A HISTORY OF THE EFFECTS OF AERIAL FOREST SPRAYING IN CANADA ON AQUATIC FAUNA

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Throughout the history of aerial chemical control operations in Canada, insecticides have been interacting with aquatic ecosystems. The vast forested areas which have been the sites of these operations are scattered with rivers, lakes and streams, and insecticides have been introduced into watersheds containing the breeding and rearing areas of important commercial and sport fishing stocks, particularly of salmonoid fish. The effects of insecticides on these fish populations and the ecology of their habitat has frequently resulted in conflict between forestry and fishery interests within an area.

The Fisheries Act broadly prohibits the discharge of deleterious substances into waters inhabited by fish. Nevertheless, such waters are integral parts of forest ecosystems and their suitability for the propagation of fish depends upon the health of the surrounding forests. It has been the policy of fisheries personnel not to strictly enforce this section of the Fisheries Act, but to allow the use of chemical control agents in an enlightened manner with fisheries input in planning control programs and the use of special precautions to protect fish populations (Jackson, 1966). Implementation of these precautions has not always been adequate or sufficient to prevent losses of fish and their food organisms; but a sincere desire to eliminate these side-effects, and co-operation between forestry and fisheries personnel in planning, monitoring, and assessing experimental and operational spray programmes have resulted in the development of policies and techniques compatible with the interests of both groups. Whenever undesirable side effects have been discovered and evaluated, every possible precaution has been taken to avoid their recurrence (Fettes and Buckner, 1972).

Bioassays, Insecticides and Application Procedures

Most of our knowledge of the effects of insecticides on aquatic ecosystems has come from monitoring activities carried out by fisheries researchers in spray areas. Progressive changes have indeed been adopted in major control projects during the past 15 years, in respect of insecticides, dosage rates and application procedures. These changes have resulted primarily from information provided through field monitoring programs and bioassay studies of the effects of insecticides on fish, and through continuing laboratory and field research on numerous candidate insecticides as effective pest control agents.

The relative toxicity of different insecticides to fish can be established by comparing values determined by bioassays. Bioassay work was carried out by several researchers in the late 1950's to determine the toxicity of DDT to salmonoids present in spray areas. Alderdice and Worthington (1959) studied the toxicity of the 1957 Northern Vancouver Island spray to coho salmon under yearlings.

They found a 48-hour LC_{50} (the concentration at which 50% of the test fish die in 48 hours) of 20.7 ppm for the emulsifier but were unable to establish a LC_{50} for the spray formulation as concentrations both above and below theoretical concentrations of 0.5 to 0.3 ppm were less toxic than these concentrations. They concluded however, that concentrations of the spray formulation below 0.05 ppm would be safe to young coho salmon with the criterion for safety being the absence of mortality within an exposure time of one week at $10^{\circ}C$. Gagnon (1958) found that ¼ lb per acre DDT (0.32 ppm) caused high mortality among Atlantic salmon and trout in circulating hatchery water with salmon fry, salmon parr and trout fry showing decreasing sensitivity in that order. Further experiments confirmed that the sensitivity of Atlantic salmon to DDT decreased with age. The LC_{50} of DDT to salmon fry in standing water was 0.072 ppm, corresponding to an application of one tenth of a pound per acre, but mortality did not occur for almost four days at this concentration. Mixing the DDT with water made it more toxic to fish than when it was present in an oil solution on the surface of the water. Bioassays conducted by Keenleyside (1958) on young salmon in 1957 gave estimated 24- and 48-hour LC_{50} values of 0.049 and 0.047 ppm for DDT in oil.

Early findings of field and laboratory studies documenting the adverse side-effects of chemical control programs using DDT on aquatic fauna led to a conference early in 1958 at which the conflicts between protecting fishery and forestry resources were discussed (Anonymous, 1958). As a direct result of this conference, an Inter-departmental Committee on Forest Spraying Operations was established

(Prebble, 1960). This committee has been instrumental in coordinating the development and implementation of chemical control methods with limited adverse side-effects on non-target organisms. It has coordinated experimental spray and bioassay programs to evaluate and determine the toxicity to fish of potentially useful insecticides. Most of the bioassay work has been conducted by personnel of the St. Andrews Biological Station of the Fisheries Research Board of Canada. Their findings over the years are summarized in the following table which compares the LC_{50}'s (for the stated time periods) of various insecticides to Atlantic salmon parr. This table has been compiled from data from Keenleyside (1958) and unpublished reports of the St.Andrews Biological Station.

Toxicity to Atlantic Salmon Parr of Some Insecticides Used Against Forest Insects in Canada
(all numbers refer to concentrations as parts per million)

Insecticide	Family	LC_{50}'s 24-hr.	48-hr.	4 day	7 day	Apparent lethal threshold
Zectran	Carb.	16.0	13.0	-	-	13.0
Phosphamidon	O.P.	15.0	11.0	-	7.0	less than 0.1
Dylox	O.P.	13.0	6.5	3.0	-	1.3
Baygon	Carb.	-	2.0	-	-	2.0
Fenitrothion	O.P.	2.0	1.4	-	-	0.9
Matacil	Carb.	-	1.1	1.1	-	1.1
Malathion	O.P.	-	0.13	-	-	0.13
DDT (in oil)	O.C.	-	0.05	-	-	-
Malathion (in oil)	O.P.	-	0.03	-	-	-

Carb. - carbamate O.P. - organophosphate O.C. - organochloride

The extremely low toxicity to salmon parr of the organophosphate and carbamate insecticides which have replaced DDT is easily seen from this table. Phosphamidon shows a very low short-term toxicity but continues to display toxicity down to very low concentrations when exposure times are in the order of several weeks (Sprague, 1966). This apparently low lethal threshold would not be significant under field conditions as phosphamidon is rapidly degraded and fish would not be exposed to initial concentrations long enough to be affected.

