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Insecticide use: Contexts and ecological consequences

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Abstract. Constraints to the sustainability of insecticide use include effects on human health, agroecosystems (e.g., beneficial insects), the wider environment (e.g., non-target species, landscapes and communities) and the selection of insecticide-resistant traits. It is possible to find examples where insecticides have impacted disastrously on all these variables and others where the hazards posed have been (through accident or design) ameliorated. In this review, we examine what can currently be surmised about the direct and indirect long-term, field impacts of insecticides upon the environment. We detail specific examples, describe current insecticide use patterns, consider the contexts within which insecticide use occurs and discuss the role of regulation and legislation in reducing risk. We consider how insecticide use is changing in response to increasing environmental awareness and inevitably, as we discuss the main constraints to insecticide use, we suggest why they cannot easily be discarded.

Key words: Ecology, Ecotoxicology, Environment, Risk, Insecticides

Abbreviations: BSE – Bovine spongiform encephalopathy; Bt – Bacillus thuringiensis; DDT – Dichlorodiphenyltrichloroethane; DEFRA – UK Department for Environment, Food & Rural Affairs; EPA – US Environmental Protection Agency; FAO – Food and Agriculture Organization; GAO – US Government Accounting Office; IGR – Insect growth regulator; IPM – Integrated pest management; JHA – Juvenile hormone analogue; LC – Lethal concentration; LD – Lethal dose; OP – Organophosphate; ULV – Ultra low volume; UNEP – United Nations Environment Program

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Introduction

> "Farming looks mighty easy when your plow is a pencil, and you're a thousand miles from the corn field." Dwight D. Eisenhower, Presidential Speech, September 25, 1956.

Of the world's total land area of 150 million km^2, 10% is under arable production, 55% supports meadows, pastures and forests and the remainder is not suitable for agricultural use. Much of that arable production is dependent on "conventional" farming methodologies (i.e., production practices that use synthetic agrochemical inputs) that, despite Malthusian pronouncements to the contrary (Ehrlich, 1968; Brown, 1998), have continued to meet the demands of human population growth. Globally, since 1960, the population has doubled (UN, 1999), agricultural productivity has risen 2.6-fold, but the arable area under production increased by just 10% (FAO, 2004a). Although productivity across Europe, Asia, the Americas and Australasia has rocketed (DEFRA, 2005; FAO, 2004b) Africa's overall yields (despite some local successes), continue to decline due to an intractable combination of drought, civil unrest, land degradation, poor farming methods, and unfavorable land tenure and ownership systems (UNEP, 1999).

Notwithstanding these overall triumphs, we are becoming increasingly aware of the consequences of agricultural intensification. Intensive agriculture is responsible for air and groundwater pollution, the eutrophication of water systems, greenhouse gas emissions; and is the dominant anthropogenic source of ammonia, the main cause of acid rain. The extent and methods of agriculture (though not necessarily insecticides per se) have demonstrably led to extensive and permanent loss of biodiversity in many localities (for invertebrate communities, see Schultz and Liess, 1999; Heaney et al., 2001; Benton et al., 2002; for butterflies in the UK, see Longley and Sotherton, 1997; for farmland birds in Europe, see Krebs et al., 1999; Donald et al., 2001; Newton, 2004; and for amphibians in Australia and North America, see Anon., 1999a; Blaustein and Keisecker, 2002; Davidson et al., 2001, 2002). We must assume that these patterns are being repeated globally. Furthermore, the sustainability of modern agriculture itself is affected by its own impacts on land degradation, salinization (i.e., accumulation of salts), the availability of water; and the reduction of crop, livestock, and agroecosystem diversity.

Globally, agriculture operates within an increasingly free-market economy; albeit currently heavily subsidized in some large developed nations. It is driven by the weather, demand, supply, and competition; and it prioritizes profit over social need. Currently, approximately 15% of the world's 6 billion people suffer malnutrition (FAO, 2005). By 2050, this system, with all its attendant pressures on the environment, must feed a world population of 9 billion people. As many northern hemisphere countries choose (or are driven) to opt out of agricultural self-sufficiency, much production is shifting to the developing world. For example, the last decade saw a 12% decline in self-sufficiency in staple foods in the UK (Davies, 2005) and everyday foodstuffs are now as likely to be grown in Peru or Ethiopia as they are locally. These shifting production patterns are driven by the global expansion in trade that has encouraged developing countries to pursue competitive export-led agricultural systems (Altieri and Rojas, 1999) but for many commodities, markets have been over-supplied and prices have fallen. The ensuing struggle to maintain agricultural profits in both the developing and developed world escalates intensification and promotes the conflict between the farmer and environment that is described as "the tragedy of the commons" (Hardin, 1968). With increasing globalization of food production there comes a collective responsibility to protect the livelihoods and profits of farming communities whilst preserving existing biodiversity and "ecosystem services" provided by agricultural landscapes.

In this context, the ecological consequences of insecticide use are of major concern. Although other aspects of modern agriculture often have a greater environmental impact, insecticides are among the agricultural tools most popularly associated with environmental harm. Their expressed purpose is to kill pests; consequently, they may have lethal or sublethal impacts on non-target organisms (such as organisms that recycle soil nutrients, pollinate crops, and prey on pest species) and reduce and/or contaminate food supplies for organisms at higher trophic levels.

This article presents a comprehensive overview of the ecological impacts of insecticide use and provides some context as to why insecticides continue to play such a key role in modern agriculture. Previous syntheses of the literature have tended to deal with insecticides briefly and as a subset of agrochemicals and pesticides in general (i.e., Andow and Davies, 1989) or were written some decades ago, prior to the development of many current chemistries and before many modern examples of insecticide impacts and risk assessments had been researched and written (i.e., Metcalf, 1980). This review focuses on the literature of the last 20 years. We refer readers to Andow and Davies (1989) and Metcalf (1980) for a further historical evaluation of the subject. We do not attempt to address the whole of the vast literature on insecticide ecotoxicology; most of it conducted in laboratories under highly artificial conditions. Such studies, though useful for identifying potential hazards, do not illuminate the true environmental and ecological consequences of insecticide use in the field. Table 1 is provided as a guide to the insecticides referred to herein. It lists the common active ingredients, their mode of action and their approximate date of introduction.

Current trends in insecticide use

Pimentel (2005) states that pesticide use in US arable systems returns about \$4 per \$1 invested for pest control. The attractiveness of conventional pest management methods is therefore clear. Those costs however, do not include the societal or ecological costs of agriculture. Annual environmental and social costs associated with agricultural pesticide use in the US amount to \$10 billion, and \$2 billion for water surveillance and pesticide clean-up alone (Pimentel, 2005). As US crop and animal revenues are approximately \$200 billion per annum, this represents about 4% of farm revenues (Fare et al., 2006). In the UK, the government estimated the costs associated with agricultural water pollution to be 1–2% of total agricultural value (DEFRA, 2000; Pretty et al., 2000). The environmental and health costs of pesticide use in

Table 1. Insecticide groups, mode of action and date of introduction.

Primary site and mode of action	Insecticide type	Common examples	First use[a]
Acetylcholinesterase inhibitors: *Block the action of the enzyme acetylcholinesterase, disrupting the transmission of impulses between nerve cells*.	Carbamates	Aldicarb, Bendiocarb, **Carbaryl**, Carbofuran, Carbosulfan, Methiocarb, Methomyl, Pirimicarb, Thiodicarb	1956
	Organophosphates	Acephate, Chlorpyrifos, Diazinon, Dimethoate, Fenitrothion, Fenthion, **Malathion**, Methamidophos, Monocrotophos, Parathion, Pirimiphos, Profenofos, Temephos	1950
GABA-gated chloride channel antagonists: *Interfere with chloride channels in the nerve membrane, disrupting ion transfer and the transmission of impulses between nerve cells.*	Cyclodiene organochlorines	**Chlordane**, Endosulfan, gamma-HCH (Lindane)	1945
	Phenylpyrazoles (Fiproles)	**Fipronil**	1993
Sodium channel modulators: *Interfere with sodium channels in the nerve membrane disrupting ion transfer and the transmission of impulses between nerve cells*.	Organochlorine	**DDT**	1943
	Pyrethroids	**Allethrin**, Bifenthrin, Cyfluthrin, lambda-Cyhalothrin, Cypermethrin, **Deltamethrin**, Fenvalerate, Permethrin, Resmethrin	1952 1977
	Pyrethrins	Pyrethrins (**pyrethrum**)	1850s
Nicotinic Acetylcholine receptor agonists/antagonists: *Mimic the action of the neurotransmitter acetylcholine thus blocking receptors and disrupting with the transmission of impulses between nerve cells.*	Neonicotinoids	Acetamiprid, **Imidacloprid**, Nitenpyram, Thiacloprid, Thiamethoxam	1991
	Nicotine	**Nicotine**	1930s
	Spinosyns	**Spinosad**	1996
Chloride channel activators: *Bind to and activate chloride channels in the nerve membrane disrupting ion transfer and the transmission of impulses between nerve cells*.	Avermectin	**Abamectin**, Emamectin benzoate	1985
Juvenile hormone mimics and analogues: *Competes, mimics or interferes with the juvenile hormones essential for insect development*.	Juvenile hormone analogues and mimics	Hydroprene, Kinoprene, Methoprene, **Fenoxycarb**, Pyriproxyfen	1993
Compounds of unknown or non-specific mode of action (selective feeding blockers)	Cryolite	**Cryolite**	1929
	Pymetrozine	**Pymetrozine**	1999
Microbial disruptors of insect midgut membranes (includes transgenic crops expressing *Bacillus thuringiensis* toxins).	Bacillus *species*	**Bacillus thuringiensis** *subsp*. israelensis, Bacillus sphaericus, Bacillus thuringiensis *subsp*. aizawai, Bacillus thuringiensis *subsp*. kurstaki, Bacillus thuringiensis *subsp*. tenebrionis	1961
Inhibitors of oxidative phosphorylation: *Disrupt electron transport within cells*.	Diafenthiuron	**Diafenthiuron**	1997
	Chlorfenapyr	**Chlorfenapyr**	1985
Inhibitors of chitin biosynthesis: *Inhibit the normal formation of the insect exoskelton*.	Benzoylureas	Novaluron, **Diflubenzuron**, Teflubenzuron	1983
	Buprofezin	**Buprofezin**	1988
Ecdysone agonists and moulting disruptors: *Interfere with the process of insect molting* .	Diacylhydrazines	**Halofenozide**, Tebufenozide	1999
	Azadirachtin	**Azadirachtin**	1985

