FENITROTHION RISK ASSESSMENT

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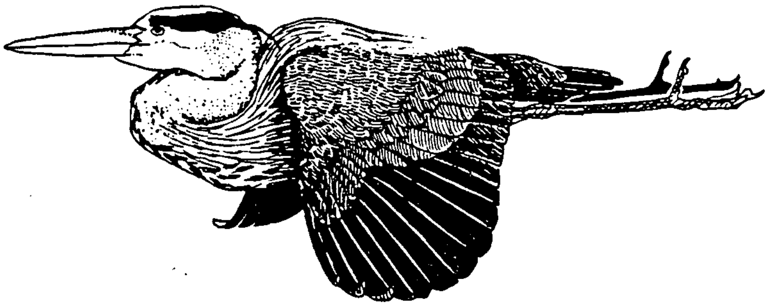

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PREFACE

The following Technical Report was prepared as part of a special review of the insecticide fenitrothion, initiated by Agriculture Canada under the authority of the *Pest Control Products Act*. The fenitrothion risk assessment contained in this report is intended to help Agriculture Canada render a decision on the future regulatory status of fenitrothion. Due to the technical nature of this risk assessment, it is being released as a part of the Technical Report Series issued by the Canadian Wildlife Service of Environment Canada.

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1. Physicochemical properties

The vapour pressure of fenitrothion is reported to be 2.14×10^{-4} mmHg at 25°C, which indicates that fenitrothion has an intermediate to high volatility under field conditions. Fenitrothion has the potential to volatilize from soil, particularly moist soil. Volatilization from the surface microlayer is the major means of dissipation for fenitrothion sprayed on natural waters. Water solubility values reported for fenitrothion are 5 mg/L at 10°C, 14 mg/L at 30°C, and 31 mg/L at 50°C, which indicate that fenitrothion is soluble in water. The octanol/water partition coefficient (log K_{ow}) for fenitrothion is reported to be 3.43 at 20°C, which indicates that fenitrothion has a potential for bioaccumulation in the environment. No raw data or experimental methodologies were included by the proponent to support these reported physicochemical values.

2. Transformation

The half-lives of fenitrothion via hydrolysis are 191–200 d at pH 5, 180–186 d at pH 7, and 100–101 d at pH 9. These results indicate that fenitrothion is hydrolysed under basic conditions and is more resistant to hydrolysis under neutral and acidic conditions. The major transformation product observed at pH 9 was 3-methyl-4-nitrophenol. The hydrolysis of fenitrothion and the formation of 3-methyl-4-nitrophenol are likely to proceed very slowly under the environmental conditions found in most eastern Canadian watercourses.

Phototransformation of fenitrothion on soil surfaces and in air is considered slow. Phototransformation studies conducted in water at 25°C indicated that fenitrothion had half-lives of 3.7 and 141 d under light and dark conditions, respectively, indicating that the compound can undergo phototransformation quite readily in water. The major transformation product observed was p-nitro-m-cresol, which disappears through photolytically mediated polymerization to form substances resembling fulvic or humic acids, which may then be bound to organic matter or remain in solution.

Aerobic soil transformation studies conducted using a number of soil types indicated that fenitrothion and its major transformation product 3-methyl-4-nitrophenol are microbially transformed and are not persistent in forest soils.

Aerobic aquatic transformation studies using sediments have shown that fenitrothion and its major transformation product 3-methyl-4-nitrophenol are not persistent under aerobic aquatic conditions and that microorganisms in natural sediments and water play an important role in the decomposition of fenitrothion. Anaerobic aquatic transformation studies have indicated that fenitrothion and its major transformation products aminofenitrothion and 3-methyl-4-nitrophenol are associated with sediments and are not persistent under anaerobic aquatic conditions.

3. Mobility

Laboratory studies have indicated that fenitrothion has the potential to be mobile in coarse-textured soils with low organic matter content. Fenitrothion will desorb from all soils and sediment. The major transformation product 3-methyl-4-nitrophenol was observed to be more mobile than the parent compound.

4. Field dissipation

Canadian field dissipation studies have shown that fenitrothion is present in low concentrations (<0.005–0.1 μg/g) in forest soil and is not persistent following operational aerial applications to control the spruce budworm *Choristoneura fumiferana*. Fenitrothion and its major transformation products do not leach appreciably in forest soils when applied according to label instructions.

Concentrations of fenitrothion in lotic (flowing) aquatic systems measured following operational spraying have ranged from 1.3 to 127 μg/L. Fenitrothion concentrations usually declined to less than 1.0 μg/L within 24–48 h and to less than 0.5 μg/L within 1–6 d. The half-life of fenitrothion in stream water is estimated to be 6–10 h.

Maximum aqueous concentrations of fenitrothion in lentic (standing water) systems after operational sprays have usually occurred within 2 h after the start of spraying and have ranged from 0.38 to 2500 μg/L. Mean surface water concentrations of fenitrothion in small ponds directly oversprayed with fenitrothion at an application rate of 210 g a.i./ha ranged from 20 to 1500 μg/L. A large fraction (up to 70%) of the fenitrothion applied to lakes and ponds is rapidly volatilized from the surface layer (half-life <0.5 h). Fenitrothion concentrations at depths of 0.3 and 1 m in acid bog ponds were 0.25–0.5 times the concentrations observed in the surface layer 15 min postspray. However, concentrations observed at the greater depths were not as transitory (half-life about 1 d) as the high residues observed in surface water. Aqueous fenitrothion residues reappeared 1 year after application to an acid bog pond, suggesting that some component of that ecosystem, possibly moss, acted as a reservoir.

Concentrations of fenitrothion in aquatic sediments after operational spraying were less than 0.5 μg/g and fell below detectable levels 2 d postspray in a small pond in a spruce-fir forest in New Brunswick. Aminofenitrothion, the major transformation product, persisted for less than 4 d.

Numerous Canadian field studies have indicated that fenitrothion can persist and possibly accumulate for periods of more than 1 year at concentrations of approximately 1 μg/g in conifer foliage.

Environmental toxicology

1. Soil microorganisms and arthropods

Concentrations of fenitrothion found in forest soil immediately following operational applications to control spruce budworm are not expected to adversely affect soil microorganisms or microbial processes. Populations of soil and leaf litter arthropods were reduced by fenitrothion application, but none of the species was eliminated, and numbers recovered by the following year. However, the effects of repeated applications of fenitrothion on the soil invertebrate community and the effects of any changes in the invertebrate community on soil processes remain unknown.

2. Nontarget terrestrial arthropods

Arthropod biomass and numbers may be reduced by up to 35% and 50%, respectively, following operational applications of fenitrothion to control spruce budworm. Despite this large mortality, most invertebrate populations are affected only temporarily, because a proportion of the population of any arthropod species is likely to avoid contact with the insecticide owing to the existence of refugia in unsprayed foliage within the spray block and within individual trees. Swaine jack-pine sawfly *Neodiprion swainei* and balsam fir sawfly *N. abietis* populations may be suppressed by persistent fenitrothion residues in conifer needles. Long-term population densities of arboreal predaceous insects and spiders are relatively stable, but a few species (e.g., the ladybeetle *Mulsantina hudsonica*) remain scarce in sprayed areas.

The variability of many invertebrate predator populations over time makes interpretation of results difficult. Conventional fenitrothion spraying to control spruce budworm larvae may virtually eliminate late larval and pupal parasites from spray blocks by killing adult parasites before they have found a host suitable for oviposition. Thus, continued fenitrothion spraying may be weakening a section of the biocontrol complex operating upon budworm larvae and pupae.

3. Pollinators

Laboratory studies have indicated that fenitrothion is highly toxic to honeybees and other native pollinators. Numerous field studies have observed mortality in foraging bees and a decline in the foraging activity of honeybees in fenitrothion-treated plots compared with control plots. Aerial applications of fenitrothion at 210 g a.i./ha have caused a high mortality of native pollinators, including bumblebees, solitary bees, and vespid wasps. Time for recovery of pollinator populations ranged from 3 to over 10 years, depending upon the severity of the reductions. Fenitrothion spraying has also been observed to result in shifts in pollinator behaviour. For example, bumblebees foraging in habitats sprayed with fenitrothion foraged on fewer plant species than bees in unsprayed locations, perhaps because of reduced competition in the treated areas.

Population reductions of honeybees and wild bees have been correlated with reduced seed and fruit set (plant fecundity) in various plants in natural and agricultural ecosystems. Crop losses of blueberries in New Brunswick between 1970 and 1977 due to fenitrothion spraying in surrounding forests were estimated at 670 000 kg. The implementation of buffer zones around blueberry fields has eliminated this problem in recent years. Forest plant species for which an association between reduced fecundity and fenitrothion spraying has been demonstrated include wild sarsaparilla *Aralia nudicaulis*, with a 30% reduction in fruit production; corn lily *Clintonia borealis*, 17–36% reduction; bunch-berry *Cornus canadensis*, 44% reduction; sheep-laurel *Kalmia angustifolia*, 52% reduction; wild lily-of-the-valley *Maianthemum canadense*, 27% reduction; and red clover *Trifolium pratense*, 78% reduction. The effects of the observed reductions in plant fecundity, specifically fruit and seed set, on the population biology of affected plants are unknown.

The impact of fenitrothion on forest pollinators and pollination is a definite concern. Fenitrothion-mediated reductions in pollinator abundance undoubtedly add some element of risk to the sexual reproduction of boreal forest herbs. However, population-level effects such as changes in age structure, changes in the ratio of male to female plants in an area, and inbreeding in plants have not been investigated to date. Impacts on plant populations are likely to be difficult to measure, because many flowering forest plants are long-lived perennials that, besides sexual reproduction, have well-developed clonal growth. In the absence of studies addressing sublethal effects on pollinators and the consequences of reduced production of seed and fruit for forest wildlife dependent on those food resources, a complete understanding of the impact of spraying with fenitrothion on the forest ecosystem due to effects on forest pollinators is not possible at this time.

4. Algae

The 96-h EC50 of fenitrothion for *Selenastrum capricornutum* was 1300 μg/L. Operational applications to control spruce budworm are not expected to be toxic to algae inhabiting small ponds.

5. Aquatic invertebrates

The 48-h static LC50 of fenitrothion for *Daphnia magna* was reported as 8.6 μg/L, with a no-observed-effect concentration (NOEC) of <2.0 μg/L. The 48-h flow-through LC50 of Sumithion 8E to *D. magna* was reported as 2.3 μg/L, with a NOEC of 1.0 μg/L. These reported laboratory acute toxicities indicate that Sumithion and its active ingredient fenitrothion are highly toxic to aquatic invertebrates. These acute toxicity values are well within the range of concentrations measured in small lentic habitats.

Field studies examining the effect of operational applications of fenitrothion on aquatic invertebrates in lotic systems have generally indicated an increase in drift of invertebrates from a number of different orders, which in some cases included dead organisms. Most peaks in drift occurred within the first 12 h after treatment, and the effects generally lasted less than 24 h. In the majority of these studies, no decreases in benthic populations of aquatic invertebrates were observed after spraying. In those cases in which short-term reductions in some groups did occur, the effects were transitory.

Operational sprays of fenitrothion should have only minor effects on the invertebrates associated with large lentic water bodies, as large lakes are protected by no-spray buffer zones that limit the amount of insecticide contacting the water. In addition, dilution in large bodies of water would serve to further reduce the hazard to resident biota.

An experimental ground application of fenitrothion at a rate of 210 g a.i./ha to small bog ponds produced residue concentrations that were within the range of those that have been measured after operational spraying. The application reduced emergence of a number of orders of invertebrates for a period of 12 weeks. Benthic biomass was reduced by about 50%, and recovery to control densities required about 1 year. The implications to wildlife inhabiting these areas are unknown at present.

Small ponds other than bog ponds are difficult to see from the air and hence difficult to protect with buffer zones; in addition, they have little forest canopy to screen out deposit, a small capacity for dilution, and low water turnover rates. The invertebrates inhabiting these ponds could therefore be potentially at risk from forest spraying with fenitrothion. Mean surface water concentrations of fenitrothion in small ponds (<0.5 ha) directly oversprayed with fenitrothion at an application rate of 210 g a.i./ha ranged from 20 to 1500 μg/L. These concentrations are 10–200 times greater than the 96-h LC50s for some aquatic invertebrates (see above). It must be recognized, however, that maximum surface water concentrations are rapidly attenuated as a result of dilution, transformation, and volatilization and are not directly comparable with concentrations of fenitrothion used to develop laboratory toxicity values. Therefore, the toxicities measured by conventional procedures (i.e., 24- to 96-h LC50s) may not accurately predict the effects of the short-pulse exposures to fenitrothion that organisms receive in the field. However, field monitoring has shown that concentrations of fenitrothion at depths of 0.3 and 1 m in acid bog ponds, although 0.25–0.5 times the concentrations observed in the surface layer, were not as transitory (half-life about 1 d) as the high residues observed in surface water. When water sampled from a small pond in New Brunswick that was operationally sprayed with fenitrothion in 1991 was bioassayed against *D. magna* in the laboratory, 100% mortality of test organisms resulted after 48 h at dilutions down to 12.5%.

Small ponds (<5 ha in size) are currently not protected by buffer zones in most provinces, and the impact of the forestry use of fenitrothion on aquatic invertebrates in these habitats may be substantial. The potential exists for significant effects on vertebrate fauna that depend on energy flow through these trophic levels, but no studies that address this concern have been conducted to date.

6. Amphibians and reptiles

Laboratory studies have shown that fenitrothion can be toxic to frog eggs and tadpoles, with the toxicity depending upon the duration of exposure. Acute LC50s for tadpoles of the green frog *Rana clamitans* are 9.9 mg/L after 24 h, 4.9 mg/L after 96 h, and below 4 mg/L after 160 h (with this exposure duration, there was 100% mortality at 4 mg/L). Amphibian eggs appear to be more susceptible than tadpoles to the toxic effects of fenitrothion; when developing Indian bullfrog *Rana tigrina* eggs were subjected to long-term fenitrothion exposures in the laboratory, effects on larval development were seen at fenitrothion concentrations as low as 0.01 mg/L.

Symptoms of fenitrothion exposure of subadult amphibians include developmental arrest, hemorrhagia, swimming difficulty, buoyancy problems, poor pigmentation, and curvature of the body axis. With green frog tadpoles, behavioural changes occurred at fenitrothion concentrations above 1 mg/L. Because initial fenitrothion residues in the surface layer of small forest ponds may be above this concentration following operational spray programs, there may be only a small margin of safety for developing amphibians during forest insect control programs using fenitrothion. However, because these concentrations do not persist for very long, the greater threat to amphibians may stem from a reduction in their invertebrate food resource.

In eastern Canada, field monitoring studies have not detected mortality of adult amphibians after operational fenitrothion applications. This is not unexpected, because of the demonstrated ability of adult amphibians to tolerate acute organophosphorus (OP) insecticide exposure. Most monitoring studies have been concerned with finding dead adult amphibians and have not examined longer-term effects on amphibian populations that might occur through removal of invertebrate food resources or effects on amphibian developmental stages. Census data collected during a recent study in northern New Brunswick indicated that mink frog *Rana septentrionalis* population densities were consistently lower in areas where fenitrothion had been used most frequently in the 5 years preceding the census. These data suggest that fenitrothion may be having detrimental impacts on the long-term viability of these frog populations, although the influence on frog densities of other habitat quality parameters measured at the study sites could not be factored out.

No information was found on the effects of fenitrothion on reptiles.

7. Fish

The acute toxicity (96-h LC50) of technical fenitrothion to various species of fish ranges from 1000 to 5000 μg a.i./L. Similar levels of toxicity are observed with Sumithion

20F, a flowable formulation. Of the various life stages tested, rainbow trout *Oncorhynchus mykiss* embryos were the most tolerant of fenitrothion, with a 24-h LC50 of >34 000 µg a.i./L. At less than acute toxic concentrations, fenitrothion can induce sublethal effects in fish, such as reduction in feeding activity (1000 µg a.i./L), swimming inhibition (480–750 µg a.i./L), and a decrease in reaction distance to prey (5.5 µg a.i./L).

The bioconcentration factor for fenitrothion is relatively low. The values reported range from 30 in common carp *Cyprinus carpio* muscle to 2300 in whole guppy *Poecilia reticulata*. Fenitrothion, however, is cleared rapidly from fish, with depuration half-lives reported to be between 3 h and 2 d. The half-life of fenitrothion in the guppy was 15 h.

Based on comparisons of fenitrothion concentrations detected in lotic water with concentrations required for acute toxic effects to fish, the hazard associated with direct effects of fenitrothion on fish in streams is low. The hazard due to indirect effects resulting from aerial spraying with fenitrothion on fish inhabiting lotic ecosystems is also expected to be low because of the transitory effects that have been observed on benthic invertebrates inhabiting these systems.

The overspraying of small ponds with fenitrothion can result in fenitrothion concentrations as high as 2500 µg a.i./L in surface water. Such concentrations are higher than those known to cause sublethal or lethal effects in fish food (acute NOEC of 1.0 µg/L for *D. magna*) and fish (acute NOEC of 440 µg/L for rainbow trout). Recent studies have demonstrated mortality of invertebrates and trout in bioassays conducted using water collected from a small pond oversprayed with fenitrothion during an operational application. In Canada, spray buffer zones are not required for small water bodies (e.g., in New Brunswick, nondesignated rivers and lentic bodies <5 ha in size), and thus these habitats are not protected from overspraying. The number of water bodies under 5 ha in Canada is high (e.g., 10 151–16 610 within the New Brunswick forest spray area). Because such ponds provide substantial areas of good quality fish habitat, particularly for brook trout *Salvelinus fontinalis*, there is concern that the unprotected aquatic fauna in small ponds are at risk.

8. Birds

Birds can be exposed to chemical insecticides following aerial applications through four routes: 1) inhalation of respirable particles; 2) dermal contact with the spray cloud and contaminated vegetation; 3) ingestion of contaminated food; and 4) ingestion of residues during preening of contaminated plumage. There is little information on the relative importance of the different routes of exposure, although all four routes probably contribute to total exposure. Compared with similar OP insecticides, fenitrothion appears to be relatively toxic through the dermal route of exposure.

The sensitivity of birds to OP insecticides such as fenitrothion is highly variable. Factors thought to be important in a bird's response to OP insecticide exposure include age, sex, body condition, nutritional status, and ambient temperature at the time of spraying.

Variables such as habitat structure can influence exposure to the spray. Small birds are clearly more sensitive to fenitrothion than large birds, placing songbirds at highest risk from the spray.

The available data show that, following exposure to fenitrothion at current application rates, impacts on songbirds may include alterations in behaviour, decreased reproductive success, and direct mortality. Effects on the behaviour of free-living birds at these application rates can include reduced singing activity, changes in foraging strategies, and an inability to fly and elude capture.

In one laboratory study that examined bird behaviour following fenitrothion administration, Zebra Finches *Poephila guttata* given fenitrothion at a dose of 1.04 mg/kg body weight had significantly diminished behavioural activity levels for 2 d following the treatment.

In the laboratory, the threshold for sublethal effects on reproduction following chronic (18–19 weeks) exposures of fenitrothion in the diet of Northern Bobwhites *Colinus virginianus* and Mallards *Anas platyrhynchos* was between 10 and 30 mg/kg feed. Residues in this range have been detected in food items of birds following operational applications of fenitrothion, but little information is available on the persistence of these residues.

In three field studies in which the reproductive success of naturally nesting songbirds was monitored, data collected following aerial fenitrothion applications at rates ranging from 280 to 1000 g a.i./ha indicated that fledging success was reduced in the treated areas. For instance, a single application of 280 g a.i./ha resulted in a lower proportion of young fledged on the treated plot during the year of treatment and a reduced recruitment of individuals in the year following the treatment. Some birds were found to be in breeding condition late in the season, suggesting that the spray had disrupted earlier nesting attempts.

When fenitrothion was applied at 420 g a.i./ha followed by 210 g a.i./ha 8 d later, most White-throated Sparrow *Zonotrichia albicollis* breeding attempts were disrupted, and reproductive success in the sprayed area was only one-third of that in a nearby control area. Behavioural responses of the adult birds included territory abandonment, inability to defend a territory, disruption of normal incubation activities, and clutch desertion. A further effect was seen on the adult population in the study area, which was reduced by one-third as a result of mortality and territory abandonment after the first spray. In this study, brain cholinesterase (ChE) activities following sprays at the above-normal application rate were inhibited to levels that can sometimes be seen following conventional (210 g a.i./ha) fenitrothion applications. This suggests that reduced reproductive success can occur following sprays at the 210 g a.i./ha application rate.

