

**SESSION 7B**

**ECOLOGICAL IMPLICATIONS  
OF THE LONG-TERM USE OF  
PESTICIDES**

**CHAIRMAN**      **PROFESSOR H. F. van EMDEN**

**SESSION**  
**ORGANISER**      **DR V. K. BROWN**

**INVITED PAPERS**      **7B-1 to 7B-5**

THE ECOLOGY OF ARABLE FARMLAND

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ABSTRACT

Arable farmland is characterised by periodic violent disruptions to its flora and fauna. This is most obvious in the crops themselves, as cultivation, pest control and harvesting, each has a dramatic effect; but the same is often true of the semi-natural habitats surrounding crops. Such areas usually have important agricultural functions and so require periodic management. Animals and plants respond to these changes by either surviving in protected places or by movement. The former is usual in plants, but animals often depend on their mobility to survive. The spatial dynamics of organisms inhabiting farmland is poorly understood, but it is fundamental to the interpretation of any long-term effects of pesticide use.

In order to understand and, if possible, to predict the ecological implications of the long-term use of pesticides, one has to take into account the basic ecology of wildlife on arable farmland. Unfortunately, most of our knowledge in Ecology results from studies of more natural systems, and apart from research on pests, and those organisms having an impact on pest populations, rather little is known about the ecology of arable farmland. This is a major gap in our understanding when one realises that this form of land use occupies 80 % of lowland Britain. Indeed, the current Joint Agriculture and the Environment Programme, funded by AFRC, ESRC and NERC, results from a recognition of this gap by the Research Councils.

Unlike more natural systems, arable farmland undergoes periodic violent disruptions to its flora and fauna, as a direct result of crop husbandry. This is most marked in the areas covered by crops, as cultivation and harvesting have sudden and dramatic effects on those species living in the fields, by modifying the microclimate and by reducing the availability of food and shelter for many species. Thus, irrespective of the use of pesticides, there are periodic changes in the crop that must be tolerated by organisms inhabiting it.

To a lesser extent, the same sorts of problems face those species inhabiting the semi-natural habitats surrounding crops. Such areas usually have important agricultural roles associated with boundaries ( hedges and fence lines ), access ( headlands and tracks ) or drainage ( ditches and streams ). As such, they require periodic management if they are not to become overgrown and to lose their function. Even a simple process such as cutting can change the species composition of vegetation, and have a marked effect on those animal species dependent upon it ( Morris, 1979, 1981; Morris & Lakhani, 1979; Morris & Plant, 1983 ).

Animals and plants inhabiting farmland generally cope with these sudden changes in their habitats by either surviving in protected places, or by movement. The former is most usual in plants, which persist as perennial parts or seed, but animals frequently depend upon their mobility to survive. Obviously, the timing of the disruption will be very important in those species with immobile stages in their life cycle, as with most insects, and local extinction of their populations can result from management at the wrong time. By comparison, vertebrate animals, such as birds and mammals, are likely to be far less affected.

Arable farmland is also characterised by small areas of semi-natural habitats separated by extensive areas of crop. Compared to more natural habitats, farmland habitats are highly fragmented and patchy, and they tend to be separated by areas that are inimical to wildlife. Added to this, these patches tend to be of very unnatural shapes. Apart from woodlands, most semi-natural habitats occur in narrow, linear strips around the edges of fields, and the implications of this to the survival and dispersal of plants and animals is unknown.

Much of our thinking about the spatial dynamics of species inhabiting patchy environments has been influenced by the classical theory of Island Biogeography of MacArthur & Wilson (1967). They proposed an equilibrium model to explain the diversity of biota inhabiting islands, in which species diversity was a balance between immigration and extinction rates. They further proposed that immigration depended on the extent of isolation of the island (distance from the mainland), whilst extinction rates were a function of the size of the island, it being higher on small islands than on large ones.

Application of this theory to terrestrial habitats has had mixed success. As a general rule, the number of species present is related to the area of the habitat, but this relationship is not a simple one. Not all patches of habitat are equal,

because populations of different species depend upon one another, as well as on the physical conditions in the habitat. Thus an insect parasitoid can be present only if its host is both present and sufficiently abundant. A break in any food chain will result in the loss of species at the higher trophic levels, and so area *per se* will be a poor measure of habitat suitability. We need to know the carrying capacity of the habitat for the species in question, that is the maximum number that can be supported throughout their lives.

Similarly, distance to the next patch of habitat may be a poor measure of isolation. Of course, MacArthur & Wilson measured isolation as the distance of the island from the mainland, whereas farmland ecology does not have the equivalent to a mainland, but is concerned with distances between patches. In this case, we need to know the distance from a source of immigrants, but even this may be far from straightforward. Movement of many species is dependent upon movements of air, water, or some animal carrier, and is non-random in direction. Movement may also be density-dependent, so that the degree of isolation of a patch of habitat may vary between years, as populations rise and fall.

Mobility is such an obvious attribute of most animals that one might have supposed that its role in population ecology would be well known. This is not so, however; largely, I suspect, because it is so difficult to study. Most field studies of animal populations either totally ignore movement, or assume that immigration and emigration balance one another out. The same is true of current ecological theory ( May, 1976 ) which considers that the persistence of animal populations is primarily the result of regulation through density-dependent processes, *i.e.*, a function of the internal dynamics of the population rather than of periodic recruitment from outside. Such regulatory processes are perhaps more relevant to stable natural systems than to highly changeable agricultural habitats, but even in more natural systems they have proved very difficult to demonstrate in field populations ( Dempster, 1983; den Boer, 1986, 1987 ). There is, on the other hand, some evidence to suggest that recruitment from immigration makes an important contribution to population persistence in some species ( Connor *et al.*, 1983; Dempster, 1989 ). Mobility is, I suspect, the key to survival of most animal species inhabiting arable farmland, but its role is very difficult to quantify in all animals, vertebrates and invertebrates, alike.

In the past, studies of animal dispersal have relied upon mark and recapture techniques. For these, individuals are given a mark and released into a particular area. After a period of time, the surrounding country is searched

for their presence. The likelihood of recapture depends on the density of marked individuals and as one moves away from the point of release, this is reduced by the increased area. One has to make judgements as to how far individuals might move to determine the size of the area to be searched, but the point is soon reached when the area becomes too large to be searched adequately. As a result, such studies are biased towards finding most marked individuals close to the point of release, so underestimating movement. Added to this, one has to be certain that marked individuals behave normally, and there is evidence to show that capture, handling and marking have an effect on the frequency of recapture of small organisms, such as insects ( Singer & Wedlake, 1981; Morton, 1984 ).

To understand the role of movement in the population dynamics of an organism, one needs to quantify emigration and immigration. Mark and recapture may give some information on emigration, but it will rarely give any measure of immigration, unless the whole population can be marked, and unmarked individuals can be identified as immigrants. This is, perhaps, feasible with the use of techniques which transfer a mark to an animal via its food plants, as for example, with the marking of phytophagous insects by a rare element, such as rubidium, by its application and absorption by plants ( Frazer & Raworth, 1974; Alverson *et al.*, 1980; Wolfenbarger *et al.*, 1982; Padgham *et al.*, 1984 ), but in my opinion, the best hope of quantifying movement rests with the study of natural population markers.

The chemistry of individual animals depends upon their genetic make-up and upon the chemical environment in which they live. Research looking for genetic markers has concentrated mainly on isoenzymes, but current research on genetic finger-printing makes this a promising and largely untapped field of study. Research on the chemical composition of animals has shown that, in some cases, this can be related to the locality in which they were reared (McLean, & Bennett, 1978; Turnock *et al.*, 1979; Bowden *et al.*, 1979; Dempster *et al.* 1986 ). As another alternative approach to this problem, I am currently looking into the possibility of using the presence of the mustard oils from different species of Cruciferae, and their metabolites, in the orange-tip butterfly, to measure immigration into an area where *Cardamine pratense* is its only foodplant. The use of natural markers of this sort might overcome many of the problems associated with mark and recapture, but we are still some way from having a satisfactory method for quantifying movement in any animal.

Whilst our understanding of the spatial dynamics of different types of organism on arable farmland is very limited, there is little doubt that the ability of species to recolonise areas after local elimination is an important aspect of their biology, affecting long-term impacts of pesticide use, at least in the case of non-persistent pesticides. Clearly, this depends upon their innate mobility, the frequency and extent of pesticide applications, and the availability of refuges within the range of movement of the species. The spatial and temporal scales of any study of the impacts of pesticide use must take these features into account. Unrealistically small experimental plots can result in short-term effects being swamped by rapid recolonisation, so giving a false impression of lack of impact. On the other hand, long-term effects are likely to result from either large areas being sprayed or from repeated applications of the pesticide preventing recovery of the flora and fauna.

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## LONG-TERM ECOLOGICAL EFFECTS OF HERBICIDES: FIELD STUDIES

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## ABSTRACT

Herbicides are not very toxic to fish, birds or mammals but they are toxic to some soil or aquatic invertebrates. They can kill organisms which degrade organic matter in soils and therefore the rate at which it is broken down to release plant nutrients. They have considerable influence on the incidence of damage by arthropod pests and plant pathogens by; decreasing the numbers of alternative hosts, removing shelter for parasites, predators and antagonists and decreasing their diversity. Herbicides could percolate through earthworm burrows and contaminate groundwater in direct drilled fields.

Examples of each of these effects are given based on the results of field experiments. These summarize the toxicity of the herbicide cyanazine to soil organisms such as springtails; the indirect effects of the herbicide pyrazone on the incidence of pests and their predators in sugar beet fields; the effects of paraquat and simazine on the rate of breakdown of organic matter in soil; and the influence of earthworm burrows on the contamination of groundwater by alachlor and atrazine.

## INTRODUCTION

Herbicides are used in extremely large quantities and account for 85% of all the U.S. pesticide use (Edwards and Regnier, 1989) and 45% of world pesticide use (Edwards, 1986). Since they have low toxicity to fish, birds and mammals, it is usually assumed that their environmental effects are small (Edwards, 1991). However, they have major effects on soil organisms and soil processes in agricultural fields, since they kill all the plants other than the crop, thereby leaving the soil bare and with little ground cover. This in turn influences the amount of organic matter that reaches soil (Figure 1). Some herbicides are toxic to soil organisms (Edwards and Thompson, 1973; Edwards, 1989) which are important in breaking down soil organic matter (Edwards, 1988).

Weeds provide alternative hosts for many insect pests and plant pathogens. Moreover, they provide shelter for parasites or predators of pests and act as hosts for antagonists of plant diseases. When weeds are controlled by herbicides, some



of these natural control mechanisms for pests are decreased, and insect pest and plant disease attacks increase.

Herbicides such as paraquat and diquat have made the practice of direct drilling (no till) cropping possible by killing weeds and sowing a crop without cultivations. However, it has been suggested that the improved drainage, resulting from larger earthworm populations and the absence of cultivations may cause contamination of ground water by herbicides. This paper describes practical field examples assessing various ecological effects in field experiments.