Modified treatment procedures have been used by themselves and in conjunction with new insecticides to protect aquatic life in streams and rivers within spray areas. Up until 1960 streams had been used as spray block boundaries in New Brunswick and this resulted in them frequently being sprayed twice, once during the spraying of each adjoining block (MacDonald, 1968). In 1961 a system was introduced where rectangular spray blocks were used and spray planes were directed from higher-flying aircraft. Streams were no longer used for plot boundaries and DDT spray was shut off when spray planes crossed streams visible to the guide plane. In 1963 phosphamidon buffer or safety zone spraying was introduced in DDT spray areas in New Brunswick. A swath of forest approximately 1,200 feet wide bordering both sides of major streams or lakes was sprayed with phosphamidon rather than DDT to protect aquatic life. The introduction of phosphamidon safety zone spraying in 1963 and its use over increasingly larger proportions of the watersheds in DDT spray areas in subsequent years reduced the adverse effects of the DDT sprays.

The use of phosphamidon safety zone spraying largely overcame the undesirable side-effects of DDT upon the aquatic environment. However, the lack of serious side effects upon other components of forest ecosystems, its effectiveness against budworm and its relative safety to persons handling it in its pure form left DDT as one of the best insecticides available in the late 1960's for large-scale operational control programs in eastern Canada (Joy, 1968). Although phosphamidon had been shown to have no significant effects on aquatic organisms, its high cost and high toxicity to birds made it less suitable for treating large areas. DDT, however, had been shown to have a very long half-life and toxic breakdown products and evidence had been found of sub-lethal effects of DDT residues on reproduction in fish (Burdick et al, 1964).

In 1968 the use of DDT in operational control programs against forest insects in Canada was banned. The only subsequent use, in New Brunswick in 1969, was over a small acreage to provide a check on the efficacy of the substitute insecticide. The organophosphate insecticide fenitrothion was chosen for large scale application to spruce budworm infested forests in 1969 on the basis of results of experimental applications of this insecticide which had been conducted yearly since 1966. Phosphamidon safety zone spraying was no longer necessary as fenitrothion had been shown to cause no mortality among salmon populations at the operational dosage applied (2 oz per acre in two separate applications). Fenitrothion is still in widespread use today but researchers continue to look for and evaluate new, even safer control materials. Particular emphasis is being placed on the development of effective spray formulations containing bacteria, viruses and hormones as "biological insecticides" which have no adverse side-effects on non-target organisms.

DDT - Effects on Field Populations

Pioneer work in Canada on the effects of DDT on aquatic ecosystems was carried out by the Ontario Department of Lands and Forests during DDT field trials conducted over spruce budworm, *Choristoneura fumiferana* (Clem.) infested forest in Algonquin Park in 1944 and 1945 and in areas north of Lake Superior in 1945 and 1946 (Ontario Department of Lands and Forests, 1945 and 1949). Studies of the effects on fish by Langford (1949) indicated that DDT was toxic to fish in their diet, when present in water as an emulsion and to a lesser extent when present as a surface film. A small proportion of the fish populations in sprayed streams was killed due to gorging on knocked down flying insects or coming into contact with a surface slick of DDT in oil. Effects on aquatic invertebrates as reported by Savage (1949) were more severe. Invertebrate life in streams in sprayed areas was drastically reduced but there were no effects on organisms inhabiting the shallow margins of lakes. Surface forms (water striders and whirligig beetles) were killed by the surface slick and blackflies and other inhabitants of rapids were eliminated as the DDT was emulsified by tubulence. Free living caddisflies, mosquito larvae, tadpoles and crayfish were particularly sensitive to the spray but clams, snails, copepods, cladocerans and carnivorous aquatic insects exhibited considerable resistence to insecticide poisoning. Insects present as eggs or pupal forms were also resistant and prevented the loss of entire insect populations. It was concluded from these studies that if streams were protected from being sprayed directly their invertebrate populations would be unlikely to be reduced by more than 40% for a significant length of time and would recover within two years. During this period the growth of fish in the streams might be reduced due to lack of food.

In 1952, aerial forest spraying against spruce budworm began in New Brunswick using DDT. Kerswill (1967) has summarized the development of the fisheries research program related to the spraying operations in New Brunswick, and effects of the latter on aquatic ecosystems from the early 1950's to the early 1960's. In 1952 and 1953 systematic visual patrols were made of lakes and streams in the sprayed areas. Numbers of dead Atlantic salmon parr, *Salmo salar* L., brook trout, *Salvelinus fontinalis* Mitchell, and aquatic insects were seen in rivers and streams and dead brook trout and minnows were found in lakes (Kerswill and Edwards, 1967). In 1954 the spray areas included watersheds where investigations of salmon populations by Fisheries Research Board of Canada scientists from the St. Andrews Biological Station had been in progress since 1950. This provided an opportunity to compare populations of young salmon and other fish exposed to an aerial application of ½ lb per acre DDT to unsprayed populations of the previous four years. Studies were also made of mortality of salmon parr held in live-boxes in sprayed and unsprayed streams. Results of investigations with the caged fish showed that the fish were unaffected by spraying directly overhead but sizeable kills occurred within a few days of upstream spraying. Three weeks after spraying 63 to 91% mortality had occurred among caged fish in the sprayed streams as compared to negligible mortality in control streams (Kerswill and Elson, 1955). The annual fall population census showed that salmon fry had been practically eliminated from sprayed streams and small and large parr reduced to lesser extents. Aquatic insects also appeared to be almost completely absent from sprayed streams compared to normal populations at control areas. The spraying had no effect upon runs of adult salmon entering sprayed areas in the fall to spawn.