A single fenitrothion application of 500 or 1000 g a.i./ha for locust *Schistocerca gregaria* control in Senegal affected nesting success of Singing Bush Larks *Mirafra javanica* and Buffalo Weavers *Bubalornis albirostris*. Explanations advanced for the decreased reproductive success were that the insecticide terminated the process of reproduction or caused adults to abandon the nests before the young were fully fledged. Fledgling larks were also found to have severely reduced ChE levels.

In a study in which songbirds were provided with artificial nest boxes, a single application of 300 g a.i./ha had no apparent influence on reproductive success except for a nonsignificant trend towards slower growth resulting in lower final weights of the nestlings.

Decreased reproductive success is likely the result of a reduction in available food resources, direct toxicity to the nestlings, and direct toxicity to the adults, affecting their ability to forage and care for their young.

Mortality of songbirds has been observed at application rates as low as 140 g a.i./ha. Because the toxicity curves for fenitrothion are steep, small increases in the effective application rate (as a result of overswathing, navigational errors, or the vagaries of spray drift) may result in sharply increased mortality. ChE measurements support the conclusion that some songbird mortality is expected following operational fenitrothion sprays. Estimates of the extent of the mortality following standard operational sprays are difficult to make, however, as the relationship between levels of ChE depression and mortality is only approximate. Similarly, mortality estimates cannot be made from carcass searches, as songbird carcasses are very difficult to find and quickly disappear through activities such as scavenging and decomposition. The ChE data indicate, however, that songbird mortality occurs at a greater rate than would be apparent if only carcass searches were relied on to monitor spray effects.

Long-term effects of the spray on songbird populations are difficult to determine. Available census data for maritime songbirds do not allow an evaluation of population trends in the areas where fenitrothion has been used most heavily, and other potential influences on songbird population sizes are difficult to factor out.

The available data indicate that effects on songbirds ranging from behavioural alterations and reduced reproductive success to mortality may occur following operational fenitrothion applications. To obtain a measure of the extent and frequency of potential effects on songbirds, brain ChE activities have been monitored. The ChE monitoring data indicate that exposure of songbirds to the spray can be highly variable and unpredictable. However, the data also indicate that a large proportion of the songbird population may receive a significant exposure to the spray.

For instance, on average, almost half of the White-throated Sparrows collected following fenitrothion applications ranging from 140 to 280 g a.i./ha had a brain ChE activity level that was less than 80% of the value seen in control birds, confirming that they had received a significant exposure to the insecticide. Almost 17% of the birds collected had a brain ChE activity level below 50% of normal, which indicates that they had been exposed to a potentially life-threatening dose. These figures likely underestimate the true magnitude of the exposure of the songbird population, as it is the less affected birds that tend to be collected during postspray collection programs. Further, White-throated Sparrows are likely to be less affected by exposure to fenitrothion than smaller birds such as warblers, which, because of their small size, may be more sensitive to the toxic effects of the insecticide.

The ChE data show that White-throated Sparrows (and other forest songbirds) may be at risk from the current fenitrothion spray program because 1) ChE measurements indicate a proportion of the White-throated Sparrow population receives a heavy exposure to the pesticide during operational applications, which inhibits their ChE activity by more than 50%; 2) some of the birds with this level of ChE activity will likely die from the exposure; and 3) a larger proportion of less exposed birds may suffer from effects associated with sublethal levels of ChE inhibition, including abnormal behaviour, difficulties with flying, anorexia, increased vulnerability to predation, and compromised physiological ability to deal with natural stressors.

Other birds may also be at risk from fenitrothion applications. Waterfowl raising broods in the treated areas may suffer from the removal of their invertebrate food resource. This risk would be greatest where birds are raising broods in small ponds, as these are not buffered by setbacks during the operational spray program. American Black Ducks *Anas rubripes* and Ring-necked Ducks *Aythya collaris* are two species that might be vulnerable to insecticide-induced food removal during their brood-rearing period, as they nest in these types of habitats. Although American Black Duck broods are able to move among wetlands to avoid localized invertebrate population depressions, which might mitigate fenitrothion impacts, large-scale spray operations such as those practised in New Brunswick might result in reduced invertebrate biomass in all potential brood-rearing habitat within a brood's home range.

In summary, the available data support the following general observations concerning forest birds. ChE data collected following operational applications indicate that songbird impacts will occur following fenitrothion treatments. A proportion of the bird population will receive a significant exposure to the insecticide; some of the exposed birds may die, and others will suffer from sublethal effects. These conclusions are supported by sporadic findings of dead or incapacitated songbirds following operational sprays. Current understanding of the biological relationships between ChE inhibition and sublethal impacts does not allow prediction of the outcome of sublethal ChE depression, but present application rates appear to be able to adversely affect reproductive success. An estimate of the total mortality resulting from any application cannot be made, and possible influences of the insecticide on the long-term status of bird populations in the spray areas cannot be assessed. ChE monitoring data reveal that high exposures may occur frequently. With present application technology, there is no known means to diminish or prevent these exposures. Because of the range of effects seen following fenitrothion applications, and because the ChE data indicate that these effects may occur frequently, concerns are raised for forest songbird populations in fenitrothion treatment areas.

Nesting waterfowl may breed in small, unbuffered ponds that are sensitive to fenitrothion and that also tend to receive direct applications of the compound. The data suggest that success of breeding waterfowl may be hampered by an insecticide-induced depletion of their prey resource.

Impact assessment

The weight of evidence accumulated with respect to the identified and potential negative impacts caused by the forestry use of fenitrothion on nontarget fauna, including terrestrial arthropods, pollinators, aquatic invertebrates, amphibians, fish, and songbirds, and their potential ecological implications, supports the conclusion that the large-scale spraying of fenitrothion for forest pest control, as currently practised operationally, is environmentally unacceptable.

FENITROTHION RISK ASSESSMENT

Environmental chemistry and fate

1. Physicochemical properties

The vapour pressure of fenitrothion is reported by the proponent to be 2.14×10^{-4} mmHg at 25°C (Sumitomo Chemical America, Inc. 1991), which indicates that fenitrothion would have an intermediate to high volatility under field conditions, according to Kennedy and Talbert (1977). No raw data or experimental methodologies were included to support this value. The volatility of fenitrothion from a sandy loam soil (0.8% O.M., pH 6.5) at 25°C maintained at 50% and 75% field moisture content indicates that fenitrothion has the potential to volatilize from soil and that volatilization is higher in moist soils (Sumitomo Chemical America, Inc. 1991). Fenitrothion sprayed on the surface of natural water in the laboratory disappeared rapidly, with a half-life of 0.5 h. At least 70% of the disappearance was due to volatilization. It was concluded that volatilization from the surface microlayer was the major dissipation process for fenitrothion sprayed on natural waters (Maguire 1991).

Water solubility values for fenitrothion are reported to be 5 mg/L at 10°C, 14 mg/L at 30°C, and 31 mg/L at 50°C (Sumitomo Chemical America, Inc. 1991), which would indicate that fenitrothion is soluble in water. No raw data or experimental methodologies were included to support these values.

The Henry's law constant calculated using the vapour pressure and water solubility values reported by Sumitomo Chemical America, Inc. (1991) (5.57×10^{-6} atm·m^3/mol) also indicates that fenitrothion has the potential to volatilize from water and moist soil.

The octanol/water partition coefficient (log K_{ow}) for fenitrothion is reported to be 3.43 ± 0.03 at 20°C (Sumitomo Chemical America, Inc. 1991), which indicates that the compound has a potential for bioaccumulation in the environment. No raw data or experimental methodologies were provided to support this value.

2. Transformation

The half-lives of fenitrothion via hydrolysis are reported to be 191–200 d at pH 5, 180–186 d at pH 7, and 100–101 d at pH 9 (Sumitomo Chemical America, Inc. 1991). These results indicate that fenitrothion is hydrolysed under basic conditions and is more resistant to hydrolysis under neutral and acidic conditions. The rate of hydrolysis also increases with increasing temperature. The major transformation product observed at pH 5 and 7 was O-methyl-O-hydrogen-O-(3-methyl-4-nitrophenyl) phosphorothioate. At pH 9, 3-methyl-4-nitrophenol was the major transformation product (Sumitomo Chemical America, Inc. 1991). The hydrolysis of fenitrothion and the formation of 3-methyl-4-nitrophenol proceed very slowly under the environmental conditions likely to be found in most eastern Canadian watercourses (Fairchild *et al.* 1989).

Phototransformation studies conducted using a sandy loam Japanese soil (1.6% O.M., pH 6.2) showed that fenitrothion had half-lives of 85 and 182 d under light and dark conditions, respectively, suggesting that transformation by photolysis is slow under the conditions of these studies. Phototransformation studies conducted in water at 25°C demonstrated that fenitrothion had half-lives of 3.65 and 140.9 d under light and dark conditions, respectively, indicating that the compound can undergo phototransformation quite readily in water (Sumitomo Chemical America, Inc. 1991). Maguire and Hale (1980) concluded that the main pathway of degradation for fenitrothion in surface and subsurface water was photolysis to p-nitro-m-cresol. This transformation product disappears through photolytically mediated polymerization to form substances resembling fulvic or humic acids, which may then be bound to organic matter or remain in solution (Miyamoto 1977). Phototransformation studies conducted in air indicated that transformation under the conditions of these studies is slow. The major atmospheric transformation product observed was 3-methyl-4-nitrophenol (Sumitomo Chemical America, Inc. 1991).

Aerobic soil transformation studies conducted using a sandy loam soil from Nebraska (1.6% O.M., pH 6.2) showed that radiolabelled fenitrothion had a calculated DT50 (dissipation half-life) of 36 d in a year-long study. The major transformation product was 3-methyl-4-nitrophenol, which increased to 20% of the applied radioactivity after 3 d, then decreased to <1% of the applied radioactivity by 30 d. Bound residues also comprised a significant portion of the radioactivity (19.7% of applied) after 365 d. In another study using an organic soil (44.4% O.M., pH 3.5) and a sandy loam soil (5.4% O.M., pH 5.2) from Maine, the DT90s were 29 and 15 d, respectively. The major transformation product observed was 3-methyl-4-nitrophenol, which had DT50s of 6 and 13 d in the organic and sandy loam soils, respectively. Further transformation of 3-methyl-4-nitrophenol resulted in the formation of CO_2 and soil-bound material (Sumitomo Chemical America, Inc. 1991). The results of these studies suggest that fenitrothion and its major transformation product 3-methyl-4-nitrophenol are microbially transformed and are not persistent in forest soils.

A study using water and sediments collected from a river and pond in Japan found the half-lives of fenitrothion in the river and pond water to be 10 and 16 d, respectively (Sumitomo Chemical America, Inc. 1991). The results indicated that fenitrothion and its major transformation product 3-methyl-4-nitrophenol are not persistent under aerobic aquatic conditions and that microorganisms in natural sediments and water play an important role in the mineralization of fenitrothion.

A study using a flooded sandy loam soil from Nebraska (1.6% O.M., pH 6.2) and radiolabelled fenitrothion found that fenitrothion was not persistent when applied under anaerobic aquatic conditions. Major transformation products included aminofenitrothion and 3-methyl-4-nitrophenol. Bound residues accounted for 73.9% of the applied amount after 365 d (Sumitomo Chemical America, Inc. 1991). It should be noted that this study used a flooded soil and not a sediment, as is required to satisfy Canadian data requirements. Zitko and Cunningham (1974) observed that fenitrothion partitioned into sediments and suggested that the formation of aminofenitrothion is expected under anaerobic conditions. These studies indicate that fenitrothion and its major transformation products, aminofenitrothion and 3-methyl-4-nitrophenol, are associated with sediments and are not persistent under anaerobic aquatic conditions.

3. Mobility

Laboratory studies examining the adsorption/desorption of fenitrothion indicated that fenitrothion has a high mobility on silty clay loam, silty clay, and sand soils and a medium mobility on sandy loam soil. Fenitrothion is strongly adsorbed to sediment. The results also indicated that adsorption is positively correlated with organic matter content and that fenitrothion will desorb from all soils and sediment (Sumitomo Chemical America, Inc. 1991).

Soil column leaching studies conducted using three Japanese soils (a clay loam and two sandy loams) indicated that fenitrothion has the potential to be mobile in coarse-textured soils with a low organic matter content. Soil column leaching studies conducted using a sandy loam soil (1.6% O.M., pH 6.2) from Nebraska indicated that fenitrothion was moderately mobile and that its major transformation product 3-methyl-4-nitrophenol was more mobile than the parent compound. The increased level of nonextractable residues with time suggests decreased mobility of fenitrothion with extended soil contact (Sumitomo Chemical America, Inc. 1991).

The results of soil TLC analysis indicate that fenitrothion is mobile, that its transformation product 3-methyl-4-nitrophenol has an intermediate mobility, and that the transformation products fenitrooxon and desmethylfenitrothion have a low to intermediate mobility, according to Helling and Turner's (1968) classification.

4. Field dissipation

Sundaram (1974) found that fenitrothion residues in soil samples were low (ranging from <0.005 to 0.1 μg/g) and that these residues disappeared within 45 d following an operational spray. Yule and Duffy (1972) found that a fenitrothion application of 275 g a.i./ha produced a deposit of up to 0.04 μg/g in forest soil and that these residues were undetectable after 32–64 d. Sundaram (1984) observed that fenitrothion concentrations were 0.018–0.063 μg/g in litter and 0.013–0.034 μg/g in soil in a plantation that was sprayed experimentally for 5 consecutive years at 280 g a.i./ha. Following an application at a similar rate, Morin *et al.* (1986) observed that DT50 values for fenitrothion in sandy soils, clay loam soils, and leaf litter were 6.34, 7.98, and 16.4 d, respectively. These and other Canadian field dissipation studies indicate that fenitrothion is present only in low concentrations in forest soil and is not persistent following operational aerial applications.

Residues of fenitrothion were not detected below a depth of 15 cm following application of a Sumithion reference standard (96.6% purity) to soils from Illinois and Florida at a rate of 2800 g a.i./ha (Sumitomo Chemical America, Inc. 1991). No characteristics of these soils were given. Canadian field studies have indicated that fenitrothion and its major transformation products do not leach appreciably in forest soils when applied according to label instructions. The low deposits and short persistence of fenitrothion and its major transformation products in forest soils following applications for spruce budworm *Choristoneura fumiferana* control are major reasons for the low observed vertical mobility.

Concentrations of fenitrothion in lotic (flowing) aquatic systems following operational spraying range from 1.3 μg/L (Mallet and Cassista 1984) to 127 μg/L (Morin *et al.* 1986) (see Table 4.2, p. 121 in Fairchild *et al.* 1989). Many of the higher maximum concentrations recorded were in surface film samples (Morin *et al.* 1983, 1986) or in samples from streams that were not protected by buffer zones and that had sparse forest canopy cover (Coady 1978). Mean maximum concentrations for spray applications ranged from 0.59 μg/L (Sundaram *et al.* 1984) to 18.4 μg/L (Flannagan 1973). Fenitrothion concentrations usually declined to less than 1.0 μg/L within 24–48 h and to less than 0.5 μg/L within 1–6 d (Gaboury *et al.* 1981).

Maximum recorded concentrations of fenitrothion in lentic (standing water) aquatic systems after operational spraying usually occur within 2 h after the start of spraying and range from 0.38 μg/L (Major and Dostie 1983) to 2500 μg/L (see Table 4.3, p. 122 in Fairchild *et al.* 1989). Mean surface water concentrations of fenitrothion in small ponds (<0.5 ha) directly oversprayed at an application rate of 210 g a.i./ha ranged from 20 to 1500 μg/L. Mean outflow concentrations ranged from 0.2 to 26.0 μg/L (Ernst *et al.* 1991).

The mean and median half-lives of fenitrothion in lentic aquatic systems were 18.9 and <8.5 h, respectively (see Table 4.5, p. 123 in Fairchild *et al.* 1989). Malis and Muir (1984) reported that fenitrothion had half-lives of 1.0 and 1.6 d under unshaded and shaded conditions, respectively, in small ponds (pH 7.5–8.8) treated with fenitrothion to produce an initial concentration in water of 70 μg/L. This study indicated that photolytic transformation contributed significantly to the disappearance of fenitrothion from these ponds. Fairchild (1990) observed similar half-lives in bog ponds (pH 4) after treatments producing fenitrothion concentrations in water of 40–80 μg/L. These results suggest that pH had no effect on the persistence of fenitrothion in the ponds in these two studies.

Morin *et al.* (1986) found that the high initial fenitrothion concentration of 543 μg/L observed in a pond was reduced by 95% within 3.3 h after spraying. Maguire and Hale (1980) observed that fenitrothion sprayed on water volatilized very quickly from surface slicks (half-life 18 min at 20°C). Thus, a large fraction (70%) of fenitrothion applied to lakes and ponds is volatilized from the surface film (Maguire and Hale 1980; Maguire 1991), and the portion remaining in the water is transformed through photolysis (Maguire and Hale 1980; Malis and Muir 1984). These two processes generally operate independently of pH. Fairchild (1990) observed that fenitrothion concentrations at depths of 0.3 and 1 m in acid bog ponds were 0.25–0.50 times the concentrations observed in the surface layer at 15 min postspray. However, concentrations observed at greater depths (half-life about 1 d) were not as transitory as the high residues observed in surface water (half-life <0.5 h). Fairchild (1990) observed that aqueous fenitrothion residues reappeared 1 year after application to an acid bog, suggesting that some component of that ecosystem, possibly moss, acted as a reservoir. This is a topic that requires further study.

Concentrations of fenitrothion in aquatic sediments after operational spraying were less than 0.5 μg/g (see Table 4.4, p. 123 in Fairchild *et al.* 1989). Maguire and Hale (1980) observed that fenitrothion concentrations in sediment from a small pond in a spruce-fir forest in New Brunswick fell below detectable levels 2 d postspray, and aminofenitrothion, the major transformation product, persisted for less than 4 d.

Yule and Duffy (1972) found that an aerial application of fenitrothion at 275 g a.i./ha resulted in a deposit of 2–4 μg/g on coniferous foliage (red spruce *Picea rubens*, white spruce *P. glauca*, and balsam fir *Abies balsamea*), which decreased by about 50% within 4 d and by 70–85% within 2 weeks; 10% of the initial deposit persisted for most of the year. Sundaram (1974) found that fenitrothion residues persisted in detectable amounts (0.005 μg/g) on conifer foliage for up to 155 d. Mallet and Cassista (1984) observed that the maximum residue detected in coniferous foliage was 0.53 μg/g.

Yule (1974) concluded that residues of fenitrothion persisted and accumulated in coniferous tree foliage over a number of years with repeated annual applications; the maximum accumulated residue observed was 1 μg/g in fresh balsam fir foliage. Eidt and Mallet (1986) observed that there was some tendency for fenitrothion to accumulate in balsam fir foliage with successive years of spray, but only until a peak near 1.5 μg/g was reached. McNeil *et al.* (1979) observed residues in jack pine *Pinus banksiana* foliage of 0.27, 0.41, and 0.54 μg/g in samples collected 1 month after the final spray in three different sites with 1, 2, and 3 years of successive treatment history, respectively. A pine plantation was sprayed experimentally for 5 consecutive years at 280 g a.i./ha; 11 months after the last spray, fenitrothion concentrations in bark ranged from 0.089 μg/g in eastern white pine *Pinus strobus* to 0.291 μg/g in red pine *P. resinosa*, and concentrations in foliage ranged from 0.068 μg/g in eastern white pine to 0.179 μg/g in red pine (Sundaram 1984).

Yule and Varty (1975) observed that hardwood species (e.g., red maple *Acer rubrum*) appeared to collect 3–4 times the concentration of fenitrothion that was observed on conifers; however, residues decreased to less than 1.0 μg/g within 10 d, and no fenitrothion was transferred to the soil stratum in the autumn leaf fall. These and other studies indicate that fenitrothion can persist and possibly accumulate for periods of more than 1 year at levels of approximately 1 μg/g in conifer foliage.

