IMPACTS OF AGRICULTURAL HERBICIDE USE ON TERRESTRIAL WILDLIFE: A REVIEW WITH SPECIAL REFERENCE TO CANADA

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ABSTRACT

Changes in agriculture since World War II are briefly described. We review the existing literature to examine the extent to which birds and mammals living in terrestrial habitats have been affected by the increasing use of agricultural herbicides. Wildlife (including birds, mammals and their resources) are exposed to herbicides primarily through direct overspray, spray drift and vapour drift. Except for a few specific cases, vertebrate wildlife are unlikely to be exposed to herbicide levels that are acutely toxic. In contrast, herbicide use has induced changes in wildlife resources through a variety of effects on different taxonomic group species (e.g. plants, soil organisms, insects, other invertebrates). Coupled with field studies on birds and mammals, there is sufficient evidence to suggest that herbicide use has adversely affected wildlife, primarily through a variety of toxic effects on plants, and simplification in habitat composition, heterogeneity and interspersion. The significance of our findings to pesticide regulation, and the direction and need for research are discussed. In particular, toxicity testing guidelines for nontarget plant protection that have been developed in Canada need to be adopted to support pesticide registration. Large scale, long-term interdisciplinary research of different farming systems is needed, particularly in North America, to integrate and evaluate the agricultural and ecological costs and benefits of herbicide use (along with other pesticides), and to more fully understand the relationships among wildlife, crop production, noncrop habitats, pesticide use, the entire farming system, and the socio-economic costs to the farmer and public at large.

RÉSUMÉ

Les changements dans les pratiques agricoles survenus depuis la dernière guerre mondiale sont décrits brièvement. Plus particulièrement, nous présentons une revue de la littérature sur les effets de l'utilisation toujours croissante d'herbicides sur les oiseaux et mammifères généralement retrouvés dans les habitats terrestres associés aux milieux agricoles. La faune et les habitats sont exposés aux herbicides lors de l'épandage par contact direct et indirectement par la dérive d'herbicides sous forme de gouttelettes ou de vapeur. A l'exception de certains cas spécifiques, il est peu probable que la faune de milieux agricoles soit exposée à des doses atteignant des niveaux de toxicité aiguë. Par contre, l'utilisation d'herbicides a induit des changements au niveau des ressources utilisées par la faune et ce, à différents niveaux taxonomiques tels que plantes, organismes des sols, insectes et autres invertébrés. A l'instar des études de terrain sur les oiseaux et les mammifères, on a trouvé suffisamment d'évidences dans la littérature qui montrent que l'utilisation d'herbicides affecte négativement la faune principalement par des effets toxiques sur les plantes et par la réduction de la composition, l'hétérogénéité et la distribution des habitats fauniques. L'importance de ces résultats pour la réglementation des pesticides, de même que pour l'orientation que devrait prendre la recherche dans ce domaine est discutée. Plus particulièrement, les lignes directrices élaborées récemment au Canada pour évaluer les effets des pesticides chimiques sur les plantes devraient être adoptées pour l'homologation des pesticides. De plus, des études approfondies des différents types d'exploitation agricole, particulièrement en Amérique du Nord, sont nécessaires afin d'intégrer et d'évaluer les coûts et les bénifices de l'utilisation des herbicides (et des autres pesticides) tant au niveau écologique qu'agricole et afin de nous permettre de mieux comprendre les interactions entre la faune, les habitats fauniques en milieux agricoles, la production agricole, l'utilisation de pesticides et tous les coûts socio-économiques pour le fermier et le public en général.

INTRODUCTION

Since World War II, agriculture has been undergoing dramatic changes, including the introduction of synthetic chemicals, the use of larger and more sophisticated machinery, the development of high yield plant and animal varieties, a shift in market forces towards cash crops, and the emergence of fewer, larger and more specialised farms (McEwen and Stephenson, 1979; Pimentel and Perkins, 1980; Conacher and Conacher, 1986; O'Connor and Shrubb, 1986; Eijsackers and Quispel, 1988a). Associated with these changes has been a huge increase in the use of herbicides as a result of the development of new and specialized products, skilful marketing by chemical companies, active promotion or endorsement by agricultural researchers and extension officers, the development of conservation tillage methods and technology, and machinery capable of delivering herbicides more conveniently. Herbicides are now the most widely used pesticides in agriculture worldwide, both in terms of the volumes used and the areas treated (Conacher and Conacher, 1986). In this respect, compared with other pesticides, herbicides offer the greatest potential to contaminate and cause damage to the environment. North America is the main market for herbicides, followed by western Europe. Products for use on corn, small grains, soybeans, rice, sugarbeet and sugarcane account for 78% of the herbicide market worldwide (Schwinn, 1988).

As elsewhere, the nature and extent of agriculture in Canada has been changing. About 11% of Canada is considered suitable for agriculture. In 1986, farmland occupied 67.8 million hectares, a slight decline from its historical peak in 1951 (Table 1). While the total area under farm ownership in Canada has been stable since 1951, the relative importance of different crops has changed (Figure 1). From 1911 to 1986, cultivation of

Table 1. Statistics on farming and pesticide use in Canada (abstracted from Statistics Canada 1951-1986).

	1951	1961	1971	1976	1981	1986
Total number of farms	623 091	480 903	366 128	338 578	318 361	293 089
Size of farms (ha)	113	122	145	163	187	202
Total farming area (millions of ha)	70.4	69.8	68.6	68.4	65.8	67.8
Field Crops (% total farming area)	34.1	33.7	37.3	38.7	43.8	45.9
Herbicide use (millions of ha sprayed)*			8.5		15.2	22.9
Insecticide use (millions of ha sprayed)*			0.9		1.6	4.6

* Each ha was counted only once regardless of the number of applications

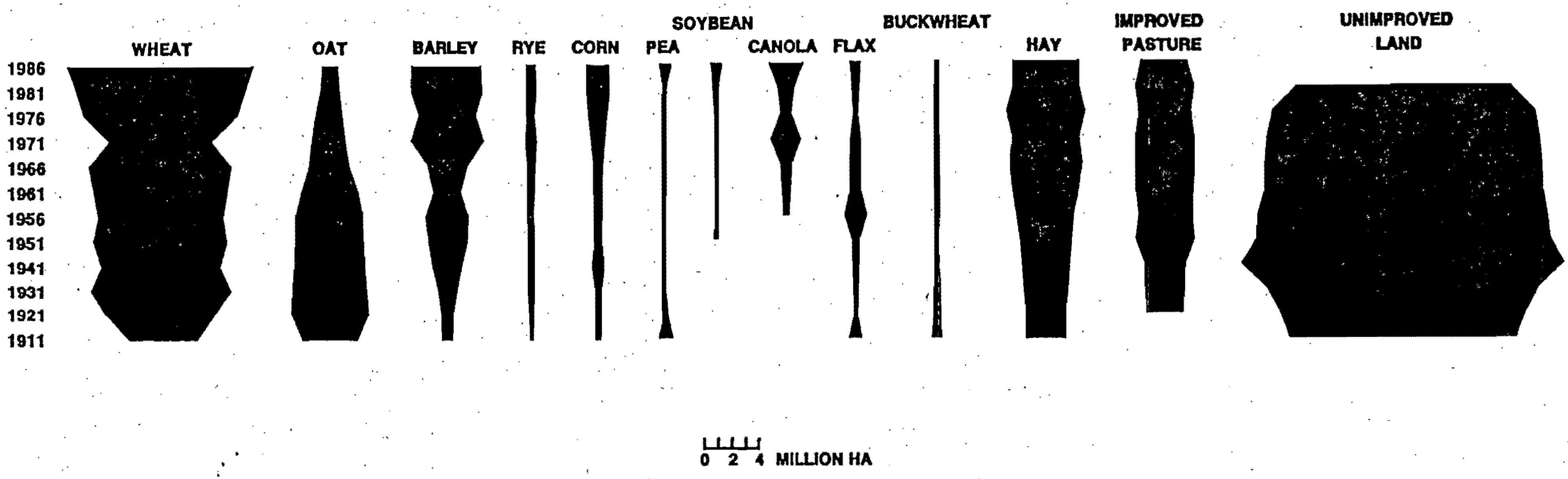

Figure 1. Changes in areas of major crops, hay, improved pasture and unimproved lands in Canadian farmland from 1911-1986. Improved pasture includes pasture or grazing lands which have been cultivated, drained, irrigated, fertilized, seeded, sprayed, etc. Unimproved lands includes areas of native pasture or hay land that have not been cultivated, brush pasture, grazing or waste lands, sloughs, marsh, rocky land, etc. Abstracted from Statistics Canada (1951-1986).

wheat, barley, corn, soybeans and canola increased substantially while oat and buckwheat decreased. Hay and "improved" pasture (defined in Canada as land used for pasture or grazing and which has been cultivated, drained, irrigated, fertilized, seeded, sprayed, or otherwise manipulated) also increased, while "unimproved" lands (defined as areas of native pasture or hay land that have not been cultivated, brush pasture, grazing or waste lands, sloughs, marsh and rocky land, etc.) decreased. From 1951-1986, the total number of farms in Canada decreased by over 50% while average farm size just about doubled (Table 1). Agricultural production has generally been rising through more intensive use of farmland, often at the expense of wildlife. For example, more than 80% of wetlands in agricultural regions of Canada have been lost through drainage, dyking and infilling, with dramatic adverse impacts on associated wildlife such as ducks (Boyd, 1985). Pesticide use has increased substantially since 1971 (Table 1). Herbicide use increased three-fold since 1971, with 22.9 million hectares of farmland (34% of total farmland) being treated in 1986. Insecticide use is much less extensive, but increased five-fold since 1971, with 4.6 million hectares (7% of total farmland) being treated in 1986. Insecticide use is, however, more unpredictable than herbicide use, due to the fluctuating nature of insect outbreaks, and 1986 was a year with a large outbreak of grasshoppers.