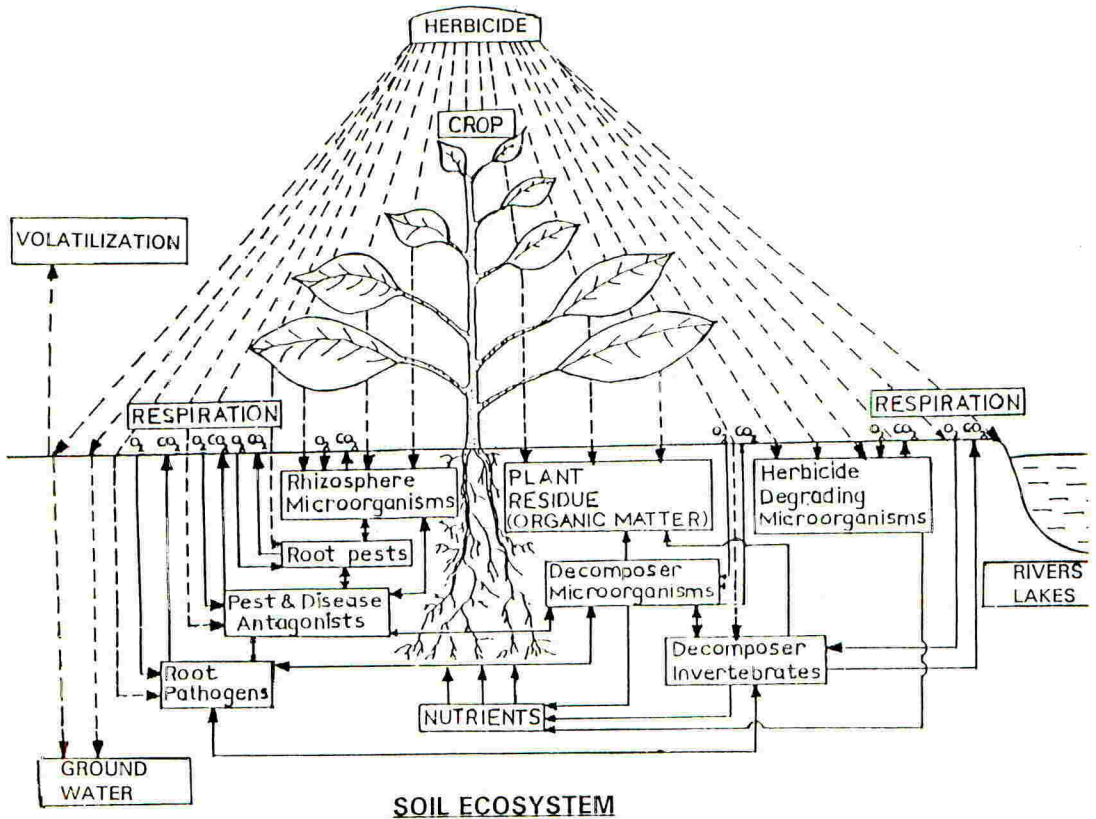


Figure 1. Ecological impacts of herbicides (from: Edwards, 1989)

### EFFECTS OF HERBICIDES ON SOIL ORGANISMS

There is an extensive literature on the effects of pesticides on microorganisms, which is based largely on laboratory experiments (Edwards, 1989). In these, and in the field experiments reported to date, there appear to be no long term effects of herbicides on soil microorganisms, other than when extremely large doses of herbicides are used. Most herbicides have no direct toxicity to soil-inhabiting invertebrates, other than the triazine herbicides which are moderately toxic to many

soil-inhabiting invertebrates (Edwards and Stafford, 1979). It is difficult to assess the direct toxicity of herbicides to soil organisms in field experiments (Domsch, 1983), because their populations are influenced not only by the direct effects of the herbicide, but also by the indirect effects of the weeds. Weeds die and provide organic matter in soil upon which soil organisms grow. The only way in which the direct toxicity of a herbicide to soil organisms can be assessed in the field is by comparing populations of soil organisms in herbicide-treated plots with those in plots kept weed-free by hand-weeding. This was demonstrated in a field experiment where a triazine herbicide, cyanazine, was applied to replicated plots at a rate of 2 kg AI/ha in April. Populations of soil-inhabiting invertebrates were assessed by periodically taking soil cores from treated plots control plots with weeds and control plots that had been hand-weeded. The invertebrates were extracted in high-gradient Tullgren funnels and then counted. The effects of cyanazine on springtails (Collembola) are summarized in Figure 2. Populations were decreased by hand-weeding but the direct toxicity of cyanazine can be assessed by comparing populations in treated plots with those in weed-free control plots.

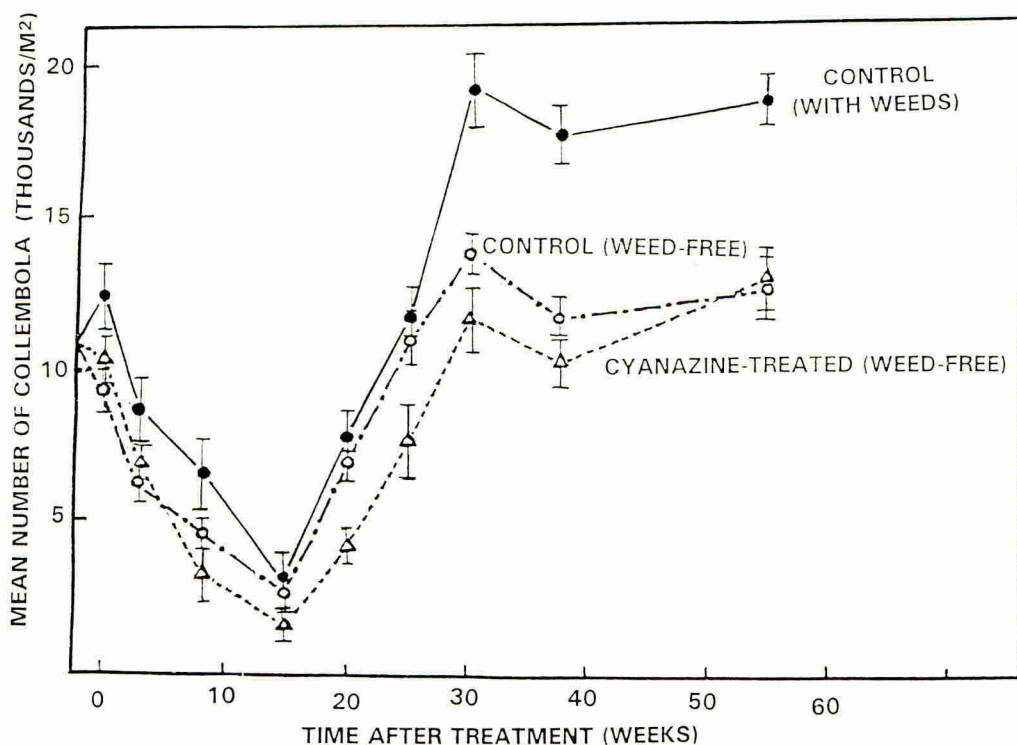


Figure 2. Effects of cyanazine on populations of Collembola

## EFFECTS OF HERBICIDES ON ORGANIC MATTER BREAKDOWN

The breakdown of organic matter in soil and the release of the nutrients it contains is mediated by the activities of the soil fauna and microflora, which physically fragment it and progressively decompose it. The effects of a herbicide on the rate of breakdown of organic matter in soil provides an excellent overall index of the effects of a herbicide on this dynamic soil process which is essential to soil fertility (Edwards, 1988).

The method used was based on cutting 5 cm diameter disks of filter paper or deciduous tree leaves, e.g. oak, beech or elm, with a cork borer. Batches of fifty such disks were enclosed in nylon mesh bags of 5 mm aperture mesh (Edwards and Heath, 1963) and eight bags were buried in pesticide-treated soil in the field to a depth of 5 cm and dug up at regular intervals for assessment of the extent of breakdown. Losses due to invertebrates were assessed by washing disks and measuring their surface area by light transmission, after laying them out on a glass plate or on a moving translucent band. The amount of light passing through them was measured on a photoelectric cell and the data recorded. Each measurement was compared with that from an intact set of 50 disks. After each measurement, disks were replaced in the mesh bag and reburied (Edwards, 1988). The effects of paraquat and simazine, applied at recommended doses to field plots, on the breakdown of organic matter using this method are summarized in Figure 3. Clearly, paraquat accelerated this process slightly but simazine decreased the rate of organic matter breakdown very considerably.

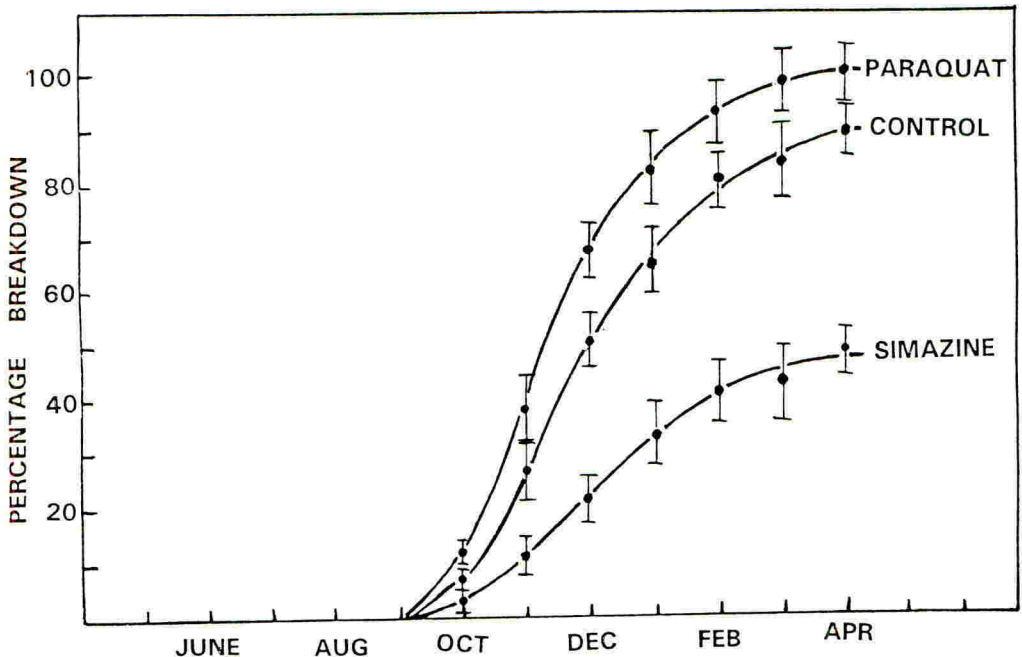


Figure 3. Effects of paraquat and simazine on organic matter breakdown (based on 8 mesh bags/treatment)

## EFFECTS OF HERBICIDES ON THE INCIDENCE OF PESTS

There is good evidence that weeds are important in influencing the incidence of insect pests, by providing alternative hosts and increasing the diversity of natural enemies (Pimentel, 1991). However, there are not many demonstrations of such interactions in the field. In an experiment which investigated the effects of pesticides on the soil-inhabiting fauna of sugar beet fields, Thornhill and Edwards (1985) investigated the effects of pyrazone applied, pre-emergence at 2.8 kg AI/ha, to replicated field plots, on the incidence of predatory beetles and on populations of various sugar beet pests. The indirect effects of the herbicide on both predators and pests were assessed using pitfall traps. Significantly fewer flea beetles of the species *Chaetocnema concinna* were trapped in the herbicide-treated plots, than in control plots, in two successive years (Figure 4). This could have been due to greater activity of predatory beetles such as *Bembidion lampros* (Herbst) or loss of the alternative host *Polygonum convolvulus* L.