Environmental toxicology

1. Soil microorganisms/microbial processes and soil and litter arthropods

Laboratory experiments indicated that fenitrothion concentrations of 10 and 14 μg/g in forest soils had no significant effect on populations of actinomycetes, yeast, fungi, and bacteria. Soil respiration was also unaffected at these concentrations (Sumitomo Chemical America, Inc. 1991). Fenitrothion concentrations of 1 and 5 μg/g in a forest soil did not alter nitrifying or denitrifying processes and had no effect on leaf litter or cellulose decomposition or soil respiration (Spillner *et al.* 1979a; Sumitomo Chemical America, Inc. 1991). Growth of the mycorrhizal fungus *Suillus luteus* was completely inhibited at 15 μg/g; the no-observed-effect concentration (NOEC) was found to be 5 μg/g (Sumitomo Chemical America, Inc. 1991). Spillner *et al.* (1979b) observed that fenitrothion and its transformation products were not detrimental to the microbial community indigenous to the forest soil environment after exposure to fenitrothion at 7.4 μg/g in moist soils. Thus, the concentrations of fenitrothion found in forest soil immediately following operational applications to control spruce budworm (0.005–0.1 μg/g) are not expected to adversely affect soil microorganisms or microbial processes.

A fenitrothion application of 209 g a.i./ha in one study and two applications of 140 g a.i./ha in another study reduced population levels of soil and leaf litter arthropods, including members of the Sarcoptiformes, Trombidiformes, Mesostigmata, and Collembola. The mesostigmatid mites were the group most affected, and some of the populations of other groups had not fully recovered to pretreatment levels after 61 d (the last sampling date). In no case, however, were any of the arthropod populations eliminated (Sumitomo Chemical America, Inc. 1991). Carter and Brown (1973) investigated certain arthropod predator elements of the soil fauna of a mature red spruce stand following two aerial applications of fenitrothion at 140 and 210 g a.i./ha. They observed that one species of centipede, one species of pseudoscorpion, and three species of harvestmen were found in fewer numbers during the year of spraying, but that numbers recovered to pretreatment levels the following year. Spiders of the family Erigonidae showed no marked difference in numbers during the course of the study. Most other species of spiders were encountered in reduced numbers during the spray year but recovered to pretreatment levels the next year. The effects of repeated annual fenitrothion applications on the soil invertebrate community and the effects of any changes in the invertebrate community on soil processes remain unknown.

2. Nontarget terrestrial arthropods

Fenitrothion is a broad-spectrum insecticide, and thus operational applications usually result in a large, immediate kill of arthropods from a variety of taxa, but especially Lepidoptera larvae, sawfly larvae (Diptera: Diprionidae/Tenthredinidae), aphids (Homoptera: Aphididae), and perching flies (Diptera) (Buckner 1975a; Varty 1975, 1980; Fraser 1983). Arthropod biomass may be reduced by up to 35% and numbers by up to 50% (Millikin 1990). However, despite this large kill, insect populations may be affected only temporarily.

This is because a proportion of the population of any arthropod species is likely to avoid contact with the insecticide owing to the existence of refugia in unsprayed foliage within the spray block and within individual trees (Varty 1977, 1980; Millikin 1990).

Hymenopterous insects are particularly sensitive to fenitrothion. It has been reported that Swaine jack-pine sawfly *Neodiprion swainei* and balsam fir sawfly *N. abietis* populations may be suppressed by persistent fenitrothion residues in conifer needles in frequently sprayed forests (McNeil *et al.* 1979; Eidt and Mallet 1986).

The effect of fenitrothion treatment on parasitism of spruce budworm has received considerable attention (e.g., Varty 1971, 1975, 1977; Buckner 1975a; Magasi *et al.* 1980). Conventional fenitrothion spraying to control spruce budworm larvae may virtually eliminate late larval and pupal parasites by killing adult parasites before they have found a host suitable for oviposition (Varty 1971; McNeil and McLeod 1977). Effects on early larval and egg parasitism are minimal because susceptible stages are not present at the time of spraying. Adulticide sprays may be far more hazardous because many parasites, including those attacking eggs and early larvae, will also be in the susceptible adult stage (Varty 1973, 1975). Adulticide spraying is not currently practised in Canada.

Spruce budworm predators in balsam fir crowns include resident hemipterous bugs (mirids, anthocorids, assassin bugs, stink bugs), beetles (ladybeetles), cecidomyid midges, syrphid flies, lacewings, mites, spiders, and harvestmen (Varty 1980). Fenitrothion spraying results in extensive mortality of predaceous insects, mites, and harvestmen (up to 50% of the population exposed). This mortality is patchy in both space (because of variable deposit) and time (because areas may not be sprayed in consecutive years), however, and recolonization from untreated areas and areas of low deposit is generally rapid (Varty 1977). As a consequence, long-term population densities of most arboreal predaceous insects and spiders are relatively stable, even when areas are sprayed over a number of consecutive years (Varty 1975, 1977). Populations of a few species (e.g., the ladybeetle *Mulsantina hudsonica*), however, have been severely reduced in sprayed areas of New Brunswick (Varty 1977, 1980). It should be noted that this information was obtained from surveys, and a well-designed, well-controlled, scientific study examining the effect of fenitrothion spraying on predators of the spruce budworm has not been conducted.

The fact that spruce budworm epidemics persist in perennially sprayed areas despite a very high kill of larvae has led some to suggest that the budworm outbreak in eastern Canada is being perpetuated by an insecticide-induced breakdown of natural biocontrol mechanisms. The evidence on parasite and predator abundance does not offer any substantial support to this hypothesis (Varty 1975, 1977). Likewise, there is no evidence to suggest that any irruptions of minor pests have been triggered by local disruption of biocontrol mechanisms as a result of spraying (Varty 1977; Magasi *et al.* 1980). Once again these conclusions have been derived from survey results and not a scientific study. These surveys have not been sufficient to characterize the parasitoid response at large but have been restricted to a few important species parasitizing spruce budworm larvae.

Nevertheless, it is apparent that continued fenitrothion spraying may be weakening a section of the biocontrol complex operating upon budworm larvae and pupae. Although these biocontrol mechanisms do not contain full-blown budworm epidemics, they do contribute to budworm mortality, slowing population increases and hastening the collapse of declining populations.

3. Pollinators

Laboratory studies have indicated that fenitrothion is highly toxic to honeybees. Okada and Hoshiba (1970) reported an LD50 of 0.03 µg/bee and an LD90 of 0.1 µg/bee. Johansen and Atkins (1978) observed an LD50 of 0.176 µg/bee and an LD90 of 0.294 µg/bee. Stevenson (1978) reported a contact LD50 of 0.018 µg/bee and an oral LD50 of 0.019 µg/bee. Fenitrothion also has a high toxicity to other native pollinators. The LD50s for *Andrena erythronii*, *Megachile rotundata*, and *Bombus terricola* have been reported as 0.10, 0.40, and 0.76 µg/bee, respectively (Kevan and Plowright 1989).

Using the methods of Atkins (NRCC 1981) to calculate roughly the dosage of fenitrothion that honeybees would receive at usual rates of application (140–280 g a.i./ha) and to estimate the mortality that those rates of application would cause, it appears that a high mortality of honeybees would be expected to occur under field conditions. It must be recognized, however, that these methods were developed for agricultural use; in a forest environment, the tree canopy would intercept much of the pesticide, thereby reducing the exposure to pollinators. In an area with no protective canopy that was sprayed with fenitrothion (i.e., assuming that 100% of emitted spray products are deposited at ground level), honeybee mortality would be predicted to be 55–60% at an application rate of 210 g a.i./ha. However, if 50% of the pesticide was intercepted by the forest canopy (i.e., assuming 50% of the emitted spray is deposited at ground level), mortality would be reduced to 15–20%.

Numerous field studies have been conducted addressing the impact of operational applications of fenitrothion on honeybees and native wild pollinators. Buckner (1974, 1975b), McLeod and Mortensen (1975), and Buckner and McLeod (1977) observed mortality in foraging bees and a decline in the foraging activity of honeybees in fenitrothion-treated plots compared with control plots. Much higher numbers of dead bees were collected at the hive in treated areas compared with control areas, which may indicate a serious loss to the foraging component of the hive. Plowright (1977) and Plowright *et al.* (1978) observed that caged bumblebees in fenitrothion spray blocks suffered significantly greater mortality (58–100%) than those in unsprayed locations. When cages were placed under dense coniferous canopy, however, mortality was not significantly different between treatment and control. High mortalities were also observed among exposed solitary bees and vespid wasps (Plowright and Rodd 1980).

In experiments using laboratory-reared colonies of bumblebees set out in the forest prior to spraying, survival of bees was significantly lower in treated forests than in control forests. Bacilek (1982) conducted a rigorous experiment in which honeybees were trained to use a specific flight path over which fenitrothion was sprayed. A 41% kill of flying bees exposed to the fenitrothion spray was observed.

Kevan and Lear (1973), Kevan (1974a, 1975a, 1975b, 1976, 1977a, 1977b), and Kevan and LaBerge (1979) observed that populations of blueberry pollinators were significantly lower in areas where fenitrothion was used than in areas not sprayed. Wood (1974, 1977, 1979, 1980a, 1980b, 1982a, 1982b) observed population reductions (up to 90%) of pollinators on blueberry fields in the vicinity of forests treated with fenitrothion. Plowright (1977, 1980), Plowright *et al.* (1978), Plowright and Thaler (1979), and Plowright and Rodd (1980) conducted surveys of forest pollinators in fenitrothion spray blocks and untreated forests and found that populations of many species were significantly depressed by spraying. Species of bumblebees that emerged early in the spring (e.g., *Bombus ternarius* and *B. terricola*) showed the greatest population reductions, whereas those species that emerged late (i.e., that were still hibernating at the time of spraying) were actually more common in treated than in control areas. It was suggested that this may have been due to a reduction in competitive stress, resulting in "ecological release" of late-emerging species (Plowright and Rodd 1980). The ecological replacement of early-emerging by late-emerging species in fenitrothion spray blocks was further investigated by Thorpe (1979), who observed that early-emerging species of solitary bees (Andrenidae, Halictidae) were at peak activity levels for both foraging and reproduction when fenitrothion spray residues were high. The Colletidae, Anthophoridae, and Megachilidae species were late emergers and were thought to be less vulnerable.

The time required for the recovery of pollinator populations from fenitrothion exposure is a topic on which there is disagreement in the literature. Wood (1979) concluded that recovery took 1–3 years in blueberry fields that were sprayed once. Kevan (1977c) concluded that recovery of pollinator populations required from 3 or 4 years to over 10 years, depending on the severity of the population reduction. Kevan and LaBerge (1979) documented 5 years of continued recovery of pollinator populations in blueberry fields sprayed over a number of years. The recovery time may be influenced by a number of factors. In areas where fenitrothion spraying is repeated yearly, one does not expect the rapid recovery that characterizes isolated plots that receive only a single application. Further, those species that are fairly mobile (e.g., bumblebees) are likely to show more rapid recolonization of sprayed areas than those that form nesting aggregations (e.g., solitary bees). The size of the spray blocks may also influence recolonization by pollinators; recolonization would be slower in the centre of large spray blocks.

Fenitrothion spraying has also been observed to result in shifts in pollinator behaviour. Plowright *et al.* (1978) observed that bumblebees foraging in habitats that had been sprayed with fenitrothion showed much narrower food-niches (i.e., foraged on fewer plant species) than bees in unsprayed locations. Therefore, plant species that hold a subordinate position in the hierarchy of bee preference may be most affected by reduction in pollinator density.

Population reductions of honeybees and wild bees have been correlated with reduced seed and fruit set (plant fecundity) in various plants in natural and agricultural ecosystems. Beginning in 1970, several blueberry growers in New Brunswick reported unusually low wild pollinator activity in fields located near fenitrothion spray blocks, accompanied by lower than normal harvests (Kevan 1974b; Kevan and Collins 1974). Crop losses of blueberries in New Brunswick between 1970 and 1977 due to fenitrothion spraying in surrounding forests were estimated at 670 000 kg (Kevan and Plowright 1989). Subsequent research documented the extent of bee mortality (summarized in Kevan and Plowright 1989), and no-spray buffer zones of 3.2 km were implemented around blueberry fields (Sexsmith 1987). The effectiveness of no-spray buffer zones has not been documented, but there have been few reports of blueberry crop failures in recent years.

Forest plant species for which an association between reduced fecundity and fenitrothion spraying has been demonstrated include wild sarsaparilla *Aralia nudicaulis*, with a 30% reduction in fruit production; corn lily *Clintonia borealis*, 17–36% reduction; bunchberry *Cornus canadensis*, 44% reduction; sheep-laurel *Kalmia angustifolia*, 52% reduction; wild lily-of-the-valley *Maianthemum canadense*, 27% reduction; and red clover *Trifolium pratense*, 78% reduction (Plowright and Thaler 1979; Plowright and Rodd 1980; Thaler and Plowright 1980; SOMER 1985). In order for a plant to be considered susceptible to fenitrothion spraying, the plant must 1) require insect visitation for fruit production; 2) be pollinated by insects that are vulnerable to the insecticide; and 3) bloom during the period when pollinator populations are most reduced by the spray. All of the above plant species satisfy these three requirements (Kevan and Plowright 1989).

The effects of the observed reductions in plant fecundity, specifically fruit set and seed set, on the population biology of affected plants are unknown, although Kevan and Plowright (1989) speculated on some possibilities. Theoretically, reduced fruit and seed production should be reflected in lower recruitment and/or restricted dispersal of seedlings. However, many forest plant species reproduce primarily by vegetative means, and recruitment of seedlings is normally quite low. Another possible effect of reduced sexual reproduction might be reduced genetic variation in those species that can produce self-fertilized seeds. None of these possible scenarios has been studied.

The impact of fenitrothion on forest pollinators and pollination is a definite concern. Fenitrothion-mediated reductions in pollinator abundance undoubtedly add some element of risk to the sexual reproduction of boreal forest herbs. However, population-level effects such as changes in age structure or areal-sex ratio (the ratio of male to female plants in an area) or inbreeding in plants have not been demonstrated, primarily because the appropriate studies have not been conducted. Impacts on plant populations are likely to be difficult to measure, because almost all of these plants are long-lived perennials that, besides sexual reproduction, have well-developed clonal growth (Barrett and Helenurm 1987). In the absence of studies addressing sublethal effects on pollinators and the consequences of reduced production of seed and fruit for forest wildlife dependent on those food resources, a complete understanding of the impact of spraying with fenitrothion on the forest ecosystem due to effects on forest pollinators is not possible at this time.

4. Algae

The 96-h EC50 for *Selenastrum capricornutum* under laboratory conditions is 1300 μg/L. The NOEC is 610 μg/L after 96 h (Sumitomo Chemical America, Inc. 1991). Because the EC50 value is near the upper limit of the range of mean surface water concentrations for fenitrothion (20–1500 μg/L) observed by Fairchild *et al.* (1989) in small ponds operationally sprayed with fenitrothion, and because these concentrations have been observed to disappear rapidly under field conditions, effects on algae and other primary producers are not expected to be significant. Caunter and Weinberger (1988) reported a doubling of the half-life of fenitrothion in aquatic media without algae compared with media with algae present. Thus, differences in phytoplankton productivity between ponds may affect the exposure of invertebrates to fenitrothion in water.

5. Aquatic invertebrates

5.1 Laboratory toxicity studies

Aquatic invertebrates have shown a wide range of sensitivities to fenitrothion, with 24-h LC50s ranging from 0.4 μg/L for *Culex tarsalis* (Diptera) to 40 000 μg/L for *Eriocera spinosa* (Diptera) (Wildish and Phillips 1972). Summaries of the toxicities of fenitrothion to a wide range of invertebrate taxa are available in a number of publications (Wildish and Phillips 1972; NRCC 1975; Symons 1977; Johnson and Findlay 1980; Woodward and Mauck 1980; Gaboury *et al.* 1981; Sanders *et al.* 1983; Poirier and Surgeoner 1986). For instance, Wildish and Phillips (1972) listed static 96-h LC50 values for Trichoptera, Odonata, and Neuroptera of 28, 3, and 15 μg/L, respectively.

The 48-h static LC50 of fenitrothion for *Daphnia magna* was reported as 8.6 μg/L, with a NOEC of <2.0 μg/L. The 48-h flow-through LC50 of Sumithion 8E (fenitrothion 76.8% EC, or emulsifiable concentrate) for *D. magna* was reported to be 2.3 μg/L, with a NOEC of 1.0 μg/L. The 21-d flow-through EC50 of technical fenitrothion for *D. magna* was reported as 0.19 μg/L (Sumitomo Chemical America, Inc. 1991).

These reported acute and chronic toxicities indicate that Sumithion and its active ingredient fenitrothion are highly toxic to aquatic invertebrates. Further, some of these acute and chronic toxicity values are within the range of concentrations that have been measured in lentic aquatic systems following operational applications. Because of its toxicity to non-target arthropods, the World Health Organization (WHO 1992) recently recommended that application rates of fenitrothion be limited and that the compound should never be sprayed over water bodies or streams.

5.2 *Effects in lotic systems*

Drift is a natural phenomenon and refers to the downstream transport by the current of benthic organisms inhabiting lotic watercourses. Active entry into the water column is a behavioural response of some aquatic invertebrates to stress stimuli, such as predation, competition, contact, or physical or chemical irritants (Wiley and Kohler 1984; Scherer and McNicol 1986). Passive entry into the drift is the result of accidental or unintentional release from the substrate. Dosage rates in field studies examining the effects of fenitrothion on invertebrate drift range from 140 to 420 g a.i./ha. Summaries of these studies are contained in Tables 4.7, 4.8, and 4.9 in Fairchild *et al.* (1989). The majority of these studies indicated an effect of the treatments on invertebrate drift, which in some cases included dead organisms. Orders of insects most frequently affected included Plecoptera, Ephemeroptera, Hemiptera, and Diptera. Most peaks in drift occurred within the first 12 h after treatment, and the effects generally lasted less than 24 h (Eidt 1977; Morrison and Wells 1981). The only drift-monitoring study conducted since 1980 (Holmes 1982) showed very limited effects in one stream (increased duration of drift, not magnitude) and none in another.

Summaries of studies examining the effects of fenitrothion on benthic invertebrates in lotic ecosystems after aerial spraying are provided in Tables 4.10 and 4.11 in Fairchild *et al.* (1989). The majority of these studies found no decreases in benthic populations after spraying; however, some of the studies observed short-term reductions in some groups of aquatic invertebrates following treatment. In the majority of cases in which reductions were observed, the effects were transitory. This is probably due to the resilient nature of this ecosystem (i.e., recolonization as a result of drift from untreated areas upstream). Small streams in New Brunswick are not currently required to be buffered when fenitrothion or any other insecticide is applied with TBM and larger aircraft (TBMs are the main aircraft applying fenitrothion during current operational programs).

5.3 *Effects in lentic systems*

Only five studies examining the effects of experimental fenitrothion applications on aquatic invertebrates in lentic ecosystems have been reported; these are outlined in Tables 4.12 and 4.13 in Fairchild *et al.* (1989). Three of these studies examined direct applications of fenitrothion to lakes during forestry operations (Kingsbury 1977, 1978a, 1978b). The limited effects observed (e.g., decrease in shallow-dwelling Ephemeroptera and temporary reduction in surface populations of zooplankton) indicate that operational sprays should have only minor effects on the invertebrates associated with large lentic water bodies. In operational forest spray programs, lakes are protected by no-spray buffer zones that limit the amount of insecticide contacting the water. In addition, lakes have a substantial capacity for dilution. Effects may be noted on invertebrates inhabiting the littoral zone; however, these effects are transitory. In the fourth study, an agricultural application to the shallow water of a rice paddy at double the application rates used in forestry caused 100% mortality of benthic invertebrates and frogs (Thirumurthi *et al.* 1973).

In the fifth study, Fairchild (1990) conducted an experimental ground application of fenitrothion at a rate of 210 g a.i./ha to small bog ponds. The application resulted in water column concentrations comparable with those measured in similar water bodies after operational sprays. Emergence of a number of orders of invertebrates was reduced and did not recover until 12 weeks after treatment. Benthic biomass (primarily Ceratopogonidae and Chironomidae) was reduced by about 50%, and recovery to control densities required more than 1 year. During this period, the benthic community structure shifted to become dominated by Nematoda and Annelida, which resulted in an increase in the amount of energy and nutrient cycling within the ponds and a reduction in the contribution of these components to the surrounding terrestrial ecosystem. The implications to wildlife inhabiting these areas are unknown at present.

