ kg} \times \frac{1\ year}{365\ days}$$

Where,
a) H_R is the daily per capita release of the chemical (g/person/day);
b) *PV* is the production volume of the chemical being evaluated that is produced annually in the USA that is discharged into domestic wastewaters (kg/year); and
c) *Pop* is the 2003 U.S. resident population (2.908×10^8 persons) (U.S. Bureau of the Census, 2004-2005).

The chemical's concentration in untreated wastewater is then reduced by the fraction removed during wastewater treatment processes. The remaining chemical is discharged into surface water (*e.g.*, a river or stream), where it is assumed that it is instantaneously diluted, with no further removal. The surface water concentration is calculated using the following generalized equation.

$$SWC = \frac{H_R \times \frac{1}{Q_H} \times \left(1 - \frac{WWT}{100}\right) \times \frac{10^6\ \mu g}{g}}{SDF}$$

Where,
a) *SWC* is the surface water concentration (µg/L);
b) Q_H is the household wastewater volume released daily (it is estimated to be 388 L per person per day), it includes only domestic and residential POTWs;
c) *WWT* is the wastewater treatment removal (percent removed prior to discharging into a body of water, %); and
d) *SDF* is the stream dilution factor.

The Stream Dilution Factor (*SDF*) is the volume of the receiving stream flow divided by the volume of the wastewater released from the POTW or effluent flow (*SDF* = *SF*/*EF*). This value is estimated by the Stream Dilution Factor Program (SDFP) for 36 industrial categories. The Stream Dilution Factor Program (SDFP) estimates surface water concentrations using the 50th and 10th percentile stream dilution factors for all active POTWs, for streams to which the wastewater facilities discharge. SDFP retrieves receiving stream flow data for facilities, calculates dilution factors for each facility, ranks the flow data and stream dilution factors, and reports the results in terms of percentiles. For E-FAST 2014's DtD module, SDFP calculates these values from 9,619 POTWs for which stream flow and effluent flow information is available in the Main Facility File. The Main Facility File contains facility and basic receiving stream information for a large number of direct discharging facilities. The Main Facility File is used by three of the E-FAST 2014 modules, including the DtD and the Probabilistic Dilution Model (PDM) modules.

There are four types of stream flows that the developers of the model have deemed adequate for the protection of aquatic life and human health (acute and chronic). These are 1Q10, 7Q10, 30Q5 and Harmonic Mean. The 30Q5 and Harmonic Mean flows are used for human life exposure and are not further discussed here. Additionally, flows have been characterized to represent mid-sized stream flows and smaller stream flows. For the protection of aquatic life, the developers of the model have deemed the following stream flows (SFs) suitable:

a) SF_{1Q10} is the stream flow that corresponds to the single day of lowest flow over a 10-year period (*i.e.*, lowest 1-day flow during any 10-year period). The 1Q10 stream flows provide acute surface water concentrations to compare with acute concentrations of ecological concern or acute endpoints for aquatic animals and plants; and

b) SF_{7Q10} is the stream flow corresponding to seven consecutive days of lowest flow over a 10-year period (*i.e.*, lowest consecutive 7-day average flow during any 10-year period). The 7Q10 stream flows give chronic surface water concentrations to compare with chronic concentrations of concern or chronic endpoints, for aquatic animals.

The manual describes results provided by the model related to small stream flows and mid-sized stream flows as shown below.

a) 10th Percentile Results — exposure results are calculated based on the high-end surface water concentrations (*i.e.*, upper 10th percentile). These results represent the bounding high-end exposures. These flows are used to represent small streams.

b) 50th Percentile Results — exposure results are calculated based on the median surface water concentrations (*i.e.*, 50th percentile). These results represent central tendency exposure. These flows are used to represent mid-sized stream flows.

It should be noted that the DtD module of E-FAST 2014 is a first tier model. It does not take into account processes such as degradation prior to treatment at the facility, or after the effluent is released to the body of water, or partitioning (*i.e.*, sorption by sediment or particulate matter). As indicated earlier, the model used for these assessments assumes that the freshwater bodies of water receive the POTW effluents (*e.g.*, rivers or ponds); however, EFED calculated risk quotients for estuarine/marine species, assuming that in saltwater bodies of water the EECs are similar to the ones calculated by E-FAST's DtD module.

5.4 *Description of Model Inputs*

5.4.1 Production Volume/Usage and POTW Removal

There are two main input values in the E-FAST's DtD module: the production volume (PV), and the percent removal from wastewater treatment (WWT). The PV can be obtained from the registrant(s) sources or an estimate can be supplied by the Biological and Economic Analysis Division (BEAD). The reported usage for the synthetic pyrethroids and pyrethrins was discussed in **Section 5.2** above.

Model results are also sensitive to the WWT, which in turn is dependent on the physicochemical properties of the active ingredient of concern and the extent of wastewater treatment (*e.g.*, primary, secondary, tertiary, or ultrafiltration). An estimate of WWT is available from the Sewage Treatment Plant Fugacity Model (STPWIN™) of EPISUITE v.4.11 (USEPA 2012). This model provides estimates of the fate of organic chemicals in conventional wastewater treatment plant that uses activated sludge secondary treatment, using the default half-lives of 10,000 hours (~417 d) in the primary tank, the aeration tank, and the settling tank. This may be considered a conservative half-life value. Note that alternatively EPISUITE provides the second option to enter half-lives derived from monitoring experiments, or the third option to use model-estimated half-lives for the above mentioned processes. According to the STPWIN™ Help manual, EPI Suite's STP program was conservative (*i.e.*, leading to higher concentrations) predicting removal percent (WWT) 88% of the time using its default half-lives of 10,000 hours for 29 of 33 chemicals evaluated, for primary clarifier, aeration vessel and settling tank; however, the evaluation was based on a set chemicals which are not pesticides. A potentially more suitable and reliable alternative, is data derived from a bench scale study that may be required either during the new chemical registration process of the chemical or during Registration Review, to further refine this input parameter. Finally, for a few chemicals, when available, WWT can be obtained from actual monitoring studies of influent and effluent from POTWs. As will be seen later, there is monitoring data for pyrethroids from 31 POTW facilities in California (Markle *et al.*, 2014).

Table 6 provides a summary of removals by various mechanisms for eight pyrethroid insecticides predicted by STPWIN™. As shown in the table, the module predicts that for all these chemicals, the main removal mechanism is sludge sorption. The total biodegradation is low while the release to air is minimal. For all the chemicals, the removal is above 90%.

Table 6. Removal Percent of Eight Pyrethroids in Wastewater Treatment Plants Obtained from EPISUITE v.4.11 and Its STPWIN Module[1]

Process	**Bifent.**	**Fenprop.**	**Cyhalot.**	**Permet.**	**Cyflut.**	**Cypermet**	**Esfenval.**	**Deltamet.**	**Pyrets.[2]**
Sludge sorption	93.2	91.4	93.1	92.7	91.2	93.0	92.1	92.1	90.9
Total Biodegradation	0.78	0.77	0.78	0.78	0.77	0.78	0.77	0.77	0.77
Total to Air	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Total Removal	94.0	92.2	93.8	93.4	91.9	93.8	92.9	92.8	91.7

[1] All results were rounded to two decimal places or three significant figures.

[2] Based on results for pyrethrin 1, which has been designated as the representative pyrethrin in the available

environmental fate studies.

5.4.2 Bench Scale Study

Based on experience with DtD modeling with pyrethroids, OPP requested additional data from registrants to improve modeling of the fate and removal efficiency of certain pyrethroids in POTWs. In response, PWG submitted a bench scale study simulating four processes that occur in POTWs: primary settling, aerobic biological treatment, anaerobic digestion, and ultra-filtration (Cleary and McGrath, 2012, MRID 48762906)[4]. Pyrethroids studied in these processes included: permethrin, deltamethrin, bifenthrin, cyfluthrin, *lambda*-cyhalothrin, cypermethrin, esfenvalerate, and fenpropathrin. Although in treatment plants, they occur simultaneously, these processes were evaluated separately from each other (*i.e.*, they were treated as modules). This treatability study is useful in understanding the relative contributions of the different processes that occur at a treatment facility. Removal processes included primary settling, which showed very limited removal (this was an unexpected result), and aerobic and anaerobic digestion, which show moderate levels of removal. Only ultrafiltration appeared to remove over 90% of the material in the bench scale study. Results from modelling (using EPISUITE v.4.11 giving total removal) and monitoring data (discussed below) indicate levels of removal of above 90%. However, direct comparison of the bench scale study results to modelling and monitoring data is confounded by the fact that the bench scale study design does not enable determination of an overall removal efficiency based on the sum total of the simulated treatment processes. Therefore, the utility of the bench scale study mostly relates to how separate processes affect pyrethroid removal and not for an estimate of the overall removal efficiency of pyrethroids from POTWs. The study was considered supplemental, providing limited information about the individual processes, and their relative contributions in a treatment plant.

5.5 Summary of Model Outputs

In this assessment, the removal percentages were obtained from the model EPISUITE v.4.11, as shown in **Table 6**. The usage information was supplied by BEAD (see **Table 5**, and USEPA 2015b). The usage data from **Table 5** was converted from pounds active ingredient/year to kilograms active ingredient/year. The input values are summarized in **Table 7**.

Model outputs from the E-FAST 2014 model are provided in **Table 7**. These include the acute and chronic EECs (µg/L). Both the peak and chronic EECs are the same, since they represent stream dilution factors of 1.0 at the 10th percentile (*i.e.*, no dilution or effluent dominated body of water, and small receiving bodies of water or small streams). It was noted that for bifenthrin, the output acute and chronic EECs exceeded the limit of

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[4] A bench scale study is a study conducted in a small scale basis, to mimic the processes that occur in a POTW plant. In the pyrethroids bench scale study, four processes were simulated. A number of issues were observed in the study, and its results were not used for the modeling in E-FAST 2014.

solubility by around 50%, which is by itself extremely low (s = 0.014 µg/L). Given that the wastewater treatment removals (WWTs) were of a similar level for all the chemicals, then the EECs appear to be very dependent on the reported usage or PV. The highest predicted EECs were for pyrethrins, followed by cypermethrin, with respective reported usages of 196,000 and 75,000 kg a.i./year. For an example output file from E-FAST 2014 for permethrin, refer to **Appendix C**.

Table 7. Estimated Environmental Concentrations for Eight Pyrethroids and Pyrethrins

Chemical	**Reported Usage (kg/year)**	**Wastewater Removal / (%)**	**Peak EEC = Chronic EEC[1] (µg/L)**
Bifenthrin	14,500	94.0	0.014 – 0.0211 [2]
Cyfluthrin	910	91.9	0.00179
Lambda-Cyhalothrin	1,360	93.8	0.00205
Cypermethrin	15,400	93.8	0.0232
Deltamethrin	1,590	92.8	0.00278
Esfenvalerate	6,000	92.9	0.0103
Fenpropathrin	N/A	92.2	N/A
Permethrin	7,710	93.4	0.0124
Pyrethrins	40,400	91.7	0.0814

N/A = does not apply, no applicable down-the-drain uses for fenpropathrin.

NA = not available.

1 Stream flow SF_{1Q10} (acute) and SF_{7Q10} (chronic) stream dilution factor = 1.

2 Exceeded the limit of solubility for bifenthrin (0.014 µg/L). The actual concentration predicted by E-FAST 2014 was 0.0211 µg/L.

5.6 POTW Monitoring Data

The available information on the occurrence of pesticides in U.S. POTW influent, effluent and biosolids was reviewed and is summarized here with a focus on the following questions:

> Which pesticides are most commonly detected in POTWs and how does this relate to their intended uses?
>
> What is the removal efficiency of pesticides by wastewater treatment processes and how does this compare to estimates based on modeling?

Although a number of country-wide surveys of pesticides and other micropollutants in POTW wastewater have been conducted in Europe (*e.g.*, Loos *et al.*, 2013; Luo *et al.*, 2014), an analogous U.S. wide survey was not identified in this review. Instead, a number of state-wide and POTW-specific surveys were identified, some of which are mainly about pyrethroids and are summarized below.

5.6.1 Pyrethroids in POTW Influent and Effluent

5.6.2 California POTWs

In a comprehensive state-wide survey, Markle *et al.* (2014) sampled 31 POTWs in California for the presence of eight pyrethroids in influent, effluent, and biosolids. This effort was conducted by the PWG, in response to pyrethroid re-evaluation activities by both the California Department of Pesticide Regulation and the USEPA. The POTWs they surveyed represent approximately 40% of the treated municipal wastewater in California and include primary, secondary and tertiary treatment as terminal wastewater treatment processes. Samples were taken from January through March, 2013 during dry weather period. Consecutive grab samples were taken from influent, effluent and biosolids (when available) and did not account for hydrologic retention time between entry to the POTW and discharge. Extensive quality control measures were instituted including separate analytical measurement by two laboratories.

Results indicate high detection frequencies (*e.g.*, 43% to 100%) for 7 of the 8 pyrethroids sampled in POTW influent (**Table 8**). Frequencies of detection exceeded 80% for bifenthrin, cyfluthrin, *lambda*-cyhalothrin, cypermethrin and permethrin. Fenpropathrin was the least detected pyrethroid in influent at 4.5% (3 of 67 samples), and is the only pyrethroid sampled that is not registered for residential uses in California. This suggests residential uses of these products are contributing to their loadings to California POTWs. By far the highest maximum and median influent concentrations reported are for permethrin (3,800 and 230 ng/L, respectively), which may be related to its topical use to treat lice infestations (a pharmaceutical use that is regulated by the Food and Drug Administration (FDA)).

Table 8. Summary of pyrethroid measurements in influent from 31 California POTWs (Source: Markle *et al.*, 2014)

Chemical	# of Detects	% Detected	LOQ (ng/L)	Max. (ng/L)	Min. (ng/L)	Average[1] (ng/L)	Median[1] (ng/L)
Bifenthrin	64	96%	5	74	ND	15	9.7
Cyfluthrin	59	88%	5	55	ND	11	7.4
Lambda-Cyhalothrin	54	81%	5	72	ND	5.6	2.8
Cypermethrin	54	81%	5	200	ND	35	21
Deltamethrin	29	43%	10	210	ND	8	3.3
Esfenvalerate	31	46%	5	360	ND	8.1	1.7
Fenpropathrin	3	4.5%	5	130	ND	4.6	1.7
Permethrin	67	100%	50	3800	30	330	230

ND = Not detected. Units of ng/L = parts per trillion.

A total of 67 influent samples were collected (62 samples + 5 repeats).

[1] Median and average values were calculated assuming the limit of quantitation for non-detects.

In POTW effluent, the greatest detection frequencies are observed for bifenthrin (82%), followed by cypermethrin (81%), permethrin (65%), cyfluthrin (60%), *lambda*-cyhalothrin (48%) and esfenvalerate (32%; **Table 9**). Comparatively, the rates of detection for deltamethrin and fenpropathrin are much lower (16% and 3%, respectively) in effluent than influent. Consistent with the influent sampling results, the greatest maximum and median concentrations in POTW effluent are observed for permethrin (170 and 9.4 ng/L, respectively). Cypermethrin showed the next highest effluent concentrations with maximum and median values of 13 and 1.3 ng/L, respectively. Maximum and median concentrations for the other six pyrethroid are 2 and 1 orders of magnitude below that for permethrin.

Table 9. Summary of pyrethroid measurements in effluent from 31 California POTWs (Source: Markle *et al.*, 2014)

Chemical	# of Detects	% Detected	LOQ (ng/L)	Max. (ng/L)	Min. (ng/L)	Average[1] (ng/L)	Median[1] (ng/L)
Bifenthrin	51	82%	0.5	3.9	ND	0.89	0.6
Cyfluthrin	37	60%	0.5	4	ND	0.6	0.3
Lambda-Cyhalothrin	30	48%	0.5	1.6	ND	0.3	0.2
Cypermethrin	50	81%	0.5	13	ND	2.11	1.3
Deltamethrin	10	16%	1.0	1.2	ND	0.31	0.3
Esfenvalerate	20	32%	0.5	0.6	ND	0.25	0.2
Fenpropathrin	2	3.2%	0.5	0.8	ND	0.22	0.2
Permethrin	40	65%	5.0	170	ND	20	9.4

ND = Not detected. Units of ng/L = parts per trillion.

A total of 67 effluent samples were collected.

[1] Median and average values were calculated assuming the limit of quantitation for non-detects.

It is also of interest to evaluate the removal of pyrethroids by POTW treatment, since this information can help inform modeling approaches for estimating pyrethroid loadings from POTWs. Influent and effluent data from Markle *et al.* (2014) were used to calculate percent removal of pyrethroids using the following equation:

$$\%\ \text{Removal} = \left(1 - \frac{\text{Effluent Concentration}}{\text{Influent Concentration}}\right) \times (100\%)$$

When the effluent concentration was reported below limits of quantitation (LOQ), the concentration was equated to the LOQ. When the influent was reported to be below the LOQ, no calculation was made. On average, pyrethroid concentrations measured in POTW effluent are approximately 10% those measured in influent, representing a reduction of approximately 90% (**Figure 3**). The higher mean % removal indicated for esfenvalerate (97%) and fenpropathrin (99%) are based on very few samples and are therefore considered highly uncertain.

In terms of POTW-specific factors affecting pyrethroid concentrations, there was typically a large reduction in pyrethroid concentrations in effluent from primary to secondary treatment, although only one plant sampled had primary treatment as its terminal treatment process. The relationship between secondary and tertiary treatment was less clear, whereby some POTWs containing secondary treatment had higher concentrations in effluent compared to those with tertiary treatment and vice versa.

It is noted that the study by Markle *et al*. (2014) was not specifically designed to estimate % removal efficiency of pyrethroids because samples were taken concurrently from influent and effluent without regard to the retention time of treated water in the POTW. Therefore, differences between concentrations of pyrethroid in influent and effluent may reflect, not only partitioning and degradation processes associated with wastewater treatment, but also variation in pesticide loadings over time. Nonetheless, average % removal efficiencies based on the monitoring data (90-99%) are quite similar to those calculated using the STPWIN™ model summarized in **Table 6** (91-94%).

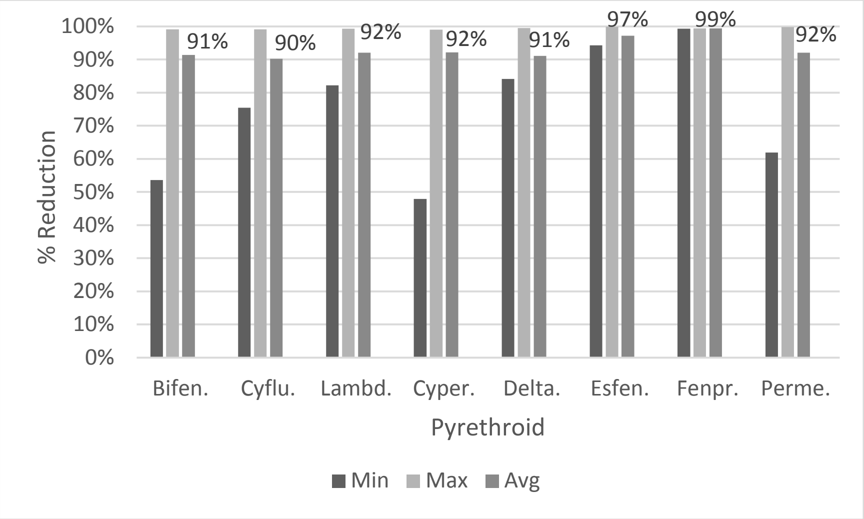

Figure 3. Percent reduction in pyrethroid concentrations in POTW effluent relative to influent (Source: Markle *et al.*, 2014)

5.6.3 Sacramento POTW

In contrast to the previous study which conducted limited sampling of POTW wastewater across many facilities, Weston *et al.* (2013) focused their efforts on a single facility, the Sacramento Regional County Sanitation District Treatment Plant. Concentrations of eight pyrethroids in influent and effluent were sampled over multiple time periods from November 2010 to January 2012. Twelve, 24-h composite samples were taken monthly from influent and seven 24-h, flow-weighted composite samples were taken from effluent (4 during rain events and 3 during dry events). Importantly, the timing of effluent samples was adjusted to account for the retention time of the wastewater in the plant. This facilitates more accurate estimation of % removal efficiency compared to the previous study by Markle *et al.* (2014). Weston *et al.* (2013) also sampled three POTW wastewater interceptors during the course of this study, one of which (City interceptor) received both municipal sewage and storm water runoff while the other two (Folsom and Laguna interceptors) received only municipal sewage.

Results from this study indicate that four pyrethroids were detected in all (100%) of the 12 monthly POTW influent samples (permethrin, bifenthrin cypermethrin, and *lambda*-cyhalothrin). Among these, permethrin was the dominant pyrethroid detected in terms of overall concentration and typically ranged between 200 and 400 ng/L (ppt). Cypermethrin and bifenthrin were generally found between 20 and 40 ng/L in influent while cyhalothrins were found up to 30 ng/L. Cyfluthrin was detected once in influent during the study, while deltamethrin, fenpropathrin and esfenvalerate were not detected in any of the 12 influent samples. Attempts to correlate temporal peaks in influent concentrations with known use pattern or sales data were not successful. Analysis of pyrethroids in the wastewater interceptor upstream of the treatment plant suggest that storm water runoff was not the dominant source of pyrethroids to the plant. Concentrations of permethrin in the City interceptor (receiving stormwater) were slightly lower than those which did not receive stormwater. Furthermore, all the interceptors sampled contained substantially lower concentrations of permethrin than what was found in the POTW influent, suggesting that other sources of permethrin to the plant are likely. The other pyrethroids were found in similar concentrations in the three interceptors compared to POTW influent. The authors speculate that indoor uses of pyrethroids, container washing and possibly improper disposal of unwanted pesticide may be leading to the loadings to the Sacramento POTW.

In terms of effluent quality, permethrin was again the dominant pyrethroid detected in all but one of the seven effluent samples, ranging from 12-45 ng/L. Bifenthrin and cyhalothrins ranged from 1-5 ng/L in effluent and were detected 43% and 29% of the time, respectively. Concentrations of permethrin, bifenthrin and cyhalothrin were up to 2 times the respective 96-h EC_{50} values reported for the freshwater amphipod, *Hyalella azteca*. However, attempts to correlate observed toxicity to *H. azteca* in effluent samples with toxic units or TIE procedure were not definitive in terms of the cause of toxicity. Removal efficiencies of the pyrethroids from the POTW influent generally ranged from 90-95%, which is similar to the findings by Markle *et al.* (2014) in their California-wide POTW survey.

5.6.4 Weston and Lydy (2010)

In this study, Weston and Lydy sampled three California POTWs (Sacramento, Stockton, and Vacaville) for the presence of 8 pyrethroids and chlorpyrifos during three dry and three wet seasons in 2008 and 2009. The authors indicate that except for a small portion of the Sacramento POTW influent, all plants contained sanitary sewer systems that were separate from stormwater systems. They further note that the Stockton POTW included tertiary treatment via routing secondary treated wastewater through 240 ha of treatment ponds which yielded a retention time of about 30 days. Results above 1 ng/L are considered by the authors to be reliable. A total of 18 POTW samples were taken. Other samples of agricultural drains and urban runoff were also analyzed but are not relevant for this discussion here.

Weston and Lydy (2010) report that of all the samples, quantifiable concentrations of one or more pyrethroids were found in 67% of the samples taken. Across all three facilities, chlorpyrifos (40%), bifenthrin (39%) and permethrin (33%) were most commonly detected (**Table 10**). Generally, the highest concentrations of pyrethroids and chlorpyrifos are seen with the Sacramento POTW. In terms of toxicological relevance, 22% of the effluent samples containing bifenthrin, 17% containing *lambda*-cyhalothrin, and 6% of the samples containing cypermethrin exceeded the respective EC_{50} or LC_{50} values for *H. azteca*. The authors note that the presence of pyrethroids is surprising especially given the low levels of suspended solids in the effluent (< 8 mg/L). They suggest that sewer disposal of household pesticides, use of pet and lice control shampoos and laundering of permethrin-treated clothing may be potential sources of pyrethroids to the POTWs. Despite 25-50% greater flows in wet weather, Weston and Lydy (2010) report similar concentrations in effluent during dry and wet weather flows, which indicates that pesticide loadings from urban/residential runoff may be contributing to loadings to POTWs.

Table 10. Pyrethroids and chlorpyrifos in effluent from three California POTWs (Source: Weston and Lydy, 2010)

POTW[1]	Bifen.	Cyfl.	Cyp.	Delt.	Esfen.	Fenp.	*L.* Cyh.	Perm.	Chlor.
Maximum Concentration Detected (ng/L)									
Sacr.	2.7	1.7	17.0	0	3.7	0	5.5	17.2	24.1
Stock.	4.8	0	0	1.3	0	0	0	7.9	5.5
Vaca.	6.3	0	0	2.7	0	0	2.8	7.6	0
Overall Detection Frequency (n=18) [2]									
	39%	6%	6%	11%	6%	0%	17%	33%	40%
Frequency exceeding EC_{50} or LC_{50} [3]									
	22%	0	6%	NA	NA	NA	17%	0	0

[1] Sacr. = Sacramento; Stock. = Stockton; Vaca. = Vacaville.

[2] Detection frequency = # samples > 1 ng/L / total samples from all 3 plants (n=18).

[3] Frequency of exceeding EC_{50} or LC_{50} for *H. azteca* (Bif =3.3 ng/L; Cyf = 1.9 ng/L; Cyp = 1.7 ng/L; L. Cyh = 2.3 ng/L; Per = 21.1 ng/L and Chlor = 96 ng/L).

5.6.5 Suffern, NY POTW

In a study submitted to the Agency (MRID 48072901), the software TOXCHEM+ was used to estimate the fate of eight pyrethroids in the Village of Suffern Sewage Treatment Plant (Suffern, NY). The software is used to predict the behavior of a chemical in a treatment plant. Since the software's approach is not relevant to this assessment, it will not be described here. Nonetheless, in the study, sampling of the POTW was conducted and results are reported below. **Table 11** summarizes the measured and predicted concentrations in filtered and total effluent waters, and sludge cake. Since the measured filtered effluent waters were reported as <10 ng/L, all the predicted values were within this range. The measured and predicted total effluent water concentrations were within the same order of magnitude of each other.

Table 11. Measured and Predicted Concentrations in Effluent (Filtered & Total) and Sludge

Chemical\Conc. (ng/L)	Measured Filtered Effluent	Predicted Filtered Effluent	Measured Total Effluent	Predicted Total Effluent	Measured Sludge Cake	Predicted Sludge Cake
Bifenthrin	<10	0.371	<10	0.628	103100	40580
Cyfluthrin	<10	1.50	1.88	2.09	55830	43320
Cypermethrin	<10	2.76	9.5	5.08	249100	286600
Deltamethrin[1]						

Chemical\Conc. (ng/L)	**Measured Filtered Effluent**	**Predicted Filtered Effluent**	**Measured Total Effluent**	**Predicted Total Effluent**	**Measured Sludge Cake**	**Predicted Sludge Cake**
Esfenvalerate	<10	3.0	<10	10.5	6560	5940
Fenpropathrin	<10	0.137	<10	0.193	4310	6280
Lambda-cyhalothrin	<10	0.23	0.54	0.58	10030	7010
Permethrin	<10	4.7	<10	6.9	567000	555000

[1] For deltamethrin the concentrations were too low.

5.6.6 Sacramento, CA POTW

In another similar study (MRID 48857505), the software TOXCHEM+ was also used to estimate the fate of eight pyrethroids in the Sacramento Regional County Sanitation District (SRCSD) (Elk Grove, California) treatment plant. In SRCSD plant, 23 locations were sampled, including 18 wastewater and five biosolids samples in three sampling events in three different days. Analyses for eight pyrethroids was conducted. **Table 12** summarizes the ranges in measured and predicted concentrations in influent, dechlorinated final effluent, primary sludge, and secondary sludge. These results are taken to be representative of overall results (of measured *vs.* predicted concentrations), since several other measured and predicted concentrations were reported in this study. Cypermethrin and permethrin had the highest measured influent concentrations, and the highest measured and predicted effluent and sludge concentrations.

Table 12. Ranges in Measured and Predicted Concentrations in Final Effluent, Primary Sludge and Secondary Sludge

Chemical	**Measured Plant Influent**	**Measured Dechlorinated Final Effluent**	**Predicted Dechlorinated Final Effluent**	**Measured Primary Sludge**	**Predicted Primary Sludge**	**Measured Mixed Sludge**	**Predicted Mixed Sludge**
	ng/L	**ng/L**	**ng/L**	**ng/mL**	**ng/mL**	**ng/mL**	**ng/mL**
Bifenthrin	18.8-19.7	1.49-1.81	1.09-2.26	<2.5-2.77	3.59-5.78	<2.5-2.61	1.39-2.43
Cyfluthrin	6.95-9.55	0.556-0.608	0.32-0.69	<2.5	1.63-2.52	<2.5	0.51-0.81
Cypermethrin	46.1-48.6	3.81-4.68	1.72-2.72	4.51-6.41	7.95-13.0	4.60-5.92	2.53-4.05
Deltamethrin[1]							
Esfenvalerate[1]							
Fenpropathrin[1]							
Lambda-cyhalothrin	10.6-16.5	1.05-1.47	0.77-1.77	<2.5	2.10-4.37	<2.5	0.77-1.66
Permethrin	352-368	24.7-26.2	10.2-20.1	36.5-50.0	65.4-103	31.8-34.9	21.3-32.0

[1] An insufficient number of detections of deltamethrin, esfenvalerate and fenpropathrin did not allow for calibration of the model.

5.6.7 Comparison of the Suffern, NY and Sacramento, CA POTW Facilities

Discrepancies were observed in the predicted distribution of pyrethroids in various compartments between the NY study and the study involving a plant in CA. For example, for bifenthrin the model predicts 24% is emitted through volatilization in NY, compared to

1.8% in CA. For the same chemical, the model predicts that 1.6% is in the effluent in NY, compared to 10% in the CA study. It is acknowledged, however, that there are differences in treatment plant design. In addition, more usage may be registered in one region of the U.S. than in others. Compared to the CA study, the NY study was more limited in scope with fewer samples monitored and analyzed.

5.7 Comparison of Monitoring and Modeling Results

It is instructive to compare the results of POTW monitoring to that predicted by down-the-drain modeling (DtD) using E-FAST described earlier, as a way of evaluating model predictions. A comparison of EECs predicted by E-FAST's Down-the-Drain module is presented in **Table 13** along with the monitored concentrations in effluent summarized in **Table 9**. With bifenthrin, the predicted concentrations in POTW effluent is 14 ng/L (*i.e.*, the limit of solubility) to 21.1 ng/L (*i.e.*, the actual model-predicted concentration), which is more than an order of magnitude above the median (0.6 ng/L) concentration measured in California POTWs by Markle *et al.* (2014). However, it is about an order of magnitude above the maximum concentration detected in California POTW effluent (3.9 ng/L). Similarly, for *lambda*-cyhalothrin, cypermethrin, and deltamethrin, the predicted concentration was above the maximum concentration detected in California POTWs. For cyfluthrin, both the predicted and the maximum concentration were of the same order of magnitude, although the predicted concentration was slightly below the maximum concentration detected in California POTWs. For permethrin, the predicted concentration was around an order of magnitude below the maximum concentration detected in California POTWs, which may be due to the additional permethrin uses for lice control, which are considered drug uses, and are not considered in this assessment. These products are regulated by FDA, but if the active ingredient in these products is also a pesticidal active ingredient, the risk assessor should consider the pesticidal uses in the ecological risk assessment. Another possible reason may be that permethrin usage in California may be higher than elsewhere in the United States. Nonetheless, for both cyfluthrin and permethrin, the predicted concentration is above the median measured concentration.

Table 13. Estimated Environmental Exposure Concentrations of Pyrethroids and Pyrethrins, from POTW Discharges, Compared to Monitored Concentrations (Markle *et al.*, 2014)

Chemical	Reported Usage (kg)	WWT (% Removal)	Median Measured Conc. (ng/L)[1]	Maximum Measured Conc. (ng/L)	Predicted Conc. (ng/L)	<1 Order of Magnitude Difference?
Bifenthrin	14,500	94.0	0.6	3.9	14 – 21.1[2]	Yes
Cyfluthrin	910	91.9	0.3	4	1.79	Yes
Lambda-Cyhalothrin	1,360	93.8	0.2	1.6	2.05	Yes
Cypermethrin	15,400	93.8	1.3	13	23.2	Yes
Deltamethrin	1,590	92.8	0.3	1.2	2.78	Yes
Esfenvalerate	6,000	92.9	0.2	0.6	10.3	Yes
Fenpropathrin	N/A	92.2	0.2	0.8	N/A	N/A
Permethrin	7,710	93.4	9.4	170	12.4	No[3]
Pyrethrins	40,400	91.7	NA	NA	81.4	NA

WWT = Percent removal from wastewater treatment; N/A = not applicable (fenpropathrin does not have applicable indoor down-the-drain uses).

[1] Median values were calculated assuming the limit of quantitation for non-detects.

[2] The actual output concentration in E-FAST 2014 was 21.1 ng/L. This value exceeded the limit of solubility for bifenthrin (14 ng/L).

[3] Uses for lice control, which are considered pharmaceutical uses, and possible higher usage of this chemical in California than elsewhere in the nation, may be the reasons why the maximum monitored concentration is above the predicted EEC for permethrin.

5.8 Pesticides in POTW Biosolids

Section 405(d) of the Clean Water Act (CWA) requires the USEPA to identify and regulate toxic pollutants that may be present in biosolids (sewage sludge) at levels of concern for public health and the environment. Historically, the focus of identification and regulatory efforts has been on industrial chemicals, pharmaceuticals, metals, and selected antimicrobial chemicals (USEPA 2009). However, recent studies have raised attention on the occurrence of conventional pesticides in biosolids, which often are treated and applied to land. Potential consequences of land-applied biosolids that contain appreciable amounts of pesticides include alteration of soil and terrestrial biota, runoff to surface waters and contamination of ground water.

In addition to quantifying pyrethroid concentrations in POTW influent and effluent, the previously summarized study conducted by Markle *et al.* (2014) also measured pyrethroids in biosolids from 24 of the POTWs included in the survey (**Table 14**). In terms of overall detection frequency, results mirror those described previously for influent and effluent, with the highest detection frequencies reported for bifenthrin (96%), permethrin (92%), cypermethrin (90%), and cyfluthrin (87%). The maximum concentration of permethrin (11,000 ng/g d.w.) [or 11,000 µg/kg d.w.] is 10X that of the pyrethroids with the next highest maximum concentrations (bifenthrin, cypermethrin). Median concentrations are greatest for permethrin (1,200 ng/g d.w.), bifenthrin (120 ng/g d.w.) and cypermethrin (79 ng/g d.w.). Permethrin was also reported in sewage sludge from the U.K. (Rogers *et al.*, 1989) and Switzerland (Plagellat *et al.*, 2004).

Table 14. Summary of pyrethroid measurements in biosolids from 24 California POTWs (Source: Markle *et al.*, 2014)

Chemical	# of Detects	% Detected	LOQ (ng/g)	Max. (ng/g)	Min. (ng/g)	Average[1] (ng/g)	Median[1] (ng/g)
Bifenthrin	50	96%	2.5	1100	ND	150	120
Cyfluthrin	45	87%	2.5	190	ND	34	29
Lambda-Cyhalothrin	27	52%	2.5	200	ND	29	28
Cypermethrin	47	90%	2.5	1000	ND	110	79
Deltamethrin	16	31%	5.0	78	ND	28	24
Esfenvalerate	16	31%	2.5	42	ND	15	14
Fenpropathrin	3	5.8%	2.5	71	ND	12	6.8
Permethrin	48	92%	25	11000	30	1500	1200

ND = Not detected. A total of 52 biosolid samples were collected.

[1] Median and average values were calculated assuming the limit of quantitation for non-detects.

As a consequence of these and other reports of conventional pesticides in POTW biosolids, OPP has undertaken efforts along with counterparts in the Office of Water to develop approaches to screen uses of conventional pesticides for their potential to end up and persist in biosolids. The initial efforts focused on identifying pesticide uses with the greatest potential for releases down the drain (**Table B1** in the **Appendix B**). Subsequently, efforts have focused on developing models for evaluating the potential risks associated with pesticides in land-applied biosolids. One approach being evaluated is adapting the current Office of Water Biosolids Core Risk Assessment Model (BCRAM) for a first tier assessment. Other approaches being investigated include adapting existing OPP models (*e.g.*, PRZM) and exposure scenarios for evaluation of land applied biosolids.

6 Aquatic Receptors and Ecological Effects Characterization

In first tier ecological risk assessments, effects characterization describes the types of effects a pesticide can produce in an aquatic (or terrestrial) organism. This characterization is based chiefly on registrant-submitted studies that describe acute and chronic effects toxicity information for various aquatic and terrestrial animals and plants. Data from the open literature are also considered; in this case, when updated ECOTOX queries were available, papers were screened to see if data from any of those studies were more sensitive than available data from submitted studies and, if so, reviewed according to approved procedures (USEPA, 2011b). Toxicity testing reported in this section does not represent all species of aquatic organisms. Only a few surrogate species for freshwater fish are used to represent all freshwater fish (2000+) species in the United States, although the most sensitive useable endpoints available are used in risk calculation. Estuarine/marine testing is usually limited to a crustacean, a mollusk, and a fish. Also, amphibians are not usually tested. The risk assessment assumes that freshwater fish serve as a surrogate for aquatic-phase amphibians.

Acute and chronic laboratory studies with pyrethroids and pyrethrins on aquatic organisms provide information regarding survival, growth, and reproduction. Data on the technical grade active ingredient (TGAI) and the effects on freshwater and estuarine/marine fish and invertebrates, aquatic vascular and nonvascular plants (algae), and freshwater and estuarine/marine sediment dwelling organisms are used to evaluate the potential risk from aquatic exposure resulting from indoor uses.

A comparison of freshwater and estuarine/marine fish most sensitive endpoints is provided in **Table 15**. A comparison of freshwater and estuarine/marine invertebrates' most sensitive endpoints is provided in **Table 17**. Vascular and non-vascular plant toxicity data is summarized in **Table 18**. In comparing the endpoints, it should be taken into account that a number of species are tested for each chemical, and only the most sensitive endpoints are included in these tables; in other words, species may vary across chemicals. Further information about the below included endpoints, selected species, study identification, and effects observed, is presented in the **Attachment II**.

For freshwater fish, the most sensitive acute toxicity endpoint is for cyhalothrin (0.029 µg/L) and the least sensitive endpoint for pyrethrins (5.1 µg/L). On a chronic basis, the most sensitive NOAEC is for bifenthrin [0.004 µg/L, however this is an estimated value that is based on the most sensitive pyrethroid (*i.e.*, tefluthrin, which is not included in this assessment)], and the least sensitive endpoint is for pyrethrins (1.9 µg/L).

It appears that, *in general*, the estuarine/marine fish are less sensitive than freshwater fish to pyrethroids. On an acute basis, for estuarine/marine fish the lowest endpoint is for cypermethrin (0.58 µg/L) and the highest is for esfenvalerate (>19.3 µg/L). For bifenthrin, the acute endpoint is 1270 times higher than the limit of solubility for the chemical (17.8 µg/L compared to a solubility of 0.014 µg/L). (As will be shown later, the acute RQ values would always be below the LOC for estuarine/marine fish, based on its limit of solubility.) On a chronic basis, the most sensitive NOAEC belongs to deltamethrin (0.024 µg/L), followed very closely by cyfluthrin (0.025 µg/L). The least sensitive NOAEC is for fenpropathrin (0.81 µg/L).

Table 15. Comparison of Freshwater and Estuarine/Marine Fish Most Sensitive Endpoints for Pyrethroids and Pyrethrins

	Freshwater Fish / (µg/L)		**Estuarine/Marine Fish / (µg/L)**	
Chemical	**96-hr LC_{50}**	**NOAEC/LOAEC**	**96-hr LC_{50}**	**NOAEC/LOAEC**
Bifenthrin	0.15	0.004 (estimate[1])	17.8	0.1/0.14
Cyfluthrin	0.068	0.0042/0.008	4.05	0.025/0.084
Lambda-cyhalothrin	0.029	0.031/0.062	0.807	0.25/0.38
Cypermethrin	0.39	0.051/0.077	0.95	0.125 (ACR[2])
Deltamethrin	0.15	0.017/0.035	0.58	0.024/0.049
Esfenvalerate	0.142	0.017/0.036	>19.3	0.63/1.3
Fenpropathrin	2.2	0.06/0.091	3.1	0.81/2.0
Permethrin	0.79	0.052 (ACR[2])	2.2	0.143 (ACR[2])
Pyrethrins	5.1	1.9/3.0	16	0.7/1.7

For additional details and MRID citations, refer to **Attachment II**.

[1] Estimated based on the most sensitive pyrethroid (tefluthrin).

[2] Estimated value, based on acute-to-chronic ratio (ACR).

For freshwater and estuarine/marine fish, the acute endpoints exceed the limit of solubility of some of the pyrethroids. This is the case for bifenthrin (both freshwater and estuarine/marine fish acute endpoints), cyfluthrin (estuarine/marine fish), deltamethrin (estuarine/marine fish), and esfenvalerate (estuarine/ marine fish). Endpoints measured at above the limit of solubility appear to indicate lower toxicity; however, these measurements are uncertain. The pyrethroids are highly hydrophobic, and should not be bioavailable above the limit of solubility. **Table 16** shows the limit of solubility of the eight pyrethroids in this comparative assessment, plus pyrethrins.

Table 16. Solubility Values for Synthetic Pyrethroids and Pyrethrins

Chemical	**Water Solubility (µg/L)**
Bifenthrin	0.0140
Cyfluthrin	2.32
Lambda-cyhalothrin	5.00
Cypermethrin	3.97
Deltamethrin	0.200

Chemical	**Water Solubility (µg/L)**
Esfenvalerate	6.00
Fenpropathrin	10.3
Permethrin	5.50
Pyrethrins	200 – 9,000[1]

[1] Range given for pyrethrin 1 – pyrethrin 2.

Freshwater and estuarine/marine invertebrates are more sensitive to pyrethroids and pyrethrins than freshwater and estuarine/marine fish. For example, for freshwater invertebrates, the least sensitive acute endpoint belongs to the pyrethrins (0.76 µg/L), while the most sensitive endpoint is for *lambda*-cyhalothrin (0.00008 µg/L). Note that the range of endpoints is very wide (*i.e.*, four orders of magnitude). On a chronic basis, the least sensitive endpoint is for pyrethrins (0.04 µg/L), and the most sensitive NOAEC belongs to deltamethrin (0.000026 µg/L).

For estuarine/marine invertebrates, the most sensitive acute endpoint is for cyfluthrin (0.0022 µg/L), and the least sensitive acute endpoint is for the pyrethrins (1.4 µg/L). Meanwhile, on a chronic basis, the most sensitive NOAEC also belongs to cyfluthrin (0.00007 µg/L), and the least sensitive belongs to the pyrethrins (0.25 µg/L).

In general, the chemical showing the least toxicity to fish and invertebrates are the pyrethrins, with the exception of estuarine/marine fish, for which the acute endpoints for bifenthrin and esfenvalerate are less sensitive than for pyrethrins, and the chronic NOAEC for fenpropathrin is less sensitive than for pyrethrins.

Table 17. Comparison of Freshwater and Estuarine/Marine Invertebrates Most Sensitive Endpoints for Pyrethroids and Pyrethrins

	Freshwater Inverts / (µg/L)		**Estuarine/Marine Inverts / (µg/L)**	
Chemical	**96-hr EC_{50}**	**NOAEC/LOAEC**	**96-hr EC_{50}**	**NOAEC/LOAEC**
Bifenthrin	0.000493	0.000050/0.00009	0.00397	<0.0006/0.0006[2]
Cyfluthrin	0.025	0.00012/0.00025	0.0022	0.00007/0.00017
Lambda-cyhalothrin	0.00008	0.00022/0.00031	0.00491	0.0002/0.0008
Cypermethrin	0.00056	<0.00005/0.00005[2]	0.0054	0.000781/0.00197
Deltamethrin	0.0002	0.000026/0.000050	0.0037	0.00047/0.00073
Esfenvalerate	0.000848	0.0000309 (ACR[1])	0.00466	0.00017/0.00025
Fenpropathrin	0.00305	<0.0015/0.0015[2]	0.021	0.012/0.024
Permethrin	0.0066	0.0042/0.0074	0.018	0.0024/0.0046
Pyrethrins	0.76	0.04/0.10	1.4	0.25/0.64

For additional details and MRID citations, refer to **Attachment II**.

[1] Estimated value, based on acute-to-chronic ratio (ACR).

[2] Non-definitive endpoint.

Although no plant toxicity data were readily available for cyfluthrin, it appears that aquatic plants are much less sensitive to pyrethroids than freshwater and estuarine/marine fish and invertebrates based on the plant data that is available. In many cases, the endpoints were non-definitive, due to low toxicity.

Table 18. Comparison of Vascular and Non-vascular Plants Most Sensitive Endpoints for Pyrethroids and Pyrethrins

	Vascular Plants / (μg/L)		Non-vascular Plants / (μg/L)	
Chemical	**EC_{50}**	**NOAEC**	**EC_{50}**	**NOAEC**
Bifenthrin	>330[1]	330	>290[1]	290
Cyfluthrin	NA	NA	>2[1]	NA
Lambda-cyhalothrin	>0.508[1]	0.508	>310[1]	≥310[1]
Cypermethrin	>1.62	1.62	25000	60
Deltamethrin	>779[1]	779	>3.1[1]	3.1
Esfenvalerate	>8.6[1]	8.6	>5.6[1]	5.6
Fenpropathrin	>1000[1]	1000	63	24
Permethrin	>3.2[1]	3.2	>4.4[1]	0.65
Pyrethrins	1230	480	105	29

NA = No available toxicity data.
For additional details and MRID citations, refer to **Attachment II**.

The endpoints selected for RQ calculations are presented in the footnotes to each of the RQ tables in **Section 7.1** (*Risk Estimation*).

7 Risk Characterization

Risk characterization provides the final step in the risk assessment process. In this step, exposure and effects characterization are integrated to provide an estimate of risk relative to established levels of concern (LOCs; **Section 7.1**). The results are then interpreted for the risk manager through a risk description and synthesized into an overall conclusion (**Section 7.2**). In addition, the risk description also contains a discussion of relevant sources of uncertainty in the risk assessment and sensitivity of the risk assessment findings to important methodological assumptions.

7.1 Risk Estimation

Results of the exposure modelling and toxicity effects data are used to evaluate the likelihood of adverse ecological effects on non-target species. For the assessment of pyrethroids and pyrethrins, the risk quotient (RQ) method is used to compare exposure and measured toxicity values (refer to **Appendix A**). Estimated environmental concentrations (EECs) are divided by the most sensitive acute and chronic toxicity values. The RQs are then compared to the Agency's levels of concern (LOCs). These LOCs, summarized in **Appendix A**, are the Agency's interpretive policy and are used to analyse potential risk to non-target organisms and the need to consider regulatory action. These criteria are used to indicate when a pesticide's use as directed on the label has the potential to cause adverse effects on non-target organisms. LOCs currently address the following risk presumption categories:

Aquatic Animals:

- **Acute risk** - potential for acute risk to non-target organisms which may warrant regulatory action in addition to restricted use classification,

- **Acute risk, listed species –** listed species may be potentially affected by use,
- **Chronic risk –** potential for chronic risk may warrant regulatory action, listed species may potentially be affected through chronic exposure.

7.1.1 Freshwater Fish Risk Quotients

For freshwater fish, there were only a few exceedances of levels of concern (LOCs) (for bifenthrin, *lambda*-cyhalothrin, cypermethrin, and esfenvalerate). For these chemicals, the acute RQs were only slightly above the listed (endangered and/or threatened) species LOC (LOC = 0.05) (RQ range 0.06-0.14). The acute non-listed LOC (0.5) was not exceeded by any of the evaluated pyrethroids and pyrethrins. On a chronic basis, only bifenthrin exceeded the LOC (LOC = 1.0), with an RQ range of 3.5 – 5.3. For all the remaining pyrethroids, the chronic RQs were below the LOC (RQs ≤ 0.61).

Table 19. Freshwater Fish Risk Quotients for Eight Pyrethroids and Pyrethrins

Chemical	Peak EEC = Chronic EEC[1] (μg/L)	96-hr LC$_{50}$ (μg/L)	Acute RQ	NOAEC (μg/L)	Chronic RQ
Bifenthrin	0.014 – 0.0211[2]	0.15	***0.09 – 0.14***	0.004	**3.5 – 5.3**
Cyfluthrin	0.00179	0.068	0.03	0.0042	0.43
Lambda-Cyhalothrin	0.00205	0.029	***0.07***	0.031	0.07
Cypermethrin	0.0232	0.39	***0.06***	0.051	0.45
Deltamethrin	0.00278	0.58	<0.01	0.017	0.16
Esfenvalerate	0.0103	0.142	***0.07***	0.017	0.61
Fenpropathrin	N/A	2.2	N/A	0.06	N/A
Permethrin	0.0124	0.79	0.02	0.052	0.24
Pyrethrins	0.0814	5.1	0.02	1.9	0.04

N/A = does not apply, there are no applicable down-the-drain uses for fenpropathrin.

LOC for acute risk 0.5, acute restricted use 0.1, acute endangered species 0.05, and chronic risk 1.

[1] Stream flow SF$_{1Q10}$ (acute) and SF$_{7Q10}$ (chronic) stream dilution factor = 1.

[2] Exceeded the limit of solubility for bifenthrin (0.014 μg/L). The actual concentration from E-FAST 2014 was 0.0211 μg/L.

7.1.2 Estuarine/Marine Fish Risk Quotients

For estuarine/marine fish there were no exceedances of any of the acute or chronic listed and/or non-listed LOCs (acute listed species LOC = 0.05; acute non-listed species LOC = 0.5; chronic LOC = 1.0). On an acute basis, the RQs were ≤0.02 for all the chemicals. The chronic RQs were ≤0.21 for all the chemicals.

Table 20. Estuarine/Marine Fish Risk Quotients for Eight Pyrethroids and Pyrethrins

Chemical	Peak EEC = Chronic EEC[1] (μg/L)	96-hr LC$_{50}$ (μg/L)	Acute RQ	NOAEC (μg/L)	Chronic RQ
Bifenthrin	0.014 – 0.0211[2]	17.8	<0.01	0.1	0.14 – 0.21
Cyfluthrin	0.00179	4.05	<0.01	0.025	0.07
Lambda-	0.00205	0.807	<0.01	0.25	0.01

Chemical	**Peak EEC = Chronic EEC^1 (µg/L)**	**96-hr LC$_{50}$ (µg/L)**	**Acute RQ**	**NOAEC (µg/L)**	**Chronic RQ**
Cyhalothrin					
Cypermethrin	0.0232	0.95	0.02	0.125	0.19
Deltamethrin	0.00278	0.58	<0.01	0.024	0.12
Esfenvalerate	0.0103	>19.3^3	<0.01	0.63	0.02
Fenpropathrin	N/A	3.1	N/A	0.81	N/A
Permethrin	0.0124	2.2	0.01	0.143	0.09
Pyrethrins	0.0814	16	0.01	0.7	0.12

N/A = does not apply, there are no applicable down-the-drain uses for fenpropathrin.

LOC for acute risk 0.5, acute restricted use 0.1, acute endangered species 0.05, and chronic risk 1.

[1] Stream flow SF$_{1Q10}$ (acute) and SF$_{7Q10}$ (chronic) stream dilution factor = 1.

[2] Exceeded the limit of solubility for bifenthrin (0.014 µg/L). The actual concentration from E-FAST 2014 was 0.0211 µg/L.

[3] Non-definitive endpoint.

7.1.3 Freshwater Invertebrates Risk Quotients

For freshwater invertebrates, on an acute basis, all the RQs exceeded one or more of the LOCs. Cyfluthrin and pyrethrins exceeded only the listed species LOC (acute listed species LOC = 0.05; RQs of 0.07 and 0.11, respectively). The remaining pyrethroids exceeded all the acute LOCs, including the acute non-listed LOC (LOC = 0.5). The highest RQ was for bifenthrin (RQ = 42.8).

On a chronic basis, only the natural pyrethrins and permethrin slightly exceeded the LOC (LOC = 1.0; RQs = 2.0-2.9). All the synthetic pyrethroids and natural pyrethrins exceeded the chronic LOC (1.0) with RQs ranging from 2.0 for pyrethrins (only slightly above the LOC), to >464 for cypermethrin.

