Effects of Pollutants at the Ecosystem Level
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CASE 7.7
A Case Study of the Use of Fenitrothion in New Brunswick: The Evolution of an Ordered Approach to Ecological Monitoring

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7.7.1 INTRODUCTION

Fenitrothion is an organophosphorus insecticide used to control a number of closely related moths that are collectively called spruce budworm. The spruce budworm inhabits and feeds on coniferous forest in areas throughout North America, particularly on pure or mixed stands of balsam fir, alpine fir, and white, red and black spruce. Its preferred habitat is a mixed stand of balsam fir and spruce; such stands comprise approximately 15×10^6 acres in New Brunswick. They are the mainstay of the pulp and paper industry and spruce is also utilized for sawlogs. In addition, forest areas support or interact with a very large number of organisms, all of which fulfil important, interrelated ecological roles as well as being of aesthetic value.

The budworm is a natural part of such a complex ecosystem. It, and the forest it inhabits, are thought to have coexisted for thousands of years. There appear to have been at least seven major budworm outbreaks in Eastern Canada in the past 200 years, each extending over millions of acres (Baskerville, 1976). Such outbreaks have allowed the survival of both the insect and the forest since an outbreak tends to destroy the mature forest and a forest fire can follow, burning the dead trees and leaving ideal conditions for the germination of surviving seeds (Baskerville, 1976). A cycle induced by the fires exists whereby firspruce forests naturally tend to become even-aged and therefore present ideal conditions for future destructive budworm outbreaks. At the same time, the renewal processes lead to a very healthy young forest.

The spruce-budworm problem arose when man wished to compete with the budworm for the coniferous forest harvest. After an outbreak in 1913–1919, it seemed that the budworm might severely limit the industrial use of the forest that was gradually being established in New Brunswick. Spruce trees were replacing the large, old, white pines that had previously been heavily cut for sawlogs (Baskerville, 1976). In the 1950s, the intensity of the spruce harvest increased further as the industry became involved in supplying pulp for paper mills. The technology of the paper mills was designed specifically to utilize a continuous annual harvest of spruce and fir trees, which was incompatible with the sporadic but, to all intents and purposes, complete harvest of the budworm.

A crop-protection programme was introduced in 1952. Since then, vast areas of New Brunswick have been sprayed with assorted biocides each year. This case study will evaluate the usefulness of the ecological monitoring programme that has been conducted for fenitrothion. The principles of experimental strategies which can be applied to maximize the usefulness of monitoring studies will be identified.

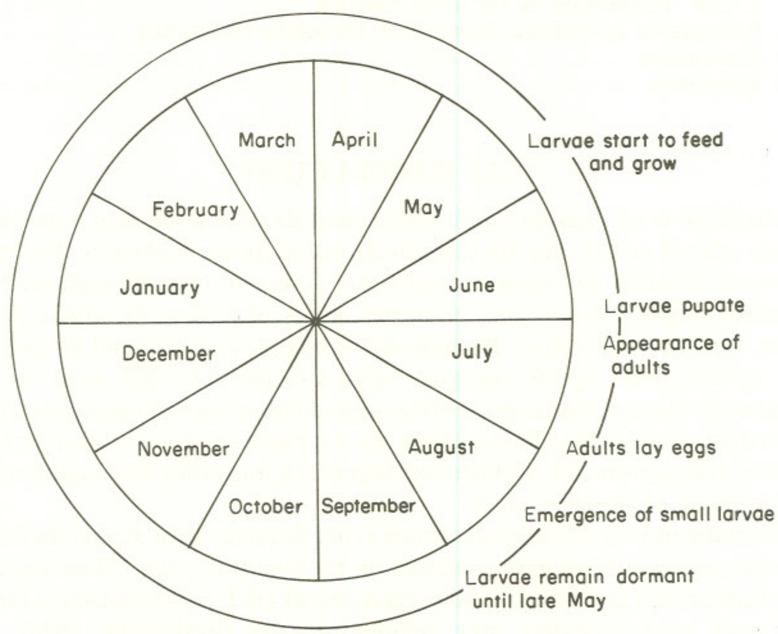

Figure 7.7.1 Lifecycle of spruce budworm in New Brunswick

7.7.2 DEVELOPMENT OF THE PROBLEM

7.7.2.1 Lifecycle of the Budworm

The annual life cycle of the budworm in New Brunswick is represented in Figure 7.7.1. Soon after emerging, in late June or early July, female moths lay their eggs. If the population is small, eggs will be laid locally. However, when the population reaches a critical size and local host trees have become heavily defoliated, females disperse over an area as large as 40–160 km and their subsequent egg-laying spreads the outbreak (Baskerville, 1976). Small larvae emerge in late July and August but do not begin feeding until the following spring. The actual dates of the budworm life history vary from year to year and from place to place, depending on climatic conditions. Under normal conditions, parasites, predators and disease cause a 99 per cent total mortality of immature life stages. Consequently, although each female lays about 200 eggs, most of the time the budworm population naturally remains low.

However, it is obvious that very small reductions in mortality can cause explosive increases in population levels. This is illustrated in Figure 7.7.2. For example, if the mortality rate falls to 90 per cent, a 10 000-fold increase could result

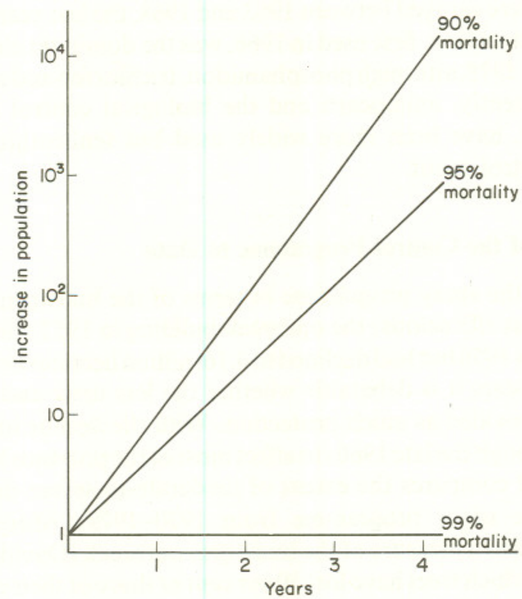

Figure 7.7.2 Effect of decreased generation mortality on budworm population levels over four generations (assuming each female lays 200 eggs and the male: female ratio is 1 : 1)

in 4 years. If this happens to take place in a highly susceptible older forest, an outbreak occurs which results in the severe defoliation of trees. Budworm larvae normally prefer current foliage and, since fir and spruce trees retain a portion of each year's needles for at least eight years, it can take several years to kill a tree. When budworm numbers are exceptionally high and new foliage is limited, the budworm will consume old foliage and this hastens the death of the tree. Mature forests are thus particularly susceptible to destruction. Outbreaks have a natural time limit; factors such as starvation, lack of egg-laying sites or prolonged cool and wet climatic conditions have been postulated to eventually cause budworm populations to collapse back to endemic levels.