Table 1. continued

Primary site and mode of action	Insecticide type	Common examples	First use[a]
Mitochondrial complex I electron transport inhibitors: *Disrupt electron transport within mitochondria.*	Rotenone	**Derris**, Rotenone	1850s
Voltage-dependent sodium channel blockers: *Interfere with sodium channels in the nerve membrane by disrupting ion transfer and the transmission of impulses between nerve cells.*	Indoxacarb	**Indoxacarb**	2000

Note: This is not intended as an exhaustive list. Information herein was partly derived from the Insecticide Resistance Action Committee (http://www.irac-online.org) list in which insecticides are ranked by mode of action. Within any grouping, the toxicity of insecticides (in terms of species specificity and effective dose) and their environmental persistence can vary widely.[a] Dates refer to the insecticide example given in bold.

1996 in the UK were equivalent to those of habitat loss and soil erosion, but well below those of gas emissions, food poisoning and of the single most costly event of that year, the British bovine spongiform encephalopathy (BSE) outbreak (Pretty et al., 2000).

Constraints to the sustainability of insecticide use include effects on human health, agroecosystems (e.g., beneficial insects), the environment in general (e.g., non-target species, landscapes and communities) and the selection of insecticide-resistant traits in pest species. For all these categories, it is possible to find both examples where insecticides have been used disastrously and others where the hazards posed have been (through accident or design) ameliorated. The World Health Organization (WHO) has estimated that some 20,000 people die annually from pesticide exposure (WHO, 1990) but those chemicals also protect yields, profits and public health. Some insecticides have been shown to devastate natural enemy populations in some systems (e.g., Boza Barducci, 1972; Matlock and de la Cruz, 2002), but in others, particularly with some of the more novel insecticides, there appear to be minimal impacts (Naranjo et al., 2004). Insecticides have had profound effects on predatory bird populations in some instances (e.g., Sibly et al., 2000) but others have been used in seemingly sensitive ecosystems for decades without evidence of impact upon non-target organisms (Resh et al., 2004; Holmes, 1998). Some have been used so intensively that the evolution of resistance has compromised their use within generations (Ishaaya and Horowitz, 1995; Devine et al., 2001; Zhao et al., 2002), but for others, resistance remains rare or easily managed. In relation to the latter point, it is of interest that although resistance can be a constraint on field efficacy, it seldom signals the end of all useful applications for that chemical. Of the 544 species listed as resistant by the pesticide-resistant arthropod database (Anon., 2006), almost 30% appear on the list by virtue of a single uncorroborated citation reflecting, at best, a unique observation of a single population. Moreover, even for those species such as the yellow fever mosquito (*Aedes aegypti*), the cotton whitefly (*Bemisia tabaci*), and the German cockroach (*Blattella germanica*), where resistance has been convincingly argued in hundreds of publications, traditional insecticides still play a role in their control.

Despite increasing awareness of the hazards posed by their use, the area treated with insecticides in the developed world has remained static over the last decade. In the UK, approximately 6,000,000 ha of arable land were treated annually between 1990 and 2003 (Anon., 2003b). This is the equivalent of a quarter of the total landmass of that country. The statistics reflect multiple applications to the same areas. In California, 6–8,000,000 ha were treated annually between 1992 and 2001 (Epstein and Bassein, 2003; Wilhoit, 2002; Wilhoit et al., 1999). Generally, the overall weight of active ingredients used has declined due to newer compounds tending to have greater intrinsic insecticidal activity. For example, the LC_{50} of the organophosphate (OP) profenofos (registered in the USA in 1982) against susceptible populations of the silverleaf whitefly (*B. tabaci*) is ca. 4 ppm while the LC_{50} of the juvenile hormone analogue (JHA) pyriproxyfen (granted emergency approval in the USA in 1996) against the same *B. tabaci* strain is 1000 times less (El Kady and Devine, 2003).

It is important to note that generally, even in developed countries with powerful environmental legislation and active lobbying groups, insecticide use is not on the wane. There is a widely held perception that the concept of "integrated pest management" (IPM) has been or will be successful in reducing pesticide use (Allen and Rajotte, 1990; Grewell et al., 2003; Metcalf, 1980). IPM refers to "a decision support system for the selection and use of pest control tactics, singly or harmoniously coordinated into a management strategy based on cost/benefit analyses that take into account the interests of and impacts on producers, society, and the environment" (Kogan, 1998: 248). One of its main emphases is on decreasing chemical inputs. However, in truth, and particularly for high value crops such as vegetables, synthetic pesticides remain a major line of defense in most pest management programs, integrated or otherwise. IPM will be discussed in more detail in a later section.

Newer compounds with lower impacts

The use of the oldest and most toxic cyclodienes, carbamates and OPs is slowly decreasing; at least some have been banned by most countries but overall, they retain a 50% worldwide market share. The synthetic pyrethroids, introduced in the late 1970s (Elliot et al., 1978) now account for 20% of global insecticide sales and have vastly improved mammalian and avian toxicity profiles over their forebears. Other newer insecticide classes, some of which have very specific activity against particular arthropod orders, have been introduced over the last 15 years. These developments have been driven by a growing environmental awareness – the "scramble for green credibility among corporations" (Kroma and Flora, 2003: 25) and increasingly rigorous registration, harmonization and risk assessment processes such as those proceeding in Europe, which are also re-assessing older compounds whose toxicological profiles may no longer be acceptable (Anon., 2003a).

These new insecticides have several advantages over the older classes. Low mammalian toxicity allows for a shorter pre-harvest interval, greatly simplifying the logistics of their use. Most are less likely to harm natural enemies and other non-target species; thus reducing their impact on the wider environment and making them, at least in principal, more compatible with an IPM ethos. For example, indoxacarb (an oxadiazine insecticide registered in 2000) is very effective against lepidopteran (butterfly and moth) larvae but allows most predators and immature wasp parasites which attack these caterpillars to survive (Hewa-Kapuge et al., 2003; Studebaker and Kring, 2003). Tebufenozide and methoxyfenozide [dibenzoylhydrazine insect growth regulators (IGRs)] disrupt the molting process in Lepidoptera but do not affect beneficial insects (Dhadialla et al., 1998; Hewa-Kapuge et al., 2003). Benzoylphenylurea IGRs (such as diflubenzuron and teflubenzuron) disrupt chitin synthesis and have a broader spectrum of activity but, because they become active only or mainly after being ingested, direct effects on parasitic wasps (hymenopterans) are minimized. Pyriproxyfen, a pyridine compound developed in the late 1980s, is a juvenile hormone analogue (JHA) that inhibits egg production and metamorphosis. It is active primarily against sucking insects and has little effect on hymenoptera. It is also an effective mosquito larvicide and, although it can be toxic to aquatic organisms, the lethal dose differential between the majority of non-target aquatic organisms and mosquito pupae is so large as to make it a good choice for mosquito control in sensitive environments (Sihuincha et al., 2005).

In the developed world, an increasing percentage of treatments are now made with these newer compounds, which are termed "reduced risk" insecticides by the Environmental Protection Agency (EPA). A Government Accounting Office study (GAO, 2001) concluded that although overall agricultural pesticide use had not declined in the US between 1992 and 2000, the use of the "riskiest pesticides" had declined by 14% by weight of active ingredient. Similar shifting patterns of insecticide use are occurring throughout the developing world but these changes are gradual. Furthermore, few insecticides can be ecologically risk-free. The pyrethroids can be acutely toxic to fish and have broad spectrum effects on invertebrates (Mian and Mulla, 1992; Smith and Stratton, 1986). Pyriproxyfen is highly toxic to predatory coccinellids (ladybirds) and can disrupt IPM programs (Grafton-Cardwell and Gu, 2003). Diflubenzuron is toxic to crustaceans and can badly disturb aquatic environments (Lahr et al., 2001) and contact with teflubenzuron can affect the fecundity of hymenopteran (i.e., wasp) parasitoids (Furlong et al., 1994). Newly developed compounds may also turn out to be more damaging than revealed by the eco-toxicological data packages submitted during the registration process. In 2000, during the final stages of registration, the EPA declined to allow the insecticide chlorfenapyr to be used in cotton pest control because of the chronic risk that it presented to reproduction in birds (EPA, 2006). Each country's registration process, however, is idiosyncratic; chlorfenapyr remains registered for use, and is widely used in cotton in Australia and throughout Asia.

