Protecting Aquatic and Riparian Areas from Pesticide Drift

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Abstract

Pesticide use has become integral to agricultural production around the world. Pesticides are known to move from treated agricultural areas into field margins, and subsequently into the broader environment, through transport in air and water. Although there are many routes of transport of pesticides, this paper will limit its discussion to spray drift. Air in field margins may be contaminated with pesticides because of application drift and post-application vapour loss. The impact of drift is greatest in field margins which, in intensively farmed areas, can be the primary source of habitat for wildlife. Sensitive plants and animals as well as the water quality of water bodies in field margins can be affected either directly or indirectly. Impacts due to spray drift can be mitigated by use of appropriately equipped and operated delivery systems (e.g., low boom heights, low-drift nozzles, favourable weather conditions), appropriate product choice (e.g., low vapour pressure, low water solubility), and through the use of buffer zones.

Introduction

In Canada, agricultural production occurs in all ten provinces but the extent to which the landscape has been permanently modified by agricultural activities in the various ecozones varies greatly. Depending upon the ecozone, agriculture may occupy from < 1 (Pacific Maritime ecozone) to 90% (Prairie ecozone) of the landscape (McRae et al. 2000). As the percentage of the landscape in agricultural production increases, field margins become more valuable as wildlife habitat.

Field margins refer to areas adjacent to agricultural fields or to cropped or grazed areas. In the Prairie ecozone, which has only 10% of the landscape available for wildlife habitat, field margins frequently consist of narrow strips of vegetation adjacent to fields such as fence lines or edges of road rights-of-way. In other situations, field margins may be treed areas unsuitable for cultivation or riparian areas along rivers or streams. Natural wetlands and constructed reservoirs may also be present in field margins. The vegetation growing within these areas may consist of native plant species, which can include low growing grasses, woody plants such as wild rose (*Rosa woodsii*), shrubs such as willow (*Salix spp.*) and tall trees such as aspen (*Populus tremuloides* Michx.). However, planted vegetation, such as trees in shelterbelts and grasses in waterways, may be present. Thus, field margins can vary greatly, not only in vegetation type, area and shape, but also in their ability to provide adequate habitat for wildlife. The presence of plants, insects and animals in field margins determines how farmers manage them to prevent their possible movement into adjacent crops.

Pesticides are known to move from treated agricultural and forested areas into the broader environment. Since many field margins are immediately adjacent to cropland treated with pesticides, greater interception of pesticides, due to off-target transport, and greater adverse effects on vegetation and wildlife within these areas would be expected relative to the broader environment. It is generally accepted that transport of pesticides into field margins, and subsequently into the broader environment, is associated with atmospheric transport and transport in water. Pesticide presence in air occurs by three main routes of entry. These include application drift, post-application vapour loss and wind erosion of treated soil (Figure 1). Once in the atmosphere either as spray droplets, vapour or sorbed to wind-eroded sediment, pesticides, or their photodegradation products, may be transported relatively short (field margins) or long (broader environment) distances before the removal processes of atmospheric wet (precipitation) and dry (particulate) deposition return them to the earth's surface.