Table 21. Freshwater Invertebrates Risk Quotients for Eight Pyrethroids and Pyrethrins

Chemical	**Peak EEC^1 (µg/L)**	**96-hr EC$_{50}$ (µg/L)**	**Acute RQ**	**NOAEC (µg/L)**	**Chronic RQ**
Bifenthrin	0.014 – 0.0211^2	0.000493	**28.4 – 42.8**	0.000050	**280 – 422**
Cyfluthrin	0.00179	0.025	***0.07***	0.00012	**15**
Lambda-Cyhalothrin	0.00205	0.00008	**26**	0.00022	**9.3**
Cypermethrin	0.0232	0.00056	**41**	<0.00005^3	**>464**
Deltamethrin	0.00278	0.0002	**14**	0.000026	**107**
Esfenvalerate	0.0103	0.000848	**12.1**	0.0000309	**333**
Fenpropathrin	N/A	0.00305	N/A	<0.0015^3	N/A
Permethrin	0.0124	0.0066	**1.88**	0.0042	**2.9**
Pyrethrins	0.0814	0.76	***0.11***	0.04	**2.0**

N/A = does not apply, there are no applicable down-the-drain uses for fenpropathrin.

LOC for acute risk 0.5, acute restricted use 0.1, acute endangered species 0.05, and chronic risk 1.

[1] Stream flow SF$_{1Q10}$ (acute) and SF$_{7Q10}$ (chronic) stream dilution factor = 1.

[2] Exceeded the limit of solubility for bifenthrin (0.014 µg/L). The actual concentration from E-FAST 2014 was 0.0211 µg/L.

[3] Non-definitive endpoint.

7.1.4 Estuarine/Marine Invertebrates Risk Quotients

For estuarine/marine invertebrates, on an acute basis, all the chemicals exceeded at least one LOC. For pyrethrins and *lambda*-cyhalothrin the listed species LOC was exceeded (RQs of 0.06 and 0.42, respectively). For all the remaining chemicals the acute non-listed species LOC (0.5) was exceeded and the RQs ranged from 0.69 (permethrin) to 5.31 (bifenthrin).

On a chronic basis, the only chemical that did not exceed the LOC (chronic LOC = 1.0) was pyrethrins with an RQ of 0.33. The remaining chemicals exceeded the LOC with a range of 3.8 (deltamethrin) to 61 (esfenvalerate).

Table 22. Estuarine/Marine Invertebrates Risk Quotients for Eight Pyrethroids and Pyrethrins

Chemical	Peak EEC = Chronic EEC^1 (µg/L)	96-hr EC$_{50}$ (µg/L)	Acute RQ	NOAEC (µg/L)	Chronic RQ
Bifenthrin	0.014 – 0.0211^2	0.00397	**3.53 – 5.31**	<0.0006^3	**>23 – >35**
Cyfluthrin	0.00179	0.0022	**0.81**	0.00007	**26**
Lambda-Cyhalothrin	0.00205	0.00491	***0.42***	0.0002	**10**
Cypermethrin	0.0232	0.0054	**4.3**	0.000781	**30**
Deltamethrin	0.00278	0.0037	**0.75**	0.00073	**3.8**
Esfenvalerate	0.0103	0.00466	**2.21**	0.00017	**61**
Fenpropathrin	N/A	0.021	N/A	0.012	N/A
Permethrin	0.0124	0.018	**0.69**	0.0024	**5.2**
Pyrethrins	0.0814	1.4	***0.06***	0.25	0.33

N/A = does not apply, there are no applicable down-the-drain uses for fenpropathrin.

LOC for acute risk 0.5, acute restricted use 0.1, acute endangered species 0.05, and chronic risk 1.

[1] Stream flow SF$_{1Q10}$ (acute) and SF$_{7Q10}$ (chronic) stream dilution factor = 1.

[2] Exceeded the limit of solubility for bifenthrin (0.014 µg/L). The actual concentration from E-FAST 2014 was 0.0211 µg/L.

[3] Non-definitive endpoint.

7.1.5 Vascular and Non-vascular Aquatic Plants

For vascular plants, there were no exceedances of the listed and non-listed species LOC (LOC = 1.0). All RQs available were ≤0.01. There were no vascular plant endpoints for cyfluthrin. Based on the overall risk picture for all the remaining pyrethroids and pyrethrins, it appears that aquatic vascular plants are not sensitive to pyrethroids.

Table 23. Vascular Plants Risk Quotients for Eight Pyrethroids and Pyrethrins

Chemical	Peak EEC^1 (µg/L)	NOAEC (µg/L)	Listed RQ	EC$_{50}$ (µg/L)	Non-listed RQ
Bifenthrin	0.014 – 0.0211^2	330	<0.01	>330^3	<0.01
Cyfluthrin	0.00179	NA	NA	NA	NA
Lambda-Cyhalothrin	0.00205	0.508	<0.01	>0.508^3	<0.01
Cypermethrin	0.0232	1.62	0.01	>1.62^3	<0.01
Deltamethrin	0.00278	779	<0.01	>779^3	<0.01

Chemical	Peak EEC[1] (µg/L)	NOAEC (µg/L)	Listed RQ	EC$_{50}$ (µg/L)	Non-listed RQ
Esfenvalerate	0.0103	8.6	<0.01	>8.6[3]	<0.01
Fenpropathrin	N/A	1000	N/A	>1000[3]	N/A
Permethrin	0.0124	3.2	<0.01	>3.2[3]	<0.01
Pyrethrins	0.0814	480	<0.01	1230	<0.01

N/A = does not apply, there are no applicable down-the-drain uses for fenpropathrin.
NA = not available.
LOC for acute risk 1, and acute endangered species 1.
[1] Stream flow SF$_{1Q10}$ and stream dilution factor = 1.
[2] Exceeded the limit of solubility for bifenthrin (0.014 µg/L). The actual concentration from E-FAST 2014 was 0.0211 µg/L.
[3] Non-definitive estimate; no effects seen at the highest test concentration.

For non-vascular plants, there were no exceedances of the listed and non-listed species LOC (LOC = 1.0). All RQs available were ≤0.02. There was no reported NOAEC for cyfluthrin. Based on the overall risk picture for all the remaining chemicals, it appears that aquatic non-vascular plants are not sensitive to pyrethroids.

Table 24. Non-vascular Plants Risk Quotients for Eight Pyrethroids and Pyrethrins

Chemical	Peak EEC[1] (µg/L)	NOAEC (µg/L)	Listed RQ	EC$_{50}$ (µg/L)	Non-listed RQ
Bifenthrin	0.014 – 0.0211[2]	290	<0.01	>290[3]	<0.01
Cyfluthrin	0.00179	NA	NA	>2[3]	<0.01
Lambda-Cyhalothrin	0.00205	≥310[3]	≤0.01	>310[3]	<0.01
Cypermethrin	0.0232	60	<0.01	25000	<0.01
Deltamethrin	0.00278	3.1	<0.01	>3.1[3]	<0.01
Esfenvalerate	0.0103	5.6	<0.01	>5.6[1]	<0.01
Fenpropathrin	N/A	24	N/A	63	N/A
Permethrin	0.0124	0.65	0.02	>4.4[3]	<0.01
Pyrethrins	0.0814	29	<0.01	105	<0.01

N/A = does not apply, there are no applicable down-the-drain uses for fenpropathrin. NA = not available.
LOC for acute risk 1, and acute endangered species 1.
[1] Stream flow SF$_{1Q10}$ and stream dilution factor = 1.
[2] Exceeded the limit of solubility for bifenthrin (0.014 µg/L). The actual concentration from E-FAST 2014 was 0.0211 µg/L.
[3] Non-definitive estimate; no effects seen at the highest test concentration.

7.2 Risk Description

7.2.1 Factors Affecting the Risk Quotients

Synthetic pyrethroids and pyrethrins can be characterized as very highly toxic to freshwater and estuarine/marine fish and invertebrates on an acute basis (*i.e.*, LC$_{50}$ or EC$_{50}$ <0.1 mg/L). **Table 15** shows the most sensitive LC$_{50}$s and NOAECs for freshwater and estuarine/marine fish and **Table 17** presents the most sensitive EC$_{50}$s and NOAECs for freshwater and estuarine/marine invertebrates. Freshwater fish are generally more sensitive to pyrethroids and pyrethrins than estuarine/marine fish on an acute and chronic basis. In general, freshwater invertebrates are more sensitive to pyrethroids than estuarine/marine invertebrates on an acute basis. On a chronic basis, a trend is not clear for freshwater invertebrates, when compared to estuarine/marine invertebrates. Comparison of **Table 15** against **Table 17** shows clearly that freshwater and estuarine/ marine invertebrates are orders of magnitude more sensitive to pyrethroids and pyrethrins than freshwater and estuarine/marine fish.

Besides each chemical's toxicity, the RQs are also a function of the EECs, which are in turn dependent of the reported usage (or the production volume for esfenvalerate), and the wastewater removal at the treatment plant (WWT). The wastewater removals for these chemicals lied in the very narrow range from 91.7% (pyrethrins) to 94.0% (bifenthrin), based on EPISUITE v.4.11 estimates. These removals were similar to those estimated based on monitoring of POTWs in California (Markle *et al.*, 2014). In terms of the reported usage, provided by BEAD, the chemical with the highest usage is pyrethrins, with 89,000 lb a.i./year, followed by cypermethrin and bifenthrin with less than half the usage than pyrethrins (34,000 and 32,000 lb a.i./year, respectively). Another chemical with relatively high usage is permethrin (17,000 lb a.i./year). Meanwhile, for the remaining chemicals, the usage is around an order of magnitude lower than permethrin (2,000 – 6,000 lb a.i./ year for the remaining chemicals).

7.2.2 Summary of Risk Quotients

Table 25 provides an overall summary of all the risk quotients for freshwater and estuarine/marine animals for eight pyrethroids and pyrethrins (see also **Tables 19-22** for additional details, including endpoints, EECs, etc.)

Acute and chronic risks quotients (RQs) for freshwater and estuarine/marine fish were generally orders of magnitude lower than for freshwater and estuarine/marine invertebrates. For freshwater fish, there were a few exceedances of the acute listed species LOC (but the RQs did not exceed the non-listed species LOC), and one chronic LOC exceedance, while for estuarine/marine fish there were no exceedances of acute and chronic LOCs.

The highest freshwater fish acute RQ (bifenthrin, RQ = 0.14) does not exceed the acute non-listed LOC (0.5). However, if one considers the reported limit of solubility of bifenthrin and caps the acute EEC at the limit of solubility, the RQ is 0.09.

On a chronic basis, the only chemical that exceeds the LOC (1) for freshwater fish is bifenthrin, with RQs of 3.5 – 5.3. For bifenthrin the NOAEC is an estimate, based on the lowest NOAEC of all synthetic pyrethroids (*i.e.*, NOAEC for tefluthrin[5]). This NOAEC was chosen given the lack of a measured value and the uncertainty that would be introduced if an acute-to-chronic ratio was used, using an LC_{50} of 17.8 µg/L for estuarine/marine fish,

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[5] Tefluthrin is one of the chemicals supported by PWG; however, it is not included in this comparative assessment, plus it has no indoor or residential uses.

which is several orders of magnitude higher than the limit of solubility of bifenthrin (0.014 µg/L).

Table 25. Summary of Risk Quotients for Down-the-Drain Uses of Pyrethroids and Pyrethrins for Freshwater and Estuarine/Marine Fish and Invertebrates.

Chemical	**Risk Quotients**							
	FW Fish		**E/M Fish**		**FW Inverts**		**E/M Inverts**	
	Acute	**Chronic**	**Acute**	**Chronic**	**Acute**	**Chronic**	**Acute**	**Chronic**
Bifenthrin	*0.09 – 0.14*	**3.5 – 5.3**	<0.01	0.14 – 0.21	**28.4 – 42.8**	**280 – 422**	**3.53 – 5.31**	**>23 – >35**
Cyfluthrin	0.03	0.43	<0.01	0.07	*0.07*	**15**	**0.81**	**26**
Lambda-Cyhalothrin	*0.07*	0.07	<0.01	0.01	**26**	**9.3**	*0.42*	**10**
Cypermethrin	*0.06*	0.45	0.02	0.19	**41**	**>464**	**4.3**	**30**
Deltamethrin	<0.01	0.16	<0.01	0.12	**14**	**107**	**0.75**	**3.8**
Esfenvalerate	*0.07*	0.61	<0.01	0.02	**12.1**	**333**	**2.21**	**61**
Fenpropathrin[1]	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Permethrin	0.02	0.24	0.01	0.09	**1.88**	**2.9**	**0.69**	**5.2**
Pyrethrins	0.02	0.04	0.01	0.12	*0.11*	**2.0**	*0.06*	0.33

A light shaded and italics "*RQ*" means it exceeds the listed species LOC. A dark shaded and bolded "**RQ**" represents LOC exceedances of the listed species and non-listed species LOCs.

[1] N/A = There are no applicable down-the-drain uses for fenpropathrin.

For aquatic invertebrates the overall risk picture is different than for fish, since for both freshwater and estuarine/marine invertebrates there are multiple exceedances of LOCs for all the chemicals evaluated. For freshwater invertebrates, the lowest acute RQs were for cyfluthrin and pyrethrins, with RQs of 0.07 and 0.11, exceeding only the acute listed species LOC (0.05). Even though cyfluthrin is more acutely toxic to freshwater invertebrates than pyrethrins (EC_{50}s of 0.025 for cyfluthrin and 0.76 µg/L for pyrethrins), the acute EEC for pyrethrins is higher as a result of its higher usage, resulting in a higher EEC and consequently RQ. On an acute basis, for all the remaining pyrethroids the RQs exceed the acute non-listed species LOC (0.5) (RQ range **1.88 – 42.8**). On a chronic basis, all chemical's RQs exceeded the chronic LOC (1), with values ranging from 2.0 to >464. Meanwhile, for estuarine/marine invertebrates, on an acute basis all nine chemicals exceeded at least one LOC. The lowest RQ (0.06) for pyrethrins exceeded only marginally the listed species LOC (0.05). The highest acute RQ was for bifenthrin (RQ = 5.31). On a chronic basis, only the pyrethrins did not exceed the chronic LOC. For all the other pyrethroids, the RQs ranged from 3.8 (deltamethrin) to 61 (esfenvalerate).

7.2.3 Number of Days an Acute Concentration of Concern is Exceeded

The Probabilistic Dilution Model (PDM) is itself another module of E-FAST that may be run in tandem with DtD. It is intended to address aquatic ecological exposures and risks. PDM is used by the DtD module to calculate daily concentrations and to predict the number of days per year a chemical's concentration of concern (COC) in an ambient water body will be exceeded by the discharge from a facility. Up to three COCs may be entered in EFAST 2014.

For additional characterization, one representative chemical was selected to run PDM. The chemical selected was permethrin and only acute exposure to aquatic invertebrates was addressed. The permethrin RQs were deemed to be representative of other pyrethroids since they are approximately in the middle, compared to the RQs for other pyrethroids and pyrethrins.

The acute COC may be calculated by multiplying the acute endpoint (*i.e.*, 96-hr EC_{50} for invertebrates) by the level of concern (*e.g.*, 0.05 for listed species). For permethrin, the acute COC for freshwater invertebrates and listed species is 0.00033 µg/L (*i.e.*, acute endpoint x 0.05 = 0.0066 µg/L x 0.05). Meanwhile, the acute COC for estuarine/marine invertebrates and listed species is 0.00090 µg/L (*i.e.*, 0.018 µg/L x 0.05). Since three COCs can be entered, an additional COC was entered, related to an acute non-listed LOC (0.5) for freshwater invertebrates. Therefore, the additional COC entered was 0.0033 µg/L for freshwater invertebrates and non-listed species (*i.e.*, 0.0066 µg/L x 0.5). **Table 26** summarizes the PDM results for permethrin (for the example output file, see the **Appendix C**).

Table 26. Number of Days per Year and Percent of Year a Concentration of Concern Is Exceeded for Permethrin

Taxa	LOC	Concentration of Concern (COC) (µg/L)	Number of Days Concentration of Concern is Exceeded (# days)	Percent of Year Concentration of Concern is Exceeded (%)[1]
Freshwater invertebrates	0.05 (acute listed species)	0.00033	327	89
Freshwater invertebrates	0.5 (acute non-listed species)	0.0033	154	42
Estuarine/marine invertebrates	0.05 (acute listed species)	0.00090	268	74

[1] Percent value was rounded to the nearest digit.

As indicated in the table, for freshwater invertebrates the *acute listed species* concentration of concern is exceeded 327 days/year (*i.e.*, 89% of the year) and the *acute non-listed species* concentration of concern is exceeded 154 days/year (*i.e.*, 42% of the year). For estuarine/marine invertebrates, the *acute listed species* concentration of concern is exceeded 268 days/year (74% of the year). All these numbers of days are related to the high end scenario (10th percentile), and results provide additional line-of-evidence that the chemical has the potential to affect both freshwater and estuarine/marine invertebrates.

7.2.4 Comparison of RQs for Small *vs.* Medium-Sized Recipient Streams

As it has been described earlier, for the protection of aquatic life, the developers of the model have deemed two types of stream flows (SFs) suitable for ecological effects. They include the SF_{1Q10} (single day of lowest flow over a 10-year period), providing acute surface water concentrations to compare with acute endpoints. For chronic exposure, the stream flow SF_{7Q10} (seven consecutive days of lowest flow over a 10-year period) is used to compare with chronic endpoints for aquatic animals. Additionally, exposure results are calculated based on high-end surface water concentrations (*i.e.*, upper 10th percentile). These results represent the bounding high-end exposures and small streams, and they are used for risk assessment purposes. Besides the 10th percentile results, the model provides the 50th percentile results. These are exposure results calculated based on the median surface water concentrations (*i.e.*, 50th percentile). They represent central tendency exposure and mid-sized stream flows.

For characterization purposes, for permethrin (taken as a representative pyrethroid), at the 50th percentile, the acute exposure concentration is $6.15\text{x}10^{-4}$ µg/L and the chronic concentration is $5.10\text{x}10^{-4}$ µg/L. The stream dilution factors are 20.08 and 24.22, respectively. By way of comparison, the concentration for the 10th percentile is $1.24\text{x}10^{-2}$ µg/L (for both, acute and chronic exposure), with a stream dilution factor of 1.00 (for acute and chronic exposure). This means that the 10th percentile concentration is up to 20-24 times higher than the acute and chronic 50th percentile concentrations (for the example output file, see the **Appendix C**). EFED calculated RQs for mid-sized stream flows (50th percentile) and the compared results for permethrin are shown in **Table 27**.

For permethrin it was found that there are no exceedances of the LOCs for freshwater or estuarine/marine fish. For *small stream flows* (high-end scenario), there are exceedances of all the acute listed and acute non-listed and chronic LOCs for freshwater and estuarine/marine invertebrates. Meanwhile, it was found that for *mid-sized stream flows* (lower-end scenario), there is only a single slight exceedance of the listed species LOC for acute exposure for freshwater invertebrates (RQ = 0.09; LOC = 0.05). There are no exceedances of the acute non-listed, or chronic LOCs for freshwater and estuarine/marine invertebrates, or freshwater or estuarine/marine fish.

For freshwater and estuarine/marine invertebrates, the chronic LOC is exceeded for small stream flows, but not for mid-sized stream flows.

Table 27. Risk Quotients for Down-the-Drain Uses of Permethrin for Freshwater and Estuarine/Marine Fish and Invertebrates: Small & Medium-Sized Stream Flows

Chemical Permethrin (represented body of water)	**Risk Quotients**							
	FW Fish		**E/M Fish**		**FW Inverts**		**E/M Inverts**	
	Acute	**Chronic**	**Acute**	**Chronic**	**Acute**	**Chronic**	**Acute**	**Chronic**
Low stream flows EEC = 0.0124 µg/L[1]	0.02	0.24	0.01	0.09	**1.9**	**3.0**	**0.69**	**5.2**

Chemical Permethrin (represented body of water)	**Risk Quotients**							
	FW Fish		**E/M Fish**		**FW Inverts**		**E/M Inverts**	
	Acute	**Chronic**	**Acute**	**Chronic**	**Acute**	**Chronic**	**Acute**	**Chronic**
Mid-sized stream flows Acute EEC = $6.15x10^{-4}$ µg/L; chronic EEC = $5.10x10^{-4}$ µg/L[2]	<0.01	0.01	<0.01	<0.01	*0.09*	0.12	0.03	0.21

A light shaded and italics "*RQ*" means it exceeds the listed species LOC. A dark shaded and bolded "**RQ**" represents LOC exceedances of the listed species and non-listed species LOCs.

[1] Based high-end surface water concentrations (*i.e.*, upper 10th percentile), representing small streams. The stream dilution factor is 1.00 (both acute and chronic).

[2] Based on median surface water concentrations (*i.e.*, 50th percentile), representing mid-sized streams. The stream dilution factor is 20.08 (acute) and 24.22 (chronic).

7.2.5 Comparison of Modelled and Monitored Concentrations in POTWs

Another line of evidence is the monitoring of pyrethroids. Monitoring data from POTWs in California corroborate the presence of synthetic pyrethroids in POTW effluents. For example, monitoring of 31 POTWs by Markle *et al.* (2014) in 67 effluent samples, and tested for eight pyrethroids, demonstrated that the pyrethroid detected with the lowest frequency (3.2% or 2 samples) was fenpropathrin. This result is not unexpected, since fenpropathrin does not have residential uses. Other pyrethroids were detected from 16% frequency (deltamethrin) to 81-82% frequency (cypermethrin and bifenthrin). Comparison of monitored and modelled results indicated that measured and predicted concentrations are comparable. Refer to **Table 13**, in **Section 5.7** for further details. One notable exception is permethrin, which is a chemical that is also used for lice control in human medications (shampoos and lotions); however, these products are regulated by FDA; if the active ingredient in these products is also a pesticidal active ingredient, the risk assessor should consider only the pesticidal use in the ecological risk assessment.

Table 28. Summary of Risk Quotients for Down-the-Drain Uses of Pyrethroids and Pyrethrins for Freshwater and Estuarine/Marine Fish and Invertebrates, Based Upon Maximum Monitored Concentrations (based on Markle *et al.*, 2014)

Chemical	**Max. Conc. µg/L**	**Risk Quotient Based on Monitored Concentration**							
		FW Fish		**E/M Fish**		**FW Inverts**		**E/M Inverts**	
		Acute	**Chronic**	**Acute**	**Chronic**	**Acute**	**Chronic**	**Acute**	**Chronic**
Bifenthrin	0.0039	0.03	0.98	<0.01	0.04	**7.9**	**78**	**0.98**	**>6.5**
Cyfluthrin	0.004	*0.06*	0.95	<0.01	0.16	*0.16*	**33**	**1.8**	**57**
Lambda-Cyhalothrin	0.0016	*0.06*	0.05	<0.01	0.01	**20**	**7.3**	*0.33*	**8.0**
Cypermethrin	0.013	0.03	0.25	0.01	0.10	**23**	**>260**	**2.4**	**17**

Chemical	Max. Conc. µg/L	Risk Quotient Based on Monitored Concentration							
		FW Fish		E/M Fish		FW Inverts		E/M Inverts	
		Acute	Chronic	Acute	Chronic	Acute	Chronic	Acute	Chronic
Deltamethrin	0.0012	0.01	0.07	<0.01	0.05	**6.0**	**46**	*0.32*	**2.6**
Esfenvalerate	0.0006	<0.01	0.04	<0.01	<0.01	**0.71**	**19**	*0.13*	**3.5**
Fenpropathrin[1]	0.0008	<0.01	0.01	<0.01	<0.01	*0.26*	>0.53	0.04	0.07
Permethrin	0.170	*0.22*	**3.3**	*0.08*	**1.2**	**26**	**40**	**9.4**	**71**
Pyrethrins[2]	NA	NA	NA	NA	NA	NA	NA	NA	NA

A light shaded and italics "*RQ*" means it exceeds the listed species LOC. A dark shaded and bolded "**RQ**" means it exceeds the listed species and non-listed species LOCs.

[1] N/A = There are no applicable down-the-drain uses for fenpropathrin; however, it was observed in the monitoring report.

[2] NA = There is no available monitoring data for pyrethrins. Chemical was not included in the Markle, *et al.*, 2014 monitoring report.

Comparison of RQs derived based on the maximum monitored concentration against the acute and chronic endpoints yields the following findings and are detailed in **Table 28**:

- All RQ values based on maximum monitoring concentration for bifenthrin, *lambda*-cyhalothrin, cypermethrin, deltamethrin and esfenvalerate are lower than the previously calculated RQs, based on modelling.
- All RQs for cyfluthrin are slightly above previously calculated from the modelling results, but they are of the same order of magnitude.
- RQs for permethrin are higher than previously calculated from modelling results. This may be due to the additional uses for permethrin for lice control in humans (*considered drug uses*), which may have caused permethrin concentrations in the environment (*i.e.*, in POTW effluents reaching bodies of water), to be higher than predicted based solely on pesticidal uses of the chemical. The drug uses of permethrin are under the purview of the Food and Drug Administration (FDA) and are evaluated separately by the sister Agency. Another factor that may have affected the RQs for permethrin based on the usage information is that BEAD could not locate usage information on the products used for the treatment of clothes or the pretreated clothes, which are used to prevent mosquito bites. Based on usage information, the RQs for freshwater and estuarine/marine fish for permethrin were below the LOCs; however, based on the highest monitored values, the RQs exceed LOCs for both freshwater and estuarine/marine fish.
- A set of RQs could be derived for fenpropathrin, based on a limited number of detections. For fenpropathrin, there is only one exceedance and only of the acute listed species LOC for freshwater invertebrates. Previously in this assessment, no RQs were calculated since fenpropathrin does not have residential uses and therefore there was no usage data or production volume.
- No RQs could be derived for pyrethrins, based on monitoring, since there is no available monitoring data for this chemical.

7.2.6 Uncertainties and Other Issues

7.2.6.1 Usage Information and Production Volume

The Biological and Economic Analysis Division (BEAD) provided usage information for several synthetic pyrethroids (USEPA 2015b). In short, BEAD concluded, based on private market research data to determine the estimated amounts of pyrethroids used indoor in the U.S. by Pest Control Operators (PCOs) and consumers, that overall approximately 225,000 pounds A.I. of pyrethroids and pyrethrins (combined) are estimated to be used indoors annually in the U.S., in ways that have some likelihood of resulting in DtD exposure. BEAD considered this a very high-end estimate, "since it assumes that all indoor treatments targeting flies, bedbugs, and cockroaches are equally likely to result in DtD exposure every time an application is made" (USEPA 2015b). Information is provided for seven pyrethroids assessed in this document plus the pyrethrins; in addition, a number of other pyrethroids were included (but not assessed in this document). For a characterization of the provided usage and adjustment factors, refer to USEPA (2015b). It should be noted that the EECs are highly dependent on the usage. The reason is that these pyrethroids and pyrethrins show a narrow range of wastewater treatment removals. Therefore, the estimated usage uncertainties affect the risk quotients to a high degree. Stakeholders have indicated that the lack of source characterization frustrates design of targeted mitigation measures to reduce POTW discharges. This issue is acknowledged. This PRA relies on the best available usage data (or production volume), for each of the chemicals evaluated. Also, stakeholders have indicated that there is no method to estimate influent loads and that EFED just assumes 100% of the production of a pesticide is discharged down indoor drains. This is a conservative assumption, although it should be stressed that BEAD applied certain correction factors to the usage information, to refine their estimates of pyrethroid and pyrethrins usage.

7.2.6.2 Wastewater Treatment Removal

Wastewater treatment plant removal efficiencies (WWT) were calculated using the Sewage Treatment Plant (STPWIN™) module of EPI Suite v.4.1. According to the STPWIN™ Help manual, EPI Suite's STPWIN™ program was conservative (*i.e.*, leading to higher concentrations) predicting removal percent (WWT) 88% of the time (using its default half-lives of 10,000 hours for 29 of 33 chemicals evaluated, for primary clarifier, aeration vessel and settling tank); however, the evaluation was based on a set of chemicals which are not pesticides. Stakeholders have indicated that EFED uses of the average or a single available literature wastewater treatment plant removal efficiency value, with no accounting for the range of removal efficiencies in real POTWs. Comparison of modeled WWTs against estimated monitoring removal values from POTWs in CA are favorable. However, even though a relatively large data set of monitored concentrations from California POTWs was available, it is uncertain whether the estimated WWTs are representative of other sites across the U.S. A potentially more suitable and reliable alternative, is data derived from a bench scale study that may be required either during the new chemical registration process of a chemical or during Registration Review, to further refine this input parameter (see next paragraph). Finally, for a few chemicals, when available, WWT can be obtained from actual monitoring studies of influent and effluent from POTWs.

For the pyrethroids, the Division requested a bench scale study, with the expectation to obtain reliable removal efficiency data for each pyrethroid. It was found that the study did not provide suitable removal efficiency information. EFED then compared monitoring data from 31 California POTWs (of variable treatment design), against EPI Suite's STPWIN estimates. Data were comparable. Thus, the STP – EPI Suite estimates were used in this draft Part I of the Tier 1 Preliminary Aquatic Ecological Risk Assessment (PRA) for Eight Pyrethroids and Pyrethrins. EFED acknowledges that there were limitations in the California monitoring data (*e.g.*, use of grab samples, see next paragraph); however, it was the best available information available, which was submitted to EFED, and also published in the open literature. For pyrethroids, which bind to organic matter tightly, EFED believes that setting the removal efficiency to 0 (zero) is unrealistically conservative. Additionally, the results of the draft PRA were consistent with the expected behavior of the pyrethroids, and the monitoring and modeling results were similar.

It should be acknowledged that in the study by Markle *et al.* (2014), in which the monitoring of 31 POTWs in California was reported, tested samples were grab samples, as opposed to 24-hour composite samples. The composite samples represent a longer period of time and avoid the variability inherent to a single grab sample. Despite this uncertainty, it was noted a good agreement between the modeled (E-FAST's DtD), and measured values.

7.2.6.3 E-FAST 2014 Uncertainties

Concentrations derived from the E-FAST 2014 model are uncertain. The model assumes no degradation and no sorption of the chemical with organic matter or with the sediments in the body of water. Pyrethroids are known to bind to sediments, particulate matter and suspended organic matter. The E-FAST model is intended to be a Tier 1 model, and concentrations are considered conservative. Nonetheless, comparison of modeled concentrations *vs.* maximum monitored concentrations from California POTWs indicated similar concentrations (within about one order of magnitude).

In natural water, it appears that pyrethroids are less bioavailable than in pure water, at the same concentrations. For example, Parry & Young (2013) reported the distribution of pyrethroid insecticides in dechlorinated final effluent from Sacramento Regional Wastewater Treatment Plant. In sorption studies for four pyrethroids to suspended particulate matter, it was found that the estimated K_{iDOC} values were about $2x10^5$ L/kg for permethrin, cypermethrin and *lambda*-cyhalothrin, and $8x10^5$ L/kg for bifenthrin. A large percentage of the pyrethroid load was further associated with dissolved organic carbon (DOC). They observed in samples taken at different times of the years 2010-2011, that <6% of the total pyrethroid load was dissolved (bioavailable) across sampling and chemicals. However, these results should be taken with caution. When the whole water concentrations and the estimated dissolved concentrations were compared against the observed toxicity of the water to *Hyalella azteca*, it was found that neither of them correlated well with the toxicity (assuming additive pyrethroid toxicity, based on toxic units). The study authors speculated that toxicity may have been attributable to other factors.

Certain stakeholders have noted that many POTWs have zero dilution. Peak and chronic EEC results in E-FAST 2014 used by EFED are the higher level estimates, 90th percentile and a stream dilution factor of 1.00 (one) for acute and chronic exposure, representing small receiving or effluent dominated bodies of water. This is a conservative approach.

The comment about use of dilution factors has been raised in certain Problem Formulations for Registration Review, and more recently, in meetings with stakeholders. Evaluation of E-FAST's DtD module results indicated that for the high end scenario, the stream dilution factor (SDF) is 1.00, which appears to be the minimum allowed value. The SDF is the volume of the receiving stream flow divided by the volume of the wastewater treatment facility effluent flow. The high end scenario concentration represents small receiving streams (10th percentile and SDF = 1.00). A SDF of 1.00 also represents "effluent dominated" surface bodies of water. These results may be compared against the alternative SDFs on the order of 20-24 for the 50th percentile concentration that represents midsized receiving streams. The SDF of 1.00 is used in the risk assessment, while SDFs of 20-24 may be used for characterization of results. For characterization, for one chemical, EFED calculated RQs for the 50th percentile and stream dilution factor of ~20 to ~24, representing a mid-sized receiving body of water.

7.2.6.4 Use of E-FAST for Estuarine/Marine Bodies of Water

In prior consultation with OPPT, it was pointed out that the DtD module better represents freshwater receiving bodies of water; however, in the majority of the risk assessments to date, and in the draft Pyrethroids and Pyrethrins PRA, it has been assumed that the effluent from a POTW can come in contact with estuarine/marine bodies of water. OPP considers exposure in saltwater environments, including discharges into estuarine/marine environments by comparing estimated exposure concentrations (EECs), to toxicity endpoints from estuarine/marine organisms, especially when they are more sensitive than freshwater taxa. The EECs from the DtD module of E-FAST 2014 are therefore compared against saltwater endpoints. This is somewhat similar to what the Division does with the results from agricultural modeling and the standard ecological pond. Note, however, that estuarine/marine bodies of water are larger than freshwater bodies of water, so it appears that this is a conservative approach.

7.2.6.5 Uncertainty Regarding Possible Pest Outbreaks

The observed EECs may be higher or lower than predicted by E-FAST 2014 depending on the usage in households in a certain areas leading to POTWs. For example, there could be high pest pressure in a particular site during a period of time that leads to higher observed environmental concentrations (compared to predicted EECs), in that particular POTW. E-FAST 2014's DtD module does not account for possible pest outbreaks in a certain region that would cause a sizable use of a single pyrethroid. In such a case, it is possible that a single plant would receive a higher pesticide amount and as a consequence, the effluent would show higher pesticide levels than predicted by E-FAST 2014.

7.2.6.6 Use of the National Per-Capita Wastewater Release

Furthermore, E-FAST 2014 uses a per capita wastewater release based on the national average. The amount of water used regionally may be largely variable, presumably leading to differences in EECs. Another comment that has arisen regarding the DtD module is that it assumes even per-capita use across the entire nation. According to stakeholders, water conservation has greatly reduced per capita flows in the west, southwest, and southeast US. These lower flows mean less dilution. The per capita flow is not an optional input value in the model. This is a limitation of the model and it is discussed in the modeling assumptions. Currently, for a national Tier 1 assessment, the model is the best available approach for the down-the-drain uses assessed.

7.2.6.7 Issues of Biosolids, Recycled Water and Air Emissions

Stakeholders have indicated that EFED has not modeled biosolids, recycled water, or air emissions from POTWs. Regarding air emissions, EPI Suite predicts short half-lives for pyrethroids, in air, due to OH-radical emissions. For example, for bifenthrin, which appears to be the pyrethroid with the highest volatilization potential due to its extremely low solubility, EPI Suite predicts an OH-radical reaction half-life of only 0.36 days. Furthermore, pyrethroids show low vapor pressures and Henry's Law Constants, and very high K_{OC}s and K_{OW}s. Based upon these physicochemical and fate properties, the potential air emissions should be limited (further details about the fate of the individual pyrethroids can be found in the **Attachment III**). Pyrethroids and pyrethrins should not persist in air for long periods of time.

EFED and PRD have worked with the Office of Water to obtain a better handle on the potential exposure from application of biosolids to land. However, for the purposes of the down-the-drain assessment, EFED considers removal of pyrethroids to be a "sink" that reduces the direct exposure to water from down-the-drain uses. In addition, the Divisions believe that the exposure from land-applied biosolids is likely to be significantly less than the exposure that could occur from direct application of pyrethroids in agriculture according to product labels.

7.2.6.8 Ecological Effects Data Gaps and Uncertainties

Although the aquatic toxicity dataset was fairly robust for this group of chemicals, some data gaps were present. For bifenthrin, chronic fish toxicity data were not available and so an estimated endpoint was used in risk calculations. For bifenthrin, cypermethrin, permethrin and pyrethrins, some or all of the vascular and non-vascular aquatic plant toxicity data were missing. This is an uncertainty and risk could not be estimated nor precluded.

Several of the toxicity studies also had non-definitive endpoints. For esfenvalerate, the estuarine/marine fish acute endpoint was a non-definitive, greater than (>), value due to low toxicity. For cyfluthrin, *lambda*-cyhalothrin, deltamethrin, esfenvalerate and fenpropathrin, some or all of the vascular and non-vascular aquatic plant endpoints were also non-definitive, greater-than (>), values. However, in all cases, information was sufficient to provide boundaries of the risk estimate resulting in clear evidence that the RQs were below LOCs. Therefore, data were sufficient for risk estimations.

An additional uncertainty is the unknown potential for synergists, such as piperonyl butoxide and MGK-264, to persist in POTW discharges. Information on their persistence through POTW treatment processes has not yet been quantified. Since they are present in some pet shampoos and other end-use products containing pyrethroids, they may also enter POTWs. Because their synergistic mechanism of binding mixed-function oxidases that would break down pyrethroids, is associated with acute toxicity, rather than chronic toxicity and considered to be a short-term effect; for this Part I of the comparative risk assessment, synergism was not assessed. However, synergists that persist in POTW effluents, along with pyrethroids may enter the aquatic environment and be available for simultaneous exposure to aquatic organisms. Therefore, the likelihood of enhanced toxicity due to DtD synergists is an uncertainty.

For freshwater and estuarine/marine fish, the acute endpoints exceed the limit of solubility of some of the pyrethroids. This is the case for bifenthrin (both freshwater and estuarine/marine fish acute endpoints), cyfluthrin (only estuarine/marine fish), deltamethrin (both freshwater and estuarine/marine fish), and esfenvalerate (only estuarine/ marine fish). Endpoints derived at above the limit of solubility may be uncertain.

In summary, the toxicity data gaps and resulting uncertainties affecting the risk conclusions were:

- Chronic risk to fish from bifenthrin use due to missing toxicity data, although the approach taken appears to be conservative;
- Risk to aquatic plants from bifenthrin, cypermethrin, permethrin and pyrethrins use due to missing toxicity data;
- Potential for toxicity enhancement from synergists due to lack of information synergist persistence in POTW discharges; and,
- Endpoints derived at above the limit of solubility, which is typically very small for all the pyrethroids.

Acclimation/Adaptation

The risk characterization for aquatic organisms is based on standard toxicity studies using laboratory-reared cultures. Recently, two studies have been published which document much greater tolerance of field-collected freshwater amphipods (*Hyalella azteca*) from urban and agriculturally-impacted streams in California with a known history of pyrethroid contamination (Weston *et al.,* 2013; Clark *et al.,* 2015). Compared to laboratory populations, Weston *et al.* report that field-collected *H. azteca* from urban impacted sites were up to 550 times less sensitive on an acute exposure basis to two common pyrethroids (cyfluthrin, bifenthrin). Using multiple lines of evidence, Weston *et al.* report that the resistant populations of field-collected *H. azteca* originate from genetically-distinct clads[6] and that the increased tolerance to pyrethroids likely pertains to genetic mutations associated with voltage-gated sodium channel membrane proteins, which are the target receptors of pyrethroid insecticides. Similarly, Clark *et al.* report field collected aquatic *H. azteca* to be less sensitive on an acute exposure basis to bifenthrin and cypermethrin by up to two orders of magnitude. Interestingly, Clark *et al.* report a substantial reduction in pyrethroid resistance of field collected *H. azteca* when reared in the laboratory over three generations, although the F3 generation populations were still about an order of magnitude less sensitive compared to populations that originated in the laboratory. These authors also report a strong seasonal variation in sensitivity of field-collected amphipods to these pyrethroids, which may be related to temporal variation in pyrethroid exposure history and subsequent activation of compensatory mechanisms associated with pesticide resistance (*e.g.*, enzyme activation).

Considering the well-documented occurrence of insect resistance to pyrethroids and other insecticides, these examples of pyrethroid resistance with aquatic arthropods are not surprising. However, they do have implications for this ecological risk assessment, particularly since *H. azteca* is consistently among the most sensitive aquatic invertebrate species to pyrethroids. One implication is that acute risks to pyrethroid-impacted populations of aquatic invertebrates may be much lower (by up to two orders of magnitude) than predicted from this ecological risk assessment. However, it is important to caveat this interpretation because there is evidence that suggest *H. azteca* may experience periods of more or less tolerance depending on the prior history of pesticide exposure.

Another implication is that pyrethroids are already altering the genetic and/or epigenetic composition of aquatic crustaceans (amphipods), which could have deleterious consequences for overall organism and population fitness. For example, populations that become genetically tolerant to pyrethroids due to repeated exposures over generations may be genetically less diverse compared to less impacted populations. This loss of genetic diversity may have deleterious consequences when organisms are challenged by other anthropogenic or natural stressors, thereby compromising population health and fitness (*e.g.*, Reed and Frankham, 2003). Currently, the evolutionary consequence of genetic tolerance to chemical stressors is not well understood. Furthermore, there is uncertainty when extrapolating results from *H. azteca* to other species, which may have a lower or greater capacity for acclimating or adapting to pyrethroid perturbations.

7.2.6.9 Methodologies for Effects Assessment

Another comment stakeholders have made is that methodologies for effects assessment once environmental concentrations are estimated by POTW modeling are also an issue. Toxicity estimates used in EFED risk calculations often come from submitted studies using recommended species (*e.g.*, *Daphnia magna*), but when more sensitive toxicity endpoints are available, these are also used if the study is determined to be sound. NPDES permit

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[6] *H. azteca* is currently considered a “species complex” consisting of multiple genetically distinct populations across geographic regions.

limits are more likely based on effluent toxicity to species selected for routine testing rather than the more detailed toxicity profile compiled for pesticide registration (*e.g., Ceriodaphnia dubia*, rather than *Daphnia magna;* https://www.epa.gov/npdes/npdes-permit-limits#wet). In both cases, species tested are viewed as surrogates for other species present in the environment and the selection of test species based on multiple considerations. If more sensitive toxicity data are available, EFED encourages its submission to the Division.

8 Conclusions

As shown in **Section 3.4.1**, the risk hypothesis in this assessment:

> *Pyrethroids and pyrethrins, when used indoors, in accordance with current labels, may result in off-site movement of the compound via wash-off into surface waters via drains and municipal wastewater treatment plants, leading to exposure of non-target aquatic plants and animals. This potential exposure pathway may result in adverse effects upon the survival, growth, and/or reproduction of non-target aquatic animals, but not to non-target aquatic plants*.

The assessment concludes that the use of bifenthrin, cypermethrin, cyfluthrins, deltamethrin, esfenvalerate, cyhalothrins, permethrin and pyrethrins, when used in accordance with registered labels, can result in acute and/or chronic risk LOC exceedances for freshwater and estuarine/marine invertebrates, from the indoor down-the-drain exposure to POTWs which in turn result in releases to certain bodies of water. Additionally, it results in acute risk listed species LOC exceedances for freshwater fish for bifenthrin, *lambda*-cyhalothrin, cypermethrin and esfenvalerate, plus chronic risk LOC exceedances for freshwater fish for bifenthrin. There are no potential acute or chronic risk LOC excedances for estuarine/marine fish, or acute listed or non-listed species LOC exceedances for vascular and non-vascular plants from the down-the-drain uses for bifenthrin, cypermethrin, cyfluthrin, deltamethrin, esfenvalerate, *lambda*-cyhalothrin, permethrin, and the pyrethrins. Since it has no residential indoor down-the-drain uses, for fenpropathrin there is no potential risk to freshwater or estuarine/marine aquatic animals or aquatic plants.

Two additional lines of evidence appear to support the conclusion that pyrethroids have a potential to cause acute and/or chronic risk for freshwater and estuarine/marine invertebrates: First, the monitoring of 31 POTWs in California shows that the pyrethroid concentrations in effluents are within an order of magnitude than it was predicted by E-FAST's DtD module. Second, an analysis of the percent of days in a year that a concentration of concern (COC) is exceeded for the example chemical permethrin (excluding the pharmaceutical uses) for acute exposure is 89% (acute listed species COC for freshwater invertebrates), 42% (acute non-listed species COC for freshwater invertebrates) and 74% (acute listed species COC for estuarine/ marine invertebrates). These high percentages suggest that there is risk of concern.

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Appendix A. Risk Quotient Method

Risk characterization integrates the results of the exposure and ecotoxicity data to evaluate the likelihood of adverse ecological effects. The means of this integration is called the quotient method. Risk quotients (RQs) are calculated by dividing exposure estimates by acute and chronic ecotoxicity values.

RQ = EXPOSURE/TOXICITY

RQs are then compared to OPP's levels of concern (LOCs). These LOCs are used by OPP to analyze potential risk to nontarget organisms and the need to consider regulatory action. The criteria indicate that a pesticide used as directed has the potential to cause adverse effects on nontarget organisms. LOCs currently address the following risk presumption categories: (1) acute risks - regulatory action may be warranted in addition to restricted use classification, (2) acute restricted use - the potential for acute risk is high, but may be mitigated through restricted use classification, (3) acute endangered species - endangered species may be adversely affected, and (4) chronic risk - the potential for chronic risk is high regulatory action may be warranted. Currently, EFED does not perform assessments for chronic risk to plants, or chronic risk from granular/bait formulations to birds or mammals.

The ecotoxicity test values (measurement endpoints) used in the acute and chronic risk quotients are derived from required studies. Examples of ecotoxicity values derived from short-term laboratory studies that assess acute effects are: (1) LC_{50} (fish and birds), (2) LD_{50} (birds and mammals), (3) EC_{50} (aquatic plants and aquatic invertebrates) and (4) EC_{25} (terrestrial plants). Examples of toxicity test effect levels derived from the results of long-term laboratory studies that assess chronic effects are: (1) LOAEL or LOAEC (birds, fish, and aquatic invertebrates) and (2) NOAEL or NOAEC (birds, fish and aquatic invertebrates). For birds, mammals, fish and aquatic invertebrates the NOAEL or NOAEC generally is used as the ecotoxicity test value in assessing chronic effects, although other values may be used when justified. Risk presumptions and the corresponding RQs and LOCs, are tabulated below.

Table A-1. Summary of the levels of concern used for the risk quotient method.

Risk Presumption	RQ	LOC
Birds and mammals		
Acute Risk	Diet-based EEC/LC$_{50}$ or dose-based EEC/LD$_{50}$	0.5
Acute Restricted Use	Diet-based EEC/LC$_{50}$ or dose-based EEC/LD$_{50}$ (or LD$_{50}$ < 50 mg/kg)	0.2
Acute Endangered Species	Diet-based EEC/LC$_{50}$ or dose-based EEC/LD$_{50}$	0.1
Chronic Risk	Diet or dose-based EEC/NOAEC	1
Aquatic Animals		
Acute Risk	EEC/LC$_{50}$ or EC$_{50}$	0.5
Acute Restricted Use	EEC/LC$_{50}$ or EC$_{50}$	0.1
Acute Endangered Species	EEC/LC$_{50}$ or EC$_{50}$	0.05
Chronic Risk	EEC/NOAEC	1
Terrestrial and Semi-Aquatic Plants, and Aquatic Plants		
Acute Risk	EEC/EC$_{25}$	1
Acute Endangered Species	EEC/EC$_{05}$ or NOAEC	1
Terrestrial Invertebrates		
Acute risk to bees	EEC (adult contact)/LD$_{50}$ (adult contact) EEC (adult or larval oral)/LD$_{50}$ (adult or larval oral)	0.4
Chronic risk to bees	EEC (adult contact)/NOAEC (adult contact) EEC (adult or larval oral)/NOAEC (adult or larval oral)	1

Appendix B. Identification of Pesticides of Concern Likely to End Up in POTWs

A number of indoor pesticide uses are considered to have lower potential for substantive releases to POTWs based on labeled uses. These include labeled applications of indoor foggers, baits, crack and crevice treatment, and bed and mattress treatments where a hydrological connection to sewer systems is considered highly unlikely or at most, rare. Considerable discussion arose around the use of 'spot on' treatments for pets (*e.g.*, flea and tick control) as well as insecticide-impregnated collars. With spot on treatments, it is expected (and advised on some pesticide labels) that shampooing soon after application of spot on treatments would reduce the efficacy of such treatments, and those would not be cost effective and discouraged. Regarding pet collars, the potential substantive releases to POTWs are considered low based on their expected slow release rate of pesticides from the collars.

It is important to note that the pesticide uses identified in **Table B-1** do not represent *all* potential sources of pesticide input to POTWs. Rather, they represent those uses that are currently being assessed as part of DtD modeling in OPP environmental risk assessments. For example, pesticides may potentially be released by industrial discharges to POTWs from pesticide manufacturers. However, such releases are subject to regulation under other environmental statutes and regulatory programs (*e.g.*, state and federal pretreatment programs under the authority of the Clean Water Act), and not under FIFRA. It is recognized that certain outdoor residential uses of pesticides may contribute to pesticide loadings to storm water systems which are connected to POTWs. Modeling of outdoor residential use of pesticides in OPP environmental assessments is presently focused on direct loadings to surface water. Information from the open literature suggests that some POTWs may experience greater flow during wet weather events even when direct connections to storm water inputs are not apparent (Weston and Lydy, 2010). Presumably, such inputs represent groundwater intrusion and/or fugitive inputs from storm water runoff. For these and other sources of pesticides to POTWs unaccounted for in **Table B-1**, OPP is relying on targeted monitoring data to ascertain inputs to and discharges from POTWs.

Table B-1. Identification of Pesticides of Concern Likely to End Up in POTW Effluents and/or Biosolids (EFED and AD Comparison Chart)

Description of Use	**EFED[1]**	**AD**
Products registered for indoor domestic residential uses, certain indoor commercial facility uses (*e.g.*, supermarkets, storage facilities, restaurants, and warehouses), and institutional uses (*e.g.*, hospitals)		
Pet lotions or shampoos (*e.g.*, treatment for fleas and ticks)	Y	n/a[2]
Pesticide-containing pet collars and spot-on treatments	N[A]	n/a[2]
Products for the treatment of shoes/clothing/textiles (*e.g.*, miticides, sanitizers, deodorizers)	Y	Y[3]
Pre-treated clothes/textiles, bed sheets, linens, etc.	Y	Y[3]
Bed and mattress treatments (except products to treat bed sheets)	N[B]	N[4]
Drain treatments that convey water to sanitary sewer systems (root herbicides)	Y	n/a[5]

Description of Use	EFED[1]	AD
Storm drain/storm system treatments (*e.g.*, root herbicides and filtration media for storm water filtration systems)	Y[6]	Y[6]
Sewage system treatments (*e.g.*, filtration media for municipal wastewater filtration)	Y[6]	Y[6]
Carpet treatments (except materials preservatives) removed from carpets during shampooing then subsequently disposed with wash water down-the-drain	Y	Y
Lice shampoos, skin lotion treatments (*e.g.*, for mites)	N[12]	n/a[2]
Toilet bowl cleaners	n/a[7]	Y
Shower, sink, and bathtub cleaning products	n/a[7]	Y
Sanitizers and deodorizers applied to sinks, showers, dishes, countertops, etc. that are subsequently rinsed down-the-drain	n/a[7]	Y[3]
Dishwashing liquids and detergents	n/a[7]	Y
Laundry detergents used in residences, commercial laundries, and institutional establishments (*e.g.*, hospitals)	n/a[7]	Y
Laundry detergents used in industrial laundries	n/a[7]	Y[8]
Products, such as insecticides, intended to treat wood, walls, floors, spot treatments, or crack and crevice	N[C]	N
Foggers	N[C]	N
Sprays (surface and space)	N[C]	Y[3]
Aerosols	N[C]	N
Antifouling products	n/a[7]	Y
Egg washing treatments and/or fruit and vegetable rinses	Y	Y
Products registered for outdoor domestic residential uses		
Pool, spa, and fountain treatments (some localities require draining to sanitary sewer)	Y	Y
Residential and domestic products intended for *outdoor* applications (except pool, spa, and fountain treatments)	N[D]	N[9]
Wood preservatives and stains (excludes industrial application during processing)	N	N[10]
Oil and gas well production process water / recovery	N	Y[11]

Footnotes:

[A] Pet collars and spot on treatments have a low potential for substantive release down the drain going to POTW.

[B] Bed mattress treatments do not lead to a complete exposure pathway down the drain.

[C] These uses will not lead to a substantial amount of chemical going down the drains and/or they do not represent complete exposure pathways.

[D] This use does not lead to direct down the drain exposure going to POTWs.

[1] If monitoring of POTW effluent shows that a pesticide is present, this may be an indication that an assessment may be required.

[2] Not an antimicrobial pesticide use pattern

[3] This type of end-use for an antimicrobial pesticide would typically be listed on a label along with several other end uses. As a screen to estimate potential exposures and risks to humans and aquatic organisms from releases of an antimicrobial pesticide from domestic wastewater treatment plants, the contribution from all of these end-uses would be considered rather than this end use alone. The total production volume for the antimicrobial pesticide would be used as an input to the DtD module of E-FAST and the resulting estimate of exposure would include, but not be limited to, this specific use.