The results of the 1954 investigations showed that DDT spraying had serious side-effects on young salmon and as a result many of the research projects of the St. Andrews Station were directed toward discovering the scope and nature of these effects. Investigations to study the recovery of fish and aquatic insect populations affected in 1954 were carried out and sprays within the study areas in 1956 and 1957 were monitored for their effect. These studies showed that young salmon, brook trout and eel, *Anguilla rostrata* Le Suer populations were reduced in number after spraying (Keenleyside, 1959). Minnows appeared to be ralatively unaffected. The number of underyearling salmon was reduced by 90% in most streams treated with DDT at ½ lb per acre and small and large parr were generally reduced by about 70% and 50% respectively (Kerswill, 1957; Elson, 1967). Juvenile salmon populations seemed to recover in three years if an area was not resprayed within this period. Trout populations recovered within two years. Respraying budworm infested areas at intervals of less than three years prevented the recovery of juvenile salmon populations and was predicted to seriously decrease the number of adult salmon returning to the area to spawn in future years. Runs of adults consist of fish of several successive year classes and if spraying in successive years has reduced the strength (i.e. numbers) of closely grouped year classes, future runs of adult fish may be small and adversely affect commercial and sport fishing. Kerswill (1958) forecast reductions in adult salmon stocks caused by severe mortality among year classes sprayed as young. This mortality would not be reflected in commercial fishing and angling catches until several years after the spraying, when the affected year classes were entering the fishery.

Studies on the effects of DDT spraying on aquatic insects were begun in 1955 by F. P. Ide in New Brunswick and by G. Filteau in spray areas within the Gaspé. Populations were studied primarily by trapping emerging adults with cages placed in streams and occasionally by collecting larval forms from rocks in the stream bed. These studies showed that immediately following spraying emergence of adult insects ceased almost completely for from three to six weeks as most larval insects were killed by the spray (Ide 1957 and 1967; Filteau, 1959). During this period a thick mat of filamentous algae appeared in the stream bed due to the absence of herbivorous insect larvae. Insects in a large river were affected more slowly and to a lesser extent than insects in streams, probably due to greater dilution of the insecticide. Recovery of insect populations began with the emergence of massive number of midges (Diptera, Fam. Chironomidae) which rapidly increased the number of insects emerging each day to above pre-spray levels. The volume of emerging insects remained very low until the emergence of caddisflies (Trichoptera) which had survived the spraying in resistant pupal stages. The recovery of insect populations in the years following spraying was fairly slow, especially among the larger forms. In the year after spraying large numbers of chironomids continued to emerge but the number of species of mayflies (Ephemeroptera) emerging was only half the number at the control station and caddisflies and large stoneflies (Plecoptera) were almost totally absent.

Changes in bottom fauna populations resulted in changes in the diet of young salmon and influenced their populations accordingly (Keenleyside, 1967). Reduction of the number of larger aquatic insects by spraying forced the surviving salmon fry and parr to feed much more extensively on the large numbers of chironomids and other diptera larvae which appeared several weeks after spraying. The continued scarcity of larger insect forms in the year after spraying resulted in the parr feeding heavily on snails, worms and small fish, all of which were unimportant items in the pre-spray diets of salmon parr. Over a three year recovery period after spraying chironomids became less important and mayflies more important in parr diets as their respective populations returned to normal levels. Caddisfly larvae were the last items to reappear in the diets of young salmon in significant numbers. The results of these changes in the bottom fauna and resulting changes in the diets of young salmon can be seen in the effects on the year class of salmon born after the 1954 spraying. The large number of fry spawned by heavy runs of adult salmon in 1954 had a bountiful supply of suitable food in the large number of chironomid larvae present in their streams and also had few competitors or predators because populations of older salmon, trout and eels had been reduced by spraying. As this year class thrived under these conditions more larger inset forms became available to them as aquatic insect populations began to recover. Most of this year class was not exposed to further DDT spraying and subsequently resulted in stronger than normal runs of smoult to the sea when they matured.

In 1958 and 1959 experimental applications of DDT at various dosages were conducted in New Brunswick. A significant finding was that DDT was effective against budworm at rates of 1, ½ and ¼ pounds in one gallon of formulation per acre providing spray coverage and the resulting deposit was adequate (Fettes, 1960). The critical measurement for dosage effectiveness was confirmed to be droplets of spray per unit area and not gallons emitted per acre. This showed that lower doses of DDT could be effective against budworm providing sufficient number of drops of spray per unit area could be deposited. Reduction of the dosage of DDT used would be significant with respect to the effects on aquatic life as monitoring of experimental applications of ¼ lb per acre DDT had shown this dosage to have no observable effect on salmon and limited effect on aquatic insects. Fisheries Research Board monitoring in 1958 and 1959 showed caged salmon fry and parr, and planted and native fish populations suffered no significant mortality when exposed to applications on DDT at ¼ lb per acre whereas ½ lb per acre reduced numbers of caged fry and to some extent planted hatchery parr (Kerswill and Edwards, 1967).