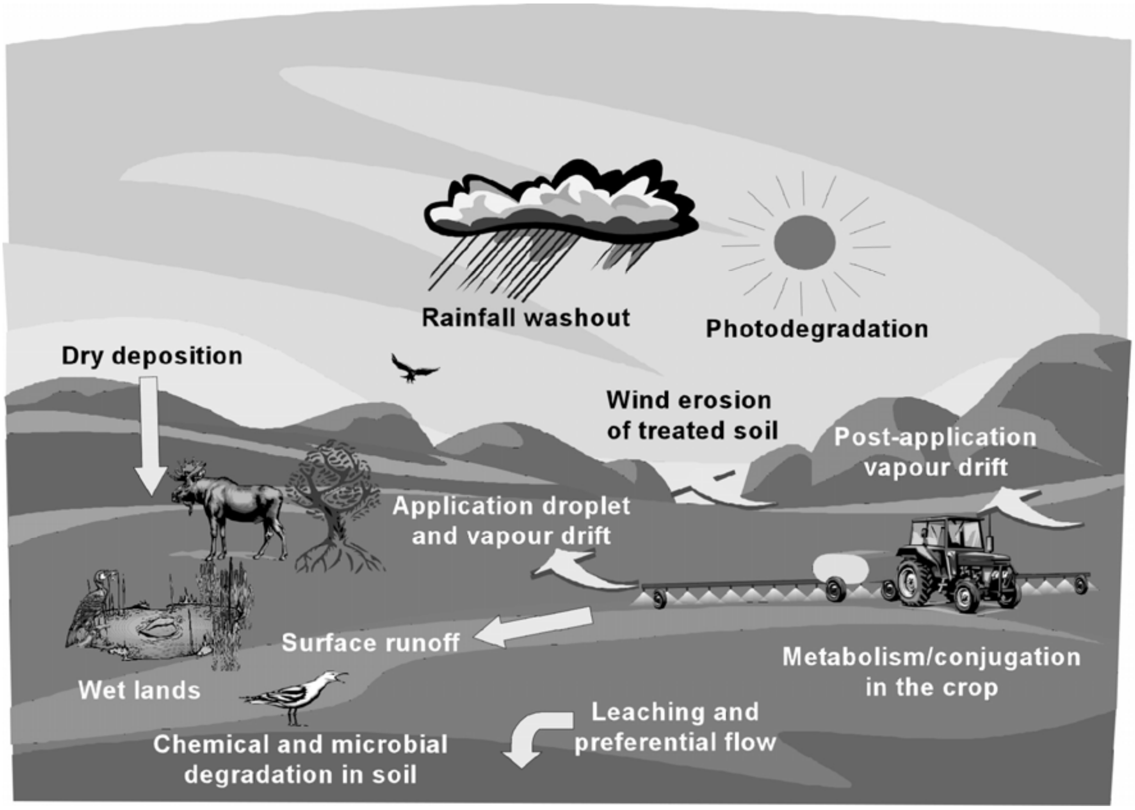

Figure 1. Routes of entry of pesticides into the atmosphere and into surface and ground waters and mechanisms of pesticide transformation in air, soil and plants.

Through atmospheric deposition, pesticides can deposit in field margins directly onto plant surfaces, wildlife, surface waters and the first few millimetres of soil. Deposition into surface waters must first pass through a surface film of organic material (Maguire and Tkacz 1988; Southwood et al. 1999) that is present in varying thickness depending on the water body and the season. Pesticides may then partition into the water column and, from there, into suspended or bottom sediments. The atmospheric dispersion and deposition of pesticides has been reviewed recently by Van Duk and Guicherit (1999).

Pesticides may also enter field margins in surface (irrigation, rainfall or snow melt) runoff and contaminate soil (by infiltration), plants (by plant uptake) and surface (receiving) waters (Figure 1). Pesticides entering a lake, wetland or constructed reservoir are those applied, or atmospherically deposited, within its watershed. Further transport is generally restricted to dispersion within the water body and partitioning into sediment. Surface runoff may traverse riparian areas and enter flowing water (river or stream). Then, not only is it possible for pesticides to be transported long distances, but a diversity of pesticides may be present due to multiple watersheds that can occur along the reach of a river or stream. Surface water bodies in field margins may also contain pesticides because of recharge with ground water contaminated through the leaching processes of matrix flow and/or preferential flow. Preferential flow occurs via preferential pathways such as cracks or fissures in the soil or continuous macropores consisting of insect burrows or cavities left by decayed plant roots.

The public is increasingly aware of the diffusion of pesticides into the environment and their potential to impact human health and environmental quality. Public interest in human health issues with respect to pesticide use is high and people remain concerned with air quality and the safety of their drinking water and food supply. Effects of pesticides on wildlife habitat and biodiversity are also important as evidenced by the popularity of wildlife- and fish-related activities, and strong commitment to the protection and conservation of abundant and diverse wildlife (Filion et al. 1993).