7.7.2.2 Crop Protection

New Brunswick has engaged in a spruce–fir crop protection programme involving the aerial spraying of larvae with assorted biocides since 1952. The history of this programme (summarized in Table 7.7.1) is complex because different spray strategies have been used. On average, 17 per cent of the total forest area has been sprayed annually since 1952. Until 1962, mainly DDT was used, but as the environmental problems caused by DDT and the extreme persistance of its metabolites were recognized (Brooks, 1974), new chemicals were tested. DDT and phosphamidon were sprayed between 1963 and 1968, the last year in which DDT was applied. Fenitrothion, first used in 1966, was the dominant chemical sprayed between 1968 and 1978, although phosphamidon, trichlorfon and aminocarb were also utilized. Recently, aminocarb and the biological control agent, *Bacillus thuringiensis* (Bt), have been more widely used but fenitrothion remains the predominant control agent.

7.7.2.3 Efficacy of the Control Programme to Date

The net result of the spray programme in terms of the budworm, is difficult to evaluate. DDT was efficacious; the outbreak building in 1952 affected about 2.5 million hectares by 1956 but had declined to 0.16 million hectares by 1962 (Prebble, 1974). In recent years it is debatable whether the less toxic and less persistent pesticides have provided as much protection. With the decline in DDT use, the outbreak rose through the late 1960s to affect most of the province by 1973 (Figure 7.7.3). Table 7.7.2 compares the extent of moderate-to-severe defoliation with the extent of the spray programme from 1970–1979. 'Moderate-to-severe defoliation' is used as a measure of the budworm outbreak (Davidson, 1981) and indicates areas in which trees have lost 30 per cent or more of their current foliage. This comparison reveals no correlation between the quantity of pesticide applied and the extent of defoliation in subsequent years (see also Desaulniers, 1977).

Because the growth in budworm population is suppressed by the spray programme, the natural pulsing cycles and population collapses may no longer

Table 7.7.1 History of the spray programme in New Brunswick[1]

Year	Area sprayed[2] per year (ha)	Primary[3] pesticides	Application[4] rate ($g\,ha^{-1}$)	Solvents	Emulsifier(s)
1952–1962	8×10^4–2×10^6	DDT	290–1165		
1963–1966	3×10^5–8×10^6	DDT Phosphamidon	290		
1967	4.0×10^3	DDT Phosphamidon Fenitrothion	290		
1968	2.0×10^5	Phosphamidon Fenitrothion DDT	 290		
1969	1.3×10^6	Fenitrothion Phosphamidon	145	Atlox 3409F	Arotex 3470
1970	1.7×10^6	Fenitrothion Phosphamidon	145–218	Atlox 3409F Toximol MP 8	Arotex 3470
1971	2.4×10^6	Fenitrothion	145–218	Atlox 3409F Toximol MP 8	 Arotex 3470
1972	1.9×10^6	Fenitrothion Phosphamidon	145–290	Atlox 3409F Toximol MP 8	Arotex 3470
1973	1.8×10^6	Fenitrothion Phosphamidon	145–218	Atlox 3409F Toximol MP 8	Arotex 3470

Table 7.7.1 (*Cont.*)

Year	Area sprayed[2] per year (ha)	Primary[3] pesticides	Application[4] rate ($g\,ha^{-1}$)	Solvents	Emulsifier(s)
1974	2.3×10^6	Fenitrothion	145–218	Atlox 3409F	Arotex 3470
		Phosphamidon		Toximol MP 8	
		Trichlorfon	466		
1975	2.7×10^6	Fenitrothion	180	Atlox 3409F	Arotex 3470
		Phosphamidon		Toximol MP 8	
1976	4.2×10^6	Fenitrothion	217–280	Atlox 3409F	Arotex 3470
		Trichlorfon	580		
1977	1.6×10^6	Fenitrothion	210	Atlox R	Arotex R
		Aminocarb	70	No. 2 fuel oil	Nonyl Phenol
		Trichlorfon	580	Water	
1978	1.5×10^6	Fenitrothion	217–290	Atlox R	Arotex R
		Aminocarb	70	Diluent 585	Nonyl Phenol
1979	1.6×10^6	Aminocarb	70–90	Diluent 585	Nonyl Phenol
		Fenitrothion	218	Atlox R	Arotex R

[1] Data from: NRCC, 1975; Schneider, 1976; Baskerville, 1976; Varty, 1977, 1980.
[2] Only includes major spray programmes.
[3] Does not include pesticides applied to small areas for experimental reasons.
[4] Number of applications varied (1–5 times).

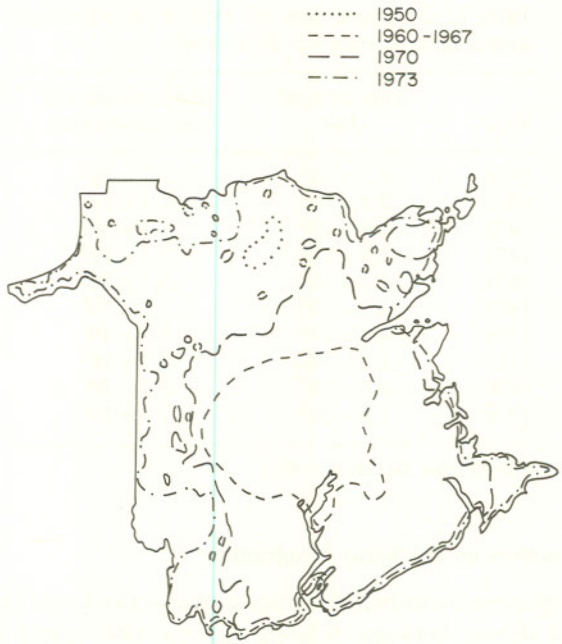

Figure 7.7.3 Margin of areas of generally persistent outbreaks in New Brunswick (adapted from Prebble, 1974)

take place. The spray programme helps to maintain an abundant food source and ample egg-laying sites. In fact, populations now tend to migrate from defoliated areas to areas where defoliation has been limited by previous protective measures. 'Return migration' has become possible, making the opportunity for persistence of the outbreak essentially limitless (Baskerville, 1976). As well, insecticides possibly reduce predator numbers (NRCC, 1975; Varty, 1980) and recent improvements in forest-fire control have favoured a higher component of balsam fir in forest stands (Newfoundland Royal Commission, 1981). The budworm outbreak has spread to include virtually the entire host forest in New Brunswick (Figure 7.7.3) and appears able to persist indefinitely.

Such a persistent, semi-outbreak condition has been tolerable to the forest industry. Millions of acres of wood have been lost, about as much as would have been lost in a single uncontrolled epidemic, but the loss has been spread over many years and has allowed an annual harvest to be perpetuated.