In this paper, we review the existing literature to examine the extent to which wildlife living in terrestrial habitats has been affected by the use of agricultural herbicides. Our review is extensive but not exhaustive. Compared to aquatic environments, our knowledge of exposure and ecological effects of toxic chemicals in the terrestrial environment is much more limited (Hurlbert, 1975; OECD, 1989). Firstly, we briefly review the environmental chemistry and fate of agricultural herbicides to establish the potential exposure of wildlife and other nontarget organisms in terrestrial, farmland habitats.

Secondly, we review herbicide impacts on wildlife from direct toxicity. We then focus on secondary impacts of agricultural herbicide use through adverse modification of wildlife resources by reviewing effects on different taxonomic groups (e.g. plants, soil organisms, other invertebrates, insects, etc.) and on habitat composition, heterogeneity and interspersion, and by reviewing field studies for wild mammals and birds. Lastly, we discuss the significance of our findings to pesticide regulation, and the direction and need for research.

EXPOSURE OF NONTARGET ORGANISMS

The environmental fate of herbicides is related to chemical and physical properties of the products, amount and frequency of use, methods of application, abiotic and biotic characteristics of the environment and meteorological conditions (Klingman *et al.*, 1982). At recommended rates of use in agriculture (now ranging from 4.5 g to 3 kg of active ingredient per hectare), the half-life of herbicides range from up to one month (e.g. 2,4-D) to 3 to 12 months (atrazine, trifluralin, metsulfuron methyl), to more than one year for picloram, pendimethalin, chlorsulfuron, and ethametsulfuron (Brown, 1978; Hassall, 1982; Grover and Bowes, 1981; Walker and Bond, 1977; Anderson and Barrett, 1985; Agriculture Canada, 1989; 1992). Tebuthiuron, a herbicide registered in 1974 in Canada for noncrop use, was estimated to persist between 2.9 to 11 years in soils of Arizona (Emmerich *et al.*, 1984; Johnsen and Morton, 1991). Under certain use conditions, some herbicides or their metabolites can remain active for longer times, for example, if applied in the fall or under anaerobic conditions such as those characteristic of aquatic habitats (Faust and Aly, 1964; Nicholaichuk and Grover, 1983). Residues can accumulate to toxic concentrations with consecutive treatments, and metabolites of herbicides such as atrazine can exhibit persistent and toxic properties (Woodwell, 1967).

Wildlife and other nontarget organisms can be exposed in varying degrees to agricultural herbicides or their breakdown products through direct overspray, from drift during and volatilization after application, by leaching from the soil, by runoff, by removal via water- or wind-eroded soil sediments, by contact with contaminated surfaces, inhalation of contaminated air, by consumption of contaminated food and water, and by ingestion through preening. Of the potential routes of exposure, direct overspray, spray drift and vapour drift are of particular concern (Elliott and Wilson, 1983). Noncrop habitats adjacent to fields may be sprayed directly with herbicides in attempts to eliminate potential reservoirs of unwanted plant species or their associated crop pests and pathogens (van Emden, 1965; Bendixen *et al.*, 1979; Best, 1983; Boatman, 1987; Heitefuss, 1988). In the Canadian prairies, an estimated 5-10% of herbicide application is performed aerially (Sheehan *et al.*, 1987). Aerial application can result in significant nontarget exposure due to overspray of noncrop habitats interspersed within or adjacent to sprayed areas, and to downwind drift of herbicide droplets or vapour (Holst, 1982; Sheehan *et al.*, 1987).

On average, less than 50% of products applied by air deposit on their target (Maybank *et al.*, 1978; Renne and Wolf, 1979; Grue *et al.*, 1986) although it can range between 28% to 100% of the application rate (Ware *et al.*, 1970). Although herbicides can drift long distances from the site of application, the normal range of drift is about 300 to 500 metres (Conacher and Conacher, 1986). Compared to aerial application, deposition of agricultural herbicides applied by ground equipment is usually more confined to the target area, particularly if nozzle type and orientation and hydraulic pressure are adequately set (Maybank *et al.*, 1978; Elliott and Wilson, 1983; Gohlich, 1983). Generally, spray drift is about 5-10% (up to 18%) of the active ingredient for a single swath with tractor-mounted, flat fan nozzles operated under typical herbicide application conditions (Norby and Skuterud, 1975; Bode *et al.*, 1976; Grover *et al.*, 1979; Sheehan *et al.*, 1987). For a 100 m wide field sprayed with similar ground equipment, ground deposit densities could reach extremes of about 1.0% of the applied dosage at 10m downwind from the field edge, 0.2% at 30m and 0.06% at 100m (Elliott and Wilson, 1983). Two herbicides widely used in Canada, triallate and trifluralin, have shown significant vapour drift (Grover, 1983). Thus, whether applied by air or from the ground, there is considerable potential for agricultural herbicides to affect wildlife and their habitats in Canadian farmland.

EFFECTS ON WILD BIRDS AND MAMMALS

1. DIRECT TOXICITY

Herbicides are generally much less acutely toxic to birds and mammals than other pesticides, particularly insecticides (Table 2) (cf. Pimentel, 1971; Brown, 1978; Hassall, 1982). However, some herbicides (e.g. MCPA, bromoxynil) do exhibit higher acute toxicity to birds and mammals than insecticides such as malathion. Except for paraquat, diquat and dinoseb, 5-day subacute dietary toxicities of herbicides, with death as the endpoint, are greater than 5,000 ppm for birds (Heath *et al.*, 1972; Hill and Camardese, 1986; USEPA, 1988). Based on laboratory testing with sheep, cattle, dogs and cats, wild mammals are thought to be at low risk from herbicides as a result of dietary exposure (Kimmins, 1975). Embryotoxicity has been observed when bird eggs are exposed in the laboratory (Hoffman and Albert, 1984). Herbicides such as paraquat, trifluralin, propanil, diclofop-methyl and a mixture of bromoxynil and MCPA were more embryotoxic than other herbicides and all the

Table 2. Toxicity of the 10 herbicides and insecticides most used in Canada based on a 1988 survey of Pesticide Registrants. Toxicity classes are from Bascietto (1985) and Farringer (1985). Toxicities from Worthing (1983, 1987) and Hudson et al. (1984).

Acute oral toxicity LD_{50} (mg/kg)	HERBICIDES	INSECTICIDES
	Avian toxicity	
Practically nontoxic (>2000)	Atrazine, difenzoquat, 2,4-D, diclofop-methyl, metolachlor, triallate, trifluralin, glyphosate	Carbaryl, benzene hexachloride (lindane)
Slightly toxic (500-2000)	MCPA	Malathion
Moderately toxic (51-500)		Azinphos-methyl, fenitrothion
Highly toxic (10-50)	Bromoxynil	Chlorpyrifos, fonofos, dimethoate, terbufos
Very highly toxic (<10)		Carbofuran
	Mammalian toxicity	
Practically nontoxic (>2000)	Metolachlor, trifluralin glyphosate	Malathion
Slightly toxic (500-2000)	Atrazine, triallate, diclofop-methyl, MCPA	Carbaryl, dimethoate, fenitrothion
Moderately toxic (51-500)	Bromoxynil, 2,4-D, difenzoquat	Chlorpyrifos, benzene hexachloride (lindane)
Highly toxic (10-50)		Azinphos-methyl, fonofos
Very highly toxic (<10)		Carbofuran, terbufos

insecticides tested. In a review of 134 avian reproduction studies done for the registration of 69 pesticides in Canada, six herbicides (of 22 herbicides) demonstrated effect at concentrations realistically encountered in the environment and among them, two herbicides exhibited developmental effects at levels lower than those giving rise to detectable parental toxicity (Mineau *et al.* 1994).

In general, the low acute toxicity and the lack of documented cases of mortality suggests that birds and mammals living in terrestrial habitats adjacent to fields treated with agricultural herbicides are unlikely to be exposed to levels that are acutely toxic. However, as will be outlined below, more needs to be known about potential effects from recurrent exposure to low levels of herbicides, potential effects on avian reproduction, and differential sensitivity of birds and mammals in relation to age, sex or size.

2. ADVERSE MODIFICATION OF WILDLIFE RESOURCES

Different taxonomic groups at a variety of trophic levels (e.g. primary producers, consumers, decomposers) play important roles in agroecosystems in soil stabilization and aeration, the recycling of organic matter and nutrients, the degradation of contaminants, and the provision of resources such as food, shelter and nesting sites for wild birds and mammals (cf. Pimentel and Edwards, 1982). The use of agricultural herbicides (and other pesticides) can interfere with these functions by altering plant biochemistry, developmental processes and morphology, changing population dynamics, species composition and diversity, interrupting energy and nutrient flows, degrading water quality, and changing the composition, heterogeneity and interspersion of habitats for wildlife. These disruptions may be exacerbated in agroecosystems already stressed by the effects of cultivation, chemical fertilisers, other contaminants, or severe climatic conditions (Conacher and Conacher, 1986).

We have summarized our review of the impacts of agricultural herbicide use through effects on resources used by wildlife as follows: (a) plants, (b) soil organisms, (c) other insects, invertebrates, etc., (d) mammals and (e) birds (Table 3). Each will now be dealt with in turn.

2.a) <u>Plants</u>

Use of agricultural herbicides can alter biochemical and developmental processes in plants, plant morphology, species abundance, composition and diversity, and the composition, heterogeneity and spatial interspersion of farmland habitats. As will be seen later on, these effects, to varying degrees, can impair organisms at other trophic levels (e.g., invertebrates, insects, mammals and birds). For example, plants are important to wildlife as shelter, nesting cover, and food resources. Many crop and noncrop plants are used directly by wildlife for food (Figure 2). Their importance varies among families, among genera and possibly among species. Weeds and crops in the Gramineae family are particularly important in the diet of many species. Habitat use by wildlife can also vary substantially among different crop and noncrop habitats (Freemark *et al.*, 1991).

2.a.i) Biochemical, developmental and morphological effects

By modifying biochemical and developmental processes (e.g., increasing levels of mineral salts, carbohydrates and moisture, changing protein and vitamin composition, delaying maturation), herbicides can increase plant susceptibility to pests and disease (Oka and Pimental, 1976; Heitefuss, 1988) and lower their suitability as animal food (Ries, 1976). Most of the work demonstrating these effects has been done with crop plants. As early as

Table 3. Summary of impacts of agricultural herbicide use on wildlife through effects on wildlife resources.