In this review, pesticide entry into the atmosphere as spray drift will be discussed as essential background information for understanding the movement of pesticides into field margins. Impacts of pesticide transport on vegetation and wildlife within field margins will then be discussed together with strategies to mitigate or minimize those impacts. Pesticides with currently registered uses in Canada will be emphasized in this review.

Entry of Pesticides into Air

Application Drift

The amount of spray drift leaving the treated swath depends primarily on the droplet size spectrum of the spray (Cross et al. 2001; Maybank et al. 1974), the height of droplet release (Nordby and Skuterud 1975), and weather conditions such as wind speed and atmospheric turbulence (Threadgill and Smith 1975). Under typical daytime conditions with winds between 10 and 25 km h^{-1}, Wolf et al. (1993) showed that about 95% (±4%) of fine to medium sprays deposited on the treated swath whereas the remaining 5% (±4%) moved off-swath in the direction of the wind.

Off-swath, the spray has been shown to be subject to several processes. First, the largest droplets deposited under the force of gravity with their size and evaporation rate, together with atmospheric turbulence, determining the distance they travelled prior to deposition (Bache and Johnstone 1992). The deposition profile has typically followed a linear pattern on a log-log scale (Wolf and Caldwell 2001). At 0 to 5 m downwind, fallout deposit amounts accounted for < 1% of the applied amount, whereas from 10 to 50 m, deposit amounts were typically < 0.1%. Some studies have shown that these amounts are dependent on atmospheric and topographic conditions. For example, deposition was suppressed in a turbulent atmosphere (Maybank and Yoshida 1969), channelled along low-lying areas (Allwine et al. 2002), or intercepted (filtered) by roughness elements along the soil surface (Miller et al. 2000).

Smaller droplets tend to remain air-borne, moving upward and downward with turbulent eddies, and become vertically mixed and diluted in the process (Bache and Johnstone 1992). The atmospheric loading that results from this latter process accounts for the majority of the pesticide loss from droplet drift (Majewsky 1991) that may then be engaged in long-distance transport prior to being removed from the atmosphere through the processes of precipitation washout (Hill et al. 2002) and dry deposition (Waite et al. 1999). Vertical structures, such as shrubs and trees, may intercept drifting droplets; with larger amounts being intercepted the nearer such structures are to the application swath (Raupach et al. 2001). Though some authors have suggested the use of shelterbelts or hedgerows as collectors to reduce airborne drift concentrations (Ucar and Hall 2001), others have proposed to use buffer zones or conservation headlands to protect these same hedgerows from exposure to drift (Longley et al. 1997).

Research clearly indicates that the amount of pesticide present in air and depositing on soil, plant and water surfaces within field margins depends, in large measure, on emitted droplet size spectrum and other operational parameters, distance from the field margin, and local weather conditions. However, amounts deposited are difficult to predict due to the heterogeneous nature of natural areas and associated changes in turbulence, interception, and dispersion (Marrs et al. 1991b). As a result, conventional drift models may not accurately predict movement into field margins and biological effects may therefore need to be studied directly rather than inferred.

Post-Application Vapour Losses

Soil-incorporated pesticides, and those used as seed treatments, may move as vapours to the soil surface where they sorb to the upper few millimetres of dry surface soil. Greater vapour movement to the soil surface occurs with pesticides with larger Henry's Law constants [ratio of concentration in air to concentration in (soil) water]. Vapour loss to the atmosphere of pesticides sorbed to the soil surface may occur when the surface soil becomes moist by rain or heavy dew. The emission of pesticides from both soil and plant surfaces has been recently reviewed (Van den Berg et al. 1999).

Vapour losses of pesticides from soil and plant surfaces determine post-application concentrations of pesticides in air within field margins. The magnitude of post-application vapour losses from soil and plant surfaces is dependent upon several factors. Losses of pesticides from plant surfaces depend primarily on the vapour pressure of the pesticide, as well as the rate of uptake by the plant, the rate of photodegradation on plant surfaces, and atmospheric turbulence to move vapours away from plant surfaces. Vapour losses following postemergence application to wheat (*Triticum aestivum* L.) were of the order 2,4-D *iso*-octyl ester (Grover et al. 1985) > bromoxynil *n*-butyrate plus octanoate (1:1) (Grover et al. 1994) > diclofop-methyl (Smith et al. 1986) (Table 1).