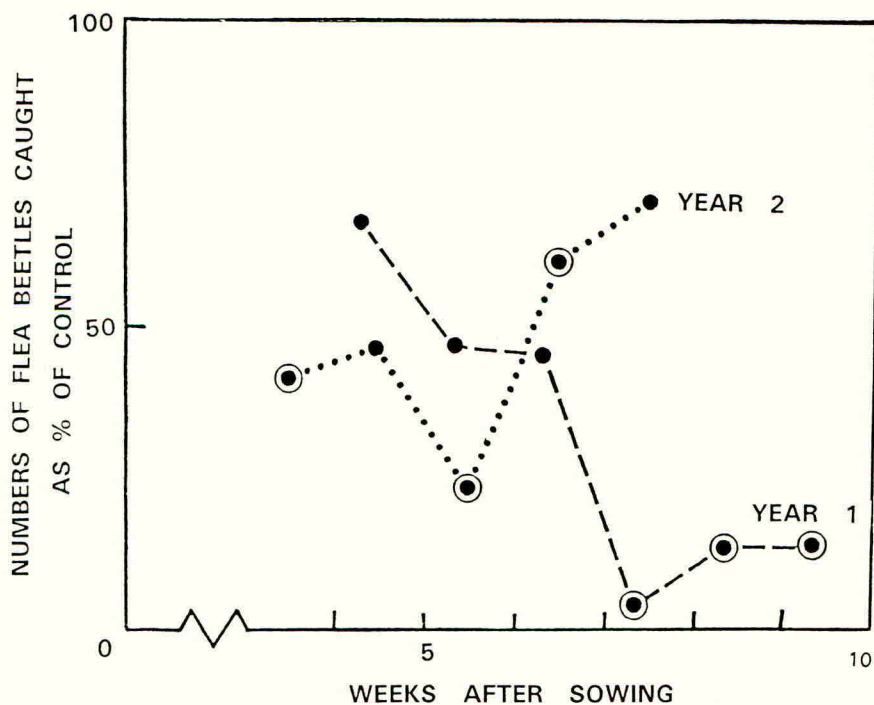


Figure 4. Indirect effects of pyrazone on flea beetle populations (⊙) Significantly different from no-herbicide treatment  $P=0.05$

## EFFECTS OF HERBICIDES ON GROUNDWATER POLLUTION

There have been many reports of contamination of ground-water by the more soluble herbicides, such as atrazine and alachlor, sometimes to levels above the designated human safety levels. There has been considerable discussion as to whether a major source of this contamination is the increasing adoption of direct drilling or no till practices, that substitute the use of herbicides for cultivations. Such practices not only increase herbicide use but also encourage the formation of undisturbed cracks and crevices and promote greatly increased earthworm populations. These create burrows through which water and herbicides in solution percolate (Edwards and Lofty, 1982). Although there has been considerable speculation as to how this affects groundwater contamination, there are not many field experiments that have assessed this problem quantitatively.

An experiment by Edwards, W.M. et al (1991) assessed the movement of herbicides down earthworm burrows. Sample bottles were installed to collect water flowing to the 45 cm depth, in *L. terrestris* burrows, by excavating laterally into the wall of a pit to intercept burrows at the desired depth. A 125 ml plastic bottle was placed on the soil shelf beneath each burrow with a short section of polyethylene tubing leading from the burrow directly into the bottle. The upper end of this tube was tapered using a pencil sharpener to facilitate inserting it tightly into the burrow, thereby insuring that all water flowing in the burrow at that depth would enter the

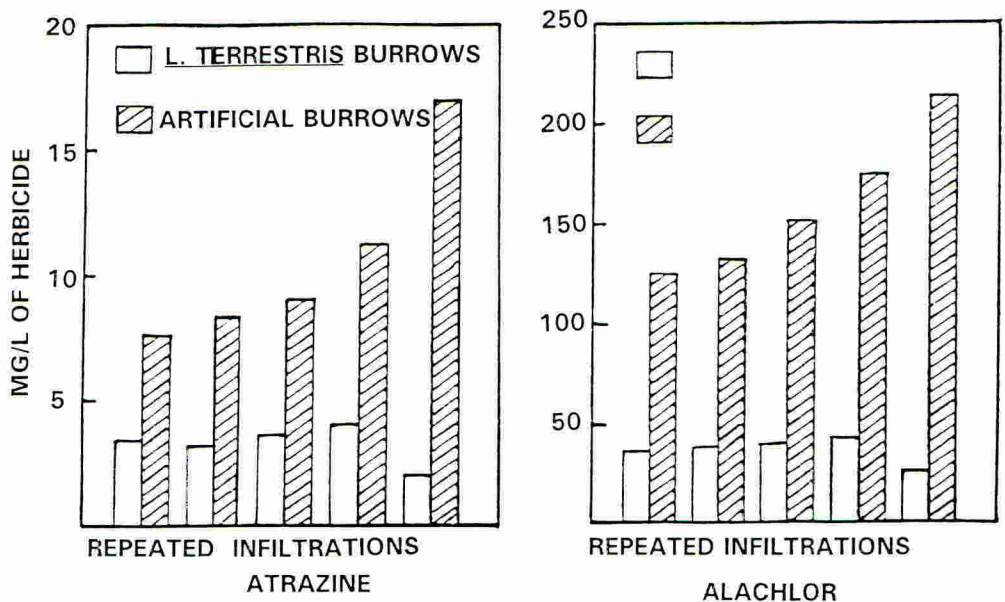


Figure 5. Infiltration of alachlor and atrazine down ten earthworm burrows and artificial channels (Input concentrations 29.4 mg/l atrazine; 223.0 mg/l alachlor)

bottle. A longer tube led from the bottle to the soil surface serving as an access port to the buried sampler both tubes passed through holes in the bottle lid that were sealed with silicon caulking.

The bottle was held in place by moist soil packed firmly around and under the bottle to insure that the sample tube would remain securely in place under the earthworm burrow. After instrumenting burrows with samplers, the pit was refilled. Following each subsequent rainfall event, the contents of the samplers were collected by inserting a slender extraction tube into each bottle through the above-ground access port and pumping the contents into individual containers, for volume measurement and transport to the laboratory. Additional details of sampler installation are presented by Edwards, W.M. et al. (1989).

The infiltration of atrazine and alachlor was assessed by pouring 50 ml aliquots of the herbicides into the surface openings of newly-abandoned burrows and adjacent artificially-made holes of the same diameter. The liquid passing down the burrow was collected and analyzed by passing them down a solid phase extraction column and then analyzing on a Varian Model 3500 Capillary gas chromatograph.

The results are summarized in Figure 5. Clearly the amounts of herbicide passing down the earthworm burrows were much less than those moving down the comparable artificial holes which passed more herbicides with successive infiltrates. Probably the herbicides were adsorbed onto the linings of the earthworm burrows which were high in organic matter and very active microbially but not onto those of the artificial burrows.

## CONCLUSIONS

These four field experiments differ in approach and execution but they serve to show that herbicides can have extremely diverse ecological effects that are often so complex that they can be very difficult to assess. Clearly, all of the experiments demonstrate adverse ecological and environmental effects of herbicides that could not be investigated or predicted by laboratory experiments. They show how difficult it is to assess the effects of individual agricultural practices such as herbicides on agroecosystems, because any single input can impact upon many organisms or processes. There is a need for well-defined protocols for field experiments that would provide data on the ecological and environmental impact of pesticides. Such data is essential for the development of integrated pest management (IPM) programs (Edwards et al, 1991) and sustainable agriculture systems (Edwards et al, 1990).

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## A STUDY OF REPEAT APPLICATIONS OF MECOPROP TO PLANT COMMUNITIES IN MICROCOSMS

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## ABSTRACT

There has been an increasing awareness of potential impacts of herbicide drift on vegetation in nature reserves and field margin habitats adjacent to treated areas. Previous work has suggested safe buffer zones should be between 6-10m downwind of the sprayer. In the present study, eight native dicotyledonous species, with and without a perennial grass (*Lolium perenne*), were grown in standardized microcosms and exposed up to 8m downwind of mecoprop drift in each of 3 years. The aim was to investigate the likely responses of vegetation in this buffer zone.

*Stachys sylvatica* and *Lolium perenne* (where sown) increased near the sprayer, but *Digitalis purpurea*, *Galium mollugo*, *Hypericum hirsutum*, *Lychnis flos-cuculi*, *Primula veris* and *Ranunculus acris* showed a reduction in either performance or yield in at least one year. *L. flos-cuculi* showed a reduced seed yield in the third year of the experiment, but neither *L. flos-cuculi* nor *R. acris*, the 2 species which produced sufficient seed for testing, showed any effect on seed viability. The implications of these results for the persistence of attractive plant communities in sensitive areas are discussed.

## INTRODUCTION

Herbicide drift on to field boundaries and other sensitive communities from sprayed agricultural land has often been considered by conservationists to be one of the main reasons for their depauperate flora. Surprisingly, there have been few attempts to investigate this problem. In a previous study (Marrs *et al.*, 1989a,b) we showed, using single species bioassays downwind of a sprayer, that most impacts were likely to be found < 10m downwind. These experiments took no account of either differential interception or inter-specific competition between species following repeat applications. In a subsequent experiment Marrs *et al.* (1990a) used artificially created microcosms to investigate both of these aspects downwind of mecoprop applications in each of two seasons. This paper reports additional information for the third year of treatment including a detailed assessment of the effects of mecoprop drift on flowering, seed production and seed viability. Significant effects on these characteristics could translate into changes in plant community composition or structure. Mecoprop is a widely used herbicide for the control of



broad-leaved weeds (Roberts, 1982), as grasses are generally resistant. Here, we applied it under conditions which simulate its use in practice.

The microcosm approach was chosen as a hybrid between field experiments and classic competition experiments. Field experiments of herbicide impact on semi-natural vegetation (Yemm & Willis, 1962; Parr & Way, 1984; Marrs, 1985; Marshall, 1988; Willis, 1988) require treatment plots with relatively homogenous vegetation, especially if herbicide doses are low. Competition experiments (De Wit, 1969; Jolliffe *et al.*, 1984) have also proved useful for two species comparisons (Breeze *et al.*, 1990), but are difficult to apply to multi-species communities. Here, artificial communities were created in microcosms (*sensu* Grime *et al.*, 1987).

## METHODS

### Experimental procedure

In 1988, 60 experimental microcosms were created. Each microcosm consisted of a container (27 cm diameter x 12 cm deep) filled with SAI GP compost in which individuals of the following dicotyledous species were planted: *Digitalis purpurea*, *Filipendula ulmaria*, *Galium mollugo*, *Hypericum hirsutum*, *Lychnis flos-cuculi*, *Primula veris*, *Ranunculus acris* and *Stachys sylvatica* (nomenclature follows Clapham, Tutin & Moore, 1987). The plants were produced from seed sown in the autumn of the previous year and matched plants transferred into the microcosms when they had 4 expanded leaves. A standard arrangement was used with the taller species (when mature) positioned on the downwind side. Half of the microcosms were sown with the grass *Lolium perenne* Majestic at a rate equivalent to 20 kg seed ha<sup>-1</sup> in the April before first spray, to create an additional treatment comparison (grass *versus* no grass). At the time of first spraying the grass had reached a height of c. 6 cm. After the microcosms were produced, they were assigned randomly to treatment position, i.e. downwind of the sprayer or unsprayed control.