HABITAT(S)	EFFECT		REFERENCE
A. PLANTS			
Crop(C)/Noncrop (NC)	Mortality	NC:	Hanson 1952; Marrs et al. 1989.
	Biochemical (e.g., higher protein, carbohydrate or nitrate content; metabolic interference)	C:	Maxwell & Hardwood 1960; Ishii and Hirano 1963; Adams & Drew 1965; Oka & Pimentel 1976; Ries 1976; Ibenthal & Heitefuss 1979; Heitefuss 1988.
	Developmental (e.g., delayed seed germination, growth and maturation; decreased nodulation; bud/flower abortion)	C:	Fox 1948; Elliott & Wilson 1983; Oka & Pimentel 1976; Heitefuss 1988; Schnelle & Hensley 1990.
	Morphological (e.g., necrosis, epinasty, stem distortion, defoliation)	C:	Way 1964; Norby & Skuterud 1975; Grover et al. 1976; Elliott & Wilson 1983; Poster 1986.
		NC:	Hanson 1952; Conacher & Conacher 1986; Marrs et al. 1989.
	Shifts in species abundance/composition	C:	Frankton 1955; Fryer & Chancellor 1970; Rademacher et al. 1970; Day 1978; Harvey 1979; LeBaron & Gressel 1982; Frankton & Mulligan 1987; Hume 1987; Holt & LeBaron 1990; Primiani et al. 1990.
		NC:	Willis & Yemm 1966; Thilenius et al. 1974; Tomkins & Grant 1977.
	Species diversity - increased	C:	McCurdy & Molberg 1974; Tomkins & Grant 1977; Guthery et al. 1987;
		NC:	Tomkins & Grant 1977; Machlica & Shipman 1983.
	- decreased	C:	Tomkins & Grant 1977; Chancellor 1979; Guthery et al. 1987; Eijsackers & Quispel 1988b.
		NC:	Tomkins & Grant; Machlica & Shipman 1983; Boutin et al. 1994.

Table 3. Herbicide impacts (cont'd)

HABITAT(S)	EFFECT	REFERENCE
A. PLANTS (Cont'd)		
Farmland: North America (NA) United Kingdom (UK) Europe (EUR)	Shifts in crop types	NA: Fawcett 1982; Lovett 1982; Clark et al. 1986. UK: O'Connor & Shrubb 1986; Hurle 1988.
	Reduced diversity/interpersion of habitat types	NA: van Emden 1965; Alteiri & Letourneau 1982; Smutz 1987. UK: Potts & Vickerman 1974; O'Connor & Shrubb 1986. EUR: Baudry et al. 1988; Eijsackers & Quispel 1988b; Hanski & Tiainen 1988; Mader 1988.
	Reduction, degradation and/or isolation of noncrop habitats (e.g., hedgerows, fencerows, shelterbelts, woodlands, wetlands)	NA: Merriam 1978; Best 1983; Smutz 1987; Best et al. 1990. UK: Potts & Vickerman 1974; O'Connor 1984; O'Connor & Shrubb 1986; Boatman 1987. EUR: Baudry et al. 1988, van Dorp et al. 1988.
B. SOIL ORGANISMS		
Cropland *Grassland* *Conifer plantation* *Swards*	Acute Toxicity	Edwards 1984.
	Transitory or no change in abundance of microorganisms	Brown 1978; Smith 1982; Chakravarty & Chatarpaul 1990.
	Increased decomposition/degradation rate by microorganisms	Ilijin 1962; Smith et al. 1991; Smith & Aubin 1991.
	Invertebrate abundance	
	- increased/decreased	Fox 1964.
	- no change	Davis 1965.
	No effect on soil fertility	McCurdy & Molberg 1974; Fryer 1981b; Smith et al. 1991.

Table 3. Herbicide impacts (cont'd)

HABITAT(S)	EFFECT	REFERENCE
B. SOIL ORGANISMS (Cont'd)		
	Adverse effects from reduced food, availability, decreased plant cover, reduced plant species diversity.	Edwards & Thompson 1973; Way & Cammell 1981; House et al. 1987; House 1989.
C. OTHER INVERTEBRATES, ETC.		
Laboratory	Acute toxicity	Adams 1960; Hassan 1983.
Farmland	Delayed growth & development	Adams 1960.
	Species abundance - increased	Putman 1949; Maxwell & Hardwood 1960; Ishii & Hirano 1963; Adams & Drew 1965; Oka & Pimentel 1976; Klingauf 1988.
	- decreased	Dempster 1969; Fryer & Chancellor 1970; Lamp et al. 1984; Sotherton 1984, 1985; Ali & Reagan 1985.
	Reduced species diversity	Lewis 1969; Southwood & Cross 1969; Shelton & Edwards 1983.
	Reduced abundance/diversity of predatory insects	Altieri & Letourneau 1982; Mader 1988; Wratten & Thomas 1990.
D. SMALL MAMMALS		
Grassland *Pasture* *Rangeland* *Old Field*	Diet composition - no effect	Johnson 1964.
	- changed	Johnson 1964; Fagerstone et al. 1977.
	Decreased survival from diet shift or greater foraging dispersal	Keith et al. 1959; Tietjen et al. 1967; Hull 1971.

Table 3. Herbicide impacts (cont'd)

HABITAT(S)	EFFECT	REFERENCE
D. SMALL MAMMALS (Cont'd)		
	Lower reproductive success from diet shift	Spencer & Barrett 1980.
	Species abundance - no change	Johnson & Hansen 1969.
	- decreased	Keith et al. 1959; Tietjen et al. 1967; Johnson & Hansen 1969; Hull 1971; Lillywhite 1977; Borreco et al. 1979; Spencer & Barrett 1980.
	- increased	Johnson & Hansen 1969.
	Decreased species diversity	Lillywhite 1977.
Forests	Species abundance/composition - no effect	Sullivan & Sullivan 1982; Freedman et al. 1988; Santillo et al. 1989b.
	- shifts	Kirkland 1978; Savidge 1978; D'Anieri et al. 1987; Ritchie et al. 1987; Santillo et al. 1989b.
Farmland	Increased use of cultivated fields	Wegner & Merriam 1990.
	Lowered dispersal/local abundance/persistence from reduction, degradation and/or isolation of noncrop habitat (e.g., hedgerows, fencerows, shelterbelts, woodlands)	Fahrig & Merriam 1985; Johnson & Beck 1988; Merriam 1990; Merriam & Lanoue 1990.
E. LARGE MAMMALS		
Forests	Browse - no effect	Sullivan & Sullivan 1979.
	- increased	Krefting & Hansen 1969; Kufeld 1977; Thompson et al. 1991.

Table 3. Herbicide impacts (cont'd)

HABITAT(S)	EFFECT		REFERENCE
E. LARGE MAMMALS (Cont'd)			
	Browse - decreased		Kennedy & Jordan 1985.
	Habitat use - no change		Darr & Klebenow 1975.
F. BIRDS			
Farmland North America (NA) United Kingdom (UK) Europe (EUR)	Regional population decline from changes in plant species abundance/composition	UK:	Southwood & Cross 1969; Fryer & Chancellor 1970; Potts 1977, 1984, 1986; O'Connor & Shrubb 1986; Sotherton et al. 1988.
	Decreased chick survival from diet shift or greater foraging dispersal	NA: UK:	Warner 1984, Warner et al. 1984. Potts 1970, 1980; Sotherton 1982; Green 1984; Sotherton et al. 1985; Rands 1986.
	Higher abundance with greater diversity/interspersion of habitat types	NA: UK: EUR:	Warner 1984; Warner et al. 1984; Clark & Weatherhead 1986; Clark et al. 1986; Vander Haegen et al. 1989. O'Connor & Shrubb 1986. Robertson et al. 1990.
	Population declines with changes in crop types - Regional	NA: UK:	Clark et al., 1986; Clark & Weatherhead 1986. O'Connor & Shrubb 1986; Galbraith 1988; Inglis et al. 1990.
	- Local	NA:	Smutz 1987.
	Lowered dispersal/local abundance/diversity from reduction, degradation and/or isolation of noncrop habitat (e.g., hedgerows, fencerows, shelterbelts, woodlands, wetlands).	NA: UK:	Wegner & Merriam 1979; Sheehan et al. 1987; Freemark 1988; Johnson & Beck 1988; Merriam 1988; Best et al. 1990; O'Connor 1991; Freemark & Collins 1992; Villard et al. 1992; Boutin et al 1994. Arnold 1983; Osborne 1984; Rands 1985; O'Connor & Shrubb 1986; Lack 1987.

Table 3. Herbicide impacts (cont'd)

HABITAT(S)	EFFECT	REFERENCE
F. BIRDS (Cont'd)		
	Higher species diversity with greater diversity/interspersion of habitat types	EUR: Robertson et al. 1990.
Broadleaved herbs	Redistribution of nests	NA: Dwernichuk & Boag 1973.
Sagebrush/brushy field	Local abundance - decreased	NA: Best 1972; Schroeder & Sturges 1975; Wiens & Rotenberry 1985.
	- increased	NA: Wiens & Rotenberry 1985.
	- no change	NA: Best 1972, Beaver 1976.
	Shifts in species composition/diversity	NA: Wiens & Rotenberry 1985.
Forests	Local abundance - decreased	NA: Slagsvold 1977; Savidge 1978; Biggs & Walmsley 1988; Santillo et al. 1989a.
	- no change	NA: Freedman et al. 1988.