In 1960, reduction of the DDT dosage to ¼ lb per acre was tested operationally over about half the total New Brunswick spray area while ½ or 3/8 lb per acre was used to treat the rest of the area. In the years following 1960 DDT was almost exclusively applied at the dosage of ¼ lb per acre, but complications in the budworm outbreak involving differences in the kind of damage to balsam fir and red spruce often resulted in the necessity of two applications of DDT at this dosage about ten days apart - the first application being made to preserve balsam fir foliage and the second to reduce surviving budworm populations on red spruce (Macdonald, 1964). Monitoring of the effects of these sprays on aquatic organisms was carried out cooperatively between personnel of the St. Andrews Biological Station of the Fisheries Research Board and the Fish Culture Branch of the Department of Fisheries. Results of these studies showed that mortality among caged salmon fry and parr and trout fingerlings was approximately proportional to the total dosage of DDT applied. Two separate ¼ lb per acre treatments several days apart were similar in their effect to a single ½ lb per acre treatment. They caused a 90% reduction among wild salmon underyearlings and appeared to reduce parr to much the same extent as ½ lb per acre treatments (Elson, 1967). Adult eels held in areas sprayed twice with ¼ lb DDT per acre and feeding on contaminated salmon parr were unaffected (MacDonald and Penney, 1967). Single ¼ lb per acre treatments reduced underyearlings by about 50% and effected parr to a relatively insignificant extent. There was evidence that DDT caused fish mortality to a decreasing extent up to 20 miles downstream from the spray area. This was also seen in the high mortality among caged fish downstream from watersheds in ¼ lb per acre DDT treatment areas sprayed only once but over a period of several days (Kerswill, 1962; Kerswill and Edwards, 1967). Effects on salmon and trout in experimental areas treated with ¼ lb per acre DDT in one application were considerably less severe than those shown in operational spray areas where the application of ¼ lb per acre DDT was spread over several days. Surber (square-foot) samples showed bottom fauna in experimental ¼ lb per acre DDT spray areas was reduced by up to 97% over the spray period (Ide, 1967). Large numbers of drifting insects affected by the spray were collected within an hour of spraying and for several hours afterwards from screens set across treated streams.

Another effect of DDT observed in New Brunswick was delayed mortality in the fall of salmon parr in watersheds treated with DDT earlier in the year. Mortality of this nature was observed in the field during the fall of 1954, 1957, and 1960 when stream temperatures were approaching freezing for the first time (Kerswill and Edwards, 1967; Elson, 1967). Following these observations salmon parr were collected from a sprayed and unsprayed stream in 1960 and subjected in the laboratory to gradulaly lowering temperatures and then held near one degree centigrade. Parr from the unsprayed streams survived for a month with no mortality but 80% mortality occured among parr from the treated stream (Fisheries Research Board of Canada, 1961). Delayed mortality of salmon parr was observed again and studied in the field in 1964, 1965 and 1966. It occurred in both phosphamidon safety zone protected streams and unprotected streams within the DDT spray areas (MacDonald, 1971). The extent and distribution of mortality varied from year to year. Native parr were held in cages in 1965 and 1966 during the period when water temperatures were approaching freezing and daily patrols were conducted along these streams to search for dead fish. Some correlation between mortality among caged fish and the number of dead native parr found in the same stream was evident, but the greatest number of parr found dead over a six week period in a single stream was only 34, even though 55% mortality occurred among caged fish in the stream. Parr from spray areas contained significantly greater DDT and DDD metabolite residues than parr from unsprayed streams but there were no significant differences between DDT residues in parr collected live and dead from spray areas. This indicates that mortality was governed by factors other than the concentrations of DDT and its metabolites accumulated by the fish. Parr populations in streams where delayed mortality occurred had below average condition factors (a measure of the ratio of weight to length) than populations in unsprayed streams but this was also true for populations in treated streams where no fall mortality occurred. Another factor which was linked to delayed mortality was sexual maturity. From 80 to 100% of the caged and native parr found dead were sexually mature males, even though females made up approximately half of the samples of native fish collected by electro-seining. No underyearling salmon were found dead. This evidence indicates that delayed mortality among salmon parr in DDT treated streams may be initiated by a drop in water temperature but results from physical changes and physiological stresses associated with sexual maturation.

In 1955 an outbreak of black-headed budworm, *Acleris gloverana* (Wals.), was assuming damaging population levels on northern Vancouver Island. The British Columbia Loggers' Association formed a Pest Control Committee to study the feasibility of combating the outbreak by means of aerial chemical control. Spray trials were conducted with DDT in 1956 to determine the timing and dosage of chemical application effective against this insect (Brown et al, 1958). When the outbreak persisted into 1957 an operational spray program was planned to apply 1 lb. DDT per acre to a portion of the infested forest (British Columbia Loggers' Association, 1957). Personnel of the Federal Fisheries Department and the British Columbia Fish and Game Commission were invited to participate in planning the operation to suggest safety measures to avoid damage to salmon and trout stocks present in the spray area. This resulted in several modifications of the original flight plan. Areas of low value timber containing streams of significant salmonoid populations were removed from the areas to be sprayed. Spray block boundaries were shifted away from all streams so they would not be sprayed twice as adjacent blocks were sprayed. Spray planes were instructed to keep one swath away from major streams and to shut off spray when crossing the streams. Fisheries personnel observed the actual spraying from another plane to ensure that these protective measures were carried out and that spray planes avoided spraying streams whenever possible.