The microcosms were exposed to drift once in 1988, 1989 and 1990 using the same method each year. Five replicate microcosms with *Lolium perenne* and five without grass were placed at 0 m (i.e. receiving the full application rate from the outer 50 cm of the spray jet) and at 1 m, 2 m, 4 m and 8 m downwind from the sprayer. Four sequential upwind swathes from the 0 m position were sprayed in each experiment. Five microcosms with *Lolium perenne* and 5 without the grass remained unsprayed to act as controls. After exposure to the spray drift, the microcosms were placed in a glasshouse for 24 h and watered carefully to prevent herbicide being washed off, before transfer to an open-air plunge bed. Thereafter, plants were maintained using standard horticultural procedures. After the first year of the experiment all microcosms were given 10 g of fertilizer (Nitrophoska N:P<sub>2</sub>O<sub>5</sub>:K<sub>2</sub>O:MgO 12:12:17:2) each Spring.

Mecoprop was sprayed at 2.4 kg AI ha<sup>-1</sup> in June of each year with a tractor-mounted Team sprayer to reproduce field conditions. The sprayer had a 6 m boom, which was fitted with 12 Lurmark Red 03-F80 (BCPC code F110/1.20/3) fan nozzles, and held 80 cm above the vegetation surface. A tank pressure of 2 bar and a tractor speed of 6 km h<sup>-1</sup>, which distributed c. 200 litres solution ha<sup>-1</sup>, was used throughout. Wind speed and direction

were monitored with a R500 recording anemometer. In both years, the wind speed during spraying was between 2-3 m s<sup>-1</sup> at a height of 2 m.

### Recording

Recording was done in 5 ways:

- (1) In September of each year all individual dicotyledonous plants plus the entire herbage of Lolium perenne were harvested at ground level, dried at 80°C and weighed.
- (2) In early June 1989 before spraying the recovery of plants from the first application of mecoprop was assessed using a series of non-destructive measures of performance (e.g. leaf length, flower number).
- (3) During the summer of 1990 the numbers of flowers produced by each dicotyledonous species were counted weekly.
- (4) During the summer of 1990, seed was collected weekly from each dicotyledonous flower head as it was produced. The seeds were extracted, sieved and weighed.
- (5) Seed viability was assessed using standardized germination tests.

### Data analysis

The following assumptions were made; (1) that the data for each species in the unsprayed 'control' microcosms was the norm, and (2) that any deviation from the above data was an effect brought about directly or indirectly by exposure to herbicide drift in the zone downwind of the sprayer.

As the recording of the grass and no grass treatments were undertaken separately, data for each treatment were analyzed individually. Three separate analyses were done for each data set in each year:

- (1) A contrast between means at 8m and the untreated control. This was to assess whether there was any significant effect at the greatest test distance.
- (2) A contrast between means (expressed as a percentage of untreated values) at 0m and the pooled mean of 1,2,4 and 8m. This was to test whether plants given the full herbicide dose were reduced in performance relative to those in the downwind zone.
- (3) Regressions between the measure expressed as a percentage of untreated values and distance downwind; both linear and quadratic equations were fitted (n=25). Only significant equations are presented, and where both linear and quadratic equations were significant, the one which accounted for the greatest variation is discussed.

Data on seed viability was available for a limited number of individuals of 2 species (L. flos-cuculi and R. acris). Here, Kendall rank correlation coefficients were calculated between viability and distance downwind.

## RESULTS

### Analysis 1 (8m versus untreated)

Out of 131 possible contrasts only 7 were significant (Table 1), with 2 types of significant response found:

(1) Stimulation at 8m: yield in 1988 - D. purpurea S. sylvatica and performance in 1989 - H. hirsutum, all in the no grass treatment and R. acris in the grass treatment.

(2) Suppression at 8m: yield in 1988 - L. perenne where sown and yield in 1990 of F. ulmaria and P. veris in the no grass treatment.

No significant differences were found in the flowering performance or seed production in 1990.

### Analysis 2 (0m versus pooled mean 1,2,4,8m)

Out of 131 possible contrasts 27 proved significant (Table 2) separating species into 2 significant responses:

(1) Stimulation at 0m: F. ulmaria, H. hirsutum and S. sylvatica in the no grass treatment and L. perenne where sown.

(2) Suppression at 0m: G. mollugo and L. flos-cuculi showed reduced yield in 1989 and 1990 and reduced performance in 1989 in both grass treatments. P. veris showed a reduced performance in 1989, in the grass treatment.

No significant differences were found in flowering performance or seed production in 1990.

### Analysis 3: Relationship between performance and distance downwind

Only F. ulmaria showed no significant relationship with distance downwind from the sprayer. All other species produced at least one significant regression in one of the grass treatments in at least one year (yield, Table 3; other performance variables, Table 4). Two types of significant response were detected in either yield or performance in 1989:

(1) Species which showed an increase near the sprayer: L. perenne and S. sylvatica (grass only).

(2) Species which showed a reduction near the sprayer: D. purpurea, G. mollugo, H. hirsutum, L. flos-cuculi, P. veris, and R. acris.

No significant relationship was found for flowering performance in 1990.

#### Analysis 4: Effects of seed viability

Correlation coefficients between seed viability and distance downwind of the sprayer were not significant; *L. flos-cuculi*  $r_k = 0.07$  and *R. acris*  $r_k = -0.07$  and  $-0.25$  for the no grass and grass treatments respectively.

#### DISCUSSION

The aims of this work were to investigate the cumulative responses of mixtures of perennial species, typical of semi-natural communities, in the 0-8m zone downwind of sprayed areas. This was the distance suggested by Marrs *et al.* (1989a,b) as a suitable buffer zone for protecting sensitive vegetation. The microcosm approach used here is an artificial model, to investigate likely impacts.

The result that few species were affected at 8m downwind compared to untreated controls tends to confirm that buffer zones of this order of magnitude are reasonable. However, the ecological significance for the few species whose performance was reduced or stimulated at this distance requires further investigation. Stimulated growth downwind of sprayers has been noted in similar studies (Marrs *et al.*, 1990b). What is perhaps of more surprise is that so few species were adversely affected at the 0m position where they received a recommended dose of mecoprop. Clearly, many of these perennial species have considerable ability to recover from herbicide damage.

Herbicide spray drift affected species in different ways. First, there were often differential responses between the microcosms sown with *L. perenne* and those without. This effect may well be caused by differential deposition/interception of herbicide drift in the contrasting vegetation structures (Marrs *et al.*, 1990b). Second, differential effects were noted between species. Thus, the balance of species in sensitive habitats may shift depending on the amounts of drift deposited and their effects on different species. Third, the results from the non-destructive measures of performance show that species can be affected in the year following exposure to spray. Finally, some species appeared to recover during the second growing season even allowing for the fact that they received a second dose of herbicide. Exposure to spray drift may cause visual symptoms of damage and reduced performance early in the next growing season, but for some perennial species these effects are temporary as there is clearly compensatory growth later in the season. This compensatory growth may be a direct effect of exposure to low doses of a hormone type herbicide. A similar result has been noted in other studies of spray drift (Marrs *et al.*, 1989a,b, 1990b). Whether, this recovery can be maintained *ad infinitum* in response to continuous annual exposures remains to be determined. However, in the 1990 assessments of flowering performance and seed production, which were measured over the entire season, no single species showed a reduced flowering performance and only one a reduced seed production near the sprayer. Moreover, neither of the 2 species tested had a detectable reduction in seed viability in this experiment.

The increase in yield of *L. perenne* near the sprayer in the

microcosms where it was sown was expected, because grasses tend to be resistant to mecoprop (Roberts, 1982). Continuous exposure to drift of mecoprop, therefore, may well lead to a dominance of this perennial grass in the longer term. Increased dominance of grasses has been reported in several experiments where herbicides/growth retardants have been applied at recommended rates to semi-natural grasslands (Marshall, 1988; Parr & Way, 1984) and parallel the effects of foliar insecticide drift (e.g. Gange & Brown, this volume).

The results for this study appear to confirm the earlier suggestions that a buffer zone of 0-10m is adequate for the protection of field margin habitats and other sensitive plant communities, although within this zone there may be changes in both community composition, flowering and seed production. The implications of these results for community regeneration are being investigated by further experiments.

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TABLE 1. Species which showed a significant difference in yield and performance in microcosms exposed to mecoprop 8 m downwind of the sprayer and those untreated.

Measures	No. of contrasts	Type of response	Year	Species	Grass (+/-)	8m	Untreated	LSD (P<0.05)
Yield (ln g+1)	51	+ ve	88	<u>D. purpurea</u>	-	1.853	1.583	0.215
			88	<u>S. sylvatica</u>	-	2.717	2.266	0.413
	- ve	88	<u>L. perenne</u>	+	1.607	2.102	0.479	
		90	<u>F. ulmaria</u>	-	0.200	0.367	0.154	
		90	<u>P. veris</u>	-	0	0.064	0.061	
Performance	89	+ ve	89	<u>H. hirsutum</u>	-	8.2	3.0	5.02
			89	<u>R. acris</u>	+	19.8	6.6	11.06

Table 2. Species which showed a significant difference in microcosms when placed directly under a spray boom and exposed to mecoprop. Results are of the contrast between mean at 0 m (0) and the pooled mean of microcosms downwind (1,2,4,8 m), both expressed as a % of untreated. Significance \* =  $P < 0.05$ , \*\* =  $P < 0.01$ , \*\*\* =  $P < 0.001$ .

Measures (No. of contrasts)	Species	Year	Grass (+/-)	0m	Pooled mean (1,2,4,8m)	Significance
Yield (51)						
0 > rest	<u>F. ulmaria</u>	90	-	124	53	*
	<u>H. hirsutum</u>	89	-	4587	158	**
		90	-	164	5	**
	<u>L. perenne</u>	89	+	226	94	***
		90	+	199	95	**
0 < rest	<u>S. sylvatica</u>	89	-	472	194	**
	<u>G. mollugo</u>	89	+	11	110	**
		90	+	18	194	**
		88	-	30	71	*
		89	-	0.2	52	**
		90	-	0	19	**
	<u>L. flos-cuculi</u>	89	+	199	95	*
		90	+	63	132	*
		88	-	40	87	**
		89	-	21	128	***
		90	-	98	211	***
Performance 89 (48)						
0 > rest						
Stem height	<u>S. sylvatica</u>	89	-	137	71	**
0 < rest						
Shoot no.	<u>G. mollugo</u>	89	+	12	125	***
Shoot no.			-	0	95	***
Shoot length			-	0	105	***
Flower stem no.	<u>L. flos-cuculi</u>	89	-	0	92	***
Flower no.			-	0	78	***
Leaf no.	<u>P. veris</u>	89	+	18	99	***
Leaf length			+	24	118	***
Flower stem no.			+	0	77	*
Flower stem height			+	0	81	***
Flower no.			+	0	80	*

Table 3. Regression analysis between yield of species grown in microcosms and distance downwind of the sprayer; data expressed as a percentage of untreated controls. Significance \* =  $P < 0.05$ , \*\* =  $P < 0.01$ , \*\*\* =  $P < 0.001$ .

Species	Grass (+/-)	Year	Linear (l)/ Quadratic (q)	r <sup>2</sup>	F	Significance
<u>G. mollugo</u>	+	89	q	0.33	4.3	*
	+	90	q	0.38	5.1	*
<u>L. perenne</u>	+	89	q	0.46	7.4	**
	+	90	l	0.27	6.6	*
<u>L. flos-cuculi</u>	+	90	q	0.34	4.4	*
	-	88	l	0.22	5.2	*
	-	89	q	0.70	26.1	***
<u>P. veris</u>	+	88	q	0.41	5.8	*
	+	89	l	0.29	7.5	*
<u>R. acris</u>	+	90	l	0.45	14.5	*
<u>S. sylvatica</u>	+	89	l	0.42	13.2	**



Table 4. Regression analysis between performance in 1989 of species grown in microcosms and distance downwind of the sprayer; data expressed as a percentage of untreated controls. Significance \* =  $P < 0.05$ , \*\* =  $P < 0.01$ , \*\*\* =  $P < 0.001$ .