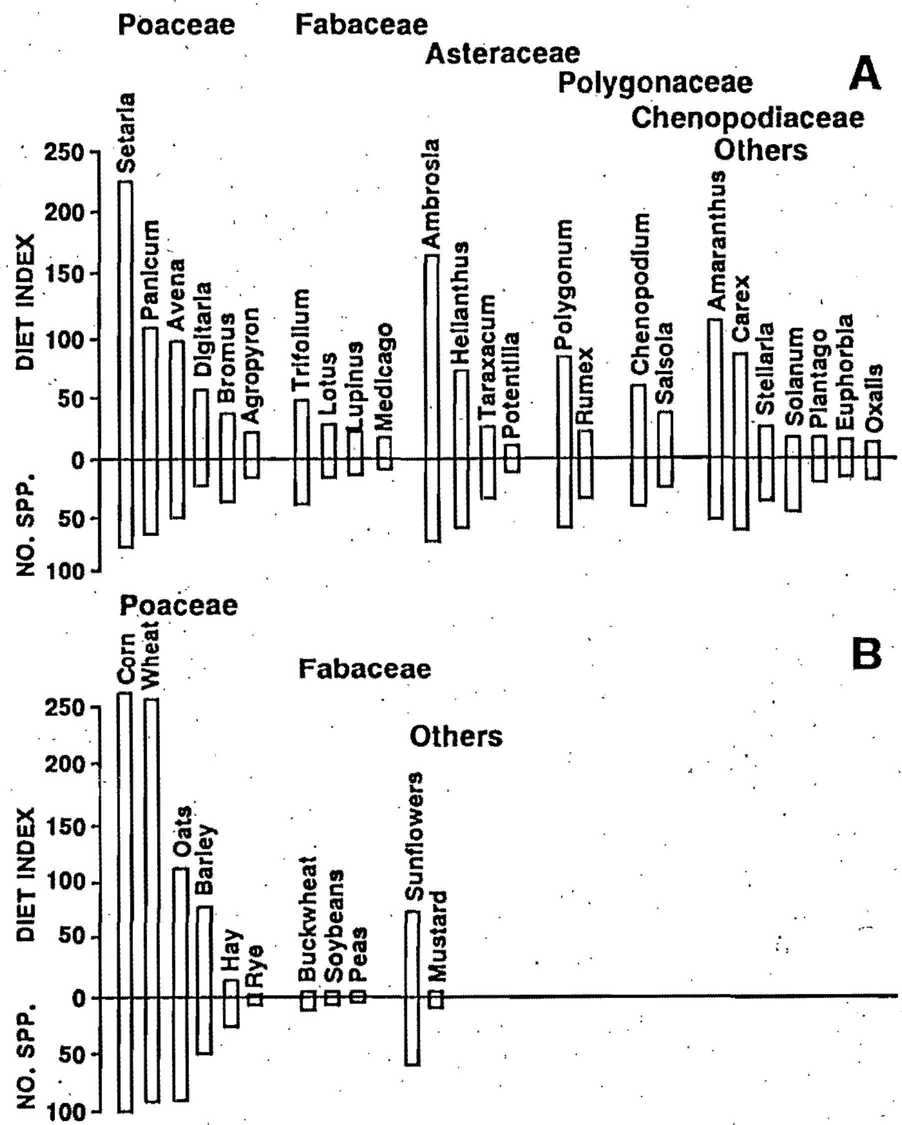

Figure 2. Importance of weed (A) and crop (B) species as food for wildlife (waterbirds, shorebirds, gamebirds, songbirds, birds of prey, fur and game mammals, small mammals, hoofed browsers, fish, amphibians, and reptiles). Adapted from Martin, Zim and Nelson 1951 as follows: The upper bars represent a diet index derived by aggregating stars (where * = 2-5% of the diet, ** = 5-10%, *** 10-25%, **** 25-50%, ***** = 50% or more) for each food item across consumer species; the lower bars indicate the number of consumer species.

1948, Fox (1948) showed that application of 2,4-D caused a delay in the germination of seeds and growth of wheat plants sufficient to favour an increase of wireworm damage. The incidence of mildew on spring wheat was enhanced by three different herbicides as a result of stress induced by metabolic interference (Ibenthal and Heitefuss, 1979). Wheat treated with 2,4-D was higher in protein content resulting in a proliferation of aphids (Adams and Drew, 1965). The concentration of nitrogenous compounds in crops was enhanced by 2,4-D, leading to an increase in pathogens and pests on corn (Oka and Pimentel, 1976), rice (Ishii and Hirano, 1963) and broad beans (Maxwell and Hardwood, 1960). An increase in nitrate content can render forage toxic to livestock (Ries, 1976). Nodulation and nitrogen fixation can be disrupted by herbicides resulting in adverse effects on growth and reproduction in crops such as dry bean (*Phaseolus vulgaris*), soybean (*Glycine max*), broad bean (*Vicia faba*), and peanut (*Arachis hypogea*) (Schnelle and Hensley, 1990 and references therein).

Efficacy of herbicides has, for obvious reasons, been well studied. Damage to plant development and morphology (e.g. necrosis, epinasty, inhibition of leaf expansion, stem distortion, bud/flower abortion, delayed flowering) at doses expected at the edge or downwind from fields sprayed with ground equipment, has been observed for a diversity of broad-leaved field crops exposed to the broad-leaved herbicides MCPA, 2,4-D and mecoprop (Elliott and Wilson, 1983; downwind deposition discussed above). One crop was damaged at 0.05% or less of the applied rates (damage expected 100m downwind), 4 at 0.4% or less (damage expected 30m downwind), 5 at 1% or less (damage expected 10m downwind), 13 at 2% or less, 16 at 5% or less, 18 at 10% or less, and 21 at 20% or less. Sensitivity to herbicide damage has been found to differ among herbicides and crops independently of plant morphology or taxonomic family (Way, 1964). Spray drift can be more toxic than direct spray. Barley plants were ten-times more damaged when exposed to the same rate of aminotriazole via spray drift versus a direct spray (Norby and Skuterud, 1975), perhaps due to better deposition on and uptake by plants of the small droplets characteristic of spray drift. Low doses characteristic of vapour drift can damage nearby crops and noncrop vegetation, as reported for clomazone in the U.S.A. (Poster, 1986), and are of concern for other herbicides (Ross *et al.*, 1990; Breeze and van Rensburg, 1991). During the 1966-1975 spraying seasons, air samples taken at sites in the Canadian prairies, not directly in the vicinity of sprayed fields, contained sufficient residues of 2,4-D butyl esters to damage susceptible plants (Grover *et al.*, 1976).

Assessment of herbicide damage to plants in wildlife habitats within and adjacent to treated fields has not been systematically addressed. Like crops, field margin species may also be affected by low doses of herbicides although few data are available. Widespread damage to farm trees and native vegetation from herbicide drift has been reported in Australia (Conacher and Conacher, 1986 and references therein). While the most common symptoms, namely yellowing and defoliation, are unlikely to be lethal, injured plants can be more susceptible to insect attack and disease (references cited in Conacher and Conacher, 1986) or suffer reproductive impairment (Fletcher *et al.*, 1993). Hanson (1952) investigated effects of aerial oversprays of the amine and ester formulations of 2,4-D on plant species associated with wetlands in North Dakota. Several dicotyledonous species were severely damaged including silverleaf potentilla *(Potentilla anserina)*, smartweed (*Polygonum coccineum*), water hemlock (*Circuta maculata*), blue lettuce (*Lactuca pulchella*), Rydberg's sunflower (*Helianthus rydbergii*) and hedge nettle (*Stachys palustris*). Monocotyledonous species were also killed or damaged by both the amine and the ester formulations, among them the dominant species, creeping spikerush (*Eleocharis palustris*), and two bulrush species (*Scirpus paludosus* and *S. acutus*). There was some seedling emergence of the dicot species four weeks after treatment but no sign of recovery for the monocots. More recently, Marrs *et al.* (1989) assessed effects of spray drift in relation to plant damage and yield for a range of plant species of conservation interest in Britain after applying each of six herbicides with a standard agricultural hydraulic ground sprayer. They observed lethal effects at two to six metres from the field edge and damage at greater distances, although damaged plants were able to recover within the growing season. Potential effects at the species level as a result of reduced competitive abilities of individual plants and subsequent shifts in dominance among species could not be assessed due to the short term of their experiment.

2.a.ii) Plant species abundance, composition and diversity

Farmers, agronomists and the agrochemical industry have constantly had to adapt to the emergence and expansion of weeds in farmland (Fisher, 1989). In the 1970's, proso millet (*Panicum miliaceum*), a species unknown a few years earlier, emerged as an important weed in corn and soybean (Harvey, 1979). More recently, nodding beggarticks (*Bidens cernua*) a common wetland species in Canada, has become a problem in farmland reclaimed from wetlands (D. Benoit, Agriculture Canada, Saint-Jean-sur-Richelieu, pers. comm). Frankton (1955) described 205 weed species important in Canada. Thirty years later, Frankton and Mulligan (1987) listed an additional 25 species, amongst them are serious weeds such as kochia (*Kochia scoparia*), scentless chamomille (*Matricaria maritima*) and absinth (*Artemisia absinthium*). Herbicides usually induce a simple species replacement, as one weed species is killed off another proliferates in its place, often from an entirely different family, and often posing an equal or greater problem (Hileman, 1982).

The few, available long-term studies to monitor changes in weed flora in cultivated fields in response to herbicides suggest that continual use causes shifts in weed species abundance and composition but does not eliminate weed problems (Fryer and Chancellor, 1970; Chancellor, 1979, 1985; Mahn and Helmecke, 1979; Mahn, 1984, 1988; Hurle, 1988). In a 36-year study in Saskatchewan, Hume (1987) found that 2,4-D reduced the abundance of susceptible species such as lamb's quarters (*Chenopodium album*) and stinkweed (*Thlapsi arvense*) while other, more-tolerant species increased such as wild buckwheat (*Polygonum convolvulus*), green foxtail (*Setaria viridis*) and nightshade (*Solanum trifolium*). In Britain, Rademacher *et al.* (1970) observed a variety of changes in weed species abundance over a 14-year period in relation to different weed control measures in cereals, including the use of 2,4-D, MCPA and DNOC. Changes in the weed flora detected in long-term monitoring studies cannot be solely attributed to herbicide use. The relative importance of a given weed species is also influenced by many other changes in agricultural practices such as longer or less diversified crop rotation, crop selection, increased mechanization, cultivation timing and method (e.g., no- or minimum till), field drainage, earlier planting, later harvesting, fertilization, use of commercial fertilizers rather than manure, and better seed cleaning methods (Beck, 1968; Fryer, 1981a; Haas and Streibig, 1982; Hanski and Tiainen, 1988; Hurle, 1988; Streibig, 1988). In addition, there are often logistical difficulties in taking frequent measurements over a long term (Cousens *et al.*, 1988).