The problem is really a question of forest management. Is it acceptable, economically or ecologically, to continue annual spraying of biocides indefinitely? If not, what alternatives exist?

Table 7.7.2 Comparison of total area defoliated with total area sprayed; 1970–1980

Year	Area sprayed (ha)	Moderate-severe[a] defoliation (ha)
1970	1.7×10^6	5.7×10^5
1971	2.4×10^6	1.6×10^6
1972	1.9×10^6	1.7×10^6
1973	1.8×10^6	3.2×10^6
1974	2.3×10^6	3.4×10^6
1975	2.7×10^6	3.5×10^6
1976	4.2×10^6	6.9×10^5
1977	1.6×10^6	4.7×10^5
1978	1.5×10^6	6.7×10^5
1979	1.6×10^6	1.3×10^5

[a] Data from Davidson (1981).

7.7.2.4 Perspectives on the Spray Programme

The spray programme is continually being assessed at two levels. One level involves the emotional debate between different groups with vested interests in the situation; the other concerns scientific environmental monitoring and risk-benefit analysis.

Vested-interest groups include the wood industry and secondary forest product users on one side, and farmers, fishermen and public-interest groups on the other. Issues that have been raised include the following:

1. Does the spray programme have any adverse effect on the fishing industry? There is considerable evidence that DDT (at 290–1165 g ha^{-1}) caused severe economic loss to the Atlantic salmon industry (Logie, 1975). Fenitrothion at the application rates used (210–290 g ha^{-1}), is not directly toxic to fish but is toxic to aquatic invertebrates (NRCC, 1975). The potential long-term secondary effect of this reduction in invertebrate populations on fish productivity is one of the critical issues.
2. To what extent do pesticides damage other beneficial invertebrates, including pollinators, and in this case, what is the resulting damage to fruit crops? It is now clear that fenitrothion applied at 210–290 g ha^{-1} can cause up to a 100 per cent reduction in fruit crop (e.g. Bridges Brothers Ltd versus Forest Protection Ltd, The Supreme Court of New Brunswick, 1976; Thaler and Plowright, 1980) and the overall long-term implications have raised concern.
3. Do pesticides adversely affect other species, particularly birds and man? It is now clear that small crown-canopy birds can be affected by fenitrothion at present application rates (210–290 g ha^{-1}) (NRCC, 1975; Pearce *et al.*, 1976).
4. What is the net efficacy of the spray program?

These issues are obviously important but the concept of discontinuing the spray program also raises questions, including the following:

1. What would be the general economic, social and environmental damage of a full-scale, widespread budworm epidemic?
2. The budworm can severely damage the seed-bearing capacity of trees. Could this, in extreme situations, affect the natural replacement cycle?
3. After an outbreak, could a viable forest industry ever be established in a way that could avoid or deal with the possibility of another outbreak?
4. What will be the effects of increased fire-hazard in defoliated stands?
5. Would the tourist industry survive the aesthetic damage to recreation areas and would this damage be acceptable to the public at large?

These issues, and others that have been raised, need not contradict one another. Is it to anyone's advantage to allow the entire forest ecosystem to self-destruct? Equally, is it rational to disperse toxic chemicals, which may have deleterious effects on the ecosystem and may not even be effective as control agents, into the environment? Budworm control alternatives have been discussed by Baskerville (1976). They conclude that the idea, perpetuated by many professionals as well as by the public at large, that a simple solution to the problem exists, has been one of the greatest impediments to understanding and effectively dealing with the situation.

Interest groups can be expected only to voice questions; it is the scientists' responsibility to try to answer the particular issues raised by the interest groups, as well as to monitor and define general ecological effects.

By 1977, the Canadian scientific community was aware that these complex issues were not going to be resolved if a *laissez-faire* approach to monitoring continued (Varty, 1976; NRCC, 1977). It was recognized that indicative ecological monitoring requires recognition of the complex interrelations between forest management and ecological interactions. Since 1976, forest managers in Canada have been encouraged to cooperate with researchers in an effort to delineate the nature of spray programmes more precisely and to include the needs of the scientist in developing operational programmes, since monitoring and subsequent risk prediction apply only to relatively specific situations (NRCC, 1977). It has been accepted that, for economic and pragmatic reasons, the scope of ecological and toxicological monitoring of a pesticide used in large-scale operations must be limited and that risk-benefit analyses can take place only after the scenarios from the multiplicity of possible spray regimes are simplified to a point where indicative monitoring is possible (Varty, 1976).

This case study will concentrate on establishing principles which can be applied to ecological monitoring to maximize the indicative nature of the data obtained. Since 1977, scientists have adopted a much more organized approach to ecological monitoring in New Brunswick. Fenitrothion will be used as an example to clarify these principles.

Other aspects of the spray program, such as forest management scenarios and cost-benefit analysis, will not be discussed here. For discussion of such issues, the reader is referred to Baskerville (1976).

7.7.3 PRINCIPLES OF AN ORDERED APPROACH TO ECOLOGICAL MONITORING

Monitoring studies yield maximum information when focused on the critical path for a pollutant's fate, i.e. on the environmental compartments in which significant quantities of a pollutant are likely to accumulate and on the degradation processes most likely to produce toxicants. Establishing a critical path for pollutant fate involves the systematic characterization of the interactions between a chemical and the environment. The relative sensitivity of the different organisms that inhabit critical compartments can then be determined so that key indicator species may be identified and specifically monitored. A strategy for such an analysis (summarized in Figure 7.7.4) may be simplified to the sequential steps now described.

Step one: Characterize (1) the nature of the formulation initially loaded into the environment and (2) its basic physical and chemical properties

1. Characterization of a pesticide includes a complete description of the formulated solution as well as the designated active ingredients. Toxicity is a function not only of the amount of active ingredients, but also of the amount of contaminants, the nature of the adjuvants used in the preparation of the formulated product applied in the field and the age and the storage history of both the pure chemical and the formulation. The nature of the formulation also affects its volatility, deposition and degradation patterns. Frequent changes in the formulation make it difficult to establish patterns of effects from monitoring programs. Data obtained during the first year in which a chemical is monitored only suggest possible patterns of effects and behaviour, for observed effects may arise coincidentally to the spray programme and not as a result of the programme. If any aspect of the spray programme is changed in the second year, patterns can be masked by the change and the level of uncertainty is high. Patterns can be meaningfully established only if the formulation is kept constant over reasonable time periods.

The various fenitrothion formulations used in New Brunswick since 1968 are listed in Table 7.7.1. Fortunately, in New Brunswick the formulation was consistently water-based until 1977. In other regions (e.g. Quebec), the use of an oil-based formulation, which may have contained polyaromatic hydrocarbons, further complicates the assessment. Most early monitoring studies in New Brunswick neglected to indicate which formulation was used and formulations often changed; as well, laboratory studies focused almost exclusively on the active ingredient. These problems have been suggested as factors contributing to inconsistencies in the early literature (NRCC, 1975; Varty, 1976; NRCC, 1977).