[4] Although materials preservatives are used in mattresses, these chemicals might be released to industrial WWTPs during application, but would not be expected to be released to domestic WWTPs.

[5] No known antimicrobial use

[6] These types of uses can potentially lead to acute adverse effects to microorganisms at WWTPs immediately downstream of application sites. For these types of uses that are sporadic and localized, the down-the-drain

module of E-FAST will tend to underestimate concentrations of chemicals entering WWTPs immediately downstream of application sites and corresponding concentrations of chemicals in receiving waters downstream of these WWTPs.

[7] Not a conventional pesticide use pattern

[8] Releases from this use are expected to be predominantly to industrial WWTPs; the down-the-drain module of E-FAST is not appropriate for estimating exposures from releases to industrial WWTPs.

[9] AD does, however, examine releases during processing and industrial use for residential/domestic products intended for outdoor applications (*e.g.*, pressure-treated wood products)

[10] Releases from this use are to industrial WWTPs only. The down-the-drain module of E-FAST is not appropriate for estimating exposure from releases to industrial WWTPs. The appropriate module of E-FAST to use is the General Population Exposures from Industrial Releases module.

[11] Releases from this use can be to domestic or industrial WWTPs.

[12] These products are regulated by FDA, but if the active ingredient in these products is also a pesticidal active ingredient, the risk assessor should consider the pesticidal uses in the ecological risk assessment.

Appendix C. Example Output File for Permethrin

INITIAL REVIEW EXPOSURE REPORT

CHEMICAL ID: tmpcas

ENVIRONMENTAL RELEASES OF CHEMICALS IN HOUSEHOLD PRODUCTS

SCENARIO #: 1 EXPOSED POPULATION: Adult

WWT REMOVAL (%)	PRODUCTION VOLUME (kg/yr)	HOUSEHOLD RELEASE DAYS	PRETREATMENT RELEASE (g/person/day)	POST-TREATMENT RELEASE (g/person/day)	BODY WEIGHT (kg)	BCF (L/kg)
93.40	7710.00	365.00	7.26E-05	4.79E-06	80.00	0.00

High-end scenario

CONCENTRATION OF CONCERN (ug/L)	NUMBER OF DAYS CONCENTRATION OF CONCERN EXCEEDED	PERCENT OF YEAR CONCENTRATION OF CONCERN EXCEEDED
3.30E-04	326.61	89.48
3.30E-03	153.95	42.18
9.00E-04	268.48	73.56

10th PERCENTILE DRINKING WATER AND FISH INGESTION EXPOSURE ESTIMATES						
Exposure Units	Drinking Water Results	Fish Ingestion Results	ASSUMPTIONS			
			ED (years)	AT (years)	DW IR (L/day)	FISH IR (g/day)
Cancer						
$LADD_{pot}$	1.48E-08 mg/kg/day	N/A mg/kg/day	57.00	78.00	1.04	7.50
$LADC_{pot}$	1.14E-06 mg/L	N/A mg/kg	57.00	78.00	NA	NA
Acute						
ADR_{pot}	2.60E-07 mg/kg/day	N/A mg/kg/day	NA	1 day	3.03	279.00

50th PERCENTILE DRINKING WATER AND FISH INGESTION EXPOSURE ESTIMATES						
Exposure Units	Drinking Water Results	Fish Ingestion Results	ASSUMPTIONS			
			ED (years)	AT (years)	DW IR (L/day)	FISH IR (g/day)
Cancer						
$LADD_{pot}$	8.75E-10 mg/kg/day	N/A mg/kg/day	57.00	78.00	1.04	7.50
$LADC_{pot}$	6.70E-08 mg/L	N/A mg/kg	57.00	78.00	NA	NA
Acute						
ADR_{pot}	1.18E-08 mg/kg/day	N/A mg/kg/day	NA	1 day	3.03	279.00

50th PERCENTILE AQUATIC EXPOSURE ESTIMATES - SURFACE WATER				
DESCRIPTOR	Harmonic Mean	30Q5	7Q10	1Q10
STREAM DILUTION FACTOR	134.85	39.66	24.22	20.08
CONCENTRATION (µg/L)	9.16E-05	3.12E-04	5.10E-04	6.15E-04

10th PERCENTILE AQUATIC EXPOSURE ESTIMATES - SURFACE WATER				
DESCRIPTOR	Harmonic Mean	30Q5	7Q10	1Q10
STREAM DILUTION FACTOR	7.95	1.80	1.00	1.00
CONCENTRATION (µg/L)	1.55E-03	6.86E-03	1.24E-02	1.24E-02

Chapter 5

Occurrence and Sources of Pesticides to Urban Wastewater and the Environment

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Municipal wastewater has not been extensively examined as a pathway by which pesticides contaminate surface water, particularly relative to the well-recognized pathways of agricultural and urban runoff. A state-of-the-science review of the occurrence and fate of current-use pesticides in wastewater, both before and after treatment, indicates this pathway is significant and should not be overlooked. A comprehensive conceptual model is presented to establish all relevant pesticide-use patterns with the potential for both direct and indirect down-the-drain transport. Review of available studies from the United States indicates 42 pesticides in current use. While pesticides and pesticide degradates have been identified in wastewater, many more have never been examined in this matrix. Conventional wastewater treatment technologies are generally ineffective at removing pesticides from wastewater, with high removal efficiency only observed in the case of highly hydrophobic compounds, such as pyrethroids. Aquatic life reference values can be exceeded in undiluted effluents. For example, seven compounds, including three pyrethroids, carbaryl, fipronil and its sulfone degradate, and imidacloprid, were detected in treated wastewater effluent at levels exceeding U.S. Environmental Protection Agency (US EPA) aquatic life benchmarks for chronic exposure to invertebrates. Pesticides passing through wastewater treatment plants (WWTPs) merit prioritization for additional study to identify sources and appropriate pollution-prevention strategies. Two case studies, diazinon and chlorpyrifos in household pesticide products, and fipronil and imidacloprid in pet flea control products, highlight the importance of identifying neglected sources of environmental contamination via the wastewater pathway. Additional monitoring and modeling studies are needed to inform source control and prevention of undesirable alternative solutions.

Introduction

Pesticide pollution has long been recognized in agriculturally impacted surface waters. A growing body of work indicates pesticide pollution is common in urban waterways as well (*1–5*). This pollution has been directly linked to urban and agricultural runoff associated with rainfall (stormwater) and irrigation. There are abundant agricultural and urban runoff monitoring data, mechanistic field and laboratory transport studies, and robust modeling tools to predict the environmental fate of specific chemicals under various outdoor agricultural and urban application scenarios (*6–8*).

Much less is known about the occurrence of pesticides contained in treated municipal wastewater effluent discharging to surface waters. Unlike most urban or agricultural runoff, municipal wastewater is treated prior to discharge into receiving waters. Limited data exist on the efficacy of typical municipal wastewater treatment technologies for pesticide removal; however, available results suggest that these treatment processes — which were not designed to address trace chemical contaminants — are insufficient to reduce pesticide concentrations below aquatic toxicity thresholds (*9–11*).

Treated wastewater effluent continuously discharged into surface waters represents an ongoing source of contaminants recalcitrant to removal. Treated wastewater effluent can dominate flow in streams and rivers in arid regions, as well as in estuarine environments with limited hydrodynamic exchange with the ocean (*12*). An understanding of the relative contribution of pesticides in wastewater effluent is essential to developing suitable management strategies for total pesticide loading to surface water.

The goal of this chapter is to provide a state-of-the-science review of the occurrence and fate of pesticide active ingredients ("pesticides") in wastewater influent and in effluent discharged to surface waters that serve as habitat for aquatic life. We do this through: (1) presenting a robust conceptual model of pesticide uses ("use patterns") available for down-the-drain transport; (2) summarizing all available journal-published monitoring data for current-use pesticides in United States (U.S.) wastewater treatment plants (WWTPs) influent and effluent; (3) presenting case studies that detail significant pesticide pathways; and (4) identifying gaps in monitoring and specific use patterns where research efforts should be focused. Other WWTP emissions and products (e.g., biosolids, air emissions, recycled water) and other uses of treated effluent (e.g., for direct or indirect potable use) are acknowledged, but are beyond the scope of the monitoring data literature review provided.

This review focuses primarily on discharges to indoor drains that flow to separated municipal sewer systems designed to only carry indoor discharges; it does not address combined sewer systems that mix urban runoff with wastewater from indoor drains. While combined sewer systems are not uncommon in older urban areas of the U.S., most modern sanitary sewer systems do not provide significant drainage for urban runoff arising from precipitation events.

Pesticide-use patterns are strongly influenced by government regulation; therefore, the scope of this review was limited to the U.S., because of the relatively uniform regulatory structure in place. Of note, a significant proportion of U.S. monitoring data is from the state of California. For purposes of this review, we will not consider metals or antimicrobial pesticides (e.g., triclosan, triclocarban). Although there are pesticide products that contain metals as an active ingredient, additional nonpesticidal sources complicate the interpretation of available data. Similarly, antimicrobial active ingredients are present in products regulated as pesticides, as well as in personal care products regulated by agencies designed to protect human health. Compounds used both as pharmaceuticals and pesticides were also excluded, such as the blood thinner and rodenticide warfarin.

Regulatory Framework Relevant to Urban and Consumer Pesticide Applications

The U.S. Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA) requires that all pesticide products are registered by the U.S. Environmental Protection Agency (US EPA) and places controls on the sale and use of pesticides. FIFRA requires pesticide manufacturers to submit supporting studies to demonstrate the efficacy and safety of proposed products. The US EPA then reviews the environmental fate and potential risks of pesticide products. Following federal registration, additional supporting studies may be required prior to registration in any particular state.

The US EPA includes municipal wastewater ("down-the-drain") modeling as a part of its pesticide registration evaluation and periodic reviews (*13*). The current US EPA model framework would benefit from an improved understanding of which pesticide-use patterns result in down-the-drain transport. Furthermore, information on the fraction of pesticide applied that are dislodged and reach indoor drains via specific-use patterns would improve modeling capabilities.

Product labels evaluated and approved by US EPA during pesticide registration, specify use patterns and application requirements. Pesticide labels are considered enforceable and regulators have the authority to assess fines and penalties for pesticides not applied according to label directions. State and local authorities can implement additional mitigation measures to address off-site pesticide transport through professional applicator permit conditions or through regulations.

Unlike professional applications, consumer use of pesticides, though widespread, has relatively limited regulation. This has crucial implications for the composition of wastewater, as consumer applications often dominate the pesticide-use patterns most likely to result in down-the-drain pesticide transport. It is not practical in such cases to enforce or to instruct individual consumers on safe pesticide use, the more difficult source reduction approach must be used to prevent and mitigate wastewater pesticide contamination. Gaining a robust understanding of pesticide-use patterns that result in down-the-drain transport and relative source contribution is necessary to develop successful source reduction measures.

Another U.S. law, the Clean Water Act also requires that states implement enforceable effluent pollutant limits on wastewater dischargers, including WWTPs. In California, where much of the monitoring data were developed, the State and Regional Water Quality Control Boards develop and implement these limits. Pesticides in wastewater effluent have posed a significant regulatory challenge for California water-quality regulators, particularly after a study found pyrethroids in the effluents of 28 of 31 municipal WWTPs, in some cases at concentrations higher than US EPA aquatic life benchmarks (*10*). For example, the Central Valley Regional Water Quality Control Board developed an amendment of a water-quality control plan to address the occurrence of pyrethroids in the entire Central Valley basin, including contributions from WWTPs (*14*).

WWTPs are legally responsible for limiting chemicals discharged to the environment; however, local municipal agencies like WWTPs cannot regulate the sale and use of pesticides in their service areas. California's Department of Pesticide Regulation (DPR), in partnership with the US EPA, has the regulatory authority over use and sale of pesticides in the state. Collaborative efforts between DPR and WWTPs to generate useful data to inform regulatory decisions are well underway.

A Conceptual Model of Pathways by Which Pesticide Sources Enter Wastewater Systems

A comprehensive conceptual model elucidates the multiple sources and pathways by which pesticides can enter municipal wastewater (Figure 1). The model must consider the entire region drained by the sewer system, also known as the *sewershed*. Refined conceptual models specific to particular pesticides or product types can be used to identify key sources whose control would most effectively reduce levels of pesticides in wastewater and receiving waters. Such models also enable enhanced evaluation of pesticide products during the registration process (*6*).

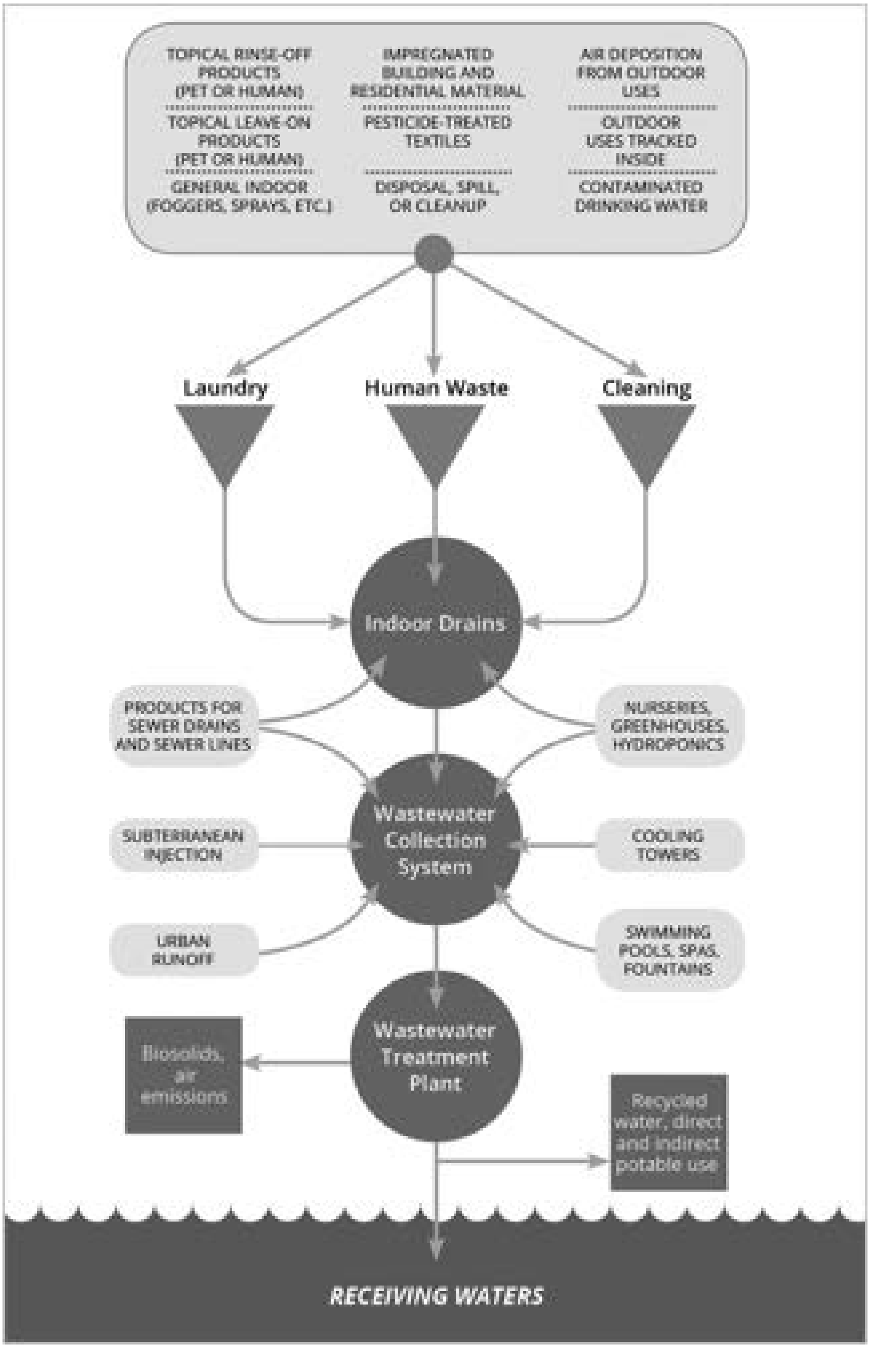

Figure 1. Conceptual model of sources of current-use pesticides to municipal wastewater. Black text is used to describe sources.

Readily identifiable and direct sources of pesticides to municipal wastewater are topical products intended to be rinsed down the drain, such as pesticidal shampoos. Examples for humans include over-the-counter shampoo treatments for lice (pediculicides) with pyrethrins or permethrin, or prescription-strength products with ivermectin, malathion, or spinosad. Examples for pets include flea and tick shampoos containing pyrethrins, permethrin, pyriproxyfen, and s-methoprene.

Other topical pesticide products may not be designed specifically for rinse-off application, but nevertheless enter municipal wastewater through bathing and cleaning activities. For example, after human dermal application of insect repellents containing *N*,*N*-diethyl-*m*-toluamide (DEET), the compound is washed from the skin while bathing and enters the municipal wastewater system. DEET has been widely detected in both wastewater influent and effluent (*15*).

Topical spot-on or spray pesticide products for flea and tick control are commonly applied to pets; pesticides include fipronil, imidacloprid, s-methoprene, pyriproxyfen, pyrethrins, permethrin and other pyrethroids, etofenprox, dinotefuran, indoxacarb, spinetoram, and selamectin (*16*–*18*). These pesticides enter municipal wastewater through multiple pathways including pet bathing (*19*), transfer to humans via petting (*20*–*25*) followed by washing and bathing; and transfer to pet bedding (*23*–*26*), interior surfaces, and house dust (*27*–*30*), followed by cleaning and laundering activities that result in down-the-drain discharges. Commercial pet grooming facilities are likely to discharge notable levels of pesticides from products used to treat pets (*19*).

Bathing, residential cleaning, and laundry activities are expected to result in pesticide discharge to municipal wastewater from a variety of other urban applications, including: (1) indoor pest-control products such as sprays, foggers, and crack and crevice treatments; (2) pesticide-impregnated construction and building materials; and (3) pesticide-treated clothing, pet bedding, and other textiles. Disposal of indoor-use pesticides, including improper cleanup of accidental spills by either professional applicators or consumers, likely results in larger sporadic discharges to wastewater. Commercial laundry facilities serving professional pesticide applicators or agricultural workers may also release larger loads of pesticides to the municipal sewer system.

Pesticides more generally associated with outdoor uses and urban runoff can also make their way into wastewater via transport indoors followed by washing, cleaning, and laundry activities. Pesticides in outdoor-use products can be tracked indoors via shoes, clothing, and skin (*27*, *31*), with higher levels observed for professional pesticide applicators and agricultural workers (*28*, *31*, *32*). Indoor contamination can also result from air deposition of volatile or spray pesticide applications from nearby outdoor settings (*33*).

Another potential indirect source of pesticides to wastewater is human waste contaminated via pesticide ingestion and via other indoor or occupational exposures. Some pesticides have been observed in human urine (*34*); however, due to lack of data, this indirect pathway is only suspected for other pesticides.

Contaminated drinking water can be a source of pesticides to municipal wastewater systems. Pesticides applied in the vicinity of both surface water and groundwater supplies can result in broad, low-level environmental contamination.

Because conventional drinking water treatment technologies were not designed to remove pesticides, these compounds may persist in finished drinking water. For example, recent studies in the U.S. have documented neonicotinoid insecticides (*35*) (clothianidin, imidacloprid, and thiamethoxam) and herbicides (*36*) (atrazine and metolachlor) in finished drinking water. While such findings have implications for human exposure to pesticides, they can also contribute to the presence of these compounds in wastewater.

Additional sources of pesticides to wastewater include herbicides designed to be flushed through sewer drains and sewer lines to kill roots penetrating pipes; products to control bacteria and algae in swimming pools, hot tubs, spas, and decorative fountains or water features draining to the municipal sewer system; specialized biocides used in cooling towers; insecticides and fungicides used in hydroponic cultivation, particularly for cannabis; and pesticides used at plant nurseries, including large chain retailers with nursery departments. More diffuse sources of pesticides traveling via urban stormwater runoff or subsurface flows can also infiltrate wastewater collection systems via cracks or leaks in sewer pipes, even when flows are not deliberately directed to sewers. Infiltration is suspected to provide an indirect, underground point of entry for other outdoor urban applications of pesticidal products (including injected termiticides). A sewer system's vulnerability to infiltration increases with deterioration of pipes, typically a function of infrastructure age.

All pesticides entering municipal wastewater collection systems are subjected to wastewater treatment. Conventional treatment technologies are designed primarily to handle human waste and food waste compounds present at relatively high concentrations, and often have limited efficacy in eliminating unique pesticide compounds present at nanogram/liter (ng/L) concentrations. Any contamination that does not partition to solids or degrade during treatment is discharged to receiving waters via treated wastewater effluent.

Monitoring data are sparse for many of the products or use patterns emphasized in this conceptual model. For example, many sources are associated with nonprofessional or consumer applications; unfortunately, consumer pesticide use practices are poorly characterized. Door-to-door surveys suggest widespread pesticide use in residences (*37*), and surveys of store shelves indicate ready access to an evolving array of pesticides in consumer-use products (*38*). Other sources of pesticides that are both poorly understood and may increase in use over time include those associated with construction and building materials, textiles such as clothing or mattresses, and hydroponic cannabis-growing operations. These gaps in understanding limit our ability to identify the most significant sources of pesticides found in wastewater.

Comprehensive Review of Available Current-Use Pesticide Influent and Effluent Data for the United States

Municipal wastewater has long been recognized as a pathway for discharge to receiving waters of contaminants derived from pharmaceuticals, personal care and cleaning formulations, and other consumer products; however, relatively few studies have evaluated this pathway for pesticides in current-use pesticides. This dearth of data is not surprising given that, prior to this publication, there has been no comprehensive conceptual model describing the potential pathways by which pesticides enter wastewater.

Presented here is a compilation of data from peer-reviewed publications describing the occurrence of current-use pesticides in influent and effluent (Table 1). The data compilation was limited to the U.S., and metals, antimicrobials, and pesticides also used as pharmaceuticals were excluded, as they may be derived from multiple additional sources not governed by pesticide regulation. Wastewater treatment processes vary from plant to plant. In this review we did not distinguish the type or level of treatment for specific monitoring results. In the U.S., municipal WWTPs utilize primary and secondary treatment at a minimum while advanced or tertiary treatment is common in densely populated city centers.

This extensive review of the scientific literature revealed wastewater influent and/or effluent detections for 20 insecticides and degradates, one insect repellent, 18 herbicides and degradates, two fungicides, and one wood preservative. The literature review found no detections for 39 additional pesticides and degradates. This review found information on a total of 81 pesticides in wastewater, which represents a small fraction of the hundreds of pesticides registered for use in the U.S. While information on a limited number of additional pesticide analytes may be available in grey literature, this does not alter the fact that there is a substantial shortage of data on current-use pesticides.

Some studies provide paired influent and effluent data that can be used to estimate removal efficiency of conventional wastewater treatment technologies. High levels of removal, 80–100% reductions observed following treatment, were only seen in studies of pyrethroids and high removals did not occur in all sampled WWTPs (*9*, *10*). This is not unexpected, as conventional wastewater treatment is focused on nutrient and pathogen removal, rather than removal or degradation of low levels of bioactive compounds with wide-ranging physicochemical properties. For some compounds, paired influent and effluent data are not available, preventing an estimate of removal efficiency.

Table 1. Occurrence of Pesticides in Wastewater Influent and Effluent in the U.S.

Pesticides & Degradates	*Inf./Eff.*	*Range (ng/L)*[a]	*Median (ng/L)*[b]	*DF (%)*	*No. of Samples*	*No. of Facilities*	*References*
2,4-D	Eff.	<100–1890	<100	3	102	52	(*39*)
2,4-DB	Eff.	<610–7440	<610	10	102	52	(*39*)
2,4-Dichlorophenol*	Eff.	<19–470	<19	62	102	52	(*39*)
Acetamiprid	Inf.	3–4.7	3.2	100	5	1	(*40*)
	Eff.	0.6–5.7	1.3–1.7	76	17	13	(*40*)
Acetamiprid-N-desmethyl*	Inf.	<0.6	<0.6	0	5	1	(*40*)
	Eff.	1.1–1.6	1.2	100	5	1	(*40*)
Acetochlor	Eff.	<0.89–240	1.3	61	38	3	(*41–43*)
Atrazine	Inf.	1–67	2–18.4	100	19	4	(*44–46*)
	Eff.	<7–390	<7–29	82	67	16	(*41–44*, *46*, *47*)
Bifenthrin	Inf.	<0.1–74	7.7–20.3	96	80	32	(*9*, *10*)
	Eff.	<0.1–14.1	<1–10.3	71	92	34	(*9*, *10*, *48*, *49*)
Carbaryl	Eff.	<0.49–663	<41	9	140	55	(*39*, *41–43*)
Chlorpropham	Eff.	<7.7–72.4	<7.7	3	102	52	(*39*)
Chlorpyrifos	Inf.	<1–81.9	15.2	85	13	1	(*9*)
	Eff.	<1–24.1	<1–3	40	30	5	(*9*, *41*, *42*, *49*)

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Table 1. (Continued). Occurrence of Pesticides in Wastewater Influent and Effluent in the U.S.

Pesticides & Degradates	*Inf./Eff.*	*Range (ng/L)*[a]	*Median (ng/L)*[b]	*DF (%)*	*No. of Samples*	*No. of Facilities*	*References*
Clothianidin	Inf.	<0.9–666	18	80	5	1	(*40*)
	Eff.	<0.9–347	12.5–45.3	47	17	13	(*40*)
Cyfluthrin	Inf.	<0.8–55	<1–8.85	74	80	32	(*9*, *10*)
	Eff.	<0.2–4	<1–0.3	42	90	34	(*9*, *10*, *49*)
Cypermethrin	Inf.	<0.8–200	18–27.3	99	80	32	(*9*, *10*)
	Eff.	<0.167–17	<1–1.3	56	90	34	(*9*, *10*, *49*)
DEET[c]	Inf.	413–42,300	413–10,100	100	18	4	(*44*, *45*, *50*)
	Eff.	<5–13,600	25–675	85	171	69	(*39*, *43*, *44*, *50–53*)
Deltamethrin	Inf.	<1.6–210[d]	<3.33	42	67	31	(*10*)
	Eff.	<0.2–2.7	<1	15	81	34	(*10*, *49*)
Diazinon	Eff.	<5–150	<5–38	64	25	22	(*41*, *42*, *47*, *51*)
Dicamba	Eff.	<300–760	<300	3	102	52	(*39*)
Dichlorprop	Eff.	<300–370	<300	1	102	52	(*39*)
Diuron	Eff.	<4–775	<4	46	102	52	(*39*)
Esfenvalerate	Inf.	<1.6–360[d]	<1.67–2.3	46	67	31	(*10*)
	Eff.	<0.167–3.7	<1	27	81	34	(*10*, *49*)
Fenpropathrin	Inf.	<0.8–130 [e]	<1.67	4	67	31	(*10*)

Pesticides & Degradates	*Inf./Eff.*	*Range (ng/L)*[a]	*Median (ng/L)*[b]	*DF (%)*	*No. of Samples*	*No. of Facilities*	*References*
	Eff.	<0.167–0.8	<1	2	81	34	(*10*, *49*)
Fipronil	Inf.	<20–146	30–70.5	66	41	33	(*11*, *54*)
	Eff.	<0.5–340	30–104	67	57	40	(*11*, *41*, *42*, *54*, *55*)
Fipronil amide*	Inf.	<0.3	<0.3	0	8	8	(*11*)
	Eff.	<0.3–19.8	1.25–6.7	95	21	13	(*11*, *55*)
Fipronil desulfinyl*	Inf.	<0.5–5.5	<0.8	19	16	8	(*11*)
	Eff.	<0.5–30.8	<0.8–9.4	56	32	15	(*11*, *41*, *42*, *55*)
Fipronil sulfide*	Inf.	<0.5–5.2	1.95–2.05	81	16	8	(*11*)
	Eff.	<0.5–52.2	<5–8.4	81	32	15	(*11*, *41*, *42*, *55*)
Fipronil sulfone*	Inf.	<0.5–31.2	8–23.1	94	16	8	(*11*)
	Eff.	<0.5–79.1	<5–30.7	88	32	15	(*11*, *41*, *42*, *55*)
Fluridone	Eff.	<7.7–27	<7.7	1	102	52	(*39*)
Glyphosate	Eff.	<100–2000	<100	27	11	10	(*47*)
Imazapyr	Eff.	<40–17,200	<40	9	102	52	(*39*)
Imidacloprid	Inf.	30–306	51.4–161	100	21	17	(*11*, *40*)
	Eff.	18.5–305	48.3–164	100	25	21	(*11*, *40*)[f]
Lambda-cyhalothrin	Inf.	<0.8–72	2.4–16	78	80	32	(*9*, *10*)

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Table 1. (Continued). Occurrence of Pesticides in Wastewater Influent and Effluent in the U.S.

Pesticides & Degradates	*Inf./Eff.*	*Range (ng/L)*[a]	*Median (ng/L)*[b]	*DF (%)*	*No. of Samples*	*No. of Facilities*	*References*
	Eff.	<0.167–5.5	<1	41	90	34	(*9*, *10*, *49*)
Mecoprop	Eff.	<0.28–72	4	80	35	1	(*43*)
Metolachlor	Eff.	<0.9–98	<6–75	74	38	3	(*41–43*)
Pentachloro-phenol	Eff.	<100–300	<100	2	102	52	(*39*)
Permethrin	Inf.	30–3800	180–315	100	80	32	(*9*, *10*)
	Eff.	<1–170	<1–21.4	64	90	34	(*9*, *10*, *49*)
Prometon	Eff.	<4–64	<10	4	105	54	(*39*, *41*, *42*)
Propiconazole	Eff.	<20–9020	<20	3	102	52	(*39*)
Simazine	Eff.	<4–56	<4	1	105	54	(*39*, *41*, *42*)
Terbuthylazine	Eff.	<4–61	<4	1	102	52	(*39*)

Pesticides & Degradates	*Inf./Eff.*	*Range (ng/L)*[a]	*Median (ng/L)*[b]	*DF (%)*	*No. of Samples*	*No. of Facilities*	*References*
Thiabendazole	Eff.	24–27	25.5	100	2	2	(*52*)
Triclopyr	Eff.	<300–3900	<300	11	102	52	(*39*)

Inf. = Influent; eff. = effluent. DF = detection frequency. MDL = method detection limit. * indicates compound is a degradate [a] If minimum is nondetect, the lowest MDL is reported. [b] Range of medians reported by the studies. [c] (*15*) conducted a broader review on DEET and reported a maximum concentration of 8480 and 14,000 ng/L, and a DF of 100% (sample size = 71) and 88.1% (sample size = 310) in influent and effluent, respectively, in wastewater treatment plants in the US. [d] The maximum concentration is substantially greater than the second largest value (29 and 29 ng/L for deltamethrin and esfenvalerate, respectively.) [e] There are three detections out of 67 samples: 360, 100, and 1.3 ng/L. [f] (*39*) sampled effluent from 52 WWTPs in Oregon and analyzed for imidacloprid. DF was 9.8% (10 out of 102 samples) at MDL=20 ng/L with a median (median of detections) of 237 ng/L and maximum of 387 ng/L. The study was not included in the table because the MDL was relatively high, which resulted in a considerably lower DF, compared to other studies. Pesticides analyzed but not detected [MDL, ng/L]: alachlor [5], azinphos-methyl [50], α-hexachlorohexane [5], benfluralin [10], butylate [4], carbofuran [20], cis-permethrin [6], cyanazine [18], dacthal [3], dieldrin [9], dinotefuran [32.6], disulfoton [20], EPTC [2], ethalfluralin [9], ethoprophos [5], fipronil desulfinyl amide [9], fonofos [3], linuron [35], malathion [27], metribuzin [6], molinate [2], napropamide [7], parathion [10], parathion-methyl [15], pebulate [4], pendimethalin [22], phorate [11], propachlor [10], propanil [11], propargite [20], propyzamide [4], tebuthiuron [16], terbacil [34], terbufos [17], thiacloprid [0.1], thiamethoxam [0.3], thiobencarb [5], tri-allate [2], trifluralin [9] (*40–42*).

Relative Ecotoxicity of Pesticides in Effluent

For those pesticides for which effluent monitoring data exist, compounds found at concentrations exceeding aquatic toxicity thresholds are typically prioritized for source identification and management action. The continuous discharge of treated municipal wastewater effluent containing pesticides at such levels suggests a potential for harm, particularly to sensitive aquatic species in highly impacted ecosystems, such as effluent-dominated streams.

Pesticides—particularly insecticides—in WWTP effluent can exceed aquatic toxicity based reference values. For example, observed WWTP pesticide effluent concentrations (Table 1) exceeded the following US EPA chronic invertebrate aquatic life pesticide benchmarks (*56*): the pyrethroids bifenthrin (1.3 ng/L), lambda-cyhalothrin (2 ng/L), and permethrin (1.4 ng/L); carbaryl (500 ng/L); fipronil (11 ng/L) and its degradate, fipronil sulfone (37 ng/L); and imidacloprid (10 ng/L). Other pesticides detected in effluent at levels above 50% of the lowest available US EPA aquatic life pesticide benchmark include: the pyrethroids cyfluthrin (7.4 ng/L) and deltamethrin (4.1 ng/L); chlorpyrifos (40 ng/L); diazinon (170 ng/L); and imazapyr (24,000 ng/L).

While identifying effluent pesticide levels exceeding reference values is useful for prioritization, this alone is not proof of harm. The potential for adverse impacts on aquatic species depends not only on discharged pesticide concentrations, but also on site-specific factors in the receiving waters. Such factors include: (1) dilution; (2) the presence of the pesticide in question in other discharges (e.g., urban stormwater runoff); (3) the presence of other contaminants that may cause additive, synergistic, or antagonistic effects (e.g., related pesticides and pharmaceuticals); and (4) the presence of substances that alter bioavailability or toxicity (e.g., dissolved organic carbon). Processes such as biodegradation and partitioning in receiving waters can also have long-term implications for the potential for adverse impacts to wildlife.

Gaps in available ecotoxicity data must also be acknowledged, as a lack of understanding of potential risks could lead to unexpected impacts. For example, relatively few studies of pesticide toxicity relevant to saltwater species and estuarine or marine receiving waters are available. Fewer ecotoxicity studies are available for pesticide degradates, metabolites, and transformation products (e.g., disinfection byproducts) relative to parent compounds. Additionally, few reference values (e.g., US EPA pesticide aquatic life benchmarks) have been developed to specifically address these compounds.

Nevertheless, the presence of a pesticide in effluent at levels exceeding reference values (e.g.,US EPA pesticide aquatic life benchmarks and other aquatic toxicity thresholds) signals the need for a closer examination of its sources, uses, and pathways to wastewater.

Case Studies Illustrating Use of WWTP Monitoring Data and Conceptual Models

Compound-specific conceptual models can guide targeted examinations of: (1) the relative quantities of the identified active ingredient in available pesticide products; (2) the pathways of transport relevant to these products; and (3) the relative contributions of different types of wastewater discharge to the sewer system, including residential and key commercial or industrial facilities. Two case studies that illustrate this approach can provide evidence to guide management actions designed to reduce the presence of pesticides in surface water.

Case Study: Diazinon and Chlorpyrifos

Toxicity testing in the late 1980s found that effluent from the Central Contra Costa Sanitary District WWTP (Martinez, California) was acutely toxic to *Ceriodaphnia dubia.* In accordance with the Clean Water Act and the California Porter Cologne Act, the San Francisco Regional Water Quality Control Board required toxicity identification evaluations (TIEs) to determine the cause of the toxicity. The TIE studies suggested that the combination of two organophosphate pesticides, diazinon and chlorpyrifos, was causing the effluent toxicity. At the time, these pesticides were commonly found in products available directly to consumers, including lawn and garden products, indoor pest control products, and flea and tick treatments for pets (*57*).

DPR partnered with Central Contra Costa Sanitary District to conduct wastewater sampling to better understand potential sources. Sampling included influent and subsewershed sites (i.e., residential areas and commercial locations). Commercial sampling focused on sites expected to introduce higher relative pesticide loads to the wastewater catchment, including pet groomers, kennels, and pest-control businesses. Diazinon and chlorpyrifos were detected in all 37 influent daily-composite samples, with mean values of 310 and 190 ng/L, respectively. Pesticide concentrations reported in residential sewage ranged from ND–4300 and ND–1200 ng/L for diazinon and chlorpyrifos, respectively. Commercial sampling locations contained the highest measured concentrations: 20,000 ng/L of diazinon in sewage from a kennel, and 38,000 ng/L of chlorpyrifos in sewage from a pet groomer.

Mass balance calculations determined that the overall mass contribution from residential sewage dominated the total pesticide mass entering the WWTP. Although the residential sewage concentrations were much lower, due to the higher residential flow rate, the residential contribution (82%) greatly exceeded the commercial contributions (6%) (*57*). This subsewershed study highlighted the need to understand pesticide sources, pathways, and relative contributions to establish a robust conceptual model and inform effective mitigation solutions.

As noted previously, the US EPA conducts registration reviews for actively-registered products. As a part of the re-registration review process in the early 2000s, concerns over human health arose for both pesticides. In 2000, registrants voluntarily agreed to terminate almost all indoor residential uses of chlorpyrifos in 2001, and all indoor residential uses of diazinon in 2002 (*58*, *59*).

Limited available long-term monitoring data suggest a general reduction in chlorpyrifos and diazinon WWTP influent concentrations as a result of this near complete phase-out of their indoor uses. Weston et al. (*9*). reported a median of 15.2 ng/L for chlorpyrifos in influent from another California WWTP sampled 2010–2012, representing an order of magnitude reduction from 1996 results. Similarly, the median diazinon influent concentration reported in a US EPA WWTP survey conducted in 2005–2008 was <10 ng/L (*60*). Conducting long-term monitoring in parallel with mitigation measure implementation would ensure that source control measures do indeed result in reduced chemical loading.

Of note, the data presented in Singhasemanon et al. (*57*) were not included in Table 1, as they primarily represent contributions from products no longer in current use. Current consumer insecticidal product replacements now typically contain active ingredients such as pyrethroids, and more recently fipronil and imidacloprid. Unfortunately, the use reduction of organophosphates has coincided with an increase in pyrethroid occurrence in wastewater influent. As noted previously, effluent levels of pyrethroids, as well as newer replacements fipronil and imidacloprid, now exceed US EPA aquatic life benchmarks.

Case Study: Fipronil and Imidacloprid in Pet Flea and Tick Treatments

To keep homes and pets free of fleas and ticks, treatment of dogs and cats with pesticides has been common for several decades. Shortly before the phase-out of most pet flea shampoos in the early 2000s, a new class of spot-on flea control products for pets entered the market. Fipronil and imidacloprid are common active ingredients in these popular topical products (*18*).

While occurrence data for both fipronil and its degradates (collectively fiproles) and imidacloprid in WWTP influent and effluent are sparse, these compounds are typically detected in available studies (Table 1). In one such study, the per capita influent loads for fiproles (54 ± 9 nmol/person/d, mean ± standard deviation) and imidacloprid (190 ± 80 nmol/person/d) for 7 Northern California WWTPs had low load variability, suggesting ubiquitous, low-level contributions from sources within the service areas (*11*). The authors outlined a conceptual model specific to fiproles and imidacloprid, that included all potential sources to wastewater, and the means by which pesticides derived from these sources might enter wastewater (*11*); these sources are a subset of those included within the comprehensive conceptual model provided in Figure 1.

Comparison of per capita pesticide loads in influent with active ingredient concentrations in individual pesticide applications suggested that widespread use of spot-on or spray flea control products might be the primary source of fiproles in wastewater (*11*). An estimate of influent fiprole load per fipronil-treated dog was found to be consistent with levels of the active ingredient in spot-on or spray products. Other potential sources, including use of crack-and-crevice treatments, outdoor pesticide applications tracked indoors, contaminated drinking water, and episodic discharges from spills, cleanup, or improper disposal, were found unlikely to be major contributors. The similarity of use patterns for imidacloprid suggested it was likely to be transported via comparable pathways (*11*).

Sadaria et al. (*11*) found multiple pathways by which fipronil and imidacloprid derived from flea control products can enter wastewater: (1) bathing of treated pets by professional groomers or pet owners in the home; (2) washing human hands contaminated via pet contact; (3) human waste following ingestion of trace levels of the pesticide as a result of pet contact; and (4) cleaning and laundering of residential surfaces, including pet bedding, that came into contact with pets or contaminated house dust. A subsequent study examined fiproles in rinsate from bathing fipronil-treated dogs 2, 7, or 28 days after treatment (*19*). Results confirmed pet bathing as a direct pathway of fiproles derived from spot-on products to municipal wastewater, with fiproles detected in 100% of samples and levels generally decreasing with increasing time from application (*19*). Additional calculations suggested washing 25% of fipronil-treated dogs in a service area within 7 days of treatment could account for the entire fiprole load of the sewershed, indicating spot-on products containing fipronil are likely to be an important fiprole source (*19*). While comparable data are not available for imidacloprid, the compound's higher solubility could result in significant wash off during pet bathing. In addition, targeted sampling of wastewater discharged from a pet-grooming operation confirmed the release of fipronil, pyrethroids, and imidacloprid to the wastewater catchment (*19*).

Additional evidence supports other pathways identified in the conceptual model. As noted previously, fipronil and imidacloprid in spot-on products can be readily transferred to humans via petting (*20*, *21*, *23*, *25*). Pesticides transferred to the hands of companions may enter wastewater via washing, or via unintentional ingestion followed by elimination. The human waste pathway is known to be relevant for imidacloprid, as it has been detected in human urine (*34*), but has not been investigated for fipronil (*61*).

Pesticide active ingredients in flea treatment also commonly appear in house dust. Fipronil and degradates were observed in nearly every sample of house dust examined in two studies of homes in Texas and California (*29*, *30*). While fipronil in house dust may also be derived from indoor- and outdoor-use products for non-flea pests such as ants, reported concentrations were more than 20 times higher in residences housing a dog treated with a spot-on fipronil product relative to those without treated pets (*29*). Imidacloprid was also detected in house dust from 32 of 38 California houses sampled (*30*).

Spot-on products containing each of these pesticides have also been observed to transfer to pet bedding (*23*, *26*). Cleaning and laundering are known to transfer contaminants associated with house dust and textiles to the wastewater system (*62*), and can be expected to transfer fipronil and imidacloprid as well.

Levels of these pesticides in wastewater before and after treatment indicate both fiproles and imidacloprid are relatively persistent, with little removal occurring via common WWTP treatment technologies (*11*, *40*, *54*). As noted previously, concentrations in effluent commonly exceed US EPA aquatic life benchmarks (*56*). Flea control products containing these pesticides may therefore pose risks to surface waters receiving discharges of municipal effluent, particularly when dilution of that effluent is limited.

Regional actions informed by these recent studies have already begun. The Bay Area Clean Water Agencies (BACWA), a joint powers authority that includes municipalities and special districts providing sanitary sewer services to more than 6.5 million people in the San Francisco Bay Area, has prioritized engagement with state and federal agencies to address the impacts of flea control pesticides, including providing comments to US EPA highlighting the need to include pet products in models used in pesticide risk assessment and regulation (*63*, *64*).

BACWA has distributed consumer education materials and findings from recent studies (*11*, *19*), which have also been highlighted via local media.

Priority Data Gaps

Available monitoring data, although sparse, highlight the need to address pesticide loading to surface water from WWTP effluent. Existing studies indicate that some pesticides (pyrethroids, fipronil, imidacloprid, and carbaryl) exceed aquatic life reference values and suggest the potential for harm to aquatic ecosystems, particularly to sensitive aquatic species in highly impacted ecosystems, such as effluent-dominated streams and estuaries. These and any other pesticides exceeding aquatic life reference values are high priorities for additional study to identify sources and appropriate pollution prevention strategies.

Developing strategies that continue to provide protection from pests while reducing overall pesticide loading will require a robust, quantitative understanding of use patterns and subsequent down-the-drain transport. Pesticide-specific customization of the comprehensive conceptual model (Figure 1) is an essential first step to build the knowledge to develop effective mitigation solutions. Refining this conceptual model for specific active ingredients can elucidate key data gaps, inform monitoring designs, and ultimately inform effective mitigation measures.

In the case of chlorpyrifos and diazinon, a conceptual understanding of potential sources based on registered uses led to a focused investigation of subsewershed contributions, characterizing sewage concentrations and loadings from residential and commercial sites (*57*). Study calculations to fill this data gap revealed low-level, ubiquitous residential sources to be of greater importance than large mass pulses (*57*). This case study illustrates how cooperative relationships between wastewater agencies and pesticide regulators are needed to ensure necessary data are obtained to inform potential mitigation.

A refined conceptual model (*11*) identified the need to confirm suggested contamination pathways, an important data gap in the case of fipronil- and imidacloprid-based flea and tick control. A study of the most direct contamination pathway (e.g., bathing treated animals in locations discharging to the sewer) suggested it is likely to provide significant mass transfer (*19*). However, presence of flea control active ingredients on pet bedding (*23*, *26*), pet owners (*20*, *21*, *23*, *25*), and house dust (*29*, *30*) indicate true source control at the site of application may be needed to significantly reduce down-the-drain transport.

Further WWTP influent and effluent monitoring is necessary to document occurrence or absence of additional as yet unexamined pesticides. More than 1000 pesticides are currently registered. The pesticide market continually shifts to adapt to changing needs and to produce alternatives to replace pesticides or product types most heavily scrutinized by federal and state regulators. Pesticides with the use patterns identified in the conceptual model, particularly those where parent compounds or degradates have relatively high aquatic toxicity, should be the highest priority for monitoring effluent discharged to surface waters that serve as habitat for aquatic life. Long-term monitoring to evaluate spatial and seasonal patterns and to track temporal trends resulting from mitigation or regulatory actions would fill additional data gaps for these prioritized pesticides.

There is also a need to identify and screen for degradates and metabolites of pesticides, including degradates formed during wastewater treatment (e.g., disinfection byproducts). The degradation products of some pesticides are known, but very few have been measured in WWTP influent and effluent. In some cases, degradate aquatic toxicity is comparable to, or greater than, the toxicity of the parent compounds. Identifying potentially harmful degradates is an area of intensive research that often utilizes high-resolution mass spectrometry to search for both known degradates and previously unidentified transformation products (*30*, *65*). However, these techniques may not be sufficiently sensitive to rule out the presence of pesticides at ppt levels.

Focused investigations of specific sources and sites within sewersheds are needed to better understand pesticide contributions from use patterns identified in the conceptual model. Several suspected high-use indoor pesticides sources are poorly understood and merit prioritization. For example, irrigation water from nursery operations discharging to wastewater collection systems (including stores that temporarily hold plants before sale) has received little study. Legalization of cannabis cultivation in many states may lead to an increase in hydroponic indoor growing systems and associated pesticide applications. Intensive use of pesticides such as for bed bug mitigation and subsequent cleaning activities is another identified data gap. While professional pest control operators are a highly-regulated group intimately familiar with pesticide handling requirements, the laundering of uniforms used during application is likely a concentrated source to wastewater. Similarly, the commercial laundering of uniforms for large groups (e.g., the military) whose clothing has come in contact with pesticides is likely to introduce large pulses of pesticides to sewer systems. Finally, to inform mitigation and predictive modeling of pesticide discharges, it is important to gain a better understanding of the fraction of certain pesticide uses that is dislodgeable and available for transport down the drain, including impregnated building and construction materials, foggers, and sprays.

Developing advanced engineering solutions to expand the capacity of wastewater treatment to reduce trace organic chemicals, present in the ppb to sub-ppt concentrations, has been an area of intense research over the past 20 to 30 years (*66*). However, due to the diverse chemical properties of pesticides, source control is more likely to provide financially feasible and effective mitigation, rather than implementing costly and potentially ineffective upgrades that add wastewater treatment technologies for removal of specific pesticides.

Enhanced understanding of compound-specific removal in wastewater treatment will improve our ability to prevent and manage risk. Available data provide some insights, but are too sparse to reflect the diverse design and operations of WWTPs. Use of additional or alternative treatment technologies, such as reverse osmosis or advanced oxidation, may also impact concentrations of pesticides and transformation products and such data can inform improved predictive modeling.

Addressing data gaps concerning pesticide wastewater treatment removal efficiency and incorporating this information into currently used risk evaluation modeling tools, such as the US EPA Exposure and Fate Assessment Screening Tool (E-FAST) (*13*), could inform development of effective mitigation solutions and could prevent future registration of products that pose a risk to surface water through down-the-drain transport. The E-FAST model relies on removal predictions based solely on physical and chemical properties rather than chemical-specific removal studies. This approach can introduce inaccuracies in modeling. For example, Parry and Young (*67*) measured the distribution of pyrethroids in secondary treated effluent and found additional settling time would not result in improved removal efficiency. The observed association between pyrethroids and dissolved organic matter present in wastewater may account for the over-predicted removal of pyrethroids by the E-FAST model (*68*). Predictive modeling must also recognize long-term trends, such as expected decreases in per capita water use which may result in increases in contaminant concentrations in influent.

Conclusion

Pesticide contamination of aquatic ecosystems occurs via WWTP effluent discharges as well as via agricultural and urban runoff. This state-of-the-science review of the occurrence of pesticides in wastewater derived primarily from indoor, down-the-drain inputs indicates that, for some pesticides, continuous discharges of WWTP effluent have the potential to adversely impact vulnerable aquatic biota. Protecting the quality of water resources that receive these effluent discharges is essential, particularly in regions with effluent-dominated streams, or embayments with limited hydrodynamic exchange with the ocean.

Addressing the data gaps identified in this review will improve the ability to prevent and manage these risks. The knowledge gained will not only allow for informed mitigation solutions, but also enhanced evaluation of pesticide products prior to registration and use. Pollution prevention is a key strategy to improve water quality for municipal wastewater pathways.

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Disclaimer

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REPORT

Results from the Swedish National Screening Programme 2008

Subreport 1. Biocides: 3-Iodo-2-propynyl butyl carbamate (IPBC) and 2,2-dibromo-2-cyanoacetamide (DBNPA)

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Per Wiklund, Eva Brorström-Lundén

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Results from the Swedish National Screening Programme 2008

Subreport 1. Biocides: 3-Iodo-2-propynyl butyl carbamate (IPBC) and 2,2-dibromo-2-cyanoacetamide (DBNPA)

Sammanfattning

IVL Swedish Environmental Research Institute has performed a “Screening Study” of IPBC and DBNPA as an assignment from the Swedish Environmental Protection Agency. IPBC belongs to the category preservatives/disinfectants and is used as a fungicide. It is used in the paper and pulp industry to prevent the formation of slime and as a preservative in cosmetics, paints, coatings etc. DBNPA is mostly used to reduce the occurrence of slime forming microorganisms with in the paper and cellulose industry. It is also used a biocide in cooling systems. IPBC is moderately soluble and not very persistent in water, hydrolysis is expected to be main route of dissipation. DBNPA has high water solubility, a short half-life and is rapidly degraded in water by hydrolysis. The overall objectives of the study were to determine the concentrations of IPBC and DBNPA in the Swedish environment. A sampling strategy was developed and the selection was based on the usage pattern of the substances. IPBC was found in air at two paint companies representing point sources and in the centre of Gothenburg, representing an urban diffuse source. In water, IPBC was found in high levels in the influent water at two paint companies and in both the in- and effluent water from three STPs. It was not found in background areas or in urban surface water, storm water and sediment, STP sludge or storm water sludge. The results indicate that if IPBC is being used, there is a possibility for the compound to be distributed to the environment since in can be detected in air and influent and effluent water from the paint industries and in STP waters. However, IPBC does not seem to reach other urban or background areas which may be due to the fact that IPBC is not very persistent in water. DBNPA was found below LOD in all samples.

Nyckelord samt ev. anknytning till geografiskt område eller näringsgren

IPBC, DBNPA, screening, point sources, background

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Sammanfattning

IVL Svenska Miljöinstitutet AB har på uppdrag av Naturvårdsverket genomfört en screening av 3-jod-2-propynyl butylkarbamat (IPBC) and 2,2-dibrom-2-cyanoacetamin (DBNPA).

IPBC tillhör kategorin konserveringsmedel/desinfektionsmedel och används som fungicid. Det används inom pappersmassaindustrin för att förhindra slembildning och som konserveringsmedel i kosmetika och färger. DBNPA används i störst utsträckning för att förhindra tillväxt av slembildande mikroorganismer inom pappers- och cellulosaindustrin. Ämnet används också som biocid i kylvattensystem inom industrin.