Shifts in plant species abundance and composition in response to herbicides have been observed in short term studies. For example, a single application of 2,4-D changed the ratio of graminoid:forb above-ground biomass in a Wyoming alpine meadow from 3:7 to 8:3 for the next four years with no change in total biomass (Thilenius *et al.*, 1974). Willis and

Yemm (1966) found substantial changes in the frequency of different species within a sward sprayed with 2,4-D, maleic hydrazide or a combination of both. Not only did the frequency of many dicotylodenous species decrease as expected, but some grass species were also affected with tall oat-grass (*Arrhenatherum elatius*) declining and bluegrass (*Poa pratensis*) and fescue-grass (*Festuca rubra*) increasing. Tomkins and Grant (1977) studied the effects of six herbicides applied once on a recently disturbed and once on an old field over a three-year period. Numbers of monocot species proliferated in response to the application of auxin-mimic herbicides (picloram, 2,4-D and 2,4,5-T). In contrast, simazine caused a reduction in monocots. Diuron and paraquat reduced both monocots and dicots. Plant species diversity had not totally recovered three years after the application of diuron and increased after the use of paraquat and simazine. Fluctuations in numbers of individuals, numbers of species and species phenology were much less pronounced in the old field habitat than in the recently disturbed habitat. In southern Québec the vegetation in hedgerows and woodland edges where herbicides had been regularly used was compared with similar habitats where no herbicides had been used for at least six years (Boutin *et al.*, 1994). Significant reductions in species diversity and cover was observed where herbicide had been sprayed in fields adjacent to hedgerows and woodland edges.

Although shifts in vegetation are primarily ecological, species composition can also be altered as plants develop resistance to herbicides (Day, 1978). Triazine herbicides, in particular, have induced resistance in several weed species such as common groundsel (*Senecio vulgaris*), lamb's quarters (*Chenopodium album*) and wild oats (*Avena fatua*) (LeBaron and Gressel, 1982). Four weed species are known to have developed resistance to sulfonylurea herbicides such as chlorsulfuron and metsulfuron (Holt and LeBaron, 1990; Primiani *et al.*, 1990). The development of resistance is related to a species' morphological and anatomical characteristics (Hodgson, 1970), its life history characteristics such as delayed germination (Harper, 1956), and most importantly, to changes in physiological processes at the site of herbicide activity within the plant (Radosevich, 1977).

Herbicides may also alter the diversity of plant species. Use of herbicides in cultivated fields generally decreases plant species diversity (Chancellor, 1979). Guthery *et al.* (1987), in an experiment performed in the fall on sites that were cleared, levelled, tilled and then sprayed, found that 12 of 14 herbicides used reduced plant species diversity. Continuous application of 2,4-D and MCPA decreased total weed emergence over a 25-year period (1947-1972), but increased the number of weed species (McCurdy and Molberg, 1974), perhaps due to reduced competition from dominant weeds (Harper, 1957). In an 11-year study of old fields and five herbicides (bromacil, diuron, tandex, fenuron, picloram), Machlica and Shipman (1983) found that plant species diversity increased two years after spraying but after 10 years, patterns in species diversity varied with prevailing abiotic factors and successional stage prior to treatment.

2.a.iii) Habitat composition, heterogeneity, and interspersion

In addition to affecting plant species abundance, composition and diversity, herbicide use has also strongly affected the composition, heterogeneity and interspersion of farmland habitats. Herbicides have reduced the need for crop rotation for weed control and allowed farmers to specialize in monoculture and continuous cropping (Day, 1978; Hileman, 1982; Lovett, 1982; O'Connor and Shrubb, 1986; Hurle, 1988). Acreages of densely seeded or perennial crops such as oats and hay which aided in weed control have declined in parts of Canada (Clark *et al.*, 1986) as elsewhere (Fawcett, 1982). In Canada, however, only oats has declined as a whole (Figure 1). Increasing use of herbicides, other pesticides, and chemical fertilizers, greater use of machines, and the expansion of monoculture cropping have resulted in low within-field and between-field diversity (Alteiri and Letourneau, 1982; Hanski and Tiainen, 1988; studies reviewed above). In addition, farmers have eliminated hedgerows, woodlands, wetlands, meadows, grasslands, and other noncrop habitats within and adjacent to fields in efforts to increase the area of cultivation, to facilitate use of larger machinery and to remove potential reservoirs of weeds and other pests. In Canadian farmland, unimproved lands (areas of native pasture or hay land that have not been cultivated, brush pasture, grazing or waste land, sloughs, marsh and rocky land) have decreased steadily since 1941 (Figure 1). The elimination of field margin habitats is especially well documented in Britain (Potts and Vickerman, 1974; O'Connor and Shrubb, 1986) and, to a limited extent, in Canada (Merriam, 1978; Smutz, 1987; Boutin *et al.*, 1994), the USA (Best *et al.*, 1990) and elsewhere (Baudry *et al.*, 1988; van Dorp *et al.*, 1988). Increased spraying of herbicides to control weeds in hedgerows has been documented in some countries (Best, 1983; Boatman, 1987). Intensification of agriculture (including herbicide use and its associated practices) has contributed to an overall reduction in habitat heterogeneity within farmland, an increasing uniformity in the spatial arrangement of remaining habitats, and greater isolation of natural or semi-natural habitats (van Emden, 1965; Merriam, 1978; O'Connor, 1984; O'Connor & Shrubb, 1986; Mader, 1988).

Both the theoretical and applied ecological literature indicate that changes in composition, heterogeneity and interspersion of farmland habitats influence species richness and abundance (Hansson, 1979; Hanski and Tiainen, 1988; Merriam, 1988, 1990; see below). For plants, species diversity of arable weeds is expected to decrease because of decreasing within-field heterogeneity, quite separately from declines associated with the use of herbicides (Hanski and Tiainen, 1988). In Finland, the number of weed taxa in winter cereal fields was highest in the region with the greatest between-field and within-field variation, in part because herbicides had been used for a shorter period of time and in smaller quantities, fields were smaller, and subsurface drainage was less widespread (Eijasackers and Quispel, 1988b).

(b) Soil Organisms

Most herbicides are not acutely toxic to soil organisms, except for TCA and monuron at very high doses, and some triazine herbicides (Edwards, 1984). Curry (1970) suggested that cultivation is more detrimental to the majority of soil invertebrates than herbicide use. Fox (1964) found diverse effects of herbicides on soil invertebrates in grasslands. Abundance of earthworms was decreased by atrazine and TCA. Wireworm abundance was decreased by atrazine and monuron but unaffected by 2,4-D. Millipede abundance was decreased by monuron, increased by dalapon and TCA but unaffected by 2,4-D. The abundance of springtails was decreased by atrazine and monuron, increased by dalapon and TCA but unaffected by 2,4-D. Mite abundance was decreased by monuron, increased by dalapon and TCA but unaffected by 2,4-D. Fox concluded that when vegetative cover was decreased significantly, soil invertebrates were less abundant; when only the floristic composition was modified, soil fauna were not greatly altered. In a nine-year study of MCPA applied in cereal crops, Davis (1965) found that, despite marked differences in weed species (e.g., more grass species in MCPA plots), soil arthropods were negligibly affected.

Decomposers and other soil microorganisms (bacteria, protozoa and fungi) generally exhibit a transitory reduction in abundance or no change in response to herbicides (Brown, 1978; Smith, 1982). A few experimental studies have observed an increase in decomposition rate (based on CO_2 production) in response to herbicides (Ilijin, 1962).

Adaptation of soil microorganisms has been observed in fields which have been treated repeatedly, resulting in faster herbicide degradation compared to untreated soil (Smith and Aubin, 1991; Smith *et al.*, 1991). Most soil microbial populations in a site being prepared for a conifer plantation were unaffected in the long term by glyphosate or hexazinone (Chakravarty and Chatarpaul, 1990). However, the growth of ectomycorrhyzal fungi was significantly reduced by glyphosate in the first two months, an effect of concern given the importance of these microorganisms for seedling establishment. Some limited evidence from long-term studies in cultivated fields indicates that soil fertility was not adversely affected by repeated use of 2,4-D or MCPA in Saskatchewan (McCurdy and Molberg, 1974; Smith *et al.*, 1991), or MCPA, linuron, simazine or triallate in England (Fryer, 1981b).

Although apparently not acutely toxic, herbicides have caused adverse secondary effects on soil organisms by reducing food availability (e.g., plant roots, organic detritus) (Edwards and Thompson, 1973), decreasing plant cover (Fox, 1964; Curry, 1970; Way and Cammell, 1981), and reducing plant species diversity (House *et al.*, 1987; House, 1989). More basic research on biological processes in soil and on the ecology of soil organisms needs to be done before more accurate prediction and assessment of herbicide impacts on soil organisms can be made (Edwards, 1984). Long-term effects of repeated herbicide use have not been well studied in cultivated fields, and not at all in adjacent habitats, and pose a source of concern. The collective effects on soil organisms of different herbicides, combined with other pesticides, chemical fertilisers and cultivation, also needs to be investigated (Conacher and Conacher, 1986).

(c) Insects and other Invertebrates

In an early experiment by Adams (1960), 2,4-D killed a large proportion of adult beetles and delayed the growth and development of beetle larvae by 60%, thereby reducing the beetle's effectiveness to control aphids. Hassan (1983 cited in Klingauf, 1988) tested 77 pesticides and demonstrated that 54% of herbicides tested were harmful or moderately harmful to the parasitic wasp, *Trichogramma cacoeciae* and several beneficial arthropods.

An early study done by Putman (1949) showed that grasshopper nymphs were more abundant in summer fallow plots treated with 2,4-D than in untreated plots. The effect was attributed either to better nutritional quality of the remaining vegetation used by the grasshopper or to an increased rate of hatching on treated plots due to higher soil temperatures where broad-leaved plants had been killed. As noted earlier (Section 2a.i), increased abundance in response to changes in the content of protein or nitrogenous compounds of treated crops have been observed for aphids (Adams and Drew, 1965) and insect pests (Oka and Pimentel, 1976; Ishii and Hirano, 1963; Maxwell and Hardwood, 1960).