In one other study, Fairchild *et al.* (1987) observed that mosquito and midge larvae inhabiting pitcher plant *Sarracenia purpurea* leaves are affected by concentrations of fenitrothion to which they may be exposed following operational applications.

Small ponds other than bog ponds are difficult to see from the air and hence difficult to protect with buffer zones; in addition, they have little forest canopy, a small capacity for dilution, and low water turnover rates. The invertebrates inhabiting these ponds could therefore be potentially at risk from forest spraying with fenitrothion. Mean surface water concentrations of fenitrothion in small ponds (<0.5 ha) directly oversprayed with fenitrothion at an application rate of 210 g a.i./ha ranged from 20 to 1500 μg/L (Ernst *et al.* 1991). These concentrations are well within the range of fenitrothion 96-h LC50s for some aquatic invertebrates (see Section 5.1). It must be recognized, however, that maximum surface water concentrations are rapidly attenuated as a result of dilution, transformation, and volatilization (Kingsbury 1977; Maguire and Hale 1980) and are not directly comparable with concentrations of fenitrothion used to develop laboratory toxicity values. Toxicities as measured by conventional procedures (i.e., 24- to 96-h LC50s) may not predict the effects of the short-pulsed exposures to fenitrothion that organisms receive in the field. However, W.R. Ernst (personal communication) has indicated that water sampled from a small pond that was operationally sprayed with fenitrothion in 1991 and bioassayed against *D. magna* in the laboratory resulted in 100% mortality after 48 h at dilutions down to 12.5%.

The impact of the forestry use of fenitrothion on aquatic invertebrates in small ponds may therefore be substantial. The ecological implications of such impacts have not been investigated, but the potential exists for significant effects on vertebrate fauna that depend on these ponds to provide invertebrate food resources.

6. Amphibians and reptiles

No studies on the effects of fenitrothion on reptiles have been found. Turtles may be exposed to fenitrothion by feeding on tadpoles contaminated with residues (see Section 6.2.1).

6.1 *Laboratory studies*

In static bioassays in the laboratory, the 24-h and 96-h LC50s of fenitrothion for tadpoles of the green frog *Rana clamitans* were 9.9 and 4.9 mg/L, respectively (Lyons *et al.* 1976). Although fenitrothion had a half-life of only 3 d in the test vessels used in this study, total mortality of the animals occurred within 160 h at all of the acute toxicity test concentrations (4–11 mg/L). In chronic toxicity tests using fenitrothion concentrations ranging from 0.25 to 3 mg/L, symptoms of fenitrothion exposure included extensive hemorrhagic regions, jaw twitching, ecdysis, swimming difficulty, buoyancy problems, and colour change. The behavioural changes occurred at fenitrothion concentrations above 1 mg/L. After exposure to 1 mg/L, fenitrothion residues in the tadpoles were 5.30 mg/kg body weight at 1 h and peaked at 11.5 mg/kg after 1 d; this level remained relatively constant for 5 d and then declined with a half-life of about 7 d. As transient concentrations above 1 mg/L have been seen in small forest ponds sprayed operationally (see Section 5.3), tadpoles may contain fenitrothion residues high enough to adversely affect their predators.

In another laboratory study, Mohanty-Hejmadi and Dutta (1981) examined the effects of five organophosphorus (OP) pesticides on the developmental stages of the Indian bullfrog *Rana tigrina*. These studies employed long, continuous exposures of the animals to fenitrothion, a situation that does not normally occur in the field. Fenitrothion was the most toxic of the five compounds tested (demeton-S-methyl, malathion, fenitrothion, dimethoate, and DDT + methyl parathion), and the eggs were the most susceptible of the developmental stages tested. Effects of fenitrothion exposures at concentrations as low as 0.005 mg/L during the egg stage included developmental arrest at the feeding, limb bud, or hind limb stages, smaller size at the time of metamorphosis, and a prolongation of the normal period of larval development. For example, at 0.05 mg/L, only 48% of the tadpoles metamorphosed within 58 d of rearing, whereas 80% of the controls metamorphosed within 36 d of rearing. Dutta and Mohanty-Hejmadi (1978) argued, from similar results seen with the OP insecticide dimethoate, that an increase in the development time of the larvae would prolong the period that the animals must stay in temporary ponds and increase their susceptibility to both predation and desiccation. In addition, a smaller size at metamorphosis may mean a diminished chance of survival later.

Pawar *et al.* (1983) and Pawar and Katdare (1983, 1984) observed mortality and developmental effects when eggs and tadpoles of the narrow-mouthed frog *Microhyla ornata* were exposed to the OP insecticides fenitrothion and malathion. The animals were exposed to technical grade fenitrothion with the test concentrations renewed every 24 h. The 96-h LC50s for the yolk plug-stage embryo and the 8-d-old tadpole were 3.21 and 1.14 mg/L, respectively (Pawar and Katdare 1984). At 3 mg/L, the number of embryos that hatched was less than 75% of the number of unexposed embryos hatching. The most common developmental abnormality caused by fenitrothion was blisters on the bodies of the tadpoles, which appeared at a concentration of 3 mg/L. At concentrations of 5 mg/L and above, abnormalities such as curvature of the body axis, poor body pigmentation, feeble blood circulation, and retarded growth were prominent. The authors advanced several hypotheses to explain the effects of OP insecticides on developing amphibians including cholinesterase (ChE) inhibition, a disturbance in the osmoregulatory mechanism, and interference with the enzyme ATPase (Pawar and Katdare 1983).

The toxicity of fenitrothion to embryos of the clawed toad *Xenopus laevis* was related to the temperature of the test medium; the 24-h LC50 for embryos exposed at 18 and 25°C was >10 mg/L, but this value was reduced to 0.33 mg/L at 30°C (Elliott-Feeley and Armstrong 1982). After exposure to 10 mg/L for 24 h, most embryos were abnormal and did not survive past the feeding stage. Fenitrothion concentrations which resulted in 50% of the exposed embryos exhibiting morphological abnormalities (including altered body shape, microcephaly, edema, and abnormalities of the heart, spinal cord and notochord) which caused death of the animals after hatching were 4.2, 0.37, and 0.17 mg/L at 18°C, 25°C, and 30°C, respectively.

6.2 Field studies

6.2.1 Residues

After a natural pond received an aerial application of fenitrothion in the Lyons *et al.* (1976) study, the insecticide was readily taken up by tadpoles. This may explain why fenitrothion was more toxic to the tadpoles than the other OP insecticide tested (the 24-h LC50 for Orthene was 6433 mg/L). Tadpoles collected 1 h after the simulated operational application (280 g a.i./ha applied by a Cessna aircraft equipped with Micronair rotary atomizers) contained fenitrothion residues of 0.61 mg/kg in whole body samples, which were 185 times the concentration in a water sample collected at the same time. The authors concluded that fenitrothion levels in shallow ponds following operational applications could approach harmful levels for these tadpoles, but concentrations affecting the larvae should not usually be reached given the pond concentrations witnessed in their studies (0.003–0.025 mg/L). They further concluded that ingestion of contaminated tadpoles by Mallards *Anas platyrhynchos* should not result in mortality of the birds but might result in sublethal effects. Presumably these effects would be more severe in birds that are more sensitive to the toxic effects of fenitrothion than are the relatively insensitive Mallards.

In a similar study in Manitoba, Lockhart *et al.* (1977) collected adult frogs after an aerial fenitrothion application of 280 g a.i./ha. Frogs collected within 1 d of spraying near a stagnant water pool contained fenitrothion residues of 0.03–0.17 mg/kg wet weight. Initial surface water concentrations (1 h after the application) were 0.001 and 0.701 mg/L in a stream and in the pool, respectively, and initial subsurface concentrations were 0.001 and 0.009 mg/L. One explanation for the smaller degree of bioaccumulation seen in this study compared with the Lyons *et al.* (1976) study is that the more terrestrial adult frogs may be less exposed than the larvae collected in the earlier study.

Ernst *et al.* (1991) recently measured fenitrothion residues in small lentic ponds of the type that might be used as amphibian breeding habitat in New Brunswick. Surface water samples (0–1 cm water depth) were collected after an operational application of fenitrothion at 210 g a.i./ha during the annual spruce budworm control program. The small ponds (<0.5 ha in size with a depth of 1 m or less) were not protected from the spray by overhanging vegetation or spray buffer zones. Surface water concentrations ranged from 0.006 to 2.5 mg/L. The highest mean surface water concentration following a single spray was 1.5 ± 0.5 mg/L. As mentioned in Section 6.1, a static 96-h exposure to 1 mg/L fenitrothion in the laboratory caused behavioural changes in green frog tadpoles. Although Ernst *et al.* (1991) calculated dilution and dissipation factors indicating that fenitrothion concentrations would decrease from initial levels fairly rapidly, these lower concentrations may still be in the range of concentrations toxic to early developmental stages of these frogs (see Section 6.1). However, as these concentrations do not persist for very long, the greater threat to frogs may stem from the effects of the insecticide on their invertebrate food resources.

6.2.2 Population effects

Field studies of impacts on amphibians after operational fenitrothion applications have usually employed simple counts of pre- versus postspray calling or visible adult frogs or attempts to find dead amphibians. The majority of the studies have been uncontrolled and unreplicated, and the data have not been subjected to statistical analysis. Pearce and Teeple (1969), for example, censused a pond near the Miramichi River in New Brunswick before and after an operational fenitrothion application of 140 g a.i./ha. The census consisted of walking slowly once around the pond during the evening and counting all frogs seen in the water or on shore. During six nights of prespray and three nights of postspray censuses, the investigators observed no adult amphibian mortality, and they concluded that the application had no detrimental effect on frogs or toads.

Similar results were reported by Rick and Price (1974). Two fenitrothion applications of 140 g a.i./ha were made 9–17 d apart in Fundy National Park, New Brunswick. The investigators reported no acute effects on three species of salamanders, six species of frogs, or frog or salamander larvae and no immediate depression in the populations of green frogs at 14 ponds in the park. However, no data on long-term effects were collected.

Buckner (1974) reported seeing large numbers of salamander eggs and larvae in a pond on a fenitrothion treatment plot after two applications at 140 g a.i./ha. Close observation revealed no mortality after the second application, and a revisit to the pond 6 weeks after treatment revealed many larval salamanders.

The U.S. Forest Service (1971) reported that no effects on amphibians were seen in northern Maine forests that had been treated with fenitrothion at a rate of 140 g a.i./ha.

In a simulated operational application, fenitrothion was sprayed onto small (250 m^2) ponds 20 to 30.5 cm deep at rates of 224 and 896 g a.i./ha (Mulla *et al.* 1963). Tadpoles of the bullfrog *Rana catesbeiana* were introduced within 1 h of the application and their survival monitored for 24 h. No dead bullfrogs were found after 24 h.

In India, on the other hand, Thirumurthi *et al.* (1973) studied the effects of fenitrothion applied at 500 g a.i./ha from a height of 2–3 m by helicopter. The application was made to rice fields flooded with impounded water to a depth of about 10–15 cm. Little information was provided (e.g., the species of frogs censused), and the concentration of the pesticide in the water was not measured; however, the investigators counted 21 dead frogs and no live frogs after the application and so assumed that the insecticide caused 100% mortality of the frogs in the impoundment. The maximum expected concentration of fenitrothion in the impoundment given the application rate and the water depth would be approximately 0.33 mg/L.

In one other field study, Bendell *et al.* (1986) conducted pitfall trapping for adult amphibians in 50-ha jack pine forest blocks that were treated twice with fenitrothion at 210 g a.i./ha. In a preliminary report, the authors concluded that fenitrothion (and the microbial insecticide *Bacillus thuringiensis* [B.t.] applied as Futura XLV at 30 B.I.U./ha) may have decreased populations of the wood frog *Rana sylvatica* in the treatment blocks (there were too few captures of other amphibians to make further comparisons). In this study, the insecticides appeared to have prevented hatching of the eggs, survival of larvae, or metamorphosis of the larvae into adults. In the fenitrothion spray blocks, almost the same number of adult wood frogs were caught during trapping periods 3 weeks before the spray (five frogs) and at 4 weeks after the treatment (six frogs), indicating little recruitment into the adult population. In the unsprayed control plot, the number of frogs caught during the second trapping period increased over 80-fold (five frogs caught prespray and 431 caught postspray). Numbers were similar in the B.t. spray block.

Pearce and Price (1975) discussed why it is unlikely that widespread direct mortality of adult amphibians will be detected after operational forestry applications of OP insecticides such as fenitrothion. One of the reasons is that adult amphibians appear to be tolerant of OP insecticide exposure (Hall 1990) and able to survive even very high levels of ChE inhibition. Adult male Indian bullfrogs, for instance, survived (in a moribund state) exposure to the OP insecticide phosalone even after inhibition of ChE activity to beyond 90% (Balasundaram and Selvarajan 1990). Thus, it is unlikely that acute toxicity will cause extensive direct mortality of adult amphibians after operational applications of fenitrothion. However, as mentioned above, adverse impacts might occur with subadult amphibians, or the animals might suffer through an insecticide-induced removal of their food resources.

Possible effects of fenitrothion on long-term viability of amphibian populations were recently studied in an investigation of frog populations in northern New Brunswick (D.F. McAlpine and N.M. Burgess, unpublished data). In this study, mink frog *Rana septentrionalis* populations in ponds in areas with different fenitrothion spray histories were censused. The spray histories were categorized as low, medium, or high amounts of fenitrothion applied based on the spray frequency in the 5 years preceding the study: zones classified as "low" had no fenitrothion applications between 1986 and 1990, "medium" areas had one application in that 5-year period, and "high" spray areas had three applications within the 5-year period. Overall means in frog counts across the spray zones showed a decline in frog numbers from zones of low and medium spray application to the high spray zone (Table 1).

Statistical analyses of these data indicated that there were significant differences in mean frog counts not only between spray zones, but also between ponds *within* the spray zones, indicating variability in frog numbers between different ponds in the individual spray zones. Analysis of the overall mean frog counts from the different spray zones indicated that the high spray zone had significantly lower mean counts than either the low spray zone or the medium spray zone. The low spray zone, on the other hand, had lower frog counts than the medium spray zone.

Frog habitat quality data (abundance of emergent and submergent vegetation and water chemistry parameters) were collected at the same time as the frog counts. A multiple regression analysis indicated that variables significantly related to frog densities were the abundance of submergent vegetation, the fenitrothion spray history, and the amount of sulphate in the water (a measure of acidity). These three variables alone could account for about 53% of the variation in the frog counts between different ponds.

Some of the possible influences that were not measured in this study include the abundance of predators, the level of human disturbance, and the potential for immigration into the study ponds from nearby areas that were unsprayed or contained large water bodies or running water where the impact of the insecticide might be diminished. Nevertheless, the census results for the mink frog suggest that frequent applications of fenitrothion may be having a detrimental impact on frog populations in New Brunswick. Because these results represent the outcome of only one season of fieldwork, and because a number of different variables may influence mink frog density in the area, further work is required to confirm these conclusions and investigate cause-and-effect relationships between fenitrothion applications and frog densities. Possible explanations for the reduction in frog densities in the frequent application areas might be removal of frog food resources by the broad-spectrum insecticide or a reduction in frog reproductive success as a result of the types of developmental effects observed in the laboratory by Pawar and Katdare (1983) and Mohanty-Hejmadi and Dutta (1981). As amphibians are important components of ecosystem integrity (Orser and Shure 1972; Burton and Likens 1975), the entire question of insecticide effects on amphibian populations demands further study.

7. Fish

The acute toxicity (96-h LC50) of technical fenitrothion to various species of fish ranges from 1000 to 5000 μg a.i./L (Symons 1977; Thellan *et al.* 1987; Eidt *et al.* 1989; Ernst and Doe 1989). Specific 96-h LC50s for various species are 4300 μg/L for channel catfish *Ictalurus punctatus*, 1000 μg/L for bluegill sunfish *Lepomis macrochirus*, 3200 μg/L for fathead minnow *Pimephales promelas*, 1000 μg/L for rainbow trout *Oncorhynchus mykiss* (all from Sanders *et al.* 1983), and 900 μg/L for Atlantic salmon *Salmo salar* (Wildish *et al.* 1971).

Table 1. Geometric least-squares mean frog counts per transect in New Brunswick forest ponds with different fenitrothion spray histories (from D.F. McAlpine and N.M. Burgess, unpublished data)[1]

Study pond	No. of transects	Spray zone[2]	Mean frog counts/transect	95% Confidence limit
1	12	Low	7.094	3.860 - 12.477
2	11	Low	20.952	11.871 - 36.450
3	10	Medium	65.425	36.864 - 115.513
4	13	Medium	12.054	6.980 - 20.349
5	10	High	3.862	1.721 - 7.680
6	10	High	3.990	1.793 - 7.908
7	10	High	3.523	1.532 - 7.077
8	12	High	6.151	3.259 - 11.001
9	16	High	0.711	0.097 - 1.657

[1]Frog count data were transformed using natural logarithms ($\ln x + 1$) to meet assumptions of normality and homogeneity of variances. To overcome the unbalanced design (different numbers of transects between ponds), least-squares means were used to determine unbiased estimates for the mean frog count at each pond. The ln-transformed means and standard errors were back-transformed to give the geometric means shown in the table.

[2]See text.

In a comparison of the conventional liquid technical formulation of fenitrothion with Sumithion 20F, a flowable formulation, Ernst and Doe (1989) observed similar levels of toxicity: 96-h LC50s for the rainbow trout were 1700 μg a.i./L for the technical product and 2200 μg a.i./L for the 20F formulation, respectively; the 96-h LC50 was 3100 μg a.i./L for both the technical product and formulation for the threespine stickleback *Gasterosteus aculeatus*..

Klaverkamp *et al.* (1977) reported that various life stages of the rainbow trout exhibited varying degrees of susceptibility to fenitrothion. The embryo was the most tolerant, with a 24-h LC50 of >34 000 μg a.i./L, followed by 10- to 11-d-old yolk sac fry at <34 000 μg a.i./L. The least tolerant was the fingerling stage, at 3400 μg a.i./L.

The acute NOEC for rainbow trout has been reported as 440 μg a.i./L (Sumitomo Chemical America, Inc. 1991).

At less than acute toxic levels, fenitrothion can induce sublethal effects in fish. For example, feeding activity is reduced at 1000 μg/L (Symons 1973), and swimming is inhibited at 500 μg/L (24-h exposure) (Peterson 1974). Bull and McInerney (1974) reported an increase in coughing and swimming inhibition after a 2-h exposure to formulated fenitrothion at concentrations of 0.04 μL formulation/L (formulation = 11% fenitrothion). Scherer (1975) reported that goldfish *Carassius auratus* avoided water containing fenitrothion, with an avoidance threshold of 10 μg/L.

Lockhart *et al.* (1977) reported that brain ChE activities in wild fish did not differ before and after exposure to fenitrothion in stagnant water with a surface concentration of 700 μg/L.

The bioconcentration factor for fenitrothion is relatively low. The values reported range from 30 in common carp *Cyprinus carpio* muscle (Tsuda *et al.* 1990) to 2300 in whole guppy *Poecilia reticulata* (De Bruijn and Hermens 1991). Clearance of fenitrothion is rapid; half-lives have been reported to be between 3 h and 2 d (Fairchild *et al.* 1989). The half-life of fenitrothion in the guppy was 15 h (De Bruijn and Hermens 1991).

Based on comparisons of fenitrothion concentrations detected in lotic water with concentrations required for acute toxic effects to fish, the hazard associated with direct effects of fenitrothion on fish in streams is low. The hazard due to indirect effects resulting from aerial spraying with fenitrothion on fish inhabiting lotic ecosystems is also expected to be low because of the transitory effects that have been observed on invertebrates inhabiting these systems.