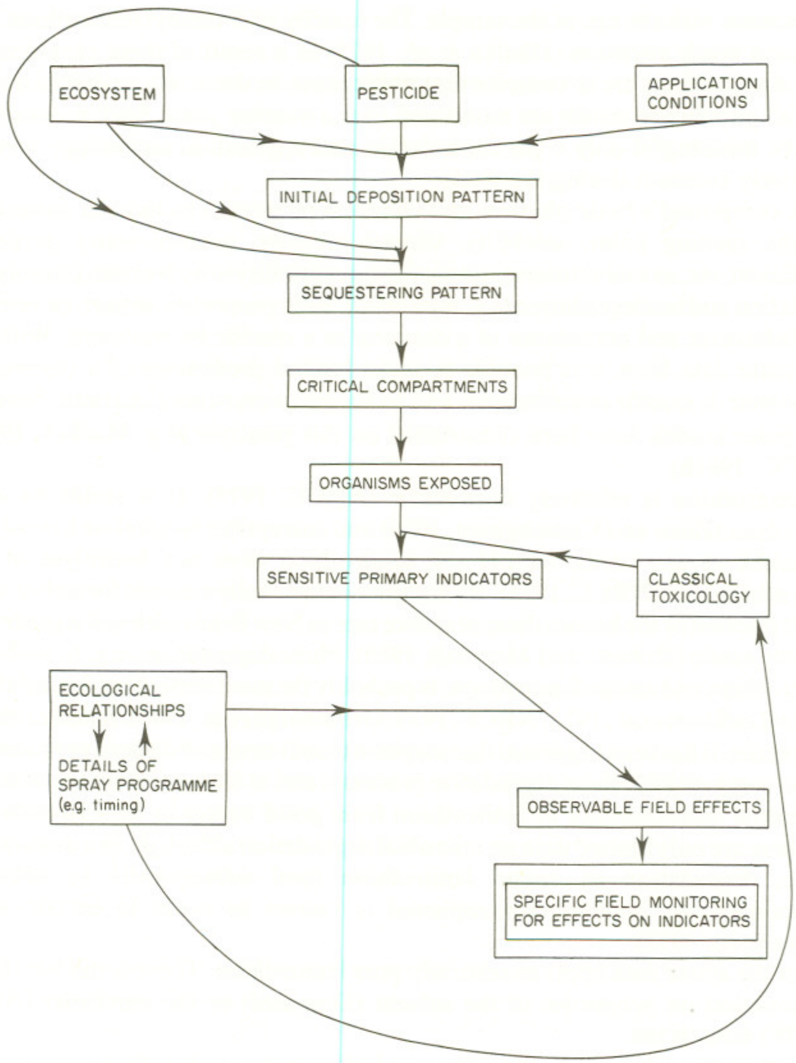

Figure 7.7.4 Strategy for an organized approach to ecological monitoring. Arrows indicate a rational sequence of investigation

One reason for the need to characterize the formulation stems from the variation in the toxicity of batches of technical fenitrothion (NRCC, 1975). In part, the variability has been shown to arise because of the formation of the *S*-methyl structural isomer of fenitrothion which may be more toxic than the parent compound (Greenhalgh and Shoolery, 1977). Also, the relative amount of this isomer, and possibly those of other degradation products and contaminants, tends to increase with the age of the sample. The toxicity of *S*-methyl fenitrothion has received much attention (Hladka *et al.*, 1977) as a result of these variations.

Some questions are so complex that they cannot be decisively answered unless overall population trends are monitored over a number years. Such monitoring can be meaningful only if the formulation and application technology remain relatively constant during the study.

2. A compound's basic physical and chemical properties include its molecular weight, melting point, solubility, vapour pressure, octanol-water partition coefficient, etc, as well as information on its susceptibility to hydrolysis, photolysis, oxidation and biodegradation (e.g. microbial). Such properties, in part, determine the behaviour and persistance of a chemical in a specific environment. With an adequate data base, it is possible to make limited predictions of a chemical's behaviour in specific environments and to suggest worst-case situations. Several computer models have been constructed on this principle (e.g. Mackay, 1979; NRCC, 1981b).

Fenitrothion is relatively nonvolatile (NRCC, 1975). It is stable to acid hydrolysis (Zitco and Cunningham, 1974) and susceptible to alkaline hydrolysis (Truchlik *et al.*, 1972). At $pH \leq 9$, fenitrothion does not hydrolyse at an appreciable rate (NRCC, 1975). Hydrolysis could usually account for only a very small fraction of the known disappearance rate as New Brunswick soil is generally slightly acidic (Roberts and Marshall, 1980). Photodegradation, e.g. to carboxy fenitrothion and amino fenitrothion, is probably the most important degradation route (Lockhart *et al.*, 1973; NRCC, 1975; Greenhalgh *et al.*, 1980). Based on these properties, it has been suggested that sorption to sediment and dispersive processes could compete with these degradative processes and at least partially account for the initial disappearance of fenitrothion from pond waters in New Brunswick.

There are only limited data on microbial degradation although, in a qualitative sense, degradation to amino fenitrothion (and subsequently to nitroso-compounds) or 3-methyl-4-nitrophenol is known to occur in certain soils (Miyamoto, 1977).

All of the above data apply to relatively pure fenitrothion. There is still very little information on properties of the solvent (Dowanol) or the emulsifier (Atlox 3409F) now in use.

Step two: Establish the nature of the loading of pollutant into the environment

This is the first step toward establishing the primary distribution pattern of a chemical in the environment, and hence towards elucidating which species are likely to be exposed to toxicologically significant levels of the compound. Forest pesticides are applied in the form of an aerial spray and such techniques are not amenable to fine control. After the pesticide is released from the aircraft, it forms a cloud which disperses as it descends. Atmospheric patterns (e.g. wind speed and direction), temperature, spray droplet size and the topography of the area all affect the amount of spray deposited and the deposition pattern (Armstrong, 1977). The amount deposited can be extremely uneven (at least ± 100 per cent of the theoretical deposition) and spray drift can cause a significant fraction of the total application to be deposited in nontarget areas (Crabbe *et al*., 1980 a, b).

The early history of fenitrothion use has been fraught with application problems. It has been applied over millions of hectares of forest (Table 7.7.1) and the deposition pattern has been uneven; some areas have received doses much higher than the supposed maximum (290 $\mathrm{g\,ha^{-1}}$) (Carrow, 1974). In New Brunswick, spray drift has caused increases of up to $3\,\mu\mathrm{g\,m^{-3}\,d^{-1}}$ in the total phosphorus content of the air in nearby towns downwind from spray plots (Yule *et al*., 1971). Areas have been accidently sprayed twice with fenitrothion (NRCC, 1975) and, in at least one case, a plot was sprayed with the wrong compound—phosphamidon instead of fenitrothion (NRCC, 1975). Under ideal conditions, only an average 50 per cent of the emitted spray is deposited in the target area (Armstrong, 1977). The need for a standard test protocol involving measurement of spray deposit, weather conditions, equipment and other aspects of aerial application has been recognized (Armstrong, 1977) in recent years attempts have been made to even out the deposition patterns. Computer models are being developed to facilitate prediction of deposition patterns and drift potential (e.g. Crabbe, 1980a, b), but human error and other unpredictable variables will still cause a high degree of uncertainty.