Table 1. Post-application vapour losses of herbicides following postemergence application to wheat.

Herbicide	Percent Loss	Time Period	Reference
Diclofop-methyl	< 1	5 days	Smith et al. 1986
Bromoxynil butyrate	16	5 days	Grover et al. 1994
Bromoxynil octanoate	7	5 days	
2,4-D *iso*-octyl ester	21	7 days	Grover et al. 1985

Post-application vapour losses from the soil surface are determined by the physical-chemical properties of the pesticide (e.g., vapour pressure), the degree of incorporation or penetration of the pesticide into the soil, the extent of binding of the pesticide to soil components, the half-life of the pesticide in soil (i.e., its susceptibility to microbial degradation), the tillage system, and the environmental conditions, such as soil moisture and atmospheric turbulence above the soil surface, following application. For example, much greater vapour loss occurred when trifluralin was applied to moist soil with no incorporation compared to the same treatment on dry soil (Glotfelty et al. 1984) or when the herbicide was incorporated shortly after application (White et al. 1977; Grover et al. 1988) (Table 2). Using micro-climatological data, Harper et al. (1976) showed that when the soil surface was dry, trifluralin fluxes were controlled by the availability of soil surface water content and its effect on the adsorption of the herbicide to soil components. Cumulative seasonal losses of triallate were also dependent on soil surface moisture conditions (Grover et al. 1988) as were losses of unincorporated granular formulations of ethalfluralin, trifluralin and triallate over a 2-wk period (Smith et al. 1997). Losses of surface-applied atrazine and alachlor after 35 d were greater from conventionally tilled corn fields than from no-till fields (Wienhold and Gish 1994). Losses were further reduced when a starch-encapsulated formulation was used. Recently, Rice et al. (2002) reported pesticide vapour loss from freshly tilled soil for several surface-applied pesticides to be as follows: □endosulfan (29%), trifluralin (14%), chlorpyrifos (10%), metolachlor (6.5%), atrazine (3.6%) and □-endosulfan (2.5%). When applied by herbigation under flood-irrigation conditions, cumulative vapour loss of EPTC from irrigation water and wet soil was 74% after 52 h (Cliath et al. 1980).

Impacts of Pesticide Use on Air and Water Quality

Impacts on Air Quality

Once pesticides have entered the air through application drift and post-application vapour loss, they are subject to atmospheric transport. The extent to which they move in the atmosphere before depositing to the earth's surface through the processes of atmospheric deposition can vary greatly. Distances moved can be local (less than one to tens of kilometres), regional (hundreds of kilometres) or long-range (thousands of kilometres) in nature. This movement of pesticides in the atmosphere has been reviewed recently by Van Duk and Guicherit (1999).

Impacts on Field Margins

The impact of pesticides on organisms in boundary habitats is subject to considerable debate. It is clear from the previous discussion that pesticides are frequently found in non-target areas. However, there is a significant lack of information on the biological consequences of their presence in these areas. Biological effects of pesticides on selected indicator non-target organisms are documented, and

Table 2. Post-application vapour losses of herbicides following preemergence surface-applied, soil-incorporated and herbigation applications.

Herbicide	Percent Loss	Time Period	Reference
Preemergence, moist surface soil, not incorporated			
Trifluralin	50%	3 to 7 hours	Glotfelty et al. 1984
	90%	2 to 7 days	
Preemergence, dry surface soil, not incorporated			
Trifluralin	2 to 25%	50 hours	Glotfelty et al. 1984
Preemergence, granular, not incorporated			
Trifluralin	15%	14 days	Smith et al. 1997
Ethalfluralin	12%	14 days	
Triallate	19%	14 days	
Preemergence, no till, not incorporated			
Atrazine	4%	35 days	Wienhold and Gish 1994
Alachlor	9%	35 days	
Preemergence, conventional tillage, not incorporated			
Atrazine	9%	35 days	Wienhold and Gish 1994
Alachlor	14%	35 days	
Preemergence, incorporated			
Trifluralin	22%	120 days	White et al. 1977
Trifluralin	24%	67 days	Grover et al. 1988
Triallate	18%	67 days	
Surface-applied to freshly tilled soil			
α-Endosulfan	29%	21 days	Rice et al. 2002
Trifluralin	14%	21 days	
Chlorpyrifos	10%	21 days	
Metolachlor	6.5%	21 days	
Atrazine	3.6%	21 days	
β-Endosulfan	2.5%	21 days	
Flood irrigation, herbigation			
EPTC	74%	52 hours	Cliath et al. 1980