The hazard to fish attributed to direct oversprays, however, may be greater in small lentic habitats (ponds) where surface water concentrations of fenitrothion as high as 2500 μg/L have been detected. Even though the fenitrothion concentration in the water at depths of 0.3 and 1 m may be 2–4 times less than concentrations observed in the surface layer

(Fairchild 1990), laboratory data indicate that such levels of fenitrothion in water may be toxic to fish (NOEC = 440 μg/L for rainbow trout) and to fish food (NOEC = 1.0 μg/L for *D. magna*). Ponds can also act as slow-release reservoirs that allow fenitrothion concentrations to remain elevated in downstream water for periods longer than normally expected in flowing streams. This scenario could enhance any adverse effects in the downstream habitat.

Small beaver ponds in New Brunswick provide substantial areas of good quality fish habitat, particularly for brook trout *Salvelinus fontinalis* (D. Morantz, personal communication). The streams typically dammed by beaver are small to medium in size and tend to have low flow in summer. Beaver dams increase stream depth, thereby improving fish-rearing conditions in summer. The dams themselves offer excellent depth and cover conditions for fish. For these reasons, stream areas immediately upstream and downstream of the dams often have high concentrations of trout. Depth, cover, shading, and stream braiding conditions associated with beaver ponds also contribute to increased invertebrate production, which further enhances these habitats for trout. Because spray buffer zones are not required for small water bodies (i.e., nondesignated rivers and lentic systems <5 ha in size) in New Brunswick, areas such as beaver ponds are not protected by provincial buffer zones.

The overspraying of small ponds with fenitrothion can result in fenitrothion concentrations in water that may cause sublethal or lethal effects in the aquatic invertebrate and/or fish populations. Because these areas can be productive fish habitat, there is cause for concern. Precautions against overspraying fish habitat should be taken.

8. Birds

8.1 Toxic mode of action

Fenitrothion, as an OP insecticide, has the same basic mode of action in both insects and vertebrates. All OP (or anticholinesterase) insecticides disrupt nervous activity by blocking synaptic transmissions in the cholinergic tracts of the nervous system. Fenitrothion exhibits its toxic mode of action by inhibiting the enzyme acetylcholinesterase (AChE) in nervous tissue. The normal function of AChE is to catalyse the hydrolysis (to choline and acetic acid) of the neurotransmitter acetylcholine (ACh). ACh is released through nervous stimulation at nerve synapses. Inhibition of AChE occurs as a result of binding of the pesticide to the enzyme, which blocks the enzyme's active sites. When the activity of AChE is inhibited, the neurotransmitter ACh accumulates to toxic levels. Although birds rapidly metabolize and excrete fenitrothion, the bound enzyme is fairly stable, so that complete recovery from intoxication is slow (taking longer than 10 d) and includes the biosynthesis of new enzyme (Blaber and Creasey 1960; O'Brien 1960).

Reviews of the mode of action of OP insecticides and symptoms of OP insecticide poisoning can be found in NRCC (1975), WHO (1986), Ecobichon (1991), and Grue *et al.* (1991). Grue *et al.* (1991) listed the symptoms that have been reported in humans and mammalian and avian wildlife. Ecobichon (1991) summarized the symptoms of OP insecticide poisoning (mainly from reports of poisoned humans) as those stemming from stimulation of the autonomic nervous system (increased secretions, bronchoconstriction, contraction of pupils, gastrointestinal cramps, diarrhea, urination, and bradycardia) and the junctions between the nerves and muscles of the autonomic nervous system (tachycardia, hypertension, muscle fasciculations, tremors, muscle weakness and/or flaccid paralysis) and those resulting from effects on the central nervous system (restlessness, emotional lability, loss of coordination, lethargy, confusion, loss of memory, generalized weakness, convulsion, cyanosis, and coma). Birds typically show loss of coordination, high carriage, wing drop, wing shivers, falling, tremors, salivation, loss of righting reflex, tetany, laboured breathing, contraction of pupils, secretion of tears, and wing beat convulsions (Tucker and Crabtree 1970; Hudson *et al.* 1984; Grue *et al.* 1991).

The immediate symptoms of fenitrothion poisoning are related to the inhibition of AChE. With accumulation of ACh at the nerve endings, there is continual stimulation of nervous activity at the synapses, which leads to an inability to transmit nervous impulses. Loss of AChE activity may then lead to a range of effects resulting from excessive stimulation. In mammals, death is generally caused by respiratory failure due to blocking of the brain's respiratory centre, bronchospasm, and paralysis of the respiratory muscles (WHO 1986). In birds, however, fenitrothion may also cause death through starvation. Mortality in Common Grackles *Quiscalus quiscula* from dietary exposure to fenitrothion was largely a result of pesticide-induced anorexia (Grue 1982). Similarly, Zebra Finches *Poephila guttata* that died as a result of fenitrothion poisoning lost an average of 12.6% of their initial body weight after receiving a single oral dose of the compound (Holmes and Boag 1990a). Further, birds exposed to a high dose of fenitrothion had a significantly depressed core temperature. Holmes and Boag (1990a) speculated that death could therefore result from respiratory failure, cold stress, or weight loss and starvation as a result of reduced food consumption induced by the toxicity.

Fenitrothion is not among the anticholinesterase pesticides that cause delayed neurotoxicity in birds (Farage-Elawar and Francis 1988).

8.2 Species differences

Little has been written concerning species differences in sensitivity to OP insecticide poisoning. Mammals appear to be relatively insensitive to the toxic effects of fenitrothion, as well as many other OP insecticides, whereas insects and birds are more sensitive (with the exception of the apparently tolerant Mallard; see Table 2). Among bird species, smaller birds are more sensitive to the toxic effects of OP insecticides, including fenitrothion (see Section 8.4.1).

NRCC (1975) suggested several reasons for the low toxicity of fenitrothion to mammals compared with insects: differences in ChE inhibition, more rapid detoxification of fenitrothion (through dealkylation involving glutathione-S-alkyl transferase, an enzyme apparently absent in insects; Nag and Gosh 1989), differences in penetration and translocation of the insecticide to target sites, and differences in activation of fenitrothion to its toxic metabolite. Eto (1979) similarly attributed the large differences in toxicity between insects and mammals to the net difference in rates of activation, detoxification, and transfer to the target and to differences in the sensitivity of the target enzyme.

Detoxification mechanisms may also explain the differential toxicity of fenitrothion to birds and mammals. In a recent review, Walker and Thompson (1991) reported that differences in the abilities of birds and mammals to detoxify fenitrothion may be explained by differences in the amounts of two types of enzymes - the "A" and "B" esterases. The "B" esterases include cholinesterases such as AChE and butyrylcholinesterase. The "A" esterases include the phosphoric triester hydrolases (PTH), enzymes that hydrolyse neutral OP triesters such as fenitrothion and its metabolite fenitrooxon. The PTH enzymes are well represented in mammals, but birds have little or no PTH in serum or plasma and relatively low levels in liver microsomes. Insects may also not have these PTH enzymes. According to Walker and Thompson (1991), the important "B"-type neurotransmitter enzymes such as AChE are protected from organophosphate inhibition by the "A"-type esterases in mammals, but not in birds and insects.

8.3 Routes of exposure

Birds can be exposed to fenitrothion following applications of the insecticide through four routes: 1) inhalation of respirable spray droplets and vapours; 2) dermal contact with the spray cloud and contaminated vegetation; 3) ingestion of contaminated food; and 4) ingestion of residues during preening of contaminated plumage.

Busby *et al.* (1989) summarized data concerning the different routes of exposure for wild birds. Although there is little quantitative information on the relative importance of different routes of exposure, all four routes appear to have some importance in the total exposure of the forest avifauna.

The potential risk from inhalation of respirable particles by forest birds during fenitrothion applications has not been studied. Exposure to the spray cloud through inhalation may seem to be of only minor importance, as respirable droplets containing the pesticide make up only a small portion by volume or mass of an aerial spray and because droplets are present in the air for at most only a few hours after spraying. However, these characteristics of the spray cloud may vary with the spray formulation and the conditions under which it is applied, and exposure via the inhalation route may be important in some cases. When the application is made in the early morning (when bird activity is intense) and weather conditions favour a slow dissipation of fenitrothion vapour (Crabbe *et al.* 1980), the potential for inhalation exposure may be high. Busby *et al.* (1983) argued that inhalation exposure may have contributed to an acute response (manifested as a marked and rapid ChE depression) observed in White-throated Sparrows *Zonotrichia albicollis* exposed to a 420 g a.i./ha fenitrothion application.

Table 2. Toxicity of fenitrothion to various species (from Smith 1987)

Test animal	Sex	Age (months)	Compound purity[1]	LD50 (mg/kg)
Mule deer *Odocoileus hemionus*	♂	13	95	>727
Rat *Rattus norvegicus*	♂	A[2]	tech.	740
	♀	A	tech.	570
Mallard *Anas platyrhynchos*	♂	3—4	95	1190
	♀	3	95	1662
Northern Bobwhite *Colinus virginianus*	♂	5	—[3]	32.0
	♂	2—3	—	27.4
	♀	5	—	23.6
Ring-necked Pheasant *Phasianus colchicus*	♂	3	—	55.6
Sharp-tailed Grouse *Tympanuchus phasianellus*	♂	6—7	—	53.4

[1]Purity of the test compound; tech. = technical-grade (pure) material.

[2]A = adult.

[3]Not provided.

Forest songbirds may be particularly sensitive to airborne pesticides because of their high respiratory rate and small body size. Exposure of birds would be expected to be greater than exposure of other organisms of similar body mass because fresh air passes through birds' respiratory systems during both inhalation and exhalation. For instance, given the respiratory frequency and tidal volume of the European Starling *Sturnus vulgaris* (92/min and 0.67 mL, respectively; Powell and Scheid 1989) and the atmospheric fenitrothion concentrations at the spray line for the first hour after application (40 ng/L of fenitrothion vapour and droplets smaller than about 40 μm in diameter; Crabbe *et al.* 1980), an 88-g (Feare 1984) European Starling would receive approximately 2.46 ng or about 28 ng of fenitrothion/kg body weight by the inhalation route during this hour (assuming that all of the inhaled material is absorbed into the bird). This value refers to exposure only to fenitrothion vapour and small droplets, and any inhalation by the birds of aerosolized material (droplets greater than about 40 μm) would raise the exposure. Further, volatilization of fenitrothion from an early-morning application has been reported to produce sustained low concentrations of fenitrothion vapour during the afternoon (Crabbe *et al.* 1980), which would lengthen the duration of exposure and increase the amounts inhaled.

Spray chamber exposures of small passerines to 1-h average airborne fenitrothion concentrations of 52 ng/L have led to significant brain AChE inhibition (Mineau *et al.* 1990). In this study, the birds were simultaneously exposed via other routes, and the contributions of inhalation, preening, and dermal absorption could not be separated.

No studies have been conducted with fenitrothion in which the contributions to toxicity of the different routes of exposure of birds could be separated. In the first investigation of its kind, Driver *et al.* (1991) studied the relative contributions to ChE inhibition of the different routes of exposure in Northern Bobwhites *Colinus virginianus* exposed to simulated agricultural applications of the OP insecticide methyl parathion. To measure the amount of inhalation exposure, the birds were covered in impermeable coverings so that their exposure could be limited to that occurring through inhalation only. Total airborne concentrations of the insecticide were below those that have been seen following operational fenitrothion applications (Crabbe *et al.* 1980; see above): 8 ng/L at the time of generation of the spray cloud, decreasing to 1.3 ng/L at 4 h after the spray cloud was produced. Inhalation of the pesticide aerosol contributed significantly to the reduction in ChE activity in the exposed birds. The peak impact was a 17% depression of ChE activity 1 h after the birds had been exposed.

Preening and dermal absorption may be a continued source of exposure to fenitrothion. However, maximum loss of fenitrothion residues on spruce and fir foliage occurs within 5 d of application, and most of the residues dissipate within 10–14 d after spraying (the half-life on conifer foliage is 2–4 d; Yule and Duffy 1972; Sundaram 1974). Bird exposure through direct contact with the contaminated foliage thus tends to occur primarily within the first 5 d after the application.

Data collected for Mallards indicate that fenitrothion has relatively high dermal toxicity (see Table 6, Section 8.4.1). If a similar situation occurs with passerine species, the dermal route of exposure may be important in the case of forest applications of fenitrothion. Morgan (1968) observed mortality in adult House Sparrows *Passer domesticus* when 10 μL technical fenitrothion was applied to the bases of their feet (according to the fenitrothion technical-grade label, about 12.5 mg a.i., equivalent to an oral dose of approximately 550 mg/kg body weight). The results of this study indicate that uptake through the feet may be an important route of exposure. However, data concerning the levels of dislodgeable fenitrothion residues on fresh conifer foliage following operational applications were not found. In the laboratory, Sundaram and Sundaram (1987) found average dislodgeable residues on conifer foliage of about 32 μg/g fresh weight in the first 5 d after the trees had been treated with Sumithion 20F, but the delivery rate from their atomizers (406 g a.i./ha) was calibrated to provide a high deposit of 270 g a.i./ha on the trees. In a simulated field study (Sundaram 1986, 1989) in which formulated fenitrothion was applied at a rate of 340 g a.i./ha to plastic-sheltered trees, dislodgeable foliage residues of approximately 50 μg/g dry weight were measured during the first 4 d following the application. It is difficult to relate these dislodgeable foliage residues to the amount of fenitrothion a songbird might acquire through normal contact with fenitrothion-contaminated surfaces.

Driver *et al.* (1991) determined that dermal absorption of deposited residues on Northern Bobwhites exposed to aerosolized methyl parathion was sufficient to produce toxicity. The authors assumed that the rapid decrease in ChE activity in the exposed birds was the result of uptake of the pesticide through the eyes (a contribution of 12% of the observed ChE depression at 1 h after exposure), whereas longer-term ChE depression (42% ChE depression by 48 h postspray) was the result of percutaneous uptake. It would be interesting to conduct a similar investigation of the contribution to toxicity from the dermal absorption of aerosolized fenitrothion; fenitrothion has a much higher acute oral to dermal toxicity ratio in Mallards (the only species for which this ratio can be calculated) than does methyl parathion (see Table 6, Section 8.1.4).

No information is available on the potential exposure of forest passerines to fenitrothion from preening of residues deposited on their feathers during or after an operational application. In the laboratory, Mineau *et al.* (1990) measured residues on the feathers of captive Zebra Finches exposed to a spray cloud of fenitrothion at concentrations similar to those seen during operational applications. When deposits were equivalent to 38, 51, or 139 g a.i./ha, feather residues were 1.31 $\pm$ 0.54, 1.21 $\pm$ 0.52, and 5.21 $\pm$ 2.41 mg/kg wet weight, respectively. With an application rate simulating an overswathing situation (a deposit of 255 g a.i./ha), the feather residue was 62.5 $\pm$ 28.5 mg/kg.

In the field, results from a British study that examined fenitrothion residues on Coal Tits *Parus ater* after an application of 300 g a.i./ha to a Scottish lodgepole pine *Pinus contorta* plantation indicated that residues on these small birds could be quite high (Hamilton *et al.* 1981). Maximum whole body fenitrothion residues of birds collected 1 d after the spray were 7.45 mg/kg, with at least 95% of total fenitrothion residues on the plumage and skin (or approximately 7 mg/kg on the feathers and skin). Hamilton *et al.* (1981) concluded that direct contact with the spray cloud and contact with contaminated foliage were the principal pathways of fenitrothion uptake in birds after spraying.

To calculate the potential exposure of forest passerines from preening of residues deposited on their feathers, more information is required on the residue burden on the plumage following operational applications, on the frequency of preening and the proportion of the body covered during preening, and on the preening efficiency (i.e., the proportion of insecticide that is removed from the feathers). Further, information is needed on the weight of the feathers relative to the total body weight of songbirds.

Driver *et al.* (1991) observed intense preening in free-ranging Northern Bobwhites for up to 2 min immediately following exposure to methyl parathion spray in their wind tunnel experiments and assumed that a Northern Bobwhite would ingest 50% of the deposited insecticide on its feathers. They were able to determine that ingestion of methyl parathion from preening could account for 8–10% of the measured brain ChE inhibition in the first 8 h after exposure.

Given the data gaps mentioned above, it is difficult to calculate the amount of fenitrothion that may be ingested by preening songbirds. If it is assumed that the feathers contribute approximately 14% to the total body weight of songbirds (P. Mineau, personal communication), then some rough calculations can be made. For instance, a Coal Tit weighs about 9.3 g (Perrins 1979) and therefore has a feather weight of about 1.3 g. This would mean that the Coal Tits collected by Hamilton *et al.* (1981) (see above) would have a total of about 9.1 μg of fenitrothion on their feathers. If the Coal Tit ingests 50% of the 9.1 μg fenitrothion deposited on its feathers by preening (Driver *et al.* 1991), the oral dose from this route would be about 0.5 mg/kg body weight. Similarly, if an 11.7-g Zebra Finch (Mineau *et al.* 1990) with a feather weight of 1.6 g ingests 50% of the 8.3 μg fenitrothion deposited on its feathers after an aerial application during which 139 g a.i./ha reaches the canopy, the dose to the bird would be about 0.4 mg/kg body weight.

Dermal absorption and ingestion of residues through preening may also be important routes of exposure for nestlings that receive fenitrothion residues from contact with the spray cloud or with the contaminated plumage of their parents.

The fourth potential route of exposure for forest songbirds is through the ingestion of contaminated food items. This longer-term exposure may occur through the ingestion of arthropods or plant material contaminated with fenitrothion. Fenitrothion residues have been measured in food items of songbirds (mostly invertebrates) at concentrations ranging from 0.01 to 25 mg/kg (Busby *et al.* 1989). Busby *et al.* (1989) provided data on the levels of fenitrothion residues that have been found on moss capsules (*Polytrichum commune*, a favourite White-throated Sparrow food item), substrate insects, flying insects, and spiders after operational fenitrothion applications. They listed the species of birds that feed on spruce budworm larvae and pupae and so may be particularly vulnerable as a result of their ingestion of contaminated food items following application. They also pointed out that nestlings may be more vulnerable than adults as a result of their high ingestion rates relative to body size.

Data for the White-throated Sparrow can be used to estimate the exposure to fenitrothion resulting solely from the ingestion of contaminated invertebrates. In this scenario, a White-throated Sparrow with a body weight of 27.4 g and a food consumption of 28.9% of its body weight per day in dry weight of food (Kenaga 1973) is consuming contaminated invertebrates with wet weight fenitrothion residues of up to 17 mg/kg (Busby *et al.* 1989). With a moisture content of 67% for arthropods (Ricklefs 1974), the maximum dry weight residues on the arthropods would be 51.5 mg/kg. Thus, the White-throated Sparrow could ingest 14.9 mg fenitrothion/kg body weight per day if it was feeding exclusively on arthropods contaminated with the maximum level of fenitrothion residues measured by Busby *et al.* (1989). Given the LD50 of 25.0 mg/kg for the larger Red-winged Blackbird *Agelaius phoeniceus* (see Table 3, Section 8.4.1), this might be close to the median lethal values for the White-throated Sparrow. In one acute lethal toxicity study conducted with the White-throated Sparrow (Forsyth and Martin 1993; see Table 4, Section 8.4.1), the LC50 was found to be similar to the maximum arthropod residues calculated above.

As mentioned above, it is possible that nestlings would be more vulnerable than adults to the effects of exposure to contaminated food because of their high ingestion rates relative to body size. One-day-old nestling White-throated Sparrows ate 176% of their body weight in insects, 4-d-old nestlings ate 72%, and 7-d-old nestlings consumed 77% (Busby 1982). Controlled dosing experiments showed that the survival and growth of nestling White-throated Sparrows given food contaminated with fenitrothion at concentrations that might occur following operational applications could be affected (Pearce and Busby 1980a). Effects varied from slight growth impairment to death, depending upon the age of the nestlings and the dosage rate employed. It has been suggested that altricial songbird nestlings are vulnerable to OP insecticide poisoning because they have low levels of AChE or a poorly developed blood-brain barrier or detoxifying system (Busby *et al.* 1989). At the same time, vulnerability of the adults may increase, as their food consumption is higher in the breeding season as a result of increased energy demands.

The above information indicates that exposure of songbirds to fenitrothion from aerial forest spraying can be quite significant. Each route of exposure can contribute to the overall toxicant burden that the bird receives and to the subsequent physiological response in the animal. Overall, the importance of the various routes of uptake of fenitrothion applied as an aerial spray in terms of their contribution to the toxic response probably follows the order dermal > preening > food > inhalation.