Despite close adherence to these precautions as confirmed by the fisheries observers, considerable fish mortality occurred in the spray areas (Crouter and Vernon, 1959). Much of this could be attributed to the difficulty of spotting streams from the air and unpredictable drift of spray products into the streams. Due to the timing of spray application only coho salmon fry, *Oncorhynchus kisutch* Walbaum, and trout were present in the streams as other species of salmon fry and steelhead (rainbow trout) smolts had completed migration to the sea or movement into lakes. Live pens containing coho fry, hatchery fingerling trout and native steelhead smolts were established in streams within treatment and control areas to assess effects on these populations. Mortality of coho fry in these cages varied from none to severe, reflecting the range in the amount of insecticide entering various streams in the spray area due to differences in visibility from the air, drift of spray, forest cover, meteorological conditions and other factors. Mortality of hatchery fingerling trout was somewhat less, and of steelhead smolts light. The DDT was applied as an emulsion and as a result had the greatest effect in the areas sprayed and very limited effect further downstream. This is the opposite of findings from New Brunswick where DDT sprayed in a solvent oil had a greater effect downstream than at the place where it entered the stream. Coho fry abundance estimated from seine hauls in the Keough River after spraying showed large numbers a week after spraying, few a month later and none in October, when fry were still found in control streams. It is possible that fry in this river were displaced downstream in search of food since the bottom fauna was reduced by the spray in terms of number of organisms and weight of aquatic insects present. Similar reductions in bottom fauna occurred in some other treated streams. Analysis of stream water for DDT content showed fluctuation from trace amounts to 0.4 ppm within a two-week period after spraying but generally remained above the "safe" level of 0.05 ppm arrived at by Alderdice and Worthington (1959) for three days after spraying at the four stations where DDT content of the water was monitored.

Coho salmon in the streams affected by the 1957 DDT spraying on northern Vancouver Island have a predominant 3-year life cycle and the spawning runs consist almost exclusively of fish of the same age class. Concern was therefore expressed by fisheries officers that the number of adult salmon of the affected year-class returning to spawn would be insufficient to restore Coho populations in the most seriously affected streams for many cycles.

Examination of estimates of the number of adult Coho salmon returning to spawn in these streams in succeeding years fails to reveal such an effect. The numbers of individuals present as fry in 1957 and returning to spawn in the fall of 1959 fall within the range of estimates of spawning runs during the 6 years preceding DDT spraying and are not noticeably lower than the numbers of adults unaffected by spraying and returning to spawn in 1958 and 1960. However, commercial fishing in 1959 was curtailed owing to a strike of fishermen, and runs into the spawning streams may have been proportionately greater than usual on that account. Examination of similar data on spawning runs of pink, chum, spring and sockeye salmon in streams within the treated areas reveals no effects of DDT spraying.

An operational control program using DDT in fuel oil at the dosage of ¼ lb per acre was conducted on northern Moresby Island, British Columbia in 1960. A small stream was intentionally sprayed preliminary to the operational program to evaluate the hazard of this dosage to native fish populations and their food (Todd and Jackson, 1961). Results showed no significant effect on native fish or aquatic insect populations although some mortality occurred among caged fish restricted to the top six inches of the stream. No significant mortality occurred among caged fish held at other depths. In order to protect fish populations in the operational area ¼-mile-wide swaths along both sides of major salmon streams were left unsprayed. No effects were evident on fish populations or aquatic insects in these streams although small amounts of drift entered the streams and some mortality among caged fish restricted to the surface of the streams again occurred.

DDT-Laboratory Studies on Sublethal Effects

Recently, numerous laboratory studies have been conducted to look for sublethal effects of DDT on fish. One of the first sublethal effects of DDT studied was its raising of the lower lethal temperature of brook trout and salmon. This has already been discussed with respect to delayed mortality of salmon in DDT treated streams. Temperature selection may also be affected by sublethal exposure to DDT according to results of studies conducted at the Biological Station at St. Andrews. A variety of experiments with brook trout showed that sublethal DDT exposure affected the central nervous system in various ways. Their lateral line nerve was rendered hypersensitive to an experimental stimulus (Anderson, 1968), the cold-blocking temperature for a simple reflex was altered (Anderson and Peterson, 1969), and their ability to learn a visual conditioned avoidance response was impaired (Anderson and Prins, 1970). The ecological significance of these effects are unknown but the effects seem to disappear several weeks after exposure to the insecticide. Unpublished studies revealed no effect of sublethal DDT exposure on the swimming performance of salmon smoults. The effects of exposure to sublethal concentrations of DDT on the learning ability of Atlantic salmon parr were examined by the use of a shuttlebox conditioning apparatus (Hatfield and Johansen, 1972). Learning ability appeared to be enhanced by 24-hour exposure to the 96-hour LC_{50} and no effect was observed at one tenth this concentration. The vulnerability of Atlantic salmon parr to predation by large brook trout in concrete pools was unaffected by 24-hour exposure to the 96-hour LC_{50} of DDT (Hatfield and Anderson, 1972).

DDT - Residue Studies

Since the cessation of DDT applications in New Brunswick, several reports have been published on the disappearance of DDT and its metabolites from water, sediments, and fish in this area. Water sampled before, during and up to two years after the final operational application of DDT contained concentrations of DDT exceeding a steady background level of 0.5 ppb of pp'DDT for only a few hours after treatment (Yule and Tomlin, 1971). The highest concentration of DDT found was about 17 ppb and occurred as a surface film associated with the formulating oil during and immediately after treatment. Bottom sediments collected a year after the end of the DDT spraying showed a downstream dilution gradient of DDT residues from tributary to estuary and much of the original DDT they contained had been decomposed to DDE and DDD. DDT residues in Atlantic salmon parr from streams with various treatment histories were analysed and the results showed that the total amount of all DDT residues (y) in ppm remaining after a given number of years (t) roughly followed the equation $y = \frac{1.91}{t}$ (Sprague et al, 1971). Total DDT decreased to low or undetectable concentrations in 2.5 years or less after treatment but the common metabolite DDE decreased slowly over 12.5 years. Lobsters, fish and shellfish collected from estuarial and coastal waters of New Brunswick and Prince Edward Island in the fall of 1967 contained average or less than average DDT residues compared to the same or similar species elsewhere in North America and Northern Europe (Sprague and Duffy, 1971). Salmon parr, trout and sculpins collected in 1973 from the Keough River in British Columbia, where heavy fish mortality was reported following the application of 1 lb per acre DDT in 1957, contained total DDT residues ranging from 0.004 to 0.015 ppm (Buckner et al, 1975 a).