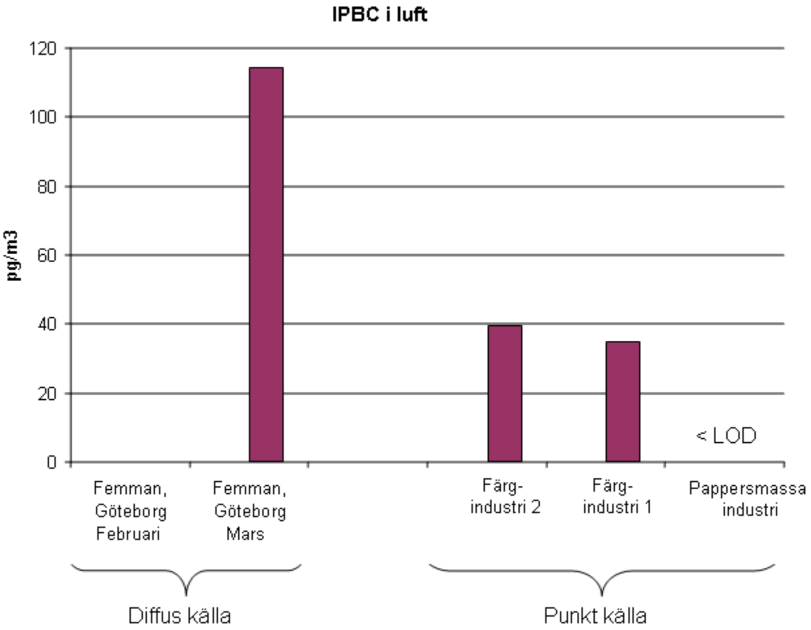
IPBC är måttligt lösligt i vatten och mindre persistent på grund av att föreningen kan genomgå hydrolys. Vattenlösligheten och det låga ångtrycket gör att fördelning till luft är mindre trolig. Spridning och retention av IPBCs i miljön beror på dess fysikaliska och kemiska egenskaper och inte på biotransformationsprocesser.

DBNPA har lågt log Kow och lågt ångtryck vilket ger föreningen hög vattenlöslighet. Ämnet har dock kort halveringstid i vatten och bryts snabbt ner genom hydrolys.

Syftet med föreliggande screening var att utreda förekomsten av IPBC och DBNPA i miljön och i vilken omfattning. En provtagningsstrategi utarbetades utifrån substansernas förutspådda emissionskällor och dess fördelning i miljön. De utvalda provtagningsplatserna representerar punktkällor så som färgindustri och pappersmassaindustri, diffusa källor i urban miljö, avloppsreningsverk samt bakgrundsområden. Proverna bestod av luft, ytvatten, sediment, jord, ingående och utgående avloppsvatten, avloppsslam, dagvatten och dagvattenslam.

IPBC återfanns i luft utanför två färgindustrier samt i centrala Göteborg, se figur IPBC i luft nedan. I ingående vatten till två färgindustriers interna reningsverk återfanns IPBC i höga koncentrationer, se figur IPBC i vatten från potentiella punktkällor. I det utgående vattnet var koncentrationerna betydligt lägre. I tre avloppsreningsverk hittades IPBC i både det utgående och ingående vattnet medan det var under detektionsgränsen i två reningsverk., se figur IPBC i vatten från avloppsreningsverk. IPBC kunde inte detekteras i fasta matriser så som jord, sediment eller slam oberoende om det var nära punkt eller diffus källa, eller i urbant vatten. Prover tagna från platser definierade som backgrundområden innehöll inte heller IPBC.

Resultaten visar att IPBC hittas i miljön där det används och skulle kunna ha en spridning vidare ut i miljön via luft och avloppsvatten. IPBC hittas dock inte i ytvatten från urban miljö.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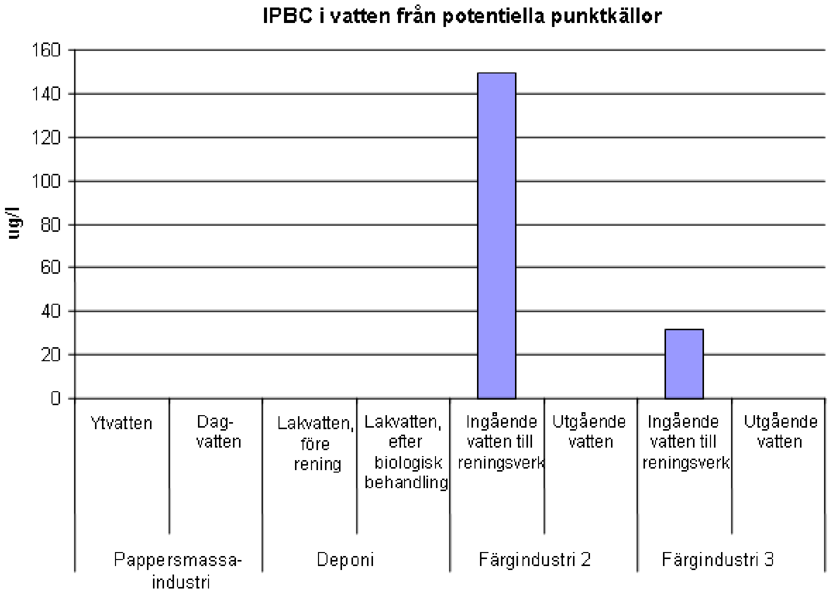
IPBC i vatten från potentiella punktkällor
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Dag-vatten
Lakvatten, före rening
Lakvatten, efter biologisk behandling
Ingående vatten till reningsverk
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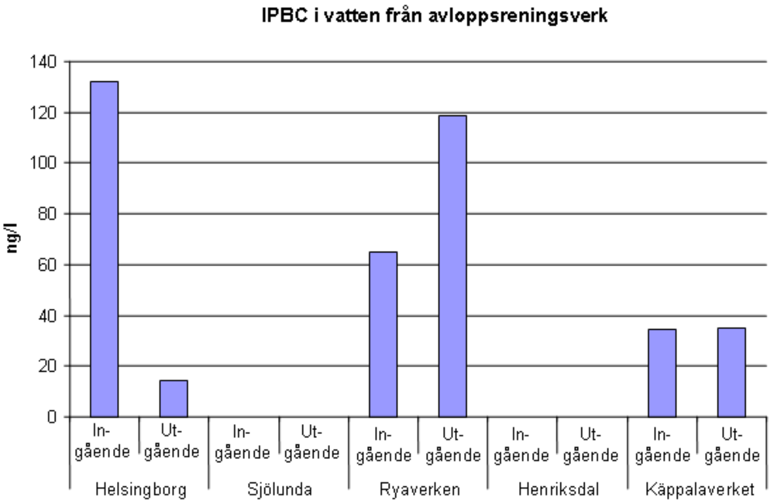


Halterna av DBNPA var under detektionsgränsen i samtliga prover vilket till största sannolikhet beror på den snabba nedbrytningen av föreningen.

Summary

As an assignment from the Swedish Environmental Protection Agency, IVL Swedish Environmental Research Institute has during 2008/2009 performed a "Screening Study" of 3-Iodo-2-propynyl butyl carbamate (IPBC) and 2,2-dibromo-2-cyanoacetamine (DBNPA).

IPBC belongs to the category preservatives/disinfectants and is used as a fungicide. It is used in the paper and pulp industry to prevent the formation of slime and as a preservative in cosmetics, paints, coatings etc. DBNPA is mostly used to reduce the occurrence of slime forming microorganisms with in the paper and cellulose industry. It is also used a biocide in cooling systems.

IPBC is moderately soluble and not persistent in water, hydrolysis is expected to be the main route of dissipation. The water solubility and low vapor pressure of IPBC make the compound not likely to partition to the atmosphere or undergo long range transport or deposition. The fate of IPBC in receiving waters is dependent on physicochemical processes rather than biotransformation.

DBNPA has low vapor pressure and high water solubility which make the compound to be retained mainly in the water phase. However, DBNPA has a short half-life and is rapidly degraded in water by hydrolysis.

The overall objectives of this screening study were to determine the concentrations of IPBC and DBNPA in the Swedish environment and to assess the possibility of current emissions. A sampling strategy was developed in order to determine concentrations of these compounds in different matrices. Selected sampling sites represent point sources in urban environments, diffuse sources and background areas.

IPBC was found in air at two paint companies representing point sources and in the centre of Gothenburg, representing an urban diffuse source. In water, IPBC was found at high levels in the influent water to treatment plants at two paint companies and in both influent and effluent water from three STPs. It was not found in background areas or in urban surface water, storm water sediment, STP sludge or storm water sludge.

The results indicate that if IPBC is being used, and that there is a possibility for the compound to be distributed to the environment since in can be detected in air and influent and effluent water from treatment plants at paint industries and in STP waters. However, IPBC does not seem to reach urban or background areas.

DBNPA was not found in any of the samples. This is probably due to the rapid degradation of the compound.

Table of contents

1 Introduction

As an assignment from the Swedish Environmental Protection Agency, a screening study has been performed by IVL during 2008/2009. This screening includes biocides, unintentionally produced substances and fuel additives. These substances/substance groups are emitted and distributed in the environment via a variety of sources, e.g. different point sources and/or diffusive sources. Table 1 shows the major reason for their concern as well as the number of the report where individual results are presented.

Table 1. Substances / substance groups included in the screening.

Substance / Substance group		Banned/ restricted	HPV^{a}	Indications of toxicity	B/P^{b}	Sub-report #
Biocides	**3-Iodo-2-propynyl butyl carbamate (IPBC) 2,2-Dibromo-2-cyanoacetamide (DBNPA)**			**x**		**1**
	Glutaraldehyde		x	x		2
	Difenacoum	x		x	x	3
Unintentionally produced substances	Nitro-PAH 3-Nitrobezantron Oxy-PAH Heterocyclics Brominated dioxins and aromatics			x	x	4
Fuel additives	Methyl *tert*-butyl ether (MTBE) Ethyl *tert*-butyl ether (ETBE)		x	x	x	5

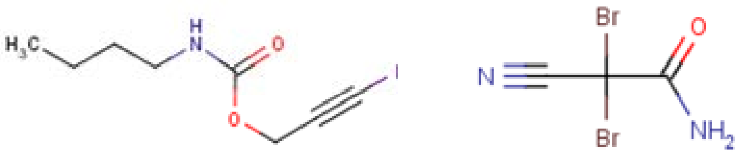
$^{a)}$ High Production Volume
$^{b)}$ Bioaccumulation/Persistence

The overall objectives of the screening studies are to determine the concentrations of the selected substances in a variety of media in the Swedish environment, to highlight important transport pathways, and to assess the possibility of current emissions in Sweden.

Due to the variety in emission sources and use as well as differences in chemical properties, the screening has been carried out in five subprojects. This sub-report concerns the screening of **IPBC and DBNPA**. Results for the other chemicals are presented in subreport 2, 3, 4 and 5.

2 Chemical properties, fate and toxicity

Figure 1 shows the chemical structure of IPBC and DBNPA and Table 2 shows the names, CAS numbers and some chemical and physical data of the compounds. The chemical and physical data of IPBC and DBNPA are gathered from ChemIDplus.



IPBC DBNPA

Figure 1. Chemical structure of 3-Iodo-2-propynyl bytylcarbamate (IPBC) and 2,2-Dibromo-2-cyanoacetamide (DBNPA).

Table 2. Chemical and physical data of 3-Iodo-2-propynylbytylcarbamate (IPBC) and 2,2-Dibromo-2-cyanoacetamide (DBNPA).

Name	CAS#	MW (g/mol)	Melting Point (˚C)	Log Kow	Water Solubility (mg/L)	Vapor Pressure
3-Iodo-2-propynyl bytylcarbamate (IPBC)	55406-53-6	281	64-66.5	2,8	156 at 20 ˚C	9.96E-8 Atm at 20 ˚C
2,2-Dibromo-2-cyanoacetamide (DBNPA)	10222-01-2	240	124.5	0.82	15 000 at 20 ˚C	9.00E-4 mm Hg at 20 ˚C

2.1 Properties and fate

IPBC belongs to the category preservatives/disinfectants and is used as a fungicide. It is also used in the paper and pulp industry to prevent the formation of slime and as a preservative in cosmetics, paints, coatings etc. DBNPA is mostly used to reduce the occurrence of slime forming micro-organisms within the paper and cellulose industry. It is also used as a biocide in cooling systems.

IPBC is moderately soluble in water, and is not likely to adsorb to suspended solids or sediment, it is more likely to remain in the dissolved phase (Juergensen et al. 2000). It is not persistent in the water column; hydrolysis is expected to be the main route of dissipation. IPBC is not expected to bioaccumulate and has a bioconcentration factor of <4.5 (Juergensen et al. 2000). The water solubility and low vapor pressure of IPBC makes the compound not likely to partition to the atmosphere or undergo long range transport or deposition. The fate of IPBC in receiving waters is dependent on physicochemical processes rather than biotransformation. The persistence of IBPC in natural water was estimated to be less than seven days (Juergensen et al. 2000).

For DBNPA, the low vapor pressure and high water solubility should make the compound to retain mainly in the water phase. Therefore there is a possibility for the compound to be distributed to the environment and water recipient and less probably be distributed to soils and sediments. However, the short half life in water due to hydrolysis is probably the main factor that determines the residence time in water and the distribution of it from the emission source. The half –life of DBNPA in water has been set to a mean of 0.5 h when used in model calculations for risk assessment (Klaine et al. 1996).

In water DBNPA is rapidly hydrolysed to dibromoacetonitrile (Blanchard et al. 1987). The presence of organic material in the water leads to a second pathway where monobromonitrilopropionamide is being formed.

2.2 Toxicity

IPBC is listed as one of the most commonly used industrial antifungal agents according to "World Guide to Industrial Biocides", as described by Nakashima et al. 2000. It has been detected in metalworking fluids (Henricks-Eckerman et al. 2008) and a few cases of contact allergy to it have been reported (Bryld et al. 1997, Majoie et al. 2000).

IPBC is potentially toxic to nontarget organisms in the aquatic environment arising from accidental release or leaching of IPBC-containing formulations. IPBC has been reported to affect fathead minnows at levels of 0.019 mg/L and *Daphnia magna* at levels of 0.07 mg/L (Juergensen et al. 2000). More acute and chronic toxicity for fresh water fish and invertebrates are reviewed and summarized in Juergensen et al 2000. Information regarding the environmental toxicology, chemistry and fate of IPBC in the Canadian environment has been gathered and used to develop an interim Canadian guideline for the protection of fresh water life for IPBC and al level of 1.9 μg/L is recommended (Juergensen et al. 2000).

Due to a short half life of DBNPA in aquatic environments, it has been concluded that it does not present a significant risk to aquatic ecosystems from possible exposure from industrial cooling systems where DBNPA is being used (Klaine et al. 1996).

A 48-h EC50 of 0.96 mg/L for *Daphnia magna* for DBNPA was referred to in Rigol et al. 2004 where a bioluminicence inhibition test of the bacteria *Vibrio fisheri* gave a EC50 of 0.50 mg/L for DBNPA (Rigol et al. 2004).

3 Production, use and emission

IPBC is imported to Sweden as a fungicide and is used in the production of paint and lacquer and in work up fluids in the metal industry as a preservative (www.kemi.se). It occurs in 512 preparations in Sweden in 2007 and 93 tonnes were used in the same year (SPIN 2009). IPBC is one of the most commonly present allergens in shampoos in the U.S (Zirwas and Moennich 2009).

IPBC may enter the environment through leachate or storm water runoff from freshly treated lumber (Juergensen et al. 2000).

The main use of DBNPA in the U.S. is as a biocide in cooling systems, which also represents the greatest potential for environmental exposure. DBNPA has been manufactured and sold by The Dow Chemical Company since 1973, the greatest producer of the compound in the U.S. In Sweden, DBNPA occurs in 41 preparations and 149 tonnes was used per year in 2007 (SPIN 2009). It is used as a biocide in the manufacture of pulp, paper and paper products in Sweden.

4 Previous measurements in the environment

The information of the detection of IPBC in the environment is very limited. IPBC is thought to have moderate potential for movement through soil, groundwater contamination may be possible. In contrast, it has been suggested that IPBC should not threaten groundwater because it appears to rapidly degrade in both aquatic and terrestrial environments (Juergensen et al. 2000). However, IPBC was detected in sediments close to lumber mills using IPBC formulations in the range of 0.19-0.49 µg/g dry weight (Juergensen et al. 2000). With a log K_{ow} of 2.81 and a bioconcentration factor of < 4.5, IBPC has a low potential for accumulation and it has not been found in biota.

DBNPA has been analysed in waters from a recycling paper mill that uses biocides for the improvement of the water quality (Rigol et al. 2002, Rigol et al. 2004). The concentration of DBNPA in the process water was 59 µg/L and was 4.2 µg/L in the outlet primary treatment water (Rigol et al. 2004).

5 Sampling strategy and study sites

5.1 Screening program

A sampling strategy was developed in order to determine concentrations of IPBC and DBNPA in different matrices in the Swedish environment.

The sampling programme was based on the identification of possible sources due to the use and the possibility for the compounds to distribute in the environment. The sampling programme is summarized in Table 3 for IPBC and in Table 4 for DBNPA. Details on the samples can be found in Table A1 and A3 in the Appendix.

For IPBC air samples were collected at background areas and urban areas. Samples were also collected at three different paint production industries, i.e. possible point sources due to the use of the compound. Different types of water samples (influent, effluent, leachate and storm water) as well as sediment were also sampled at the point sources. The point sources consist of samples from the paper and pulp industry, one landfill and paint companies. Samples from diffuse sources were collected in the cities of Gothenburg and Stockholm were air, surface water, soil, sediment and storm water samples were collected. From Sewage treatment plants (STP), both influent and effluent water were sampled as well as sludge.

DBNPA has a similar sampling programme as IPBC. Due to the physical and chemical properties, the most relevant matrices for DBNPA are water (surface water, process water and STP influent and effluent water). The short half life of the compound makes point sources where DBNPA is being used of great interest. Several different water samples have been collected at a paper and pulp industry.

In order to determine background levels, samples of soil, sediment, surface water and fish were analysed from three background lakes, classified as reference lakes by the Swedish Museum of National History.

Due to the low log Kow of the compounds and especially DBNPA, i.e. they will not bioaccumulate, biota samples were not considered in the program.

Table 3. Sampling program for IPBC

Site	Air	Surface water	Sedi-ment	Soil	Storm water	Storm water sludge	Industry water	STP Water (in)	STP water (out)	STP sludge	Leach-ate	Total
Background												
Råö	2											2
Lakes		3	2									2
Diffuse sources												
Urban area	2	3	3	2	5	3						18
STP								5	5	5		15
Point sources												
Paper & pulp industry	1	1		2	1							5
Paint production industry	2						4					6
Landfill											2	2
Total	7	7	5	4	6	3	4	5	5	5	2	53

Table 4. Sampling program for DBNPA

Site	Air	Surface water	Sedi-ment	Soil	Storm water	Storm water sludge	Industry water	STP Water (in)	STP water (out)	STP sludge	Leach-ate	Total
Background												
Råö	2											2
Lakes		3	2									5
Diffuse sources												
Urban area	2	3	3	2	5	3						18
STP								5	5	5		15
Point sources												
Paper & pulp industry	1	1			1		2					7
Landfill											2	2
Total	5	7	5	4	6	3	2	5	5	5	2	49

6 Methods

6.1 Sampling

The staff at the different sewage treatment plants collected de-watered **sludg**e samples from the anaerobic chambers. The sludge was transferred into glass jars and stored in a freezer (-20 °C) until analysed. **Influent** and **effluent** waters were sampled in 1 or 2 L glass bottles.

Surface **sediment** (0-2 cm) samples were collected by means of a Kajak sampler. The sediment was transferred into glass jars and stored in a freezer (-20 °C) until analysed. One sediment sample from the background lake Tärnan was provided from the specimen bank at the Swedish Museum of Natural History.

Surface water samples from background lakes and from the city of Stockholm were sampled in glass bottles and the pH was adjusted to pH 2 with H_3PO_4 and stored in at 6 °C.

The upper 2-3 cm of surface **soil** was collected in glass jars.

Storm water samples were provided by the Swedish Road Administration (Vägverket) and “Gatukontoret” in Göteborg.

Air samples were collected using a low volume air sampler (LVS) with a flow of approximately 1 m^3/h. The air was passed through a glass fiber filter (MG160, Munktell) where the particles were collected and then through a glass column packed with the adsorbents XAD-2 (Amberlite) and polyurethane foam (PUF). Sampling duration was one week at the point sources and two weeks at the background station for IPBC. For DBNPA, the sampling duration was 2-3 days at the background station and 1-2 days at the point sources. Prior to sampling, glass fiber filters were heated to 400 °C, and the adsorbents columns were cleaned by Soxhlet extraction with acetone. After sampling, the filter and columns were wrapped in aluminum foil and send to the laboratory, where they were stored in a freezer (-20 °C) until analysed. Additional filters and columns used as field blanks were sent back to the laboratory unexposed.

All glass equipment used was muffled (400 °C) before use.

6.2 Sample preparation

6.2.1 Solid samples

IPBC and DBNPA were extracted from the solid samples simultaneously. Sludge (0.5 g), storm water sludge (0.5 g), sediment (1.5-2 g) and soil (1.5-2 g) were ground with Na_2SO_4 and extracted twice with 30 ml acetone by one hour rotation. The volumes were reduced and the solvent was exchanged to methanol (5 ml) before HPLC-MS/MS analysis.

6.2.2 Water samples

IPBC and DBNPA were extracted from the water samples simultaneously. An Oasis HLB SPE column (200 mg, Waters) was activated with methanol and acidic water before the water was extracted. The trapped analytes were then eluted with 5 ml acetone and the solvent was exchanged to methanol. The final sample volume was 1 ml.

Volumes used: surface water: 400 ml, storm water 100 mL, influent water from sewage treatment plant: 100 ml, effluent water from sewage treatment plant: 200 ml, and leachate water 100 mL.

6.2.3 Air samples

IPBC and DBNPA were extracted from the adsorbent using 100 ml of methanol. The solvent volume was reduced to 2 ml by a rotary evaporator.

6.3 Instrumentation

The extracts were analyzed applying high performance liquid chromatography using a Prominence UFLC system (Shimadzu) with two pumps LC 20AD, degasser DGU-20A5, auto sampler SIL-20ACHT and column oven CTO-20AC. The analytical column was a Thermo HyPurity C8 50 mm x 3 mm, particle size 5 µm (Dalco Chromtech). The column temperature was 30 °C. For analysis, 10 µl sample extract in methanol were injected.

IPBC: Mobile phase A was 2 mM ammonium acetate in water. Solvent B was methanol, adopted from Henriks-Eckerman et al. (2008). The flow rate of the mobile phase was 0.4 ml/min. A gradient elution was performed: 0-2 min 30% B, 2-10 min linear increase to 95% B, 10-14 min isocratic 95% B, 14-16 min linear decrease to 30% B. Equilibration time. 5 min.

DBNPA: Mobile phase A was 0.1% formic acid in water. Solvent B was methanol. The flow rate of the mobile phase was 0.4 ml/min. A gradient elution was performed: 0-3 min 10% B, 3-6 min linear increase to 95% B, 6-10 min isocratic 95% B, 10-12 min linear decrease to 10% B. Equilibration time. 7 min.

The effluent was directed to an API 4000 triple quadrupole mass spectrometer (Applied Biosystems).

For the analysis of IPBC electrospray ionisation (ESI) in negative mode was used. Two MRM transitions according to Table 5 was recorded. Calibration was done using standard curves based on areas vs. known concentrations of the actual compound. The sample extracts were adjusted to known volumes. No internal standards were used since it was not possible to find representative compounds.

Table 5. Instrumental parameters for determination of IPBC.

	Precursor ion m/z	Product ion m/z	Declustering potential, V	Collosion energy, V
MRM1	282	165	52	27
MRM2	165	127	100	50

For DBNPA it was not possible to obtain stable MRM signals. Probably the compound is thermally degraded in the ion source. Instead the ions m/e 79 and 81 (bromine) were selected by both quadrupoles in the instrument .

6.4 Quality control

6.4.1 IPBC

Detection was done using two independent MRM transitions. MRM 282/165 was used for quantification and MRM 165/127 was used as qualifier. Also, the retention time should match the authentic standard compound within ± 0.1 min.

For each matrix, two solvent method blanks were prepared in parallel with the samples to assess possible interferences and contamination from the background.

The background contamination in the blank samples was subtracted from the measured sample values. Limit of detection (LOD) was defined as three times the standard deviation of the noise of the blank water samples and adjusted for the different volumes used In the solid matrixes LOD was calculated as ten times the blank level and for the air samples as three times the level in the field blank.

6.4.2 DBNPA

The relationship for the two bromide isotopes 79 and 81 is 1:1. A standard curve was prepared and showed a linear relationship between 1000 ng/ml and 25 ng/ml. In this range the relationship between the two isotopes are 1:1 but divides from this at concentrations lower than 25 ng/ml. Limit of detection (LOD) was calculated from this value, i.e. 25 ng/ml was multiplied with the final volumes for the sample extracts and then divided with the sample amount.. It was not possible to calculate a LOD value from noise in the solvent blank samples.

Samples with a strong signal for m/z 81 were also analysed on GC-MS/MS in the full scan mode to compare the spectra with an authentic standard.

For each matrix, solvent method blanks were prepared in parallel with the samples to assess possible interferences and contamination from the background.

7 Results and discussion

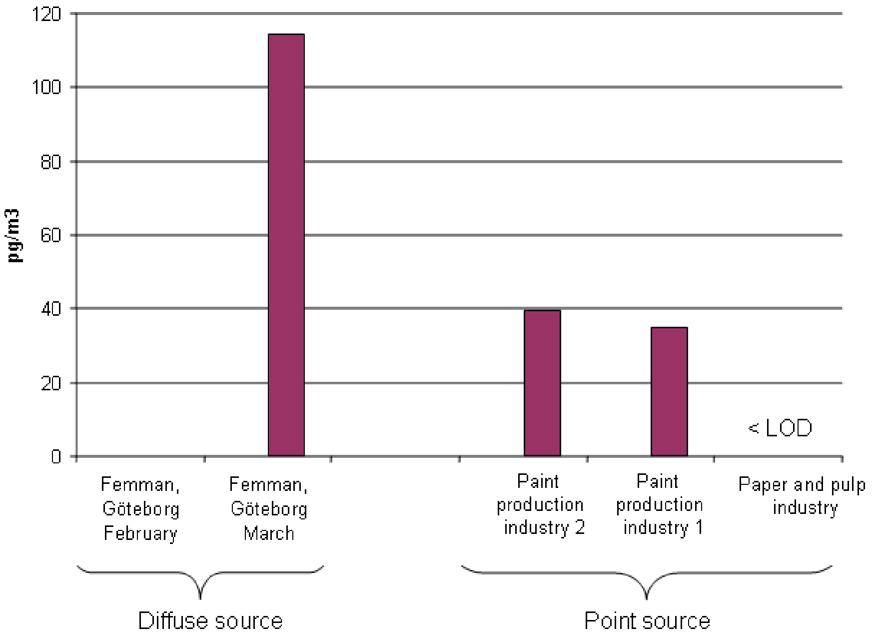
7.1 IBPC

IPBC could be found in one air sample representing an urban diffuse source and in two air samples from potential point sources. In water, IPBC was found in several samples at both point sources and diffuse STP sources. It was not found in background areas or in urban surface water, storm water, soil and sediment, STP sludge or storm water sludge. The results from in air and water are presented in figures below. All results and LOD can be found in Table A2 in the Appendix.

7.1.1 Air

IPBC was found in air from both diffuse and point sources, see Figure 2. Femman is a mall in the centre of Gothenburg. The air sampler was up in two periods, following each other directly. At the first sampling occasion, in the end of February, the concentration was below LOD and at the next occasion in the beginning of March, the concentration was 110 pg/m^3. This may be due to that work was being performed at Femman were products were used that contained IPBC. This also shows that the detection of IPBC in the environment may be a coincidence due to the sampling occasion.

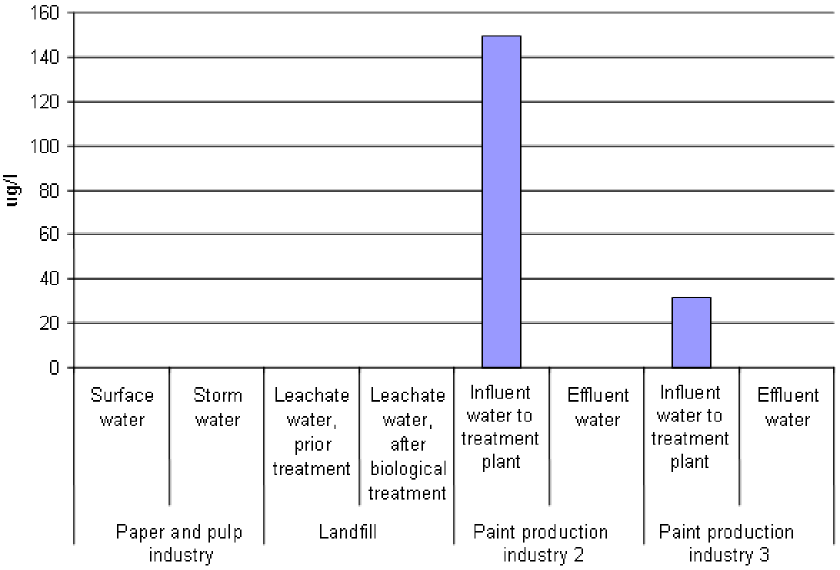
IPBC is a component in paint and IPBC was found at in the air both Paint production industry 1 and 2. IPBC is also used in the paper industry but the concentration where below LOD in the air sample from the paper and pulp industry in this study. It could be due to that IPBC is not being used at this company or that it was not detectable at the sampling occasion.



Figur 2. Concentrations of IPBC (pg/m^3) in air from a diffuse urban area and from point sources, i.e. paint industry and paper and pulp industry.

7.1.2 Water, Point sources

IPBC was detected at high concentrations, 150 and 32 ug/l respectively, in the influent waters to the treatment plants at two paint industries, see Figure 3. The effluent waters from these companies also contained IPBC but at much lower levels, 0.032 and 0.22 ug/l. IBPC was not detected in surface or storm water at the other potential point source, a land fill. The land fill, contained 0.035 ug IPBC/l in the water prior to treatment.

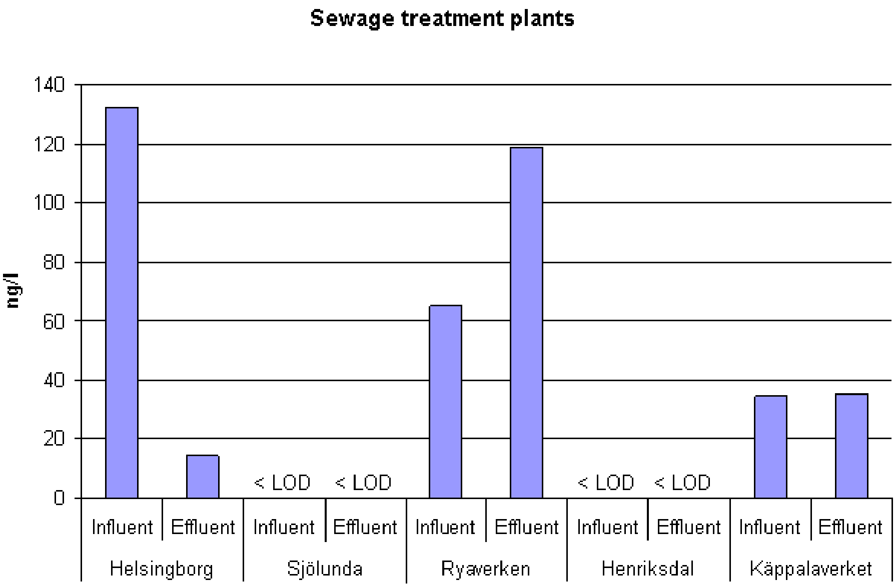


Figur 3. IPBC concentrations (ug/l) in influent and effluent water at treatment plants at paint industries, paper and pulp industry and from a landfill (recycling company).

7.1.3 Water, Diffuse sources

IPBC was found in both influent and effluent waters at three STPs, Helsingborg, Ryaverket and Käppalaverket, see Figure 4. Only at Helsingborg there was a trend with lower concentrations in the effluent water compared to the influent water. At Sjölunda and Henriksdal, the concentrations were below LOD. The effluent water from Paint production industry 3 is connected to Sjölunda and with respect to dilution and hydrolysis, it seem reasonable that the concentrations at Sjölunda is below LOD. However, IPBC was found in similar or higher levels in the effluent water compared to the influent water at Käppalaverket and Ryaverket. Therefore it is a possibility that IPBC may enter the water after treatment. Also, the detection in the effluent water enhances the risk for IPBC to reach the environment.

The Canadian guideline for the protection of fresh water life is 1.9 ug/L (Jueregensen et al 2000) and the effluent water at the STPs where IPBC could be detected is at least 14 times lower.



Figur 4. Concentrations of IPBC (ug/l) in STP influent and effluent water.

7.2 DBNPA

DBNPA was not be found in any of the samples. In the environment DBNPA is rapidly degraded primarily through hydrolysis but it may also be degraded during the analytical procedure.

All samples contained a peak, often with an asymmetric shape, at the expected retention time. However, the relationship between the bromide isotopes 79 and 81 was far from 1:1 in all samples. Due to this, the peaks were considered NOT to be DBNPA. The calibration curve showed that sample extracts must contain levels above 25 ng/ml to ensure the identity of the compound. When MRM is used with in mass spectrometry it is desirable to use an ion pair with masses from the parent compound and a fragment that is specific for the compound. This was not possible to do for DBNPA since it is too instable to pass the analytical instrument with out being degraded, therefore only m/z 79 and m/z 81 could be used. The LODs for the different matrices, calculated from 25 ng/ml as described in section 6.4.2, are summarized in Table 6. Details for respective sample can be found in Table A3 in the Appendix.

Table 6. Limit of detection (LOD) for the different matrices analysed in the study.

Matrix	**LOD**	**Matrix**	**LOD**
Surface water	300 ng/L	Air	< 1 ng/m^3
Storm water	1250 ng/L	Sediment	< 60 ng/g f.v.
Leachate water	1250 ng/L	Soil	< 60 ng/g f.v.
Influent water	1250 ng/L	Sludge	< 250 ng/g f.v.
Effluent water	625 ng/L		

8 Conclusions

- IPBC was found in air in urban areas, in air close to point sources and in waste water at industries where it is being used.
- IPBC occured in both influent and effluent water at STPs. The occurrence in effluent water indicated that it has a potential to reach the environment.
- IPBC was not found either in background or urban surface water, sediment and soil.
- DBNPA was not found in any of the samples. This is probably due to the rapid degradation of the compound.

9 Acknowledgement

The staff at the municipal sewage treatment plants are acknowledged for their help during sampling.

Annika Potter and Katarina Hansson at IVL in Gothenburg are acknowledged for their assistants in collecting and providing samples.

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Table A1. Sample information from the screening programme for IPBC.

Sample ID	Category	Site	Sampling date	Matrix	DW (%)	Notes
7806	Background	Råö	2009-02-27 - 2009-03-10	Air		
7847	Background	Råö	2009-03-30 - 2009-04-15	Air		
7674	Background	Gårdsjön	2008-11-20	Sediment	4	
7708	Background	Tärnan	2008-09-23	Sediment	10	
7679	Background	Gårdsjön	2008-11-20	Surface water		
7615	Background	Tärnan	2008-11-30	Surface water		
7616	Background	Largen	2008-11-31	Surface water		
7794	Point source	Paper and pulp industry	2009-03-06 - 2009-03-13	Air		
7790	Point source	Paint production 1	2009-02-26 - 2009-03-05	Air		
7795	Point source	Paint production 2	2009-02-25 - 2009-03-04	Air		
7811	Point source	Paper and pulp industry	2009-03-05	Soil	72	
7812	Point source	Paper and pulp industry	2009-03-05	Soil	72	
7831	Point source	Landfill	2009-04-22	Leachate water		Prior treatment
7832	Point source	Landfill	2009-04-22	Leachate water		After biological treatment
7813	Point source	Paper and pulp industry	2009-03-05	Surface water		
7789	Point source	Paper and pulp industry	2009-03-04	Storm water		
7777	Point source	Paint production 3	2009-02-23	Influent water		Prior chemical treatment
7837	Point source	Paint production 2	2009-04-21	Influent water		Prior treatment
7838	Point source	Paint production 2	2009-04-21	Effluent water		After treatment
7778	Point source	Paint production 3	2009-02-23	Effluent water		After chemical treatment
7810	Diffuse, urban	Göteborg, Femman	2009-02-24 - 2009-03-02	Air		
7807	Diffuse, urban	Göteborg, Femman	2009-03-04 - 2009-03-12	Air		
7669	Diffuse, urban	Stockholm, Vårbergstoppen	2009-02-13	Soil	28	
7670	Diffuse, urban	Stockholm, Årstafältet	2009-02-16	Soil	68	
7585	Diffuse, urban	Stockholm, St Essingen	2008-11-08	Sediment	18	
7583	Diffuse, urban	Stockholm, Årstaviken	2008-11-08	Sediment	16	
7584	Diffuse, urban	Stockholm, Riddarfjärden	2008-11-08	Sediment	16	
7582	Diffuse, urban	Stockholm, St Essingen	2008-11-08	Surface water		
7578	Diffuse, urban	Stockholm, Årstaviken	2008-11-10	Surface water		

Sample ID	Category	Site	Sampling date	Matrix	DW (%)	Notes
7581	Diffuse, urban	Stockholm, Riddarfjärden	2008-11-08	Surface water		
7845	Diffuse, urban	Stockholm, Årstafältet	2009-04-27	Storm water		
7846	Diffuse, urban	Stockholm, Huddinge	2009-04-27	Storm water		
7771	Diffuse, urban	Göteborg, Odingsplatsen	2009-02-23	Storm water		
7773	Diffuse, urban	Göteborg, Gårda	2009-02-23	Storm water		
7775	Diffuse, urban	Göteborg, Korsvägen	2009-02-23	Storm water		
7772	Diffuse, urban	Göteborg, Odingsplatsen	2009-02-23	Storm water sludge	22	
7774	Diffuse, urban	Göteborg, Gårda	2009-02-23	Storm water sludge	19	
7776	Diffuse, urban	Göteborg, Korsvägen	2009-02-23	Storm water sludge	41	
7766	Diffuse, STP	Henriksdal	2009-01-30	Sludge	27	
7726	Diffuse, STP	Käppalaverket	2008-12-15	Sludge	17	
7760	Diffuse, STP	Ryaverken	2009-01-29	Sludge	32	
7841	Diffuse, STP	Helsingborg	2009-04-20	Sludge	25	
7782	Diffuse, STP	Sjölunda	2009-02-23	Sludge	23	
7722	Diffuse, STP	Henriksdal	2008-12-16	Influent water		
7724	Diffuse, STP	Käppalaverket	2008-12-16	Influent water		
7839	Diffuse, STP	Helsingborg	2009-04-21	Influent water		
7779	Diffuse, STP	Sjölunda	2009-02-23	Influent water		
7761	Diffuse, STP	Ryaverken	2009-01-29	Influent water		
7723	Diffuse, STP	Henriksdal	2008-12-16	Effluent water		
7725	Diffuse, STP	Käppalaverket	2008-12-16	Effluent water		
7840	Diffuse, STP	Helsingborg	2009-04-21	Effluent water		
7784	Diffuse, STP	Sjölunda	2009-02-23	Effluent water		
7762	Diffuse, STP	Ryaverken	2009-01-29	Effluent water		

Table A2. Concentration of IPBC in samples from the screening programme.

Sample ID	Category	Site	Matrix	Sampling date	Unit	Conc
7806	Background	Råö	Air	2009-02-27 - 2009-03-10	pg/m^3	< 20
7847	Background	Råö	Air	2009-03-30 - 2009-04-15	pg/m^3	< 20
7674	Background	Gårdsjön	Sediment	2008-11-20	ng/g f.v.	< 0.3
7708	Background	Tärnan	Sediment	2008-09-23	ng/g f.v.	< 0.3
7679	Background	Gårdsjön	Surface water	2008-11-20	ng/l	< 5
7615	Background	Tärnan	Surface water	2008-11-30	ng/l	< 5
7616	Background	Largen	Surface water	2008-11-31	ng/l	< 5
7794	Point source	Paper and pulp industry	Air	2009-03-06 - 2009-03-13	pg/m3	< 20
7790	Point source	Paint production 1	Air	2009-02-26 - 2009-03-05	pg/m^3	35
7795	Point source	Paint production 2	Air	2009-02-25 - 2009-03-04	pg/m^3	39
7811	Point source	Paper and pulp industry	Soil	2009-03-05	ng/g f.v.	< 0.3
7812	Point source	Paper and pulp industry	Soil	2009-03-05	ng/g f.v.	< 0.3
7831	Point source	Landfill	Leachate water	2009-04-22	ng/l	35
7832	Point source	Landfill	Leachate water	2009-04-22	ng/l	< 10
7813	Point source	Paper and pulp industry	Surface water	2009-03-05	ng/l	< 5
7789	Point source	Paper and pulp industry	Storm water	2009-03-04	ng/l	< 10
7777	Point source	Paint production 3	Influent water	2009-02-23	ng/l	31500
7837	Point source	Paint production 2	Influent water	2009-04-21	ng/l	149000
7838	Point source	Paint production 2	Effluent water	2009-04-21	ng/l	32
7778	Point source	Paint production 3	Effluent water	2009-02-23	ng/l	220
7810	Diffuse, urban	Göteborg, Femman	Air	2009-02-24 - 2009-03-02	pg/m^3	< 20
7807	Diffuse, urban	Göteborg, Femman	Air	2009-03-04 - 2009-03-12	pg/m^3	110
7669	Diffuse, urban	Stockholm, Vårbergstoppen	Soil	2009-02-13	ng/g f.v.	< 0.3
7670	Diffuse, urban	Stockholm, Årstafältet	Soil	2009-02-16	ng/g f.v.	< 0.3
7585	Diffuse, urban	Stockholm, St Essingen	Sediment	2008-11-08	ng/g f.v.	< 0.3
7583	Diffuse, urban	Stockholm, Årstaviken	Sediment	2008-11-08	ng/g f.v.	< 0.3
7584	Diffuse, urban	Stockholm, Riddarfjärden	Sediment	2008-11-08	ng/g f.v.	< 0.3
7582	Diffuse, urban	Stockholm, St Essingen	Surface water	2008-11-08	ng/l	< 2.5
7578	Diffuse, urban	Stockholm, Årstaviken	Surface water	2008-11-10	ng/l	< 2.5
7581	Diffuse, urban	Stockholm, Riddarfjärden	Surface water	2008-11-08	ng/l	< 2.5

Sample ID	Category	Site	Matrix	Sampling date	Unit	Conc
7845	Diffuse, urban	Stockholm, Årstafältet	Storm water	2009-04-27	ng/l	< 10
7846	Diffuse, urban	Stockholm, Huddinge	Storm water	2009-04-27	ng/l	< 10
7771	Diffuse, urban	Göteborg, Odingsplatsen	Storm water	2009-02-23	ng/l	< 10
7773	Diffuse, urban	Göteborg, Gårda	Storm water	2009-02-23	ng/l	< 10
7775	Diffuse, urban	Göteborg, Korsvägen	Storm water	2009-02-23	ng/l	< 10
7772	Diffuse, urban	Göteborg, Odingsplatsen	Storm water sludge	2009-02-23	ng/g f.v.	< 1
7774	Diffuse, urban	Göteborg, Gårda	Storm water sludge	2009-02-23	ng/g f.v.	< 1
7776	Diffuse, urban	Göteborg, Korsvägen	Storm water sludge	2009-02-23	ng/g f.v.	< 1
7766	Diffuse, STP	Henriksdal	Sludge	2009-01-30	ng/g f.v.	< 1
7726	Diffuse, STP	Käppalaverket	Sludge	2008-12-15	ng/g f.v.	< 1
7760	Diffuse, STP	Ryaverken	Sludge	2009-01-29	ng/g f.v.	< 1
7841	Diffuse, STP	Helsingborg	Sludge	2009-04-20	ng/g f.v.	< 1
7782	Diffuse, STP	Sjölunda	Sludge	2009-02-23	ng/g f.v.	< 1
7722	Diffuse, STP	Henriksdal	Influent water	2008-12-16	ng/l	< 10
7724	Diffuse, STP	Käppalaverket	Influent water	2008-12-16	ng/l	34
7839	Diffuse, STP	Helsingborg	Influent water	2009-04-21	ng/l	130
7779	Diffuse, STP	Sjölunda	Influent water	2009-02-23	ng/l	< 10
7761	Diffuse, STP	Ryaverken	Influent water	2009-01-29	ng/l	65
7723	Diffuse, STP	Henriksdal	Effluent water	2008-12-16	ng/l	< 5
7725	Diffuse, STP	Käppalaverket	Effluent water	2008-12-16	ng/l	35
7840	Diffuse, STP	Helsingborg	Effluent water	2009-04-21	ng/l	14
7784	Diffuse, STP	Sjölunda	Effluent water	2009-02-23	ng/l	< 5
7762	Diffuse, STP	Ryaverken	Effluent water	2009-01-29	ng/l	120

Table A3. Sample information and results from the screening programme for DBNPA.

Sample ID	Category	Site	Sampling date	Matrix	DW (%)	Notes	Conc
7798	Background	Råö	2009-02-25 - 2009-02-27	Air			< 1 ng/m^3
7814	Background	Råö	2009-03-27 - 2009-03-30	Air			< 1 ng/m^{3}l
7674	Background	Gårdsjön	2008-11-20	Sediment	4		< 60 ng/g f.v.
7708	Background	Tärnan	2008-09-23	Sediment	10		< 60 ng/g f.v.
7679	Background	Gårdsjön	2008-11-20	Surface water			< 300 ng/l
7615	Background	Tärnan	2008-11-30	Surface water			< 300 ng/l
7616	Background	Largen	2008-11-31	Surface water			< 300 ng/l
7797	Point source	Paper and pulp industry	2009-03-05 - 2009-03-06	Air			< 1 ng/m^3
7811	Point source	Paper and pulp industry	2009-03-05	Soil	72		< 60 ng/g f.v.
7812	Point source	Paper and pulp industry	2009-03-05	Soil	72		< 60 ng/g f.v.
7831	Point source	Landfill	2009-04-22	Leachate water			< 1250 ng/l
7832	Point source	Landfill	2009-04-22	Leachate water			< 1250 ng/l
7813	Point source	Paper and pulp industry	2009-03-05	Surface water			< 300 ng/l
7789	Point source	Paper and pulp industry	2009-03-04	Storm water			< 1250 ng/l
7787	Point source	Paper and pulp industry	2009-03-03	Influent water			< 1250 ng/l
7788	Point source	Paper and pulp industry	2009-03-03	Effluent water		After biological treatment	< 625 ng/l
7809	Diffuse, urban	Göteborg, Femman	2009-02-23 - 2009-02-24	Air			< 1 ng/m^3
7796	Diffuse, urban	Göetborg, Femman	2009-03-02 - 2009-03-04	Air			< 1 ng/m^3
7669	Diffuse, urban	Stockholm, Vårbergstoppen	2009-02-13	Soil	28		< 60 ng/g f.v.
7670	Diffuse, urban	Stockholm, Årstafältet	2009-02-16	Soil	68		< 60 ng/g f.v.
7585	Diffuse, urban	Stockholm, St. Essingen	2008-11-08	Sediment	18		< 60 ng/g f.v.
7583	Diffuse, urban	Stockholm, Årstaviken	2008-11-08	Sediment	16		< 60 ng/g f.v.
7584	Diffuse, urban	Stockholm, Riddarfjärden	2008-11-08	Sediment	16		< 60 ng/g f.v.
7582	Diffuse, urban	Stockholm, St. Essingen	2008-11-08	Surface water			< 300 ng/l
7578	Diffuse, urban	Stockholm, Årstaviken	2008-11-10	Surface water			< 300 ng/l
7581	Diffuse, urban	Stockholm, Riddarfjärden	2008-11-08	Surface water			< 300 ng/l
7845	Diffuse, urban	Stockholm, Årstafältet	2009-04-27	Storm water			< 1250 ng/l
7846	Diffuse, urban	Stockholm, Huddinge	2009-04-27	Storm water			< 1250 ng/l
7771	Diffuse, urban	Göteborg, Odingsplatsen	2009-02-23	Storm water			< 1250 ng/l

Sample ID	Category	Site	Sampling date	Matrix	DW (%)	Notes	Conc
7773	Diffuse, urban	Göteborg, Gårda	2009-02-23	Storm water			< 1250 ng/l
7775	Diffuse, urban	Göteborg, Korsvägen	2009-02-23	Storm water			< 1250 ng/l
7772	Diffuse, urban	Göteborg, Odingsplatsen	2009-02-23	Storm water sludge	22		< 250 ng/g f.v.
7774	Diffuse, urban	Göteborg, Gårda	2009-02-23	Storm water sludge	19		< 250 ng/g f.v.
7776	Diffuse, urban	Göteborg, Korsvägen	2009-02-23	Storm water sludge	41		< 250 ng/g f.v.
7766	Diffuse, STP	Henriksdal	2009-01-30	Sludge	27		< 250 ng/g f.v.
7726	Diffuse, STP	Käppalaverket	2008-12-15	Sludge	17		< 250 ng/g f.v.
7841	Diffuse, STP	Helsingborg	2009-04-20	Sludge	25		< 250 ng/g f.v.
7782	Diffuse, STP	Sjölunda	2009-02-23	Sludge	23		< 250 ng/g f.v.
7760	Diffuse, STP	Ryaverken	2009-01-29	Sludge	32		< 250 ng/g f.v.
7722	Diffuse, STP	Henriksdals	2008-12-16	Influent water			< 1250 ng/l
7724	Diffuse, STP	Käppalaverket	2008-12-16	Influent water			< 1250 ng/l
7839	Diffuse, STP	Helsingborg	2009-04-21	Influent water			< 1250 ng/l
7779	Diffuse, STP	Sjölunda	2009-02-23	Influent water			< 1250 ng/l
7761	Diffuse, STP	Ryaverken	2009-01-29	Influent water			< 1250 ng/l
7723	Diffuse, STP	Henriksdals	2008-12-16	Effluent water			< 625 ng/l
7725	Diffuse, STP	Käppalaverket	2008-12-16	Effluent water			< 625 ng/l
7840	Diffuse, STP	Helsingborg	2009-04-21	Effluent water			< 625 ng/l
7784	Diffuse, STP	Sjölunda	2009-02-23	Effluent water			< 625 ng/l
7762	Diffuse, STP	Ryaverken	2009-01-29	Effluent water			< 625 ng/l

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The alarm reaction of coho salmon parr is impaired by the carbamate fungicide IPBC

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Abstract

To determine whether the carbamate fungicide IPBC alters the olfactory-mediated behavioral and physiologic alarm responses of coho salmon parr (*Oncorhynchus kisutch*), groups of coho were exposed to skin extract (an alarm pheromone source) under a variety of conditions. In the 3 min following skin extract exposure, freezing behavior was significantly increased (In the 3 min following skin extract exposure, freezing behavior was significantly increased under darkness (IR lighting) but not ambient lighting (25.3 ± 2.6% and 7.5 ± 5.7%, respectively; Δ calculated as: [(time (s) after/time (s) before) − 1] × 100%), and so IR was used for further experiments. Physiologically, following skin extract exposure, plasma cortisol concentration was increased at 0.5 h (58.1 ± 14.6 ng/ml versus 4.32 ± 1.31 ng/ml, exposed versus control), hematocrit (Hct) was increased at 2 h (50.4 ± 1.0% versus 41.7 ± 1.6%), and leucocrit (Lct) was decreased at 0.5 and 2 h (0.534 ± 0.114 and 0.13 ± 0.01% versus 1.23 ± 0.20%). After 0.5 h exposures to 0, 1, 10 and 100 μg/l IPBC and skin extract, the time spent dashing (>5 cm/s) increased significantly (323 ± 118%) in the first minute after skin extract exposure, but was absent in IPBC-exposed coho. Freezing behavior increased after skin extract exposure with control and 1 μg/l IPBC exposures (11.0 ± 3.0% and 17.7 ± 11.0%, respectively), but was absent after 10 μg/l and decreased after 100 μg/l IPBC. Physiologically, Hct and plasma lactate concentration were significantly increased above controls after 1 μg/l IPBC exposure (Hct: 45.7 ± 1.6% versus 34.0 ± 1.6%, lactate: 12.8 ± 1.2 mM versus 3.30 ± 1.2 mM). After 10 μg/l exposure, IPBC alone elicited a stress response similar to skin extract. However in the 100 μg/l treatment group the stress parameters were not different from controls. These findings suggest that the behavioral and physiologic alarm responses of juvenile salmonids may be impaired by acute exposure to ≥1 μg/l IPBC.