8.4 *Laboratory studies*

8.4.1 Toxicity

Acute oral toxicities of fenitrothion (LD50s) to several bird species can be seen in Table 3. The metabolite fenitrooxon is much more toxic than the parent compound in laboratory tests; acute oral LD50s for Ring-necked Pheasant *Phasianus colchicus* and Mallard are 10.6 and 12.5 mg/kg, respectively, compared with 55.6 and >259–1662 mg/kg for the parent compound (Sumitomo Chemical America, Inc. 1991). Thus, fenitrooxon shows similar high toxicities to these bird species even though the toxicities of the parent compound to these species are quite different (Table 3).

Lethal dietary toxicities of fenitrothion (LC50s expressed as mg a.i./kg feed) are listed in Table 4.

There is a general inverse relationship between avian weight and acute sensitivity to fenitrothion (see Tables 3 and 4); birds of smaller species are more likely to succumb or become intoxicated after fenitrothion exposure. This is possibly because of their higher metabolic rates and food consumption relative to body weight (Busby *et al.* 1989).

It should be noted, however, that the endpoint measured in these tests (mortality) may not be the most sensitive measure of fenitrothion's toxicity to wildlife; concentrations of a compound lower than those producing mortality may impair reproduction or the ability of an animal to survive in the long term (Busby *et al.* 1989). Further, it is difficult to extrapolate from a single value (i.e., the median lethal dose) for a toxicant administered under controlled conditions in the laboratory to the toxicity of that compound to animals in the wild. This is because there are numerous intervening variables that influence the response of animals in their natural environment. Factors such as age, sex, developmental differences, body condition, nutritional status, ambient temperature, and route of exposure can modify an animal's response or sensitivity to toxic chemicals, including fenitrothion (Busby *et al.* 1989). One of the most important variables in this list is the age of the birds. Altricial nestlings are more sensitive to OP insecticide intoxication than older birds because of some of the factors mentioned above. Young birds also have a more limited thermoregulatory ability and a higher rate of metabolism than older birds and may therefore be particularly vulnerable to the effects of OP insecticides.

One other characteristic of fenitrothion that may increase its potential for adverse impacts on forest birds is its high dermal penetration and toxicity (Hudson *et al.* 1979, 1984). In laboratory tests, fenitrothion appears to show low dermal toxicity to Mallards compared with other OP insecticides (Table 5). However, unlike the other OP insecticides tested, the dermal route of exposure appears to be more important than the oral route for fenitrothion toxicity (Table 6). Further, experiments by Morgan (1968) indicated that dermal absorption of fenitrothion applied directly to the feet of adult House Sparrows resulted in mortality. Mineau *et al.* (1990) provided evidence that the dermal route may be an important

Table 3. Fenitrothion LD50s (as mg a.i./kg body weight) for various bird species

Species	Sex	LD50 (mg/kg)	Reference[1]
Mallard *Anas platyrhynchos*	♂	1190	1
	♀	1662	1
		>259	2
Ring-necked Pheasant *Phasianus colchicus*	♂	55.6	1
Japanese Quail *Coturnix c. japonica*		161	2
	♂	115	3
	♀	140	3
	♂	84.85	4
	♀	73.87	4
Sharp-tailed Grouse *Tympanuchus phasianellus*	♂	53.4	1
Northern Bobwhite *Colinus virginianus*	♂	32.0	1
	♂	27.4	1
	♀	23.6	1
		23	2
Rock Dove *Columba livia*		42.24	5
Red-winged Blackbird *Agelaius phoeniceus*		25.0	6
Zebra Finch *Poephila guttata*		32	2
	♂+♀	20.3[2]	7

[1] 1 = Hudson *et al.* 1984; 2 = Sumitomo Chemical America, Inc. 1991; 3 = Kadota and Miyamoto 1975; 4 = Hattori *et al.* 1974; 5 = Hattori 1974; 6 = Schafer 1972; 7 = Holmes and Boag 1990a.

[2] LD50 not reported in the study; LD50 provided is result of probit analysis by Canadian Wildlife Service

route of exposure to fenitrothion for forest-dwelling birds. The importance of this exposure route and the resulting toxicity in forest-inhabiting birds have not been adequately addressed (see Section 8.3).

8.4.2 Effects on eggs and egg production

A number of laboratory experiments have examined the effects of fenitrothion exposure on eggs and egg production. During a fenitrothion application, the eggs may be directly exposed to the spray, or a proportion of the insecticide may be transferred to the eggs through contact with the contaminated parental plumage; effects on the embryos inside the eggs will occur only if the insecticide can be transferred across the eggshell to the embryo (see below). Alternatively, the female can transfer to the embryos fenitrothion residues that she has accumulated before egg laying. Laboratory studies have shown that this route is unlikely, however; White Leghorn hens (*Gallus gallus*) serially dosed with 2 mg/kg fenitrothion for 7 d transferred only 0.2% of the dose to their eggs (or about 0.03 mg/kg of the 14 mg/kg cumulative dose; Mihara *et al.* 1979).

The ability of OP insecticides to move across the eggshell barrier has been reported. Embryos of Mallard eggs dipped at day 3 of development in a number of OP insecticides (not including fenitrothion) exhibited shortening and contortion of the axial skeleton (Hoffman and Albers 1984). Kulczycki (1975) reported that an aqueous (0.2%) solution of fenitrothion sprayed on Ring-necked Pheasant eggs at "realistic" (unspecified) field rates resulted in significantly reduced hatching success and an increased proportion of problems in the hatched chicks. The impact was especially pronounced when the eggs were sprayed late in incubation; hatching success of eggs sprayed at day 20 of incubation was about one-half of that seen in control eggs. Further, hatchlings suffered primarily from incomplete paralysis of the legs and paralysis of the neck muscles, symptoms that are commonly associated with administration of anticholinergic agents to hatched birds. These results suggest that quantities of the insecticide that pass through the eggshell are sufficient to affect the cholinergic system not only of the developing embryo but also of the hatching chick.

Laboratory studies of the effects of fenitrothion on egg production have been conducted with Japanese Quail *Coturnix c. japonica*, Northern Bobwhites, and Mallards administered fenitrothion in their feed. Egg production was reduced in chronic exposures when Japanese Quail were fed diets containing 50 mg fenitrothion/kg feed for 4 weeks (Kadota and Miyamoto 1975). The dose groups were 0, 1.5, 15, and 50 mg/kg feed. No effects on behaviour, mortality, body weight, or food consumption were observed. Brain ChE activity was significantly inhibited in the 15 and 50 mg/kg dose groups. Miyamoto (1978) reported that fenitrothion at 10 ppm in the feed of Northern Bobwhites or 100 ppm in the feed of Mallards did not adversely affect adult growth and behaviour; egg production, weight, quality, or hatchability; or growth or viability of the young. Finally, Kadota and Miyamoto (1975) reported that fenitrothion concentrations of 10 ppm in the feed of Northern Bobwhites and 30 ppm in the feed of Mallards did not affect the adult birds, their egg production, or the hatchability or success of their chicks.

Table 4. Fenitrothion LC50s (as mg a.i./kg feed) for various bird species

Species	LC50 (ppm)	Reference[1]
Mallard *Anas platyrhynchos*	2482	1
Ring-necked Pheasant *Phasianus colchicus*	453	1
Japanese Quail *Coturnix c. japonica*	440	1
Northern Bobwhite *Colinus virginianus*	157	1
Common Grackle *Quiscalus quiscula*	78	2
White-throated Sparrow *Zonotrichia albicollis*	50	3

[1]all tests consisted of feeding the birds fenitrothion-contaminated food for 5 d followed by a 3 d observation period.

[2]1 = Hill *et al*. 1975; 2 = Grue 1982; 3 = Forsyth and Martin 1993.

Table 5. Acute dermal toxicity of organophosphorus insecticides to Mallards (from Hudson *et al*. 1979)

Compound	Sex	Age (weeks)	Purity[1]	LD50 (mg/kg)
Demeton	♂	42—44	92	24
Dicrotophos	♂	53—61	80	14.2
Disulfoton	♂	42—48	97	192
Fenitrothion	♂	55—61	95	504
Fensulfothion	♀	55—60	90	2.86
Fenthion	♂	42—44	99	44
Methyl parathion	♀	20—23	80	53.6
Mevinphos	♀	55—58	100	11.1
Monocrotophos	♂	54—57	75	30
Parathion	♂	43—45	99.5	28.3
Phorate	♀	55—61	88	203
Phosphamidon	♀	54—60	85	26.0

[1]Purity of the tested compound in percent active ingredient.

Table 6. Ratio of acute oral toxicity to acute dermal toxicity of organophosphorus insecticides to Mallards (from Hudson *et al.* 1979)

Compound	Acute oral LD50 (mg/kg)	Acute dermal LD50 (mg/kg)	Ratio[1]
Demeton	7.19	24	0.30
Dicrotophos	4.24	14.2	0.30
Disulfoton	6.54	192	0.03
Fenitrothion	1190	504	2.36
Fensulfothion	0.75	2.86	0.26
Fenthion	5.94	44	0.13
Methyl parathion	60.5	53.6	1.13
Mevinphos	4.63	11.1	0.42
Monocrotophos	4.76	30	0.16
Parathion	2.40	28	0.09
Phorate	2.55	203	0.01
Phosphamidon	3.81	26.0	0.15

[1]The ratio is the acute oral LD50/acute dermal LD50; when multiplied by 100, this is the dermal toxicity index (DTI) of Hudson *et al.* (1979).

In the current data submission (Sumitomo Chemical America, Inc. 1991), reproduction studies with the Northern Bobwhite and Mallard were provided. With the Northern Bobwhite, treatment-related mortalities and clinical signs of toxicity were apparent at fenitrothion concentrations of 45.4 and 65.6 mg/kg feed. Egg production was affected at 30 mg/kg (the lowest dose tested), and hatchability of eggs was reduced at 65.6 mg/kg. Food consumption and adult weight gain were significantly reduced at 65.6 mg/kg. Reductions in egg production at 30, 45.4, and 65.6 mg/kg were dose dependent and appeared to be treatment related. As egg production was lowered in the lowest dose group in this study, a no-observed-effect level (NOEL) could not be established.

With the Mallard, fenitrothion did not cause mortality or overt signs of toxicity in adult birds and had no adverse effects on adult growth and behaviour, egg production, egg weight and quality, hatchability, or growth and viability of young at concentrations of 51, 74, or 107 mg/kg (nominal) in the feed. Treatment-related effects on adult body weight were seen at 74 and 107 mg/kg (nominal). The NOEL for this study was approximately 40 mg/kg feed (51 mg/kg nominal treatment rate with an approximately 80% inclusion rate of the compound in the feed).

From the above laboratory data, it appears that the threshold for sublethal effects on reproduction following chronic exposures of fenitrothion in the diet of Japanese Quail, Northern Bobwhites, and Mallards is between 10 and 30 mg/kg feed. Concentrations in this range have been detected in food items of birds following operational applications of fenitrothion (Busby *et al.* 1989), but little information is available on the persistence of these residues on or in these food items following the applications. Persistent high concentrations in bird food items over a 4-week chronic exposure period may be unlikely. Few data are available on the effects of dietary concentrations of fenitrothion on reproduction in forest birds (see Section 8.5).

In another study (Sumitomo Chemical America, Inc. 1991), the influence of a direct spray of fenitrothion to control ectoparasites on egg-laying hens was studied. Heisdorf Nelson and Rockhorn hens were sprayed with dilutions of a 10% EC formulation of fenitrothion. The highest concentration tested was a 10-fold dilution of the 10% EC formulation, and the lowest was a 40-fold dilution of the same formulation. The treatments were made with an ordinary garden sprayer to wet the birds. Groups of 10 birds were housed together in 1.5-m^3 cages that were enclosed in plastic for 15 min during six successive spray applications. Corresponding aerial concentrations in the birds' holding cages could not be calculated. The applications were separated by a 1-week period. The egg-laying rate for the Rockhorn hens was decreased at all concentrations compared with that of the control birds (no statistical significance was provided for the results). There were no mortalities of the hens after their exposure and no effects on the egg-laying rate of the Heisdorf Nelson hens or on the body weight or food consumption of any of the hens. Because of the method of application, it is difficult to relate these concentrations to what a bird might receive from direct contact with the spray cloud during a forestry application.

In one laboratory study of the effects of fenitrothion on songbird reproduction, Holmes and Boag (1990b) studied the effects of two doses of fenitrothion - 1.04 mg/kg body weight (expected to reduce the level of their brain AChE activity by about 50%) or 3.80 mg/kg body weight (expected to reduce AChE activity by about 70%) - administered by gavage to Zebra Finches. Although there was an inverse relationship between reproductive success and dosage rate, this relationship was not significant. Further, fenitrothion had no effect on the timing of reproductive events or on the size of the young at fledging.

Data collected on reproductive success in the Holmes and Boag (1990b) study were confounded by two factors. First, two of the females died in the high dose group within 2 d of dosing, which limited the sample size. Secondly, the birds were dosed at the egg-laying stage, a part of the reproductive cycle that may not be susceptible to disruption by OP insecticide exposure. Further, dosing at this stage would allow the adult birds time to recover to their normal behavioural activity levels before the hatching of their chicks (when behavioural effects caused by the exposure might influence parental care). As the authors stated, it would be simple to change the experimental protocol for this study and dose the adult birds at different stages of the incubation or nestling period. A further study should examine the effects on adult songbirds and their offspring when the adults are exposed to fenitrothion both through their diet and through contact with the spray cloud and contaminated surfaces. Intoxication of the nestlings through contact with the contaminated plumage of their parents and effects stemming from changes in the behaviour of the adult birds (e.g., changes in their nest attentiveness or food-provisioning behaviour) could then be examined.

8.4.3 Effects on songbird behaviour

One laboratory study has closely examined the influence of fenitrothion on the behavioural activity patterns of songbirds (Holmes and Boag 1990b). As mentioned in Section 8.4.2, Zebra Finches were administered fenitrothion at doses of 1.04 or 3.80 mg/kg body weight by gavage. These doses affected the level and diurnal pattern of perch-hopping activity in the treated birds. Behavioural activity (measured by perch hopping) was significantly depressed in the dosed birds on the day they were treated - 40% in the 1.04 mg/kg dose group and 80% in the 3.80 mg/kg dose group. One day after the treatment, there were still significant differences in activity between the dosed groups and the control birds, but the behaviour of the dosed birds returned to normal by 2 d after treatment. From an earlier study (Holmes and Boag 1990a), brain AChE activity was expected to be depressed on average by about 50% and 70% in the low and high dosage groups, respectively, and was expected to remain depressed for about 4–10 d.

8.4.4 Effects on waterfowl

Fenitrothion applied to ponds biweekly (by ground application to the water and surrounding vegetation at a rate of 448 g a.i./ha) over a 12-week period was reported to have no effect on adult Mallards inhabiting the ponds (Keith and Mulla 1966). As the ducks were not sprayed during the treatment and were supplied with uncontaminated feed during the study - a supplemental food source that would also prevent the animals from being affected by an insecticide-induced reduction in their invertebrate food resource - they may have been able to avoid exposure to the insecticide, and the absence of gross effects is not surprising. The birds did not breed on the artificial ponds, so any effects on reproductive success as a result of the application could not be examined.

8.5 Field studies

8.5.1 Fenitrothion use

Currently, fenitrothion is used in Canada mainly for control of eastern spruce budworm in New Brunswick forests as part of the largest and most continuous forest protection program in the world. In New Brunswick, fenitrothion spray operations have been conducted every year since the first experimental applications in 1966. The applications are made from the latter part of May to mid-June at dosage rates between 140 and 280 g a.i./ha (two applications at 210 g a.i./ha, spaced 5-6 d apart, are typically made; the maximum total allowed for two treatments for spruce budworm control is 420 g a.i./ha). Over the past two decades, most of New Brunswick has been sprayed at least once with fenitrothion, and some parts of the province have received up to 14 double applications (every 1.4 years, on average).

8.5.2 Field monitoring

Field monitoring of the effects of fenitrothion on forest songbirds commenced with the first experimental applications of the compound in 1966 (Pearce 1967, 1968). Intensive groundwork in the forests of New Brunswick after fenitrothion applications in 1967 included searches for dead and incapacitated birds in the treated blocks. Birds were found to be behaving abnormally in areas that had been treated by TBM aircraft emitting fenitrothion at 560 g a.i./ha. These birds had indications of OP insecticide poisoning (e.g., the birds were lethargic, were unable to maintain equilibrium when perching or to elude easy capture by the investigators, or were lying incapacitated on their backs on the ground). In addition, some dead birds were found, and some birds died after being captured by the researchers. Similar intensive fieldwork during 1988 in Newfoundland following the first of two fenitrothion applications of 210 g a.i./ha for hemlock looper *Lambdina fiscellaria* control revealed birds exhibiting the same types of abnormal behaviour (S. Fudge and Associates Ltd. 1989). Because fenitrothion has been used in wide-scale operational applications in eastern Canada in each of the 20 years between these studies, including, on average, applications to between 1 and 2 million hectares of New Brunswick forest annually (Busby *et al.* 1989), concerns about the impact of this insecticide on forest songbirds seem warranted.

Pearce (1974) supplied data on bird effects following early spray programs. Behavioural changes, reduced singing activity, and reduced movements were seen at application rates of 138 g a.i./ha and above. Birds found dead in sprayed areas treated with

206 g a.i./ha included Magnolia Warblers *Dendroica magnolia*, Blackburnian Warblers *D. fusca*, and American Redstarts *Setophaga ruticilla* (three of each). All birds had substantial whole body residues of fenitrothion, suggesting it as the cause of death.

However, the occurrence of songbird mortality following fenitrothion applications can be quite variable. Although Busby *et al.* (1989, Table 3.16) pointed out that dead birds have been found following applications of 140 g a.i./ha, only limited mortality was seen following an application at 10 times this rate (Lehoux *et al.* 1982). In small Scottish spray programs, there has been no evidence of bird fatalities due to fenitrothion applications at a dosage of 300 g a.i./ha (Crick and Spray 1987).

The effects of fenitrothion on forest songbirds have recently been reviewed by the Canadian Wildlife Service. In this report, Busby *et al.* (1989) reviewed data concerning 1) the use pattern of fenitrothion in New Brunswick over the past 20 years; 2) the toxicology of fenitrothion to avian species; 3) deposit, residues, and fate of the compound following operational applications; 4) routes of exposure to fenitrothion for birds inhabiting treated forests; 5) the techniques used to measure the impact of those exposures on the resident birds; and, finally, 6) the specific effects that may occur in birds inhabiting the treated forest, including alterations in brain ChE activity and reproductive and behavioural effects.

The Busby *et al.* (1989) report includes data published up to 1988 on the effects of fenitrothion on birds. As the authors stated in their introduction, the report represents both a compilation of a very diverse literature and a reanalysis of data available in the open literature and in unpublished reports. The following critical assessment of the risk of fenitrothion to forest songbirds from operational applications of fenitrothion is based on Busby *et al.* (1989), with incorporation of limited, recently published data.

Although the manufacturer was required to submit any new data collected on environmental impacts for the fenitrothion reevaluation, no data were submitted that were usable in the evaluation of the impacts of fenitrothion on forest songbirds.

8.5.3 Field measurements of toxicity

Before the development of techniques to measure ChE activity, methods used to examine impacts of OP insecticides on songbirds in the field involved population censuses (singing bird censuses, territory mapping, or mist-netting studies), carcass searches, nesting studies, or observations of abnormal behaviour. Busby *et al.* (1989) reviewed some of the difficulties, biases, and inadequacies of these methods and concluded that all of them tend to underestimate the actual impact on birds and provide only limited data on the true impact of a pesticide application. This is because of biases inherent in the methods themselves (e.g., the difficulty in finding carcasses and their rapid removal by scavengers, which can lead to large underestimates of actual mortality if spray impacts are measured by carcass searches alone) or biases associated with the techniques used (e.g., the small plot biases that are associated with a labour-intensive method such as territory mapping). Further, these rather crude methods are not capable of detecting impacts such as nestling mortality and low levels of adult mortality (Peakall and Bart 1983). Sublethal impacts such as prevention of breeding attempts and desultory incubation would also not be detected with these methods.