Species	Performance measure	Grass (+/-)	Linear (l)/ Quadratic (q)	$r^2$	F	Significance
<u>D. purpurea</u>	Leaf length	+	l	0.21	6.2	*
	Flowering stem height	+	l	0.21	6.2	*
<u>G. mollugo</u>	Stem no.	+	q	0.58	15.2	***
		-	q	0.40	7.5	**
	Stem length	-	q	0.61	16.7	***
<u>H. hirsutum</u>	Stem no.	-	q	0.27	4.0	*
	Stem length	-	q	0.59	15.9	***
<u>L. flos-cuculi</u>	Flowering stem height	+	q	0.31	5.0	*
		-	q	0.81	47.4	***
	Stem no.	+	l	0.17	4.7	*
		-	l	0.43	17.3	**
	Flower no.	+	l	0.46	19.6	**
		-	l	0.40	15.6	**
<u>P. veris</u>	Leaf no.	+	q	0.56	13.8	**
		-	l	0.20	5.9	*
	Leaf length	+	l	0.26	8.0	**
	Flowering stem no.	+	q	0.27	4.1	*
		-	l	0.46	19.3	**
	Flowering stem height	+	l	0.38	14.3	**
		-	l	0.20	5.9	*
	Flower no.	+	l	0.21	6.1	*
	-	l	0.39	15.0	**	
<u>R. acris</u>	Flowering stem height	+	l	0.28	8.7	*
<u>S. sylvatica</u>	Stem no.	+	l	0.30	9.7	*

## EFFECTS OF INSECTICIDE APPLICATION ON WEED AND PASTURE PLANT COMMUNITIES

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## ABSTRACT

Long-term experiments, involving the use of insecticides to manipulate insect numbers, have become an important feature of ecological methodology in recent years. These experiments provide a parallel to the situation where insecticides are applied over a long period of time to perennial crops, such as pasture grass. The reduction of foliar- and/or root-feeding insects can have dramatic effects on pasture plant communities, but also on weed communities which are often subjected to insecticidal drift.

Application of foliar insecticide in both weed and pasture communities increases the performance of vigorous-growing perennial grasses, which outcompete forb species and thereby leads to a depression in plant species richness. In contrast, soil insecticide application increases perennial forb growth at the expense of perennial grasses in a weed community, but acts like foliar insecticide in a pasture community. The consequences of these results in land subjected to changing use and habitat restoration programmes are discussed.

## INTRODUCTION

The increasing commitment of farmers and land owners to incorporate semi-natural habitats into farmland highlights the need for greater awareness of the action of pesticides on these plant and animal communities. We currently know relatively little about the impact of pesticides on naturally developing grassland, especially in the long term. It is well established that grassland covers more than 65% of the agricultural area in Britain. Furthermore, herbage losses of over £500 million per year have been attributed to the action of invertebrate pests such as insects and molluscs (Clements *et al.*, 1991). Moreover, while cereal crops cover less area than grass, they may receive up to 20 times as much annual pesticide application (Clements *et al.*, 1991). Although these chemicals are effective against the target organisms, we know very little about their effects on surrounding natural vegetation. Clearly, the development of semi-natural habitats, such as grassland, may be seriously affected by either the direct action of insect herbivores or spray drift from nearby crops. Meanwhile, the introduction of the set-aside scheme and the maintenance of field margins and headlands to provide habitats for plant and animal species of conservation value, reservoirs for predators and parasites of arable crop pests (Morris & Webb, 1987) and cover and food for game birds (Rands & Sotherton, 1987) are making early successional or weed communities an integral part of the farmland scene. The effects of pesticides on these habitats need to be considered, not least because such areas are often in close proximity to crops and are therefore highly vulnerable to insecticide drift (Dover *et al.*, 1990).

Since their advocacy by Harper (1977), long-term experiments, involving the manipulation of insect numbers by the judicious application of pesticides, have become a feature of ecological methodology. Using these methods, recent work at Silwood Park has demonstrated that both above- and below-ground insect herbivores may be important in the dynamics of early successional plant communities (Brown, 1989; Brown & Gange, in press). In this paper, we

explore the effects of the long-term use of insecticides on semi-natural vegetation in farmlands. We take two examples: one a typical weed community, developing on abandoned land, such as the fallow option of set-aside, and the other a mature pastureland dominated by perennial grasses. The results are compared to those of insecticide application in improved grassland (Clements *et al.*, 1990) and their relevance to the management of farmland is discussed.

## MATERIALS AND METHODS

### Experimental sites

There were two experimental sites, each 400m<sup>2</sup>, on ex-arable land at Silwood Park, Berkshire. These were on acidic, sandy soil and had previously been part of a four-year rotation of winter wheat, field beans, Brassicas and fallow. Each site was created by treating the area with a weedkiller in autumn to destroy perennial weeds and then shallow ploughing in winter. The land was harrowed the following March and the vegetation left to develop naturally. At the beginning of the study, one site comprised vegetation developing on bare ground and will be referred to as the "weed community". The other site was at least 17 years old and was mature grassland; it will be referred to as the "pasture community". Each site was fenced to exclude rabbits and mollusc numbers were reduced by the application of slug pellets (Mifaslug, Farmers Crop Chemicals Ltd). Plots (3m x 3m), separated by 2m 'walkways', were established on each site and allocated to treatments (see below) at random.

The same experimental treatments were initiated on each site: foliar-feeding insects were reduced by application of Dimethoate-40 (Portman Agrochemicals Limited) at 2-3 weekly intervals in summer and monthly in winter. This was applied at the standard agricultural rate of 0.336 kg AI ha<sup>-1</sup> (equivalent to 750ml per plot) (Martin & Worthing, 1976). Control plots were sprayed with an equal volume of water. Soil-dwelling insects were reduced by the application of Dursban 5G containing 5% *m/m* chlorpyrifos (DowElanco) at the rate of 18g per plot at six-weekly intervals. A complete reduction in herbivory was also obtained by the application of both compounds, thereby making the experiments fully factorial. The results of total herbivore removal, detailing the interactions between treatments, will be published elsewhere. Both experiments contained five replicates of each treatment. The experiments are ongoing, but the results presented here describe the first four years of insecticide application. The advantages of the selected insecticides in manipulative field experiments, in addition to their methods of application and tests of possible phytotoxicity, have been described previously (Brown, 1989; Gange *et al.*, in press a).

### Sampling

The developing vegetation in the weed community was sampled at fortnightly intervals in the first season (May-October) and at monthly intervals in three subsequent seasons. The pasture community was sampled monthly for four seasons. A 38cm linear steel frame, containing ten 3mm diameter point quadrat pins was placed randomly five times in each plot. All touches of living plant material on each pin were recorded to species. Data were condensed to provide information on the total touches of all vegetation (cover abundance) and the total touches of plant species belonging to different life-history groups (Gibson *et al.*, 1987). Here, only the annual forbs (non-graminaceous species), perennial forbs and perennial grasses are considered. Cover abundance of these life-history groups was expressed as a percentage of the total vegetation in each plot, to provide relative measures of their contribution to the plant community. The proportional data were analysed according to dates using a split-plot analysis of variance, employing the pesticide treatments as main effects. All proportions were first subjected to the arc-sine transformation.

## RESULTS

Effects of insecticide application on the weed community

Annual forbs dominated the community during the first year of colonisation (Fig. 1a). Although the cover abundance of this life-history group was increased by both forms of insecticide application (Brown & Gange, in press), the contribution of this group to the total vegetative cover in the three treatments was not altered. However, during the second and third years of application, foliar insecticide significantly reduced the proportion of the annual forbs in the community (year 2:  $F = 4.3$ ,  $P < 0.05$ ; year 3:  $F = 9.1$ ,  $P < 0.01$ ) (Fig. 1b,c). In contrast, there was a significant increase in the proportion of annual forbs in those plots treated with the soil insecticide during years three and four (year 3:  $F = 10.9$ ,  $P < 0.01$ ; year 4:  $F = 5.9$ ,  $P < 0.05$ ) (Fig. 1c,d). By the fourth year of colonisation of bare ground, annual forbs are only a minor component of the vegetation. The difference between treatments in this year was mainly because seedling recruitment was enhanced by the application of soil insecticide. *Vicia hirsuta* and *V. sativa* were the commonest annual forbs in the sward at this time.

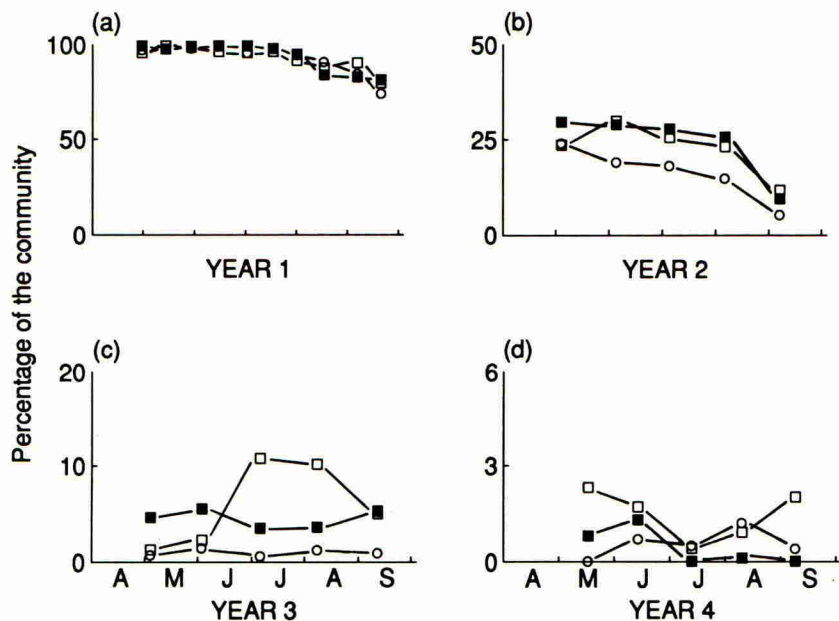


Fig. 1 Influence of foliar and soil insecticide application on the percentage contribution of annual forbs to the weed community during the first four years of the colonisation of bare ground. (○) foliar insecticide, (□) soil insecticide, (■) control.

Perennial forbs only became established towards the end of the first year of colonisation (Fig. 2). The application of foliar insecticide significantly decreased the proportion of this life-history group in the community during years two and three (year 2:  $F = 3.9$ ,  $P < 0.05$ ; year 3:  $F = 6.1$ ,  $P < 0.05$ ) (Fig. 2b,c). Although the contribution of perennial forbs decreased in this treatment, there was no effect of application on the total cover abundance of these plants (Brown & Gange, in press). Thus, the proportional decrease must be due to another life-history group increasing in abundance (see below). In contrast to the application of foliar insecticide, soil insecticide significantly increased the cover abundance of perennial forbs during years two, three and four (Brown & Gange, in press), with their contribution to the total vegetation also being significantly increased in years three and four (year 3:  $F = 10.1$ ,  $P < 0.01$ ; year 4:  $F = 10.1$ ,  $P < 0.01$ ) (Fig. 2c,d). These differences were visually apparent in mid season, when the forbs were in flower. Indeed, 12 species of perennial forbs were only found in soil insecticide-treated plots. The most abundant of these were *Glechoma hederacea*, *Stellaria graminea* and *Trifolium hybridum*.