It is useful to consider the drivers which influence insecticide use patterns. The GAO (2001) quotes EPA officials as suggesting that decreases in the use of the riskiest group of insecticides occurred because they were: (a) discontinued by EPA regulatory action, (b) discontinued because of business decisions by manufacturers, (c) non-competitive with newer or cheaper alternatives, (d) less effective due to pest resistance, or (e) used less frequently with crop varieties genetically modified to resist insects. Clearly, change as a result of ecological or environmental awareness was not a decision-making factor in these cases for farmers. It is a common observation that, in the northern hemisphere, voluntary reductions in insecticide use are often contemplated but seldom implemented because prophylactic treatment regimes are easier to implement than more complicated decision processes, such as IPM, that require monitoring pest threshold population levels. Additionally, farmers are unwilling to act unilaterally (and risk competitive disadvantage), and voluntary schemes act outside the "incentives culture" that farmers have become accustomed to (Lohr et al., 1999).

Insecticides, dogma and pragmatism

The application of insecticides even within prescribed regulatory guidelines can have detrimental environmental consequences. These effects are exacerbated by inappropriate use and there are many examples of insecticide misuse and abuse. In the worst cases, the effects of insecticides are difficult to distinguish from those of general agricultural mismanagement. The Aral Sea in Central Asia is considered by the Food and Agriculture Organization (FAO) and United Nations Environment Program (UNEP) to represent the world's worst example of how poorly planned and executed agricultural practice has devastated a once productive region. Residues of organic and OP insecticides are rife in the area (Ataniyazova et al., 2001; O'Hara et al., 2000) and although there is little in the published literature on the effects of such massive agricultural pollution on the ecology of the Aral Sea (with the exception of very specific changes in invertebrate diversity; see Aladin and Potts, 1992; Nazarova, 2006), the effects on human ecology are believed to have been devastating (Jensen et al., 1997, Zetterström, 1999, 2003). Even in developed countries using approved and regulated insecticides in well-legislated systems, there is plentiful evidence of ongoing ecological and environmental degradation resulting from pesticide use. Many of the clearest examples describe the cumulative effects of insecticide residues in rivers that drain agricultural areas. In California, the waters and sediments of the Salinas River, which empty into the Monterey Bay National Marine Sanctuary, are acutely toxic to a variety of aquatic invertebrates (Anderson et al., 2003; Phillips et al., 2004). This is also true of the Alamo and New Rivers in California's Imperial Valley where 8 years of observations (1993–2001) showed the impacts of OP pollution on macro invertebrates to be sustained and severe (de Vlaming et al., 2004).

Examples such as the above are well publicized and, not surprisingly, the link between insecticides and the environment in the public psyche is almost exclusively negative. It is therefore a common but often unsupported assumption that insecticides are culpable for many global changes in biodiversity and ecology. This dogma is reinforced by the contentious nature of insecticide use and the existence of many vested interests. Insecticide use patterns influence agrochemical company and farm profits, and are a major issue for those who design agricultural strategy, subsidy and development policies. Such dogma also drives consumer behavior, triggers strong personal, political and ethical responses, and influences governmental and non-governmental approval ratings. The rigid perceptions held by opposing camps ensures that, although the science that characterizes the effects of insecticides on the environment may be empirical and disinterested, the interpretation of that data may not be. An abundance of raw data recording the effects of an insecticide does not ensure that only one clear recommendation can be made regarding the risk that it poses.

The tendency to over or under interpret results regarding insecticide impacts leads to confusion over the true extent of their effects and influence. The accumulation of pesticide residues in the food chain through the consumption of crabs and fishes from pesticide-sprayed rice paddy fields was thought to be the cause of a musculo-skeletal condition in humans (Krishnamachari and Bhat, 1978; Mohan, 1987), but more recent studies have shown that the disease was the result of a high rate of inbreeding among affected communities (Agarwal et al., 1997). The BSE crisis in the UK was improbably attributed on occasion to OP use (Gordon et al., 1998; Purdey, 1996). The recent decline in vulture populations in India was thought to be pesticide induced (Anon., 1999b; Nair, 1999; Prakash, 1999) until it was found to be the result of an unexpected interaction between a veterinary medicine (diclofenac) and vulnerable avian kidneys (Green et al., 2004; Oaks et al., 2004). Global declines in amphibian populations are strongly associated with agrochemical use (Blaustein and Kiesecker, 2002; Davidson et al., 2001; Sparling et al., 2001), but it seems unlikely that insecticides per se are a major contributory factor (Cohen, 2001; Houlahan et al., 2000; Pounds et al., 2006; Relyea, 2005).

An example that illustrates how difficult it is to reach consensus is the controversy regarding the impact of insecticidal seed treatments on honeybees (*Apis mellifera*) in Europe. In some parts of the globe, bee populations have declined drastically over recent decades (e.g., Kremen and Ricketts, 2000; Kremen et al., 2002), due in part to parasitic mites and to protozoan and bacterial disease outbreaks. However, in 1994, French beekeepers reported unusual behavior and bee mortality among bees feeding on sunflowers treated with the neonicotinoid imidacloprid at seed planting. These effects were reported to be accentuated year on year and occurred just after flowering. Imidacloprid was implicated as the causal agent of the problem. In 1998, the French *Commission des Toxiques* reviewed its impact and found that the data did not "indisputably" link this insecticide or its metabolites to effects on bees. Further complementary studies were recommended; nonetheless, in 1999 imidacloprid seed treatments for sunflowers were suspended until the data were available. This was seen as a victory for the "precautionary principal" and was instituted, in part, simply because sunflowers had been shown to translocate imidacloprid throughout their tissues. The same rationale and continuing problems with bee mortality and honey yields resulted in the suspension of the same treatment in maize in 2004. An alternative seed treatment (fipronil) was also banned in sunflowers when its residues were found in flowering sunflower crops. A European Commission review of the ban is pending.

Much of the evidence for and against the deleterious effects of imidacloprid seed treatments for sunflowers on bee populations is in the form of government and industry reports, but it is certainly true that under some circumstances, sunflower pollen and nectar can exhibit residues of imidacloprid in the range 1–10 ppb (Bonmatin et al., 2003). Other authors claim that residues in the nectar and pollen of seed treated sunflowers in the field are consistently lower than 1.5 ppb (Schmuck et al., 2001). Acute toxicity in bees occurs at about 60 ppb (Suchail et al., 2001a; Nauen et al., 2001), although chronic effects have been reported at far lower concentrations, for example, 50% death after eight consecutive days of ingesting 0.1–10 ppb imidacloprid (Suchail et al., 2001b). This latter result is clearly damning, but it has never been repeated; therefore, it is contested. Despite 30 consecutive days of feeding 2–20 ppb imidacloprid, Schmuck et al. (2001) found no effects on mortality, feeding activity, wax/comb production, breeding performance or colony vitality. Moreover, Schmuck (2004), reporting the findings of four independent studies, found no increase in honeybee mortality nor of behavioral abnormalities during 10 days of exposure to sucrose solutions treated with imidacloprid at 0.1, 1.0 or 10 ppb. In a review of a number of industry-funded and independent trials, Maus et al. (2003) reported that bee colonies placed next to sunflower or rape fields sown with imidacloprid-dressed seeds suffered no significant effects in terms of mortality, foraging activity, colony development, brood status or pollen and nectar stores. In 2005, the French Food Standards Agency published a study examining the effect of feeding honeybee colonies on imidacloprid-tainted syrup (0.5–5.0 ppb). Their development and survival were compared with control hives over the course of several months. Mortality, population size, hive activity, fecundity and frequency of disease were all monitored. There were no differences suggesting any toxic influence by imidacloprid (Faucon et al., 2005). No contrary field evidence has been published, despite the fact that this has been a pressing issue for almost a decade.

On balance, it would appear that imidacloprid did not directly influence bee survival. The remaining potential source of risk is in the form of sublethal, behavioral impacts of imidacloprid seed treatments. The presence of imidacloprid in artificial honeybee food sources has been shown to exert such effects; most convincingly by decreasing attendance and the proportion of actively feeding bees at contaminated (6–24 ppb) artificial food sources (Colin et al., 2004; Decourtye et al., 2003, 2004). It is unclear however, whether this was the result of sublethal toxicity or of a less insidious (i.e., repellent or antifeedant) effect.

Despite the bans, honey harvests from sunflowers in France have continued to worsen over the years. The 2000 harvest was just 30–40% of the 1995–1996 yields and disease remains a major factor. One French national survey noted that 76% of apiaries suffered from at least one serious disease (Faucon et al., 2002). In other countries and in oilseed rape crops, which serve as key pollen and nectar sources for bees at particular times of the year, imidacloprid seed treatments have not been associated with bee declines. Investigative studies in major honey producing countries such as Canada (where imidacloprid seed treatments are used in the potato crop) and Argentina (in sunflowers) produced no data that would support a ban. In 2003, the International Commission for Plant-Bee Relationships, having reviewed the argument for the deleterious effects of seed treatments on bees, concluded that "the decline in bee health and colony performance, reported in a number of countries, was unlikely to have a single cause" (Lewis, 2003: 11).