Keywords: Coho salmon; *Oncorhynchus kisutch*; Olfaction; Alarm response; Behavior; Cortisol; Carbamate; Pesticide; IPBC

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1. Introduction

For salmonids, olfaction facilitates indispensable behaviors such as migration and predator avoidance. By necessity, olfactory sensory neurons (OSNs) have a relatively unprotected environmental interface, which may leave them at risk during contaminant exposure. Environmentally realistic concentrations of metals and pesticides have been shown to impair salmonid OSNs (e.g. Hara et al., 1976; Borie et al., 1981; Waring and Moore, 1997; Baldwin et al., 2003; Jarrard et al., 2004; Sandahl et al., 2004). By impairing OSNs, contaminants may also affect olfactory-mediated behavioral and physiologic responses. For example, pesticides toxic to OSNs can lead to reduced sex steroid production (Moore and Waring, 1996a, 2001) and altered migration (Scholz et al., 2000). Compared with tests of OSN function, olfactory-mediated behavioral and physiologic response tests are more ecologically meaningful (Scott and Sloman, 2004), but they are also more technically difficult to assess.

Carbamate pesticides can rapidly impair salmonid OSNs at exposure concentrations ≤1 μg/l (Jarrard et al., 2004; Tierney et al., 2006). For example, 1 μg/l exposure to the carbamate antisapstain wood preservative IPBC (3-iodo-2-propynyl-*N*-butyl carbamate) significantly impaired coho salmon OSNs in 25 min (Tierney et al., 2006). The manner in which carbamates impair olfaction is unknown; most carbamate pesticides (i.e. *N*-methyl carbamates such as carbaryl or carbofuran) impair their targets is through addition of a carbamoyl moiety (NH_2CO) to the ser-OH esteratic site of the enzyme acetylcholinesterase. IPBC's mode of fungicidal action is unknown, although it has been hypothesized to be due to the terminal iodine (Juergensen et al., 2000). Structurally, IPBC resembles other carbamates in that it contains a core of carbamic acid (see Fig. 1), but it is not an *N*-methyl

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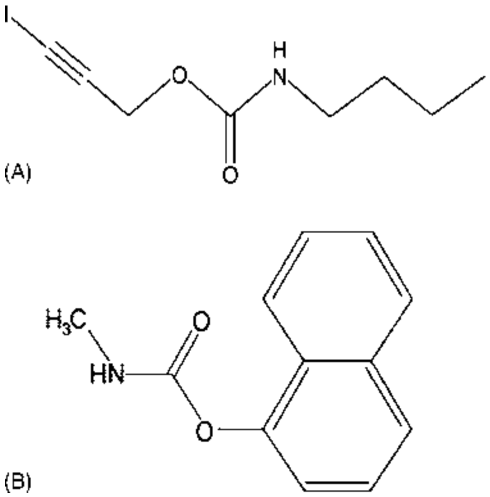

Fig. 1. The chemical structure of: (A) IPBC (3-iodo-2-propynyl-*N*-butyl carbamate) and that of the *N*-methyl carbamate and (B) carbaryl (1-naphthyl-*N*-methyl carbamate).

carbamate. IPBC has received attention in part because it can be present in runoff from lumber storage and milling sites located on salmon-producing rivers (Envirochem, 1992; Juergensen et al., 2000). Currently, the Canadian regulatory IPBC guideline for freshwater is 1.9 μg/l (CCME, 1999); US guidelines would be present on product labeling (FIFRA, 1996). The Canadian limit is well below acute lethality values for juvenile salmonids (96 h LC_{50} of 67 and 95 μg/l for rainbow trout [*Oncorhynchus mykiss*] and coho salmon [*O. kisutch*], respectively [Bailey et al., 1999; Farrell et al., 1998]), but it is not below concentrations that cause olfactory toxicity (i.e. 0.1 μg/l, Jarrard et al., 2004). Although IPBC has been identified as toxic to salmonid OSNs, it is not known how this may translate to altered olfactory-mediated physiologic or behavioral responses.

This study examined the olfactory-mediated alarm responses of coho salmon parr following a sublethal IPBC exposure. The alarm response consists of physiologic and behavioral responses, characteristic of 'fight-or-flight' reactions (Cannon, 1914). Specifically, following sensory input, central command neurons of the autonomic nervous system facilitate physiologic stress responses (Jansen et al., 1995). These responses may include hypothalamo–sympathetic–chromaffin axis-mediated release of catecholamines (e.g. epinephrine) and hypothalamo–pituitary–interrenal axis-mediated release of glucocorticoids (e.g. cortisol) (Mommsen et al., 1999). The former providing rapid (i.e. ≥seconds) enhancement of glucose and oxygen delivery to tissues such as muscle and brain, and the latter providing a following up (i.e. ≥minutes) that also enhances glucose availability, but in addition, enhances its storage (Mommsen et al., 1999). In the teleost alarm response, both primary (e.g. cortisol increases) and secondary (e.g. glucose increase) responses typically occur (Scott et al., 2003; Rehnberg and Schreck, 1987). In non-olfactory-based stress studies, other blood sample based measures including packed red and white blood cell volume (hematocrit [Hct] and leucocrit [Lct], respectively) have been used to assess secondary physiologic stress responses. For example, Hct increased from 37 to 42% (Tierney et al., 2004a) and Lct increased from 0.57 to 0.82% (Tierney et al., 2004b) in coho salmon parr 96 h after a cold-shock (18–8 °C transfer).

Physiologic changes may underlie behavioral alarm responses, which usually involve energy expenditure. For example, when the ostariophysan matrinxã (*Brycon cephalus*) perceives alarm pheromone, a period of hyperactivity (i.e. 'dashing') precedes hypoactivity (i.e. 'hiding') (Ide et al., 2003). In salmonids, a variety of alarm response studies have reported a period of hypoactivity or increased time spent frozen in alarmed fish (in rainbow trout: Brown and Smith, 1997, 1998; Mirza and Chivers, 2001, 2003; Leduc et al., 2004; in chinook salmon (*Oncorhynchus tshawytscha*): Berejikian et al., 1999, 2003; Scholz et al., 2000). As the magnitude of the alarm signal increases, the amount of inactivity increases (Mirza and Chivers, 2003). Thus far, dashing behavior has not been observed in coho salmon.

In the present study, swimming behaviors were observed and primary and secondary measures of stress were taken in coho salmon parr following skin extract exposure. Typically, salmonid alarm response behavior is examined in individuals; however, we chose to measure it in groups, which may improve ecological relevance. This also facilitates physiologic measures, which tend to be measured in five or more tank-exposed fish. To help limit any non-olfactory-based alarm reaction (e.g. visual alarm reaction propagation; Smith, 1992), a novel infrared camera and lighting system was designed to allow testing under darkness. To help resolve a suitable sampling time and skin extract concentration, the physiologic responses were measured at 0, 0.5, 1 and 2 h after exposure to three concentrations of skin extract. Finally, both the physiologic and behavioral responses of coho exposed to skin extract were characterized following exposure to three concentrations of IPBC.

2. Materials and methods

2.1. Animals

Coho salmon parr were obtained from the Fisheries and Oceans Canada (DFO) Capilano Fish Hatchery (North Vancouver, BC) in January 2005 ($n = 370$, mass 10.0 ± 0.13 g, length 10.0 ± 0.04 cm, fork length condition factor 0.995 ± 0.006; Adams et al., 1993). Coho were held for at least 3 weeks prior to experimentation in a 500 l flow-through tank supplied with dechlorinated municipal water (dissolved oxygen > 90% saturation, pH 6.8, hardness 6.12 mg/l $CaCO_3$), using a 12-h light:12-h dark photoperiod, and were fed commercial salmon feed *ad libitum* (EWOS Pacific, Surrey, BC).

2.2. Skin extract and pesticide preparation

Skin extract was prepared as in Brown and Smith (1998). In brief, fish were sacrificed by cervical dislocation, the skin was then removed, rinsed with distilled deionized water (DDW), massed and placed on ice. In the first experiment with lighting, 50 mg/l of skin extract was used as this concentration was comparable to that used in a similar study with rainbow trout (i.e.

43 mg/l; Scott et al., 2003). To achieve the desired skin extract exposure concentration, 1.7 g of skin was homogenized, filtered through glass wool and brought to a final volume of 50 ml in DDW and then frozen at −20 °C. Thus, when added to the 34 l tanks (dimensions of 50 [length] × 25 [width] × 27 [height] cm), the final skin extract concentration was ∼50 mg/l. In the second experiment, to determine the effect of skin extract concentration on alarm response, concentrations bracketing this concentration were also tested (i.e. 5 and 500 mg/l).

IPBC (3-iodo-2-propynyl-*N*-butyl carbamate; 97%; Sigma–Aldrich, Oakville, ON) was solubilized in polyethylene glycol (10 mg/ml) before being added to tanks.

2.3. Video surveillance equipment and light sources

For behavior analysis, video was acquired and captured using CNL100 CCD cameras (one per tank) streaming to PV-140V surveillance cards (Matco, St. Laurent, QC) in a 2 GHz Pentium PC. Cameras were placed 15 cm above the water, which permitted capture of the complete tank area.

For tests under ambient lighting (first experiment only), tanks were illuminated using standard, fluorescent lighting. For all other tests, individual tanks were illuminated using two custom-built infrared (IR) light sources, each consisting of a parallel array of four strings of six TSAL5100 940 nm wavelength diodes (Vishay, Shelton, CT) and two 22 Ω resistors (total 48 LEDS per tank). IR (i.e. wavelengths between ∼≥750 nm and ≤1 mm) cannot be detected by other salmonids (e.g. rainbow trout (Allen and Munz, 1983) and Atlantic salmon (Ali, 1961)), and so it is likely that coho cannot detect IR either. Here, the IR light illuminated the tanks and was relayed by the cameras, computer and software as white-light.

2.4. Experiment 1: the alarm reaction under different lighting conditions

To determine whether the behavioral alarm response could be measured in the absence of visual light, coho were tested under ambient and infrared lighting. Coho were held for 24 h in 34 l tanks, with control and treatment tanks for each lighting condition (five fish per tank; four tank replicates per lighting condition). Each tank, camera and lighting arrangement was completely shielded in black plastic to eliminate outside visual disturbances. Vehicle control (DDW) or skin extract was injected into tanks through a glass inflow tube and chased by 15 ml of DDW to ensure complete delivery. For IR experiments, the fluorescent lights were turned off and IR lights were turned on 0.5 h before injections. Video was recorded for 5 min before and after vehicle or skin extract exposure for subsequent behavioral analysis. All experiments were conducted at the same time of day.

2.5. Experiment 2: skin extract concentration and sampling time

As in Experiment 1 (Exp. 1), fish were placed into 34 l flow-through tanks and acclimated for 24 h before testing ($n = 5$ per tank). To help determine an appropriate skin extract concentration and post-skin extract exposure time that would yield a significant physiologic stress response, fish were exposed to 0, 5, 50 and 500 mg/l skin extract and sampled at 0, 0.5, 1 and 2 h post skin extract or DDW, with the entire experiment performed in duplicate. Water flow was discontinued 0.5 h prior to the delivery of the skin extract.

At each sampling time, fish were sacrificed by over anesthetization with 0.3 g/l MS-222 (Syndel Laboratories, Vancouver, BC) buffered 1:1 with $NaHCO_3$. Fish were then massed, lengthed and blood samples were collected via severed caudal peduncle into 40 μl heparinized Hct tubes. Hct tubes were centrifuged at 10,000 × *g* for 5 min, and Hct and Lct were measured within 10 min using digital calipers and a dissecting microscope. Hct and Lct measurements were performed in triplicate for most fish. Plasma was isolated and stored at −80 °C until analyzed for cortisol concentration. Each sampling was conducted between 10:00 a.m. and 12:00 p.m. to minimize the possibility of diurnal cortisol fluctuations (Peter et al., 1978). Cortisol concentrations were measured in triplicate using ELISA kits (Neogen Corporation, Medicorp, Montréal, QC).

2.6. Experiment 3: the alarm reaction after IPBC exposure

As in Exps. 1 and 2, fish were placed into 34 l flow-through tanks and allowed to acclimate for 24 h ($n = 5$ per tank). To determine whether IPBC altered the alarm response, fish were statically exposed to 0, 1, 10 and 100 (nominal) μg/l IPBC for 0.5 h, with the experiment performed in hexuplicate. Controls received the same amount of vehicle as the highest concentration (i.e. 0.1 ml/l or 0.34 ml PEG per tank). Tanks were then given 50 mg/l skin extract, a concentration shown in Exp. 2 to cause significant changes in primary and secondary stress response parameters, or vehicle. After a 0.5 h skin extract or vehicle exposure (total IPBC exposure of 1 h), fish were sacrificed and blood samples were taken and analyzed as above, except that plasma glucose and lactate concentrations were also measured (kits from Stanbio, Boerne, TX; Trinity Biotech, St. Louis, MO).

2.7. Behavioral and statistical analysis

The swimming speed of each fish in the horizontal plane was determined manually with the help of 1 cm^2 black gridlines on white plexiglas tank bottoms for every second of 3 min before and after skin extract or vehicle exposure. This behavioral scoring was conducted blind. For Exp. 1, since early observations indicated that fish were either still or in motion, the proportion of time spent either active (>0 cm/s) or inactive (frozen; ∼0 cm/s) for the 3 min before and after skin extract exposure was determined within each tank. A similar quantification of activity was used by Lindsay and Vogt (2004) and Scholz et al. (2000). Differences in the average time spent frozen before and following skin extract exposure were compared using two-way repeated measures analysis of variance (RM ANOVA) followed by Holm–Sidak post hoc test. Additionally, to determine if the change in the time spent frozen was significant for each lighting condition, the change in freezing, calculated as [after–before; time in seconds], was tested against zero using a two-tail *t*-test.

For Exp. 3, to better capture fish behavior, the ‘active’ category of Exp. 1 was split into ‘active’ (>0 and <5 cm/s) and ‘dashing’ (>5 cm/s). Since dashing has typically been observed prior to the freezing portion of the alarm response (Ide et al., 2003), the amount of time spent dashing during the first minute and the amount of time spent frozen during the third minute was compared to the average for the 3 min pre-exposure period. To ease reporting, the % change ([after–before − 1] × 100%) is also provided. Within all tanks, there was at least 1 s of each activity over the pre- and post-skin extract 3-min periods, and so the equation was never undefined. As in Exp. 1, a two-tail *t*-test of the change in activity against zero was used to detect significant changes in behavior.

For physiologic data, comparisons were made between replicate tanks, and if no differences were found, data was pooled and individual fish were considered (and thus $n = 10$ and 30 per treatment, Exps. 2 and 3, respectively). For Exp. 2, two-way ANOVA followed by a Holm–Sidak test was used to test for differences and interactions between skin extract concentration (0, 5, 50 and 500 mg/l) and exposure time (0, 0.5, 1 and 2 h) for stress parameters. For Exp. 3, the same statistical method was used to detect differences in alarm response at each IPBC concentration. All percent data were arcsine transformed prior to statistical analysis (Zar, 1999). Statistical significance was set at $p < 0.05$, and unless stated otherwise, reported values are mean ± S.E.M. JMP 5.0 (SAS, Toronto, ON), SigmaStat 3.0 and SigmaPlot 8.0 (Systat Software, Inc., Point Richmond, CA) were used for statistical analysis and graphing.

3. Results

3.1. Experiment 1: the alarm reaction under different lighting conditions

Fish were significantly more active in darkness, spending 29.9 ± 4.5% of the time active under infrared (IR) light (i.e. an average 269 ± 40 s of the total 900 possible seconds [5 fish × 3 min × 60 s/min] was spent at >0 cm/s) compared to 10.0 ± 6.6% (or 90.2 ± 59.0 s of total) under ambient lighting ($p = 0.038$) (Fig. 2). However, following skin extract exposure, activity levels were similar (4.08 ± 1.62% IR versus 7.33 ± 1.06% ambient) (Fig. 2). Freezing behavior increased in both lighting conditions (25.3 ± 2.6% under IR versus 7.54 ± 5.7% under ambient), but the increase was only significant under IR ($p = 0.008$). Subsequent experiments were conducted under IR.

3.2. Experiment 2: skin extract concentration and sampling time

Skin extract exposure elicited significant changes in all of the measured primary and secondary stress response parameters compared to controls. Values for time 0 and vehicle control groups (no skin extract) were not significantly different and so were pooled and are reported as mean ± 95% CI (Fig. 3). Plasma cortisol concentration was significantly increased over controls at 0.5 h following skin extract exposure ($p = 0.001$) (Fig. 3a),

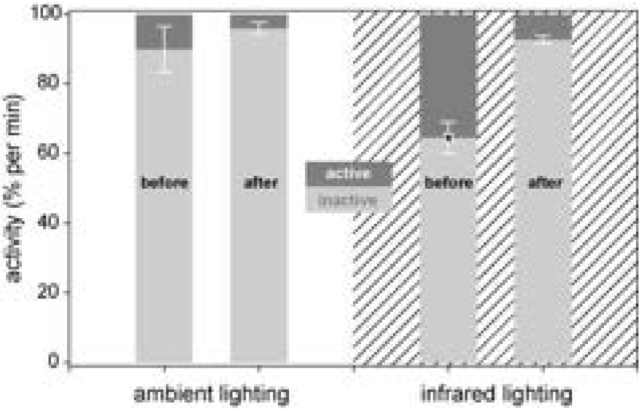

Fig. 2. The percent of time juvenile coho salmon spent either active (>0 cm/s) or inactive (∼0 cm/s) 3 min before and following skin extract exposure under either ambient or infrared lighting (darkness). Asterisk indicates significant difference (two-way ANOVA, Holm–Sidak, $n = 4$ tanks per lighting condition, five fish per tank).

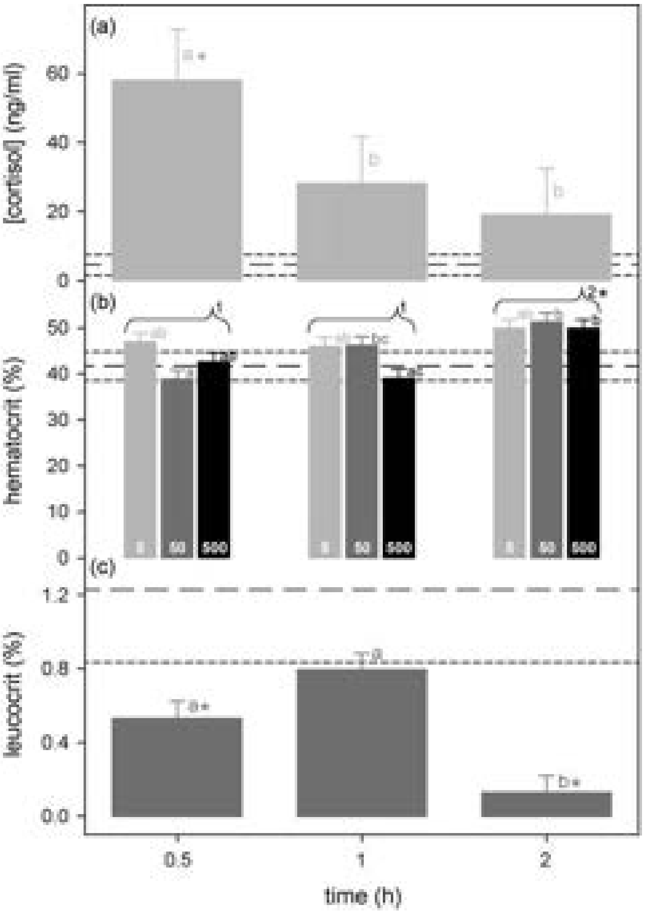

Fig. 3. (a) Plasma cortisol concentration, (b) hematocrit and (c) leucocrit of coho salmon parr at 0, 0.5, 1 and 2 h following exposure to 0, 5, 50 and 500 mg/l of skin extract, an alarm pheromone source. Plasma cortisol concentration and leucocrit were pooled as there no differences due to skin extract concentration. Similarly, time 0 and distilled water controls were pooled and are shown as mean ± 95% CI (dashed and dotted lines, respectively). Like alphanumeric superscripts indicate values that are not significantly different; asterisk indicates difference from control (two-way ANOVA, Holm–Sidak, $n = 10$ per group).

whereas Lct was significantly decreased at 0.5 and 2 h, but not at 1 h (p values of 0.04, 0.001 and 0.193, respectively) (Fig. 3c). For both plasma cortisol concentration and Lct, the amount of skin extract used (5, 50 and 500 mg/l) did not influence the response and so values are reported pooled (p values of 0.426 and 0.207, respectively) (Fig. 3a and c). With Hct, there were some differences between skin extract exposure concentrations, although overall, Hct values were significantly greater than control at 2 h post exposure ($p < 0.001$) (Fig. 3b). Since a sampling time of 0.5 h resulted in significant changes in two of three measured stress parameters (i.e. cortisol concentration increased and Lct decreased), and as 50 mg/l is a concentration similar to that used previously (i.e. 43 mg skin/l, in Scott et al., 2003), this time and concentration were used for experiments with IPBC.

3.3. Experiment 3: the alarm reaction after IPBC exposure

In vehicle-exposed tanks, dashing was significantly ($p = 0.002$) increased in the first minute following skin-extract exposure. Specifically, during the 3 min pre-skin extract exposure, there was an average of 2.69 ± 1.44 s of dashing compared to 9.83 ± 6.92 s of dashing in the 1 min following, resulting in a 323 ± 118% increase (Fig. 4a). Freezing was significantly ($p = 0.024$) increased in the third minute following skin extract exposure (11.0 ± 3.0% increase, or 264 ± 10 s before versus 292 ± 3 s after skin extract exposure) (Fig. 4b). In all IPBC-exposed fish, the time spent dashing was completely eliminated (Fig. 4a). The time spent freezing was not altered with 1 µg/l IPBC (17.7 ± 11.0%; $p = 0.030$), but was absent with 10 (4.06 ± 2.22%; $p = 0.065$) and decreased with 100 µg/l IPBC (−23.6 ± 11.4%; $p = 0.030$; i.e. skin extract caused an increase in activity at 100 µg/l IPBC) (Fig. 4b).

In skin extract-exposed fish, cortisol concentration increased ($p = 0.020$) and Hct decreased ($p = 0.008$) compared to vehicle controls (Fig. 5a and b, respectively). The stress responses were altered differently by IPBC at each exposure concentration. Specifically, in the 1 µg/l IPBC treatment group, skin extract exposure caused significant increases in both Hct ($p = 0.010$) and plasma lactate concentration ($p = 0.009$) (Fig. 5b and c, respectively). With a 10 µg/l IPBC exposure, IPBC exposure alone caused an increase in stress parameters; skin extract caused no further change or was not different from control. For example, Hct was significantly increased from control ($p = 0.029$), but was not influenced by skin extract exposure ($p = 0.513$) (Fig. 5b). No

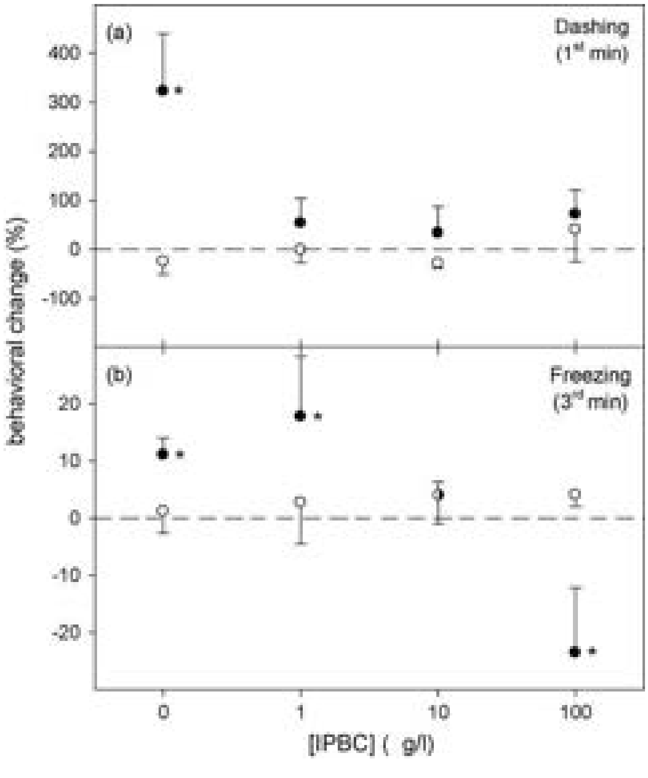

Fig. 4. (a) Changes in the proportion of the first minute spent dashing (>5 cm/s) and (b) of the third minute spent frozen (∼0 cm/s) for coho salmon after 0.5 h exposure to concentrations of the carbamate pesticide IPBC and either a distilled water control (○) or 50 mg/l skin extract (●). Percent activity change was calculated as (after/before − 1) × 100%. Values differing from zero (i.e. no change) are denoted with asterisk (two-sample t-test, $n = 6$ tanks of five fish per point).

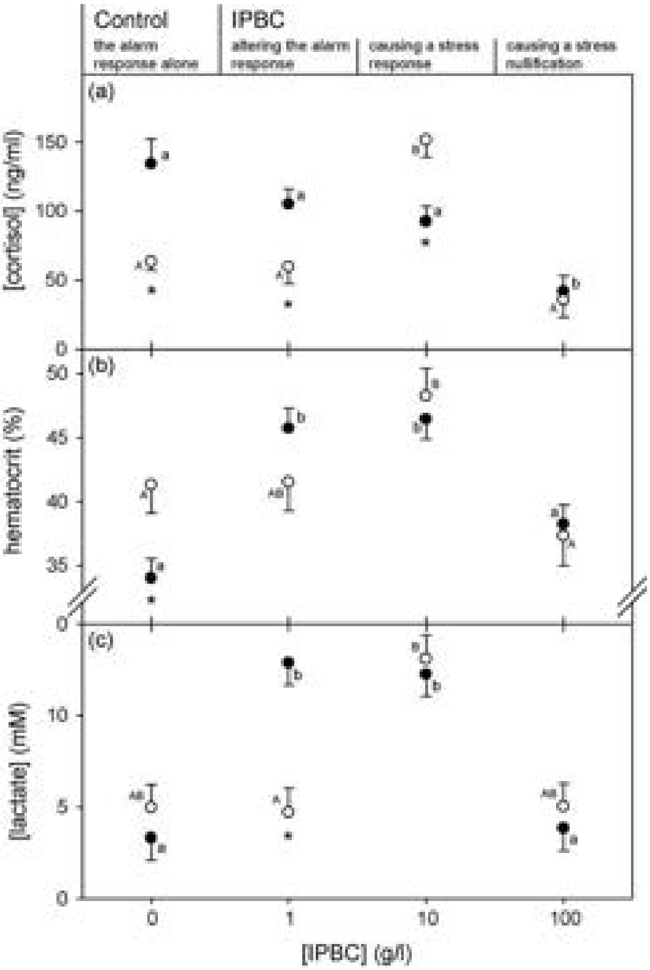

Fig. 5. (a) Plasma cortisol concentration, (b) hematocrit and (c) lactate concentration in coho salmon parr 1 h after IPBC exposure and 0.5 h after either distilled water control (○) or 50 mg/l skin extract (●). Like letters indicate values that are not significantly different from each other; differences between control and skin extract-exposed groups at each IPBC concentration are denoted with asterisks (two-way ANOVA, Holm–Sidak, $n = 30$ per point).

stress parameters were significantly different from control following 100 µg/l IPBC exposure. No changes were detected in Lct and or glucose for any groups.

4. Discussion

The goal of this research was to determine if exposure to the carbamate fungicide IPBC impaired the behavioral and physiologic olfactory-based alarm reaction in coho salmon parr. Certain coho stocks are endangered in Canada and the US, and these and other salmonids are important to aboriginal culture, recreational and commercial fisheries, marine ecosystems (e.g. food for resident killer whales, *Orcinus orca*) and terrestrial ecosystems (i.e. nutrients for temperate rainforests and terrestrial predators). Salmonids are particularly vulnerable to the possible effects of currently used pesticides, with their large-scale migrations and their reliance on multiple habitat types at different life stages.

The olfactory-mediated alarm response was selected for study since it can be crucial to successful negotiation of a predator attack (Kats and Dill, 1998). For the juvenile life stage tested here, the alarm response is of obvious ecological relevance. However, the alarm response is not the only function in which olfaction is essential to survival; olfaction is indispensable throughout the salmonid life cycle, enabling imprinting and return migration (Dittman and Quinn, 1996), and the synchronization of gamete release (Moore and Waring, 1996b). Since many of these behaviors are based on olfaction, a reduction in alarm response performance may indicate a reduction in others. The carbamate IPBC was chosen for study both because of its proximal use to major salmon-producing rivers in British Columbia and elsewhere, and because of its established toxicity to salmon OSNs (Jarrard et al., 2004; Tierney et al., 2006). Furthermore, IPBC may serve as a model for other current-use pesticides that are acutely toxic to salmonid OSNs. The IPBC concentrations used in this study (1, 10 and 100 µg/l) are potentially environmentally realistic, since IPBC has been reported to vary from non-detectable to 370 µg/l in stormwater (Envirochem, 1992). Furthermore, the method of pulse exposures used here may also be typical of IPBC exposures in the field since it is moderately soluble, does not appreciably adsorb to sediments, and can remain in solution for extended periods (Table 1).

To determine if testing could be conducted under darkness (in order to remove visual alarm propagation that can occur in groups of fish), coho were tested under infrared and fluorescent lighting. Coho salmon were more active under darkness, a finding that agrees with other studies. For example, the foraging activity of chinook salmon was greater under conditions of water turbidity, which limits vision, (Gregory and Northcote, 1993). On a larger scale, the migration activity of brown trout (*Salmo trutta*) was found to be greater at night (Bendall et al., 2005). Consequently, it is possible that for salmonids, activity is greater under the cover of darkness because the perceived predation risk is lower. This agrees with the present study: because of the greater activity under darkness, the relative increase in freezing behavior was significant only in the absence of light. For coho tested under fluorescent lighting, the increase in freezing was not significant, which suggests that these coho may have been already somewhat in predator avoidance mode. After skin extract exposure, activity was the same regardless of lighting, which may indicate skin extract induced a risk level unrelated to lighting. Similarly, Vainikka et al. (2005) noted that the antipredator responses (which included decreased activity and shoaling) of perch (*Perca fluviatilis*) did not differ with risk level. In contrast, Hartman and Abrahams (2000) noted that fathead minnows (*Pimephales promelas*) exhibited an increased dashing response following alarm substance exposure under turbid conditions. They propose that since the minnows were unable to visually assess their surroundings, the predation risk was assumed to be greater and thus there was a commensurate increase in alarm response. The difference between salmonids and minnows in this regard is unclear. Darkness was used in the present study to help isolate a chemosignal from photosignals, however, it does not remove all non-olfactory based sensory input, e.g. the mechanoreception of the lateral line system (Dijkgraaf, 1962; Ćurčić-Blake and van Netten, 2006).

Table 1
General information and some physico-chemical properties of IPBC[a]

Property	Value
Name	3-Iodo-2-propynyl-*N*-butyl carbamate
CAS number	55406-53-6
Chemical formula	$C_8H_{12}INO_2$
Molecular mass (g/mol)	281
Specific gravity (g/ml)	1.58
Water solubility (mg/l)	156
Hydrolysis half-life (days)	
pH 5	Stable
pH 7	139
pH 9	0.947
Vapor pressure (Pa)	
20 °C	0.002
30 °C	0.007
Melting point (°C)	54
Log octanol–water partition coefficient (K_{ow})	2.81
Log organic-carbon adsorption coefficient ($K_{\propto}$)	2.05 ± 0.12[b]
Bioconcentration factor	<4.5
Products	Kop-CoatNP-1 TroysanKK108-A Troysan Polyphase P-100
Uses	Antisapstain Antimicrobial Mildewcide

[a] From the review by Juergensen et al. (2000).

[b] Average for various sediments.

The experimental procedure used was not stressful since control cortisol concentrations (4.32 ± 1.31 ng/ml) compare favorably with other studies using similar fish (e.g. 2–5 ng/ml in Atlantic salmon [*Salmo salar*] parr (Shrimpton and McCormick, 1998) and 5 and 10–15 ng/ml in juvenile rainbow trout (Scott et al., 2003 and Toa et al., 2004, respectively)). However, for the final set of experiments with IPBC (Exp. 3), which were conducted in April and approximately one month following the initial measurements (Exp. 2), the control values were higher (61.8 ± 6.0 ng/ml). This may have been from natural increases in this stress hormone coincident with the approaching smoltification period (Shrimpton and McCormick, 1998; McCormick et al., 2000). Regardless of this baseline concentration, skin extract exposure elicited a plasma cortisol increase. In the coho tested earlier, skin extract exposure caused an approximately 10-fold increase in plasma cortisol concentration in 0.5 h; in the coho tested later, the difference was closer to 2-fold. As a general rule, stressful events such as handling or contaminant exposure can cause increases in plasma cortisol concentration in 1 h (reviewed by Mommsen et al., 1999).

The concentration of skin extract used (5, 50 or 500 mg/l) did not appear to influence the cortisol response, which may indicate that these concentrations were all above the HPI axis response threshold. Therefore, to facilitate a comparison with a similar study on rainbow trout that used 43 mg/l (Scott et al., 2003), 50 mg/l was used in the final set of experiments. In Scott et al. (2003) and the present study, skin extract caused significant increases in plasma cortisol concentrations within 0.5 h. In contrast, Ide et al. (2003) found that skin extract did not increase cortisol in matrinxã. Other antipredator cues, such as predator scent, can also increase plasma cortisol concentration. For example, plasma cortisol concentration was increased in coho salmon following exposure to tank water of both northern squawfish (*Ptychocheilus oregonensis*) and largescale sucker (*Catostomus macrocheilus*) (Rehnberg and Schreck, 1987). Secondary measures of stress may also indicate an olfactory-mediated alarm response. Here, we report for the first time that both Hct and Lct can change following skin extract exposure, with decreases in both noted 0.5 h following exposure. These changes may have their origins in altered cortisol concentrations since this hormone is the teleost mineralocorticoid and is leucocytolytic (Weyts et al., 1998). The olfactory-mediated physiologic response likely provides energy for antipredator behaviors, since an Hct increase can enhance blood oxygen carrying capacity (Gallaugher and Farrell, 1998), and leucocyte destruction may make protein available for gluconeogenesis (McLeay, 1973).

Coho exhibited a biphasic behavioral alarm response to skin extract (i.e. dashing occurred prior to freezing). As stated in the introduction, dashing has not been observed previously in salmonids. The difference in the present study may have been due to the testing procedure, which consisted of five coho together under IR in small tanks, or may have been influenced by the life stage used. Alternately, the concentration of skin extract was sufficient to evoke an as yet unseen behavioral response. As Mirza and Chivers (2003) noted, the magnitude of the behavioral alarm response can be related to skin extract concentration. If this is the case, it suggests that some components of the alarm response may differ in thresholds, seeing as the cortisol response was unrelated to skin extract concentration. This idea is further supported with our IPBC exposures: dashing was impaired by a lower concentration of IPBC than freezing (i.e. 1 μg/l versus 10 μg/l, respectively), which may indicate that dashing behavior has a higher response threshold.

Other pesticides can impair alarm responses at similar concentrations, for example, in chinook salmon, Scholz et al. (2000) found that the freezing response was impaired by 2 h exposure to 1 μg/l of the insecticide diazinon. Furthermore, environmental contaminants other than pesticides can impair the alarm response; Scott et al. (2003) found that the normal decreases in line crossings and feeding caused by skin extract exposure did not occur in rainbow trout exposed to 2 μg/l Cd for 1 week. Furthermore, contaminant exposure can reduce the alarm response in other types of fishes. For example, Saglio and Trijasse (1998) found that the grouping response exhibited by goldfish following skin extract exposure was decreased 24 h after exposure to 5 μg/l of the herbicides atrazine or diuron. The results of the present study together with those of others clearly establish that olfactory-mediated behavior can represent a sensitive sublethal toxicity endpoint in fish. Moreover, because of the importance of olfactory-mediated behaviors such as the alarm response, a pesticide-associated impairment of performance may have profound consequences for survival.

The physiologic alarm response of coho salmon parr was also altered by IPBC exposure. The expectation was that a decrease in this stress response would occur since a 1 μg/l IPBC exposure has been shown to impair olfactory neurons (Jarrard et al., 2004; Tierney et al., 2006) and olfactory-mediated behaviors. This was true for Hct, where skin extract alone caused an Hct decrease that was eliminated by 1 μg/l IPBC exposure, but not necessarily true for lactate, where skin extract appeared to cause a significant increase only in the presence of IPBC. It is possible that the lactate increase was motivated by multiple stressors (i.e. IPBC and skin extract together). Both Hct and plasma lactate concentration may be increased in juvenile salmonids following stressful events such as contaminant exposure (Tierney et al., 2004a and Wood et al., 1996, respectively). These findings are significant because they indicate that olfactory toxicity may need to be considered along with other measures of toxicity.

Skin extract exposure did not appear to contribute to the physiologic stress response at an IPBC exposure concentration of 10 μg/l or greater. However, three stress parameters (plasma cortisol and lactate concentration, and Hct) were increased. These IPBC-mediated increases may represent meaningful digressions from homeostasis, as Hct was increased beyond the normal healthy salmonid Hct range (i.e. from 41.3 ± 2.2% to 48.2 ± 2.2%; normal range 35–41%, Gallaugher and Farrell, 1998). Curiously, after 100 μg/l IPBC exposure, parameters were not different from 0 μg/l IPBC control. These findings are in keeping with Farrell et al. (1998), who found no change in a similar suite of primary and secondary stress parameters in juvenile rainbow trout after 24 h to 140 μg/l Polyphase P-100 (97% IPBC). Since stress parameters were increase by a concentration between 1 and 100 μg/l, IPBC may have had a hormetic effect (reviewed by Calabrese, 2005). Although the mechanism of IPBC toxicity is unknown (Juergensen et al., 2000), perhaps at the highest concentration, IPBC had a sedatious effect or it negatively impacted the hypothalamo–pituitary–interrenal (HPI) axis. In any event, the non-linear response complicates an extrapolation of IPBC effects from higher to lower concentrations.

Even though a physiologic stress response was absent after a 100 μg/l IPBC exposure, coho still exhibited a behavioral response to skin extract, indicating that there was sensory input from olfactory neurons. However, skin extract exposure at this IPBC concentration caused a decrease in freezing behavior, instead of the typical increase. It is possible that by impairing olfactory neurons, a 100 μg/l IPBC exposure effectively increased the alarm response threshold, to such a degree that normal alarm behaviors did not ensue (Mirza and Chivers, 2003). Alternately, the behavioral response may have deviated because a physiologic stress response was absent. This would indicate that a physiologic response is necessary for normal alarm behaviors to occur. Future studies could test this interplay by blocking the stress response with β-blockers (e.g. propranolol) or cortisol blockers (e.g. metyrapone).

5. Conclusions

Both behavioral and physiologic aspects of the olfactory-mediated alarm reaction were altered by environmentally realistic concentrations of the carbamate fungicide IPBC. One microgram per liter IPBC exposure eliminated dashing and aggravated the physiologic stress response. After 10 μg/l IPBC exposure, the freezing response did not occur, and increases in various physiological measures of stress were observed. In fact, on its own, stress parameters were increased after 10, but not 1 or 100 μg/l IPBC exposure, indicating a non-linear or complex, multifaceted mechanism of action which precludes a traditional concentration-response evaluation. After 100 μg/l exposure, alarm behavior was deviant (i.e. in place of a freezing response, skin extract caused an increase in activity), and a physiologic stress response was absent. Since alarm response is an important antipredator behavior, exposures to IPBC at concentrations at or below the existing Canadian regulations of 1.9 μg/l may render exposed salmon more susceptible to predation.

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Canadian Water Quality Guidelines for the Protection of Aquatic Life

IPBC

IPBC (3-iodo-2-propynyl butylcarbamate (CAS number 55406-53-60 and molecular formula $C_8H_{12}INO_2$) is a carbamate compound (Konasewich and Quintin 1994) commonly used as a pesticide with various purposes, an industrial fungicide in paints and adhesives, an antimicrobial in metal-working fluids and on canvas and cordage, and as an antisapstain (Szenasy and Bailey 1996). It is a white crystalline powder with a specific odour. IPBC is highly soluble in organic solvents (e.g., acetone and benzyl alcohol) (Bioassay Systems Corporation 1982) and is moderately soluble in water (156 $mg{\cdot}L^{-1}$ at 20°C) (Konasewich and St. Quintin 1994). IPBC has a low vapour pressure: <0.002 Pa at 20°C and 0.007 Pa at 30°C (USEPA 1997). The log octanol–water coefficient (log K_{ow}) has been estimated at 2.81 (MRI 1990). IPBC is not likely to complex or absorb with suspended solids or sediments and is more likely to remain in the dissolved phase (Szenasy and Bailey 1996).

IPBC has received full registration in Canada for use as a material preservative in paints, adhesives, caulking, etc., and temporary full registration for use as an antisapstain wood preservative and joining wood preservative (M. Raphael 1998, Pest Management Regulatory Agency, Ottawa, pers. com.). IPBC was used in antisapstain formulations, primarily Kop-Coat NP-1 (7.6% IPBC), at 60% of mills surveyed in British Columbia in 1996 (Environment Canada 1998a). In 1996, lumber mills used 36 020 kg a.i. of IPBC for antisapstain purposes, reduced from the 47 540 kg used in 1993 (Environment Canada 1998a).

The mode of action of carbamate insecticides is primarily through acetylcholinesterase inhibition (Ecobichon 1991). The mode of action of IPBC, a fungicide and antimicrobial ingredient, however, is not clearly known, but may be linked to iodine toxicity (D. Nye 1998, Troy Corporation, Newark, New Jersey, pers. com.). The primary hydrolysis metabolite, propargyl butyl carbamate (PBC), has no iodine and is approximately 1000 times less toxic to fish and invertebrates than IPBC. Although iodine is an essential element, it is considered toxic at higher concentrations. Elemental iodine (I_2) is corrosive to exposed membranes and likely interferes with the permeability of the cell membranes and denatures proteins (Bowen, 1979; Santone and Prowis 1991).

The fate of IPBC in aquatic environments depends largely on physicochemical processes, particularly hydrolysis, rather than biotransformation processes. Microbial degradation is insignificant, due to the antimicrobial properties of IPBC (Schiefer 1990). EPL Bio-Analytical Services Inc. (1990a) reported IPBC to be hydrolytically stable at pH 5 with no signs of degradation, but at pH 7 the half-life was reported to be 139 d, and at pH 9, 0.947 d. Hydrolysis may be the main route of dissipation in aquatic environments, despite the dependence on alkalinity for catalyzation (USEPA 1997). Volatilization is not likely a significant dissipation route (Agriculture Canada et al. 1989), due to IPBC's low vapour pressure and its moderate solubility in water.

As IPBC is not known to be naturally occurring, all IPBC in the environment is expected to be from anthropogenic sources. Sources could include spills and other unpermitted discharges, permitted discharges from commercial facilities using the chemical, and discharges from products treated with IPBC (Henderson 1992). A sampling study initiated by the Fraser River Action Plan (FRAP) was abandoned following repeated sampling attempts at three separate mills using Kop-Coat NP-1, because IPBC concentrations in effluent samples were too low for accurate detection. The on-site dilution study performed used samples spiked with Kop-Coat NP-1 to determine dilution recoveries. IPBC recovery was unaffected by dilution with river water; recoveries reflected those anticipated by dilution calculations with distilled water. Although IPBC may be present at the effluent outfalls, dilution in sizable receiving waters would cause rapid dissipation (Szenasy 1998). Soil and water analysis near a leaking holding dike of a Kop-Coat NP-1 spill found no detectable IPBC where levels were expected to be 760 $mg{\cdot}L^{-1}$ (Koppers Company Inc. 1987).

Table 1. Water quality guidelines for IPBC for the protection of aquatic life (Environment Canada 1998b).

Aquatic life	Guideline value ($\mu g{\cdot}L^{-1}$)*
Freshwater	1.9*
Marine	NRG†

*Interim guideline.
†No recommended guideline.

Water Quality Guideline Derivation

The interim Canadian water quality guideline for IPBC for the protection of freshwater life was developed based on the CCME protocol (CCME 1991). For more information, see the supporting document (Environment Canada 1998b).

Freshwater Life

Acute toxicity data were found for six species of fish, ranging from a 96-h LC_{50} of 0.067 mg a.i.$\cdot L^{-1}$ for rainbow trout *(Oncorhynchus mykiss)* (Springborn Laboratories 1990) to a 96-h LC_{50} of 1.9 mg P-100$\cdot L^{-1}$ for coho salmon embryos *(O. kisutch)* (Farrell et al. 1998). Chronic toxicity data were found for only one fish species, the fathead minnow *(Pimephales promelas).* Embryos were exposed to technical grade IPBC in a flow-through system <24 h post-fertilization and were observed for survival at hatch (5 d) and for reduced larval weight and length at 35 d (Springborn Laboratories Inc. 1992). The 5-d LOEC for hatch survival was 0.057 mg$\cdot L^{-1}$, while the 35-d LOEC for reduced weight gain and growth (length) was 0.019 mg$\cdot L^{-1}$.

The toxicity of IPBC to invertebrates is represented by two species, *Daphnia magna* and *Hyalella azteca*, ranging from a 48-h LC_{50} of 0.04 mg P-100$\cdot L^{-1}$ (Farrell et al. 1998) to a 24-h LC_{50} of 1.419 mg a.i.$\cdot L^{-1}$ (Inversek Research International 1989), both for *D. magna*.

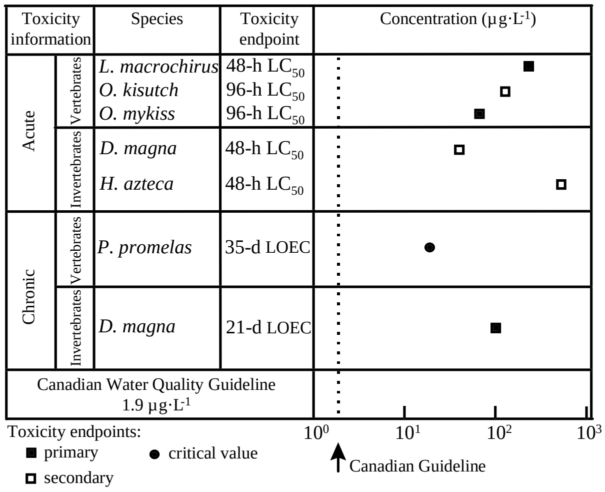

Figure 1. Select freshwater toxicity data for IPBC.

The only plant study located was an exposure of *Chlorella pyreniodosa* in a static replacement system. IPBC was algistatic at 0.5 mg$\cdot L^{-1}$ and algicidal at 1.0 mg$\cdot L^{-1}$ (United States Testing Company Inc. 1988). This study, however, was ranked as unacceptable.

The lowest concentration causing a toxic effect was the 35-d LOEC of 0.019 mg a.i.$\cdot L^{-1}$ for fathead minnows (Springborn Laboratories Inc. 1992). The guideline was derived by multiplying this LOEC by a safety factor of 0.1 (CCME 1991). This calculation results in an interim water quality guideline for IPBC of 1.9 µg$\cdot L^{-1}$ (0.0019 mg$\cdot L^{-1}$) for the protection of freshwater life

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Reference listing:

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Australian Government

Department of Health and Aged Care
Australian Industrial Chemicals Introduction Scheme

Carbamic acid, butyl-, 3-iodo-2-propynyl ester (IPBC)

Evaluation statement

22 December 2022

Table of contents

Contents

AICIS evaluation statement

Subject of the evaluation

Carbamic acid, butyl-, 3-iodo-2-propynyl ester (IPBC)

Chemical in this evaluation

Name	CAS registry number
Carbamic acid, butyl-, 3-iodo-2-propynyl ester	55406-53-6

Reason for the evaluation

Evaluation Selection Analysis indicated a potential environmental risk.

Parameters of evaluation

The chemical is listed on the Australian Inventory of Industrial Chemicals (the Inventory). This evaluation considers the environmental risks associated with the industrial use of 3-iodo-2-propynyl butylcarbamate (IPBC).

IPBC has been assessed for the risk to the environment according to the following parameters:

- Default domestic introduction volumes of 100 tonnes (t) per annum
- Industrial uses listed in the 'Summary of Use' section
- Exposure to aquatic environments and soil via release to stormwater and sewage treatment plants (STPs) as a result of consumer and commercial uses.

Summary of evaluation

Summary of introduction, use and end use

The chemical (IPBC) may be used as a fungicide and preservative in the following industrial products according to international use data:

- Adhesive and sealant products
- Apparel and footwear care products
- Arts, crafts and hobby products
- Lubricant and grease products
- Personal care products - limited environmental release
- Paint and coating products
- Plastic and polymer products
- Fabric, textile and leather products not covered by other end uses
- Ink, toner and colourant products
- Automotive care products

- Cleaning and furniture care products
- Extractive products not covered by other end uses
- Paper products
- Personal care products not covered by other end uses.

There is no information available on the volumes of IPBC in use in Australia. Data from international jurisdictions indicate that it is registered for industrial use in the European Union (EU) at 10–100 tonnes annually. Total use volume in the Nordic countries (Norway, Sweden, Finland, and Denmark) was 5–512 tonnes per year per country since 2001. IPBC is used in the USA at 20–46 t/year (2017–2020), in Canada at 0–1 t/year, and in Japan at 1–1 000 t/year.

Environment

Summary of environmental hazard characteristics

According to domestic environmental hazard thresholds and based on the available data the chemical is:

- Not Persistent (Not P)
- Not Bioaccumulative (Not B)
- Toxic (T).

Environmental hazard classification

The chemical (IPBC) satisfies the criteria for classification according to the Globally Harmonized System of Classification and Labelling of Chemicals (GHS) for environmental hazards as follows. This does not consider classification of physical hazards and health hazards:

Environmental Hazard	Hazard Category	Hazard Statement
Hazardous to the aquatic environment (acute / short-term)	Aquatic Acute 1	H400: Very toxic to aquatic life
Hazardous to the aquatic environment (long-term)	Aquatic Chronic 1	H410: Very toxic to aquatic life with long lasting effects

Summary of environmental risk

The chemical (IPBC) is a synthetic preservative and biocide. It is used as a preservative in a wide range of consumer and commercial products, including personal care products. It is also used as a film preservative in paints and coatings.

The chemical is not persistent in the environment and has a low potential for bioaccumulation. It is very toxic to aquatic life across all trophic levels, and to terrestrial plants. However, at typical environmental concentrations, IPBC readily degrades into a less harmful substance in waters and soils. While IPBC poses a risk to aquatic ecosystems if emitted in high concentrations, the rapid degradation of the chemical and the resulting reduction in toxicity are expected to mitigate the risk to the environment.

Diffuse releases of IPBC occur to surface waters and soils both directly and indirectly. Direct releases occur through the leaching of treated building facades during rainfall events, while the use of some products containing IPBC, such as personal care products, results in indirect releases via STP. The environmental concentrations of IPBC in waters and soils from both diffuse pathways is expected to be below levels of concern.

Point source releases of IPBC at high concentrations, such as the observed releases from paint manufacturers internationally, have the potential to cause adverse effects in the environment if released without treatment. However, as these releases are expected to occur below concentrations that may affect STP function, STP treatment is expected to reduce the concentrations of these IPBC releases to concentrations below levels of concern.

Conclusions

The conclusions of this evaluation are based on the information described in this statement.

The Executive Director is satisfied that the identified environmental risks can be managed within existing risk management frameworks. This is provided that all requirements are met under environmental, workplace health and safety and poisons legislation as adopted by the relevant state or territory.

Note: Obligations to report additional information about hazards under *Section 100* of the *Industrial Chemicals Act 2019* apply.

Supporting information

Rationale

This evaluation considers the environmental risks associated with the industrial uses of IPBC. This chemical is used as a fungicide and preservative (including film preservative), in personal care products, paints and coatings, textiles, paper, ink, and construction products.

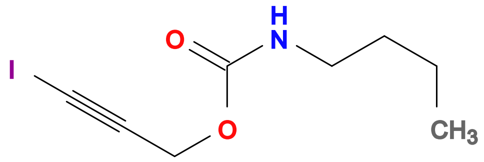
The evaluation selection analysis (ESA) of IPBC found that the industrial use of the substance may be of concern to the environment based on the high toxicity of the chemical to aquatic organisms and potential emissions to surface waters in the treated effluent discharged from sewage treatment plants (STP). This assessment will evaluate the potential for emissions of the chemical to the environment in Australia and whether risk reduction measures are required for industrial uses of this chemical.