Step three: Characterize the ecosystem factors which affect initial deposition, sequestering and persistence patterns

The distribution of a chemical in the environment depends on the properties of the environment as well as on the properties of the chemical. The initial deposition pattern can be affected by topographical factors, the weather or the geological characteristics of the area. Similarly, the primary sequestering pattern will depend on ecosystem factors such as floral structure, soil type or geological characteristics. The rates of most degradative processes, and hence the persistance of a pesticide, are also directly affected by the environment and the sequestering pattern of the compound (Step four). Some examples will clarify these principles. Pesticide aerially applied to a densely foliated area will, at least initially, be intercepted by the foliage. If application is over a more exposed area, most of the chemical will reach the ground. If deposition is onto absorbent soil, it is likely that the compound will be rapidly absorbed but, if the ground is rocky, there will be little absorption and the compound might volatilize or photolyse from the surface at a much faster rate.

Fenitrothion is mostly applied over foliated areas and up to 80–90 per cent of the initial deposit is intercepted by the crown canopy of trees. Species inhabiting areas not protected by the crown canopy (e.g. small crown-canopy birds, meadow voles) are likely candidates to suffer immediate exposure and may be designated as indicators (see Step seven).

Step four: Establish the nature of secondary mobility, sequestering patterns and biological accumulation patterns

The initial deposit into a given environmental compartment will be transferred into other compartments at rates determined by both the nature of the chemical and the environment. It is, therefore, necessary to establish mobility patterns and, from this, secondary sequestering patterns. For example, a compound initially deposited onto crown canopy foliage might eventually reach the forest floor and sequester in the soil litter or accumulate in soil invertebrates.

The amount of compound that will be sequestered depends on the relative rates of degradative and removal processes operative within the compartment receiving the initial load (see Step five). It is necessary, at this point, to determine whether the compound will simply be degraded upon deposit or be transferred in a manner that can lead to its accumulation in various other compartments including flora and fauna. As well, it is necessary to consider whether toxic degradation products are formed and sequently accumulated in specific compartments.

We have stated that a major amount of the fenitrothion initially deposited on crown canopy foliage and open areas is lost quite rapidly (NRCC, 1975), probably

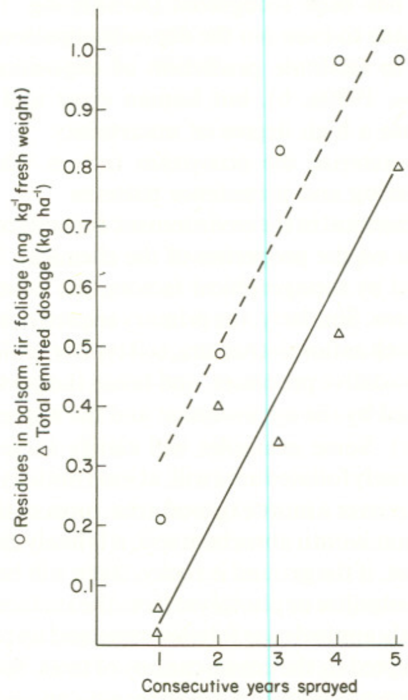

Figure 7.7.5 Persistent residues in mixed-age balsam fir foliage, and total dosage, versus the number of consecutive years a location in New Brunswick was sprayed with fenitrothion. (Adapted from Yule, 1974)

via photodegradation, but about 10 per cent is absorbed by the coniferous foliage (Yule and Duffy, 1972). Material sequestered in this matrix is relatively stable because waxy leaf pigments attenuate light and limit photolytic degradation. If there are annual applications, these residues accumulate from year to year (Yule, 1974) and are proportional to the number of annual applications and the total dosage of fenitrothion (Figure 7.7.5). This is one of the few environmental compartments in which fenitrothion is known to be persistent from one year to the next. It is likely, but has not been established, that forest needle litter will also contain such residues. Organisms inhabiting these compartments, e.g. birds, defoliating insects and possibly soil organisms (including decomposers), are likely to receive higher than average exposure to fenitrothion. Such organisms are thus possible indicators and are being monitored for acute and chronic toxic effects and for population shifts.

Fenitrothion reaching the ground, or the surface of water bodies, will be transferred to and sequestered in other compartments. The rates and extent of such transfers will depend on the properties of the local environment. For example, the extent of absorption into soil or aquatic sediments will depend on their sorptive capacity, i.e. their organic content. Since fenitrothion is only moderately hydrophobic, this effect will be less significant than for a very hydrophobic compound such as DDT. Still, conceptually, benthic organisms and filter feeders could be exposed to higher than average concentrations of fenitrothion and they are being studied as indicators. So far, there is no evidence that fenitrothion is bioconcentrated in the food chain to higher mammals (Miyamoto, 1977). However, organophosphorus insecticides are accumulated in tadpoles up to 60 times the levels in water (Hall and Kolbe, 1980) and may possibly be concentrated in the lower levels of the aquatic food chain. Animals consuming dead budworm or other defoliators may also be exposed to higher than average levels of toxicants.

Step five: Characterize the situations which will maximize and minimize persistence

The relative persistence of a pollutant is dependent upon its rates of degradation by various processes. The rates of these degradative processes can vary in different environments. Degradative processes can be directly influenced by the environment or by the partitioning pattern of the pollutant which can affect the fraction of pollutant accessible for degradation through a given process. It is theoretically possible to identify the environmental factors which will maximize or minimize the persistence of a given pollutant, although interacting effects make this extremely complex. For example, photolysis will be maximized in summer and in situations where the pollutant is exposed to direct sunlight over a large surface area (e.g. leaves or a rocky surface) and minimized in winter and in situations where light attenuation is high. The extent of microbial degradation is likely to be maximal where the bulk of the pesticide is sequestered in organic soil, organic sediment, eutrophic water bodies, etc. However, the situations which maximize microbial degradation tend to minimize photolysis. Since the extent of partitioning of pollutant into, e.g. organic soil, depends on the lipophilic nature of the chemical as well as on the soil, the effects of the environment on persistence will not be the same for different chemicals. However, within a chemical class (e.g. the class of extremely hydrophobic compounds) the patterns should be comparable.

In the case of fenitrothion, we know that microbial degradation and photolysis are the important identified degradation processes, even though microbial breakdown is not well characterized. If we assume that photolysis is often the dominant process, then fenitrothion should be least persistent when deposited on a rocky surface with little sorptive capacity. Persistence would be maximum when the compound was deposited on a sorbant material. However, if microbial degradation is dominant, the situation could be just the reverse.