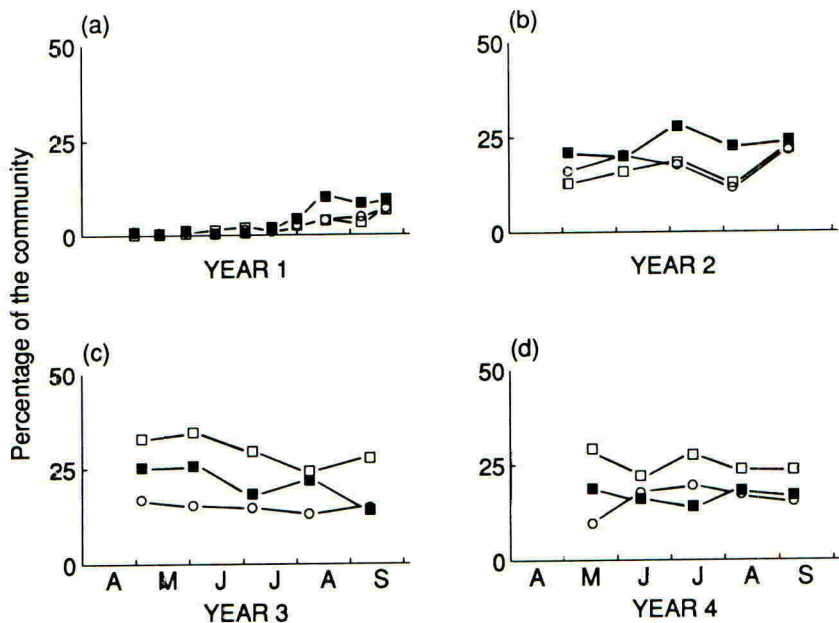


Fig. 2 Influence of foliar and soil insecticide application on the percentage contribution of perennial forbs to the weed community during the first four years of the colonisation of bare ground. (○) foliar insecticide, (□) soil insecticide, (■) control.

Like the perennial forbs, perennial grasses also became established at the end of the first season and rapidly began to dominate the community (Fig. 3). Their colonisation was increased in the foliar insecticide-treated plots, with these grasses contributing significantly more to the community than in the controls (year 2:  $F = 13.9$ ,  $P < 0.001$ ; year 3:  $F = 15.1$ ,  $P < 0.001$ ) (Fig. 3b,c). Total grass cover was also greatly increased by foliar insecticide application in years two to four, although the overall reduction in abundance in the fourth year is a likely result of the prolonged drought in that season (1990). Thus, the effect of applying foliar insecticide is to alter the competitive balance between forbs and grasses in favour of the grasses (Fig. 2b,c; Fig. 3b,c). These were mainly the vigorous-growing, highly competitive species, such as *Agrostis stolonifera* and *Elymus repens*. Once more, the application of soil insecticide had the opposite effect to that of the foliar compound since perennial grass contribution was significantly decreased by soil insecticide in years three and four (year 3:  $F = 12.3$ ,  $P < 0.001$ ; year 4:  $F = 14.4$ ,  $P < 0.001$ ) (Fig. 3c,d). Thus, in a weed community, long-term application of soil insecticide alters the balance between forbs and grasses in favour of the forbs (Fig. 2c,d; Fig. 3c,d).

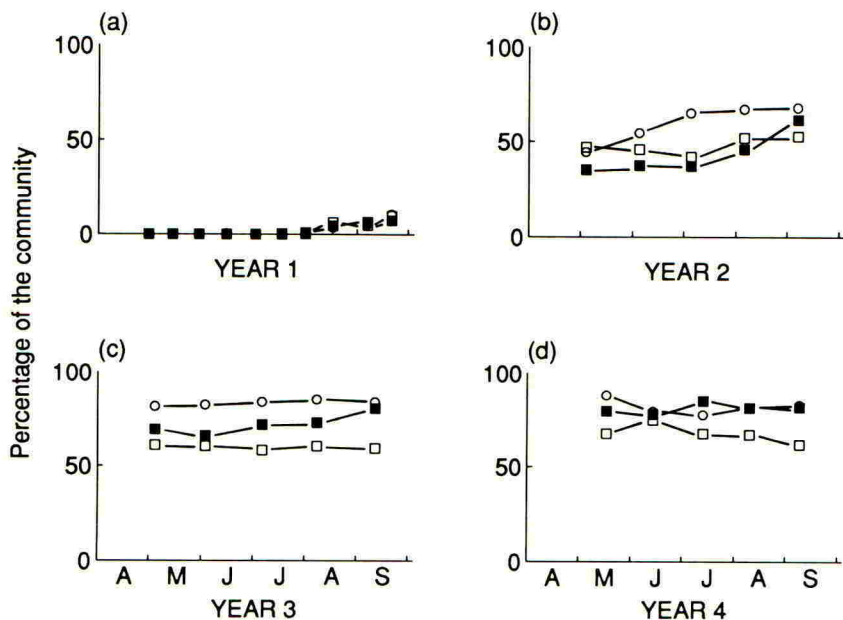


Fig. 3 Influence of foliar and soil insecticide application on the percentage contribution of perennial grasses to the weed community during the first four years of the colonisation of bare ground. (○) foliar insecticide, (□) soil insecticide, (■) control.

Effect of insecticide application on the pasture community

At the beginning of the experiment, the community was dominated by perennial forbs and grasses, with the grasses contributing 60-70% of the total vegetative cover. There were no differences between treatments in the contribution of these two life-history groups to the community (Fig. 4a; Fig. 5a) at this time. Application of foliar insecticide had no significant effect on the perennial forbs during the second year, but in years three and four, there was a reduction in the proportion of these plants in the plots which received this insecticide (year 3:  $F = 12.9$ ,  $P < 0.001$ ; year 4:  $F = 38.4$ ,  $P < 0.001$ ) (Fig. 4c,d). In contrast to the result recorded in the weed community, application of the soil insecticide also reduced the proportion of perennial forbs. This effect was more dramatic than the application of foliar insecticide, being found from year two onwards (year 2:  $F = 15.8$ ,  $P < 0.001$ ; year 3:  $F = 31.1$ ,  $P < 0.001$ ; year 4:  $F = 55.5$ ,  $P < 0.001$ ) (Fig. 4b,c,d). The most abundant forb was *Galium saxatile*, with *Rumex acetosa* and *R. acetosella* being moderately common.

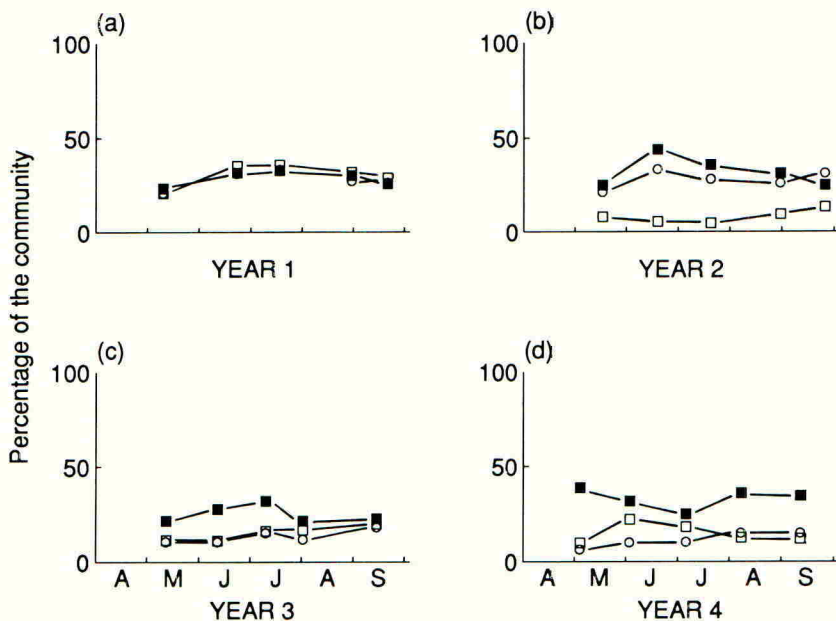


Fig. 4 Influence of foliar and soil insecticide application on the percentage contribution of perennial forbs to the pasture community over a four-year period. (  $\circ$  ) foliar insecticide, (  $\square$  ) soil insecticide, (  $\blacksquare$  ) control.

The effect of foliar insecticide application on perennial grasses was consistent with that found in the weed community, with the proportion of these plants being increased in the third and fourth years of the experiment (year 3:  $F = 7.4$ ,  $P < 0.01$ ; year 4:  $F = 7.8$ ,  $P < 0.01$ ) (Fig. 5c,d). However, soil insecticide again had the opposite effect to that found in the weed community, with application resulting in a significant increase in the proportion of perennial grasses. The change in grass abundance also occurred more rapidly than in the foliar treatment, with an increase from the second year (year 2:  $F = 19.5$ ,  $P < 0.001$ ; year 3:  $F = 35.8$ ,  $P < 0.001$ ; year 4:  $F = 74.0$ ,  $P < 0.001$ ) (Fig. 5b,c,d). Therefore, in the mature pasture community, application of foliar and soil insecticide altered the competitive balance between forbs and grasses in favour of the grasses. The two most abundant grass species were *Agrostis capillaris* and *Holcus lanatus*.

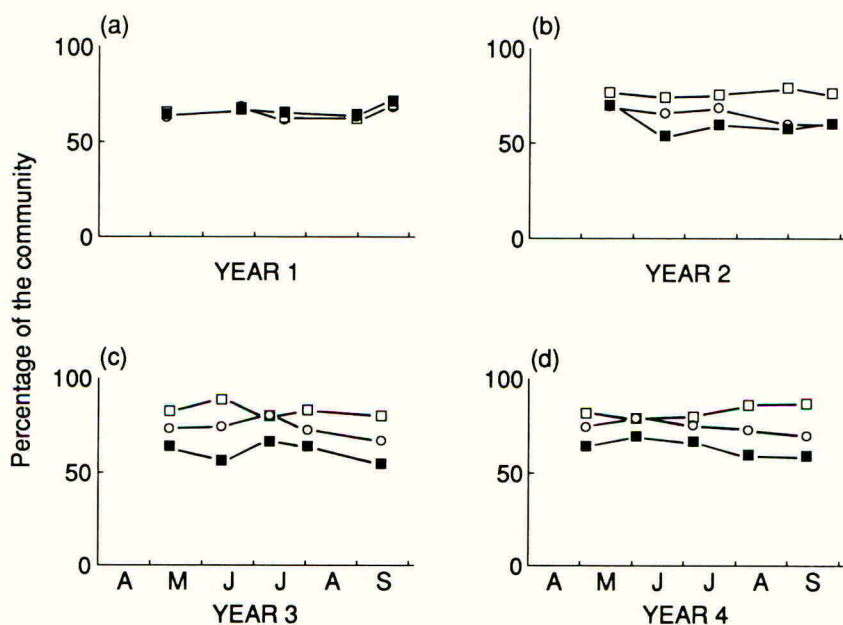


Fig. 5 Influence of foliar and soil insecticide application on the percentage contribution of perennial grasses to the pasture community over a four-year period. (○) foliar insecticide, (□) soil insecticide, (■) control.