mitigation steps recommended, as part of risk assessments conducted during modern registration processes. Legislated mitigative measures may include limits on time, rate and frequency of application, buffer zone distances from sensitive areas, and prescribed weather and application methods. However, data from laboratory, microcosm or mesocosm studies on which mitigation steps are based may not be suitable for predicting longer-term biological impacts under field conditions. Many of these studies are more concerned with documenting a threshold acute dose of pesticide such as a No Observable Effects Concentration (NOEC) rather than focusing on low-level chronic, multi-species population, or indirect effects. Data on effects of mixtures of active ingredients or of active ingredients plus fertilizers, which may be synergistic, are also rare. As a result, emphasis must be placed on documenting long-term and interactive effects of pesticides with agronomic practices on field margins.

Plants

Simulated herbicide drift studies using sensitive crop species (Eberlein and Guttieri 1994; Fletcher et al. 1996; Wall 1996; Al-Khatib and Peterson 1999) and wetland (Boutin et al. 2000; Roshon et al. 1999) and terrestrial plants (Boutin et al. 2000) indicate that direct deposition of application drift may damage sensitive plants in field margins.

It is difficult to predict the effects of herbicide drift on natural plant communities in field margins because of high variability caused by diverse natural populations, herbicide selectivity, and weather conditions. Marrs et al. (1989, 1991a) used a series of bioassay experiments with native plant species to assess spray drift in relation to plant damage and yield. At 4 m downwind of the sprayer, all species showed significant changes in growth in response to at least one of five herbicides investigated. Effects varied from growth promotion and reduction to change in flowering performance. The authors suggested that although herbicides may not kill native plants, they can alter species balance and the aesthetic value of plant communities. Breeze et al. (1992) documented a wide range of sensitivities of 14 wild plant species, to four herbicides. They concluded that individual plant effects were minimal (<10% growth reduction) if buffer zones of 10 m were observed, except for glyphosate, where distances of up to 40 m were necessary for the most sensitive species. De Snoo and van der Poll (1999) found that herbicide drift (from a range of products used in commercial practice) did not affect boundary vegetation adjacent to sugarbeet (*Beta vulgaris* L.) and potato fields, but did reduce abundance and floristic value of plants adjacent to a winter wheat field. Perry et al. (1996) found that sublethal rates of glyphosate, as might be expected from spray drift, reduced the cover abundance of grassy species but not broadleaf species. However, Marrs et al. (1991b) found few lasting effects on five native plant species placed 4 m downwind from the application of glyphosate, mecoprop, or MCPA. Canopy structure of the surrounding vegetation influenced response, suggesting that simple drift deposit models may not adequately describe risk. Kleijn et al. (1997) found inconsistent effects of fluroxypyr application to native grasses and forbs, but documented some reduction in species richness and shifts in biomass production from forbs to grasses. Nitrogen fertilizer had a larger effect on these parameters than herbicide.

Herbicide effects are not as widely studied on aquatic plants as on terrestrial plants. Davies et al. (2003) studied exposure of *Glyceria maxima*, *Lagarosiphon major*, *Myriophyllum spicatum*, and *Lemna* sp. to sulfosulfuron. No effects were noted on plants exposed to 3.33 µg L^{-1} for 21 days, but some effects occurred at 3.33 and 10 µg L^{-1} after 70 days. The study concluded that such high concentrations are unlikely to persist for a long time, therefore aquatic plant effects were improbable. Conversely, Nystrom et al. (1999) found a wide range of sensitivities of 40 species of micro-algae to the sulfonylurea herbicides metsulfuron-methyl, chlorsulfuron, and tribenuron-methyl, with sensitivities ranging from the nM to µM range. Faber et al. (1998) documented severe negative effects of glufosinate-ammonium and bialaphos on zooplankton at concentrations simulating spray drift (250 µg L^{-1}). Effects included reductions in taxa abundance and total zooplankton numbers, and effects persisted for several months to a year following exposure.