As the methods mentioned above usually fail to provide information on the true magnitude of the impact of fenitrothion applications on forest songbirds, the single method that has been relied on to provide such data is a comparison of the levels of ChE activity in birds collected from sprayed and unsprayed forests. Although subject to collection biases that tend to result in an underestimation of the true impact (see below), these data can provide a measure of bird exposure following any operational application. Thus, measurement of ChE inhibition in field-exposed animals has become an accepted method of monitoring exposure of nontarget organisms to OP insecticides.

The pattern of ChE inhibition and recovery varies depending upon the species and the level and duration of exposure. Factors that influence exposure include formulation and timing of the spray, weather conditions, aircraft and spray equipment, forest topography and structure, and accuracy of navigation (Busby *et al.* 1989; Hart 1990; Busby and White 1991). The time of maximum ChE depression may be almost immediate (i.e., on the day of spraying) or up to 11 d after the application. There is little information on the time required for ChE activity to recover to normal levels following treatment; most field studies have been terminated before recovery was complete. In the laboratory, recovery of brain ChE activity in small passerines following fenitrothion exposure may take 10 d (see Section 8.1).

Although the degree of ChE inhibition in any animal after exposure to an OP insecticide may vary in response to a number of intrinsic and extrinsic factors (Grue 1982; Busby *et al.* 1989), the mean level of brain ChE inhibition within a species appears to be dose related. However, the relationships between exposure to an OP insecticide, subsequent depression of ChE activity, and the corresponding biological consequences are still not well known (Mineau 1991); the relationship between a sublethal level of ChE depression and possible biological outcomes (e.g., effects on long-term survival or reproductive success) is especially difficult to predict. For example, what might be the effects on a bird that suffers a depression of its brain ChE activity to 60% of its normal value? If the bird survives the intoxication, does not succumb to the added stresses imposed by anorexia or an inhibited thermoregulatory capacity (Rattner and Franson 1984; Holmes and Boag 1990a), and avoids predation, it will eventually recover normal ChE activity. Because of unaccountable differences in individual birds, however (which may depend upon factors such as physiological and nutritional status), some birds may survive while others with the same level of ChE inhibition die as a result of the toxic effects of the pesticide (Zinkl *et al.* 1979).

For instance, Holmes and Boag (1990a), working in the laboratory with Zebra Finches, measured brain ChE inhibitions of 3.3–83.3% in birds dying after receiving a single oral fenitrothion dose of 3.80 mg/kg body weight and 31.9–83.7% in birds that died after being given a dose of 11.36 mg/kg. Average inhibition in the dead birds was 45.3% in the low-dose group and 62.6% in the high-dose group. In birds that survived the exposure, maximum brain ChE inhibitions of 49.5–75.3% were measured 1–3 h after dosing.

Variation in ChE inhibition in birds surviving or succumbing to fenitrothion exposure can also be seen in chronic feeding studies. In a study in which Common Grackles were fed fenitrothion, mean brain ChE inhibitions were 81% in birds dying and 59.3% in birds that survived the exposure (Grue 1982). In one other feeding experiment with fenitrothion (Forsyth and Martin 1993), brain ChE was inhibited by 57–81% (compared with controls) in White-throated Sparrows that died from the exposure and by 3.8–58% in birds that survived.

On the other hand, the *diagnostic* criteria for the consequences of measured levels of ChE inhibition are fairly well established. It has been suggested that brain ChE activity of 50% or less of the activity seen in nonexposed individuals represents a potentially life-threatening situation (Zinkl *et al.* 1979, 1980). This criterion of 50% inhibition representing a life-threatening situation is supported by the results of both laboratory and field investigations of mortalities from acute OP insecticide exposures, in which death has usually been associated with brain ChE inhibition of 50% or greater (Grue *et al.* 1991), and it is generally agreed that a measured inhibition of 50% or greater in a bird collected dead would be sufficient to diagnose the cause of death as OP insecticide-induced mortality (Ludke *et al.* 1975).

For fenitrothion field applications, Busby *et al.* (1989) summarized data indicating that fenitrothion-caused mortality in the wild is often associated with brain ChE inhibition between 40 and 80%. After an experimental fenitrothion application of 420 g a.i./ha, Busby *et al.* (1990) recovered one dead White-throated Sparrow 7 d after the forest block had been treated. Brain ChE activity of this bird was depressed by 51% compared with the mean ChE activity of unexposed White-throated Sparrows. Millikin (1987) caught two male Common Yellowthroats *Geothlypis trichas* in mist nets 4 d after an aerial fenitrothion application of 280 g a.i./ha. According to the author, the birds may have succumbed to a combination of insecticide intoxication and the stress of netting. ChE levels in these birds were inhibited by approximately 42–50%.

One other generally established criterion is that depression of brain ChE activity to a value of less than or equal to 80% of that seen in control birds is diagnostic of an individual bird's exposure to an OP insecticide (Ludke *et al.* 1975). Following operational applications, however, Busby *et al.* (1981, 1987a) found statistically significant reductions in brain ChE activity in sprayed versus unsprayed birds, even though the mean level of depression in the exposed birds was less than 20%. Therefore, it is not necessarily true that brain ChE depression of greater than 20% must be achieved before field exposure to fenitrothion can be demonstrated.

It has been argued (Mineau and Peakall 1987; Busby and White 1991) that an estimate of the mean percent level of ChE inhibition in birds diagnosed as exposed to OP insecticides can facilitate an assessment of the risk to the integrity of the population. Busby and White

(1991) proposed that concern for the integrity of the population might be justified if half the birds sampled were diagnosed as exposed to fenitrothion and had a mean level of 50% ChE inhibition compared with similarly collected unexposed birds. If, on the other hand, only a few birds were diagnosed as exposed but had the same mean level of 50% inhibition, concern for the population as a whole would be less. One of the tasks, then, for an assessment of the hazards to songbirds from operational applications of fenitrothion is to examine the proportion of birds sampled that have this degree of ChE inhibition. Before this is done, however, some of the biases inherent in this method of impact assessment must be addressed.

The limitations and biases associated with brain ChE determinations for assessing insecticide spray impacts have been addressed by several authors (Busby *et al.* 1981, 1989, 1991; Peakall and Bart 1983; Mineau and Peakall 1987). The biases stem from the fact that the method relies on the collection of live individuals exposed during the spray operation and is therefore influenced by the collection methods. For instance, birds intoxicated by anticholinesterase pesticides such as fenitrothion tend to be less active and vocal and less readily flushed during collection efforts. These effects are generally directly related to the amount of exposure the bird receives. Thus, the more intoxicated birds are much less conspicuous and are less likely to be collected during postspray sampling programs, which tend to sample visible or vocalizing birds. This bias towards the collection of less intoxicated individuals is probably only slightly compensated for by the collection of intoxicated individuals that are more conspicuous as a result of abnormal behaviour or uncoordinated movements (Mineau and Peakall 1987).

For instance, in a study with the related OP insecticide chlorfenvinphos, the flying activity of European Starlings caged in large aviaries was monitored after the birds were given a near-lethal dose of the pesticide (Fryday and Hart 1991). The birds spent significantly more time in cover (simulated bushes made from natural vegetation) and less time flying during the 8 h after administration of the compound. On the following day, the behaviour of the treated birds was similar to the behaviour of control birds.

To simulate a field study and examine this same effect, Busby and Pearce (1984) released territory-holding adult male White-throated Sparrows after giving them an oral fenitrothion dose of 10 mg/kg body weight, then observed the behaviour of the birds from blinds. The birds remained near the ground, became inactive, flew sporadically, and fed their nestlings only intermittently. Later, in the field, Busby *et al.* (1990) observed White-throated Sparrows behaving abnormally the day after an operational application; the birds made only short, erratic flights, sought dense cover, and did not sing.

The sampling of birds at various times during the period of inhibition and recovery of ChE activity may lead to a further source of error in estimating the true magnitude of ChE inhibition in an exposed population (Busby and White 1991). Maximum ChE inhibition can occur at various times after a spray application and may depend upon an individual bird's exposure and sensitivity (Busby *et al.* 1989). Birds can be collected before ChE inhibition begins, before maximum inhibition has been reached, or after maximum depression has occurred. Therefore, the average level of inhibition measured in a sample of birds collected at various times after a spray is necessarily an underestimate and does not represent the maximum level of inhibition that occurred in the population.

In summary, measurement of brain ChE activity in birds from an area treated with an OP pesticide will tend to underestimate the true amount of ChE inhibition in the population. After a review of the available data concerning ChE inhibition in birds following exposure to OP insecticides, Busby *et al.* (1989) concluded that predictive capabilities on an individual basis are limited (because it is difficult to predict the effects in any individual bird from a given level of ChE inhibition), particularly when sample sizes are small (comparisons with other individuals collected concurrently are limited) and the history of the birds is unknown (there is no information on the amount of exposure and the time between exposure and collection). However, with adequate sample sizes and when the collection techniques, data analysis, and data interpretation are approached in a systematic manner, trends in the data will reflect a level of exposure allowing a general assessment of the potential impacts of existing spray regimens. For fenitrothion, these data are available for the White-throated Sparrow.

8.5.4 White-throated Sparrows: a case study

White-throated Sparrows have been the focus of a number of field monitoring studies (Pearce and Busby 1980b, 1980c, 1980d; Busby *et al.* 1981, 1983, 1987a, 1991; Strong and Wells 1986; Fudge, Lane and Associates Ltd. 1986; Busby and Pearce 1988; S. Fudge and Associates Ltd. 1988, 1989; Millikin and Smith 1990). There are a number of reasons why the White-throated Sparrow is a suitable model for an avian impact assessment of fenitrothion (Busby *et al.* 1987b): White-throated Sparrows are abundant, inhabit ground habitats in closed and open areas in conifer forest, and are easier to work with than upper canopy birds such as warblers. Further, information is available on the effects of fenitrothion on various aspects of White-throated Sparrow biology including reproductive impacts (Busby *et al.* 1990), effects on growth and development of nestlings (Pearce and Busby 1980a), and effects on adult behaviour that might influence the provisioning of food to nestlings (Busby and Pearce 1984); data on residues in food resources (Pearce and Busby 1980d; Busby *et al.* 1989) and laboratory data on acute toxicity (Forsyth and Martin 1993) are also available.

Another reason for the usefulness of White-throated Sparrows is that impact data on these birds have been collected following recent operational applications. The operational parameters for the spray program have remained relatively constant over the last decade, and the ChE data used in this review have been gathered from birds collected in areas treated using standard application techniques. Thus, analysis and interpretation of these data will not be confounded by factors related to variation in operational techniques and spray technologies.

On the other hand, one drawback for a risk assessment that uses White-throated Sparrows exclusively is that these birds are not always the most severely affected species following aerial spray programs (Busby *et al.* 1991 and references therein). This may be at least partially due to the fact that White-throated Sparrows are larger than warblers and kinglets and hence may be less sensitive to the toxic effects of the compound. White-throated Sparrows are also opportunistic feeders and may be able to change food sources following an application to avoid contaminated food or local depressions in the invertebrate food resource.

In the exposure scenario calculated for White-throated Sparrows (see Section 8.3), it was determined that the birds could receive a potentially lethal dose of fenitrothion from the ingestion of contaminated food items alone. The likelihood of this exposure scenario might be questioned because of the fact that deposition of the chemical in the forest is highly variable, and there can be refugia of unsprayed insects, which could provide a source of uncontaminated food for the birds, even in treated areas (Moulding 1976; Millikin 1990).

A counterargument could be that the birds will preferentially feed on dead or dying treated insects, which may be easier to capture. Data to support this hypothesis are difficult to find, and no data have been found on bird predation of fenitrothion-intoxicated insects. In an anecdotal account (S. Fudge and Associates Ltd. 1989), workers monitoring effects of fenitrothion applications for hemlock looper control in Newfoundland noticed two intoxicated warblers near a large concentration of dead and dying insects they had found earlier. These observations were separated in time (by about a day), however, and no direct observations of the birds feeding on the intoxicated insects were made.

Although many birds are known to consume spruce budworm larvae (Mitchell 1952; Zach and Falls 1975; Wypkema 1982) and there is evidence that bird predation actually helps to regulate endemic budworm populations and prevent (Wypkema 1982) or contribute to the collapse of outbreaks (Blais and Parks 1964), bird predation of intoxicated larvae has not been reported. In fact, only one author has reported witnessing bird predation of pesticide-intoxicated arthropods in the field: Moulding (1976) watched a Rufous-sided Towhee *Pipilo erythrophthalmus* feed on dead and dying gypsy moth *Lymantria dispar* caterpillars strewn on the ground following a carbaryl application. In laboratory trials in which live and dead grasshoppers were offered simultaneously, Forsyth and Hinks (1991) found that captive juvenile Clay-colored Sparrows *Spizella pallida* preferred dead grasshoppers, whereas Vesper Sparrows *Pooecetes gramineus* preferred live grasshoppers. In one other study with birds (Bracher and Bider 1982), a short-term increase in bird activity in sand transects on the floor of a forest treated with the carbamate insecticide aminocarb was explained by surmising that the birds were foraging on the ground for invertebrates killed by the spray. This finding was similar to that reported by Stehn *et al.* (1976), who reported mammals scavenging dead or debilitated invertebrates in an Orthene-treated deciduous forest.

That free-living White-throated Sparrows are receiving sufficient amounts of fenitrothion to become intoxicated following operational forestry applications can be proven by examining data on ChE inhibition in birds collected from operational spray blocks. In

Table 7, summary data on brain ChE inhibition in White-throated Sparrows collected following operational fenitrothion spray programs in New Brunswick and Newfoundland are presented. The data are from birds sampled after single applications and after the first and second sprays of fenitrothion split applications, all at Canadian label rates. It is important to remember that the data in this sample most likely underestimate the true level of ChE inhibition in the population because of the reasons outlined above, although attempts were made to collect more inconspicuous and possibly heavily intoxicated animals in some of the monitoring programs.

The ChE inhibition data presented in Table 7 indicate that almost half of the White-throated Sparrows collected following fenitrothion applications ranging from 140 g a.i./ha (using ultra-ultra low volume [UULV] spray techniques) to 280 g a.i./ha had a brain ChE activity indicating that they had been significantly exposed to an anticholinesterase agent (using the ≥20% criterion of Ludke *et al.* 1975). Almost 17% had a brain ChE activity level (≥50% inhibition) indicating that they had been exposed to a potentially life-threatening dose.

One other point suggested by these data is that, with some sprays, the proportion of birds showing a potentially life-threatening inhibition of ChE activity is greater after the first application during fenitrothion split applications. This is most evident following applications in which the first spray has a great impact on the ChE activity of the birds (Fudge, Lane and Associates Ltd. 1986; Busby *et al.* 1989, 1991) (Table 7). This trend has also been observed during an analysis of the impact of split applications using ChE data collected for a number of species (Busby *et al.* 1989). Intuitively, one would expect the opposite to be the case: that a second application following the first by less than 10 d (which is standard practice) would cause an additive or synergistic depression of ChE activity in birds already inhibited by the first spray, resulting in greater inhibition than would be seen with a single application only. To explain this, Busby *et al.* (1989, 1991) hypothesized that birds that are more vulnerable (physiologically sensitive or more likely to be exposed by virtue of their habits, nest location, or other factors) are selectively removed (killed or severely debilitated) from the pool of birds that can be sampled by the first spray, thus biasing collection towards less vulnerable or less sensitive individuals when collections are made following the second spray. Busby *et al.* (1991) concluded that if this hypothesis is true, the impact of double fenitrothion applications on forest songbirds may be underestimated. It also suggests that little benefit would accrue to forest songbirds if only one application of fenitrothion is made (i.e., no split applications), as equally severe impacts may be seen with a single application alone.

The data presented in Table 7 provide only a summary of the actual measurements (mean number of birds with the two levels of inhibition), and a great deal of information is lost when the data are condensed in this manner. For instance, the range of inhibition seen in individual birds and the mean level of inhibition are not provided. In a recent analysis of the ChE inhibition data for White-throated Sparrows collected on days 0 and 1 after operational fenitrothion applications, Busby and White (1991, Figure 4) showed that the mean level of inhibition in birds diagnosed as exposed (≥20% inhibition) for application rates of 140, 210,

Table 7. Cholinesterase inhibition in White-throated Sparrows exposed to fenitrothion during recent operational spray programs

Application rate (g a.i./ha)	Spray[1]	% of birds inhibited ≥20%	% of birds inhibited ≥50%	n	Reference[2]
280	1	17	7	29	1
280	1	40	20	5	2
210	1	58	0	12	3
	2	50	7	30	
210	1	29	6	31	2
	2	37	0	27	
210	1	79	58	19	4
	2	81	24	21	
210[3]	1	0	0	23	5
	2	10	No data	20	
210	1	32	8	124	6
	2	44	6	32	
210	1	53	8	96	7
	2	63	10	137	
210[3]	1	77	55	32	8
	2	86	33	31	
140[3]	1	83	26	23	8
Means[4]	1	46.8 ± 26.6	18.8 ± 13.3	394	
	2	53.0 ± 24.4	13.3 ± 11.4	298	
	Both sprays	49.3 ± 25.9	16.8 ± 17.7	692	

[1] 1 = first application in a split application; 2 = second application.

[2] 1 = Busby *et al.* 1981; 2 = Strong and Wells 1986; 3 = Busby *et al.* 1983; 4 = Fudge, Lane and Associates Ltd. 1986; 5 = Busby *et al.* 1987a; 6 = S. Fudge and Associates Ltd. 1988; 7 = S. Fudge and Associates Ltd. 1989; 8 = Busby *et al.* 1991.

[3] UULV applications.

[4] Mean ± standard deviation.

and 280 g a.i./ha was approximately 46% (n = 3 spray events), 39% (n = 27), and 49% (n = 3), respectively. The range of inhibition for these three application rates was approximately 35–53%, 21–54%, and 39–59%, respectively.

These data indicate that White-throated Sparrows are prone to ChE inhibition following operational applications of fenitrothion. Effects associated with ChE inhibition that have been observed in the field have already been mentioned (e.g., abnormal behaviour and an inability to fly and elude capture). Recent reviews have pointed out that important responses to exposure to OP pesticides might also include anorexia, altered behavioural responses, increased vulnerability to predation, and compromised physiological ability to deal with natural stressors (Busby *et al.* 1989; Grue *et al.* 1991). Another effect is mortality.

An early review of fenitrothion (NRCC 1975) reported that immediate toxic effects in songbirds occur at a fenitrothion application rate of 206 g a.i./ha, although census work had indicated effects at 138 g a.i./ha. Mortality has been observed at rates as low as 140 g a.i./ha (Busby *et al.* 1989). Because the toxicity curves for fenitrothion are steep, small increases in the effective application rate (e.g., higher than normal deposits of the insecticide resulting from double swathing of spray lines, exceptional spray conditions, improved formulations and delivery systems that increase deposits, etc.) may result in sharply increased mortality. Although it is difficult to firmly relate the degree of ChE inhibition to mortality, in one study the estimate of songbird mortality was very similar to the proportion of birds exhibiting life-threatening (≥50%) ChE inhibition (Busby *et al.* 1990). In this study, the disappearance of 20% of free-living White-throated Sparrows was associated with a mean ChE reduction of 42%, and one dead White-throated Sparrow that was recovered 7 d after the application had a brain ChE activity level inhibited by 51%.

Given these relationships for brain ChE inhibition and mortality and the amounts of inhibition seen in monitoring programs following operational spray programs (Table 7), it is evident that the margin of safety for White-throated Sparrows exposed to fenitrothion applications is exceedingly narrow. Although this would lead one to expect quite obvious White-throated Sparrow mortality after operational applications, this is not the case. In fact, even when an application rate of 1400 g a.i./ha was used, the resulting mortality discovered during carcass searches was low (Lehoux *et al.* 1982). There are few explanations for this discrepancy. One reason may be that recent studies examining fenitrothion impacts on White-throated Sparrows have not concentrated on finding dead birds but have instead focused on locating live birds, as collection efforts have been geared to securing samples for ChE measurements. The difficulty of finding songbird carcasses has already been mentioned (see Section 8.5.3).