## DISCUSSION

These results clearly demonstrate that insect herbivores may have dramatic consequences for the establishment and subsequent development of natural plant communities. The fact that insecticide application can lead to altered proportions of different plant life-history groups shows that both foliar and subterranean insects must be considered as important causative agents of plant community structure and composition. In addition, the effect of herbivory is clearly dependent on community composition. While direct effects of herbivores on individual plants may be recorded (Gange, 1990), at the community level, the effect is mediated through plant competition (Brown, 1990). The different results of insecticide application on the weed and pasture communities may at first appear anomalous, but can be explained by a consideration of the colonisation of insects occurring during early plant succession and the availability of microsites for successful germination and recruitment.

In the ruderal phase of community development, representing recently abandoned arable land, populations of foliar-feeding insects may rapidly invade the site (Brown & Southwood, 1987; Southwood *et al.*, 1979) and soil-dwelling insects may be abundant following a crop (Blackshaw, 1988). Application of insecticide to reduce either of these modes of herbivory results in a large increase in annual forb growth (Brown & Gange, 1989a; in press). Continuing application has shown that foliar insecticide increases perennial grass growth and leads to a depression of plant species richness (Brown, 1989; Brown & Gange, in press), while soil insecticide increases perennial forb growth and enhances plant species richness (Brown & Gange, 1989b). The results presented here demonstrate the mechanisms by which these changes occur. Not only does insecticide application alter the total vegetative cover of the various life-history groups, it also alters the relative proportions of these groups in the community. Thus, foliar insecticide increased the proportion of perennial grasses in the weed community, this increase being at the expense of the perennial forbs. Perennial forb cover was not reduced compared to controls (Brown & Gange, in press) and therefore the effect is clearly an indirect one, caused by competition from the vigorously-growing grasses. It has been shown that, among the foliar-feeding insects of these early successional communities, the majority are associated with grasses (Edwards-Jones, 1988) and thus it is not unexpected that the grasses show enhanced growth when released from herbivory. In contrast, application of soil insecticide increased the proportion of perennial forbs and decreased that of perennial grasses. However, since the total cover of both life-history groups was increased in this treatment (Brown & Gange, in press), the effect is clearly not a simple competitive one. Soil-dwelling insects are known to be important agents of seed and seedling mortality (Brown & Gange, 1990). In a laboratory trial, it was found that root herbivores could reduce seedling appearance by 60% and of the seedlings which died, 85% did so before reaching the soil surface (pre-emergence mortality) (Gange *et al.*, in press b). By weekly monitoring of seedling appearance in a ruderal community for two years, we have found that soil insecticide application significantly increased the number of seedlings which recruited (Brown & Gange, unpubl.). Thus, if soil insecticide is applied at the time of forb establishment, this life-history group may be increased to a level of abundance where at least parity in competition with grasses is achieved. In addition, at this early stage in succession viable seed is still present in the seed bank, together with microsites for successful germination (Oosting & Humphreys, 1940).

When insecticide is applied to a mature pasture community, the results may in part appear to contradict those in the weed community. Foliar insecticide resulted in a release from herbivory of the dominant life-history group, the perennial grasses. As a result, the proportion of grasses increased at the expense of the perennial forbs. However, this effect was not immediate and is a reflection of the manner in which these plants grow (Gange, 1990). It is interesting that similar results have been obtained from continuous pesticide application to old mixed pasture

communities (Henderson & Clements, 1974) and highlight the need for long-term experiments of this kind. Meanwhile, soil insecticide also increased the proportion of perennial grasses and decreased that of perennial forbs. The explanation may be found in the fact that at this stage of succession, when grasses have formed a tight sward, with significant litter build up, there is virtually no recruitment of perennial forbs from seed. Microsite availability for germination is thus considerably lower than in the weed community. In addition, densities of soil-dwelling insect herbivores are high in such plant communities, reaching 2000m<sup>-2</sup> with many of the species feeding preferentially on grasses (Brown & Gange, 1990). It is not therefore surprising that grass growth is dramatically increased, at the expense of perennial forbs when this insecticide is applied. In a 20 year experiment, Clements *et al.* (1991) applied a soil insecticide, similar to that used here to a developing ryegrass (*Lolium perenne*) pasture. Pesticide application was heavy and led to the complete removal of earthworms, which in turn led to changes in a number of soil physical characteristics. Since the sward was sown to perennial grass, forb cover was minimal, however grass production was increased by 15% during the first 11 years of the study. There was also a suggestion that the proportion of invasive perennial grasses, such as *Agrostis* spp. and *Elymus repens* increased in the pesticide treatment, at the expense of the *Lolium*. It therefore appears that when an insecticide is applied to any pasture-type community, the most competitive grasses will come to dominate that community.

These results have important implications for the management of set-aside or abandoned land. The application of foliar insecticide, such as by accidental spray drift, should clearly be avoided. Otherwise swards of competitive grasses which may act as reservoirs for arable pests (both plant and invertebrate) may be produced. In particular, spray drift may be a problem in field margin conservation. Even if field boundaries are sown with wildflower mixtures, these could soon become dominated by grasses if subjected to foliar insecticide (Marshall & Smith, 1987). Application of soil insecticide, preferably in a slow-release formulation, may be beneficial to the establishment of forb-rich areas, especially those designed to be created under the Countryside Premium Scheme (Anon, 1990). Since the management of plant succession in set-aside land is by its very nature a long-term process, further experiments are required to determine which other factors, apart from insect herbivores, are important in farmland conservation.

#### ACKNOWLEDGEMENTS

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PREDICTING THE LONG-TERM IMPACT OF PESTICIDES ON PREDATORY  
INVERTEBRATES

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## ABSTRACT

This paper investigates the possible long-term side-effects of pesticides in temperate cereal systems. Clear evidence for dispersion by non-target predatory invertebrates suggests that in evaluating pesticide effects, too small an experimental scale may underestimate the time taken for these beneficial invertebrates to return to sprayed fields. One consequence may be an underestimation of the likelihood of pest resurgence in those systems where pesticides are extensively applied. A model is presented which aims to bridge the gap between short-term ecological observations, which reveal mechanisms of clear significance to invertebrate population dynamics and some long-term monitoring observations, which are so far poorly understood.

## INTRODUCTION

The dynamics of invertebrates in systems which are sprayed with pesticide, are not solely dependent upon pesticide toxicity. Factors such as how the invertebrates move between fields, their reproductive capacity and the intensity and extent of pesticide application may also be important in determining their dynamics. Even a mild compound may cause serious problems if used frequently enough or on a large enough scale. We therefore need a more comprehensive approach than simply measuring toxicity if we are ever to be in a position of defining the limits of pesticide use, beyond which unacceptable side-effects are caused.

Despite the long history of research into pesticide side-effects (eg. Croft, 1990) we are still in the phase of discovery and hypothesis formation concerning how pesticide application influences the population dynamics of predators, parasites and their prey or hosts (e.g. May & Hassell, 1988). Internationally, there are many examples of invertebrate dynamics under pesticide treatment which are difficult to understand, let alone predict: for example, resurgence and secondary pest outbreaks (Croft, 1990) which can occur widely in some systems (e.g. cotton, rice, temperate orchards), but not in others (e.g. temperate arable crops). In this paper we argue that a number of factors, all related to space and time, such as the intensity and extent of application of pesticide on a farm scale and the dispersive ability of invertebrates, within and between fields, must be considered if we are to properly understand the reasons behind such side-effects. We focus specifically upon temperate arable crops, apparently a resilient system. Firstly, we present evidence for the dispersive ability of invertebrates; we then present data indicating that localised pest re-surgence can occur as

a result of delayed predator re-colonisation. The findings of the two long-term studies of invertebrate population dynamics in cereal systems carried out to date are then summarised, emphasising how an understanding of local movement and predation can contribute to our interpretation of the results. Finally we outline a simulation model which considers explicitly the spatial and temporal components of pesticide application and invertebrate movement, to predict the long-term effects of pesticide use; specifically, the risk of local extinction.

#### MOVEMENT BY PREDATORY INVERTEBRATES INFLUENCES THE PATTERN OF RECOVERY AFTER SPRAYING

Jepson & Thacker (1990) and Thomas *et al.* (1990) have demonstrated that treated plots in cereal fields are progressively re-invaded by invertebrates which inhabit untreated control plots within the same field. This leads to a progressive equilibration of invertebrate densities within the treated and untreated parts of the crop. Areas further into sprayed plots take increasingly longer to reach densities of invertebrates comparable with the untreated control. Regression models can be used to predict the duration of effects that might be expected on a range of spatial scales (Table 1). Note however, that the presented data only apply to situations where the treated area is balanced by an equivalent area of untreated crop in the same field and that the experiments only covered the period when active recolonisation was taking place. Nevertheless, these findings have implications for the appropriate size of plot in which to experimentally test for pesticide side-effects: too small an experimental plot may lead to underestimates of the time taken for predators to re-colonise whole fields.

These experiments not only point to movement as an important factor influencing pesticide effects but also provide a means for its quantitative measurement. Thus they may be extremely useful in the parameterisation of those population models which explicitly consider invertebrate movement over farm systems.

#### THE PATTERN OF RECOVERY BY PREDATORY INVERTEBRATES HAS KNOCK-ON EFFECTS FOR PESTS

Work at the Boxworth Experimental Husbandry Farm, Huntingdonshire, UK (Burn, 1991) has indicated that under intensive treatment regimes, pesticides may reduce the capacity of predators to limit prey populations. The evidence for this comes from reductions in the rate and level of consumption of natural and artificial prey in intensively sprayed areas. Analogous data comes from more conventional, within-field experimentation (Duffield & Baker, 1990; Duffield, 1991) where apparent aphid resurgence has been detected in the centres of large, treated areas where predator reinvasion has been delayed. Duffield (1991) calculated the time in days taken for different invertebrate families to reach control (unsprayed) densities when pesticides were applied on a variety of different spatial scales (Table 2). He found that predatory invertebrates, such as the Carabidae and Linyphiidae, took more time to recover the larger the spatial scale of application, but their prey species, the Aphididae, took less time to recover the larger the spatial scale. The shorter recovery time of Aphididae in large plots may, at least in part, occur as a result of reduced predation. This finding has important implications; it suggests that pest resurgence is more likely in the centres of large sprayed plots than in the centres of the small plots that are used in field

TABLE 1. The minimum plot sizes (ha) required to detect treatment effects over specified periods, on a series of non-target arthropod families for dimethoate, applied as a summer aphicide to winter-wheat (Thacker, 1991).

Family	Time period over which significant effects are detected (days)			
	15	20	25	30
Carabidae	1.55	4.84	8.86	14.75
Linyphiidae	0.50	3.94	10.59	20.45
Staphylinidae	12.80	42.56	89.65	154.34

TABLE 2 The time taken for populations of carabids, linyphiids and aphids to recover in plots of different area treated with dimethoate (Duffield, 1991).

Family	Area treated (hectares)					
	0.01	0.06	0.25	1.00	2.25	10.0
Carabidae	6.7	9.0	11.3	14.3	16.1	17.5
Linyphiidae	2.5	3.4	4.2	5.3	6.0	7.7
Aphididae	72.8	30.0	15.4	7.9	5.3	2.6

experiments.