Amphibians

The decline of amphibians has been of concern in recent years, and probable causes are being investigated. The causes of amphibian decline appear to be complex, and so far, few studies have directly linked pesticides. Fort et al. (1999) evaluated the effect of Minnesota pond water fractions on South African clawed frog (*Xenopus laevis*) development. They found that a complex mixture of both naturally occurring and synthetic compounds were primarily responsible for the observed deformities. The potency of several compounds was also enhanced by the pond water. Direct effects of chlorpyrifos on *X. laevis* development were found by Richards and Kendall (2003). Body length and mass were significantly lower after a 96-h exposure of embryos at concentrations of 1 and 100 µg L^{-1}, respectively. Based on actual concentrations of chlorpyrifos in U.S. waters, the authors ranked the probability of effects due to the insecticide to be moderate to low. Kiesecker (2002) suggested that while theories on amphibian decline fall into two broad categories (trematode infection or chemical contamination), these two may in fact be synergistically related. The study showed that although trematode infection was necessary for deformation, infection occurred more frequently in agricultural sites. In a study of anuran development in relation to agricultural activity, Bishop et al. (1999) reported higher abnormality rates in green frogs (*Rana clamitans melanota*), northern leopard frogs (*R. pipiens*), and American toads (*Bufo americanus americanus*) exposed to water within an agricultural watershed compared to adjacent protected wetland areas. Anuran density and species diversity were also lowest in the agricultural area. The authors could not identify a single cause for these effects, but pointed to evidence of wetlands drainage, channelization of rivers, roadway construction and nutrient and particulate loading in addition to pesticide inputs within the agricultural area. There were, at the time of publication, no reports in the literature showing lethal or sublethal effects of organophosphate insecticides on amphibians at the levels detected in this study. Therefore, the factors governing amphibian welfare in field margins are likely more complicated than simple exposure to specific agricultural chemicals.

Aquatic Invertebrates

Aquatic invertebrates are commonly studied as indicator species for pesticide effects on the food chain and overall ecosystem health. Acute toxic effects are frequently documented in laboratory, microcosm, and field studies (Douglas et al. 1993; Gälli et al. 1994; Schulz et al. 2002). On a field scale, effects can be pronounced. For example, Liess and Schulz (1999) reported that rainfall-induced runoff of insecticides from arable land to a stream resulted in the disappearance of eight of the eleven abundant macroinvertebrates, and reductions in abundance of the remaining three. However, nine species recovered within 11 months, some earlier. Similar results were found by Fairchild and Eidt (1993). In their study, fenitrothion applied by air resulted in a sharp (70 to 90%) decrease in insect emergence from ponds, with recovery between 6 and 12 weeks after spraying. Farmer et al. (1995) also reported high macroinvertebrate toxicity of pyrethroids, resulting in an increase in algal growth due to reduced feeding. Toxic effects of chlorpyrifos to aquatic invertebrates in microcosms occurred at 1 $\mu g\ L^{-1}$, but delivery of that dose from spray drift deposition depended on water-body size and drift potential of the application (Biever et al. 1994).