What is clear from such examples is that we are forced to accept one of two options regarding risk assessment: (1) that methods of assessment are agreed, standardized and implemented on regional and global scales; or (2) that each region or country implements its own system, resulting in wildly different recommendations. The former runs the risk of being too inflexible and unable to keep pace with ecological or methodological developments. The latter would be something of a triumph for strongly dogmatic lobby groups because it would require acceptance that empirical data have no universal interpretation; therefore, policy can be decided subjectively.

Direct effects

The great majority of insecticide poisoning events on non-target organisms, particularly those affecting less familiar, non-charismatic species in the developing world, are likely to go unrecorded. The following examples examine a subset of direct poisoning events or shifts in population levels and species distributions. Changes at the population level may be due to direct toxicity or to sublethal effects manifested as reductions in life span, development rates, fertility, fecundity, sex ratio, and behavior (e.g., feeding, foraging and reproduction). There is a vast literature cataloguing such effects (see Stark and Banks, 2003). Reports of direct toxicity in the 1960s and early 1970s were dominated by the highly persistent organochlorines, especially dichlorodiphenyltrichloroethane (DDT), and their profound effects on predatory bird (raptor) populations (Newton, 1979). Most of these chemicals were replaced by organophosphates and carbamates over the course of the 1970s and 1980s. These too proved occasionally devastating to many bird populations, mostly as the result of birds ingesting treated seeds or grains (reviewed by White and Kolbe, 1985). During this period, the use of carbofuran granules in North American corn fields undoubtedly had a devastating effect on graniverous (grain-eating) bird species (Mineau, 1988; Mineau and Collins, 1988). The data provided led one group to estimate that 60–70 million North American birds were dying annually in the US as a direct result of pesticide exposure (Pimentel et al., 1992). These figures continue to be widely quoted (e.g., Pimentel, 2002, 2005) but it is important to note that the figures now have little relevance to the current situation in North America. Granular carbofuran formulations are mostly banned in the US and none are registered in Canada (Mineau, 2005). It cannot be assumed that other formulations or insecticides exhibit similar effects. An exhaustive orchard study of the effects of repeated liquid sprays of another carbamate insecticide, methiocarb, revealed no "serious hazard" to any of the dozens of bird species monitored (Hardy et al., 1993). It seems generally true that in the developed world, following the withdrawal and changes in use patterns of some insecticides, populations of affected bird species have largely recovered (Boatman et al., 2004).

In other parts of the globe however, OPs and carbamates remain associated with direct poisoning events. The mass poisoning of more than 5000 Swainson's hawks (*Buteo swainsoni*) in Argentina in 1995–1996 resulted in a major agrochemical company ceasing the production of monocrotophos (Goldstein et al., 1999; Winegrad, 1998). The hawks had been exposed to insecticide by ingesting treated insects following a grasshopper outbreak in the grassland *pampas*. In India, a closely monitored population of the endangered Sarus crane (*Grus antigon*) was threatened after the ingestion of monocrotophos-treated wheat seed (Pain et al., 2004). Most poisoning events are accidental, but in some instances, insecticides are used in ways which will predictably result in undesirable, widespread damage to non-target species. Spraying the organophosphate fenthion is the major means of controlling the red-billed quelea (*Quelea quelea*), which is a major pest of grain crops throughout semi-arid, sub-Saharan Africa. It is no surprise that birds of prey, and perching and songbirds (passerines) are acknowledged as common casualties; as a result of being sprayed directly and from eating affected carcasses – which may be found up to 20 km or more from the treated sites. Terrestrial arthropods are also strongly affected (reviewed by McWilliam and Cheke, 2004).

Small mammals appear generally more resilient to direct pesticide applications in the field than birds. This may be simply due to their more nocturnal (normally, pesticides are applied during the day), crepuscular and cryptic behavior. Schauber et al. (1997) suggested that the predominant effect of an application of the OP azinphos-methyl to large enclosures containing voles (*Microtus canicaudus*) and deer mice (*Peromyscus maniculatus*) was to produce a number of immediate deaths from which populations recovered, within months. McEwen et al. (1996) showed that small grassland mammal species are highly individual in their response to insecticide application due to differences in innate insecticide susceptibility and, post-spray, in their ability to compete. Deer mice were twice as sensitive as grasshopper mice *(Onychomys leucogaster)* and ground squirrels *(Spermophilus tridecemlineatus)*; post-spray trapping studies indicated decreases only in deer mice. The responses of small mammals to insecticide application are clearly specific to environment and insecticide chemistry. Live-trapping studies on deer mice following the application of malathion and carbaryl showed no post-treatment decreases in abundance (McEwen et al., 1996). An investigation of the effects of an ultra low volume (ULV) malathion application also found no effects on small mammal populations in the field (Erwin and Sharpe, 1978).

Non-target arthropods are often severely affected by insecticide use, at least in the short term. Soil dwelling populations have been recorded as being affected by the application of insecticides to agricultural, grassland, desert and forest ecosystems. Such effects typically result in transient reductions in soil fertility and productivity (reviewed by Pimentel, 1992). Aquatic invertebrate communities seem particularly vulnerable; Davies and Cook (1993) showed that aerial spraying of the pyrethroid cypermethrin against butterfly and moth (lepidopteran) pests in a Eucalyptus plantation in Tasmania resulted in contamination of several streams and 200-fold increases in the downstream displacement of invertebrates on the day of spraying. This "drift" was apparent for many days, but most species recovered after the occurrence of subsequent floods. Stoneflies (Plecoptera) and mayflies (Ephemeroptera) were most affected. Physiological changes were also noted in Brown trout (*Salmo trutta*), presumably caused by changes in diet composition and perhaps by the ingestion of cypermethrin from dead and dying invertebrates.

One of the commonest examples of insecticides disrupting arthropod ecosystems is when their use causes increases in pest numbers, termed "resurgence," and the appearance of new pest species by removing their predators and parasitoids. These effects are clearly a consequence of the direct impact that insecticides may have on non-target species and have been observed repeatedly in experimental field trials and on larger scales for broad spectrum insecticides (Metcalf, 1980; Godfray and Chan, 1990; Holt et al., 1992; Devine et al., 1998; Trumper and Holt, 1998; Van den Berg et al., 1998; Mochizuki, 2003). Grasshopper outbreaks in some North American rangelands over a 30-year period were attributed to resurgence caused by an intensive chemical control program against the grasshoppers themselves (Lockwood et al., 1988).

Similar examples exist elsewhere. At the end of the 1930s, the entire Cañete Valley in Peru was given over to cotton production. At that time the major pest was the tobacco budworm (*Heliothis virescens*). It was combated first with arsenical insecticides and then with DDT, lindane, and toxaphene. Resistance to all those pesticides evolved and other pests came to the fore as susceptible predators and parasitoids were removed from the system. By the mid-1950s, 16 pesticide applications per year were being applied, the pest complex was continuing to expand, and cotton production had crashed. In 1956, an IPM program was introduced that involved the prohibition of organic insecticides, the release of some biological control agents (mainly parasitic trichogrammatid wasps), and some changes in cropping and harvesting practices to break the Heliothis lifecycle. As a result, cotton farming once again became sustainable (reviewed by Boza Barducci, 1972). The story of cotton production in Peru is now a classic tale of IPM implementation, but progress beyond the 1960s is seldom reported. In fact, although the area now produces a small amount of organic cotton, most of the crop remains reliant on insecticide inputs made in an IPM context. Insecticide inputs can often be reduced, but they can seldom be dispensed with. Moreover, farming communities do not always apply the lessons learned to all their activities. The Cañete Valley now produces a great many other crops besides cotton in a far more deregulated environment, and farmers are currently experiencing analogous problems – now brought about by the overuse of broad-spectrum OPs and carbamates – to those that faced their cotton-growing forebears in the 1950s (Centro Internacional de la Papa, 2002a, b; Cisneros and Mujica, 1999; Holl et al., 1990). It is an unfortunate reality that insecticides are such an attractive and cheap option for pest control, that farmers will not change practice voluntarily unless some insurmountable event or enforced legislation compels them to do so.

Broad spectrum insecticides tend to have immediate, but predominantly short-term (2–3 months) effects on non-target insects (Holland and Luff, 2000; Jansen, 2000; Longley et al., 1997). Insecticide drift (the wind and temperature assisted displacement of chemical to areas that are not intentionally targeted) and direct contact with pyrethroids has been noted to reduce non-target insect numbers in unsprayed headlands (de Snoo, 1999; Longley and Sotherton, 1997), sub-field plots (Moreby et al., 1997), and in field-scale experiments (Holland et al., 1994). It should also be noted that there is considerable variation between the susceptibility of different non-target groups. For example, the pyrethroids fluvalinate and esfenvalerate did not significantly reduce catches of hoverfly (syrphid species) larvae in field plots but did affect ladybird larvae (*Adalia* spp.). Conversely, the carbamate pirimicarb had no effect on ladybird larvae but reduced the numbers of hoverfly larvae significantly (Jansen, 2000).

Studies that attempt to quantify the impacts of insecticides on the effectiveness of natural enemies as control agents for field pests are rare, but Furlong et al. (2004) found that the effect of beneficial insects was greatest at sites adopting IPM (i.e., reduced insecticide input) and least at sites practicing conventional pest management strategies. At IPM sites, the contribution of natural enemies to mortality of diamondback moth (*Plutella xylostella*) permitted the cultivation of marketable crops with no yield loss, but with a substantial reduction in insecticide inputs. The abundance and diversity of natural enemies was generally greatest at sites that adopted IPM. Studies such as these, in which lower levels of parasitism were directly associated with management practices rather than with other indirect effects (such as lack of prey following their removal by insecticides) are highly illuminating.