Reduced levels of ChE activity may also affect reproduction in forest songbirds. It has been reported that concentrations of a compound lower than those producing mortality may impair reproduction in birds, but few data are available concerning the specific effects of fenitrothion on forest songbirds. Two field studies that have examined the effects of aerial fenitrothion applications on the reproductive success of White-throated Sparrows (Millikin 1987; Busby *et al.* 1990) are described below.

Millikin (1987) and Millikin and Smith (1990) assumed that behavioural alterations caused by insecticide intoxication may change food-seeking behaviour, thus affecting parental care and influencing breeding success. To assess the long-term impacts of these sublethal effects, they compared annual return rates and fledging success over consecutive years in treated and control areas. In 1985, their study plot received an aerial application of fenitrothion at 280 g a.i./ha (an application of 293 g a.i./ha was made to a different plot by backpack sprayer in 1986).

Millikin (1987) estimated annual return rates and fledging success of White-throated Sparrows by 1) the number of fledglings caught in mist nets in treated and untreated areas; 2) observations of nests in each area; 3) observations of adult birds feeding their young; and 4) the capture dates of fledglings in treated and control areas. The mist-netting results indicated that the lowest number of birds and the lowest number of juveniles were caught on the treatment plot in the treatment year compared with all other years and compared with the control plot (statistical significance was not provided for these data). The fledging success for White-throated Sparrows on the treated plot as measured by the proportion of young caught was 14.2% in the treatment year versus 28.2% for the same year on the control plot. These results support a hypothesis of reduced fledging success in the year of application.

The number of male White-throated Sparrows returning the year after the treatment year was also lower than for other years and lower than that seen in the control area. These data suggest that the male birds might have been more affected by the spray (concurring with data collected at the same time showing that male White-throated Sparrows have a higher potential for exposure). Finally, some of the birds were found to be in breeding condition late in the season, which suggested that the spray had disrupted earlier nesting attempts in the treatment area and that renesting attempts were being made.

Millikin (1987) reported that fenitrothion caused short-term behavioural changes that affected breeding success through a lower proportion of young fledged and a reduced recruitment of individuals the year following treatment. Millikin's (1987) results also indicate that in treatment blocks where birds cannot easily forage in unsprayed areas nearby (i.e., spray blocks larger than 3 ha), fledging success is lowered and the number of birds returning the following year is decreased. Otherwise, if the birds can easily forage in unsprayed areas nearby (treatment blocks smaller than 3 ha), fledging success should be unaffected. As spray blocks in operational programs are thousands of hectares in size, reduced fledging success in areas treated operationally would be expected, especially when applications are made so that there are no untreated areas within the treatment blocks to provide food "islands" for the birds (cf. Moulding 1976).

Busby *et al.* (1990) studied the effects of an initial application of 420 g a.i./ha followed by a second application of 210 g a.i./ha 8 d later on the behaviour and reproductive success of White-throated Sparrows. The birds were inhabiting a 50-ha section of a 350-ha experimental treatment plot that received the 420 g a.i./ha application. The experimental plot was within a 10 000-ha operational spray block treated with 210 g a.i./ha. Effects on reproduction in the treated plot included territory abandonment, inability to defend a territory, disruption of normal incubation, and clutch desertion. The mean clutch size, number of eggs hatched, and brood size of the nests in the spray and control areas were not significantly different, but the mean number of young fledged and the mean number of young produced in the spray area were significantly lower than in the control area. This suggests that nests failed completely or were more or less normal in the treated area. The number of fledglings produced was one-third of the number in the control area, and the number of young produced was one-quarter of that produced in the unsprayed control area.

Busby *et al.* (1990) hypothesized that three processes could have contributed to the reduced reproductive success on the treated plot: 1) nestlings could have been exposed directly to the insecticide by contact with the spray cloud or with the contaminated plumage of their parents, or they could have been fed fenitrothion-contaminated food; 2) nestling survival may have been adversely affected by a reduction in food availability caused by the insecticide or by a change in the foraging efficiency of their parents; or 3) intoxication of the adult birds by the insecticide may have caused a reduction in the quality of parental care. Busby *et al.* (1990) were able to collect evidence that the first and third processes were occurring. They did not examine foraging effort and food abundance and so could not address the effects of a reduction in food availability and foraging efficiency on reproductive success. Support for this hypothesis, however, came from the results of Millikin's (1987) work reported above. Indeed, this effect might be even greater in operational spray blocks that are typically treated twice during the breeding season, causing a longer-duration reduction in food availability than would have occurred following the single application by Millikin (1987).

A possible criticism of the White-throated Sparrow reproductive success work conducted by Busby and colleagues (1990) is that the initial fenitrothion application was an intentional overapplication of 2 times the typical split application rate. The overapplication was intended to simulate the overexposure birds might receive if an area of forest is overswathed or for some reason receives a high deposit of the spray. This may not be an infrequent occurrence during operational spray programs; data presented by Armstrong (1977) indicate that there can be large variation in the amount of pesticide deposited during aerial spray programs (e.g., up to 237% of the amount emitted during "well-controlled" experimental spray programs).

Busby *et al.* (1990) measured brain ChE inhibition in White-throated Sparrows collected from a sprayed area adjacent to the reproductive success study plot following the 420 g a.i./ha spray. The level of inhibition resulting from the split application employing the 420 g a.i./ha spray could then be compared with the level of inhibition measured in White-throated Sparrows following conventional sprays. Busby and co-workers have claimed that similar levels of ChE depression can be seen in White-throated Sparrows following conventional 210 g a.i./ha split applications and following applications employing the 420 g a.i./ha overspray (Busby *et al.* 1989, 1990; Busby and White 1991).

In the study employing the overapplication, the mean level of brain ChE inhibition of all birds sampled was 42% following the first (420 g a.i./ha) spray and 30% following the second (210 g a.i./ha) spray (Busby *et al.* 1990). In monitoring activities following the Newfoundland fenitrothion spray program (Strong and Wells 1986; Fudge, Lane and Associates Ltd. 1986; S. Fudge and Associates Ltd. 1988, 1989), the mean brain ChE inhibition level of all White-throated Sparrows sampled ranged from 11 to 43% following first and second applications of 210 g a.i./ha. These data show that ChE inhibition following conventional sprays can be similar to that seen following a 420 g a.i./ha overapplication.

Busby and White (1991) recently summarized the White-throated Sparrow ChE data. Their Figure 4 shows the overlap in mean inhibition in birds diagnosed as exposed ($\geq$20% inhibition) between conventional 210 g a.i./ha applications and the 420 g a.i./ha spray. As mentioned earlier, the range of mean inhibition for birds diagnosed as exposed following 210 g a.i./ha sprays was 22–54%. The mean percent inhibition of exposed birds following the 420 g a.i./ha spray was approximately 50%. Busby and White (1991) also pointed out that mean percent levels of inhibition and proportions of birds with life-threatening inhibition were higher following several spray events at 210 g a.i./ha than they were following the 420 g a.i./ha spray. During bird collections following the 420 g a.i./ha spray (Busby *et al.* 1990), the whereabouts of territorial birds were known, and sampling these birds might have made the collections less biased (i.e., less likely to underestimate the true level of ChE inhibition) than those of typical monitoring programs. As ChE measurements from birds collected during typical monitoring programs tend to underestimate the true level of ChE inhibition in the population (see Section 8.5.3), the actual level of overlap between the 210 g a.i./ha and the 420 g a.i./ha sprays may be greater than that presented by Busby and White (1991).

The above data support the conclusion that a reduction in the reproductive success of White-throated Sparrows (to one-third of that seen for birds in the unsprayed forest) occurs when mean brain ChE activity is depressed to a level that may be seen following operational fenitrothion sprays. Because large tracts of forest can be treated on an annual basis during spruce budworm control programs, the White-throated Sparrow populations may not be able to compensate for this lack of reproductive success in one year by increased recruitment in unsprayed forest the following year. If these assumptions and hypotheses hold, one would expect to see some indication that insecticide applications for spruce budworm control are having adverse effects on White-throated Sparrow populations. Unfortunately, the available census data do not allow a close examination of White-throated Sparrow population trends in the areas that have been the focus of spruce budworm control efforts.

8.5.5 Other passerine studies

Field studies on the impacts of fenitrothion on songbirds have been conducted in Scotland. Fenitrothion was applied at 300 g a.i./ha for pine beauty moth *Panolis flammea* control in lodgepole pine plantations (Hamilton *et al.* 1981; Crick 1986; Crick and Spray 1987; Spray *et al.* 1987).

Hamilton *et al.* (1981) examined ChE activity in three species of songbirds. In 1979, about 50% of the Common Chaffinches *Fringilla coelebs* collected on days 1 and 4 following the spray had >20% inhibition of brain ChE. Even at 11 d after the spray, 50% of the collected birds still had >20% inhibition. The impact on Common Chaffinches in 1980 seemed more severe, as 78% of the birds taken on days 1 and 2 had >20% inhibition (the range in inhibition was about 21–75%), 90% collected on day 7 had >20% inhibition (range of 20–48%), and two birds of 12 (17%) collected 21 d after the spray had at least 20% inhibition. One Common Chaffinch was noticed that showed uncoordinated movements and was unable to fly. Brain ChE activity of this bird was inhibited by 50%, and it had a whole body burden of 1.38 mg fenitrothion/kg.

Every Coal Tit (n = 4) taken the day following the spray in 1979 showed severe inhibition (ranging from 40 to 60%), four of six birds collected at day 4 showed >20% inhibition, and one of two collected on day 11 showed >20% inhibition. Willow Warblers *Phylloscopus trochilus* collected in 1980 exhibited a curious pattern of ChE inhibition: 17 of 24 birds collected on days 1 or 2 had >20% inhibition, none of 12 collected on day 7 showed this level of inhibition, but three of 13 collected on day 21 showed >20% inhibition. The authors concluded that aerial spraying of forests with fenitrothion may present a considerable hazard to some songbird species but that a longer-term study of populations and behaviour would be necessary to investigate the nature and extent of this effect. This was the impetus for the studies reported by Spray *et al.* (1987).

Spray *et al.* (1987) conducted their work in lodgepole pine plots that were specifically sprayed to examine the effects of fenitrothion on songbirds. These researchers wanted to examine the effects of spraying on 1) annual variation in breeding populations; 2) short-term changes in abundance and activity of singing birds; and 3) breeding success, nestling growth, and parental feeding behaviour of Coal Tits using nest boxes. Four plots were monitored: two unsprayed control plots of 68 and 75 ha, a 78-ha plot sprayed in 1983, and a 64-ha plot sprayed in 1984 (both with fenitrothion at 300 g a.i./ha).

To examine annual variation in breeding bird populations, two 15-ha census plots were set up, one in the 68-ha unsprayed plot and the other in the 64-ha plot sprayed in 1984. The censuses, conducted using territory-mapping techniques, revealed that variation between years in the densities of breeding birds was naturally large and that fenitrothion had no measurable effect on the population densities the year following the treatment. The authors noted that small local population changes in the treated blocks may have been masked by the invasion of birds from adjacent unsprayed areas.

Shorter-term changes in bird abundance were measured by monitoring the singing activity of territorial birds during the week after spraying. The results indicated that fenitrothion spraying did not cause any short-term changes in the numbers of singing or calling birds, as any changes in the sprayed and unsprayed plots tended to mirror one another. Further, no dead or debilitated birds were discovered in the spray blocks, and there were no losses among parent birds using 21 nest boxes monitored following the 1983 and 1984 sprays.

The breeding performance of Coal Tits was measured by monitoring the number of eggs laid, hatched, and fledged and the mean brood size of clutches in the sprayed and control plots. The data were reported as the proportion of nest boxes in which Coal Tits successfully hatched or fledged broods. There were no significant differences in clutch size or brood size at hatching between nests in the sprayed and unsprayed areas or in brood size at fledging, although this last measure was slightly lower in sprayed areas in both years. Nestling growth and survival were also not significantly different, although the data revealed a trend for slower growth resulting in lower final weight of the nestlings exposed to the spray. No differences were found in hatching and fledging success, and no nest desertion was seen (although a slight decline in the number of visits the parent birds made to their nests was observed for 2–3 d after the fenitrothion application). From all these data, the authors concluded that, although changes in parental behaviour and nestling diet were observed and there was some evidence for the poisoning of nestlings, these did not affect nestling growth and fledging success.

Busby *et al.* (1989) summarized possible reasons for the lack of reproductive effects seen in the Scottish studies. One reason was that, because nestlings were protected from direct exposure to the spray cloud by the nest boxes, reduced reproductive success would likely be the result only of altered parental behaviour or exposure to contaminated food. A second reason was that reproductive success was reported in a manner in which the fledging of only one nestling was sufficient to consider the nest 100% successful, a method that might mask any impact of the spray on reproductive success. Finally, because details concerning the relative timing of the spray and events in the breeding phenology of the Coal Tits were not provided, it is difficult to evaluate the degree of exposure experienced by the birds.

One other report was found that discussed the impacts of fenitrothion on songbird reproductive success (Keith and Mullié 1990). Fenitrothion was applied by air at 500 and 1000 g a.i./ha in Senegal in investigations mounted to monitor the environmental impact of insecticide control programs for the desert locust *Schistocerca gregaria*. As part of the impact studies, breeding success of a number of bird species was monitored. The authors concluded that nesting success of Singing Bush Larks *Mirafra javanica* and Buffalo Weavers *Bubalornis albirostris* was probably affected by the fenitrothion treatments and that food removal, rather than direct toxicity, may have been the reason for the adverse impacts on reproductive success (the adults possibly abandoning nests before the young were fully fledged). (Similar conclusions were arrived at following aerial applications of the pyrethroid insecticide cypermethrin to control the lepidopteran pests *Tortrix veridana* and *Archips xylosteana* in Spain [Pascual and Peris 1992]. In the treated plots, where there was high arthropod mortality, there were significant adverse impacts on nestling mortality, daily survival rate, nest success, and nestling weight of Blue Tits *Parus caeruleus* nesting in nest boxes.) In Senegal, fledgling larks were found with severely inhibited ChE, however, and the authors assumed that many probably died on the treatment plots, presumably from direct toxicity. The authors concluded that reduced reproductive success was the most serious impact identified in their study (direct mortality was witnessed but was not widespread), as it would have the greatest potential for long-term effects on bird populations, and they recommended that further study of fenitrothion effects on avian reproduction be a high priority in areas where the insecticide is to be used to control locusts and grasshoppers.

8.5.6 Other forest birds

Although the focus of this avian risk assessment for fenitrothion has been on forest passerines, the possibility of adverse effects on other forest avifauna cannot be ignored. For instance, waterfowl raising broods in the treated forest may be adversely affected by the removal of their invertebrate food resource by this broad-spectrum insecticide. Fenitrothion is highly toxic to aquatic invertebrates (see Section 5), and recent studies (Fairchild *et al.* 1989; Fairchild 1990) have indicated that the compound can cause relatively long-term reductions in invertebrate biomass and diversity in small bog ponds of the type that are used as waterfowl breeding habitat in eastern Canada. At the same time, pond productivity, as measured by total invertebrate abundance and biomass, is probably the single most important factor influencing habitat selection by omnivorous ducks, at least in New Brunswick forests (Parker *et al.* 1992). This relationship occurs because of the reliance of ducklings and laying hens on animal protein. Thus, the importance of invertebrate biomass to waterfowl nesting success cannot be overlooked. Further, the close overlap between fenitrothion spraying and peak American Black Duck *Anas rubripes* hatch dates suggests that wild broods would be exposed to diminished invertebrate populations from hatching onward (Martin 1992). However, no research has examined the impacts that fenitrothion might have on waterfowl that breed in forest wetlands. A recent review of the literature, concentrating mostly on New Brunswick waterfowl (Martin 1992), represents the first attempt to examine this.

The American Black Duck, Ring-necked Duck *Aythya collaris*, Green-winged Teal *Anas crecca*, Wood Duck *Aix sponsa*, Hooded Merganser *Lophodytes cucullatus*, Common Merganser *Mergus merganser*, and Common Goldeneye *Bucephala clangula* are the most common waterfowl nesting in the forests of eastern Canada. The forested regions of New Brunswick are estimated to provide breeding habitat for at least 50% of the province's waterfowl and are especially important for American Black Ducks and Ring-necked Ducks.

American Black Ducks and Ring-necked Ducks select small wetlands for their breeding habitat; in Ontario, ponds less than 20 ha in size, and particularly those in the 1.5- to 4.0-ha size range, were preferred by both species (McNichol *et al.* 1987). The mean sizes of American Black Duck brood-rearing ponds were 4.5 and 2.1 ha in Maine and south-central New Brunswick, respectively (Renouf 1972; Ringleman and Longcore 1982); ponds modified by beaver are highly preferred brood-rearing habitat (Renouf 1972). In New Brunswick, there are typically no buffer zones around small ponds of 40 ha or less (Sexsmith 1987; the 1992 provincial spray permit placed buffer zones around ponds greater than 5 ha in size); therefore, these ponds are particularly susceptible to contamination during fenitrothion spray operations. Further, these ponds have limited volume for dilution of the chemical, have low flushing rates, have little canopy cover to protect them from direct deposition of the spray, and may be acidic, all characteristics that will tend to increase the impact of the insecticide on aquatic invertebrates (Fairchild 1990, 1991).

One published study has examined the effects of insecticide-induced reductions in invertebrate density on duckling growth and survival. Hunter *et al.* (1984) sprayed small (<5 ha) ponds with the carbamate insecticide carbaryl and examined the impacts on mixed broods of American Black Duck and Mallard ducklings. The ducklings, 7 d old when the ponds were treated, were observed for 9 d following the treatment. A 50% reduction in invertebrate biomass and numbers resulted in reduced growth rates of ducklings on the treated ponds compared with the growth rates of ducklings on untreated ponds. No mortality of the ducklings was observed, but death may have been avoided by treating the ducks when they were already 7 d old (past the age when duckling mortality is highest) and by taking the ducklings inside every night when death due to the effects of chilling and reduced body reserves might otherwise have occurred.

As mentioned above, some of the brood-rearing habitat for American Black Ducks and Ring-necked Ducks in eastern Canada is in acid bog areas. Ducklings inhabiting low-pH wetlands may be particularly sensitive to reductions in invertebrate food resources because the low diversity of aquatic invertebrates in these wetlands (Fairchild 1990) provides the ducklings with few alternative prey items. In addition, some taxa that are extremely important food items for ducklings in nonacidic wetlands (such as the Ephemeroptera) are not present in acidic ponds because of their pH sensitivity. Among the taxa that are present in small, low-pH ponds, Fairchild (1990) observed that an application of fenitrothion caused 1) a reduction in emergence for 6–12 weeks following the treatment; 2) a reduction in population densities of two of the ecologically important invertebrate families (the Chironomidae and Ceratopogonidae) by greater than 50% for 1 month after the treatment; 3) a reduction in the density of most other benthic insect taxa; and 4) a reduction in invertebrate density that lasted over the winter into the following year.

Collectively, these observations suggest that American Black Duck and Ring-necked Duck broods will be sensitive to the changes in invertebrate abundance that might be caused by contamination of their nesting areas by the aerial spraying of fenitrothion. Thus, there is a potential for fenitrothion to have an impact on duckling survival. However, the relationships between invertebrate abundance and duckling growth and survival are not well defined. The data suggest that if reductions in invertebrate numbers similar to those observed by Fairchild (1990) in bog ponds are also manifest in brood-rearing ponds, then survival of American Black Duck ducklings could be reduced (Martin 1992). As the LC50s of fenitrothion for species of Diptera, Odonata, Ephemeroptera, and Trichoptera are all less than 200 μg/L, which is within the range of concentrations observed by Ernst *et al.* (1991) in forest ponds sprayed during operational applications, then wide-scale reductions in invertebrate biomass in American Black Duck nesting areas are likely. Further, because these are important invertebrate taxa for ducklings, it also suggests that most of the important food resources for ducklings could be substantially reduced following fenitrothion applications. Although American Black Duck broods are able to move among wetlands to avoid localized invertebrate population depressions (Ringleman and Longcore 1982), which may mitigate fenitrothion impacts, the large-scale spray operations as practised in New Brunswick would most likely result in reduced invertebrate biomass in all potential brood-rearing habitat within a brood's home range. This would suggest that the impact on nesting waterfowl would be diminished only if the operation were reduced in size, with smaller blocks treated in a patchwork pattern.

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