Chemical identity

Chemical identity information for IPBC and its main expected environmental degradant, PBC (CAS RN 76114-73-3) are presented below.

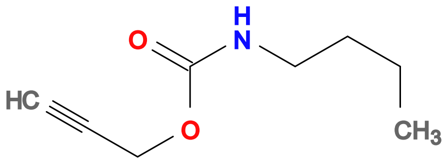
The chemical (IPBC) belongs to a large family of carbamate biocides. The target organisms of these biocides are dependent on the selection of *N*-substituent and *O*-substituent present on either side of the carbamate group, which has given rise to carbamate insecticides, herbicides, and fungicides.

The mode of action of these pesticides is also variable. Carbamate insecticides are generally *N*-methyl carbamates that inhibit acetylcholinesterase (AChE). The *N*-substituent is typically an aromatic moiety in carbamate herbicides, or a benzimidazole moiety in carbamate fungicides such as carbendazim (CAS RN 10605-21-7) (Machemer and Pickel 1994). IPBC is a carbamate ester of 3-iodo-2-propynol, bearing a butyl substituent on the nitrogen atom, however, the toxic mode of action of IPBC is not well understood. It has been proposed that IPBC affects cell membrane permeability in fungi (FRAC 2022):

Chemical name	Carbamic acid, butyl-, 3-iodo-2-propynyl ester
CAS No.	55406-53-6
Synonyms	3-iodo-2-propynyl butylcarbamate IPBC iodopropynyl butylcarbamate (INCI name) Iodocarb 3-iodoprop-2-yn-1-yl butylcarbamate 3-iodo-2-propynyl *N*-butylcarbamate carbamic acid, N-butyl-, 3-iodo-2-propyn-1-yl ester
Structural formula	
Molecular formula	$C_8H_{12}INO_2$
Molecular weight (g/mol)	281.09
SMILES	CCCCNC(=O)OCC#CI

Related chemical

The chemical (IPBC) degrades to carbamic acid, butyl-, 2-propynyl ester, or propargyl butylcarbamate (PBC), in the environment. As such, the physical and chemical properties, and the environmental fate and hazards of PBC are relevant to the environmental risk assessment of IPBC. The chemical identity and properties of PBC are detailed below:

Chemical name	Carbamic acid, butyl-, 2-propynyl ester
CAS No.	76114-73-3
Synonyms	propargyl butylcarbamate (PBC) prop-2-yn-1-yl butylcarbamate 2-propynyl *N*-butylcarbamate butylcarbamic acid propargyl ester
Structural formula	
Molecular formula	$C_8H_{13}NO_2$
Molecular weight (g/mol)	155.19
SMILES	CCCCNC(=O)OCC#C

Relevant physical and chemical properties

Measured physical and chemical properties for IPBC were obtained from the registration dossier for 3-iodo-2-propynyl butylcarbamate submitted under the Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH) legislation in the European Union (REACHa n.d.), and from the PubChem compound summary for 3-iodo-2-propynyl butylcarbamate (NCBI n.d.). Measured physical and chemical properties for PBC were obtained from the REACH dossiers for carbamic acid, N-butyl-, 2-propyn-1-yl ester (REACHb n.d.; REACHc n.d.). The Henry's Law constants for both chemicals were calculated (calc.) from the experimental (exp.) values for water solubility and vapour pressure using EPISuite (US EPA 2017):

Chemical	IPBC	PBC
Physical form	Solid	Liquid
Melting point	66°C (exp.)	-2°C (exp.)
Boiling point	Decomposition before boiling	224°C (exp.)
Vapour pressure	0.0038 Pa (20°C, exp.)	4.7 Pa (20°C, exp.)
Water solubility	168 mg/L (20°C, exp.)	4900 mg/L (20°C, exp.)
Henry's law constant	6.36×10^{-3} Pa·m^3/mol (20°C, calc.)	0.148 Pa·m^3/mol (20°C, calc.)
Ionisable in the environment?	No	No
pKa	-	-
log K_{ow}	2.81 (25°C, exp.)	1.9 (25°C, exp.)

The chemical (IPBC) is an off white solid that melts at 66°C. It is moderately soluble in water. It has a low volatility from water and when in the solid form.

Its degradation product (PBC) is a liquid at ambient temperature. It is moderately volatile, and readily soluble in water.

Introduction and use

Australia

No specific Australian introduction volume information has been identified.

The chemical has a reported use as a preservative in baby wipes (NICNAS 2013).

The chemical has non-industrial uses in Australia. It is an approved active constituent in pesticides by the Australian Pesticides and Veterinary Medicines Authority (APVMA). Four IPBC-containing products are approved for use by the APVMA for the protection of freshly sawn timber from sapstain and mould, for the dressing of pruning wounds in fruit orchards, and for the prevention of fungal disease in grapevine (APVMA).

International

Available information indicates that IPBC is an antifungal preservative (biocide) and film preservative added to many personal care products, paints and coatings, construction products, cleaning products, and other products worldwide (NCBI; Nordic Council of Ministers; REACHa; US NIEHS).

The chemical (IPBC) is used as a preservative internationally in both leave-on and rinse-off cosmetic and personal care products. The typical IPBC concentration in these products ranges from 0.005 to 0.1%. IPBC is present in some products at concentrations 0.1–5%, with the majority below 0.5%. IPBC is also used in personal care products that are primarily disposed of to landfill (wipes, nail polish).

The chemical (IPBC) is also used in cleaning and washing agents, including automotive care products, cleaning wipes, surface cleaners, and toilet cleaners. It is used in leather treatment products and shoe polish.

The chemical is used as a preservative and film preservative in surface coatings, construction products, and arts, crafts, and hobby products. This use encompasses paints, primers, lacquers and varnishes, adhesives and binding agents, sealants and fillers, surface treatment products, corrosion inhibitors, lubricants and additives, and various materials used for construction. IPBC is used in these products at concentrations ranging from 0.1 to 15%, with the majority below 1%.

The chemical is also used to preserve wood, cans, textiles, rubber, plastics, (metal) cutting fluids, oil recovery drilling mud/packer fluids, paper, and ink. It may be applied to heating, ventilation, and air conditioning (HVAC) ducts to prevent microbial growth.

Data from international jurisdictions may include industrial and non-industrial uses of IPBC. IPBC is registered under REACH for use in the European Economic Area (EEA) at up to 100 tonnes annually (REACHa n.d.). This registration volume does not include biocidal uses regulated under the Biocidal Products Regulation (BPR) (ECHAa). However, some of these uses, including uses in “Product Type 6” (Preservatives for products during storage) and "Product Type 13” (Working or cutting fluid preservatives), are categorised as industrial uses in Australia. IPBC has been used in the Nordic countries (Norway, Sweden, Finland, and Denmark) at 5–512 tonnes per year per country since 2001 (Nordic Council of Ministers n.d.). These volumes are total introduction volumes per country and include uses not registered under REACH. IPBC was used in the USA at 20–46 t/year between 2017 and 2020, according to data extracted from the 2020 Toxics Release Inventory Factsheet for IPBC (US EPAa). IPBC is used in Canada at 0–1 t/year (OECD), and in Japan at 1–1 000 t/year (NITE).

Existing Australian regulatory controls

Public

The chemical is listed in the *Poisons Standard, Standard for the Uniform Scheduling of Medicines and Poisons (SUSMP)* (TGA 2022) as follows:

Schedule 5

‘3-IODO-2-PROPYNYL BUTYL CARBAMATE (Iodocarb) in preparations containing 10 per cent or less of 3-iodo-2-propynyl butyl carbamate **except**:

a) in aqueous preparations not for cosmetic use containing 10 per cent or less 3-iodo-2-propynyl butyl carbamate; or

b) in cosmetic preparations (other than aerosolised preparations) containing 0.1 per cent or less of 3-iodo-2-propynyl butyl carbamate.'

Schedule 6

'3-IODO-2-PROPYNYL BUTYL CARBAMATE (Iodocarb) **except**:

a) when included in Schedule 5;

b) in aqueous preparations not for cosmetic use containing 10 per cent or less of 3-iodo-2-propynyl butyl carbamate (Iodocarb); or

c) in cosmetic preparations (other than aerosolised preparations) containing 0.1 per cent or less of 3-iodo-2-propynyl butyl carbamate.'

Environment

The chemical IPBC is not subject to specific environmental regulatory controls.

International regulatory status

United Nations

The chemical (IPBC) is not currently identified as a Persistent Organic Pollutant (POP) (UNEP 2001), ozone depleting substance (UNEP 1987), or hazardous substance for the purpose of international trade (UNEP & FAO 1998).

Canada

The Canadian Council of Ministers of the Environment has set an interim long term concentration threshold of 1.9 micrograms per litre (µg/L) for IPBC in the Water Quality Guidelines for the Protection of Aquatic Life (CCME).

European Union

The chemical (IPBC) is approved for use as a biocide under the EU Biocidal Products Regulation ((BPR, Regulation (EU) 528/2012)) for three product types: preservative for products during storage (PT06), wood preservatives (PT08), working or cutting fluid preservatives (PT13) (ECHAa).

The chemical (IPBC) is being reviewed for use as a biocide under the EU Biocidal Products Regulation ((BPR, Regulation (EU) 528/2012)) for the following product types: film preservatives (PT07), fibre, leather, rubber, and polymerised materials preservatives (PT09), construction material preservatives (PT10) (ECHAa).

Restrictions on the use of IPBC as a preservative in cosmetics are listed in the Cosmetic Products Regulation, Annex V. IPBC is not to be used in oral and lip products, body lotion and body cream, and products for children under 3 years of age (except in bath products / shower gels and shampoos). The IPBC concentration must not exceed 0.01% in leave-on products, 0.02% in rinse-off products, and 0.0075% in deodorants and anti-perspirants (EC).

Environmental exposure

The chemical (IPBC) is a synthetic chemical and the occurrence of this substance in the environment stems exclusively from human activity. Industrial uses of IPBC are expected to result in both direct and indirect releases into surface waters and soils.

The chemical is also used as a dry-film preservative in paints, coatings, and other construction materials. During this use, IPBC continuously leaches to the top layer of treated surfaces over time to provide long-term protection from microbial degradation. During rainfall events, IPBC is washed off treated building surfaces to become a component of building run-off (Bollmann et al. 2014; Bollmann et al. 2017; Burkhardt et al. 2012). The building run-off containing this chemical is then discharged directly onto soil and into surface waters through the stormwater drainage systems. These sources may contribute to cumulative diffuse emissions of IPBC into the environment.

Indirect source emissions of IPBC into the environment include “down the drain” disposal of IPBC-containing products. Some IPBC-containing products, such as personal care products, will be washed down drains during typical use, while some paint and adhesive residues may be improperly disposed of down drains. Products that are washed “down the drain” are expected to be treated in municipal STPs.

Factories that manufacture or formulate products containing IPBC, such as those formulating paints, cutting fluids, and personal care products, may release IPBC in wastewater. These wastewaters will be also treated in industrial and municipal STPs.

Once in STP, the removal of IPBC will occur depending on specific degradation and partitioning processes. IPBC that is not removed during these processes will be released to rivers or oceans in STP effluent. As some IPBC may adsorb to STP sludge, application of STP biosolids to land may be considered an exposure route for soil organisms (Struijs 1996).

Non-industrial uses of IPBC are also expected to contribute to emission sources into the environment. IPBC-containing products are registered with the Australian Pesticides and Veterinary Medicines Authority (APVMA) for use as antifungal and anti-sapstain agent on freshly sawn timber, in fruit orchards, and in vineyards. Release from these sources will occur to soil, surface waters, and possibly groundwater. The extent of release from non-industrial sources cannot be accurately quantified.

Environmental fate

Partitioning

The chemical (IPBC) is expected to partition to the water and soil compartments when released into the environment.

The chemical (IPBC) is an organic chemical that is neutral in the environmental pH range (pH = 4–9), is moderately soluble in water, and has low volatility. The Henry’s Law constant of 6.36×10^{-3} Pa·m^3/mol suggests that it will be very slightly volatile from water and moist soil. IPBC has moderate lipophilicity with a log K_{OW} value of 2.81. Reported organic carbon adsorption coefficients (K_{OC}) of 61–309 L/kg, depending on the type of soil, indicate it will have medium to high mobility in soil. IPBC adsorption does not appear to be highly correlated with soil organic matter content, clay content or cation exchange capacity, although differences in relative adsorption between soils may be related to differences in the extent of test item degradation (ECHAb; REACHa). In a monitoring study of biocides in surface waters, a significant proportion of the detected IPBC was found in suspended particulate (i.e., attached to particles in the water) rather than in the dissolved state (Paijens et al. 2020; Paijens et al. 2021).

Calculations with a standard multimedia partitioning (fugacity) model assuming equal and continuous distributions to the water and soil compartments (Level III output), predict that IPBC will mostly partition to water (65.3–66.1%, depending on K_{OC}), some will remain in soil (33.8–34.6%), and negligible quantities will partition to sediment (<0.1%). Following release to surface waters in STP effluent, calculations with a fugacity model with sole release to the water compartment predict that IPBC will mostly remain in water (99.9%), and very small quantities will partition to sediment (0.1%) (US EPA 2017).

IPBC is expected to degrade into propargyl butylcarbamate (PBC) and iodine species in the environment through abiotic and microbial processes. Fugacity modelling suggests that the partitioning of the PBC degradant between the environmental compartments will be similar to the partitioning of IPBC.

Degradation

The chemical IPBC is expected to undergo rapid primary degradation at environmentally relevant concentrations in water/sediment systems, and in soil. The primary degradation product, PBC, is expected to undergo further degradation.

Biodegradation studies on IPBC and PBC have been hindered by the toxicity of the chemicals to micro-organisms (Carbajo et al. 2015; REACHa). Degradation of IPBC and PBC in ready biodegradability screening tests in water, following OECD Test Guidelines (TG) 301 B and 301 F for IPBC, and OECD TG 301 F for PBC, reached 0 to 5% after 28 days. It is likely that the high concentrations of test materials (15–100 mg/L) in these tests had an inhibitory effect on the biosolids (REACHa; REACHb).

Degradation studies performed at environmentally relevant concentrations indicate the chemical is unlikely to persist in the environment. An inherent biodegradability test (OECD TG 302 B, modified Zahn-Wellens/EMPA Test) showed that IPBC degrades rapidly into propargyl butylcarbamate (PBC) in water. The biodegradation process was monitored by specific analysis of IPBC and the degradation product PBC in the aqueous phase and in the biosolids, rather than Dissolved Organic Carbon (DOC) removal. The ratio of inoculum to test material was higher than recommended in the guidelines to avoid a toxic effect of IPBC on the biosolids at higher concentrations, with an initial IPBC concentration of 1 mg/L. In the aqueous phase, the IPBC concentration decreased to 0.9–1.5% of the nominal value within two hours of incubation. In these samples, 85–89% of the maximum theoretical amount of the degradation product PBC were detected. The concentration of PBC in the water phase continuously decreased during the test period and was below the limit of quantification (LOQ) of 0.01 mg/L after 21 days of incubation. No significant amounts of IPBC or PBC were found in the biosolids (REACHa).

Another laboratory study was performed to investigate the biodegradation of IPBC in anaerobic conditions in a water/sediment system. IPBC underwent rapid primary degradation with a half-life of 1.5 hours. The major identified degradation product was PBC, which also degraded with a half-life of 11.5 days (REACHa).

Both IPBC and PBC degrade rapidly in soil at 22°C. A laboratory study was performed to investigate the aerobic soil metabolism of IPBC at temperatures of 22°C and 5°C. The initial IPBC concentration was 1 mg/kg of soil. IPBC rapidly degraded in soil at 22°C with a half-life of 2.13 hours. The primary soil metabolite was propargyl butyl carbamate (PBC). PBC further degraded in soil at 22°C with a half-life of 4.30 days. IPBC degradation was slowed at 5°C, resulting in a soil half-life of 8.60 hours. The rate of PBC degradation decreased to a greater extent than that of IPBC as temperature decreased from 22 to 5°C. Fourteen days after treatment, greater than 60% of the applied ^{14}C had been mineralized to $^{14}CO_2$ at 22°C whereas less than 5% had been mineralized at 5°C. IPBC and PBC degradation in soil is expected to be primarily microbially mediated but nonbiological processes may be involved (Juergensen et al. 2000; REACHa).

In another study performed according to OECD TG 217, with an initial IPBC concentration of 10 mg/kg soil, IPBC degraded rapidly in soil with a half-life of 1.05 days. However, the fast degradation in the beginning of the two-phase degradation kinetics was followed by a phase with a nearly constant residual IPBC concentration of 0.013 mg/kg soil (Bollmann et al. 2017).

Samples from soils impacted by IPBC spills of 700–2000 mg/L had undetectable quantities of IPBC (Detection limit 10 mg/L) 28 days following the incident, further supporting the rapid degradation of IPBC in soil (Juergensen et al. 2000).

The chemical (IPBC) was found to be hydrolytically stable in sterile aqueous solutions at ambient temperatures at pH 5 and 7 (ECHAb; REACHa). Conflicting observations were made for IPBC hydrolysis at pH 9: while a short half-life <1 day was measured in one study (REACHa), others reported long half-lives >200 days at this pH (ECHAb).

The chemical (IPBC) may undergo phototransformation in the solid state but appears to be stable to irradiation in water. No phototransformation was observed in sterilised aqueous buffer solution at pH 7 and natural pond water at pH 8.5 within 3 days of continuous irradiation (ECHAb). The decomposition rate of IPBC in wood slices dipped in an IPBC solution reached 25 to 60% after 25 to 50 days of irradiation (ECHAb ; Lee et al. 1991). A comparative study of the photodegradation of four organoiodine wood preservatives in ethanol solution and impregnated into wood showed that IPBC was the most stable to irradiation (Lee et al. 1991).

Both IPBC and PBC are rapidly degraded in the atmosphere by reactions with photogenerated hydroxyl radicals. Atmospheric oxidation modelling indicates that both chemicals have half-lives of approximately 5 hours (US EPA 2017).

Bioaccumulation

The chemical (IPBC) is not expected to bioaccumulate in organisms.

The measured log K_{OW} for IPBC is below the domestic threshold for bioaccumulation potential. A bioconcentration factor (BCF) of < 4.5 L/kg in fish has been reported for this chemical (Juergensen et al. 2000), which indicates a low potential to bioaccumulate. The calculated biotransformation half-life of IPBC normalised to 10 g fish is 23 minutes (US EPA 2017).

Environmental transport

The chemical (IPBC) is unlikely to undergo long-range environmental transport.

The chemical (IPBC) is not persistent in the aquatic environment or in soil and is therefore not expected to undergo long-range transport in water or contaminate groundwaters.

The chemical (IPBC) has a short calculated atmospheric half-life (5 hours), which limits the potential to undergo long-range transport through the atmosphere in the vapour phase. A calculated logarithmic octanol-air coefficient (log K_{OA}) of 9.25 (US EPA 2017) indicates a potential for IPBC to be transported on aerial particles. IPBC was detected in the gas phase in Arctic air samples at a Norwegian background station. However, local contamination sources from the use of the chemical as a wood preservative and uses in cosmetics and personal care products could not be excluded (Röhler et al. 2020; Röhler et al. 2021).

Predicted environmental concentration (PEC)

The estimated environmental concentration of IPBC in inland surface waters is 0.38 micrograms per litre (µg/L), and the PEC in STP influent waters is 150 µg/L, based on international monitoring data. The PEC in soils amended with STP biosolids is 0.11 mg/kg soil dry weight (dw), based on default assumptions on use volumes and release.

No domestic environmental monitoring data were identified. Standard exposure modelling assuming a default introduction volume of 100 tonnes per year, 100% release to sewers, and 68% mitigation within STPs, gives calculated IPBC concentrations of 18 µg/L in STP effluents (Struijs 1996).

However, modelled data appear to be an overestimate based on international STP effluent monitoring data. IPBC has been detected in surface waters, including STP influents and effluents, in several studies conducted in Sweden, France, Switzerland, the Netherlands, and the USA (Guardian et al. 2021; Morasch et al. 2010; NORMAN ; Norstöm 2009; Paijens et al. 2021). While the concentration of IPBC was frequently below the limit of detection, IPBC was detected in surface waters on at least one occasion in each of these studies, at concentrations up to 0.38 µg/L in an STP effluent in Switzerland (Morasch et al. 2010).

The chemical (IPBC) was detected at high concentrations in the influent water to the treatment plants at two paint industries in Sweden (150 and 32 µg/L). The effluent waters from these companies also contained IPBC but at much lower levels, respectively 0.032 and 0.22 µg/L, indicating removal efficiencies of >99% within STPs. The presence of IPBC was monitored at another potential point source, a landfill. IPBC was present at a concentration of 35 nanograms per litre (ng/L) in the landfill leachate prior to treatment but the concentration was below the limit of detection of 10 ng/L after biological treatment (Norstöm 2009).

The chemical has been detected in stormwater runoff sourced from façade leachate from buildings during rain events. IPBC was present in two out of eight sampled rain events (Paijens et al. 2021). IPBC was detected at total concentrations of 3.6–24 ng/L (dissolved state and particles) in an underground stormwater storage pond in Paris (Paijens et al. 2020).

As the measured STP effluent concentration of 0.38 µg/L (Morasch et al. 2010) is the highest measured value of IPBC in STP effluents, surface waters, or stormwater/leachates, it is taken as the worst-case PEC for surface waters. The highest IPBC concentration in an STP influent (150 µg/L) (Norstöm 2009) is used as the PEC for wastewaters entering STPs.

Very limited monitoring data in soil are available. In one study, façade leachates and soil samples from the bottom of the façades of 17 recently built or renovated houses in Denmark were analysed for biocides. IPBC was detected in the façade leachates of some houses, but was below the detection limit of 0.9 µg/kg soil in all soil samples (Bollmann et al. 2017).

The calculated PEC in soil amended with biosolids is 0.11 mg/kg soil dw at the time of application, based on an influent concentration of 56 µg/L (calculated assuming a default introduction volume of 100 t/year and 100% release to sewer), 3% partitioning of IPBC to biosolids in STPs, typical biosolids application rates and a soil bulk density of 1 500 kilograms per cubic metre (kg/m^3) (EPHC 2009; Struijs 1996). The concentration of IPBC in soil is expected to drop rapidly after application based on half-lives of 2–9 hours in soil at 5–22°C (REACHa).

Environmental effects

The chemical (IPBC) causes long lasting toxic effects at low concentrations in aquatic organisms across multiple trophic levels. IPBC is also toxic to some terrestrial organisms, particularly plants.

Effects on Aquatic Life

Acute toxicity

Acute toxicity data are available for 15 species of freshwater, marine, and euryhaline aquatic organisms, including fish, invertebrates, and algae (Adam et al. 2009; Coors et al. 2012; Coors et al. 2014; Farrell et al. 1998; REACHa; US EPAb). The following are the most sensitive acute median lethal concentrations (LC50) for fish and invertebrates and median effect concentrations (EC50) on growth rate for algae measured after standard exposure times (96 h for fish, 48 h for invertebrates, 72 h for algae), retrieved from the REACH dossier for IPBC (REACHa) and from the literature (Farrell et al. 1998). Relevant endpoints (EC50 and the concentration at which 10% inhibition is observed, EC10) for the toxicity of IPBC to aquatic micro-organisms are also reported in the table (Carbajo et al. 2015; REACHa):

Taxon	**Endpoint**	**Method**
Fish	96h LC50 = 67 µg/L	*Oncorhynchus mykiss* (rainbow trout) mortality Flow-through conditions EPA OPP 72-1
Invertebrate	48h LC50 = 40 µg/L	*Daphnia magna* (water flea) mortality Semi-static conditions Nominal concentrations (within 30% of the measured concentrations) EPA Guidelines
Algae	72h EC50 = 53 µg/L	*Desmodesmus subspicatus* (green algae) growth rate Static conditions OECD TG 201
Aquatic micro-organisms	24h EC50 = 119 µg/L	*Tetrahymena thermophila* (ciliate) biomass Standard Operational Procedure Guideline of Protoxkit F^{TM}
STP microflora	3h EC50 = 26 mg/L 3h EC10 = 6 mg/L	STP biosolids Respiration inhibition OECD TG 209

The chemical (IPBC) is very toxic to all aquatic organisms. All acute LC50 values for fish are below 1 mg/L. The endpoints for freshwater fish range from 67 µg/L for rainbow trout (*Oncorhynchus mykiss*) to 0.43 mg/L for zebra fish (*Danio rerio*). For marine and euryhaline fish, LC50 endpoints of 0.37 mg/L for starry flounder (*Platichthys stellatus*) and 0.41 mg/L for sheepshead minnow (*Cyprinodon variegatus*) have been measured (REACHa). IPBC affects fish behaviour and physiology at sub-lethal concentrations. Detrimental effects on the olfactory response of coho salmon (*Oncorhynchus kisutch*) were observed at concentrations as low as 1.3 µg/L (electro-olfactogram amplitude EC50) (Jarrard et al. 2004).

A greater variation in the toxicity of IPBC to invertebrates is observed, with endpoints ranging from 0.023 to 2.9 mg/L. A mobility EC50 of 23 µg/L was measured for the American oyster (*Crassostrea virginica*), although after a longer exposure time (96 h instead of the standard 48 h exposure for invertebrates) (US EPAb). For freshwater invertebrates, the lowest measured endpoint is an LC50 of 40 µg/L for *Daphnia magna* (Farrell et al. 1998), although a range of toxicities were observed in different studies for this species. The least sensitive invertebrate species studied is the opossum shrimp *Neomysis mercedis* with a 48 h LC50 of 2.9 mg/L (Farrell et al. 1998), although a lower 96 h LC50 of 0.090 mg/L was measured for another species of opossum shrimp, *Americamysis bahia* (US EPAb).

The acute toxicity of IPBC to two species of freshwater algae, *Desmodesmus subspicatus* (growth rate EC50 of 53 µg/L) (REACHa), and *Raphidocelis subcapitata* (biomass EC50 of 39 µg/L) (Coors et al. 2014), are in the same range.

The chemical (IPBC) is moderately to highly toxic to aquatic microorganism species found in STPs. The most sensitive species tested is the ciliate *Tetrahymena thermophila*, with an EC50 (growth) of 119 µg/L after 24 h IPBC exposure. The EC50 for bioluminescence inhibition of the bacterium *Vibrio fischeri* after 30 min IPBC exposure is 3.9 mg/L (Carbajo et al. 2015). Ecotoxicity endpoints for the inhibition of total respiration of biosolids from STPs after 3 h IPBC exposure are EC50 values of 26–44 mg/L, and an EC10 of 6 mg/L (Carbajo et al. 2015; REACHa).

The primary degradation product of IPBC is PBC. Based on the available ecotoxicity data, PBC is not expected to contribute significantly to the observed toxicity of IPBC. Measured ecotoxicity endpoints across the three trophic levels for the PBC degradant suggest that it is at least 1 000 times less acutely toxic to aquatic life than IPBC. The following are the most sensitive acute median lethal concentration (LC50) for fish, and median effect concentrations (EC50) on mobility for invertebrates and growth rate for algae, measured after 96 h exposure for fish and algae and 48 h for invertebrates, retrieved from the REACH dossiers for PBC and the Biocidal Assessment Report for the use of IPBC in product-type 8 (ECHAb; REACHb; REACHc; REACHd):

Taxon	Endpoint	Method
Fish	96h LC50 = 85 mg/L (PBC)	*Oncorhynchus mykiss* (rainbow trout) mortality Flow-through conditions US EPA FIFRA 72-1
Invertebrate	48h EC50 = 54 mg/L (PBC)	*Daphnia magna* (water flea) mobility Static conditions OECD TG 202
Algae	96h EC50 > 100 mg/L (PBC)	*Raphidocelis subcapitata* (green algae) growth rate Static conditions Nominal concentrations OECD TG 201

Chronic toxicity

The following measured no-observed-effect concentrations (NOEC) for model organisms across three trophic levels were obtained from the REACH dossier for IPBC and the US EPA ECOTOX database (REACHa ; US EPAb):

Taxon	Endpoint	Method
Fish	35d NOEC = 8.4 µg/L	*Pimephales promelas* (fathead minnow) larval length and weight Flowthrough conditions EPA OPP 72-4
Invertebrates	21d NOEC = 50 µg/L	*Daphnia magna* (water flea) mortality Flowthrough conditions Nominal concentrations OECD TG 202
Algae	72h NOEC = 4.6 µg/L	*Desmodesmus subspicatus* (green algae) growth rate Static conditions Mean measured OECD TG 201

Chronic effects are observed at very low IPBC concentrations, particularly for fish and algae. Fathead minnow larvae exposed to IPBC for 35 days following egg fertilisation have significantly reduced weights and lengths at concentrations ≥ 19 µg/L (lowest observed effect concentration, LOEC) (REACHa). The growth rate of *Desmodesmus subspicatus* is affected by IPBC at concentrations ≥ 10 µg/L (LOEC) (REACHa).

Effects on terrestrial Life

The following measured endpoints were obtained from the REACH dossier for IPBC, the European Commission’s Biocidal Assessment Report for the use of IPBC in product-type 8 (PT8, wood preservatives), the US EPA ECOTOX database, and the literature (ECHAa ; Guimarães et al. 2018; REACHa; US EPAb):

Taxon	Endpoint	Method
Terrestrial plants	21d EC50 = 4.9 mg/kg soil dry weight (dw) 21d LOEC = 1.3 mg/kg soil dw	*Avena sativa* (oat) fresh weight Nominal concentrations OECD TG 208
Annelids	14d LC50 > 1 000 mg/kg soil dw 14d NOEC = 1 000 mg/kg soil dw	*Eisenia fetida* (earthworm) mortality OECD TG 207
Arthropods	28d EC10 = 16 mg/kg soil dw 28d LC10 = 12 mg/kg soil dw	*Folsomia candida* (springtail) reproduction (EC10) and mortality (LC10) OECD TG 232
Birds	8d LC50 > 5620 ppm in diet 14d LD50 = 749 mg/kg organism (org) 21d NOEL = 292 mg/kg org	*Colinus virginianus* (northern bobwhite quail) mortality
Soil micro-organisms	28d EC50 = 312 mg/kg soil dw	Soil microflora inhibition of total respiration Nominal concentrations OECD TG 217

The chemical is highly toxic to terrestrial plants, harmful to springtails, and slightly toxic/practically nontoxic to earthworms and birds.

Endocrine effects/activity

The chemical does not appear to cause adverse effects mediated by an endocrine mode of action, however, the endocrine disruption potential of IPBC is currently under assessment by ECHA (REACHa).

In a study incorporating data from in vitro assays, chemical descriptors, and biological pathways to establish toxicity profiles of human endocrine activity for 309 chemicals, IPBC showed a low to moderate potential endocrine activity in some nuclear receptor binding assays (glucocorticoid receptor, peroxisome proliferator-activated receptor, or pregnane X receptor) and assays on xenobiotic-metabolizing enzymes activity (including cytochrome P450s, aromatase). IPBC showed no significant activity in estrogen receptor, thyroid receptor, or androgen receptor binding assays (Reif et al. 2010).

No evidence of endocrine activity in other organisms was identified. Studies on carcinogenicity and toxicity to reproduction in mice and rats did not show any significant effects of IPBC (REACHa).

Predicted no-effect concentration (PNEC)

The PNEC for aquatic organisms in this evaluation is 0.46 µg/L. The algal 72 hours NOEC value of 4.6 µg/L was used to derive the PNEC for this chemical. An assessment factor of 10 was used as there are sufficient reliable acute and chronic aquatic toxicity data available for three trophic levels.

Based on the most sensitive aquatic micro-organisms tested, the ciliate *Tetrahymena thermophila* with a 24 h biomass EC50 of 0.119 mg/L, and an assessment factor of 10, the PNEC for micro-organisms is 11.9 µg/L (EPHC 2009). However, a more representative PNEC for the microflora in STPs is obtained by considering the ecotoxicity endpoints for biosolids (EC10 = 6 mg/L), and an assessment factor of 10, leading to a PNEC for STP micro-organisms of 0.6 mg/L.

A PNEC of 98 µg/kg for soil was derived from the lowest measured endpoint for terrestrial organisms (EC50 = 4.9 mg/kg dry soil for oat), using an assessment factor of 50, as long-term toxicity tests of two trophic levels (springtail and soil micro-organisms) are available in addition to short-term ecotoxicity data across multiple trophic levels of terrestrial life (EPHC 2009).

Categorisation of environmental hazard

The categorisation of the environmental hazards of the assessed chemical according to domestic environmental hazard thresholds is presented below:

Persistence

Not Persistent (Not P). Based on measured half-lives in water and soil, IPBC is categorised as Not Persistent.

Bioaccumulation

Not Bioaccumulative (Not B). Based on measured log K_{OW} and BCF values below domestic thresholds, IPBC is categorised as Not Bioaccumulative.

Toxicity

Toxic (T). Based on available ecotoxicity values below 1 mg/L and evidence of high chronic toxicity to aquatic organisms, IPBC is categorised as Toxic.

Environmental risk characterisation

The chemical (IPBC) is very toxic to aquatic organisms, including aquatic primary producers such as algae. This toxicity is necessary for the functional use of the substance as preservative and industrial biocide, which involves controlling the growth of unicellular organisms such as bacteria, fungi, and algae. These industrial uses also have the potential to release these biocides directly or indirectly to the aquatic environment, which is of potential concern.

Based on the PEC and PNEC values determined above, the following Risk Quotients (RQ = PEC ÷ PNEC) have been calculated for release of IPBC into surface waters, STPs, and soil:

Compartment	PEC	PNEC	RQ
Surface water	0.38 µg/L	0.46 µg/L	0.83
STP micro-organisms	0.15 mg/L	0.6 mg/L	0.25
Soil (application of biosolids)	0.11 mg/kg	0.098 mg/kg	1.1

Calculated RQ values below 1 for surface waters and STP micro-organisms indicate that IPBC is not expected to pose a high risk to the environment based on estimated emissions, as environmental concentrations are below levels likely to cause harmful effects.

The chemical (IPBC) may be released to STPs as a result of the use as a preservative in consumer products. High STP influent values have also been measured at international STPs receiving industrial liquid waste, mainly from paint manufacturers. However, these influent concentrations have been observed below concentrations that may affect STP micro-organisms. High removal rates within STPs substantially reduce concentrations of IPBC in effluents and mitigate the risk to aquatic life.

The chemical (IPBC) is also released to the environmental waters and soil from the leachate of painted surfaces during rainfall events. Based on the measured concentrations of IPBC in stormwater and in affected soils, this release scenario is unlikely to pose an unreasonable risk to the environment. Additionally, IPBC degrades rapidly in the aquatic environment to form the less toxic chemical PBC, which will further reduce the risk of emissions to the environment.

The use of biosolids on agricultural land may pose a risk to soil organisms based on an RQ of 1.1. However, this RQ is based on an IPBC concentration in biosolids calculated using a default introduction volume of 100 tonnes per year with 100% release to sewer. This is expected to be a conservative upper estimate of the actual amount of IPBC reaching and partitioning to biosolids in STPs. Considering the short half-life of the chemical in soil and intermittent application of biosolids to soil, this pathway is unlikely to pose adverse effects to the environment despite the high toxicity of the chemical to terrestrial plants.

Uncertainty

This evaluation was conducted based on a set of information that may be incomplete or limited in scope. Some relatively common data limitations can be addressed through use of conservative assumptions (OECD 2019) or quantitative adjustments such as assessment factors (OECD 1995). Others must be addressed qualitatively, or on a case by case basis (OECD 2019).

The most consequential areas of uncertainty for this evaluation are:

- Monitoring studies that measure the concentrations of IPBC in Australian influents, effluents, and surface waters are not available. Additionally, studies investigating the leaching of IPBC from painted surfaces have been performed in Europe, with European weather and European paint formulations. Australian weather and paint formulated for use in Australia may differ to these European conditions. However, no information is available that suggests that use of international data is unsuitable for the purposes of this evaluation.

- The chemical (IPBC) has non-industrial uses as an antifungal agent for timber treatment, both in Australia and internationally. Consequently, the environmental monitoring data for IPBC may include releases from non-industrial sources. However, as the total IPBC concentrations are expected to be below levels of concern, further investigation into contributions from non-industrial uses are not currently needed.

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Regulation (EU) n°528/2012 concerning the making available on the market and use of biocidal products

Evaluation of active substances

Assessment Report

IPBC

Product-type 6
(Preservatives for products during storage)

September 2013

Denmark

IPBC (PT 6)

Assessment report

Finalised in the Standing Committee on Biocidal Products at its meeting on 27 September 2013

CONTENTS

1. STATEMENT OF SUBJECT MATTER AND PURPOSE

1.1. Principle of evaluation

This assessment report has been established as a result of the evaluation of IPBC as product-type 6 (In-can preservative), carried out in the context of the work programme for the review of existing active substances provided for in Article 16(2) of Directive 98/8/EC concerning the placing of biocidal products on the market[1], with the original view to the possible inclusion of this substance into Annex I or IA to that Directive.

The evaluation has therefore been conducted in the view to determine whether it may be expected, in light of the common principles laid down in Annex VI to Directive 98/8/EC, that there are products in product-type 6 containing IPBC that will fulfil the requirements laid down in Article 5(1) b), c) and d) of that Directive.

1.2. Purpose of the assessment

The aim of the assessment report is to support a decision on the approval of IPBC for product-type 6, and should it be approved, to facilitate the authorisation of individual biocidal products in product-type 6 that contain IPBC. In the evaluation of applications for product-authorisation, the provisions of Regulation (EU) No 528/2012 shall be applied, in particular the provisions of Chapter IV, as well as the common principles laid down in Annex VI.

The conclusions of this report were reached within the framework of the uses that were proposed and supported by the applicant (see Appendix II). Extension of the use pattern beyond those described will require an evaluation at product authorisation level in order to establish whether the proposed extensions of use will satisfy the requirements of Regulation (EU) No 528/2012.

For the implementation of the common principles of Annex VI, the content and conclusions of this assessment report shall be taken into account.

However, where conclusions of this assessment report are based on data protected under the provisions of Regulation (EU) No 528/2012, such conclusions may not be used to the benefit of another applicant, unless access to these data has been granted.

1.3. Procedure followed

This assessment report has been established as a result of the evaluation of IPBC as product-type 6 (In-can preservative), carried out in the context of the work programme for the review of existing active substances provided for in Article 16(2) of Directive 98/8/EC concerning the placing of biocidal products on the market.

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[1] Directive 98/8/EC of the European Parliament and of the Council of 16 February 1998 concerning the placing biocidal products on the market. OJ L 123, 24.4.98, p.1

IPBC (CAS no. 55406-53-6) was notified as an existing active substance, by members of the European Union IPBC Task Force (Arch Chemicals, Dow Benelux B.V., ISP Switzerland GmbH, Lanxess Deutschland GmbH, Troy Corp), hereafter referred to as the applicant, in product-type PT6. ISP Switzerland AG changed the legal entity to ISP Switzerland GmbH.

Commission Regulation (EC) No 1451/2007 of 4 December 2007[2] lays down the detailed rules for the evaluation of dossiers and for the decision-making process in order to include or not an existing active substance into Annex I or IA to the Directive.

In accordance with the provisions of Article 7(1) of that Regulation, Denmark was designated as Rapporteur Member State to carry out the assessment on the basis of the dossier submitted by the applicant. The deadline for submission of a complete dossier for IPBC as an active substance in Product Type 6 was 31 July 2007, in accordance with Article 9 (c) of Regulation (EC) No 1451/2007.

On 31 July 2007, DK competent authorities received a dossier from the applicant. The Rapporteur Member State accepted the dossier as complete for the purpose of the evaluation on 29 January 2008.

On 27 June 2011, the Rapporteur Member State submitted, in accordance with the provisions of Article 14(4) and (6) of Regulation (EC) No 1451/2007, to the Commission and the applicant a copy of the evaluation report, hereafter referred to as the competent authority report. The Commission made the report available to all Member States by electronic means on 29 June 2011. The competent authority report included a recommendation for the inclusion of IPBC in Annex I to the Directive for product-type PT6.

In accordance with Article 16 of Regulation (EC) No 1451/2007, the Commission made the competent authority report publicly available by electronic means on 1 July 2011. This report did not include such information that was to be treated as confidential in accordance with Article 19 of Directive 98/8/EC.

In order to review the competent authority report and the comments received on it, consultations of technical experts from all Member States (peer review) were organised by the Commission. Revisions agreed upon were presented at technical and competent authority meetings and the competent authority report was amended accordingly.

In accordance with Article 15(4) of Regulation (EC) No 1451/2007, the present assessment report contains the conclusions of the Standing Committee on Biocidal Products, as finalised during its meeting held on 27 September 2013.

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[2] Commission Regulation (EC) No 1451/2007 of 4 December 2007 on the second phase of the 10-year work programme referred to in Article 16(2) of Directive 98/8/EC of the European Parliament and of the Council concerning the placing of biocidal products on the market. OJ L 325, 11.12.2007, p. 3

2. OVERALL SUMMARY AND CONCLUSIONS

2.1. Presentation of the Active Substance

2.1.1. Identity, Physical-Chemical Properties & Methods of Analysis

Identity

IPBC, CAS No. 55406-53-6, is a fungicide produced and/or supplied by Arch Chemicals, Dow Benelux B.V., ISP Switzerland GmbH, Lanxess Deutschland GmbH, Troy Corp. at/from sites in and out of Europe. Analysis of five technical grade batches which are representative of the current manufacturing process demonstrated a mean purity of ≥ 98% w/w in compliance with European Union IPBC Task Force (Arch Chemicals, Dow Benelux B.V., ISP Switzerland GmbH, Lanxess Deutschland GmbH, Troy Corp.) specifications. All impurities above the level of 1 g/kg have been fully identified and the corresponding methods of analysis have been developed. The main identification characteristics were given in a confidential document. The active substance must be technically equivalent to the specifications given. None of the manufacturing impurities are considered to be of potential concern.

Physical and chemical properties

IPBC technical is a yellowish crystalline powder with a faint odour of iodine and a melting point of 65.8 – 66.5°C. Its relative density is 1.71 at 20°C.

The vapour pressure is found to be 2.36-4.5 x 10^{-3} Pa at 25°C. The water solubility of IPBC technical is 0.168 g/L (pH 7) at 20°C.

IPBC is very soluble in methanol (>1000 g/L) and other organic solvents. Its octanol/water partition coefficient (log P_{OW}) is 2.81 at 25°C.

The substance is stable at room temperature and is stable at 54°C for 14 days. IPBC is not highly flammable. It has no pyrophoric property and it does not undergo spontaneous combustion. IPBC is not explosive.

The recommended container material for IPBC is protected steel drums.

Methods of analysis

The identification and quantification of IPBC as manufactured is performed using HPLC-UV and GC-FID. Methods of analysis for residues are HPLC-MS/MS.

The methods developed to analyse residues in soil, water, body fluids and tissues with the respective limits of quantification of 10 µg/kg of soil, 0.1 µg/L of water, 0.05 mg/L of body fluids and 0.1 mg/L of tissues.

Residues in air were not necessary because IPBC is not volatile and spray applications only involve non-respirable particles.

An analytical method for the determination of residues of IPBC in/on food or feedstuffs is not required because the active substance is not used in a manner that may cause direct contact with food or feedstuffs. The use of IPBC as an in-can preservative result to a low concentration of IPBC in the end-product (0.01-1%). Therefore, the amount of IPBC transferred to food or feeding stuff from material treated with the end-product is considered to be negligible.

2.1.2. Intended Uses and Efficacy

The assessment of the biocidal activity of the active substance IPBC demonstrates that it has a sufficient level of efficacy against fungi and yeast and the evaluation of the summary data provided in support of the efficacy of the accompanying product, establishes that IPBC-based in-can preservative products may be expected to be efficacious. The biocidal products produced by the TF members for in-can preservation have typical concentrations in the range of 10 to 30% IPBC. In end-use products, IPBC is contained at concentrations ranging from 0.01 to 1%.

The intended uses of the substance, as identified during the evaluation process, are listed in Appendix II.

The risk of resistance formation against carbamate fungicides is regarded to be low to medium by FRAC (Fungicide Resistance Action Committee). This applies to the use of carbamate fungicides in agriculture, where yearly applications to the same fields are possible. Based on the unspecific mode of action of IPBC, the risk of resistance formation during in-can preservation is regarded to be low.

2.1.3. Classification and Labelling -proposed classification and labelling for the active substance IPBC

Proposed classification/labelling according to Directive 67/548/EEC for the active substance, IPBC, following evaluation

Classification		
Class of danger	T N	Toxic Dangerous for the environment
R phrases	R22	Harmful if swallowed.
	R23	Toxic by inhalation
	R37	Irritating to respiratory system
	R41	Risk of serious damage to the eye
	R43	May cause sensitization by skin contact
	R50	Very toxic to aquatic organisms.
S phrases	S1	Keep locked up.
	S23	Do not breathe vapour/spray
	S24/25	Avoid contact with skin and eyes

	S26	In case of contact with eyes, rinse immediately with plenty of water and seek medical advice.
	S36/37/39	Wear suitable protective clothing, gloves and eye/face protection.
	S45	In case of accident or if you feel unwell, seek medical advice immediately (show the label where possible).
	S61	Avoid release to the environment. Refer to special instructions/Safety data sheets.

Proposed classification based on Regulation EC 1272/2008:

Signal Word	Danger
Pictograms	GHS05, GHS06, GHS09
Hazard class and category code(s)	Acute Tox 3 Eye Dam. 1 Acute Tox 4 Skin Sens. 1 STOT SE3 Aquatic Acute 1 Aquatic Chronic 1*
H-Statements	H331: Toxic if inhaled H318: Causes serious eye damage H302: Harmful if swallowed H317: May cause an allergic skin reaction H335: May cause respiratory irritation H400: Very toxic to aquatic life H410: Very toxic to aquatic life with long-lasting effects*
Environmental M-factor	10 (acute), 1* (chronic)

Precautionary statements according to the latest classification and labelling guidance No. 1272/2008 have not been assigned.

* According to Commission Regulation (EU) No 286/2011 (2nd ATP)

The Committee for Risk Assessment (RAC) has in addition recently proposed to classify IPBC with STOT RE 1 based on effects seen on larynx after prolonged exposure by inhalation. This proposal has not yet been adopted by the REACH Committee.

The final classification must when adopted by the REACH Committee be considered during the authorization of biocidal products.

2.1.4. Classification and Labelling -proposed for the representative biocidal products

Classification on the basis of toxicological and environmental effects of the biocidal products shall, in the absence of experimental data, be deduced from the respective properties of the active substance(s) and the inactive ingredients on the basis of the conventional (calculation) method referred to in Article 6 and Annex II (tox.) and Article 7 and Annex III, Parts A and B (environment) of Directive 1999/45/EC. Classification on the basis of toxicological properties from experimental data is only allowed if test results on animals already exist or it can be scientifically demonstrated that the toxicological properties of the preparation cannot correctly be determined by the conventional method.

The formulation consists of two ingredients, namely 30% of the active substance IBPC (3-Iodo-2-propynyl butylcarbamate) and the solvent Dipropylene glycol. Only IPBC has properties, which lead to a classification according to the relevant guidelines.

The classification of the formulation as “harmful if swallowed”, “toxic by inhalation”, “irritating to respiratory system, “risk of serious damage to eyes” as well as “may cause sensitization by skin contact” was made on the basis of the conventional method of Directive 1999/45/EC and is based on the toxicological properties of IPBC.

The classification as “dangerous for the environment” according to Directive 2001/59/EC is based on the ecotoxicological properties of IPBC.

Classification		
Class of danger	T N	Toxic Dangerous for the environment
R phrases	R22	Harmful if swallowed.
	R23	Toxic by inhalation
	R37	Irritating to respiratory system
	R41	Risk of serious damage to the eye
	R43	May cause sensitization by skin contact
	R50	Very toxic to aquatic organisms.
S phrases	S1	Keep locked up.
	S23	Do not breathe vapour/spray
	S24/25	Avoid contact with skin and eyes
	S26	In case of contact with eyes, rinse immediately with plenty of water and seek medical advice.

	S36/37/39	Wear suitable protective clothing, gloves and eye/face protection.
	S45	In case of accident or if you feel unwell, seek medical advice immediately (show the label where possible).
	S61	Avoid release to the environment. Refer to special instructions/Safety data sheets.

The following classification is proposed in accordance with the latest classification and labelling guidance (Regulation (EC) No. 1272/2008):

Labelling elements based on the classification	
GHS pictogram	GHS05, GHS06, GHS09
Signal word	Danger
Hazard class and category code(s)	Acute Tox 3 Eye Dam. 1 Acute Tox 4 Skin Sens. 1 STOT SE3 Aquatic Acute 1 Aquatic Chronic 1*
Hazard statements	H331: Toxic if inhaled H302: Harmful if swallowed H317: May cause an allergic skin reaction H318: Causes serious eye damage H335: May cause respiratory irritation H400: Very toxic to aquatic life H410: Very toxic to aquatic life with long-lasting effects*
Environmental M-factor	10, 1*

* According to Commission Regulation (EU) No 286/2011 (2nd ATP)

Precautionary statements according to the latest classification and labelling guidance No. 1272/2008 have not been assigned.

The final classification must when adopted by the REACH Committee be considered during the authorization of biocidal products.

2.2. Summary of the Risk Assessment

2.2.1. Human Health Risk Assessment

2.2.1.1. Hazard identification

IPBC is of moderate acute toxicity by the oral route and of low toxicity by the dermal route. IPBC is classified toxic by inhalation. The substance is not irritating to skin but is a severe eye irritant and a skin sensitizer.

In the short term studies the liver and kidney were the main target organs. IPBC is neither carcinogenic, neurotoxic or genotoxic. IPBC is not toxic to reproduction or a developmental toxicant.

2.2.1.2. Effects assessment

IPBC was completely and readily absorbed via the oral route (<90%). Following absorption, the substance was widely distributed with no trend for bioaccumulation observed. IPBC was extensively metabolised with the major metabolites being the two distereomeric conformers of propargyl-N-acetic acid carbamate. Glucuronidation appeared to be the main secondary metabolism pathway. The majority of the administered radioactivity was excreted via urine (57.3% to 70.7%) with faeces being a minor route (4.4% to 7.4%); radiolabelled carbon dioxide constituted between 18.4 to 24.2% of the administered dose. There were no differences between sexes or applied doses detectable.

For IPBC, an *in vitro* study with human skin gave dermal absorption values (including skin residues) of 30, 10, and 1.6% for formulations containing 0.6, 2.3, and 17.1% IPBC, respectively. Solvents were xxxxxxx respectively.

In **acute toxicity studies**, IPBC was found to be of moderate acute toxicity by the oral route and of low acute toxicity by the dermal routes of exposure but has high acute toxicity by the inhalation route. The data support classification of IPBC for acute toxicity by the inhalation route.

IPBC is not a skin irritant, but does exhibit the potential to produce severe eye irritation. In animal studies, IPBC met the criteria for classification as a severe eye irritant.

Positive findings from guinea pig sensitisation studies (GPMTs) indicate that IPBC has skin sensitisation potential.

Following repeated oral administration of IPBC post-dose salivation was observed immediately after dosing by gavage from 30 mg/kg bw/day, but not when IPBC was administered via the diet. Food consumption was reduced from 80 mg/kg bw/day (dietary, gavage) and body weights and/or body weight gains from 40 mg/kg bw/day (dietary) or 80 mg/kg bw/day (gavage). Brain and RBC cholinesterase activities were not reduced up to and including the highest dose levels administered. Local erosions, ulceration, and/or inflammation of the stomach (forestomach and/or glandular stomach) were observed from about 20 to 30 mg/kg bw/day (dietary, gavage). Increased liver weights, sometimes accompanied by hepatocellular changes, and increased kidney weight (females only) were observed from 30 to 40 mg/kg bw/day. Increased incidence in foamy macrophage aggregates was noted in the lungs of male rats from 40 mg/kg bw/day in the 2-year study. In the 78-week mice study, an increased incidence in enlarged thyroids accompanied by foci of small vacuolated cells most likely of follicular origin and general follicular enlargement was noted at 150 mg/kg bw/day; the toxicological significance of these findings in thyroids remains unclear.

Following repeated dermal administration to rats dermal irritation persisting throughout the treatment period, and hyperkeratosis and ulceration was observed at 500 mg/kg bw/day; at 200 mg/kg bw/day, mild hyperkeratosis. No adverse systemic effects were observed.