Widespread use of herbicides was responsible, together with improved seed cleaning, for the decline of insects dependent on a number of plant species in Britain (Fryer and Chancellor, 1970). A number of studies have shown that the diversity of insects, arthropods, and the like have declined as a result of a reduction of within-field heterogeneity by herbicides. Shelton and Edwards (1983) found that insect diversity was lower in soybean fields treated with herbicides than in soybean fields supporting a variety of weed species. In addition, the abundance of phytophagous insects was enhanced by herbicides while the abundance of their predatory insects decreased. The presence of certain weeds within and adjacent to crops can influence the insect fauna and lead to decreased pest damage compared to weed-free monocultures (Altieri, 1981 and references therein). Insect diversity was generally lower in cultivated fields than in habitats with heterogenous vegetation such as hedgerows and field margins (Lewis, 1969; Southwood and Cross, 1969). The abundance of arthropods was adversely impacted by monoculture cropping associated with herbicides (Dempster, 1969; Lamp *et al.*, 1984; Ali and Reagan, 1985). The number of beneficial organisms in crops was highest near field margins with flowering plants such as hedges and headlands (Klinghauf, 1988). Van Emden and Williams (1974) reported that insect pest movement into crops could often be attributed to the absence, rather than the presence of wild plant species in hedges. Spraying of hedgerows to control weeds and other pests reduces their quality as overwintering refuges for certain key species of polyphagous predators dependent on boundaries as overwintering sites (Sotherton, 1984, 1985).

In their review of crop-weed-insect interactions, Altieri and Letourneau (1982) concluded that the exacerbation of most agricultural insect-pest problems in recent decades was associated with decreasing vegetation heterogeneity within and among crop fields and in the surrounding uncultivated habitats as a result of the increasing monoculture cultivation facilitated by herbicide use. Biological pest control was reduced because the efficiency, abundance and diversity of predatory insects within fields were adversely impacted by the removal of sources of alternate prey/hosts, pollen and nectar, shelter, nesting and overwintering sites. Reduced crop rotation has resulted in increasing damage from pests and diseases (Klingauf, 1988). Intensification of agriculture, characterized by monoculture, continuous cropping, intensive and broadscale use of herbicides, larger field sizes and reduction of noncrop habitat, has led to higher incidences of pest attack for more prolonged periods (Pimentel and Perkins, 1980; Speight, 1983). By creating experimental systems, Mader (1988) and Wratten and Thomas (1990) demonstrated the benefits of interspersing crop and noncrop habitats by showing that polyphagous predators (especially beetles, spiders, syrphids, coccinellids) inhabiting noncrop habitats, spread into neighbouring crops and, provided some measure of biological control.

In summary, herbicides can be acutely toxic to some insects and arthropods. In addition, herbicide use can have adverse, indirect impacts on the abundance, composition and diversity of invertebrates, insects, arthropods and the like by altering plant structure and/or function, and species composition, by reducing plant species abundance and diversity, and by simplifying the composition, heterogeneity and interspersion of farmland habitats. In turn, these changes have the potential to adversely affect farmland mammals and birds through changes in food resources.

(d) Mammals

Few studies were available on herbicide effects on mammals in farmland. Most studies were on small mammals in old-fields, pastures, grasslands, rangeland and forests, or on large mammals in forests. Herbicides had secondary impacts on mammals by altering food quality and quantity, lowering survival and reproductive success, or changing species abundance, composition and diversity.

The food habits of deer mice (*Peromyscus maniculatus*) altered in response to changes in plant species composition of grasslands sprayed with 2,4-D while the diets of chipmunks (*Eutamias minimus*) and montane voles (*Microtus montanus*) were unaffected (Johnson, 1964). The diet of prairie dogs (*Cynomys ludovicianus*) changed significantly following spraying of Montana rangeland with 2,4-D, although no adverse effects on their health or behaviour were observed (Fagerstone *et al.*, 1977).

A significant reduction in the abundance of pocket gophers (*Thomomys talpoides*) in pastures sprayed with 2,4-D was attributed to lowered survival as a result of herbicide control of the annual and perennial forbs preferred as food by the gophers (Keith *et al.*, 1959; Tietjen *et al.*, 1967) and to movements of animals into unsprayed areas (Hull, 1971). In rangeland treated with 2,4-D, Johnson and Hansen (1969) found that the number of pocket gophers and least chipmunks (*Eutamias minimus*) decreased, the number of montane voles (*Microtus montanus*) increased, and the number of meadow voles (*Microtus maniculatus*) was unchanged. In old field habitats, the density and reproductive success of meadow voles was reduced by spraying with 2,4-D because a marked reduction in forbs limited meadow voles to a diet deficient in protein (Spencer and Barrett, 1980). Changes in rodent species abundance and diversity were observed following herbicide use to convert bushwood to grassland. Populations of eight rodent species declined or disappeared in response to reduced shrub cover (Lillywhite, 1977).

Numerous effects on mammals and other consumers from herbicide use in forestry are expected but not well documented (Newton and Norris, 1976; Biggs and Walmsley, 1988). The studies available showed a variety of responses. The abundance and species composition of small mammals was altered in response to 2,4,5-T in some studies (Kirkland, 1978; Savidge, 1978), but not others (Freedman *et al.*, 1988) and glyphosate in some studies (Ritchie *et al.*, 1987; D'Anieri *et al.*, 1987) but not others (Sullivan and Sullivan, 1982). The abundance of insectivorous and herbivorous species of small mammals were reduced by glyphosate while species with a more diversified diet and more generalized habitat requirements, such as omnivores, were unaffected (Santillo *et al.*, 1989b). A one-year control of herbaceous vegetation in clearcuttings of Western Oregon, with atrazine and 2,4-D, altered the species composition of small mammal communities as they responded according to their habitat preferences; species preferring grassy habitats were less abundant on treated than untreated plots (Borrecco *et al.*, 1979).

Forestry studies on large mammals have focussed on changes in the quality and quantity of browse. The palatability of browse plants to captive black-tailed deer (*Odocoileus hemionus columbianus*) was unaffected by glyphosate (Sullivan and Sullivan, 1979). The supply of browse plants for white-tailed deer (*Odocoileus virginianus*) was increased by 2,4-D and as a result, deer used sprayed areas more than unsprayed areas (Krefting and Hansen, 1969). Aerial spraying of herbicides over a number of years had little detrimental effect on use of rangeland habitats in Texas by white-tailed deer primarily because the dead tops and regrowth from roots of honey mesquite (*Prosopis glandulosa* var. *glandulosa*) still provided sufficient cover (Darr and Klebenow, 1975). Elk (*Cervus elaphus*) browsing increased in oak forest following spraying of 2,4,5-T in response to an increase in the amount of forage; the effect on mule deer (*Odocoileus hemionus*) was confounded by population declines in both sprayed and unsprayed areas (Kufeld, 1977). Thompson *et al.* (1991) found that picloram, hexazinone and tebuthiuron were effective in providing openings and increasing deer forage in an oak/hickory forest in northeastern Oklahoma. Available browse for moose (*Alces alces*) was considerably more reduced in a conifer plantation treated with glyphosate than with 2,4-D (Kennedy and Jordan, 1985).

Except for wooded fencerows and shelterbelts, the effects on mammals of changes in composition, heterogeneity and interspersion of farmland habitat associated with herbicide use have generally not been well studied. Wegner and Merriam (1990) showed that *Peromyscus leucopus*, a woodland rodent in eastern Ontario (Canada) has been able to adapt to more intensified agriculture by shifting its movement and habitat use patterns to include cultivated fields. Hedgerows, wooded fencerows and shelterbelts are important to mammals and other wildlife in many ways, including protection from wind and adverse weather, escape or refuge cover, food and foraging sites, reproductive habitat and travel corridors (Forman and Baudry, 1984; Fahrig and Merriam, 1985; Henderson *et al.*, 1985; Johnson and Beck, 1988; Merriam, 1990; Wegner and Merriam, 1979, 1990). Johnson and Beck (1988) report that 10 species of small mammals have a moderate to high dependence on or use of hedgerows or shelterbelts in the USA. Shelterbelts have helped to maintain or extend the ranges of several small mammal species in the Midwest USA. Many mammal species use hedgerows, wooded fencerows and shelterbelts as travel corridors between feeding sites, as protected cover at feeding sites and/or as routes for safe dispersal of adults or young (Johnson and Beck; 1988; Merriam, 1990). Practices which destroy or simplify structural complexity (such as livestock grazing [or herbicide sprays]) seriously degrade the benefits of hedgerows, wooded fencerows and shelterbelts to mammals (Merriam and Lanoue, 1990). The intensification of agriculture (including reduced/less diversified crop rotation, monoculture cropping and herbicide use) and the reduction, degradation and/or increasing isolation of noncrop habitats such as hedgerows, wooded fencerows, shelterbelts, and woodlands pose significant, potentially adverse consequences for the dispersal, abundance and possibly the long-term survival of some small mammal species in farmland (Fahrig and Merriam, 1985; Merriam, 1988, 1990).

(e) Birds

More intensive and extensive use of herbicides, the elimination and degradation of field margin habitats such as hedgerows, changes in crops grown and treatment techniques, and improved seed cleaning have been implicated in the decline of farmland birds detected by long-term monitoring in Britain. Fryer and Chancellor (1970) conclude that birds as well as insects, declined along with a number of plant species as a result of widespread use of herbicides in conjunction with improved seed cleaning.

In lowland Britain, spring pair densities of the grey partridge (*Perdix perdix*) have been surveyed since 1933. Numbers declined by 80% between 1952 and the mid 1980's (Sotherton *et al.*, 1988). Factors contributing to this decline included increased use of herbicides (and to a lesser extent, insecticides and fungicides) (Southwood and Cross, 1969; Potts, 1977, 1984, 1986; Sotherton *et al.*, 1988), and decreased availability of suitable nesting cover from removal of field margin habitats, particularly hedgerows (Rands, 1985). During the decline phase, greater herbicide use contributed to increased chick mortality early in the season by (1) decreasing the availability of the insect food preferred by chicks as a result of direct toxicity (Sotherton, 1982) and the removal of host plants by herbicides (Sotherton *et al.*, 1985; Potts, 1970, 1980), and (2) increasing the distances which had to be travelled by chicks to obtain food (Green, 1984; Rands, 1986). In addition, chick mortality was increased during cold springs because of their dependence on aphids with the reduction in their preferred insect food and the sensitivity of aphid phenology to temperature (Potts, 1970). Population declines in the grey partridge, similar to those in Britain, have been observed in North America, Europe and Asia (Potts, 1985, 1986).