It is not only in arable landscapes that insecticides can be shown to exert effects upon the environment. The use of broad-spectrum insecticides during campaigns against the tsetse fly have had pronounced effects on non-target organisms. The use of dieldrin, DDT, and γ-BHC in the southern African savannah badly affected reptiles, small mammals, fish, birds and insects. Similar effects were detailed in tsetse control campaigns in West Africa, where dieldrin and endosulfan were applied by truck and by air to forested riverine areas. In that instance, effects on amphibians, monkeys and bats were noted and all classes of compounds including pyrethroids, caused marked population declines of insectivorous birds, fish and freshwater crustaceans for prolonged periods (reviewed by Grant, 2001). Eventually, these residual insecticide applications were replaced by less persistent aerial treatments (considered preferable, see Cockbill, 1979; Magadza, 1978) but these continued to exert an impact, particularly on freshwater crustaceans (Takken et al., 1978). Since the 1980s, the emphasis in tsetse control has shifted to insecticide treated traps and the application of dips and "pour-ons" to cattle. The species that were affected by previous treatment regimes are considered to have recovered (Grant, 2001) but even highly-targeted insecticide applications can have adverse ecological effects. Pour-on treatments for cattle have been associated with declines in scarab beetle populations caused by insecticide residues in cattle dung (Vale et al., 2004).

Indirect effects

Insecticides can have direct lethal or sublethal toxic effects upon individuals and populations or may have indirect effects that result from the removal of prey species and/or competition. There is therefore concern over the potentially insidious effects of pesticides operating through the food chain (Mills and Semlitsch, 2004; Morris et al., 2005). Impacts of insecticides on invertebrates may reduce the availability of food resources, and affect productivity and/or survival of species that depend on them. Simple measures of direct toxicity are clearly not sufficient in assessing the full ecological consequences of insecticide use, but standard methodologies for assessing insecticide effects on prey and competitor removal do not exist and would be impossible to standardize, particularly at the field scale.

It has long been accepted that the indirect effects of insecticides can be subtle and may mask or confuse the directly toxic effects at the population or community level (e.g., Emlen et al., 1958). In mark-recapture studies on the effects of the cyclodiene endrin on vole (*Microtus* spp.) populations, direct toxic effects (both lethal and sublethal) and indirect (non-toxic) effects were shown to be acting in unison (Morris, 1970). In experimental plots, the application of the insecticide killed voles and decreased numbers locally but the wider population subsequently responded as it would to depopulation by disease or trapping. Post spray, there was more recruitment of individuals (i.e., voles) to the experimental plots than to the control plots. Moreover, recruits to the treated plots survived better than individuals entering the control population, probably due to fewer aggressive interactions between voles in the less populated, treated areas. Depending on the period over which events are being observed, insecticide application can clearly have a range of non-intuitive effects on populations.

Such unpredictable ecosystem disruptions are commonplace. In a study on the impact of carbaryl on the southern leopard frog (*Rana sphenocephala*), carbaryl had no direct negative effects on tadpoles but reduced their mortality by having a lethal impact on invertebrates that preyed on them. Carbaryl also decreased the abundance of the microscopic plants and animals (periphyton) upon which the tadpoles fed. The overall effect was an increase in tadpole survival but with smaller individuals predominating (Mills and Semlitsch, 2004).

There is currently little evidence of significant population effects on birds arising from direct effects of insecticides in the UK (Boatman et al., 2004). Although many species of farmland birds are in decline, the causal factors are difficult to pinpoint and the possible indirect impacts of increasing pesticide use remain unknown for the many species that have exhibited population declines and changes in distribution coincident with agricultural intensification (Campbell and Cooke, 1997). To demonstrate the indirect effects of insecticides at the population level, it is necessary to show that such compounds impact upon food resources in a way that reduces breeding performance or survival. Such a pattern has been shown most convincingly for the grey partridge *Perdix perdix* (Potts, 1986; Rands, 1985, 1986), although data showing that pesticides have indirect effects on the available resources and reproductive capacity of other species is also available. Insecticides have been shown to cause a reduction in brood size in yellowhammers (*Emberiza citronella*) by reducing the number of invertebrate food items available (Boatman et al., 2004). During the same study no such effects were seen for skylarks (*Alauda arvensis*) but availability of invertebrate food items did affect chick condition in both skylarks and yellowhammers. The authors noted that nesting birds might benefit from a number of measures taken to reduce the impact of the insecticide, namely: minimizing applications of persistent broad-spectrum insecticides during the breeding season and providing alternative unsprayed habitats in which to forage.

Long term effects and recovery

There is a paucity of surveillance data for insecticide effects collected over long temporal scales. Studies monitoring environmental pollutant levels, however, suggest that many systems show a remarkable capacity for recovery. The dilution, dispersal and biodegradation of contaminants, all act, often over very long periods, to reduce their ecological impact (e.g., Niemi et al., 1990). For example, the use of DDT, which is one of the most infamous persistent organic insecticides, was banned in the United States in 1973. By the 1980s, there had been a more than 90% reduction of DDT in Lake Michigan fish reflecting the breakdown and dilution of the product in the environment and hence in the food chain (De Vault et al., 1996).

Apart from the persistence of the insecticide, the degree to which affected populations can recover is partly dependent upon the recruitment of new individuals from an unaffected population. Most animal populations found in farmland consist of metapopulations – discrete subpopulations that are loosely connected by dispersal (movement between groups) but that do not experience the same disturbances or fluctuations in their environments at the same time. This overall population structure permits the rapid recovery of invertebrate species within insecticide-sprayed fields (Jepson and Thacker, 1990). Simulation models that predict the impact of local applications of pesticide on the population dynamics of such populations show that the chances of a predatory invertebrate persisting in an insecticide treated field are improved if there are unsprayed refuges nearby, if the rate of application is low, or if the insecticide used is selective or not highly toxic. Less intuitively, the models also show that there appears to be an optimal dispersal rate of the predator population which maximizes their ability to persist in insecticide treated fields (Sherrat and Jepson, 1993). It is reasonable to surmise that "island" or fragmented habitats will be far less likely to re-establish than contiguous populations and environments. Careful spatial and temporal control of insecticidal application might quicken the recovery rates of non-targets affected by pesticides (just as, conversely, it may help reinvasion by pests).

Depending on their spatial and dispersive characteristics, some non-target species' populations will be only temporarily affected by insecticides and will recover if treatment is stopped. Mian and Mulla (1992) noted that when a variety of pyrethroids was used in fresh water systems, the population recovery of affected species to pre-treatment levels was noticed within weeks to months after application. Populations of fish species, dependent on those affected invertebrates for food, also recovered quickly. Giddings et al. (2001) reviewed a number of mesocosm and field studies using realistic cypermethrin and esfenvalerate concentrations (both pyrethroids) and revealed that the most sensitive organisms included freshwater shrimps (amphipods), hoglice (isopods), midges, mayflies, copepods, and waterfleas (cladocerans). The least sensitive included fish, snails (mudworms (oligochaetes), and rotifers. Populations usually recovered within months as a consequence of the presence of untreated refuges, life stages that were more insecticide-tolerant than others, rapid generation times, and the influx of immigrating adults that had not been exposed. In Giddings study, indirect effects on fish were not observed (but see Davies and Cook, 1993).

The fact that most effects are transient holds true for a variety of terrestrial species too. Honeybee colonies that had consumed low levels of the OP methamidophos in artificial diets exhibited greater mortality of eggs and larvae than control colonies, but surviving larvae developed successfully and colonies recovered rapidly from single treatments with no residual effects (Webster and Peng, 1989). The application of a range of carbamates and OPs proved lethal to populations of earthworms (lumbricids) after single field applications, but populations recovered in 20 weeks, although there were intimations of deleterious effects on mole populations due to prey removal (Potter et al., 1990). Studies on the tsetse endemic areas treated with endosulfan and deltamethrin in the mid 1980s were reassessed in 1997. Grant (2001) judged that all species considered threatened at the time had recovered. He concluded that all insecticide related effects were temporary and that numbers of even the most sensitive species of invertebrates recovered within a year.

One of the longest-term, most carefully controlled insecticide-based pest management programs regards the control of blackfly larvae (*Simulium* spp.), the adults of which vector onchocerciasis in West Africa. Fifty thousand km of rivers in 11 countries (an area of greater than 1 million km^2) were sprayed on a weekly basis from 1974 to 2002 as part of the Onchocerciasis Control Program (OCP) which officially ended in 2002 after an estimated 600,000 cases of river blindness were prevented and 25 million ha. of land were made safe for agricultural use and habitation. Insecticides, many of them broad spectrum OPs (i.e., temephos, phoxim, pyraclofos), were used in rotation with other insecticides (i.e., permethrin, etofenprox, carbosulfan and *Bacillus thuringiensis* (Bt)) to prevent the evolution of resistance by the pest. Long-term monitoring of their effects on non-target invertebrate and fish communities demonstrated few obvious impacts (reviewed by Resh et al., 2004). More recently, a number of papers have concluded that the biological variations found post-spray were ecologically acceptable (Crosa et al., 2001a, b; Paugy et al., 1999; Yameogo et al., 2001) and that no permanent damage to non-target invertebrate populations had occurred. No species at higher trophic levels, particularly the insectivorous fish group, seem to have been affected. This has not been the case in all localities where blackfly (*Simulium*) larviciding has been carried out. In some parts of the

Nile River, the use of DDT caused profound changes in the food resources available for elephant-nosed fish (*Mormyridae* spp.) and spiny eels (*Aethiomastacembelus frenatu*) and populations suffered accordingly (Paugy et al., 1999).