Phosphamidon - Effects on Field Populations

In 1962 the organophosphate insecticide phosphamidon was sprayed experimentally on a small watershed in New Brunswick. This insecticide had shown promise for budworm control in field trials conducted in 1961 and had been shown to exhibit very low toxicity to young coho salmon compared to DDT in laboratory tests conducted by the Fisheries Research Board at Nanaimo, British Columbia (Kerswill, 1962). Application of a dosage of 1 lb phosphamidon per acre was found to have no significant effect on caged salmon or native fish populations in the area. Studies on aquatic insects by means of Surber sampling, adult emergence cages, and drift screens revealed no effects on their populations (Grant, 1967). A similar experimental treatment with 1 lb phosphamidon per acre was applied directly onto a small coho salmon stream in British Columbia in 1963 to study its effects on caged and native coho fry and caddisfly larvae (Schouwenburg and Jackson, 1966). There was no effect on caged or native coho fry but it appeared that there was 100% mortality among caged and native caddisfly larvae. The theoretical initial concentration of phosphamidon in the stream water was calculated to be between 1.37 and 1.05 ppm. Bioassays conducted in the field indicated slight mortality would occur among coho fry exposed to concentrations of about 3.2 ppm phosphamidon for 24 hours.

From 1963 until the end of DDT operational spraying in 1968, phosphamidon was used for safety-zone spraying along streams and rivers in New Brunswick and significant progress was made in protecting aquatic life in DDT spray areas by increased use of this technique. Results of monitoring done in 1963 showed once again that phosphamidon by itself had no significant effects on caged and native fish or aquatic insects (Smith 1963, Grant 1967), but considerable mortality occurred among caged and native salmon within phosphamidon safety zone areas because of DDT entering unprotected streams in the DDT spray areas and being carried into the phosphamidon sprayed streams. Caged fish and population estimate studies in 1964 showed severe fish mortality in unprotected DDT spray areas, reduced losses in phosphamidon safety zone spray streams and no effect in phosphamidon spray areas (MacDonald, 1964). Caddisfly larvae held in cages in streams within 1/4 lb. phosphamidon per acre spray areas were also unaffected. In 1965 over 500 miles of streams within the DDT spray areas received phosphamidon safety zone treatment. Spray cards showed that DDT had been kept at least 1000 feet from almost all the phosphamidon sprayed streams sampled and this was reflected in the low mortality among caged fish held in these streams (MacDonald, 1965).

Immediate post-spray population estimates revealed no effects on native populations but late summer seining showed fewer salmon fry and parr in these streams than in control areas although they were still abundant. Single and double phosphamidon treatment at $\frac{1}{4}$ lb. per acre was again found to have no effect on caged fish.

Caged fish within phosphamidon safety zone spray streams in 1966 suffered only 5% greater mortality than control groups (Penney and MacDonald, 1966). Native fish populations in these streams were censused in the fall and were at least 2/3 normal density except in areas which received two applications of 1/3 lb. DDT per acre outside the protected areas. Fish populations in these areas were very low in the fall. Bottom fauna populations within safety zones were monitored in 1967. Caddisfly larvae were unaffected by spraying, mayfly nymphs and dipterous larvae were slightly affected and stonefly nymphs were reduced to very low levels. The overall effect on the total weight of aquatic insects was not severe and total biomass had recovered to control levels by mid-August.

Fenitrothion - Effects on Field Populations

Fenitrothion (Sumithion) was first evaluated for its effects on New Brunswick streams in 1966. Application of $\frac{1}{2}$ lb. per acre caused no short term mortality among caged salmon fry and parr but reduced the number and weight of fish-food organisms to about one-third their pre-spray levels (Penney and MacDonald, 1966). The following year fenitrothion was sprayed experimentally by itself and in various mixtures with phosphamidon. None of these caused short term mortality among caged salmon parr or affected native salmon fry populations (MacDonald and Penney, 1967). Fenitrothion (Novathion) sprayed at the rate of ½ lb. per acre caused a rapid decline in the number and weight of all groups of aquatic insects to very low levels. Recovery of all groups was evident beginning two weeks after treatment but only Diptera recovered to above pre-spray levels by the end of summer. Application of a mixture of 3/8 lb. per acre fenitrothion (Sumithion) and 1/8 lb. per acre phosphamidon had considerably less severe an effect on aquatic insects. Monitoring of 3/8 lb. per acre and ¼ lb. per acre applications of fenitrothion (Sumithion) in 1968 again revealed no mortality among caged salmon parr (MacDonald and Penney 1968). Results of studies on effects of these treatments upon aquatic insects differed in their findings from earlier studies as only mayfly nymph populations were reduced and the net effect upon the total weight of fish-food organisms present was insignificant. This was supported by data collected concerning growth rate of salmon parr in the higher dosage fenitrothion treated stream which showed no effect on the growth rate of these fish after treatment. In addition, the aquatic insect fauna of a stream treated with ½ lb. per acre fenitrothion (Novathion) the previous year was found to have recovered from the low levels to which it had been reduced.