O'Connor and Shrubb (1986) examined effects of farming on British birds by analyzing 35 years of monitoring and nest record data from a variety of regional samples with different histories of agricultural practices (including herbicide use). They found that bird species dependent on weeds and their seeds were particularly vulnerable to more intensive and extensive use of herbicides. Linnets (*Carduelis cannabina*) declined steadily because of high levels of chick starvation from the removal of economically-important arable weeds (e.g. *Stellaria, Polygonum, Chenopodium*) by intense and increasing use of herbicides. Concomitant with the steep increase in herbicide use in the mid-seventies, populations of Reed Buntings (*Emberiza schoeniclus*) declined as suitable nest sites in weedy hedges and fields were adversely modified by the removal of weeds by herbicides.

Populations of the Stock Dove (*Columba oenas*) were also limited by herbicides (O'Connor and Shrubb, 1986). This species declined sharply between 1951 and 1961 as a result of feeding on grain seed treated with organochlorine insecticides. Populations only partially recovered after the withdrawal of organochlorine seed dressings in 1962, because increasing use of herbicides and changes in other management practices adversely impacted food supplies. Herbicides reduced the abundance of weed seeds, particularly those in the Polygonaceae family, the major food plants of Stock Doves. The main sources of weed seeds for overwintering birds were less available because fields were increasingly sown with fall cereals rather than being left with cereal stubble or fallowed over the winter. In addition, changes in fertilization and herbicide spray techniques eliminated bare, thinly vegetated patches within cereal fields on which Stock Doves preferred to feed during the breeding season.

The switch from spring-sown to autumn-sown cereals also resulted in a reduction in Lapwing (*Vanellus vanellus*) densities, in later laying, and smaller clutches (O'Connor and Shrubb, 1986; Galbraith, 1988). The use of autumn herbicides, changes in fertilization and herbicide spray techniques, very early nitrogen dressing on winter wheat crops, and use of foliar fungicides increased plant densities in the early spring and promoted a more even stand thereby lowering its suitability as nesting habitat for Lapwings. A shift towards autumn-sown cereals was likely responsible for the decline of a number of other bird species as a result of a reduction in the feeding opportunities associated with spring cultivations (O'Connor and Shrubb, 1986).

A combination of the widespread mechanisation of cereal harvesting, the decline of summer fallows with the advent of effective herbicides, the widespread decline of spring cultivation, and an increasing loss of elms to Dutch elm disease were responsible for the decline of Rooks (*Corvus frugilegus*) in the more-heavily tilled counties of England (O'Connor and Shrubb, 1986). The breeding biology and abundance of Woodpigeon (*Columba palumbus*) was also affected by changes in the extent and type of crops grown, particularly winter cereals (O'Connor and Shrubb, 1986; Inglis *et al.*, 1990).

The abundance and species richness of farmland birds in Britain were affected to differing degrees by habitat features such as hedgerows, ponds, woodlands, field size and the proportion of fields which were arable (O'Connor and Shrubb, 1986). In response to more intense cultivation, breeding densities declined to a greater extent than the number of species. As field size increased, densities decreased for 23 of the 57 most common farmland species. The amount of hedgerow on a farm affected the overall density of breeding birds more so than the number of species. More individuals and more species hold territories in hedgerows which contain trees than in those which do not (Arnold, 1983; Osborne, 1984; O'Connor and Shrubb, 1986), not because of greater nest success but possibly increased availability of food, more song-posts, more refuges from predators or more vantage points from which to detect predators (Lack, 1987). Hedgerow bird densities were also highest where herbaceous and shrub diversity were high and where nearby woodland or scrub was abundant. The number of bird species increased with increasing hedgerow length although not in a linear fashion. Hedgerows appeared to be particularly important habitat for birds on farms with few woods (cf. also Arnold, 1983). Bird densities on farms varied proportionately with the abundance of habitats such as hedgerow, patches of shrub and trees. The number of species was affected to a lesser extent, likely because the loss of very numerous hedgerow species was partially offset by the gain of less abundant field species.

No studies have yet been conducted in North America at the broad geographic and long-term scales of the British studies reviewed above. Short-term studies of herbicide use in broad-leaved herbaceous vegetation, sagebrush, brushy fields and forestry have shown a variety of local effects on birds.

In a three-year, pre- and post-spray study on two islands (2 and 3 ha) in Alberta, Dwernychuk and Boag (1973) found that 2,4-D reduced the broad-leaved vegetation preferred as nesting cover by six species of ducks and resulted in a redistribution of nests, with aggregations in unsprayed areas and a decline in the total number of nests established. Their results are not directly applicable to ducks nesting in prairie farmland because the study was conducted in a rather uncommon habitat and at a higher application rate than that recommended for agricultural uses. However, based on a thorough review and analysis of the literature, Sheehan *et al.* (1987) cogently argued that the decline of ducks nesting in the prairie pothole region of North America may be related, at least in part, to adverse alteration of food, nesting and protective cover in uplands, sloughs and slough margins from repeated, broad scale use of herbicides.

In a two-year pre- and post-spray study in Montana, Best (1972) detected a 54% decline in the abundance of Brewer's sparrow (*Spizella breweri*) on ten, 16-ha plots where sagebrush (*Artemesia tridentata*) was totally killed by 2,4-D, while partial destruction of sagebrush and strip spray of 2,4-D did not induce any change. In contrast, the abundance of Vesper Sparrows (*Poecetes gramineus*) remained similar pre- and post-spray, presumably because they were not dependent on sagebrush cover to conceal their nests which were on the ground. Schroeder and Sturges (1975) confirmed that treatment of sagebrush with 2,4-D adOversely affected Brewer's Sparrow. Nesting was not affected immediately post-spray because the dead leaves remaining on the plant provided sufficient shade and protection from predators. Bird densities declined by 67% one year later and 99% two years later compared to untreated plots (which remained the same from year to year), and no nests were found. In a 6-year study of a transect (610m x 488m) through sagebrush rangeland in Oregon, Wiens and Rotenberry (1985) found that treatment with 2,4-D and subsequent removal of dead plants caused changes in the abundance, composition and diversity of bird species although not as rapidly as expected because of site tenacity of breeding individuals. By the end of the study, the abundance of Sage Sparrows (*Amphispiza belli*) and Sage Thrashers (*Oreoscoptus montanus*) had declined, Horned Larks (*Eremophila alpestris*) had increased in abundance, the abundance of Brewer's Sparrows had fluctuated greatly and Vesper Sparrows had appeared for the first time.

Beaver (1976) compared bird populations between two, 9.7-ha plots in a brushy field after 2,4,5-T was applied. Although the foliage of the main woody vegetation was killed on the treated plot, foliage cover remained similar between plots because damaged plants retained their leaves. No effect on the abundance or composition of breeding birds was detected after two years.

Studies in western Oregon forest showed that clearcuts unsprayed or sprayed with 2,4-D, 2,4,5-T or glyphosate differed in bird species composition but not in number of bird species or total density (Morrison and Meslow, 1984a,b). Differences in species composition were related to differences in habitat structure although many species were able to adapt. Other studies have shown that 2,4,5-T and glyphosate significantly decrease the density of some forest bird species (Slagsvold, 1977; Savidge, 1978; Biggs and Walmsley, 1988; Santillo et al, 1989a), the precise effect being tempered by behavioural adaptability of species and differential use among species of the vegetation remaining after treatment. In Nova Scotia, Freedman *et al.* (1988) found that the vegetation cover on a conifer clearcut was greatly reduced by 2,4,5-T but that the abundance of dominant breeding bird species such as Common Yellowthroat (*Geothlypis trichas*) and White-throated Sparrow (*Zonotrichia albicollis*) were unchanged after the first year.

As in Britain, agricultural practices (at least in part related to the use of herbicides) that lead to changes in land use, consolidation of fields, increases in the size of cropping units, and elimination or degradation of adjacent noncrop habitats can adversely affect the abundance and species richness of birds associated with farmland elsewhere. In Sweden, Robertson *et al.* (1990) found that agricultural regions with greater habitat heterogeneity and significantly more meadows, scrubland, deciduous woodland, reeds, farm buildings and gardens supported higher bird densities and diversity than regions with significantly more coniferous woodland and clearcuts. In Canada, Boutin *et al.* (1994) found that in southern Québec, farmland with some forested areas harboured a greater number of birds species such as warblers and thrushes, among others, that were not present in farmlands with little wooded areas. Smutz (1987) found that the nesting density of Ferruginous Hawks (*Buteo regalis*) declined with increasing cultivation in study plots (41 km^2) in the prairie region of southeastern Alberta. Swainson's Hawks (*Buteo swainsoni*) were more abundant in areas of moderate cultivation than in grassland or in areas of extensive cultivation, partly because they were better able to utilize the prey available in crop fields and were more tolerant of agricultural activity than ferruginous hawks. Clark *et al.* (1986) found that Red-winged Blackbirds (*Agelaius phoeniceus*) were less abundant in the more monocultural farmland of Southwestern Ontario than in Southwestern Quebec, and in Quebec (and to a lesser extent in Ontario), were more abundant as hayfield hectarages declined and agricultural landscape diversity increased. A computer model derived from field data indicated that maximum densities would be observed when interspersion of cropland and hayfield was high; as either habitat approached 80%, density should rapidly decline (Clark and Weatherhead, 1986). In the eastern USA, Vander Haegen *et al.* (1989) found that interspersion of corn fields, pasture and woodlands was needed to provide escape and roosting cover in close association with food needed by Wild Turkeys (*Meleagris gallopavo sylvestris*) to survive overwinter. Warner (1984) and Warner *et al.* (1984) found that the decline of the Ring-necked Pheasant (*Phasianus colchicus*) in Illinois was related, at least in part, to the replacement of hay and oats by corn and soybeans. Pheasant brood survival was lower on less diverse farms because of reduced insect abundance which forces birds to move more while foraging resulting in greater energy expenditures and increased vulnerability to predation. Also in the USA midwest, Best *et al.* (1990) found that more bird species and about five times more birds used the perimeters of cornfields than the centers. Consequently, bird species richness and abundance in cornfields decreased as field size increased. They predicted that narrower and/or more irregularly shaped fields would support more birds because they have proportionately more perimeter. Field edges bordering woodlands supported a greater abundance and more species of birds than herbaceous edges, although this had a negligible effect on bird species richness within adjacent cornfields.