Step six: Identify general sensitivity patterns and potential indicator organisms

We have discussed the importance of establishing which compartments are most likely to receive a transient or persistent exposure to a pollutant or a toxicologically significant degradation product. The next step is to analyse the organisms that live in these compartments, in terms of their relative susceptibility to damage by the pesticide or associated chemicals. Initially, this analysis involves the use of data obtained through classical laboratory toxicological studies to identify primary indicators (organisms most likely to be directly affected by contact with the toxicant). The lethal-effect dose and concentration are useful in establishing the relative sensitivity of different organisms to lethal effects. For economic reasons, it may not be practical to establish these parameters for all the organisms that are potentially exposed. Known trends in toxicological response may be used to guide the choice of laboratory tests. For example, relationships between size and sensitivity may exist for certain types of response. It must be remembered, however, that such trends are tenuous and variations in response may occur even within groups of very similar organisms. The response can also vary depending on the route of exposure and it is necessary to examine likely routes before defining relevant laboratory studies.

At the ecosystem level, effects on an individual organism are less important than effects on a species (see Chapter 4) and individual effects may be difficult to detect in the field because of natural fluctuations. Thus, individual effects are often not the best indicators of ecosystem perturbation. Indicators should be selected to reflect the actual doses likely to be encountered in the field. *It is of no use to study a highly sensitive species that is not exposed*! Subacute and chronic effects will be important at the ecosystem level only if they have the potential to cause population shifts, alter the behavioural pattern of a species or start a chain of effects. For example, teratogenic or reproductive effects could be significant if spraying coincided with the reproductive season. These ideas will be discussed in Step seven. Ideally, dose-response curves should be established for likely indicators since, if the curve is steep, small changes in deposition rates could drastically alter effect patterns. In practice, it is usually too expensive and time consuming to do this for many organisms. At best, laboratory programmes identify only a few likely sensitive primary indicators which can serve as pointers to the most productive approach to indicative field monitoring. The population responses and life cycles of the primary indicators, as well as their interactions with other components of the ecosystem, must be considered to determine the most sensitive field indicators (Step seven). It is necessary to develop a loop of laboratory–field–laboratory monitoring which allows the continued improvement of the sampling matrix and hence the quality and precision of indicative monitoring.

In the case of the fenitrothion spray programme in New Brunswick, several primary indicators have been identified. The crown canopy is a critical compartment in terms of primary exposure since much of the initial deposit is intercepted by the canopy and fenitrothion can accumulate in coniferous foliage (Step four). Organisms which inhabit this compartment, e.g. birds, will potentially suffer various levels of adverse effects as a function of their sensitivity to fenitrothion and the dose they encounter. Sensitivity varies dramatically between and within species (Figure 7.6.6). Initial field monitoring programmes found fenitrothion to have little impact on birds because these programmes did not monitor the species which, in terms of sensitivity, would be good indicators, i.e. small crown-canopy birds (NRCC, 1975). For example, no effect was

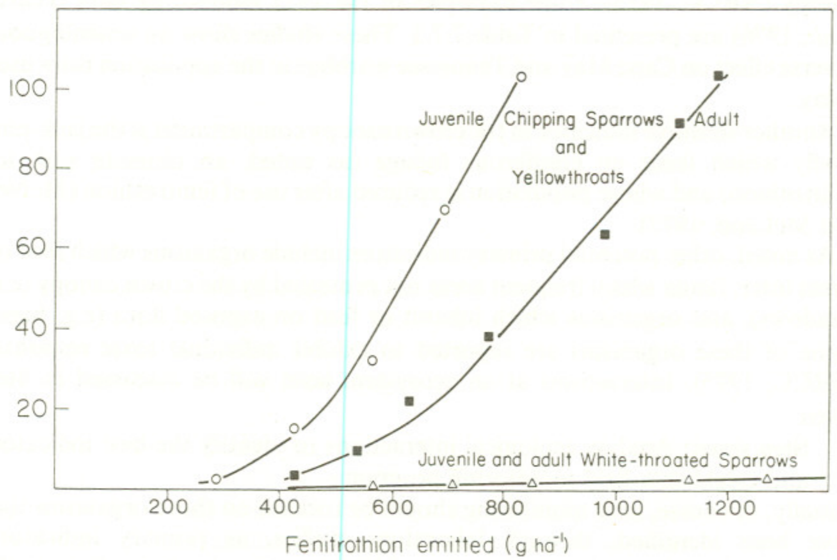

Figure 7.7.6 Mortaility curves for juvenile and adult sparrows and yellowthroats. (From Buckner, 1975. Reproduced by permission of National Research Council of Canada)

Table 7.7.3 Effect of fenitrothion on number of singing males (mean, standard deviation) recorded on transects[1] (data from Pearce *et al.*, 1976)

Species	Prespray count	Post-first spray count (175–280 g ha^{-1})	Post-second spray count (175 g ha^{-1})
Cape May warbler	14.3 ± 3.3	6.2 ± 3.7[3]	–
	11.6 ± 2.7	4.3 ± 1.2[3]	6.2 ± 2.0
Tennessee warbler	11.3 ± 6.1[2]	32.5 ± 9.5	–
	43.5 ± 5.3	38 ± 4.0[3]	23.2 ± 3.4[3]
	4.6 ± 2.4	0.3 ± 0.5[3]	0.0 ± 0.0

[1] Counts taken in several spray areas.
[2] Tennessee warbler numbers were very low pre-spray, indicating that much of the population had not yet arrived.
[3] Reduction is significant at 95 per cent level.

observed on populations of sapsucker (Rushmore, 1971), a relatively large bird which does not generally nest in or inhabit the forest crown. Another influence which may explain the original failure of monitoring programmes to detect effects is that census techniques have an unavoidable, built-in negative bias (Pearce, 1968; NRCC, 1975). Despite this, patterns of effects have been reported in small songbird populations, particularly those which inhabit the crown canopy (NRCC, 1975). Some examples of the 1975 monitoring data (Pearce *et al.*, 1976) are presented in Table 7.7.3. These studies show an unambiguous adverse effect on Cape May and Tennessee warblers at the application rates used today.

Another sensitive indicator in the crown canopy compartment is the jack-pine sawfly which feeds on coniferous foliage (as noted, an ultimate sink for fenitrothion) and whose population is reduced after use of fenitrothion (McNeil and McLeod, 1977).

As noted, other potential primary indicators include organisms which feed on forest litter, fauna which frequent areas not protected by the crown canopy (e.g. meadows), and organisms which inhabit or feed on exposed flora (e.g. bees). Some of these organisms are reported to exhibit individual toxic responses (NRCC, 1975). Interactions at an ecosystem level will be discussed in Step seven.