Terrestrial Invertebrates

Of the terrestrial arthropods, butterflies and honeybees have received the greatest attention because these species are generally attracted to field margins but have also suffered serious declines. In a recent review, Longley and Sotherton (1997a) identified a variety of factors that contribute to butterfly population size. These included inherent susceptibility to insecticides, removal of nectar sources and larval host plants, and species-dependent ecological factors determining their within-boundary behaviour and dispersal. de Snoo et al. (1998) documented a 2- to 3-fold decrease in the number of butterfly species in commercially sprayed versus unsprayed edges and ditch banks of winter wheat and potato fields. Differences in species diversity and abundance were also a factor of crop type and adjacent habitat – significantly fewer butterflies were found in the potato field margin compared to the winter wheat, and ditch banks were also favoured habitat compared to seed grass. Similar effects of crop type, pesticide, and adjacent habitat type were reported by Redderson (1994) for lauxaniid flies in field margins. Feber et al. (1996) showed that butterflies were most closely associated with the abundance of flowers of key nectar source species. Herbicides which affected flower abundance therefore reduced butterfly populations. Dover et al. (1990) showed that fields sprayed under a "Conservation Headlands" regime had higher butterfly abundance. Direct butterfly mortality due to the application of insecticides near field margins has been documented. Davis et al. (1991) found that *Pieris spp.* mortality (24 - 73%) occurred due to direct exposure to diflubenzuron drift as well as contact with *Alliaria petiolata* plants (10 - 90%) exposed to drift. Deltamethrin posed high levels of short term risk to *Pieris spp.* larvae exposed to spray drift into hedges at field margins (Cilgi and Jepson 1995). Davis and Williams (1990) suggested a buffer zone of up to 5 m for ground sprayers, and 40 m for aerial sprayers, for protection of non-target insects from selected insecticides.

Vertebrates

In an Ontario study involving 25 species of birds within an agricultural landscape, Boutin et al. (1999) reported that species generally were more significantly associated with field margins than with field interiors. It is therefore of concern that field margins may be contaminated with pesticides that originate from spray drift, runoff, leaching, or eroded soil. With the exception of the exposure of birds to granular formulations of insecticides (McLaughlin and Mineau 1995), such exposure seldom results in acute toxicity to birds and mammals. There may be sublethal effects from spray drift, as birds have low levels of esterases that hydrolyze carbamate and organochlorine insecticides. Therefore, effects of spray drift of such products on birds nesting in field margins should be documented. For example, Cordi et al. (1997) showed that primicarb and dimethoate spray drift reduced overall growth rate of nestling passerine birds in field margin hedges.

Studies on fish focus on larger bodies of water that may not necessarily comprise field margins. Nonetheless, important effects can be expected from the toxicological data. Many studies document reductions in cholinesterase activity in fish such as common carp (*Cyprinus carpio*) (Gruber and Munn 1998), and largemouth bass (*Micropterus salmoides*), bluegill sunfish (*Lepomis macrochirus*) and mosquitofish (*Gambusia affinis*) (Carr et al. 1997) in response to carbamate and organophosphate contaminated runoff from treated sites. However, chlorpyrifos has also been documented to disappear rapidly from aquatic systems, with an initial half-life of 1 to 6 days (Giddings et al. 1997). Toxicological effects of these products in flowing streams are poorly documented.

Mitigation Strategies

Atmospheric Transport

Reduction of particle spray drift has been the subject of much study. In summary, drift can be reduced by managing droplet size spectra of nozzles with spray pressure (Nordby and Skuterud 1975), low-drift nozzles (Grover et al. 1997; Wolf and Caldwell 2001), and adjuvants (Downer et al. 1995) as well as protecting spray from wind with lower boom heights (Nordby and Skuterud 1975), shrouds (Wolf et al. 1993) and air assistance (Cooke et al. 1990). Individually, these measures have been documented to reduce drift by 50 to 75% or more; therefore, a comprehensive approach can have significant effect on the magnitude of droplet transport. Drift can also be reduced by spraying under appropriate environmental (e.g., wind speed) conditions. The implications of droplet size on herbicide efficacy have been discussed elsewhere (e.g., Knoche et al. 1994; Wolf 2000).