Birds in hedgerows (or more typically, wooded fencerows) have not been well studied in North America (Best, 1983; Forman and Baudry, 1984; Shalaway, 1985). A recent review by Freemark *et al.* (1991) indicated that 57 bird species have been reported to use wooded fencerows in the Great Lakes-St. Lawrence region of eastern North America. Wegner and Merriam (1979) found that, from May to October in a single year, at least 41 bird species used wooded fencerows in farmland in Eastern Ontario (Canada). More birds and more species moved between woodland and fencerows than between woodland and fields or fencerows and fields. Birds moved along fencerows and foraged from them into fields. Poorly-vegetated fencerows restricted foraging by wood-nesting species. Shalaway (1985) found 13 species nesting in wooded fencerows in South Central Michigan (USA). Best (1983) found that fencerows with greater coverage of trees and shrubs supported more diverse and abundant avifaunas. A number of literature reviews have shown the importance of shelterbelts to birds and other wildlife in farmland (Forman and Baudry, 1984; Johnson and Beck, 1988; Freemark *et al.*, 1991). Johnson and Beck (1988) report that 29 species of birds benefit substantially and 37 species benefit moderately from shelterbelts in agricultural areas in the USA. Shelterbelts have helped to maintain or extend the ranges of several bird species in the midwest USA. The abundance and species richness of birds increases with shelterbelt area and length of perimeter, diversity of vegetation and vegetation complexity. However, insectivorous birds characteristic of larger and/or undisturbed forest habitats do not benefit appreciably from shelterbelts as nesting habitats. Species such as quail, pheasants, and songbirds may use shelterbelts as travel corridors between feeding sites, as protected cover at feeding sites and/or as routes for safe dispersal of adults or young. Practices which destroy understory vegetation (such as livestock grazing [or herbicide sprays]) seriously degrade the benefits of shelterbelts to birds. Although shelterbelts contribute substantially to avifauna populations in agricultural areas, shelterbelt benefits should not overshadow the values of maintaining and/or increasing woodland habitats.

The reduction, simplification and/or increasing isolation of noncrop habitats such as shelterbelts, wooded fencerows and particularly, woodlands, and the intensification of agriculture (including increasing field size, reduced/less diversified crop rotation, monoculture cropping and herbicide use) pose significant adverse consequences for the dispersal, abundance, species richness and possibly the regional survival of some bird species in farmland (Freemark, 1988; Merriam, 1988; Robertson *et al.*, 1990; O'Connor, 1991; Freemark and Collins, 1992; Villard *et al.*, 1992).

DISCUSSION

The widespread and increasing use of agricultural herbicides in Canada and elsewhere poses potentially serious problems for farmland wildlife. Unfortunately, the analysis of effects is highly complex given the vast array of herbicides and other pesticide products in use, the different environments in which they are used, and inadequacies in methods for detecting herbicide residues (Conacher and Conacher, 1986; Norris, 1986). The existing scientific evidence on ecological effects is often conflicting and is limited by insufficient toxicological and ecotoxicological information (Levin and Kimball, 1984; McGee and Levy, 1988), the lack of research on subtle or long-term effects at a variety of trophic levels (Conacher and Conacher, 1986), and serious gaps in our basic knowledge of the structure and functioning of ecosystems and their component organisms (Dempster, 1987). Environmental monitoring is generally not comprehensive or sensitive enough to observe significant effects (Cousens *et al.*, 1988). In addition, simultaneous changes in other agricultural practices (e.g. chemical fertilization, mechanization, crop selection) confound interpretations of ecological effects. With this in mind, our review of the existing evidence suggests that, in general, wild birds and mammals living in terrestrial farmland habitats are unlikely to be exposed to levels of agricultural herbicides that are acutely toxic. Our evaluation may need to be adjusted in future pending further research (see below).

Nevertheless, there is sufficient evidence to conclude that herbicide use can have secondary impacts on farmland wildlife, primarily mediated through a variety of toxic effects on plants, and changes in habitat composition, heterogeneity and interspersion. Herbicide use has induced changes which have a variety of, and sometimes, opposite effects on different taxonomic groups, species, and habitats in relation to biochemical, physiological or developmental processes, behaviour, abundance, composition, richness, heterogeneity, and interspersion. In general, these effects are interpreted as negative from an ecological point of view as well as from an aesthetic one (O'Connor and Shrubb, 1986; Baudry *et al.* 1988; Thomas, 1988).

Pesticide Regulation

Despite the importance of plants in primary production, nutrient cycling and the provision of food and cover for other organisms, and evidence of plant-mediated impacts, toxicity testing for the protection of nontarget plants is currently required on a case-by-case basis for the registration of pesticides in Canada or most other countries (Freemark *et al.*, 1990). Nontarget plant toxicity testing is required in the U.S.A. but only for certain use patterns, although changes are forthcoming. For terrestrial plants, only single-species tests on ten crop species are required. Crops, rather than wild species, are to be tested because of their genetic uniformity, availability and proven economic importance. How well crop species represent wild species, particularly those important as wildlife food (Figure 2), is unknown. Additional laboratory and ultimately, field testing, are triggered if a pre-defined detrimental effect is observed for at least one species at each tier.

For terrestrial plants, the Organization for Economic Co-operation and Development

(OECD, 1984) recommends testing with only three species from two or three families. No tier progression criteria are specified.

In Canada, guidelines for nontarget plant testing for registration of chemical pesticides have been developed and are being proposed (Boutin *et al.*, 1993; Boutin *et al.*, in review). Given the fact that the existing guidelines in USEPA and OECD are over ten years old, it appears essential that guidelines which incorporate the large quantity of recent toxicological and ecological research be adopted. Nevertheless, additional research is still needed to incorporate plant species which are more ecologically relevant, (Freemark *et al.*, 1990; Swanson *et al.*, 1991), to improve methods for hazard evaluation and risk assessment (Freemark and Boutin, in press), and to develop options for mitigating risks associated with herbicide use (Sotherton and Rands, 1987; Forsyth, 1989; Grue *et al.*, 1989, Freemark and Boutin, in press).

Research Needs

More research is needed on the differential sensitivity of birds and mammals to herbicides in relation to age (egg vs. young vs. adult), sex or size (Cope, 1971). In view of the high proportion of herbicides exhibiting reproductive effects (27%) in avian reproduction studies in laboratory a thorough examination of this potential problem is warranted (Mineau *et al.* 1994) . Potential effects on plants and animals from recurrent exposure to low levels of herbicides or to herbicides in combination with other pesticides also need further investigation (Hileman, 1982). Subtle or long-term effects of herbicide use on a variety of trophic levels merit further study, as do methods for improving detection of herbicide residues in the environment (Conacher and Conacher, 1986; Norris, 1986). More complete data on the quantities of herbicides used and areas treated needs to be collected in Canada as elsewhere, and made accessible (cf. Conacher and Conacher, 1986).

Large scale, long-term interdisciplinary research of different farming systems (especially in relation to weed management) is needed, particularly in North America, to integrate and evaluate the agricultural and ecological costs and benefits of herbicide use (along with other pesticides), and to more fully understand the relationships among wildlife, crop production, noncrop habitats, pesticide use, the entire farming system, and the socio-economic costs to the farmer and the public at large (Merriam, 1978; Trauger, 1982; Conacher and Conacher, 1986; O'Connor and Shrubb, 1986; Altieri, 1981, 1987; Tait, 1987; Thomas, 1988; Jordan *et al.*, 1990; Mathes and Weidemann, 1990). Incorporating aspects of landscape ecology and ecosystem management into farming systems could reduce or eliminate the need to use herbicides and chemical fertilizers and lead to better environmental management (Speight, 1983; Risser, 1985; Gulinck, 1986). Ecologically-based land use changes to increase ecological diversity and habitat heterogeneity of farmland should also be investigated (de Wit, 1988; Thomas, 1988). For example, pesticide exclusion strips approximately 6 meter wide at the edges of cereal fields have increased the abundance of gamebirds (Rands, 1985, 1986), small mammals (Tew, 1987), butterflies (Rands and Sotherton, 1986), other insects, and relatively rare arable weeds in British farmland (Sotherton *et al.*, 1985, 1988) with only a slight decrease in crop production. The practice has been implemented in some European countries (Schumacker, 1984; Snoo and Canters, 1990) and is being considered in others (Hald and Elmegaard, 1988; Hurle, 1988). In addition to providing agronomic and conservation benefits, the value of farmland for other reasons (e.g. historical, cultural, aesthetic, recreational) may also be enhanced by managing for increased ecological diversity and habitat heterogeneity within farmland landscapes (Speight, 1983; Thomas, 1988).

Changes in agricultural policy, and perhaps more importantly, agricultural technology, need to be examined in terms of their sustainability and potential effects on farmland ecology (Elliott and Wilson, 1983; O'Connor and Shrubb, 1986; Baudry *et al.*, 1988; Thomas, 1988; Allen and Van Dusen, 1988; Mineau and McLaughlin, 1994). Schemes to facilitate the adoption of revised agricultural practices that are more sympathetic to wildlife and the farmland environment should be investigated and implemented (Headley, 1980; Thomas, 1988). In particular, programs are needed to create an awareness among farmers that farm management decisions can affect the quality and quantity of wildlife habitat on farmland, and to provide information to farmers on the range of pest control alternatives and wildlife and environmental issues associated specifically with the use of herbicides and other pesticides (Conacher and Conacher, 1986; O'Connor and Shrubb, 1986).

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