Both Tables 1 and 2 indicate that side-effects will become quantitatively and qualitatively different as treatment scale approaches commercial reality. The duration of effects on natural enemies will be extended, the range of taxonomic groups affected will be greater and the chance of pest resurgence occurring will be increased. Long-term effects on predators are only likely to arise on larger scales which exceed the capacity of local reservoirs of predatory invertebrates to refill the treated area within the season of application. The next section examines research carried out on a large scale.

#### EVIDENCE FOR LONG-TERM EFFECTS FROM LARGE-SCALE STUDIES

Long-term declines in the population densities of certain invertebrates have been detected over an extended run of monitoring in Sussex farmland (Aebischer, 1990; Aebischer & Potts, 1990). Significant declines have been detected in Araneae, Cryptophagidae, Lathridiidae, Lepidoptera, Lonchopteridae, Opiliones, Parasitica, Staphylinidae and Symphyta: Aphididae also declined. Although the mechanisms for many of these changes are uncertain, indirect effects such as depletion of food reserves for mycetophagous species due to the use of fungicides, seem to have been important. A similar mechanism may explain why Parasitica declined, in parallel with Aphididae. Direct pesticide effects were more difficult to detect because insecticide use was generally at a low level and did not occur with any regularity until over half way through the study. Evidence that large-scale insecticide treatments can have a direct impact were given by Aebischer (1990) however, who detected a ten-fold reduction in sawfly populations in a 7 km<sup>2</sup> area that had been treated with dimethoate. He estimated that the populations would take 7 years to recover, using time-series models which incorporated agricultural practices and climatic variables. These models constitute an excellent tool for investigating the dynamics of a particular data-set and generalisations from these may be possible: in this case however, directly toxic compounds did not figure strongly enough for more general models of long-term insecticide effects to be developed.

Burn (1991) reported the results of a unique study, which examined the effects of three different pesticide regimes, applied to contiguous groups of fields over a series of years at the Boxworth Experimental Husbandry Farm. With the most intensive regime, both long and short-term perturbations were detected in invertebrates representing a large number of families. In particular, the predatory Coleoptera were affected, with some species becoming extinct within the experimental area. Within the Carabidae, three groups could be defined, on the basis of their phenology (Burn 1989; Jepson 1989). The most seriously affected group (Bembidion lampros and Nebria brevicollis) overwintered in the field and had surface active larvae; they were therefore exposed to autumn molluscicides and insecticide sprays as well as some spring and summer treatments. In a second group (Agonum dorsale, Bembidion lampros and Demetrias atricapillus) overwintering of adults in field boundaries provided a degree of protection however, sprays that coincided with spring colonisation or surface active summer larvae caused effects that persisted between years in some cases and there was evidence that diet depletion reduced fecundity and the ability to recover. A third group (Harpalus rufipes, Pterostichus melanarius and Trechus quadristriatus) were unaffected or increased in density within the intensive pesticide regime: these species had sub-terranean larvae, which avoided pesticide exposure and were either strong dispersers, capable of rapid

reinvasion or had flexible breeding strategies with the ability to produce offspring in the spring or autumn. Neither of the other, less intensive, pesticide regimes caused the long term extinction of a monitored species.

Overall these long-term data sets appear to indicate that:

- (i) The intensity of pesticide application influences the long-term dynamics of invertebrates.
- (ii) That trophic interactions are important in determining the long-term dynamics of some invertebrates.
- (iii) That behavioural characteristics of the invertebrates, such as dispersal rate and timing, influence their dynamics under pesticide treatment.

The outcome of most small-scale studies would lead us to expect a rapid recovery after pesticide application, even by the most vulnerable species of Carabidae (Jepson & Thacker, 1990). What then are the characteristics of large-scale treatments that lead to long-term declines as detected by Aebischer or the local extinction of some species as found by Burn? The reasons could include:

- (i) Repeated application of pesticides, within and between seasons.
- (ii) Larger scales of treatment to groups of fields.
- (iii) Treatment of whole fields, eliminating sources of recolonisation within the same crop.
- (iv) The presence of hedgerows and other landscape features which might delay recolonisation between fields.

In order to evaluate how such factors might affect the long-term impact of pesticides on predatory invertebrates we need a suitable theoretical framework. The next section presents a simulation model which incorporates many of these factors and which is capable of characterising their influence.

#### A GENERAL MODEL OF LONG-TERM EFFECTS OF PESTICIDES

Earlier models of the long-term population dynamics of invertebrates exposed to pesticides (Barclay, 1982; Waage *et al.*, 1985) all assumed entirely homogeneous populations, unaffected by immigration and emigration. The system we wish to model is perhaps best described not as a large, single, homogeneous population, nor as a system of local, isolated populations but as a system of local populations linked by dispersion. Such 'metapopulations' (Levins 1969) are now being seen as an appropriate level to explore the dynamic properties of a number of ecological systems (e.g. see Gilpin & Hanski, 1991).

A stochastic model has been written (Sherratt & Jepson, in prep) to simulate a closed farm, consisting of 16, equally-sized fields in a 4x4 matrix. The fields differ in respect to the initial number of invertebrates present, when they are initially sprayed with pesticide, how



regularly they are subsequently sprayed and how quickly invertebrates can move within them. The model assumes that invertebrates move seasonally by a diffusive process generated by a random walk, although this may be refined once more detailed information concerning the nature of invertebrate movement patterns is available. Boundaries between fields are of fixed permeability, in that if a proposed movement might take an invertebrate into another field, there is a fixed chance of the movement being rejected and an alternative movement being made. On entering a new field, the invertebrate has a chance of dying as a result of pesticide exposure. This chance declines exponentially from the time the pesticide was last applied. Invertebrates that are in the field when a fresh application of pesticide is made have a fixed probability of dying. Reproduction is density-dependent in that the more adult invertebrates there are in the field, the less offspring there will be per adult.

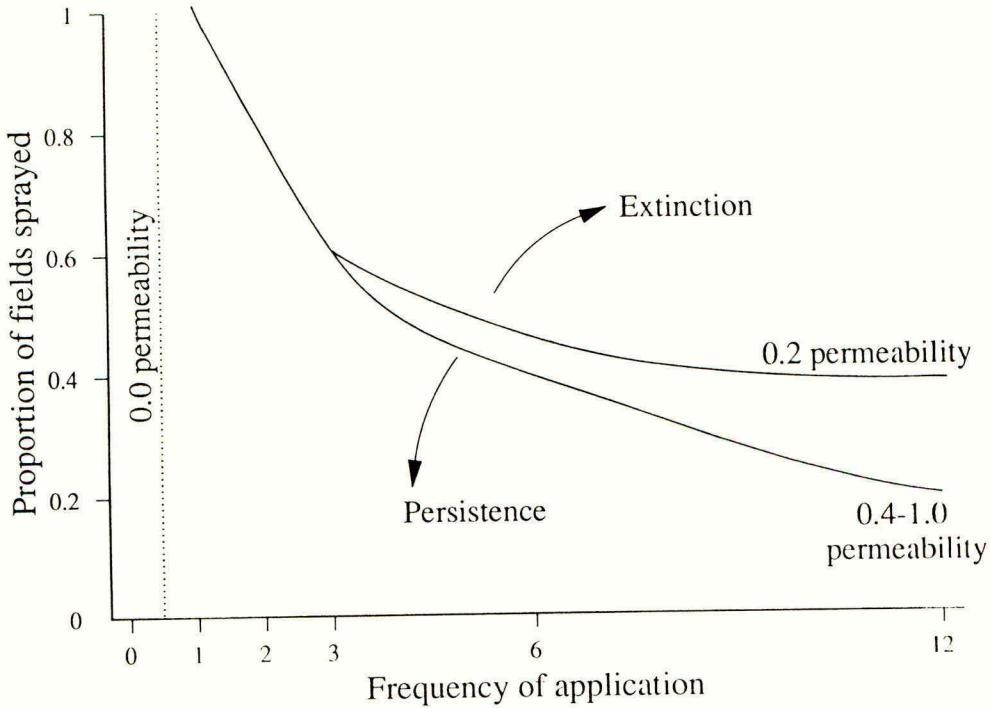
The model allows us to investigate the interactions between different key variables (such as hedgerow permeability, frequency of pesticide use and invertebrate speed of movement) and their impacts on the persistence of invertebrates both within individual fields (local persistence) and within the farm system as a whole (global persistence). The model can be scaled and parameterised using estimates from existing datasets. For example, Unal & Jepson (1991) have precisely quantified the duration of soil and foliar toxicity of cereal aphicides to small Carabidae and Coccinellidae: Jepson & Thacker (1990) have measured the time taken for sprayed areas to be fully recolonised by Carabidae, yielding a measure of their speed of movement.

The model can be explored in a number of ways. For instance, how does the frequency of pesticide application and the proportion of fields sprayed influence the persistence of invertebrates within a given field? We examine this by looking at the dynamics of invertebrates within a particular "control" field which is always sprayed and investigate what patterns of pesticide use result in the local extinction of invertebrates within this field. Figure 1 shows three lines of stability which mark the boundary between persistence within this field (on greater than 50% of simulations) and extinction. The model was parameterised as far as possible using experimental data on the dispersive speed of the Carabidae and their susceptibility to pesticides. Each line corresponds to a different permeability of field boundary (1.0 corresponds to completely permeable, 0.0 corresponds to completely impermeable). The key points to note are that:

- (i) In all three cases the more frequent (times per year) or the more widespread (proportion of fields sprayed) the pesticide application, the more likely it is that the population of predatory invertebrates will go extinct.
- (ii) The permeability of the field boundary can influence where the line marking the limit between persistence and extinction falls. If boundaries are too permeable then unsprayed or irregularly sprayed fields do not store many invertebrates in "safe-havens" for re-supply to sprayed fields, yet if boundaries are too impermeable they they are also poor reservoirs since they can supply very little to sprayed fields. Thus there appears to be an optimum field boundary permeability, maximising the range of conditions under which predatory invertebrates are likely to persist. In this example the optimum permeability appears to be approximately 0.2.

The model can be used in a variety of other ways to characterise the influence of

Figure 1. Stability boundaries showing the sorts of pesticide regimes which are likely (greater than 50% chance) to result in the extinction of a hypothetical carabid population and the sorts of pesticide regime which are likely to enable persistence of the population. The three lines represent three different stability boundaries according to how easily invertebrates can cross the field boundaries. When fields are completely isolated (i.e. 0.0 permeability), events in neighbouring fields are immaterial to the dynamics of invertebrates within a specific field, the proportion of fields sprayed is thus irrelevant and local stability is only influenced by the number of sprays per annum.



farm management (e.g. what is the influence of different cropping patterns?), pesticide properties (e.g. what is the relative influence of the initial toxicity and persistence of a pesticide in this system?) or invertebrate characteristics. As an example of the latter we might ask what sort of invertebrates are more likely to persist on the farm scale? If a species moves slowly then there will be little immigration into treated areas and consequently local extinction in sprayed fields, but not farm-scale extinction. On the other hand, if invertebrates move quickly then they would expose themselves to more chances of being killed; consequently both local and farm-scale extinction might occur. Altering the speed of invertebrates within the field systems would determine whether invertebrates of particular speeds are more likely to persist.

This modelling approach is intended to complement the traditional approaches of monitoring and experimentation in evaluating long-term side effects of pesticides. The two long-term studies on cereal systems which found that only some invertebrate families were seriously affected by particular pesticide regimes, suggest that we will need models of some form if we are ever to be able to predict the long-term consequences of pesticides on invertebrates. Precisely because agricultural systems are so complex the most straightforward and effective way of modelling such a system is by using computer simulation.

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