Buffer Zones

Buffer zones are generally employed to mitigate impacts of pesticides on sensitive ecosystems as a consequence of application drift. Within this context, buffer zones are no-spray areas defined by the distance between the downwind point of direct pesticide application and the nearest boundary of a sensitive habitat. Factors governing the depth of the buffer zone are the toxicity of the pesticide (active ingredient) to non-target organisms, the characteristics of the adjacent sensitive habitat, meteorological conditions at the time of application and the type and operating conditions of the delivery system. Although these variables are complex and some are impossible to control, a large number of researchers have suggested or defined buffer zones for the protection of field margins. For example, Davis et al. (1993) suggested buffer zone distances of 12 to 24 m for the protection of *Pieris brassicae* from triazophos and cypermethrin, respectively, Marrs et al. (1993) suggested a 20-m distance for the protection of sensitive plants from glyphosate, and Marrs and Frost (1997) found 8-m setbacks to mitigate negative effects of glyphosate, mecoprop, and MCPA. de Snoo (1999) and de Snoo and de Wit (1998) interviewed farmers and found that field margins are often sprayed intensively to prevent invasion of crops by pests. Nonetheless, a 3-m setback from ditches was considered effective at mitigating 95% of the negative effects on these habitats, and a flexible approach to buffer zone distances was deemed to gain most farmer acceptance.

Vegetative Barriers

Plant barriers in the buffer zone can capture airborne spray drift, resulting in exposure to organisms within the barrier (Longley and Sotherton 1997b). However, these same barriers can also reduce drift deposits in sensitive areas by reducing effective wind velocities and capturing spray particles (Ucar and Hall 2001, Miller 1999). Wolf et al. (2003) documented 75 to 95% reductions in drift deposits up to 30 m downwind when setback distances were vegetated with grass, shrubs, or trees. Stephenson and co-workers (Brown and Stephenson 2001; Carter et al. 2000), utilizing 2.5-m high snow fencing to simulate uniform hedge structures of 50% and 25% porosities, studied the mitigation effects of buffer zones and hedgerows on spray drift deposition into simulated wetland environments. Under moderate wind conditions (i.e., 2 to 4 m s^{-1}) for a 10-m wide buffer zone with or without artificial hedgerows, drift deposition was largely confined to the buffer zone for all trials, with trace deposits (0.01% or less) detected in the wetland area 20 to 40 m downwind of the downwind edge of the spray swath under some conditions. They concluded that a 10-m buffer zone with mixed woody vegetation would effectively protect a wetland from spray drift under wind conditions normally acceptable for spraying (i.e., less than 4 m s^{-1} or 14.4 km h^{-1}). A dense hedge was a less effective barrier to drift deposition within the wetland area than a sparse hedge. Davis et al. (1994) have observed a similar situation where instead of attenuating the wind and removing airborne drift droplets, a dense hedgerow diverts the drift cloud up and over the hedge, resulting in higher deposits farther downwind. On the basis of the safety margins observed by Roshon et al. (1999), the use of vegetated buffer zones, coupled with low-drift application methods, have the potential to reduce drift deposition into sensitive habitat to non-significant levels.

Conclusions

This review has demonstrated that pesticides can move into non-target areas during and after their application to agricultural fields in measurable and biologically significant quantities. Because of their ability to capture spray drift and to minimize erosional and runoff processes, field margins play dual, but contrasting, roles in overall ecosystem health. The retention of pesticides by field margins helps to reduce the overall effect of pesticides in the larger environment but, at the same time, can also affect, either directly or indirectly, plants or animals within them. Both of these roles deserve consideration in understanding the long-term consequences of the agricultural use of pesticides.

Use of buffer zones has received much attention as a key strategy to lessen the effects on field margins by all four routes of pesticide transport. However, although a buffer zone may reduce pesticide inputs into field margins and other environmentally sensitive areas, it does not reduce the overall amount of pesticide lost from the agricultural site. Thus, use of buffer zones must be complemented by agricultural management practices that reduce initial inputs into the environment, especially to the atmosphere. From the spray drift, post-application vapour loss, and soil erosion perspectives, this would mean use of low-drift delivery technology, low vapour pressure pesticides and agronomic practices to minimize soil erosion, respectively. In the case of surface runoff, this would similarly include proper product selection combined with agronomic practices that minimize runoff and leaching. It is use of such management practices, either alone or in combination with buffer zones, that will help to ensure long-term ecosystem health with respect to agricultural pesticide use.

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