In 1969 fenitrothion was used exclusively in the operational budworm control program in New Brunswick. Monitoring again showed fenitrothion had no effect upon juvenile salmon and no significant reductions in aquatic insect populations were found (MacDonald and Penney, 1969), as had been the case in 1968. Large scale applications of fenitrothion in 1970 again had no significant effects on caged and wild salmon or aquatic insects (Penney, 1970). Bottom fauna studies in fenitrothion treated streams in 1971 showed relatively little effect on total biomass but some impact upon stonefly nymph populations was apparent (Penney, 1971). The concentration of fenitrothion in stream water reached a peak of 0.057 ppm after spraying and 3.19 ppm fenitrothion was detected in a sample of mayfly nymphs a week after spraying, but no fenitrothion was found in mayfly nymphs from the same location beyond this date even though a second application of fenitrothion was made over the stream.

Recently, fenitrothion has been used extensively in operational spray programs in Newfoundland, New Brunswick, Quebec, Manitoba and British Columbia. Mortality was insignificant among caged salmon and brook trout in areas of Newfoundland treated with fenitrothion (Sumithion) in 1968 and 1969 to control outbreaks of hemlock looper, *Lambdina fiscellaria fiscellaria* Guen. (Hatfield and Riche, 1970). Gas chromatographic analysis of whole caged and wild fish collected live from spray areas revealed fenitrothion residues ranging from 0.0 to 0.77 ppm. Fenitrothion applications against the same insect on Anticosti Island in 1972 caused no significant mortality among caged Atlantic salmon parr (Coté and Tetrault, 1973). Four days after spraying and for several days thereafter, abnormally large numbers of dead salmon parr were caught in a blocking seine set across a sprayed river but it appears that this resulted from young salmon populations readjusting their distribution in the river in response to high water levels caused by heavy rainfall two days after the completion of spraying. Fenitrothion residues in these fish remained below 0.01 ppm but persisted at measurable levels for at least eleven days. Bottom fauna populations in streams on Anticosti Island treated with fenitrothion in 1973 appeared normal approximately three weeks after treatment (Buckner et al, 1975 b).

Several reports of fish mortality were investigated following the application of fenitrothion to approximately 10 million acres of spruce budworm infested forest in western Quebec in 1973 (Kingsbury, 1973). No significant fish mortality could be traced to insecticide poisoning at any of the locations where investigations were made. About thirty brook trout were found dead on one small lake but this was an insignificant number of fish in terms of the lake's total brook trout population of several thousand. Fish found dead the day after treatment of the surrounding area contained fenitrothion residues of 0.77 ppm which they may have accumulated by coming into contact with a surface film of insecticide dissolved in the oil carrier while preying upon shallow dwelling dragonfly nymphs.

A study was made of insecticide residues and blood chemistry parameters of rainbow trout, *Salmo gairneri* Rich. held in cages in a fenitrothion treated stream in Manitoba during operational sprays against spruce budworm in 1973 (Lockhart et al, 1973). No acute toxic effects on caged trout were found. There were some small differences in blood chemistry parameters of control and treated fish the day after treatment, but these did not persist to four days after the treatment when they were next monitored. Fenitrothion residues in whole, live trout were as high as 1.87 ppm the day after treatment but fell to below 0.02 ppm by the fourth day. Bottom fauna and juvenile salmonoid populations in a stream on Northern Vancouver Island studied one month after treatment of the surrounding forest with fenitrothion in 1973 appeared normal (Buckner et al, 1975 a).

Fenitrothion - Laboratory Studies on Sublethal Effects

Organophosphate insecticides inhibit the enzyme acetylcholinesterase whose activity is essential to nerve transmission. Acetylcholinesterase activity of brain homogenates of salmon and trout exposed to sublethal concentrations of fenitrothion in the laboratory and during operational spraying in New Brunswick was monitored in 1969 (Zitko et al, 1970). Exposure in the laboratory to concentrations of from 0.125 to 0.75 ppm fenitrothion for four days significantly reduced this activity from the normal level. There were no significant changes in acetylcholinesterase activity in salmon and trout from the operational treatment area but activity was reduced by about 20% in suckers collected alive and by about 25% in suckers found dead in the treated stream. Laboratory studies on the effects of sublethal exposure to fenitrothion on behaviour of juvenile coho salmon showed that exposure to concentrations of fenitrothion greater than 25 to 50% the 96-hour LC_{50} (1.3 ppm) reduced the frequency of all behaviours except coughing, and seemed to cause serious physiological impairment (Bull, 1971). Concentrations from 10% up to 50% the 96-hour LC_{50} reduced feeding and altered social behaviour, but short term exposure to these concentrations was concluded to have probably no significant lasting effects. Prolonged exposure to 10% of the 96-hour LC_{50} appeared to have no effects on behaviour of the test fish. Hatfield and Johansen (1972) found that 24-hour exposure of Atlantic salmon parr to the 96-hour LC_{50} of fenitrothion completely inhibited learning when measured in a shuttlebox conditioning apparatus.

Learning ability occurred with exposure to concentrations of one-tenth the 96-hour LC_{50} or lower. Similar results were found in experiments examining the effects of fenitrothion on the vulnerability of Atlantic salmon parr to predation by large brook trout in concrete pools (Hatfield and Anderson, 1972). Exposure for 24 hours to the 96-hour LC_{50} increased the vulnerability to predation of fenitrothion treated parr but exposure to one-tenth the 96-hour LC_{50} of fenitrothion had no effect. These results all indicate that concentrations of fenitrothion of 0.1 ppm and less have no sublethal effects on salmonoids. On the basis of present knowledge the estimated concentration of 0.045 ppm fenitrothion present in an average stream immediately after treatment (Wildish et al, MS 1971) can be considered a safe level for this group of fish. Studies on the effects of fenitrothion in the diet of brook trout showed that ingestion of contaminated aquatic or terrestrial insects by salmonoids is extremely unlikely to cause lethal or sublethal effects as behavioural changes in the laboratory did not occur until brook trout ingested 3,000 times the highest level of fenitrothion found in poisoned insects in treatment areas (Wildish and Lister, 1973).