Following repeated inhalation to rats decreased RBC cholinesterase activity was observed in females at 6.7 mg/m^3 *(after 2 weeks but not at study termination)* and decreased brain cholinesterase activities in females and in males at 6.7 mg/m^3. The finding is of unclear relevance since no clear dose-relationship was observed (small decrease for a large change in dose) and the normal variation seems to be wide. Results indicated that IPBC was not neurotoxic. This was also supported by the acute and 90-day neurotoxicity and 104 weeks studies in rats and 78 weeks mice study (all investigating RBC and brain cholinesterase inhibition). Histopathological findings were epithelial hyperplasia in the central region of the larynx, hyperplasia or squamous metaplasia in the ventrolateral region of the larynx, and necrosis of the underlying cartilage of the larynx at 6.7 mg/m^3 (NOAEC 1 mg/m^3). The effects on larynx are considered as a local and not a systemic effect.

IPBC was not neurotoxic when administered via the oral route.

The weight of evidence from the available well-conducted *in vitro* and *in vivo* genotoxicity studies indicates that IPBC is not a genotoxic substance.

IPBC was not carcinogenic in rats and mice up to and including the highest dose levels tested (80 and 150 mg/kg bw/day for rats and mice, respectively).

In experimental animal studies IPBC did not affect fertility and did not cause developmental toxicity. The evidence suggests that this substance does not possess significant potential with respect to toxicity to reproduction.

Reference values for the risk assessment (AEL)

Two different values will be used as basis for the risk characterisation of systemic effects:

The $\text{AEL}_{\text{long-term}}$ was derived from the 104 weeks chronic toxicity/carcinogenicity study with rats with NOAEL 20 mg/kg bw/day based on reduced mean body weight and body weight gain in both sexes and increased incidence of histopathological changes in stomach, forestomach and salivary glands.

An uncertainty factor of 100 will be applied to the NOAEL for a 10-fold factor for interspecies variability and a 10-fold factor for intraspecies variability. As absorption by the oral route was found to be close to 100%, no correction for absorption from the gastrointestinal tract has been made in the AEL setting.

$AEL_{long\text{-}term}$= 20 mg/kg bw/day / 100 = 0.2 mg/kg bw/day

As IPBC is not toxic to reproduction or a developmental toxicant the most relevant study to be chosen as a basis for setting the AEL_{acute} and $AEL_{medium\text{-}term}$ seems to be the 3 months gavage study in rats which has the lowest relevant NOAEL (~35 mg/kg bw/day) based on reduced body weight and body weight gain, increased absolute and relative kidney and liver weight and increased iron concentration.

AEL_{acute} and $AEL_{medium\text{-}term}$ = 35 mg/kg bw/day / 100 = 0.35 mg/kg bw/day

2.2.1.3. Risk characterisation for local effects

It was decided at TMIV2011 that a quantitative RA should only be carried out if suitable data is available for the product (e.g product were a dilution of the active substance). In the situation with IPBC the model product, contains formulants and thus is not just a dilution of the active substance. In line with the decision taken at TMIV2011, a qualitative local risk assessment describing the effects and including the C&L of the products has been performed.

Due to the low vapour pressure of IPBC of 0.00234 Pa (2.34 mPa) and according to the "HEEG opinion on Assessment of Inhalation Exposure of Volatilised Biocide Active Substance" IPBC is not volatilized. Exposure from inhalation is therefore considered negligible. The concern regarding local effects arises from the effects seen on larynx in rats after repeated exposure which leads to a classification of IPBC with R37: **Irritating to respiratory system.**

Only products used for in-can preservation by professionals in automated processes will be classified with R37 (10-40% a.i) and thus PPE can be prescribed. For all other uses both for professionals and amateurs the application/handling of the end products, which contains 0.01-1% IPBC, will not lead to classification of the end products with R37.

In conclusion, a local exposure assessment is not required even if a worst case approach based on the R37/STOT SE 3 classification is taken. The concentration of IPBC in all liquid formulations applied in PT6 is below 20% which is the threshold for a classification of a mixture as a respiratory irritant according to the provisions of the DPD and the CLP. Consequently, as no respiratory irritation is anticipated during application of the liquid diluted products, an exposure and risk assessment for local effects via the inhalation route is not required. It has to be emphasized that normally risk assessment for local effects are NOT performed on the active substance classifications but on the product and its resulting classification.

The representative product containing IPBC is also classified as a severe eye irritant and skin sensitiser. However, at the end use concentration of the product the risk with regard to these hazardous properties is considered to be acceptable.

Despite the moderate sensitizing property of IPBC, based on the risk assessment no specific concern is identified due to use of treated articles intentionally incorporating IPBC. Therefore, there is no need to set restrictions or specific conditions for use of IPBC in similar type of applications as evaluated in this CAR.

2.2.1.4. Systemic exposure assessment and risk characterisation

Industrial application

Exposure during industrial in-can preservation of washing, cleaning fluids, detergents, paints and coatings, fluids used in paper, textile and leather production, glues and adhesives is displayed in Table 1-1 in doc IIC. Assuming concentrations in the range of 10 to 40 %, exposure during in-can preservation is between 0.0010 to 0.0024 mg/kg bw/day when wearing PPE. These exposure figures do not exceed the long term AEL of 0.2 mg/kg bw/day (equivalent to 0.51 – 1.2% of the $AEL_{long-term}$).. The MOE is above 8357. Thus, the risk for the industrial workers during in-can preservation is considered to be acceptable.

The industrial application of in-can preserved paints and coatings results in an exposure of 0.000053 to 0.00053 mg/kg bw/day when spraying at concentrations of 0.01 and 0.1 %, respectively and wearing PPE (see Table 1-1 in doc IIC). These exposures are below 1 % of the AEL for long-term exposures of 0.2 mg/kg bw/day. The MOE is above 10000. Thus, the risk for the industrial workers during application of in-can preserved paints and coatings is considered to be acceptable.

The use of in-can preserved fluids for paper, textile and leather production (IPBC concentration: 0.1%) and glues and adhesives (IPBC concentration: 1%) result in exposures of 0.037 and 0.025 mg/kg bw/day (concentration 0.1-1 %) when wearing PPE. This equals to 18.7 and 12.54 % of $AEL_{long-term}$ of 0.2 mg /kg bw/day. The MOE is above 500. Thus, the risk for the industrial workers during application of in-can preserved fluids for paper, textile and leather production and glues and adhesives is demonstrated to be acceptable.

Professional application

When IPBC preserved washing and cleaning fluids or detergents are applied by professionals (including handling of the wet treated laundry for instance) the exposure is 0.000296 mg/kg bw/day (equal to 0.15 % of $AEL_{long-term}$ of 0.2 mg/kg bw/day) when wearing PPE. When applying the in-can preserved paints, coatings, glues and adhesives by manual spraying, brushing, or wallpapering, the exposures are in the range of 0.00089 to 0.03 mg/kg bw/day when wearing PPE. These exposures are well below the $AEL_{long-term}$ of 0.2 mg/kg bw/day (0.45 to 15. % of $AEL_{long-term}$). The MOE is above 700 in all scenarios.

Consequently, the risk for professionals during the intended biocidal uses of IPBC in PT6 has been demonstrated to be acceptable.

Amateur application

The results of the amateur/consumer exposure estimations for IPBC when using in-can preserved cleaning and washing fluids, detergents (for hand dish washing), paints and coatings (for manual spraying and brushing) and glues and adhesives (for wallpapering) are shown in Table 1-3 in doc IIC. The estimated exposures are in the range of 0.002 to 0.187 mg/kg bw/day when no PPE is worn. In all scenarios concerning amateur/consumer exposure, the AEL for longand short/medium- term exposure is not exceeded (1% of long-term AEL and max 53.43% of the short-term AEL, respectively). The MOE is above ~ 190 in all non-professionals use scenarios. These applications are therefore considered to be of acceptable risk.

Risk Characterisation for Secondary exposure

A secondary exposure and risk assessment has been performed for the following scenarios:

- Consumption of food after contact to cleaned or painted surface
- Exposure *via* residues from cleaned dishes
- Contact of child and infant to cleaned or painted surface or glues
- Contact of child and infant to residual IPBC in textiles after laundry
- Inhalation of volatilized residues indoors

For all secondary exposure assessments, worst case assumptions were made, i.e. assuming the maximum concentrations.

The oral exposure *via* residues from food in contact with cleaned or painted surfaces or from cleaned dishes was estimated to be in the range of $9.9*10^{-6}$ mg/kg bw/d $8*10^{-4}$ mg/kg bw/day for adults, children and infants, respectively, which is equivalent to 0.005% to 0.4% of the $AEL_{long-term}$.

The combined oral and dermal exposure of children and infants during playing on treated or painted surface or when touching glues containing IPBC has been estimated to amount to $3.98*10^{-3}$ and 0.0139 mg/kg bw/ day, respectively. Which is equivalent to 1.99% and 6.95% of the $AEL_{long-term}$ of 0.2 mg/kg bw/day.

The dermal exposure of children and infants via residues of IBPC in textiles after laundry results in an exposure of $1.3*10^{-6}$ to $1.86*10^{-6}$ mg/kg bw/ day, respectively which is equivalent to less than 0.1% of the $AEL_{long-term}$.

The inhalation exposure of adults, children and infants to residues volatilizing from residues was estimated to be in the range of $1.09*10^{-3}$ to $1.57*10^{-3}$ mg/kg bw/day, equivalent to < 1% of the $AEL_{long-term}$.

The risk during secondary exposure to IPBC is considered to be acceptable because the $AEL_{long-term}$ is not exceeded. The MOE is calculated to be above 1400 for all secondary exposure scenarios.

Combined exposure

Adults are the only subpopulation who may reasonably experience both primary and secondary exposure to IPBC. Since the secondary exposure adds only negligible doses to the primary exposure, no additional concern arises from the combination of all pathways. Even in the worst case scenario of a professional who is involved in industrial in-can preservation or industrial use of in-can preserved end-products, manual spraying and brushing in his or her leisure time, the MOE of the primary exposure scenario remains virtually unchanged and sufficiently high.

Overall conclusion

The use of IPBC in industrial in-can preservation and the further use of the in-can preserved end-products can be considered safe for industrial, professional and non-professional users. Furthermore, the secondary exposure is negligible or very low and does not pose a unacceptable risk for human health. Thus the overall outcome of the systemic risk assessment for humans, that has covered normal/representative use of the biocidal product together with a realistic worst case scenario as well as the material treated with it, is that proper use, i.e. use in compliance with the conditions on the label, of IPBC and in-can preserved end-products therewith is considered safe for all subpopulations.

2.2.2. Environmental Risk Assessment

2.2.2.1. Fate and distribution in the environment

IPBC is stable to hydrolysis. Direct photodegradation of IPBC in water is low and the substance may be considered photolysis stable in water.

Air will not be an environmental compartment of concern for IPBC because of the low vapour pressure of this compound. It should also be noted that the calculated DT50 of IPBC in air is only about 15 hours and IPBC is therefore not considered persistent in air.

IPBC is not readily biodegradable but is primary biodegradable according to Zahn-Wellens test. The biodegradation half-life in surface water is estimated to about 3.1 hour at 12°C. IPBC is metabolised rapidly in soil in laboratory experiments, the half-life is estimated to be 4.7 hour at 12°C. In degradation of IPBC, the primary degradate was propargyl butyl carbamate (PBC).

PBC was found in hydrolysis, aerobic soil, and anaerobic aquatic metabolism studies. In hydrolysis, PBC was the only degradation product identified.

In soil, PBC was degraded to CO_2, bound soil residues and an unidentified metabolite. In anaerobic aquatic environments (sediment/water), PBC was degraded to 2-propenyl butyl carbamate (2-PBC) and 2 unidentified degradates (less than 10%), CO_2 and possibly CH_4. The metabolite 2-PBC is only formed at a percentage > 10% in the water phase under anaerobic conditions. QSAR estimation indicates a toxicity of this metabolite is comparable to that found for IPBC. Therefore in this case it is not considered necessary to ask for experimental ecotoxicological data for this metabolite as IPBC is not likely to undergo anaerobic degradation in any of the environmental compartments that IPBC will reach when used in PT6; however, anaerobic degradation can occur in the sewage sludge if the sludge is used in biogas production, this is not included in this evaluation.

An evaluation of the degradation products iodide and iodate released from IPBC are included in the exposure and risk assessment, however for the fate and distribution in the environment and for the effect assessment of iodide and iodate data from the Swedish first Draft CAR of iodine are used.

IPBC has a medium to high mobility potential.

The bioaccumulation potential is not significant based on a log P_{ow} value of 2.8.

2.2.2.2. Effect assessment

The toxicity to aquatic organisms is documented by acute and long-term studies. Long-term NOEC values are available for all three trophic levels in the aquatic compartment: The lowest NOEC from the algae study of 0.0046 mg/L was taken as the basis for the PNEC derivation in water.

The PNEC for the sediment is calculated using the equilibrium method. However, the risk to the sediment is the same as that described for surface water. Therefore the risk of the sediment will not be considered further.

The toxicity to terrestrial organisms is documented by acute studies. Tests are available for test on earthworm, terrestrial micro-organisms and terrestrial plants. The plant test with an EC_{50} of 4.92 mg/kg was taken as the basis for the terrestrial PNEC.

The following PNEC values are used in the risk assessment for IPBC:

$PNEC_{water}$ = 0.0046 mg/L / 10 = 0.0005 mg/L

$PNEC_{stp}$ = 44.00 mg/L / 100 = 0.44 mg/L

$PNEC_{soil}$ = 4.92 mg/kg dry soil/1000 = 0.005 mg/kg dry soil

PBC was identified as a relevant metabolite of IPBC in water, sediment and soil, because it was found in degradation studies at above the limit value of 10%. Due to a relative short half-life of PBC (T½ of 31.2; 31.4 and 9.5 days at 12°C in water, sediment and soil, respectively) PBC can be regarded as a transient metabolite. In addition, the ecotoxicity of PBC is a factor of 300 – 1000 lower for fish, invertebrates and algae compared to IPBC.

The following PNEC values are used in the risk assessment for PBC:

$PNEC_{water}$ = 41.3 mg/L / 1000 = 0.0413 mg/L

$PNEC_{soil}$ = 0.149 mg/kg wet soil = 0.169 mg/kg dry soil (calculated from $PNEC_{water}$)

For the $PNEC_{STP}$ the one for IPBC is used as a worst case.

For iodine, iodide and iodate PNEC values from the first Draft CAR of iodine from SE is used.

A metabolite 2-PBC is formed at a percentage > 10% in the water phase; however only under anaerobic conditions. QSAR estimation indicates a toxicity of this metabolite is comparable to that found for IPBC. Therefore in this case it is not considered necessary to ask for experimental ecotoxicological data for this metabolite.

2.2.2.3. PBT and ED assessment

Assessment of PBT criteria

A PBT assessment is carried out for IPBC and PBC according to the REACH guidance on PBT assessment.

Persistence criteria (P)

IPBC is not readily biodegradable but is primary biodegradable according to Zahn-Wellens test. In an aerobic soil degradation study, IPBC is rapidly degraded with a DT_{50} of 2.1 hour at 22° C (DT_{50} of 4.7 hours at 12 ° C). In a water sediment study a DT_{50} of 1.4 hour at 22° C (DT_{50} of 3.1 hours at 12 ° C) was found for the water phase and a DT_{50} of 2.2 hour at 22° C (DT_{50} of 4.9 hours at 12 ° C) was found for the sediment phase. As these half-lives are below the trigger values, the P criterion for IPBC is not fulfilled.

The degradation DT_{50} of PBC in freshwater was found to be 14.2 days at 22 ºC (DT_{50} of 31.2 days at 12 ° C), in sediment the DT50 value is 14.3 days at 22 ºC (DT_{50} of 31.4 days at 12 ° C) while the degradation half-live in soil of PBC is 4.3 days at 22 ºC (DT_{50} of 9.5 days at 12 ° C). As these values are below the trigger values, the P criterion for PBC is not fulfilled.

Bioaccumulation criteria (B)

The bioaccumulation potentials are not significant based on a log P_{ow} value of 2.8 for IPBC and 1.64 for PBC which will result in bioconcentration factors (BCF) below 2000. Therefore, the B criterion is not fulfilled for either IPBC or PBC.

Toxicity criteria (T)

For IPBC the NOEC value for algae, the most sensitive aquatic species, is 0.0046 mg/l. Therefore, the T criterion is fulfilled as a chronic NOEC below 0.01 mg/L is found for IPBC. Mammalian toxicity data do not give rise to T criteria for IPBC.

For PBC mammalian toxicity data do not give rise to T criteria. For PBC no data on chronic effects are available. Therefore short-term toxicity data are compared to the trigger of 0.1

mg/L. For PBC all the short-term toxicity data are above the trigger value and a log Kow below 4.5 results in no further assessment necessary for the toxicity criteria.

Thus IPBC and PBC do not fulfil the PBT or vPvB criteria.

Assessment of Endocrine Disruption (ED)

IPBC nor PBC are not included in the EU list of potential endocrine disruptors (COM DG ENV, 2000).

2.2.2.4. Exposure assessment

For the exposure assessment, two approaches have been used, i.e. a consumption and a tonnage based approach. For the consumption based approach penetration factors of 1 and 0.5 have been applied to uses where there is a direct disperse emission to the STP. Using a penetration factor of 0.5 is a refinement that is performed as several substances are used for in-can preservation.

For the consumption based approach, emissions are calculated based on several different emission scenario documents (ESDs) as recommended in the ESD for PT6. The different uses of IPBC result in emissions directly into soil, the facility drain or the air. For the consumption based approach emissions from the formulation of the end-products are considered negligible as the process of in-can preservation is highly automated. The exposure assessment therefore covers the application phase of the end-products treated with preservative and their service life, covering all the applications listed:

PT6.1	**In-can preservation of washing and cleaning fluids and detergents**
PT6.2	**In-can preservation of paints and coatings**
PT6.3.2	**In-can preservation of fluids used in textile production**
PT6.6	**Glues and adhesives**

For the tonnage based approach exposure calculations cover both the formulation and the use of the end-products. For the formulation phase local emissions were calculated using the default values provided in A&B tables of the TGD Part II (2003) for the respective industries. For the use phase it is assumed that 100% of the used IPBC is emitted to the facility drain, except for paints where default values from the ESD for PT7 have been used as refinement (0% release for professional use and 3% release for non-professional use). For the industrial use under PT6.2 it is assumed that this part is covered by the professional use. For glues and adhesives which are used indoors, it is assumed that it result in emissions of IPBC to air but not to the facility drain. For the tonnage based approach only the emissions to the STP are considered as the other emissions are covered by the consumption based approach as these emissions are higher than those for the tonnage based approach.

As IPBC quickly degrades to PBC, iodide and iodate within the environmental compartments, PEC calculations of IPBC, PBC, iodide and iodate have been performed for the environmental compartments: STP, surface water, sediment, air, soil and groundwater were relevant. It is chosen to base the risk characterisation for the STP on the concentrations in the effluent (as suggested in the TGD); IPBC degrades totally within 4 hours in a STP and IPBC will therefore not be present in the effluent. For the evaluation of IPBC in PT8, inlet concentrations of IPBC were used to evaluate the risk for the STP. For PT8 PEC/PNEC (STP) ratios were low and it made no difference if the risk characterisation was based on IPBC or PBC; whereas for IPBC in PT6, PEC/PNEC (STP) ratios are closer to 1 so it makes a difference which ones are evaluated. It is therefore chosen to use the approach with STP effluent concentrations according to the TGD for this evaluation.

In the evaluation of iodine released from IPBC, it is chosen to consider 100% formation of both iodide and iodate. This proposed assessment is however worst case as it is expected that much less than 100% of the different iodine species will be present. However, for calculation of soil concentrations it is assumed that the total iodine concentration in soil is transformed into 14% iodide and 100% iodate. For the exposure assessment of iodide and iodate, calculations for the direct emissions to STP are based on the tonnage based approach. The reason for this is that the final risk assessment of IPBC/PBC is based on the tonnage approach and acceptable risk is found; furthermore the evaluation of iodine/iodate was added late in the process (in the final draft CAR) and for saving the amount of work only the last tier was chosen. However, the consumption based approach is applied for direct emissions to soil as this is not covered in the tonnage based approach.

2.2.2.5. Risk characterisation

IPBC released directly to the facility drain

Several of the usages of IPBC as in-can preservative result in direct emissions to a STP, these emissions can end up in the same STP and in the same environmental compartments following the STP. A cumulative risk assessment is therefore performed for these emissions based on a tiered approach. For Tier 1 all emissions to STP are evaluated together. However, it is not likely that all the industries are in the same catchment area to a STP; this approach is therefore only used as a worst case. For Tier 2 the total domestic emission is considered together while emissions from industries are evaluated separately.

Considering PBC (consumption and tonnage based approach)

In the risk characterisation it is found that the consumption based approach results in PEC/PNEC ratios much higher than those for the tonnage based approach, both when considering market penetration factors of 1 and 0.5.

For the consumption based approach the requirements for acceptable risk according to the TGD on Risk Assessment are met for all the single uses of IPBC. However, when considering the cumulative risk (Tier 1: all domestic and industrial emissions together) risk is found for the sewage treatment plant and the surface water for marked penetration factors of both 1 and 0.5. When considering the total domestic load (Tier 2) for the consumption based approach no unacceptable risk is identified for any environmental compartments for marked penetration factors of both 1 and 0.5. Also when the total domestic load is combined with any single industrial emission, no unacceptable risk is found for a marked penetration factor of 0.5. For a marked penetration factor of 1 unacceptable risk is found for the STP and the surface water when the total domestic load is combined with industrial emissions from either washing and cleaning fluids or textile production.

For the tonnage based approach for applications considered in this dossier the requirements for acceptable risk according to the TGD on Risk Assessment in the STP, the surface water compartment as well as the soil compartment are met. Also when considering the cumulative emission from all the formulations and usages no unacceptable risk is identified.

The risk to the sediment is the same as that described for surface water. Therefore the risk of the sediment will not be considered further.

Considering iodide and iodate (only tonnage based approach)

In the risk characterisation performed for the degradation products iodide and iodate unacceptable risk is identified for iodide in surface water and soil. For soil, predicted concentrations are well within the background level which is found acceptable. For surface water the predicted concentrations of both iodide and iodate are within the background level but close to the highest level. Under oxic conditions iodine is mainly present as iodate which would mean risk ratios below 1. Moreover, sorption conditions would be quite different in oxic waters. As stated in the first Draft CAR for iodine a higher sorption constant is found for oxic waters which result in lower concentrations in surface water and higher concentrations in sediment and suspended matter. Further it should be mentioned, that the calculations are considering the cumulative emission from all the formulations and usages.

For sediment and groundwater predicted concentrations of iodide and iodate are well within the background levels, which is found acceptable.

IPBC released directly to soil

IPBC is released directly to soil when used in paints applied to houses in the countryside. IPBC, PBC, iodide and iodate are evaluated as IPBC quickly degrades in soil. For the release during treatment acceptable risk according to the TGD is identified shortly after application for IPBC and for PBC considering a distance and depth of 10 cm from the treated surface. Initial risk is therefore accepted as the risk is reduced by a PEC/PNEC value far below 1 on time1 for IPBC and PBC, respectively. For release during service life the risk is unacceptable for IPBC and PBC within the initial assessment period, while the risk is acceptable within the longer assessment period when the 10 cm distance and depth is considered. When considering a 50 cm distance and depth, the PEC/PNEC values are below 1, showing an acceptable risk for both the initial and longer assessment period.

The PEC/PNEC ratios for iodide are slightly above 1 both for the application phase and the service life when considering a distance and depth of 50 cm and below 1 for the distant soil for the spraying treatment. For iodate the PEC/PNEC ratios are above 1 when considering a distance and depth of the soil of 10 cm; however, the PEC values of iodide and iodate are all well within the background level (0.5-20 mg/kg dwt). The risk is therefore found acceptable both for iodate as well as for iodide.

In line with the conclusion from the first Draft CAR of iodine from SE stating that: considering the high background iodine concentrations in the environmental compartments concerned and that iodine is an essential element to both animals and plants in rather high concentrations (higher than what corresponds to a trace element), the actual risks arising from the use should be considered to be acceptable as predicted concentrations are below the background level.

Atmosphere and groundwater

Direct exposure to air from the described use of IPBC in PT6 is considered to be low. In addition, the vapour pressure of IPBC is low and the calculated half life in air is short (15 h). PBC might reach the air compartment due to releases from the STP. The highest annual average PEC in air for PBC was calculated to be 1.21×10^{-6} mg/m^3. Consequently, air is not an environmental compartment of concern for IPBC or PBC. Exposure to air for iodide and iodate is considered to be low, as the compounds are assumed not to be volatile.

By using the FOCUS models PELMO and PEARL it could be shown that IPBC and PBC do not leach to groundwater from the soil surface, thus posing no risk to the groundwater compartment. By using the FOCUS model PEARL it could be shown that the estimated concentrations of Iodide and Iodate are within the background level of 1-70µg/L in eight of the nine considered scenarios. In one scenario the estimated concentration is only slightly above the background level of 70 µg/L. The assessment is however worst case since it was assumed that all the industries (formulation of end-products as well as the industrial use of end-products are in the same catchment area to a STP. The risk is therefore found acceptable both for iodate as well as for iodide.

2.2.3. List of endpoints

In order to facilitate the work of Member States in granting or reviewing authorisations, the most important endpoints, as identified during the evaluation process, are listed in Appendix I.

In the list of endpoints analytical methods for residues in tissue and body fluids have been added compared to the one displayed for IPBC in PT8.

3. PROPOSED DECISION

3.1. Background to the proposed decision

IPBC has a proposed classification as Acute Tox 3, Eye Dam. 1, Acute Tox 4, Skin Sens. 1, STOT SE3, Aquatic Acute 1, Acute Chronic 1 with H331, H318, H302, H317, H335, H400 and H410 (T, N, R22-23-37-41-43-50).

The assessment has been performed based on the documentation for the active substance and the representative biocidal solvent based model formulation containing 30% IPBC. The biocidal product is intended to control fungi and yeast and is for industrial in-can preservation process. The concentration of IPBC in the product to be preserved is in the range of 0.01% to 1.0%. The end-products cover washing and cleaning fluids and other detergents, paint and coatings, fluids used in textile production and glues and adhesives for indoor use.

The assessment of the biocidal activity of the active substance IPBC demonstrates that is has a sufficient level of efficacy against fungi and yeast and efficacy results show that IPBC based in-can preservative products may be expected to be efficacious. However, further efficacy data may be required on specific products to support product authorisation at Member State level.

The risk characterisation for human health indicates that there is no unacceptable risk anticipated for industrial workers during in-can preservation and during application of in-can preserved products.

In the risk characterisation for the environment, results from the consumption based approach show no unacceptable risk for all the single uses of IPBC. However, when considering the cumulative risk (Tier 1: all domestic and industrial emissions together) risk is found for the sewage treatment plant and the surface water for marked penetration factors of both 1 and 0.5. When considering the total domestic load (Tier 2) for the consumption based approach no unacceptable risk is identified for any environmental compartments for marked penetration factors of both 1 and 0.5. Also when the total domestic load is combined with any single industrial emission, no unacceptable risk is found for a marked penetration factor of 0.5. For a marked penetration factor of 1 unacceptable risk is found for the STP and the surface water when the total domestic load is combined with industrial emissions from either washing and cleaning fluids or textile production. For the tonnage based approach for applications considered in this dossier the requirements for acceptable risk according to the TGD on Risk Assessment in the STP, the surface water compartment as well as the soil compartment are met. Also when considering the cumulative emission from all the formulations and usages no unacceptable risk is identified. In the risk characterisation performed for the degradation products iodide and iodate unacceptable risk is identified for iodide in surface water and soil. For soil, predicted concentrations are well within the background level which is found acceptable. Further, there was no risk identified for contamination of groundwater at levels of 0.1 µg/L or above for IPBC and the degradation product PBC. For the degradation products iodide and iodate expected levels are within the background level of 1-70 µg/L in eight of the nine considered scenarios. In one scenario the estimated concentration is only slightly above the background level of 70 µg/L. IPBC and PBC are also not characterised as PBT substances since only the criterion for toxic (T) is fulfilled for IPBC.

Assessed from the documentation for the active substance, IPBC, and the representative artificial “dummy” product containing IPBC, the proposed manner and areas of use of products intended to control fungi and yeast may be sufficiently effective for these uses and without unacceptable risk either to human health or to the environment.

This overall conclusion relies on the fact that users of the biocidal product will be applying the basic principles of good practice and respect the conditions for use recommended on the label of the product.

3.2. Proposed decision

The overall conclusion from the evaluation of IPBC for use in Product Type 6 (Preservatives for products during storage), is that it may be possible to issue authorisations of products containing IPBC in accordance with the conditions laid down in Article 5(1) b), c) and d) of Dir. 98/8/EC.

It is therefore proposed to approve IPBC as an active substance for use in product-type 6 (preservatives for products during storage), subject to the following specific conditions:

The product assessment shall pay particular attention to the exposures, the risks and the efficacy linked to any uses covered by an application for authorisation, but not addressed in the Union level risk assessment of the active substance.

Authorisations are subject to the following condition:

For industrial or professional users, safe operational procedures and appropriate organizational measures shall be established. Where exposure cannot be reduced to an acceptable level by other means, products shall be used with appropriate personal protective equipment. Where a treated article has been treated with or intentionally incorporates IPBC, and where necessary due to the possibility of skin contact as well as the release of IPBC under normal conditions of use, the person responsible for placing the treated article on the market shall ensure that the label provides information on the risk of skin sensitisation, as well as the information referred to in the second subparagraph of Article 58(3) of Regulation (EU) No 528/2012.

3.3. Elements to be taken into account when authorising products

Products containing IPBC have been evaluated for the use to control fungi as an in-can preservative.

The following manner and area of use of products containing IPBC have been evaluated with the following specified maximum concentrations of IPBC:

PT 6 In-can preservative	Field of use envisaged for in-can preservation in:*	Concentration at which IPBC have been evaluated: **
	6.1 Washing and cleaning fluids and detergents Industrial in-can preservation process Industrial, professional and non-professional use of end-products	biocidal product: 15% - 30% end-product: 0.1%

PT 6 In-can preservative	Field of use envisaged for in-can preservation in:*	Concentration at which IPBC have been evaluated: **
	6.2 Paints and coatings Industrial in-can preservation process Industrial, professional and non-professional use of end-products	biocidal product: 30% end-product: 0.01% - 0.1%
	6.3.2 Fluids used in textile production Industrial in-can preservation process Industrial and professional use of end-products	biocidal product: 30% end-product: 0.1%
	6.6 Glues and adhesives (indoor) Industrial in-can preservation process Industrial, professional and non-professional use of end-products	biocidal product: 10% - 30% end-product: 1%

* The source for categorisation is Van der Poel and Bakker (2001) and modified after Doc. 6.4 from 41st CA meeting.

** The concentration is given in % (w/w). A product containing 30% IPBC was chosen as the representative model formulation. However, in the human health exposure and risk assessments biocidal products with an IPBC concentration in the range of 10-40% have been evaluated.

The following uses within PT6 have not been evaluated: In-can preservation of fluids used in paper production, fluids used in leather production, metal working fluids, fuels and glues and adhesives for outdoor use.

If products containing the active substance is recommended for preserving paper that comes into contact with food a dietary assessment may be necessary. This should be taken into account at product authorisation.

In the product authorisation phase, simulated use studies are required on the efficacy of the test substance over longer periods of time and on the effects of interfering substances in the products to be preserved

Dermal absorption values used in the applications for product authorisation should be justified, if available by the submission of specific dermal absorption data on the product, or by read-across to existing data if scientifically justified, or by using default values.The final classification must when adopted by the REACH Committee be considered during the authorization of biocidal products.

3.4. Requirement for further information

It is considered that the evaluation has shown that sufficient data have been provided to verify the outcome and conclusions, and permit the proposal for the approval of IPBC in accordance with Article 9 of Regulation No 528/2012.

3.5. Updating this Assessment Report

This assessment report may need to be updated periodically in order to take account of scientific developments and results from the examination of any of the information submitted in relation with Regulation (EU) No 528/2012. Such adaptations will be examined and finalised in connection with any amendment of the conditions for the approval of IPBC.

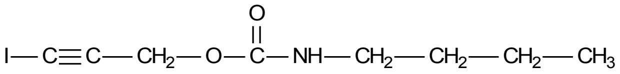
Appendix I: List of endpoints

Chapter 1: Identity, Physical and Chemical Properties, Classification and Labelling

Active substance (ISO Common Name)	IPBC, 3-Iodo-2-propynyl butyl carbamate
Function (e.g. fungicide)	Fungicide

Rapporteur Member State	Denmark

Identity (Annex IIA, point II.)

Chemical name (IUPAC)	3-Iodo-2-propynyl butyl carbamate
Chemical name (CA)	3-Iodo-2-propynyl butyl carbamate
CAS No	55406-53-6
EC No	259-627-5
Other substance No.	Not relevant
Minimum purity of the active substance as manufactured (g/kg or g/l)	980 g/kg
Identity of relevant impurities and additives (of toxicological, environmental and/or other significance) in the active substance as manufactured (g/kg)	None
Molecular formula	$C_8H_{12}INO_2$
Molecular mass	281.1 g/mol
Structural formula	 $I—C{\equiv}C—CH_2—O—\overset{O}{\overset{\Vert}{C}}—NH—CH_2—CH_2—CH_2—CH_3$

Physical and chemical properties (Annex IIA, point III)

Melting point (state purity)	65.8 – 66.5 °C (≥ 98.8 %)
Boiling point (state purity)	No boiling point (≥ 98.8 %), decomposes
Temperature of decomposition	> 85 °C
Appearance (state purity)	Technical ≈ 98%: crystalline slightly yellow solid with a faint odour of iodine Pure 99.6%: white needles
Relative density (state purity)	1.714 (98.8%)
Surface tension	69.1 mN/m at 158 mg/L
Vapour pressure (in Pa, state temperature)	2.36-4.5 × 10^{-3} Pa (at 25 °C)
Henry's law constant (Pa m^3 mol^{-1})	3.38-6.45 × 10^{-3} Pa × m^3 × mol^{-1} (at 25 °C)
Solubility in water (g/l or mg/l, state temperature)	pH___4___: 182 mg/L (20°C)
	pH___7___: 168 mg/L (20°C)
	pH___9___: 176 mg/L (20°C)
Solubility in organic solvents (in g/l or mg/l, state temperature) (Annex IIIA, point III.1)	3.5 g/L for heptane 3.6 g/L for petroleum ether 281 g/L for ethyl acetate 150 g/L for octanol > 1000 g/L for methanol
	all at 20 °C
Stability in organic solvents used in biocidal products including relevant breakdown products (IIIA, point III.2)	Stable in octanol, heptane and ethyl acetate for 96 h, storage at ambient conditions Stable in octanol, petroleum ether and methanol for 9 days when stored at 25 °C
Partition coefficient (log P_{OW}) (state temperature)	2.81 (25°C) Effect of pH is not relevant; IPBC is neither an acid nor a base. pH___5___:
	pH___9___:
	pH___7___:
Hydrolytic stability (DT_{50}) (state pH and temperature) (point VII.7.6.2.1)	pH 4: 267 days (25°C)
	pH 7: 248 days (25°C)
	pH 9: 229 – 539 days (25°C) pH 9: 11.8 days (50°C)

Dissociation constant (not stated in Annex IIA or IIIA; additional data requirement from TNsG)	Not applicable, non-ionic material
UV/VIS absorption (max.) (if absorption > 290 nm state ε at wavelength)	maxima at 191 nm and 227 nm
Photostability (DT_{50}) (aqueous, sunlight, state pH) (point VII.7.6.2.2)	No significant absorption > 290 nm. However, in ethanol solutions, irradiated with sunlight or UV lamps, ca. 25 % of the initial IPBC was degraded within 17 days of exposure. A new study demonstrates that IPBC is stable to direct and indirect photolysis in the aquatic environment. This is selected as the key study
Quantum yield of direct photo-transformation in water at Σ > 290 nm (point VII.7.6.2.2)	No significant absorption > 290 nm. Therefore, quantum yield of direct photolysis was not determined.
Flammability	Not highly flammable, not auto flammable
Explosive properties	Not explosive properties.

Classification and proposed labelling (Annex IIA, point IX.)

with regard to physical/chemical data	None
with regard to toxicological data	T, R22, R23, R37, R41, R43
with regard to fate and behaviour data	None
with regard to ecotoxicological data	N, R50

Chapter 2: Methods of Analysis

Analytical methods for the active substance

Technical active substance and the metabolite PBC (principle of method) (Annex IIA, point 4.1)	HPLC-UV GC-FID
Impurities in technical active substance (principle of method) (Annex IIA, point 4.1)	HPLC-UV GC-FID

Analytical methods for residues

Soil (principle of method and LOQ) (Annex IIA, point 4.2)	IPBC/PBC: HPLC-MS/MS, LOQ = 0.01 mg/kg
Air (principle of method and LOQ) (Annex IIA, point 4.2)	Not necessary, IPBC is not volatile and spray applications only involve non-respirable particles.
Water (principle of method and LOQ) (Annex IIA, point 4.2)	IPBC/PBC: Both for surface water, ground water and drinking water. HPLC-MS/MS, LOQ = 0.1 µg/L
Body fluids and tissues (principle of method and LOQ) (Annex IIA, point 4.2)	Relevant residues for monitoring human body fluid and tissues were PBC and IPBC. In blood and muscle IPBC

	degraded rapidly (to PBC) and it was not possible to determine IPBC residues above 70%. Analysis was done by HPLC using reversed-phase liquid chromatography and a water / methanol gradient on a C18-column. Detection was made with a MS/MS system using positive electrospray ionisation. LOQ for PBC and IPBC in urine and blood at 0.05 mg/L. LOQ for PBC and IPBC in meat at 0.1 mg/L.
Food/feed of plant origin (principle of method and LOQ for methods for monitoring purposes) (Annex IIIA, point IV.1)	Not necessary. The use of IPBC as an in-can preservative result to a low concentration of IPBC in the end-product (0.1%). Therefore, the amount of IPBC transferred to food or feeding stuff from material treated with the end-product is considered to be negligible (see Doc. IIB chapter 8.2.2.1).
Food/feed of animal origin (principle of method and LOQ for methods for monitoring purposes) (Annex IIIA, point IV.1)	Not necessary. See above.

Chapter 3: Impact on Human Health

Absorption, distribution, metabolism and excretion in mammals (Annex IIA, VI.6.2)

Rate and extent of oral absorption:	>90% based on urinary excretion (~57-71%) and exhaled air (~18-24%) within 72 hours.
Rate and extent of dermal absorption:	1.6 % (17% IPBC in xxxxxxxx) 10 % (2.4% IPBC in xxxxxxxx) 30 % (0.6% IPBC in xxxxxxx) 100% default for solutions containing <0.5%-0.6% IPBC 1.6% for solid formulations and dried solutions (based on *in vitro* human skin study (Jack & Dunsire, 1995).
Distribution:	Uniformly distributed
Potential for accumulation:	No evidence for bioaccumulation
Rate and extent of excretion:	> 77-99% within 72 hours mainly via urine (57.3 to 70.7%). Excretion in exhaled air were 18.3 to 24.0% and in faces 4.4%-7.4%.
Toxicologically significant metabolite	Iodine

Acute toxicity (Annex IIA, VI.6.1)

Rat LD_{50} oral	300 – 500 mg/kg bw,	R22

Rat LD_{50} dermal	> 2000 mg/kg bw/day
Rat LC_{50} inhalation	> 6.89 mg/L technical IPBC (for not respirable dust) 0.67 mg IPBC/L for respirable dust R23 0.763 mg IPBC/L for respirable liquid aerosol R23
Skin irritation	Non-irritant
Eye irritation	Severe eye-irritant R41
Skin sensitization (test method used and result)	Sensitizing (M&K) R43

Repeated dose toxicity (Annex IIA, VI. 6.3, 6.4, and 6.5)

Species/ target / critical effect	Rat (oral): reduced body weight and body weight gain , increased organ weights (liver and kidney) and increased iron concentration Histopathological changes in the stomach.
Lowest relevant oral $NOAEL_{short\text{-}term}$	35 mg/kg bw/day (90 day gavage rat)
Lowest relevant oral $NOAEL_{long\text{-}term}$	20 mg/kg bw/day (2 years oral rat)
Lowest relevant dermal NOAEL	90-day dermal study in rats: 200 mg/kg bw/day,
Lowest relevant inhalation NOAEL	90-day inhalation study in rats: 1.16 mg/m^3 R37: Irritating to respiratory system.

Genotoxicity (Annex IIA, VI.6.6)	The overall weight of evidence indicates that IPBC is not a genotoxic substance.

Carcinogenicity (Annex IIA, VI.6.7)

Species/type of tumour	No evidence for carcinogenic potential in rats and mice
lowest dose with tumours	Not applicable

Reproductive toxicity (Annex IIA, VI.6.8)

Species/ Reproduction target / critical effect	Rat: *Parents*: clinical signs and local effects on the stomach. *Developmental*: *in F1 generation reduced pup viability and cumulative survival index. Reduced pup weight (F1 and F2 females) and increased incidence of pups without milk in stomach and/or bitten or cannibalized at maternal toxic doses* *Reproduction:* Reduced fertility/mating index at

	maternal toxic doses.
Lowest relevant reproductive NOAEL	Parental: 10 mg/kg bw/day Developmental: 10 mg/kg bw/day Reproductive: 30 mg/kg bw/day
Species/Developmental target / critical effect	Rabbit: Maternal: not statistically significant reduced food consumption in week 1 and one animal that refused to eat due to stomach irritations resulted in body weight loss and subsequent pre-scheduled sacrifice of this animal. Developmental: no treatment related findings
Lowest relevant developmental NOAEL	Dams: 10 mg/kg bw/day Developmental: 40 mg/kg bw/day

Neurotoxicity (Annex IIIA, VI.1)

Species/ target/critical effect	Rat: *Systemic:* reduced body weight gain/body weight and food consumption at 50 and 120 mg/kg bw/day (m+f) No signs of neurotoxicity have been observed after acute and subchronic oral treatment.
Lowest relevant neurotoxicity NOAEL	120 mg/kg bw/day (90 days oral rat neurotoxicity study)

Other toxicological studies (Annex IIIA, VI/XI)

	No data available - not required

Medical data (Annex IIA, VI.6.9)

	No evidence of adverse effects to workers of manufacturing plants or professional painters. Skin sensitisation in workers/patients reported.

Summary (Annex IIA, VI.6.10)	Value	Study	Safety factor
ADI (if residues in food or feed)	0.2 mg/kg bw/day (no correction for oral absorption) relevant for the intended use	2-years rats study	100
$AEL_{long\ term}$	0.2 mg/kg bw/day	2-years rats study	100
$AEL_{medium-term}$	0.35 mg/kg bw/day	90-day gavage rat study	100
AEL_{acute}	0.35 mg/kg bw/day	90-day gavage rat study	100
ARfD (acute reference dose)	0.35 mg/kg bw/day (no correction for oral absorption)	90-day gavage rat study	100

Acceptable exposure scenarios (including method of calculation*)

Industrial use	Acceptable uses identified (with PPE) in the risk characterization of systemic effects (For details of %AEL and MOE (margin of exposure) refer to doc IIB for the different scenarios.)
Professional use	Acceptable uses identified (with PPE)) in the risk characterization of systemic effects. (For details of %AEL and MOE (margin of exposure) refer to doc IIB for the different scenarios.)
Amateur use	Acceptable uses identified in the risk characterization of systemic local effects. (For details of %AEL and MOE (margin of exposure) refer to doc IIB for the different scenarios.)
Secondary exposure	Secondary exposure levels are low and acceptable. (For details of %ADI/AEL and MOE (margin of exposure) refer to doc IIB for the different scenarios.)

* please refer to Table 1-2 and Table 1-3 in doc IIC and doc IIB regarding methods of calculation.

Chapter 4: Fate and Behaviour in the Environment

Route and rate of degradation in water (Annex point IIA, VII.7.6; Annex point IIIA, XII.2.1, 2.2)

Hydrolysis of active substance and relevant metabolites (DT_{50}) (state pH and temperature)	pH 4: 267 days (25°C) pH 4: 755 days (12°C)
	pH 7: 248 days (25°C) pH 7: 702 days (12°C)
	pH 9: 229 – 539 days (25°C) pH 9: 648 – 1525 days (12°C) pH 9: 11.8 days (50°C) no major metabolites
Photolytic / photo-oxidative degradation of active substance and resulting relevant metabolites	IPBC is stable to direct and indirect photolysis in the aquatic environment as demonstrated for sterilized buffer and natural pond water at 25°C for up to 3 days
Readily biodegradable (yes/no)	No
Inherent biodegradability	IPBC is primary biodegradable according to Zahn-Wellens test. IPBC degrades rapidly (within 2 hours) to PBC.
Biodegradation in seawater	A study on biodegradation in seawater is not required for PT 6.
Anaerobic water/sediment study:	**IPBC**:
DT_{50} total systems (nonsterile)	1.5 hours (for the total system at 22°C) 3.3 hours (for the total system at 12°C)
DT_{90} total systems (nonsterile)	5.0 hours (for the total system at 22°C) 11 hours (for the total system at 12°C)
DT_{50} total systems (sterile)	13.3 hours at 22°C 29 hours at 12°C
DT_{90} total systems (sterile)	44.3 hours at 22°C 96 hours at 12°C
	PBC:
DT_{50} total systems (nonsterile)	11.5 days at 22°C 25 days at 12°C
DT_{90} total systems (nonsterile)	38.4 days at 22°C 83 days at 12°C
Mineralization (nonsterile)	Mineralization is 10% after 120 days in nonsterile

Mineralization (sterile)	continuous N2 flow samples 21% after 119 days in nonsterile enclosed samples 42% after 93 days in nonsterile static samples Mineralization 0%
Non-extractable residues	3.9 – 6.3 % AR after 162/119 days
Distribution in water / sediment systems (active substance)	78% remained in the water phase and less than 10% in the sediment (at day 0)
Distribution in water / sediment systems (metabolites)	Propargyl butyl carbamate (PBC): Up to 88.6 % was available in the water phase (at 8 hours) Up to 13.3% was available in the sediment (at 4 hours) in nonsterile static samples. Up to 20.9 % (at day 1) was available in sterile static samples. 2-propenyl butyl carbamate (2-PBC): Surface water: - up to 34.7 % at day 59 in nonsterile static samples. - up to 35.4 % at day Day 59 in nonsterile enclosed static samples Sediment: - up to 8.0 % at day 59 in nonsterile static samples - up to 8.8 % at day 93 in nonsterile enclosed static samples

Route and rate of degradation in soil (Annex point IIIA, VII.4, XII.1.1, XII.1.4; Annex VI, para. 85)

Mineralization (aerobic)	75.3 % AR after 21 days (nonsterile, 22°C, n = 1) 5.3 % AR after 14 days (nonsterile, 5°C, n = 1) 2.3 % AR after 28 days (sterile, 22°C, n = 1)
Laboratory studies (range or median, with number of measurements, with regression coefficient)	DT_{50lab} (22°C, aerobic): 2.1 hours (n = 1) DT_{50lab} (12°C, aerobic): 4.7 hours (calculated according to Arrhenius)
	DT_{90lab} (22°C, aerobic): 7.1 hours (n = 1)
	DT_{50lab} (5°C, aerobic): 8.6 hours (n = 1)

	DT$_{50lab}$ (22°C, anaerobic): 1.5 hours in anaerobic water/sediment systems
Field studies (state location, range or median with number of measurements)	DT$_{50f}$: not required due to fast degradation of IPBC in soil (DT$_{50lab}$ = 2.1 hours at 22°C)
	DT$_{90f}$: not required due to fast degradation of IPBC in soil (DT$_{90lab}$ = 7.1 hours)
Anaerobic degradation	See anaerobic water/sediment study
Soil photolysis	Not required because the degradation of IPBC in soil is primarily microbially mediated.
Non-extractable residues	21.4% AR after 14 days which is the maximum value (nonsterile, 22°C, n = 1) 9.6 % AR after 14 days (nonsterile, 5°C, n = 1) 3.0 % AR after 28 days (sterile, 22°C, n = 1)
Relevant metabolites - name and/or code, % of applied active ingredients (range and maximum)	Propargyl butyl carbamate (PBC): 95 % AR after 12 hours DT$_{50}$: 10 days at 12^{o}C (calculated according to Arrhenius)
Soil accumulation and plateau concentration	Not required due to fast degradation of IPBC in soil

Adsorption/desorption (Annex point IIA, XII.7.7; Annex point IIIA, XII.1.2)

Ka , Kd Ka_{oc} , Kd_{oc} pH dependence (yes / no) (if yes type of dependence)	Ka: 0.676 – 2.46; Kd: 3.43 – 31.3 (n =5) Ka_{OC}: 61.0 – 309; Kd_{OC}: 457 – 4065 (n =5) Geomean 113.5 (log 2.1) Arithmetic mean: 134.5 Koc (HPLC method): 126 (log Koc = 2.1) no
K_{OC} PBC	198.1 (estimated by PCKOC v1.66)

Fate and behaviour in air (Annex point IIIA, VII.3, VII.5)

Direct photolysis in air	Not studied – no data request
Quantum yield of direct photolysis	No significant absorption > 290 nm. Therefore, quantum yield of direct photolysis was not determined.
Photo-oxidative degradation in air	DT_{50} of 15 hours (for OH radical reaction) derived by the Atkinson method of calculation.
Volatilization	IPBC is only slightly volatile (vapour pressure = 2.36 - 1.4 x 10^{-3} Pa).

Monitoring data, if available (Annex VI, para. 44)

Soil (indicate location and type of study)	No monitoring data for the EU have been reported.
Surface water (indicate location and type of study)	No monitoring data for the EU have been reported.
Ground water (indicate location and type of study)	No monitoring data for the EU have been reported.
Air (indicate location and type of study)	No monitoring data for the EU have been reported.

Chapter 5: Effects on Non-target Species

Toxicity data for aquatic species (most sensitive species of each group) for IPBC
(Annex IIA, VII. 7.1 - 7.4, Annex IIIA, XII. 2.2 and XII 2.4)

Species	Time-scale	Endpoint	Toxicity
Fish			
Rainbow trout *(Oncorhynchus mykiss)*	96 hours	Mortality	LC_{50}: 0.067 mg/L NOEC: 0.049 mg/L
Fathead minnow (*Pimephales promelas*)	35 days	Larval growth (length and weight)	NOEC: 0.0084 mg/L
Invertebrates			
Daphnia magna	48 hours	Mortality	EC_{50}: 0.160 mg/L EC_0: 0.076 mg/L
Daphnia magna	21 days	Mortality, reproduction and growth effects	NOEC: 0.050 mg/L
Algae			
Scenedesmus subspicatus	72 hours	Growth inhibition	E_bC_{50}: 0.022 mg/L E_rC_{50}: 0.053 mg/L NOEC 0.0046 mg/L
Microorganisms			
Activated sludge	3 hours	Respiration inhibition	EC_{50}: 44 mg/L

Toxicity data for aquatic species (most sensitive species of each group) for PBC
(Annex IIA, VII. 7.1 - 7.3)

Species	Time-scale	Endpoint	Toxicity
Fish			
Rainbow trout *(Oncorhynchus mykiss)*	96 hours	Mortality	LC_{50}: 85.0 mg/L
Invertebrates			
Daphnia magna	48 hours	Mortality	EC_{50}: 60 mg/L EC_0: 17 mg/L
Algae			
Selenastrum capricornutum	96 hours	Growth inhibition	E_bC_{50}: > 41.3 mg/L E_rC_{50}: > 41.3 mg/L NOEC: 21.2 mg/L

Effects on earthworms or other soil non-target organisms
(Annex IIIA, XIII.3.2)

Acute toxicity to earthworm (Annex IIIA, point XIII.3.2)	LC_{50}: > 1000 mg/kg dry soil
Reproductive toxicity to ………………………… (Annex IIIA, point XIII.3.2)	Not required

Effects on soil micro-organisms
(Annex IIA, VII.7.4)

Nitrogen mineralization	EC_{50} value could not be determined
Carbon mineralization	EC_{50}: 312.5 mg/ kg dry soil

Effects on plants
(Annex IIIA, XIII.3.4)

Toxicity to plants (*Avena sativa*)	EC_{50}: 4.92 mg/kg dry soil (based on fresh weigh reduction)

Effects on terrestrial vertebrates

Acute toxicity to mammals (Annex IIIA, point XIII.3.3)	Not required for Product type 6
Acute toxicity to birds (Annex IIIA, point XIII.1.1)	Not required for Product type 6
Dietary toxicity to birds (Annex IIIA, point XIII.1.2)	Not required for Product type 6
Reproductive toxicity to birds (Annex IIIA, point XIII.1.3)	Not required for Product type 6

Effects on honeybees (Annex IIIA, point XIII.3.1)

Acute oral toxicity	Not required for Product type 6
Acute contact toxicity	Not required for Product type 6

Effects on other beneficial arthropods (Annex IIIA, point XIII.3.1)

Acute oral toxicity	Not required for Product type 6
Acute contact toxicity	Not required for Product type 6
Acute toxicity to …………………………………..	Not required for Product type 6

Bioconcentration (Annex IIA, point 7.5)

Bioconcentration factor (BCF)	Not relevant for IPBC (see Doc. IIIA, Section 7.4.2 and Section 7.5.5)

Depration time (DT$_{50}$) (DT$_{90}$)	Not relevant for IPBC (see Doc. IIIA, Section 7.4.2 and Section 7.5.5)
Level of metabolites (%) in organisms accounting for > 10 % of residues	Not relevant for IPBC (see Doc. IIIA, Section 7.4.2 and Section 7.5.5)

Chapter 6: Other End Points

Not applicable, no other end points

Appendix II: List of Intended Uses

Summary of intended uses for IPBC-based in-can preservation products (PT6)

Object and/or situation	**Member State or Country**	**Product name**	**Organisms controlled**	**Formulation**		**Application**			**Applied amount per treatment**			**Remarks:**
(a)			(c)	Type (d-f)	Conc. of as (i)	method kind (f-h)	number min max (k)	interval between applications (min)	g as/L min max	water L/m^2 min max	g as/m^2 min max	(m)
In-can preservation	EU	n.a.	Fungi including yeasts	n.a.	10%-30%	The in-can preservation product containing IPBC is added to the products to be preserved in an automated process. The end-products cover washing and cleaning fluids and other detergents, paints and coatings, fluids used in textile production and glues and adhesives for indoor use. One application / can			The concentration of IPBC in the product to be preserved is in the range of 0.01% to 1.0%			In-can preservation process: Professionals End products can be used by professionals and amateurs.