Monitoring studies on long temporal scales which track sequential and potentially cumulative ecological effects in arable environments remain rare and are extremely valuable. Hummel et al. (2002) monitored populations of surface-dwelling arthropods over a 4-year period in vegetable production systems under a chemically based system (endosulfan and esfenvalerate applied at weekly intervals) and a biologically based one (Bt applied at weekly intervals). Pitfall trapping was used to monitor carabid beetles, staphylinid beetles and lycosid spiders. Trap catches of all groups were significantly affected by insecticide application but, despite intense insecticide pressure by two broad spectrum compounds over 4 years, invertebrate numbers were not decimated; rather they declined to 20–50% of the level found in Bt treated plots. Neither was there evidence of any cumulative worsening in effect over the 4-year period. It seems that invertebrate assemblages can be quite resilient to pesticide use; at least for those species that have a different spatial distribution in the crop than that of the insecticide target (e.g., ground-dwelling insects may be relatively resilient to foliar sprays).

Other systems however, will be more vulnerable. The effect on fresh water invertebrates of carbaryl (applied at field rates for spruce budworm moth suppression) was studied in a number of streams in Maine. These were subject to three treatment regimes: (1) streams sprayed with carbaryl for the first time, (2) streams sprayed for two consecutive years, and (3) unsprayed streams. The initial response was an enormous increase in the downstream displacement (''drift'') of invertebrates. Sampling also showed significant declines in abundance of Plecoptera (stoneflies), Ephemeroptera (mayflies), and Trichoptera (caddisflies). Stonefly larvae did not reappear in any treated stream within 60 days and streams that had been treated for the second year running had low stonefly numbers compared with those that had not been exposed. Some insect orders however (Diptera and Oligochaeta) were unaffected (Courtemanch and Gibbs, 1980).

Long-term field studies in the UK, on a range of arable crops (Young et al., 2001), suggested that there are few adverse long-term effects of pesticides on non-target organisms (including insects, spiders, earthworms and soil microbes). In that study, the application of broad-spectrum insecticides resulted in declines in the numbers of many non-target arthropods, but these usually recovered within the same growing season. Less temporary effects were seldom noted and affected only soil-dwelling collembolans (springtails). Numbers of these organisms remained comparatively low in treated plots, 2 years after application.

Risk assessment

The term ''hazard'' is used to communicate the existence of potential harm whilst ''risk'' relates to the probability of harm occurring. In recent years ecological risk evaluation schemes have evolved into complex systems of assessment and analysis. Recommendations and regulations for insecticide assessment, registration and re-evaluation are now in place in most of the developed world (reviewed by Greig-Smith, 1992). The data required to inform that process include acute, sublethal, chronic, carcinogenicity, mutagenicity, metabolism, reproduction, developmental, neurotoxicity, and mechanistic studies on mammals and birds (covering all exposure routes; oral, dermal and inhalation); and tiered tests (the results gained at one level triggering or negating the need for a subsequent layer of tests) on aquatic and terrestrial indicator species. It also includes information on the chemical's physical behavior in soil, on plant surfaces and in water. It is a current preoccupation of many governments to rank the ecological risks posed by insecticides in order to encourage and inform the registration of more benign products, thereby reducing agricultural impacts on the environment. This is dependent upon ranking pesticides based upon a manageable number of standardized tests and it is a contentious matter because, whilst no single parameter can fully describe or predict environmental impact, the inclusion of limitless tests with differing methodologies on various species makes it impossible to integrate results into standardized assessment models (see Levitan, 2000). The ongoing debate over the experimental scales and criteria needed for risk assessment (e.g., Dearfield et al., 2004; Maud et al., 2001) ensures that, at present, there are no universally applicable models available. For the time being, the process of insecticide assessment and registration remains idiosyncratic and subjective despite a seemingly vast and varied ecotoxicological literature.

The significance of this impasse is that there is no global consensus regarding the cost and benefit of using specific insecticides. For example, endosulfan is banned in Colombia, Germany, the Indian state of Kerala, The Netherlands, Singapore, Sweden, Syria and the UK, amongst others; however, in many African countries and in almost all cotton producing nations (including the USA and Australia), it is in common use. Such anomalies are particularly common in the developing world when countries lack registration procedures, locally relevant knowledge on environmental effects, or even basic information on the toxicity or efficacy of the compounds that are in use (Wiktelius et al., 1999, Everts, 1997; FAO, 1996). "Highly toxic" insecticides are the main pesticide category in use in many poorer countries, and over 50 of the 60 developing countries who responded to a questionnaire in 1993 reported that they were not studying the effects of pesticides on the environment (FAO, 1996). Studies of pesticide impacts in developing countries remain rare but, unsurprisingly, effects on ground, river, and coastal waters, fish and grazing animals have all been noted (Lacher and Goldstein, 1997; Dasgupta et al., 2007 for Bangladesh). Many countries in the developing world are already using pesticides at application rates which exceed those associated with major environmental damage in Europe and North America. This is encouraged by the availability of inexpensive, generic and locally produced insecticides (e.g. FAO, 2003). In this era of increasingly free agricultural trade, the conditions under which food and fiber crops are produced are of universal concern. Consensus would therefore be desirable – if only to ensure equality of environmental safety between the developed and developing nations.

It is important to note that the need to use any particular insecticide will change radically with time, locality and purpose. In the US, there is no license to use the OP temephos in water sources for mosquito control because of the environmental and human health risks that it poses. However, in many parts of the world temephos is used in water storage tanks to prevent the development of the mosquito *A. aegypti* (Sihuincha et al., 2005) which carries the dengue virus. In West Africa it is used in drinking water to kill the intermediate copepod host of the guinea worm (*Drancunculis medinensis*) (Sam-Abbenyi et al., 1999). These latter uses are recommended by WHO based on the risk of using temephos weighed against the risk of contracting disease. The current argument in favor of re-instating DDT for mosquito control arises from the fact that there is little reason to suspect that the intra-domiciliary application of DDT for malaria control is damaging environmentally (Roberts et al., 1997; WHO, 2006).

The overwhelming majority of the data used in existing insecticide risk-assessment models is in the form of simple lethal dose (LD) or concentration (LC) estimates. This is valuable, but the information it provides is limited because the sublethal effects of toxicants can affect populations at concentrations far lower than those seen in acute toxicity tests (e.g., Chandler et al., 2004; Guilhermino et al., 1999; Kuhn et al., 2000; Preston and Snell, 2001; Stark, 2005). Giddings et al. (2001) showed that the lowest observed adverse-effect concentrations for the pyrethroids cypermethrin and esfenvalerate in experimental mesocosms corresponded to values one tenth of those derived in simpler laboratory experiments. More realistic tests on subtler effects (behavioral, biochemical or physiological) in response to more realistic exposure (leaf residues, treated prey items, etc.) may be available in some instances, but there is usually little understanding of how to weight these in terms of their predicted impact on populations or individuals in the field or in ecological communities. Field studies of insecticide effects on non-target populations (pit-fall trapping, sweep netting, etc.) are rare, due to expense and unpredictability. Long-term trials are seldom reported.

Ecotoxicity evaluations would clearly be improved by the development of standardized methods that accounted for a larger subset of effects (direct, indirect, sublethal and demographic) that an insecticide might exert. An improved assessment process might include the use of life tables to characterize the responses of individuals (such as mortality and reproduction) over their life span to give a time-series portrait of toxicity and a measure of the effect on population growth rates. Such indicators would clearly reflect effects that cannot be seen by acute toxicity tests alone (Forbes and Calow, 1999). One example of a highly-controlled, laboratory-based, and easily replicable methodology is an exposure test for meiobenthic organisms. Using a copepod as the test species, Chandler et al. (2004) exposed larvae to the phenylpyrazole insecticide fipronil. They tracked survival, development rates, sex ratio and fecundity through mating and brood production. Mortality and fecundity rates were then used to project population distribution. The results showed that low concentrations of fipronil resulted in effects on reproduction that represented real risk to crustaceans at concentrations far below doses considered to have no effect on most aquatic test species. These kinds of tests could contribute much to models of risk assessment without having to resort to the vagaries of field trials, but they are clearly more time-consuming and expensive to carry out and would increase the costs of assessment many-fold.

The bodies that advise, legislate, and approve insecticide registration and usage need some measure of relative toxicity that is based on standardized methodologies, data requirements, and models for assessment. Moreover, these need to be robust and universally applicable (Kovach et al., 1992). In an attempt to standardize assessments, a variety of indicators have been designed that are intended to aid risk analysis. These tools grade the ecological impact of chemicals using general classifications, which are often independent of empirical considerations. Some comparisons of the impact of "integrated" and "conventional" crop production systems have been made using little or no empirical data whatsoever (Bues et al., 2004; De Jong and De Snoo, 2002) and – self-evidently and speciously – conclude that systems that use lower per ha dosages of chemicals, less toxic chemicals, and spray methods that allow less insecticide drift have less impact than those that do not. Models such as the Environmental Impact

Quotient (EIQ) (Kovach et al., 1992) consist of equations that sum the combined effects of complex variables that include human dermal and chronic toxicity, non-target effects in fish, bird, bee and beneficial arthropods and abiotic effects such as leaching potential. Such models are potentially more realistic but difficult to standardize because of the need to subjectively "weight" the variables by giving them numerical value. Maud et al. (2001) reviewed the utility of a range of risk indices (including the EIQ) for use in developing and assessing policy in the UK. There was poor correlation between the rankings of the 133 pesticides used in the evaluation when only toxicological data were used. This improved when data on recommended application rates were incorporated but there was still wide variation. Moreover, most ranking scores grouped closely together in a very limited part of the indices' potential range. The major problem identified in the application of the indices was the lack of available field data.