Step seven: Analyse ecological interactions to identify the best indicators and establish a field monitoring program

Initially, of course, field monitoring should be focused on those organisms that have been identified, through laboratory studies, as primary indicators. Emphasis should be placed on acute or chronic responses depending on the organism studied and the nature of the exposure (see Steps three to six). However, as noted, primary indicators may not be the most sensitive or useful indicators of ecosystem perturbation. Population and ecosystem level responses can be quite different from, and sometimes more dramatic than, those detected at an individual level. A field monitoring programme requires consideration of the many possibilities involving species affected via a chain of effects including the following:

1. Species that are particularly sensitive due to an interaction between their life cycle, e.g. migratory or reproductive period, and the timing of the spray programme.
2. Species that are actively foraging at the time of application.
3. Species with a critically low population (e.g. endangered species) where even a small percentage increase in mortality could be critical in terms of stability of the species.
4. Situations where an entire population might be temporarily concentrated in a small area (e.g. migrating bird flocks) making them particularly vulnerable. In such cases, unusually high loading due to variable deposition patterns could have a devastating effect on the entire flock if the dose-response curve is steep and the average application rate is close to an effect level.
5. Synergistic interactions with other pollutants or with natural factors.
6. Species with hierarchical social organization where certain individuals may be fundamental to the structure and survival of a population.

It is clear that ecological effect monitoring is extremely complex. The lack of baseline data on flora–fauna sturcture is perhaps the biggest problem. However, it is not necessary to adopt a random approach to field monitoring. Some of the interactions which help identify indicators can be rationalized. Often, field monitoring will indicate a need for further laboratory assessment.

Eventually, a sampling matrix will evolve which maximizes the opportunity for detecting small changes in complex environments. Ultimately, the most difficult point of the analysis is the prediction of long-term ecosystem consequences of apparently small perturbations, in the light of natural catastrophies, etc. (e.g. NRCC, 1981a).

The fenitrothion spray programme presents good examples of the points outlined above. As already noted, bees are often exposed to lethal concentrations of fenitrothion when it is applied to nontarget pollen and nectar sources. The overall impacts on specific populations of pollinators can range from negligible to drastic, depending on interactions between the life cycle and behaviour patterns of the species, the specific ecology of the area and the timing of the spray programme. Furthermore, it is difficult to distinguish between pesticide and environmental effects on the populations because of the lack of baseline data on temporal trends in populations.

Inital studies on the effects of fenitrothion on colonies of honeybees (Buckner, 1975) indicated that the application of fenitrothion at 275 g ha^{-1} had few long-term effects on hive stability. This result was interpreted to indicate that the pollinating force remained unaffected. Substantial numbers of dead bees were found adjacent to the hives but, since they represented only 1 per cent of the total hive population, the net impact was assumed to be very low. The key questions, the percentage of active foraging bees killed and hence the short- and long-term impact on pollinating activity, were not addressed in the discussion. Although the observed mortality was most likely restricted to the foraging bees directly exposed to the spray, experience with other pesticides has now shown that the level of kill observed in these studies is indicative of severe mortality in the foraging component (NRCC, 1981a).

While there were initial inconsistencies in the results from different field monitoring programmes (NRCC, 1981a), it is now apparent that when fenitrothion is sprayed at operational rates, significant population reductions may occur in nearly 100 species of native bees in New Brunswick (Thorpe, 1979; NRCC, 1981a). Plowright (cited in NRCC, 1981a) reports that pollinating bees are almost non-existent in the centre of spray blocks soon after spraying. Our poor understanding of the deposition patterns associated with particular spray programmes, and the limitations of the census techniques now available for pollinators, make direct quantification of the level of impact impossible. Analysis of the problem has primarily relied upon the recognition of perturbations in ecological patterns involving bees, their individual sensitivity, preferences for pollen sources and the specific pollination requirements of some plants.

Bumblebees have received particular attention in this spray programme because the overwintering queen emerges alone early in the spring, i.e. usually just before the spray programme. Bumblebee colonies are reestablished annually and, if the queen is killed before a colony is established, there would be a drastic effect on population levels over the season. Plowright *et al.* (1978) observed up to 100 per cent mortality when caged bumblebees were sprayed with fenitrothion at 206 g ha^{-1} (operational levels). The degree of mortality depended on the amount of coverage afforded by the forest canopy. Populations of bumblebees in spray areas remained depressed for several months after the spray. Solitary bees will potentially be at high risk because they forage in open meadows for long periods during the spray season.

The identification of appropriate secondary indicators of pollinator effects requires careful analysis of the preference of pollinators for different plants, the period of bloom relative to the spray period and the dependence of the plants on native bees for pollination. Preference of pollinators for specific plants has been demonstrated by the difference in the variety of pollen types collected from captured bumblebees in sprayed and unsprayed areas (Figure 7.7.7). Fewer pollen varieties were collected in sprayed areas, suggesting a bias towards a few preferred species of plants as pollinator density, and hence competition, decreased (Plowright *et al.*, 1978). All other factors being equal, the less-preferred species of plants should be the best indicators of when the pollinator force is reduced. As already suggested, the variation in blooming periods may (at

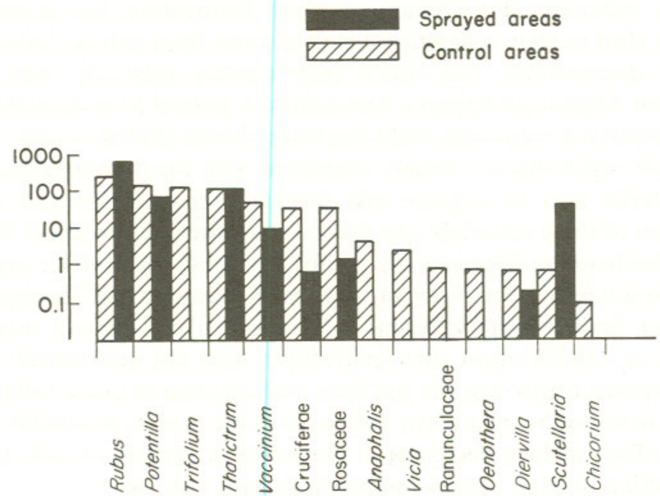

Figure 7.7.7 Ranked abundance of pollen types collected by bumblebees in fenitrothion sprayed and unsprayed areas of southwest New Brunswick. (From Plowright *et al.*, 1978)

least partly) account for contradictions in the reports about fruit-set. Thaler and Plowright (1980) have found unambiguous reductions in fruit-set of plants that bloomed shortly after application of fenitrothion. Buckner (cited in NRCC, 1975, 1981a) did not see any reduction in fruit-set in plants that bloom at periods which do not coincide with the spray period. One would anticipate that plants with short blooming periods which coincide with the spray period would be particularly vulnerable and likely indicators. The low-bush blueberry, dependent on bees for pollination, is now known to be adversely affected by the fenitrothion spray programme to such an extent that compensation has been paid to growers (e.g. $58,000 to Bridges Brothers Ltd; The Supreme Court of New Brunswick, 1976).