(a) *e.g.* biting and suckling insects, fungi, molds;
(b) *e.g.* wettable powder (WP), emulsifiable concentrate (EC), granule (GR)
(c) GCPF Codes - GIFAP Technical Monograph No 2, 1989 ISBN 3-8263-3152-4);
(d) All abbreviations used must be explained
(e) g/kg or g/l;
(f) Method, *e.g.* high volume spraying, low volume spraying, spreading, dusting, drench;
(g) Kind, *e.g.* overall, broadcast, aerial spraying, row, bait, crack and crevice equipment used must be indicated;
(h) Indicate the minimum and maximum number of application possible under practical conditions of use;
(i) Remarks may include: Extent of use/economic importance/restrictions

Appendix III: List of studies

The Doc IIIA Reference list sorted by Section No:

Section No./ Reference No.	Author(s)	Year	Title Source (laboratory) Report No. GLP; (un)published Doc. No.	Data protection	Owner
A3.1.1/01 Submitted with the PT8 BPD dossier	Jungheim	2000	Preventol MP 100 - Physicochemical properties Source: Bayer AG, Leverkusen, Germany Report No.: N 00/0070/02 LEV GLP; (unpublished) Doc. No.: 112-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	LANXESS Deutschland GmbH
A3.1.1/02 Submitted with the PT8 BPD dossier	Rodriguez, O.	1990	Melting Point of TROYSAN Polyphase P100 3-Iodo-2-Propynyl Butyl Carbamate Source: Troy Corporation, USA Report No.: TC-0236 TAL 8/20/90 GLP; (unpublished) Doc. No.: 112-002	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A3.1.1/03 Submitted with the PT8 BPD dossier	Polson, G.	1994	Physical and chemical properties of 3-iodo-2-propynylbutylcarbamate (Omacide IPBC) Source: Olin Research Center, Cheshire Report No.: 93B02IPBC GLP; (unpublished) Doc. No.: 119-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	ARCH Chemicals
A3.1.1/04 Submitted with the PT8 BPD dossier	Morrissey, M.A.	1997	Product chemistry determinations of IPEX 1000 (Color, Physical State) Source: Corning Hazleton Inc., Virginia, USA Report No.: CHW 6752-101 GLP; (unpublished) Doc. No.: 119-002	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	DOW Benelux B.V.
A3.1.3/01 Submitted with the PT8 BPD dossier	Anonymous	1990	True density of TROYSAN Polyphase P100 Source: Quantachrome Corporation, N.Y., United States Report No.: TC-0246 90-1478 GLP; (unpublished) Doc. No.: 113-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A3.2.1/01 Submitted with the PT8 BPD dossier	Görg, J.	2004	Calculation fo the Henry´s Law Constant - Active Substance IPBC 3-Iodo-2-propynyl-butylcarbamate Source: Scientific Consulting Company, Wendelsheim, Germany Report No.: 824-006 Not GLP; (unpublished) Doc. No.: 115-004	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	IPBC Task Force (ARCH, ISP, LANXESS, DOW, TROY)

Section No./ Reference No.	Author(s)	Year	Title Source (laboratory) Report No. GLP; (un)published Doc. No.	Data protection	Owner
A3.2/01 Submitted with the PT8 BPD dossier	Olf	2000	Preventol MP 100 - Vapor pressure, Physical-chemical properties Source: Bayer AG, Leverkusen, Germany Report No.: 00/024/01 GLP; (unpublished) Doc. No.: 115-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	LANXESS Deutschland GmbH
A3.2/02 Submitted with the PT8 BPD dossier	Schneider, U.	2002	Final Report: IPBC Determination of the Vapour Pressure Source: Infracor Chemistry Services Report No.: AN-ASB 0202 GLP; (unpublished) Doc. No.: 115-002	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	DOW Benelux B.V.
A3.4/01 Submitted with the PT8 BPD dossier	Seelemann	2000	Preventol MP 100 - Identity/ Spectra Source: Bayer AG, Leverkusen, Germany Report No.: N 00/0070/00 LEV GLP; (unpublished) Doc. No.: 117-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	LANXESS Deutschland GmbH
A3.4/02 Submitted with the PT8 BPD dossier	Anonymous	1997	Spectra for IPBC: GC-MS, UV, IR Source: Olin Central analytical Laboratory, Cheshire Report No.: grl 2/6/97 Not GLP; (unpublished) Doc. No.: 117-002	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	ARCH Chemicals
A3.4/03 Submitted with the PT8 BPD dossier	Lloyd, G.R.	1997	3-Iodo-Propynyl-Butyl-Carbamate (IPBC) - NMR traces Source: Olin Central analytical Laboratory, Cheshire Report No.: 19/8/97 Not GLP; (unpublished) Doc. No.: 117-003	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	ARCH Chemicals
A3.4/04 Submitted with the PT8 BPD dossier	Wojcieck, B.C.	1994	IPBC - Ultraviolet-Visible Absorption Spectrum (Amended Report) Source: Ricerca, LLC, Painesville OH Report No.: TC-0617 4257-93-0276-AS-001-002 GLP; (unpublished) Doc. No.: 117-004	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A3.5/01 Submitted with the PT8 BPD dossier	Morrissey, M.A.	1997	Solubility determination of IPEX 1000 Source: Covance Laboratories Inc., Virginia Report No.: Covance 6752-105 GLP; (unpublished) Doc. No.: 114-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	DOW Benelux B.V.

Section No./ Reference No.	Author(s)	Year	Title Source (laboratory) Report No. GLP; (un)published Doc. No.	Data protection	Owner
A3.5/02 Submitted with the PT8 BPD dossier	Jungheim	2000	Preventol MP 100 - Water solubility Source: Bayer AG, Leverkusen, Germany Report No.: N 00/0070/03 LEV GLP; (unpublished) Doc. No.: 114-002	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	LANXESS Deutschland GmbH
A3.5/03 Submitted with the PT8 BPD dossier	Cameron, B.D. Machon, A.	1986	The solubility of IPBC in buffers pH 5.0, 7.0 and 9.0 incubated at 25 °C Source: Inveresk Research Institute Report No.: TC-0244 135124 4166 GLP; (unpublished) Doc. No.: 114-004	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A3.6/01 Submitted with the PT8 BPD dossier	Siemann, L.	1990	Analysis of Polyphase P100 - Dissociation Constant (63-10) Source: Midwest Research Institute, Kansas City, United States Report No.: TC-0247 9555-F(01) GLP; (unpublished) Doc. No.: 115-003	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A3.9/01 Submitted with the PT8 BPD dossier	Jungheim	2000	Preventol MP 100 - Partition coefficient (n-octanol/water) Source: Bayer AG, Leverkusen, Germany Report No.: N 00/0070/04 LEV GLP; (unpublished) Doc. No.: 114-003	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	LANXESS Deutschland GmbH
A3.9/02 Submitted with the PT8 BPD dossier	Siemann, L.	1990	Analysis of Polyphase P100 - Octanol/Water Partition coefficient (63-11) Source: Midwest Research Institute, Kansas City, United States Report No.: TC-0248 9555-F (01) GLP; (unpublished) Doc. No.: 114-005	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A3.10/01 Submitted with the PT8 BPD dossier	Polson, G.	1997	Physical and chemical properties of 3-Iodo-2-Propynylbutylcarbamate (IPBC-100) Source: Olin Central analytical Laboratory, Cheshire Report No.: 18-94B07IPBC GLP; (unpublished) Doc. No.: 146-002	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	ARCH Chemicals

Section No./ Reference No.	Author(s)	Year	Title Source (laboratory) Report No. GLP; (un)published Doc. No.	Data protection	Owner
A3.10/02 Submitted with the PT8 BPD dossier	Lezotte, F. MacGregor, J. Chafey, K. Nixon, W.B.	2001	Determination of storage stability of IPBC technical (PROTRAM 98) at ambient and elevated temperatures (Interim Report - Elevated temperature phase) Source: Wildlife International Ltd., Easton, Maryland, USA Report No.: 526C-103 GLP; (unpublished) Doc. No.: 146-003	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	DOW Benelux B.V.
A3.10/03 Submitted with the PT8 BPD dossier	Sinning, D.J.	1999	Physical and Chemical Characteristics of TROYSAN Polyphase 100 - Stability Source: Case Consulting Laboratories, Inc., Whippany, N.J., United States Report No.: TC-0926 650-25 GLP; (unpublished) Doc. No.: 146-005	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A3.11/01 Submitted with the PT8 BPD dossier	Lindemann, M.	2004	Determination of the flammability of IPBC technical Source: Research and Consulting Company, Itingen, Switzerland Report No.: 851398 GLP; (unpublished) Doc. No.: 142-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	IPBC Task Force (ARCH, DOW, LANXESS, TROY)
A3.11/02 Submitted with the PT8 BPD dossier	Lindemann, M.	2004	Determination of the relative self-ignition temperature of IPBC technical Source: Research and Consulting Company, Itingen, Switzerland Report No.: 851402 GLP; (unpublished) Doc. No.: 142-002	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	IPBC Task Force (ARCH, DOW, LANXESS, TROY)
A3.13/01 Submitted with the PT8 BPD dossier	Olf	2000	Preventol MP 100 - Surface tension, physical-chemical properties Source: Bayer AG, Leverkusen, Germany Report No.: 00/024/03 GLP; (unpublished) Doc. No.: 116-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	LANXESS Deutschland GmbH
A3.15 Submitted with the PT8 BPD dossier	Görg, J.	2005	Statement on the explosive properties of 3-Iodopropynylbutyl Carbamate (IPBC) Source: Scientific Consulting Company, Wendelsheim, Germany Report No.: 824-009 Not GLP; (unpublished) Doc. No.: 141-002	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	IPBC Task Force (ARCH, DOW, ISP, LANXESS, TROY)

Section No./ Reference No.	Author(s)	Year	Title Source (laboratory) Report No. GLP; (un)published Doc. No.	Data protection	Owner
A3.16 Submitted with the PT8 BPD dossier	Görg, J.	2005	Statement on the oxidising properties of 3-Iodopropynylbutyl Carbamate (IPBC) Source: Scientific Consulting Company, Wendelsheim, Germany Report No.: 824-009 Not GLP; (unpublished) Doc. No.: 143-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	IPBC Task Force (ARCH, DOW, ISP, LANXESS, TROY)
A4.1/01 Submitted with the PT8 BPD dossier	Anonymous	1993	Water quality - determination of sodium and potassium - Part 1: Determination of sodium by atomic absorption spectrometry Source: International Organization for Standardization, Switzerland, International Standard, ISO 9964-1, First edition 1993-05-01; UDC 614.777:556.114:543.42:546.33 Report No.: ISO 9964-1:1993(E) Not GLP; (published) Doc. No.: 492-003	No	N.R.
A4.1/02 Submitted with the PT8 BPD dossier	Anonymous	N.I.	MT 81 Soluble Alkalinity Source: Miscellaneous Techniques and Impurities, pp. 215-217 Report No.: Not applicable Not GLP; (published) Doc. No.: 492-004	No	N.R.
A4.2a/01 Submitted with the PT8 BPD dossier	Bruckhausen, P.	2004	Development and validation of a residue analytical method for IPBC and its metabolite PBC in soil Source: Research and Consulting Company, Itingen, Switzerland Report No.: 851400 GLP; (unpublished) Doc. No.: 434-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	IPBC Task Force (ARCH, LANXESS, DOW, TROY)
A4.2c/01 Submitted with the PT8 BPD dossier	Bruckhausen, P.	2004	Development and validation of the residue analytical method for the determination of IPBC and ist metabolite PBC in drinking, ground and surface water Source: Research and Consulting Company, Itingen, Switzerland Report No.: 851401 GLP; (unpublished) Doc. No.: 435-002	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	IPBC Task Force (ARCH, DOW, LANXESS, TROY)
A4.2d/01 Submitted with the PT6 BPD dossier	Reisinger, T.	2008	Summary of Preliminary Results - Development and validation of the residue analytical method for the determination of IPBC and its metabolite PBC in Body Fluids and Tissue Source: Scientific Consulting Company, Wendelsheim, Germany Report No.: B49443 Not GLP; (unpublished) Doc. No.: 433-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	IPBC Task Force (ARCH, DOW, ISP, LANXESS, TROY)

Section No./ Reference No.	Author(s)	Year	Title Source (laboratory) Report No. GLP; (un)published Doc. No.	Data protection	Owner
A4.2d/01 Submitted with the PT6 BPD dossier	Düsterloh, K.	2008	IPBC, PBC - Development and validation of a residue analytical method for the determination of IPBC and its metabolite PBC in body fluids and tissue. Source: RCC Ltd, Itingen Switzerland Report No.: B49443 GLP; (unpublished) Doc No. 433-002	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	IPBC Task Force (ARCH, DOW, ISP, LANXESS, TROY)
A6.1.1/01 Submitted with the PT8 BPD dossier	XXXX	2000	Preventol MP 100 - Acute oral toxicity study in male and female wistar rats Source: XXXX Report No.: XXXX 30455 T4069982 GLP; (unpublished) Doc. No.: 521-003	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	LANXESS Deutschland GmbH
A6.1.2/01 Submitted with the PT8 BPD dossier	XXXX	2000	Preventol MP 100 - Acute dermal toxicity study in male and female wistar rats Source: XXXX Report No.: XXXX 30454 T3069981 GLP; (unpublished) Doc. No.: 522-004	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	LANXESS Deutschland GmbH
A6.1.3/01 Submitted with the PT8 BPD dossier	XXXX	1985	Acute inhalation limit test in rats 3-iodo-2-propynyl butyl carbamate Source: XXXX Report No.: TC-0007 Not GLP; (unpublished) Doc. No.: 523-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A6.1.3/02 Submitted with the PT8 BPD dossier	XXXX	1990	TROYSAN Polyphase P-100 - Acute inhalation toxicity study in the rat Source: XXXX Report No.: TC-0004 90-8277 GLP; (unpublished) Doc. No.: 523-002	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A6.1.4/01 Submitted with the PT8 BPD dossier	XXXX	2000	Acute skin irritation test (patch test) of Preventol MP 100 in rabbits Source: XXXX Report No.: XXXX 7891 9300/450/95 XXXX 8069193 GLP; (unpublished) Doc. No.: 565-008	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	LANXESS Deutschland GmbH

Section No./ Reference No.	Author(s)	Year	Title Source (laboratory) Report No. GLP; (un)published Doc. No.	Data protection	Owner
A6.1.4/02 Submitted with the PT8 BPD dossier	XXXX	1998	Primary eye irritation - IPEX 1000 Source: XXXX Report No.: 6042 GLP; (unpublished) Doc. No.: 566-006	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	DOW Benelux B.V.
A6.1.5/01 Submitted with the PT8 BPD dossier	XXXX.	1998	Dermal sensitization test - Buehler Method - IPEX 1000 Source: XXXX Report No.: 6044 GLP; (unpublished) Doc. No.: 567-005	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	DOW Benelux B.V.
A6.1.5/02 Submitted with the PT8 BPD dossier	XXXX	1993	TROYSAN Polyphase P-100 - The guinea pig maximization test Source: XXXX Report No.: TC-0020 14148 GLP; (unpublished) Doc. No.: 567-003	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A6.1.5/03 Submitted with the PT8 BPD dossier	XXXX	2001	Preventol MP 100 - Study for the skin sensitization effect in guinea pigs (Guinea pig maximization test according to Magnusson and Kligman) Source: XXXX Report No.: XXXX 30653 XXXX 5069983 GLP; (unpublished) Doc. No.: 567-010	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	LANXESS Deutschland GmbH
A6.2/01 Submitted with the PT8 BPD dossier	XXXX	1995	Metabolism of 14C-IPBC in rats Source: XXXX Report No.: XXXX 6491-100 TC-0457 GLP; (unpublished) Doc. No.: 512-002	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A6.2/02 Submitted with the PT8 BPD dossier	XXXX	1995	The in vitro percutaneous absorption through human abdominal epidermis of [14C]-IPBC (3-Iodo-2-Propynyl-N-Butyl-Carbamate) Source: XXXX Report No.: 155046 12367 TC0510 GLP; (unpublished) Doc. No.: 511-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation

Section No./ Reference No.	Author(s)	Year	Title Source (laboratory) Report No. GLP; (un)published Doc. No.	Data protection	Owner
A6.3.1/01 Submitted with the PT8 BPD dossier	XXXX	2001	Preventol MP 100 - 3-iodo-2-propynyl-n-butyl carbamate (IPBC) - Study for subacute oral toxicity in rats (gavage study over 4 weeks and 2 weeks recovery period) Source: XXXX Report No.: XXXX 30948 T6069830 GLP; (unpublished) Doc. No.: 532-002	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	LANXESS Deutschland GmbH
A6.3.1/02 Submitted with the PT8 BPD dossier	XXXX	1986	Iodopropynylbutyl carbamate (IPBC) 4 week dieatry dose range finding study in rats Source: XXXX Report No.: TC-0130 435046 3623 GLP; (unpublished) Doc. No.: 532-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A6.3.1/03 Submitted with the PT8 BPD dossier	XXXX	1986	Establishment of methodology and the routine analysis of Iodopropynylbutyl Carbamate in diets prepared for a 4 week dose range finding study (XXXX Project No. 435046) in the Rat Source: XXXX Report No.: 335018 4224 TC0409b GLP; (unpublished) Doc. No.: 437-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A6.3.1/04 Submitted with the PT8 BPD dossier	XXXX	1996	A 2-week range-finding study of TROYSAN Polyphase P100 in the rabbits via dietary administration Source: XXXX Report No.: 95-2395 TC0477 GLP; (unpublished) Doc. No.: 531-006	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A6.3.1/05 Submitted with the PT8 BPD dossier	XXXX	1987	Iodopropynylbutyl carbamate (IPBC) 8 week dietary dose range finding study in mice Source: XXXX Report No.: 5021 436144 TC0409c GLP; (unpublished) Doc. No.: 533-006	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A6.3.3/01 Submitted with the PT8 BPD dossier	XXXX	1994	Omacide IPBC - 2-week repeat dose inhalation toxicity study in rats Source: XXXX Report No.: XXXX 6/932373 GLP; (unpublished) Doc. No.: 531-004	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	ARCH Chemicals

Section No./ Reference No.	Author(s)	Year	Title Source (laboratory) Report No. GLP; (un)published Doc. No.	Data protection	Owner
A6.3.3/02 Submitted with the PT8 BPD dossier	XXXX	1994	Omacide IPBC - 5-day repeat dose inhalation toxicity study in rats Source: XXXX Report No.: XXXX 8/942212 GLP; (unpublished) Doc. No.: 531-005	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	ARCH Chemicals
A6.4.1/01 Submitted with the PT8 BPD dossier	XXXX	2002	Repeated dose toxicity 90-day oral toxicity study in rats with IPBC technical (Protram TM 98) Source: XXXX Report No.: 20-4-0132-01 GLP; (unpublished) Doc. No.: 533-005	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	DOW Benelux B.V.
A6.4.1/02 Submitted with the PT8 BPD dossier	XXXX	1984	90-Day subchronic oral toxicity test in rats Source: XXXX Report No.: TC-0117 GLP; (unpublished) Doc. No.: 533-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A6.4.1/03 Submitted with the PT8 BPD dossier	XXXX	1997	A subchronic (3-month) toxicity study of TROYSAN Polyphase P100 in the rabbits via dietary administration Source: XXXX Report No.: 95-2396 TC0478 GLP; (unpublished) Doc. No.: 533-003	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A6.4.2/01 Submitted with the PT8 BPD dossier	XXXX	1991	91-day dermal toxicity study in rats with TROYSAN Polyphase P-100 Source: XXXX Report No.: TC-0113 3228.14 GLP; (unpublished) Doc. No.: 534-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A6.4.3/01 Submitted with the PT8 BPD dossier	XXXX	1994	Omacide IPBC - 13-week inhalation toxicity study in rats Source: XXXX Report No.: XXXX 7/942772 GLP; (unpublished) Doc. No.: 535-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	ARCH Chemicals

Section No./ Reference No.	Author(s)	Year	Title Source (laboratory) Report No. GLP; (un)published Doc. No.	Data protection	Owner
A6.4.3/02 Submitted with the PT8 BPD dossier	Anonymous	1995	Plasma, Erythrocyte and Brain Cholinesterase Background Data Source: Not applicable Report No.: Not indicated Not GLP; (unpublished) Doc. No.: 535-002	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	ARCH Chemicals
A6.6.1/01 Submitted with the PT8 BPD dossier	Herbold, B.	2001	Preventol MP 100 - Salmonella/Microsome test plate incorporation and preincubation method Source: Bayer AG, Leverkusen, Germany Report No.: PH 30864 T0069537 GLP; (unpublished) Doc. No.: 557-008	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	LANXESS Deutschland GmbH
A6.6.2/01 Submitted with the PT8 BPD dossier	XXXX	2001	Preventol MP 100 - In vitro chromosome aberration test with chinese hamster V79 cells Source: XXXX Report No.: XXXX 30824 T1069538 GLP; (unpublished) Doc. No.: 557-007	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	LANXESS Deutschland GmbH
A6.6.3/01 Submitted with the PT8 BPD dossier	XXXX	2001	Preventol MP 100 - V79/HPRT-Test in vitro for the detection of induced forward mutations Source: XXXX Report No.: XXXX 31132 T2069539 GLP; (unpublished) Doc. No.: 557-009	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	LANXESS Deutschland GmbH
A6.6.4/01 Submitted with the PT8 BPD dossier	XXXX	1993	Omacide IPBC - Micronucleus cytogenetic assay in mice Source: XXXX Report No.: XXXX 727.122 GLP; (unpublished) Doc. No.: 557-005	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	ARCH Chemicals
A6.7/01 Submitted with the PT8 BPD dossier	XXXX	1989	3-iodo-2-propynyl butyl carbamate (IPBC) 104 week dietary carcinogenicity study in rats (Volume 1 and 2) Source: XXXX Report No.: TC-0411 435580 GLP; (unpublished) Doc. No.: 537-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation

Section No./ Reference No.	Author(s)	Year	Title Source (laboratory) Report No. GLP; (un)published Doc. No.	Data protection	Owner
A6.7/02 Submitted with the PT8 BPD dossier	XXXX	1988	3-iodo-2-propynyl butyl carbamate (IPBC) chronic dietary toxicity study in rats Source: XXXX Report No.: 5261 XXXX 435580 TC1417 GLP; (unpublished) Doc. No.: 537-002	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A6.7/03 Submitted with the PT8 BPD dossier	XXXX	1995	Review and interpretation of selected thyroid and forestomach lesions in the carcinogenicity study of 3-iodo-2-propynyl butyl carbamate (IPBC) in sprague-dawley rats Source: XXXX Report No.: TC-0476 Not GLP; (unpublished) Doc. No.: 581-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A6.7/04 Submitted with the PT8 BPD dossier	XXXX	1989	IPBC 78 week dietary carcinogenicity study in mice Volume 1 to 3 (803 pages) Source: XXXX Report No.: TC-0409 7304 436165 GLP; (unpublished) Doc. No.: 555-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A6.7/05 Submitted with the PT8 BPD dossier	XXXX	1989	IPBC 78 week dietary carcinogenicity study in mice Volume 2 to 3 (803 pages) Source: XXXX Report No.: TC-0409 XXXX 7304 GLP; (unpublished) Doc. No.: 555-002	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A6.7/06 Submitted with the PT8 BPD dossier	XXXX	1989	IPBC 78 week dietary carcinogenicity study in mice Volume 2 continued to 3 (803 pages) Source: XXXX Report No.: TC-0409 XXXX 7304 GLP; (unpublished) Doc. No.: 555-003	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A6.7/07 Submitted with the PT8 BPD dossier	XXXX	1989	IPBC 78 week dietary carcinogenicity study in mice Volume 3 to 3 (803 pages) Source: XXXX Report No.: TC-0409 XXXX 7304 GLP; (unpublished) Doc. No.: 555-004	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation

Section No./ Reference No.	**Author(s)**	**Year**	**Title Source (laboratory) Report No. GLP; (un)published Doc. No.**	**Data protection**	**Owner**
A6.7/08 Submitted with the PT8 BPD dossier	XXXX	1995	Pathology working group (PWG) report on the 78-week dietary carcinogenicity study of 3-iodo-2-propynyl butyl carbamate (IPBC) in cd-1-mice Source: Not indicated Report No.: TC-0458 275-003 GLP; (unpublished) Doc. No.: 555-005	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A6.7/09 Submitted with the PT8 BPD dossier	XXXX	1988	Results of dietary analysis for IPBC for the 78 week study in mice Source: XXXX Report No.: 436165 336802 GLP; (unpublished) Doc. No.: 437-002	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A6.8.1/01 Submitted with the PT8 BPD dossier	XXXX	1994	Omacide IPBC - Oral (Gavage) rabbit developmental toxicity dose ranging study Source: XXXX Report No.: XXXX /20/R GLP; (unpublished) Doc. No.: 551-007	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	ARCH Chemicals
A6.8.1/02 Submitted with the PT8 BPD dossier	XXXX	1994	Omacide IPBC - Oral (Gavage) rabbit developmental toxicity study Source: XXXX Report No.: XXXX /26/R GLP; (unpublished) Doc. No.: 551-006	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	ARCH Chemicals
A6.8.1/03 Submitted with the PT8 BPD dossier	XXXX	1994	Omacide IPBC - Oral (Gavage) rat development toxicity dose ranging study Source: XXXX Report No.: XXXX /18/R GLP; (unpublished) Doc. No.: 551-009	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	ARCH Chemicals
A6.8.1/04 Submitted with the PT8 BPD dossier	XXXX	1994	Omacide IPBC - Oral (Gavage) rat development toxicity (Teratogenicity) study Source: XXXX Report No.: XXXX /19/R GLP; (unpublished) Doc. No.: 551-008	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	ARCH Chemicals

Section No./ Reference No.	Author(s)	Year	Title Source (laboratory) Report No. GLP; (un)published Doc. No.	Data protection	Owner
A6.8.2/01 Submitted with the PT8 BPD dossier	XXXX	1996	Omacide IPBC - Oral (Gavage) rat one generation (expanded to two generation) reproductive toxicity study (3 Volumes) Source: XXXX Report No.: XXXX /28/R GLP; (unpublished) Doc. No.: 553-003	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	ARCH Chemicals
A6.8.2/02 Submitted with the PT8 BPD dossier	XXXX	2003	Historical control data - Reprotoxicity study in rats (Background Pregnancy Data from Multigeneration, Fertility and Pre- and Post Natal Studies on the Sprague-Dawley rat Source: XXXX Report No.: Not indicated Not GLP; (unpublished) Doc. No.: 553-005	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	ARCH Chemicals
A6.8.2/03 Submitted with the PT8 BPD dossier	XXXX	1986	TROYSAN Polyphase - Preliminary study for a two generation oral reproduction study in the male sprague dawley rat Source: XXXX Report No.: TC-0126 547-511/2 GLP; (unpublished) Doc. No.: 553-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A6.8.2/04 Submitted with the PT8 BPD dossier	XXXX	1986	TROYSAN Polyphase preliminary study for a two generation oral reproduction study in the female Sprague Dawley Rat Source: XXXX Report No.: 546-511/1 TC1390 GLP; (unpublished) Doc. No.: 553-004	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A6.8.2/05 Submitted with the PT8 BPD dossier	XXXX	1987	TROYSAN Polyphase two generation oral (dietary administration) reproduction toxicity study in the rat (one litter per generation) Source: XXXX Report No.: TC-0128 548-511/3 GLP; (unpublished) Doc. No.: 553-002	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A6.8.2/06 Submitted with the PT8 BPD dossier	XXXX	2004	Historical control data of two/one generation oral (Dietary Administration) reproduction toxicity studies 1984 to 1990 Source: XXXX Report No.: Not indicated Not GLP; (unpublished) Doc. No.: 553-006	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation

Section No./ Reference No.	Author(s)	Year	Title Source (laboratory) Report No. GLP; (un)published Doc. No.	Data protection	Owner
A6.8.2/07 Submitted with the PT8 BPD dossier	Shaw, D.	2004	To whom it may concern - IPBC purity Source: Troy Corporation, USA Report No.: Not applicable Not GLP; (unpublished) Doc. No.: 593-003	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A6.9/01 Submitted with the PT8 BPD dossier	XXXX	2002	Acute oral dose range-finding study with 3-iodopropynylbutyl carbamate (IPBC) administered by Gavage in CD rats Source: XXXX Report No.: 7071-100 TC-1414 GLP; (unpublished) Doc. No.: 541-004	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation ARCH Chemicals
A6.9/02 Submitted with the PT8 BPD dossier	XXXX	2001	Acute oral neurotoxicity study with 3-iodopropynylbutyl carbamate (IPBC) administered by gavage in CD rats - Volume 1 of 3 Source: XXXX Report No.: 7071-101 TC-1059 GLP; (unpublished) Doc. No.: 541-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	ARCH Chemicals TROY Corporation
A6.9/03 Submitted with the PT8 BPD dossier	XXXX	2001	Acute oral neurotoxicity study with 3-iodopropynylbutyl carbamate (IPBC) administered by gavage in CD rats - Volume 2 of 3 Source: XXXX Report No.: 7071-101 TC-1059 GLP; (unpublished) Doc. No.: 541-002	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	ARCH Chemicals TROY Corporation
A6.9/04 Submitted with the PT8 BPD dossier	XXXX	2001	Acute oral neurotoxicity study with 3-iodopropynylbutyl carbamate (IPBC) administered by gavage in CD rats - Volume 3 of 3 Source: XXXX Report No.: 7071-101 TC-1059 GLP; (unpublished) Doc. No.: 541-003	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	ARCH Chemicals TROY Corporation
A6.9/05 Submitted with the PT8 BPD dossier	XXXX	2002	2-week dietary range-finding and palatability study with 3-iodopropynylbutyl carbamate (IPBC) in CD rats Source: XXXX Report No.: 7071-102 TC 1415 GLP; (unpublished) Doc. No.: 542-005	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation ARCH Chemicals

Section No./ Reference No.	Author(s)	Year	Title Source (laboratory) Report No. GLP; (un)published Doc. No.	Data protection	Owner
A6.9/06 Submitted with the PT8 BPD dossier	XXXX	2001	13-week dietary neurotoxicity study with 3-iodopropynylbutyl carbamate (IPBC) in CD rats Volume 1 of 4 Source: XXXX Report No.: 7071-103 TC-1060 GLP; (unpublished) Doc. No.: 542-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	ARCH Chemicals TROY Corporation
A6.9/07 Submitted with the PT8 BPD dossier	XXXX	2001	13-week dietary neurotoxicity study with 3-iodopropynylbutyl carbamate (IPBC) in CD rats Volume 2 of 4 Source: XXXX Report No.: 7071-103 TC-1060 GLP; (unpublished) Doc. No.: 542-002	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	ARCH Chemicals TROY Corporation
A6.9/08 Submitted with the PT8 BPD dossier	XXXX	2001	13-week dietary neurotoxicity study with 3-iodopropynylbutyl carbamate (IPBC) in CD rats Volume 3 of 4 Source: XXXX Report No.: 7071-103 TC-1060 GLP; (unpublished) Doc. No.: 542-003	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation ARCH Chemicals
A6.9/09 Submitted with the PT8 BPD dossier	XXXX	2001	13-week dietary neurotoxicity study with 3-iodopropynylbutyl carbamate (IPBC) in CD rats Volume 4 of 4 Source: XXXX Report No.: 7071-103 TC-1060 GLP; (unpublished) Doc. No.: 542-004	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	ARCH Chemicals TROY Corporation
A6.9/10 Submitted with the PT8 BPD dossier	XXXX	1996	Acute Neurotoxicity Validation Study with Paraoxon in Rats Source: XXXX Report No.: XXXX 2100-004 Not GLP; (unpublished) Doc. No.: 541-007	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation ARCH Chemicals
A6.9/11 Submitted with the PT8 BPD dossier	XXXX	1996	Neurotoxicity Validation Study with Acrylamide in Rats Source: XXXX Report No.: XXXX 2100-030 Not GLP; (unpublished) Doc. No.: 541-008	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation ARCH Chemicals

Section No./ Reference No.	Author(s)	Year	Title Source (laboratory) Report No. GLP; (un)published Doc. No.	Data protection	Owner
A6.11/01 Submitted with the PT8 BPD dossier	XXXX	1988	Polyphase cholinesterase inhibition study in rats Source: XXXX Report No.: TC-0122 638784 5165 GLP; (unpublished) Doc. No.: 541-006	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A6.12.1/01 Submitted with the PT8 BPD dossier	XXXX	2003	ARCH letter to SCC - Health data (Cholinesterase levels - Rocherster) Source: XXXX Report No.: Not indicated Not GLP; (unpublished) Doc. No.: 574-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	ARCH Chemicals
A6.12.1/02 Submitted with the PT8 BPD dossier	Anonymous	2001	Medical surveillance program - Carbamates - IPBC Source: XXXX Report No.: 5.13 Not GLP; (unpublished) Doc. No.: 574-002	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	ARCH Chemicals
A6.12.3/01 Submitted with the PT8 BPD dossier	Ulfvarson, U. Alexandersson, R. Dahlqvist, M. Ekholm, U. Bergström, B. Scullman, J.	1992	Temporary health effects from exposure to water-borne paints Source: Scand J Work Environ Health 1992;18:376-87 Report No.: Not applicable Not GLP; (published) Doc. No.: 592-013	No	N.R.
A6.12.5/01 Submitted with the PT8 BPD dossier	Anonymous	2003	Material safety data sheet - Omacide IPBC 100 (According to 91/155 EC) Source: Arch Chemicals B. V. Swords / Ireland Report No.: Not applicable Not GLP; (unpublished) Doc. No.: 953-007	No	ARCH Chemicals
A6.12.6/01 Submitted with the PT8 BPD dossier	Bryld, L.E. Agner, R. Rastogi, S.C.	1997	Iodopropynyl butylcarbamate: a new contact allergen Source: Contact Dermatitis vol. 36, pp. 156-158, 1997 Report No.: Not applicable Not GLP; (published) Doc. No.: 592-003	No	N.R.

Section No./ Reference No.	Author(s)	Year	Title Source (laboratory) Report No. GLP; (un)published Doc. No.	Data protection	Owner
A6.12.6/02 Submitted with the PT8 BPD dossier	Pazzaglia, M. Tosti, A.	1999	Short Communications - Allergic contact dermatitis from 3-iodo-2-propynyl-butylcarbamate in a cosmetic cream Source: Contact Dermatitis, Vol. 41, pp. 290, 1999 Report No.: Not applicable Not GLP; (published) Doc. No.: 592-006	No	N.R.
A6.12.6/03 Submitted with the PT8 BPD dossier	Majoie, I.M. van Ginkel, J.W.	2000	The biocide iodopropynyl butylcarbamate (IPBC) as an allergen in cutting oils Source: Contact dermatitis, 2000, Vol. 43 p. 238 Report No.: Not applicable Not GLP; (published) Doc. No.: 592-007	No	N.R.
A6.12.6/04 Submitted with the PT8 BPD dossier	Bryld, L.E. Agner, T. Menné, T.	2001	Allergic contact dermatitis from 3-iodo-2-propynyl-butylcarbamate (IPBC) - an update Source: Contact dermatitis, 2001, Vol. 44, pp. 276-278 Report No.: Not applicable Not GLP; (published) Doc. No.: 592-009	No	N.R.
A6.12.6/05 Submitted with the PT8 BPD dossier	Schnuch, A. Geier, J. Brasch, J. Uter, W.	2001	The preservative iodopropynyl butylcarbamate: frequency of allergic reactions and diagnostic considerations Source: Contact Dermatitis 2002, 46, 153-156 Report No.: ISSN 0105-1873 Not GLP; (published) Doc. No.: 592-010	No	N.R.
A6.12.6/06 Submitted with the PT8 BPD dossier	Jensen, C.D. Thormann, J. Andersen, K.E.	2003	Airborne allergic contact dermatitis from 3-Iodo-2-Propynyl-Butylcarbamate at a paint factory Source: Contact dermatitis 2003, 48, 155-157 Report No.: ISSN 0105-1873 Not GLP; (published) Doc. No.: 592-011	No	N.R.
A6.12.6/07 Submitted with the PT8 BPD dossier	Brasch, J. Schnuch, A. Geier, J. Aberer, W. Uter, W.	2004	Contact Dermatitis and Allergy Iodopropynylbutyl carbamate 0-2% is suggested for patch testing of patients with eczema possibly related to preservatives Source: British Journal of Dermatology 2004, Vol. 151, page 608-615, © 2004 British Association of Dermatologists Report No.: Not applicable Not GLP; (published) Doc. No.: 592-017	No	N.R.

Section No./ Reference No.	Author(s)	Year	Title Source (laboratory) Report No. GLP; (un)published Doc. No.	Data protection	Owner
A7.1.1.1.1/01 Submitted with the PT8 BPD dossier	Jungheim	2001	Preventol MP 100 - Abiotic degradation Source: Bayer AG, Leverkusen, Germany Report No.: N 00/0070/05 LEV GLP; (unpublished) Doc. No.: 711-004	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	LANXESS Deutschland GmbH
A7.1.1.1.1/02 Submitted with the PT8 BPD dossier	Reynolds, J.L.	1994	Hydrolysis of 14C-3-iodo-2-propynyl-n-butylcarbamate (14C-IPBC) Source: Xenobiotioc Labs Report No.: XBL 94051 RPT00201 GLP; (unpublished) Doc. No.: 711-003	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	ARCH Chemicals
A7.1.1.1.2/01 Submitted with the PT8 BPD dossier	Lee, D.-H. Tsunoda, K. Takahashi, M.	1991	Photostability of organoiodine wood preservatives I. Progressive degradation and loss in fungal inhibition rate through photoirradiation Source: Mokuzai Gakkaishi, Vol. 37, No. 1, p. 76-81 (1991) Report No.: Vol. 37, No. 1 Not GLP; (published) Doc. No.: 792-005	No	N.R.
A7.1.1.1.2/02 Submitted with the PT8 BPD dossier	Lee, D.-H. Tsunoda, K. Takahashi, M.	1991	Photostability of organoiodine wood preservatives II. The photolytic process of preservatives Source: Mokuzai Gakkaishi, Vol. 37, No. 3, p. 261-265 (1991) Report No.: Vol. 37, No. 3 Not GLP; (published) Doc. No.: 792-004	No	N.R.
A7.1.1.1.2/03 Submitted with the PT8 BPD dossier	Phaff, R.	2005	AQUEOUS PHOTOLYSIS OF IPBC AND DETERMINATION OF THE QUANTUM YIELD Source: Research and Consulting Company, Itingen, Switzerland Report No.: 856160 GLP; (unpublished) Doc. No.: 712-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	IPBC Task Force (ARCH, DOW, ISP, LANXESS, TROY)
A7.1.1.2.1/01 Submitted with the PT8 BPD dossier	Grützner, I.	2002	Ready biodegradability of IPBC in a manometric respirometry test Source: Research and Consulting Company, Itingen, Switzerland Report No.: TC-1261 831172 GLP; (unpublished) Doc. No.: 713-002	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation

Section No./ Reference No.	Author(s)	Year	Title Source (laboratory) Report No. GLP; (un)published Doc. No.	Data protection	Owner
A7.1.1.2.2/01 Submitted with the PT8 BPD dossier	Seyfried, B.	2004	Inherent Biodegradability of IPBC in a modified "Zahn-Wellens /EMPA Test" Source: Research and Consulting Company, Itingen, Switzerland Report No.: 851399 GLP; (unpublished) Doc. No.: 713-007	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	IPBC Task Force (ARCH, DOW, LANXESS, TROY)
A7.1.2.2.2/01 Submitted with the PT8 BPD dossier	Blumhorst, M.R.	1992	Anaerobic aquatic metabolism study of P-100 Source: EPL Bio Analytical Services, USA Report No.: TC-0315 147-003 GLP; (unpublished) Doc. No.: 715-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A7.1.3/01 Submitted with the PT8 BPD dossier	Schneider, U.	2002	Estimation of the adsorption coefficient on soil and on sewage sludge using HPLC Source: Infracor Chemistry Services Report No.: AN-ASB 0203 GLP; (unpublished) Doc. No.: 731-003	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	DOW Benelux B.V.
A7.1.3/02 Submitted with the PT8 BPD dossier	Blumhorst, M.R.	1990	Adsorption/Desorption studies - batch equilibrium for P-100 Source: EPL Bio Analytical Services, USA Report No.: TC-0312 147-002 GLP; (unpublished) Doc. No.: 731-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A7.2.1/01 Submitted with the PT8 BPD dossier	Blumhorst, M.R.	1992	Aerobic soil metabolism study of P-100 Source: EPL Bio Analytical Services, USA Report No.: TC-0307 147-004 GLP; (unpublished) Doc. No.: 722-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A7.2.3.1/01 Submitted with the PT8 BPD dossier	Schimmelpfennig, H.	2004	Estimation of the Koc of the IPBC degradation product PBC using the PCKOC programm (v1.66) Source: Scientific Consulting Company, Wendelsheim, Germany Report No.: 824-006 Not GLP; (unpublished) Doc. No.: 731-004	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	IPBC Task Force (ARCH, DOW, ISP, LANXESS, TROY)

Section No./ Reference No.	Author(s)	Year	Title Source (laboratory) Report No. GLP; (un)published Doc. No.	Data protection	Owner
A7.3.1/01 Submitted with the PT6 BPD dossier	Görg, J. Glöckner, T.	2007	Estimation of the Atmospheric Residence Time of IPBC using the Atkinson Method - IPBC Source: Scientific Consulting Company, Wendelsheim, Germany Report No.: 824-014 743-002 Atkinson Not GLP; (unpublished) Doc. No.: 743-002	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	IPBC Task Force (ARCH, DOW, ISP, LANXESS, TROY)
A7.4.1.1/01 Submitted with the PT8 BPD dossier	XXXX	1994	Acute toxicity of Omacide IPBC to the fathead minnow (Pimephales promelas) Source: XXXX Report No.: 293- XXXX GLP; (unpublished) Doc. No.: 821-005	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	ARCH Chemicals
A7.4.1.1/02 Submitted with the PT8 BPD dossier	XXXX	1991	TROYSAN Polyphase P-100 - Acute toxicity to sheepshead minnow (Cyprinodon variegatus) under flow-through conditions Source: XXXX Report No.: TC-0299 91-10-3983 12166.0791.6103.505 GLP; (unpublished) Doc. No.: 821-003	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A7.4.1.1/03 Submitted with the PT8 BPD dossier	XXXX	1990	TROYSAN Polyphase P-100 - Acute toxicity to bluegill sunfish (Lepomis macrochirus) under flow-through conditions Source: XXXX Report No.: TC-0289 90-04-3300 12166.0789.6100.105 GLP; (unpublished) Doc. No.: 821-002	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A7.4.1.1/04 Submitted with the PT8 BPD dossier	XXXX	2001	Preventol MP 100 - Acute Fish Toxicity Source: XXXX Report No.: 1025 A/00 XXXX GLP; (unpublished) Doc. No.: 821-006	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	LANXESS Deutschland GmbH
A7.4.1.1/05 Submitted with the PT8 BPD dossier	XXXX	1994	Acute toxicity of Omacide IPBC to the rainbow trout, Oncorhynchus mykiss Source: XXXX Report No.: 294- XXXX GLP; (unpublished) Doc. No.: 821-004	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	ARCH Chemicals

Section No./ Reference No.	Author(s)	Year	Title Source (laboratory) Report No. GLP; (un)published Doc. No.	Data protection	Owner
A7.4.1.1/05b Submitted with the PT8 BPD dossier	XXXX	1990	TROYSAN Polyphase P-100 - Acute toxicity to rainbow trout (Oncorhynchus mykiss) under flow-through conditions Source: XXXX Report No.: TC-0290 90-03-3261 12166.0789.6100.108 GLP; (unpublished) Doc. No.: 821-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A7.4.1.1/06 Submitted with the PT8 BPD dossier	XXXX	1992	(Propargyl Butyl Carbamate) - Acute Toxicity to rainbow trout (Oncorhynchus mykiss) under flow-through condition Source: XXXX Report No.: TC-0305 XXXX No. 92-3-4146 12166.0991.6108.108 GLP; (unpublished) Doc. No.: 821-007	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A7.4.1.2/01 Submitted with the PT8 BPD dossier	Boeri, R.L. Magazu, J.P. Ward, T.J.	1994	Acute toxicity of Omacide IPBC to the daphnid, Daphnia magna Source: T.R. Wilbury Laboratory, Massachusetts Report No.: 292-OL GLP; (unpublished) Doc. No.: 822-002	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	ARCH Chemicals
A7.4.1.2/02 Submitted with the PT8 BPD dossier	Putt, A.E.	1992	(Propargyl Butyl Carbamate) - Acute Toxicity to daphnids (Daphnia magna) under flow-through conditions Source: Springborn Laboratories Massachusetts, USA Report No.: TC-0304 SLI No. 92-2-4122 12166.0991.6109.115 GLP; (unpublished) Doc. No.: 822-004	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A7.4.1.3/01 Submitted with the PT8 BPD dossier	Peither, A.	2001	Toxicity of Polyphase P-100 to Scenedesmus subspicatus in a 72-hour algal growth inhibition test – (Included the Analytical Report – Determination of the Concentrations of the test item in test medium) Source: Research and Consulting Company, Itingen, Switzerland Report No.: 790413 790424 TC0072 GLP; (unpublished) Doc. No.: 823-003	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation

Section No./ Reference No.	Author(s)	Year	Title Source (laboratory) Report No. GLP; (un)published Doc. No.	Data protection	Owner
A7.4.1.3/02 Submitted with the PT8 BPD dossier	Boeri, R.L. Magazu, J.P. Ward, T.J.	1994	Growth and reproduction test with Omacide IPBC and the freshwater alga, Selenastrum capricornutum Source: T.R. Wilbury Laboratory, Massachusetts Report No.: 295-OL GLP; (unpublished) Doc. No.: 823-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	ARCH Chemicals
A7.4.1.3/03 Submitted with the PT8 BPD dossier	Ward, T.J. Boeri, R.L. Magazu, J.P.	1997	Growth and Reproduction Toxicity test with Propargal Butyl Carbamate and the Freshwater Alga, Selenastrum capricornutum Source: T.R. Wilbury Laboratory, Massachusetts Report No.: TC0553 1115-TR GLP; (unpublished) Doc. No.: 823-004	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A7.4.1.4/01 Submitted with the PT8 BPD dossier	Müller	2000	Preventol MP 100 – Toxicity to bacteria Source: Bayer AG, Leverkusen, Germany Report No.: 1025 A/00 B GLP; (unpublished) Doc. No.: 842-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	LANXESS Deutschland GmbH
A7.4.1.4/02 Submitted with the PT8 BPD dossier	Mead, C.	2002	IPBC – Acute toxicity to bacteria (Pseudomonas putida) Source: Safepharm Laboratories Limited, Derby Report No.: 1597/006 GLP; (unpublished) Doc. No.: 842-003	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	ARCH Chemicals
A7.4.3.2/01 Submitted with the PT8 BPD dossier	XXXX	1992	TROYSAN Polyphase P-100 – Toxicity to fathead minnow (Pimephales promelas) embryos and larvae Source: XXXX Report No.: TC-0301 92-1-4057 12166.0791.6104.120 GLP; (unpublished) Doc. No.: 826-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation
A7.4.3.4/01 Submitted with the PT8 BPD dossier	Ward, G.S.	1991	TROYSAN Polyphase P-100 – Chronic toxicity to the water flea, Daphnia magna, under flow-through test conditions Source: Toxicon Environmental Sciences Report No.: TC-0294 J9009031b GLP; (unpublished) Doc. No.: 827-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation

Section No./ Reference No.	Author(s)	Year	Title Source (laboratory) Report No. GLP; (un)published Doc. No.	Data protection	Owner
A7.5.1.1/01 Submitted with the PT8 BPD dossier	Reis, K.-H.	2004	Effects of IPBC Technical on the Activity of the Soil Microflora in the Laboratory Source: Ibacon GmbH, Rossdorf, Germany Report No.: 17921080 GLP; (unpublished) Doc. No.: 841-002	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	IPBC Task Force (ARCH, DOW, LANXESS, TROY)
A7.5.1.2/01 Submitted with the PT8 BPD dossier	Lührs, U.	2004	Acute toxicity (14 Days) of IPBC technical to the earthworm Eisenia fetida in artificial soil Source: Ibacon GmbH, Rossdorf, Germany Report No.: 17922021 GLP; (unpublished) Doc. No.: 833-001	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	IPBC Task Force (ARCH, DOW. LANXESS, TROY)
A7.5.1.3/01 Submitted with the PT8 BPD dossier	Spatz, B.	2004	Effects of IPBC Technical on Terrestrial (Non-Target) Plants: Seedling Emergence and Seedling Growth Test Source: Ibacon GmbH, Rossdorf, Germany Report No.: 17923084 GLP; (unpublished) Doc. No.: 851-002	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	IPBC Task Force (ARCH, DOW, LANXESS, TROY)

The Doc IIIB reference list sorted by Section No:

Section No./ Reference No.	Author(s)	Year	Title Source (laboratory) Report No. GLP; (un)published Doc. No.	Data protection	Owner
B6.1.1 Submitted with the PT6 BPD dossier	Hernandez, O.	2001	Dipropylene glycol (mixture of isomers and Dominant Isomer CAS N°: 25265-71-8 & 110-98-5) Source: UNEP Publications, 2002, 1-84 Report No.: Not applicable Not GLP; (published) Doc. No.: 592-021	No	N.R.
B7.1.1/01 Submitted with the PT6 BPD dossier	Klamer, M.	2007	Field Leaching Study of IPBC from Painted Surfaces Exposed to Outdoor Conditions (Natural Rain) - Preliminary Results Source: Danish Technological Institute, Toastrup, Denmark Report No.: 1006657-17 180401 Not GLP; (unpublished) Doc. No.: 732-004	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	IPBC Task Force (ARCH, DOW, ISP. LANXESS, TROY)
B7.1.1/01 Submitted with the PT6 BPD dossier	Klamer, M.	2008	Field Leaching Study of IPBC from Painted Surfaces Exposed to Outdoor Conditions (Natural Rain) Source: Danish Technological Institute, Toastrup, Denmark Report No.: 1006657-17 Not GLP; (unpublished) Doc. No.: 732-005	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	IPBC Task Force (ARCH, DOW, ISP. LANXESS, TROY)
B7.1.1/01 Submitted with the PT6 BPD dossier	Klamer, M.	2008	1st Amendment to the report: Field Leaching Study of IPBC from Painted Surfaces Exposed to Outdoor Conditions (Natural Rain) - Preliminary Results Source: Danish Technological Institute, Toastrup, Denmark Report No.: 1006657-17 Not GLP; (unpublished) Doc. No.: 732-007	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	IPBC Task Force (ARCH, DOW, ISP. LANXESS, TROY)
B7.1.1/02 Submitted with the PT6 BPD dossier	Klamer, M. Morsing, N.	2006	Leaching of active ingredients from preservative treated wood in storage situations Source: Danish Technological Institute, Toastrup, The Netherlands Report No.: 1006657-02-22 Not GLP; (unpublished) Doc. No.: 732-003	Yes (Data on existing a.s. submitted for the first time for entry into Annex I.)	TROY Corporation