Regardless of the problems faced by those whose job it is to assess and register insecticides, the existing frameworks are actively used to shape insecticide use patterns in the developed world. Examples of such activity include the fact that since 1990 the EPA has authorized complete or partial bans on many of the most toxic insecticides, including chlordane, chlorpyrifos, disulfoton, ethion, methyl parathion, oxydemeton methyl, phorate and toxaphene. The EU directive on pesticide registration and harmonization is actively replacing older compounds with "reduced risk" chemicals (Anon., 2003a). In the UK, the Department for Environment, Food and Rural Affairs (DEFRA) is drafting arguments on the desirability of a pesticide tax (DEFRA, 2000). Despite ongoing concerns, the overall risk-benefit analysis for insecticide use in the developed world is vastly improved – particularly if one takes as a comparison the multiple and manifest problems caused by older, more problematic chemical classes such as the organochlorines and the worst of the OPs. There is thus reason for cautious optimism, but the problems of pesticide abuse remain most prevalent where there is little or no legislative machinery.

Ecological benefits

"High yield conservation" is a rather ill-defined idea (e.g., Avery, 2000), which proposes that intensively farmed, highly productive arable areas help preserve the remaining agri-suitable land (mostly forests) that remains unexploited. The hypothesis is supported by a number of scientists and commentators (CGFI, 2005) as well as by bodies which are ideologically committed to intensive agriculture such as the Hudson Institute, the Center for Global Food Issues and a number of agrochemical companies. The argument is based upon the fact that species richness is related to the area of wild habitat. As that area declines, so does the number of species it harbors (e.g., Coleman et al., 1982). The importance of wild habitat conservation is thus universally acknowledged. It is conceivable that intensive agriculture plays a part in that process by relieving the necessity of further exploiting remaining areas of wilderness through low input agriculture (as long as the impact of intensively farmed agricultural land on neighboring ground and water is minimized). Between 1961 and 2002 there was a 10% increase in the global area under arable production (FAO, 2004a) most of it secured through deforestation. There is no doubt that there is some validity to the idea of high yield conservation but few authors explicitly mention the concept when discussing agricultural production processes. Wagner et al. (2004) note that results from long-term studies in North American forests show very large yield gains following herbicide use. They note that demands for wildlife habitat and biodiversity conservation require that the growing need for timber products must be satisfied by the current area of commercially managed forest. Intensively managed, high input plantations might be essential in fulfilling that need.

The idea that there might be a tangible benefit to an ecosystem resulting from insecticide use is anathema to many, but there are occasional examples where the argument is well advanced. Gypsy moth (*Lymantria dispar* L.) is an exotic, invasive species that substantially disturbs forested ecosystems in North America. The defoliation that results when gypsy moth outbreaks are left unchecked has severe environmental impacts that can be balanced against the impact of insecticides used for gypsy moth control (mostly preparations of the Lepidoptera-specific Bt). For example, the consequences for not using insecticides for the control of the introduced gypsy moth in North America might be defoliation on a vast scale which can dramatically affect populations of native Lepidoptera (Redman and Scriber, 2000; Work and McCullough, 2000). Gypsy moths compete with native caterpillars for food, and decreases in the abundance and richness of larvae and adults from the family Arctiidae (tiger moths) have been noted in infested plots (Sample et al., 1996). Besides these impacts on Lepidoptera, many other direct and indirect effects of gypsy moth defoliation on natural ecosystems have been documented. Defoliation can result in tree death (particularly of oaks) (Davidson et al., 1999), and has been shown to increase the predation rates of forest dwelling birds, possibly by increasing the visibility or accessibility of their nests (Thurber et al., 1994). Defoliation of oaks also dramatically decreases acorn production, which can decrease the numbers of small mammals as well as alter the foraging patterns of large herbivores and omnivores such as deer and bear (Kasbohm et al., 1996; Selås, 2003). Increased temperature and sunlight on the forest floor can damage shade-loving organisms and encourage invasion by better adapted plants. Gypsy moth defoliation is thought to be one reason why red maple (*Acer rubrum*) is replacing oak (*Quercus* spp.) as a dominant species in some previously defoliated American forests (Jedlicka et al., 2004) and has also been shown to affect water quality and freshwater ecosystems by increasing nitrate content of forest streams (Eshleman et al., 2004; Townsend et al., 2004).

There are additional specific examples of a positive link between insecticides and conservation. Bevill et al. (1999) suggested that Pitcher's thistle (*Cirsium canescens*), which is endemic to an area of North America, might be protected from its insect herbivores by the use of insecticides. Such "insect exclusion intervention" (i.e., the spot treatment of rare plants) is contested as a conservation strategy by others (see Lesica and Atthowe, 2000; Louda and Bevill, 2000).

The alternative: Reduced input pest management

"Organic" agriculture, defined as farming without synthetic input, serves a rapidly growing niche market in the developed world. During the 1990s, it was one of the fastest growing markets for North American and European agriculture. Approximately 2% of California's farmland is now managed organically. Yields from organic plots are often competitive with those of conventional plots, but are more unpredictable (Trewavas, 2001). Post-harvest losses also tend to be greater for organic crops than for conventionally grown ones, which tends to result in a higher final cost of production. Despite this, organic and conventional profits are often equal because of the higher price that organic products command. However, the free market dictates that increases in the supply of organic products will result in lower prices and reduce profitability. The transition from conventional to organic agriculture is often a painful one, due to new farm investments, lag periods prior to certification and less predictable farm incomes, which farmers may not be able to afford without subsidy (FAO, 2003). In some areas however, where less competitive markets, low profit expectations, and lack of purchasing power converge, it is certainly possible to dispense with a great deal of synthetic input (for example, Cuba following the collapse of the Soviet Republic; Rosset, 1997).

For the majority of producers however, some form of IPM, which has as its central tenet the aim of reducing insecticide inputs, is the easiest and most pragmatic step in reducing the pesticide burden on the environment. IPM is usually competitive with conventional agriculture in terms of pest management results, costs and yields but is not, unfortunately, any easier to carry out. Farmers tend to adopt IPM practices because of a personal commitment to less ecologically-costly farming methods; or, more realistically, because of legislation, pesticide availability and financial disincentives (e.g., pollution taxes). In the UK, 6 years of field work (Young et al., 2001) demonstrated that a decrease in pesticide use in conventional arable crops is often practicable. In a comparison of 66 different crops, the average profit margin of lower input regimes was 2% (£12/ha) greater than for standard, higher input strategies.

IPM necessitates that the farmer can identify pest species and can appreciate that there is a numerical threshold for those insects below which yields are unlikely to be affected. It does not, as is sometimes implied, "require farmers to be para-taxonomists and ecologists" (Kaosa-ard and Rerkasem, 2000: 141). IPM normally demands, particularly in resource poor settings, only the time and the inclination to adapt to slightly more complicated decision tools; usually based on an understanding of pest threshold levels. At its most simplistic, IPM is the removal of unnecessary prophylactic insecticide applications that in turn allows unquantified benefits from the increased impacts of natural enemies.

Such simple adoptions of IPM have proved extremely successful even (or perhaps particularly) amongst the poorest of farming communities. In India, 45,000 farmers in 465 villages were recruited to a program that coached and subsidized demonstration farmers through a more complex set of decision tools for pesticide application (usually through awareness of pest thresholds below which spraying was deemed unnecessary). Once increased profits were demonstrated, other farmers in those villages followed suit. A similar scheme in Uganda, initially involving 6,000 cotton farmers and all processors who separate the fiber from the waste material, called "ginners," was expected to cover all growers by 2007 (Luseesa et al., 2003; Russell, 2004).

It is worth noting that, despite the popularity of the IPM concept (reviewed by Kogan, 1998) there has been no decrease in overall insecticide usage, even in areas where that concept is very favorably viewed (e.g., the UK and California). If the success of the IPM concept is judged by reductions in the area of land sprayed by insecticides, then it has clearly failed. Perhaps though, it is a triumph simply to have kept insecticide use static during a period of increasing agricultural intensification.

The major new pest management technology that is already making an impact upon the way that insecticides are targeted is genetically modified (GM) crops, some of which are now engineered to express Bt δ (delta) endotoxins. These toxins are generally only active against one group of herbivores; therefore, more specific than many synthetic insecticides. Their efficacy against target pests, however, rivals that of the synthetics. One of the aims of the development of insect-resistant GM plants is to provide a more targeted and sustainable means of pest control. GM cotton and maize are now grown commercially and there is widespread uptake of GM crops globally – not just among developed nations, but also where GM products have been adapted (sometimes pirated) and developed for use in developing countries (most notably China and India). In 2003, 29% of the maize and 41% of the US's upland cotton crop used Bt varieties. This latter crop is grown in Australia, China, India and the Philippines; farmers who use Bt varieties commonly cite reduced labor costs, insecticide use and/or increased yields as the major benefits (Halford, 2004; Ismael et al., 2002; Qaim and Zilberman, 2003).