Fenitrothion - Residue Studies

The persistence of fenitrothion in aqueous systems has been studied in the laboratory by Sundaram (1973). The half-lives of fenitrothion ranged from 60 hours at a pH of 5.42 to 22 hours at a pH of 8.22. Another important factor besides pH with respect to the breakdown of fenitrothion in water may be photodecomposition. Flasks of water spiked with 200 ppb of fenitrothion and kept under dark and light conditions showed significant differences in the rate of disappearance of the insecticide from them (Lockhart et al, 1973). Fenitrothion concentrations fell from 200 ppb to 10 ppb in one day in flasks exposed to sunlight but had only decreased to 142 ppb at the end of eight days in the flasks kept in the dark, but otherwise held under identical conditions. Lockhart also reports that fenitrothion concentrations in stagnant water exposed to an operational spray fell from a maximum of 75 ppb to about 2 ppb in less than four days and continued to disappear beyond this date. Similar rapid disappearance of fenitrothion has been observed in shallow ponds within experimental fenitrothion treatment areas near Ottawa (Kingsbury, 1973).

Other Chemical Insecticides

Numerous promising new insecticides have been monitored for their effects on aquatic fauna, primarily in New Brunswick. Experimental applications of DDD and malathion in 1958 and 1959 in New Brunswick produced no observable side-effects on fish but proved incapable of providing effective budworm control (Fettes, 1960; Kerswill and Edwards, 1967). In 1965, dimethoate was applied to a small coho salmon stream in British Columbia to evaluate its hazard to fish in conjunction with experimental applications against hemlock needle miner, *Epinotia tsugana* Freeman. Application of ½ lb. dimethoate per acre had no significant effect on native trout or coho fry held in live boxes and aquaria. Calculated concentrations of the insecticide reached 0.6 ppm in the aquaria (Jackson, 1965). Experimental applications of up to ½ lb. per acre of the carbamate insecticide zectran in 1967 and 1969 had no effect on aquatic insect populations (MacDonald and Penney, 1967 and 1969). Cursory examination of streams sprayed with the carbamate insecticide propoxur (Baygon) in 1968 revealed normal aquatic insect populations (MacDonald and Penney, 1968). Experimental applications of dylox (two applications at 6 oz per acre), lannate (two applications at 1 oz per acre), and matacil (two applications at $1\frac{1}{4}$ oz per acre) were monitored for side effects on aquatic organisms in 1970 and found to have no significant effects on juvenile salmon or aquatic insects (Penney, 1970). An application of $1\frac{1}{2}$ oz matacil per acre in 1971 caused a significant reduction in stonefly nymph populations and reduced the total biomass of benthic organisms to 60% the pre-spray level (Penney, 1971).

Biological Control Agents

New methods of controlling forest insects with a minimum of effects on non-target organisms are presently being sought and experimented with. Several potential methods of biological control have been developed involving aerial dispersal of bacteria, viruses, and hormones. Experimental applications of *Bacillus thuringiensis* have been shown to have no effect upon fish or bottom fauna (Todd and Jackson, 1961; Buckner et al, 1974). Application of an analogue of juvenile hormone to a stream on Anticosti Island in 1973 also had no effects upon aquatic organisms (Buckner et al, 1975 b). Future use of these and other biological control agents should totally eliminate adverse side-effects of control programs upon aquatic organisms but will depend upon their yet unproven ability to control forest insects under operational treatment conditions.

Interpreting Results of Monitoring Programs

Effects of insecticides on aquatic ecosystems have often been over-rated. This is partially due to failure to consider the ability of natural populations to recover from the effects of a destructive agent, whether natural or artificial. Weakness in the methods of monitoring insecticide effects have also led to exaggerated reports of damage to aquatic ecosystems. Hatchery fish held in live-boxes often suffer far greater mortality than native fish populations. Studies from 1964 to 1966 showed that whereas caged hatchery fish held in DDT treatment areas suffered 80 to 100% mortality, native fish suffered mortality of 20% or less even when confined to live-boxes (Penney and MacDonald, 1966). Caged fish studies are useful in comparing the relative toxicities of different insecticides and dosages, but if not used in conjunction with other methods they do not give a true picture of the effects on native fish populations. Studies on aquatic insects sometimes fail to consider natural fluctuations in populations related to life cycles, water levels, suitability of substrate, and other factors governing these populations.

Future Considerations

Much progress has been made in discovering and eliminating undesirable side-effects of forest insect control operations in Canada on aquatic ecosystems. The role of many factors related to insecticide toxicity and side-effects still remain vague, however. Spray droplet size, evaporation of descending spray, volume of spray deposited, and meteorological effects on spray deposit all influence the eventual effects of the emitted formulation on aquatic fauna (Fettes, 1962). Changes in methods of application, such as night-time applications of insecticides, may result in significant changes in the effects on non-target organisms. Continued surveillance and close scrutiny of experimental and operational insecticide applications must be maintained to provide continually updated knowledge of the ecological consequences of insect control programs (Fettes and Buckner, 1972). These studies must take into account the ecological ties between all habitats and inhabitants of forest ecosystems.

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