The implications of these observations in terms of floral diversity and stability have not been assesed, mainly because of the lack of demonstrated protocols for assessment of the problem.

The NRCC Panel (NRCC, 1981a) concluded that:

> It is in this area that we should be most concerned about our current ignorance: we are not yet able to forecast the persistence of the secondary effects of reductions in genetic variance in plant populations, although we may list some possible effects.

Pesticide-pollinator interactions emphasize the importance of an organized, systematic approach to ecological monitoring.

The fenitrothion spray programme presents other examples of possible secondary indicators. Pine foliage in which fenitrothion has accumulated is eventually shed to form a significant part of forest floor debris. Although it has not been documented, this debris may contain relatively high levels of fenitrothion. Organophosphorus insecticides in general have been shown to be toxic to certain soil organisms, including invertebrates (Edwards and Thompson, 1973). Such organisms are closely associated with the decomposition of dead plant material into its organic and inorganic constituents and in the incorporation of these materials into the soil structure (Edwards and Thompson, 1973). If fenitrothion decreases the numbers of these decomposer organisms, a decrease in soil fertility might eventually result. Edwards and Thompson (1973) report that fenitrothion ($4.5\,kg\,ha^{-1}$) is toxic to certain soil invertebrates. Spillner *et al.* (1979) report that fenitrothion does not qualitatively affect the microorganisms which degrade leaf litter and cellulose in forest soild, but these questions have not been carefully addressed in the field. Conceivable indicators for such effects include decreases in the nitrogen content of soil, the rate of decomposition of total organic matter in surface soil, etc.

The possibility that fenitrothion may accumulate in tadpoles has already been raised (Step four). If this happens, potentially toxic levels of fenitrothion could accumulate in amphibians and consequently carnivorous species could be exposed to toxic levels (Hall and Kolbe, 1980). Spray programmes aimed at the adult budworm, and hence conducted later in the summer, might be particularly hazardous to young ducks, etc. This group of organisms, which are not exposed to any direct source nor to any obvious secondary exposure vector, have not been studied.

7.7.4 CONCLUSIONS

Experience in New Brunswick demonstrates the need for a close liaison between the forest manager who mounts the control programme and the scientists who assess its impact on the forest ecosystem. The manager has an array of choices to make, e.g. which application technique to use, the nature of the product(s) to be used, the formulation, the timing and extent of the programme, etc. As well, the harvest patterns chosen by the manager strongly influence population dynamics and the effectiveness of natural control mechanisms and hence, eventually, the need to spray (Baskerville, 1976).

The integral relationship between management's decisions and the impact of the control strategy on both the pest and other organisms is depicted in Figure 7.7.8. Ultimately, each decision feeds back to the forest manager who is thus the only person who can create a stable situation conductive to the development of a productive assessment program. In the past, the absence of a stable programme with an effective liaison between managers and scientists made early efforts at assessment generally equivocal. Since the liaison has improved, a point has been reached where there is sufficient evidence to conclude that a

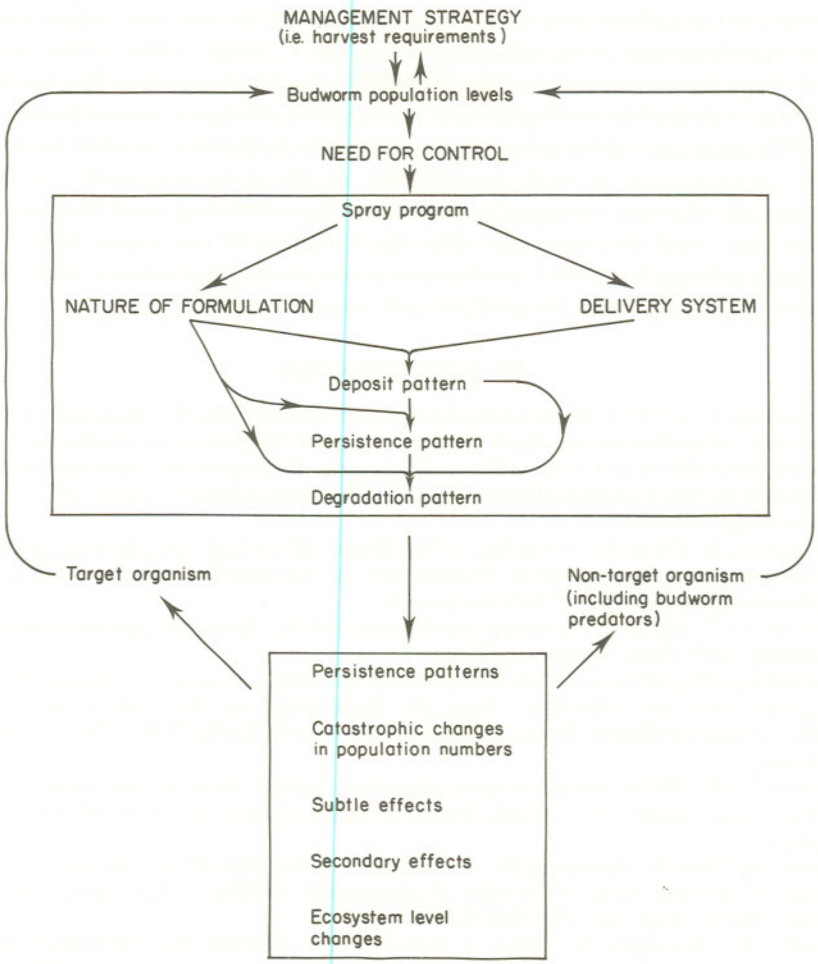

Figure 7.7.8 Management decision scheme. Decision points are capitalized. Arrows indicate possible consequences of decisions

number of indicator organisms are being affected. These include pollinators and certain other beneficial insects, small crown-canopy birds and aquatic invertebrates. To date, the overt impact of the spray programme has been judged tolerable by the forest managers (Varty, 1980). The critical question is whether undesirable long-term side-effects will be associated with a harvest policy that ecologists believe perpetuates the outbreak and hence the need to spray (Baskerville, 1976).

Unfortunately, it is also the conclusion of the ecologists that our understanding of the specific fundamental ecological relations operative in a given forest system are too rudimentary to permit assessments of the long-term implications of such programmes at the ecosystem level (NRCC, 1981a; Varty, 1980). In the face of the sheer complexity of the problem, it has been suggested that the first productive step is the development of management strategies that minimize the need for large-scale spray programmes and hence the need to consider the long-term consequences of such programmes at the ecosystem level. In New Brunswick, alternate strategies for the harvest have been suggested (Baskerville, 1976) that could fundamentally alter the dynamics of the spruce budworm, bringing natural biocontrol mechanisms more nearly into balance, and significantly reduce the need for artificial pest control strategies.

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