The major risks associated with Bt-expressing crops include the possibility of target pests evolving resistance to the expressed toxins; their potential invasiveness; and the spreading of insect resistant genes in the environment, thereby conferring such characteristics to weeds or wild relatives (e.g., flow between maize and its wild relative *teosinte* in Mexico; Amman, 2001). The possibility also exists for cross-pollination between GM and non-GM crops of the same species. This happened in 1998 when a GM variety of maize, approved only as animal feed in North America, cross-pollinated with maize crops being grown for human consumption. The contamination of the latter crop resulted in the entire crop being bought back by the seed company in question (Halford, 2004). The risk of losing biodiversity and consumer choice through such events is clearly a real one.

The effects of insect resistant GM crops on non-target arthropods, especially arthropod natural enemies of insect pests, have been studied extensively during the last decade. Like insecticides, GM plants may exert direct or indirect effects on a variety of non-target species. Natural enemy species might be affected by changes in prey or host quantity or quality. Other non-targets might be exposed when they consume prey or hosts containing GM-based plant material or by feeding on GM-affected pollen or honeydew. The ecology of the species at risk will determine their actual exposure. For example, in Bt maize the endotoxin is expressed in the leaves but not in the phloem. Moth larvae and spider mites can consume the Bt toxin, but phloem-feeding aphids do not (Dutton et al., 2002). The organisms feeding on aphids or on their honeydew are therefore less likely to be exposed to Bt toxin than those feeding on moth larvae or spider mites.

The majority of studies have not found any profoundly negative effects of GM plants on arthropod natural enemies (O'Callaghan et al., 2005), particularly when compared with conventional pest control measures, which rely on broad-spectrum insecticides. The negative effects that have been reported have generally been subtle and rather difficult to predict. For example, mortality among lacewing larvae (*Chrysoperla carnea*) increased when they were reared on moth larvae (*Spodoptera littoralis*) that had fed on maize engineered to express Bt toxins (Dutton et al., 2002; Hilbeck et al., 1998). However, when lacewing larvae were fed on spider mites that had been reared on such maize, no detrimental effects were noted. Choice tests using prey items reared on GM maize showed that lacewing larvae preferred other prey to moths and that in the field this might reduce exposure to Bt toxins (Meier and Hilbeck, 2001). Similarly, a parasitic wasp of the diamondback moth (*Plutella xylostella*) is more attracted to unmodified oilseed rape damaged by Bt-susceptible caterpillars than to Bt oilseed rape less affected by such larvae (Schuler et al. 1999, 2003). The behavior of non-target insects thus clearly affects their risk of exposure to any potentially hazardous toxins expressed by GM plants.

Ostensibly, crops manipulated to express insecticidal toxins should remove many environmental problems associated with non-directed insecticide use; moreover, the proteins involved are generally accepted to present no discernable human health risks (GM Science Review Panel, 2004; Konig et al., 2004; Lack, 2002). Neither do Bt crops pose a significant threat to non-target arthropod abundance or diversity, especially when compared with conventional crops and even when the potential risks have been examined and discussed in minute detail (e.g., the Monarch butterfly *Danaus plexippus* story; Losey et al., 1999; also Gray, 2004; Sears et al., 2001).

Implications and recommendations

Given the contentious and dogmatic nature of the debate over insecticide use, it is unsurprising that its defendants and opponents have become polarized. Among the latter there exists a common and insidious belief that insecticide use is always undesirable. There is little acknowledgment of the fact that minimal synthetic input or organic forms of agriculture are only possible on a limited scale and for some crops. However attractive the idea, such agricultural systems will not meet global needs and the need for intensive arable production systems remains. Neither is there much recognition of the fact that, at least in the developed world, the ability to predict the ecological risk posed by insecticides is improving. This, in addition to greater environmental awareness, is undoubtedly leading to better decision-making on registration issues and to improvements in the health and environmental safety profile of the insecticides that make it to the modern market.

World agriculture produces more calories per person today than it did 30 years ago, despite a 70% population increase (FAO, 2002). Globally there is enough food produced to banish world hunger but agricultural products are sold for profit, hungry people cannot afford what is produced, and Europe and North America consume more than their fair share (see UNECE, 2003). The implementation of a system that would allow the just distribution of resources would necessitate the global rejection of the current free-market neo-liberal economic model. Until this happens, short-falls in production will continue to affect many parts of the world. The rural poor account for 80% of the world's 800 million hungry people. Most of these people are dependent upon agriculture and increases in arable production and profitability can therefore have an immediate impact on poverty. For example, Peru experienced a 70% reduction in the prevalence of hunger during the 1990s that was driven by diversification into agricultural exports that increased farm incomes and that created "value added" jobs in processing and canning (i.e., the agricultural sector moved away from overproduced low input staples such as maize and potatoes) (FAO, 2005). During this time, despite the fact that it was a net food-importing country with little food security, Peru also protected its domestic agricultural market by imposing punitive import tariffs (FAO, 2000). It is no coincidence that, over this period of export growth, agrochemical sales exploded (one estimate suggested a 27% increase between 1993 and 1994 alone; PANNA, 1996).

If the best we can hope for is to optimize insecticide inputs in the face of highly unpredictable pest problems, weather patterns and farm profits, we need to ensure that the pesticides that we use are the safest available and are used as sparingly as possible. The ecological impacts of high-input agricultural systems are severe. There is no doubt that insecticides contribute to that damage through their direct toxicity to non-target species and by removing prey organisms that would otherwise be available to animals higher in the food chain. The problem is particularly marked for older, broad-spectrum insecticides and when insecticides are used as major prophylactic pest management tools rather than as components of a suite of control measures. Where ecological harm has occurred, systems can recover if insecticide application is halted but there remains in common use a slew of dangerous, outdated and persistent insecticides. The developing world in particular remains awash in the most environmentally damaging of them.

Despite this, the agrochemical industry remains disingenuous in the promotion of their products. Croplife International (2007) which represents the agrochemical industry, is currently lobbying to remove pesticide tariffs in order to improve "farmers' access to the tools they need to deal with adverse effects caused by weeds, diseases and pests. This is of particular concern for farming economies in developing countries, where these pressures are often much greater". It is hardly contentious to suggest that their more pressing agenda is one of company profit. Moreover, most governments continue to shy away from more punitive anti-pesticide legislation – partly because of the effort and cost that the imposition of measures such as pesticide taxes would entail, and partly because of the industrial and farming lobbies that threaten the legislators' electoral lifespan. Additionally, current political thinking in the developed world tends to overstress the ability of the free-market to solve environmental problems without the need for legislation (Greenhalgh, 2005).

Farming communities and agrochemical companies searching for profits in a competitive market cannot be relied upon, or even expected to, self-regulate. As a consequence of that reality, restraint in pesticide use will only be achieved by imposed regulation and by disincentive schemes. For example, the state of California levies an additional 2.1% tax on all pesticides and uses the revenue to fund pesticide-related environmental programs (CDPR, 2006). The income generated is low however (farm expenditure on pesticides is only about 5% of agricultural production costs) and although it helps fund pesticide-related environmental programs, it has had little effect on decreasing the arable acreage treated. Moreover, the tax does not discriminate between pesticides with differential environmental or health impacts. We believe that, ideally, taxes need to be targeted against the riskiest chemicals. Such ranking requires the production of robust and replicable ecotoxicology data, collected in globally standardized forms and feeding into universally recognizable assessment packages. Despite the vagaries of assessment methodologies, it is now possible to reach popular consensus on the worst offending insecticides and to institute their demise on a much greater scale than is currently being implemented.

As agriculture in the developed world becomes increasingly unsupported and uncompetitive, the burden of pesticide use will shift to less-developed countries. These will produce an increasing proportion of the world's food and fiber staples in highly competitive markets in return for hard currency. The patterns and impacts of insecticide use, in agriculturally dependent countries that do not have the resources to support regulatory control are clearly of enormous concern, yet most risk assessments are targeted at temperate region ecosystems pests and crops. Neither do the poorer nations tend to impose bans or set conditions on pesticide use. Individual farmers are left as the sole decision makers despite their limited access to accurate non-partisan information or advice. If food production, the alleviation of hunger, and the protection of the ecosystem are global responsibilities, then countries in the developed world must assume a greater role in supporting the safe use and application of pesticides globally.

Although highly desirable, universal treaties on pesticides take an age to implement. The process to establish the Stockholm Convention on persistent organic pollutants (e.g., DDT, aldrin, dieldrin) began in 1995, decades after the need to control them was recognized. It took a further 6 years to become fully ratified. The current EC Pesticide Authorizations Directive (91/414/EEC) came into force in 1993 and aimed to secure greater harmonization of the pesticide products approved by the different European Member States. The process of reviewing 865 compounds across those countries was so necessarily complex that it is not due to be completed until 2008. Whilst we await the arrival of such pan-national initiatives, the developed world is duty-bound to export its hard-won expertise in assessment and regulation to the developing world. Poorer, agriculturally dependent countries are in desperate need of technical and legislative capacitation on issues of pesticide risk and hazard, and of the financial resources necessary for its implementation. Without a more global effort in this area, we are collectively doomed to the continued repetition of insecticide overuse and mismanagement and to the steady and incremental destruction of our global environment.

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